SPACE-TIME MODELLING OF EXPOSURE TO AIR POLLUTION USING GIS

Thesis submitted for the degree of
Doctor of Philosophy
at the University of Leicester

John Gulliver

June 2002

UMI Number: U641509

All rights reserved

INFORMATION TO ALL USERS
The quality of this reproduction is dependent upon the quality of the copy submitted.

In the unlikely event that the author did not send a complete manuscript
and there are missing pages, these will be noted. Also, if material had to be removed,
a note will indicate the deletion.

UMI U641509
Published by ProQuest LLC 2015. Copyright in the Dissertation held by the Author.
Microform Edition © ProQuest LLC.
All rights reserved. This work is protected against
unauthorized copying under Title 17, United States Code.

ProQuest LLC
789 East Eisenhower Parkway
P.O. Box 1346
Ann Arbor, MI 48106-1346


UNIVERSITY OF LEICESTER
LIBRARY

TABLE OF CONTENTS

3 DEVELOPMENT OF OUTDOOR POLLUTION MODELLING 58

4 DEVELOPMENT OF EXPOSURE MODELLING 98

LIST OF FIGURES AND TABLES

Figures

Tables

ACKNOWLEDGEMENTS

Firstly and foremost, I would like to thank Professor David Briggs for his continuous enthusiasm, stimulation, patience, and kindness during the course of this research. I first worked for Dave as a placement student during 1994, as part of my first degree in Geography at Huddersfield University. After Huddersfield, I spent a year in Leicester studying for a MSc in GIS, before Dave took me on for a second time as a Research Assistant at Nene Centre for Research, University College Northampton. I had four very enjoyable years in Northampton, before moving on to Imperial College in January 2001.

This research grew out of work undertaken in Northampton as part of the EPSRC-funded Northampton Air Pollution Study (NAPS). There are a number of people who were directly involved with NAPS, without whom this research would not have been possible: Paul Elliott, Mike Ashmore, James Tate, Margaret Bell, Karen Batty and John McClatchey. I would like to thank Joe Alfano, of Northampton Borough Council, for helping with the organisation of air pollution monitoring. I would like to thank all the people at Nene Centre for Research, in particular Charlotte, Michelle, Lynn, Maria, Allison, Chris Arciszewska, and Pete King. Special thanks to Kees de Hoogh for all the help and advice, both as a fellow student in our Northampton days and now as a work colleague at Imperial College. Thanks also to Joe and Jon at Imperial College for their patience while I was writing-up.

And finally, my heartfelt thanks to Linda Beale, who as my partner and best friend has supported and encouraged me all the way. She is a star.

ABSTRACT

Space-time modelling of exposure to air pollution using GIS

John Gulliver

This thesis develops, tests and applies methods for space-time modelling of exposure to air pollution using GIS. This involves linkage of five main sub-models: a traffic model, a model of urban air pollution - combining local and 'background' pollution models - a network analysis tool for modelling exposure during journeys, and a time-activity model. The model can provide exposure estimates for individuals or population groups. The study took place entirely within Northampton, UK.

The model used to estimate hourly PM_{10} concentrations at outdoor locations gave a moderate fit to monitored data. Results were shown to be comparable with the best results from other studies. This research also found a strong, linear relationship between concentrations of PM_{10} during simultaneous monitoring of walking and in-car concentrations. This relationship was used to calibrate modelled outdoor pollution levels to give in-car concentrations. Modelled journey-time exposures for walking performed equally with predictions made using a fixed-site monitor located close to journey routes. The model did not perform as well as the fixed-site monitor in predicting in-car exposures.

The application of the model to a walk-to-school policy, in which modelled local traffic levels were reduced by 20%, demonstrated that the benefits of the reduction were not spread evenly across a sample of schoolchildren, but varied depending on the route used to school and the location of homes and schools. For those switching between car and walk there may be positive or negative effects of the policy in terms of savings in average hourly exposures, depending on their specific journey and time activity patterns.

The results from this research showed that, although the model worked reasonably well in estimating exposures, a number of improvements are needed. These include better models of background concentrations, more detailed models of in-car conditions, and extending exposure modelling to include dose-response estimates.

33,000 words

1 INTRODUCTION

1.1 RATIONALE

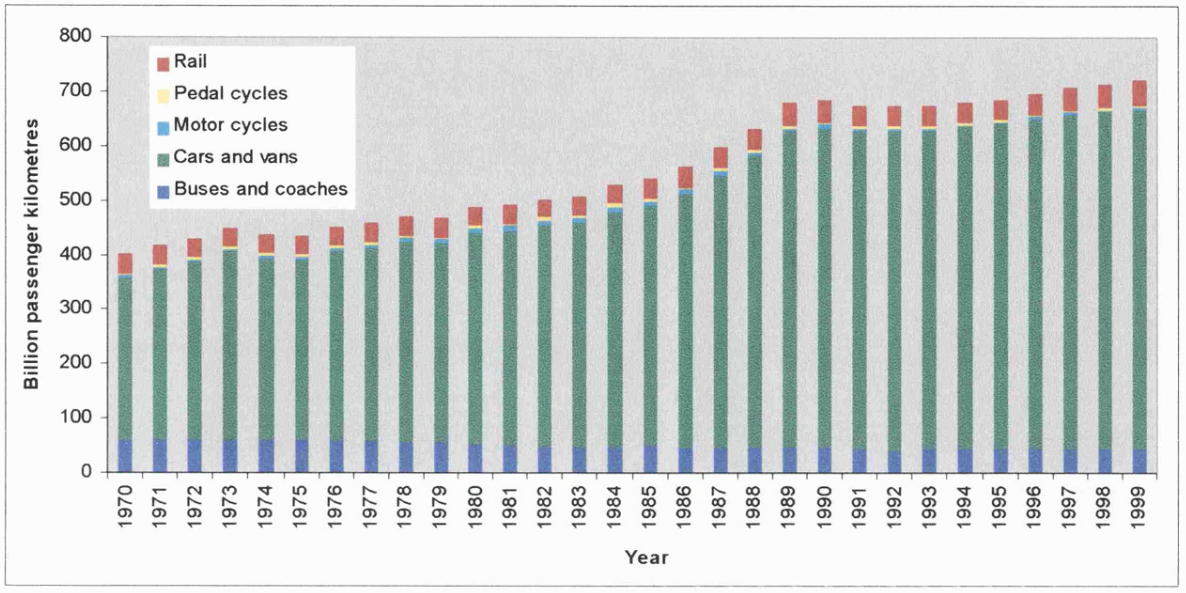
The effect of environmental air pollution on public health is a major global concern. It is estimated that, globally, 3 million people die each year and tens of thousands of people become ill as a result of poor air quality (WHO, 1999). Whilst there has been a reduction in traditional pollutants from industrial sources, such as sulphur dioxide (SO_2), the continued growth in road traffic has contributed to rising levels of traffic-related air pollution.

The rapid growth in urban populations and household incomes has led to an increase in car ownership. In the UK, seventy two per cent of households now have use of at least one car - this has increased from sixty percent of all households at the beginning of the 1980s. Over a quarter of households now have two or more cars (DTLR, 2001). The increase in car ownership has consequently led to a demand for more roads. In the ten year period to 1999 there was an increase in total road length in the UK of just over 15,000 kilometres, of which more than 95% were roads in built-up areas. In 1997, passenger journeys exceeded 700 billion kilometres for the first time, almost doubling the total distance covered by passengers in 1970 (*Figure 1.1*).

Between 1980 and 1999, fuel use by road passenger vehicles increased by a third whilst distance travelled increased by half. For cars, total trips made per person each year have risen from 520 in the mid-1980s to 640 in 1999/2000, whilst average trip length also increased from 7.8 to 8.7 miles (12.6 to 14 km) (DETR, 2000a). While walking remains the dominant mode for journeys under one mile, the car is used for 60% of journeys between one and two miles, and more than 80% of journeys between 2 and 5 miles (3.2 and 8.1 km) (DETR, 1998).

In recent times, more sustainable forms of development, cleaner fuels, traffic management schemes and national and international air quality standards have resulted in some improvements in air quality. Nevertheless, air pollution from road

Figure 1.1: Passenger transport in Great Britain (1970 –1999) (DLTR, 2001)



transport remains a major problem. In particular, there is growing concern about the health effects of human exposure to traffic-related air pollution. The health effects of traffic-related pollution are likely to be most severe in urban areas where elevated levels of pollution occur.

Evidence from a number of epidemiological studies suggests links between poor air quality and a wide range of health outcomes, including respiratory illnesses, (Schwartz, 1991; Pope *et al.*, 1995) cardio-vascular disease (Pönka, 1998, Schwartz, 1999), and reduced lung function (Hoek and Brunekreef, 1994). Research has also shown that levels of asthma and respiratory disorders tend to increase with proximity to main roads (Nitta et al, 1993). Wjst *et al.* (1993 reported a strong association with the prevalence of asthma and higher traffic volumes. For most of the population, exposure to vehicle emissions has no obvious effects at all, but for sensitive groups such as the young, elderly, and asthma sufferers, relatively small doses could be harmful. Schoolchildren, who live and make journeys along busy streets are, therefore, particularly vulnerable.

In response to these concerns, increasing efforts at national and international levels are being made to limit emissions of traffic-related air pollution. WHO (World Health Organisation) guide values, limits, and targets for different pollutants provide the basis for national and international air quality strategies. In the UK, one of the main strategies for reducing levels of air pollution has been through the introduction of air quality standards. The UK air quality standards must comply with levels specified in the European Union (EU) directives. In recent years, the UK standards have undergone several revisions. A set of air quality objectives has been devised, from recommendations by the Expert Panel on Air Quality Standards (EPAQS), to be reached by 2005 (*Table 1.1*)

Table 1.1: UK air quality objectives for 2005

Substance	***Objective level ($\mu g/m^3$)***	***Averaging period***	***AQ objective date***
Benzene	16.25	Running annual mean	31st December 2003
1,3 – Butadiene	2.25	Running annual mean	31st December 2003
Carbon Monoxide	11.6	Running 8 hour	31st December 2003
Lead	0.5	Annual mean	31st December 2004
	0.25	Annual mean	31st December 2005
Nitrogen Dioxide	200	99.8th %-ile hourly mean	31st December 2005
	40	Annual mean	31st December 2005
PM_{10}	50	90.4 %-ile 24 hour mean	31st December 2004
	40	Annual mean	31st December 2004
Sulphur Dioxide	125	99.2 %-ile 24 hourly mean	31st December 2004
	350	99.7th %-ile hourly mean	31st December 2004
	266	99.9th %-ile 15 minute mean	31st December 2005

Source: DETR, 1999

The framework for action and control in the UK is provided through Local Agenda 21 initiatives for sustainable development, the recently formed UK National Air Quality Strategy (NAQS), and advice and guidance from the Department for Environment, Food, and Rural Affairs (DEFRA) and the Environment Agency. Over the past few years there have been two significant policy developments: firstly, the legally binding agreement to reduce traffic emissions signed by Britain at the Kyoto Summit (1997), and, secondly, the recent Transport White Paper (DETR, 1998) placing a new emphasis on 'curbing the car culture' by developing more balanced and integrated forms of transport. This places a new urgency on traffic planning and air quality management.

The Transport White paper gives local authorities a key role to play in achieving national objectives. Each local authority has an obligation to declare areas that are unlikely to meet the NAQS objectives as Air Quality Management Areas (AQMAs). Special attention must be given to these AQMAs in order to achieve the NAQS objectives by 2005. Local authorities have a legal requirement to assess the contribution of local traffic to pollution in their area and take necessary action to reduce and control levels of pollution. A key objective of the White Paper is to cut the use of cars for school journeys by measures to improve the safety of children who walk or cycle to school. In the UK, the average journey length of trips to school for children aged 5 to 10 increased by 35% between 1985 and 1999 (DETR 2000a). It is envisaged that a combination of new pedestrian routes, cycle paths, more and safer crossings, increased frequency of buses, and traffic management schemes will encourage a modal shift.

The extent to which such interventions can effectively reduce air pollution and associated exposure nevertheless remains uncertain, as the processes involved are likely to be complex. For example, measures in one area may cause displacement, merely redistributing pressure to other areas. In terms of recent legislation, reductions in the number of school journeys made by car may provide new opportunities for other population groups to shift journey timing. The result may be to flatten the peak hour pollution levels, but prolong the duration of the peak.

The effects of traffic management and policy interventions have direct implications for human exposure to air pollution. Assessing human exposure to air pollution is, however, intrinsically complex. Exposures rarely occur as single, discrete events. Instead, individuals move, in space and time, at differential rates, through continually changing fields of air pollution. This complexity has, however, rarely been recognised in air quality management and epidemiological studies. In the majority of studies, exposure has been determined without using data on spatial and temporal variation in air pollution concentrations (Clench-Aas *et al.*, 1999). Indeed, most studies have simply used the nearest air pollution monitor to quantify exposure of large numbers of people, which tends to cause exposure misclassification and dilute measured associations with health. An ideal approach

would be to obtain exposure estimates for any number of individuals across a relatively wide area over a period of time. The spatial modelling facilities in GIS clearly provide the opportunity to undertake such studies (Briggs & Elliott, 1995).

In combination with GIS techniques, advances in air pollution monitoring, traffic modelling, and line-source dispersion modelling techniques offer the opportunity to model exposure, but the lack of communication between existing technologies, and high levels of expertise necessary to use them makes this a difficult task. The development of such techniques would provide an important link between traffic planning and epidemiological studies. Small numbers of systems are beginning to emerge that link GIS with models of population and air pollution. However, present systems often have low spatial resolutions, crude exposure assessment (if any), and do not take full advantage of GIS (Jensen *et al.*, 2001). There is, therefore, an urgent need for more dynamic methods of exposure assessment that are spatially- and temporally- based, and reflect the complex patterns of human movement through a continually changing field of pollution.

1.2 AIMS AND OBJECTIVES

The general aim of this thesis is:

to develop, test and apply space-time modelling of exposure to air pollution using GIS, as a basis for epidemiological and policy applications.

Within this context, the specific objectives are:

1) to review the health effects of air pollution and existing methods and models of exposure assessment

2) to develop methods for modelling air pollution at fixed-site locations and as people move through the urban environment

3) to develop a high resolution, space-time exposure model with the capability of dealing with complex patterns of human activity in a range of urban environments

4) to validate the performance of the models against monitored data

5) to demonstrate the capability of the model and apply it to examine the effects of a policy application in a detailed case study

6) to consider the implications of the results for wider application within epidemiological studies and policy applications

To this end, a mixture of monitoring and modelling is used both to develop new techniques and enhance existing methods. The approach adopted in this research involves linkage within a GIS of five main sub-models: a traffic model, a model of urban air pollution, a 'background' pollution model, a network analysis tool for modelling exposure during journeys, and a time-activity model. Traffic and air pollution monitoring techniques are used as a basis for calibrating models to local

conditions and as a basis for model validation. The study takes place entirely within Northampton, a medium-sized town in the East Midlands, UK.

1.3 STRUCTURE OF THE THESIS

The following gives a description of the structure of the research:

- Chapter 2 presents a review of the literature on the health effects of air pollution and methods and models for assessing exposure

- Chapter 3 provides a discussion of existing methods of quantifying and modelling air pollution, before describing the methods used to develop and implement a model to estimate outdoor air pollution.

- Chapter 4 provides a description of the overall framework used for exposure modelling. It then describes the methods used to develop and calibrate the individual components of an exposure model.

- Chapter 5 evaluates the performance and validation of the models, and then demonstrates the capability of the methodology and applies it to examine the effects of a policy application.

- Chapter 6 reviews the results and main findings of the research and discusses its implications for future research.

2 LITERATURE REVIEW

2.1 EPIDEMIOLOGICAL EVIDENCE

2.1.1 Introduction

Traffic-related pollutants have been shown to have a range of health effects (*Table 2.1*). In the last decade, however, the focus of attention has shifted towards particulate air pollution. The interest in particles grew from a number of US studies that found strong correlation between particles and daily death rates, independent of the effects of other pollutants (Schwartz, 1991; Schwartz and Dockery, 1992; Pope *et al.*, 1992). Although much of the particulate matter in the atmosphere is from natural sources (e.g. wind blown minerals, sea spray, particles from volatile organic compounds from plants), there is a significant proportion from anthropogenic sources, such as soot, smoke, and combustion of fossil fuels, that all may contain toxic elements. Particles derived from traffic-related combustion are of particular concern because their size means they can be breathed deep into the lungs. It is generally accepted that particles less than 10 μm in diameter (PM_{10}) carry the toxic elements that can have adverse effects on health.

There is a considerable body of epidemiological evidence to suggest that particulate air pollution has adverse effects on human health. These effects can be classified into two main groups: acute (short-term) or chronic (long-term). Typically, acute effects are observed on a daily basis and chronic effects are observed on an annual or longer basis. Clear association between air pollution and adverse health effects have been shown in studies of both acute and chronic effects. For example, Schwartz *et al.* (1991) and Pope *et al.* (1991) found strong acute associations between ambient PM_{10} and respiratory symptoms. Schwartz (1993a) and Pope *et al.* (1995) both found positive associations between PM_{10} and respiratory symptoms in studies of chronic effects. It is thought, however, that many chronic effects are simply the result of repeated exposure to elevated levels of air pollution (Schwartz, 2000).

Table 2.1: Health effects of traffic-related pollutants

Pollutant	***Health effects***
Particulate matter	PM has been associated with increased mortality, morbidity and reduced lung function. Adverse health effects have been observed in both children and adults. These effects are associated with coughing and respiratory diseases such as pneumonia, asthma and bronchitis. PM exacerbates the effects of SO_2, and vice versa (Onursal and Gautam, 1997).
NO_2	Acute exposure to NO_2 causes respiratory disease, like coughs and sore throat. NO_2 probably worsens the lung function of people with chronic bronchitis and asthma at levels higher than 300 ppb (not confirmed by all studies) (Elsom, 1996). Exposure to NO_2 is linked with increased susceptibility to respiratory infection (Onursal and Gautam, 1997)
Benzene	Benzene has toxic and carcinogenic effects. Toxic effects have been associated with the central nervous system as well as the hematological and immunological systems. Toxic effects on the nervous system have been observed with concentrations of 1,000 ppm or higher. Carcinogenic effects include leukemia (Onursal and Gautam, 1997)
PAH	PAH are mutagenic and carcinogenic. They get absorbed in the lungs and the intestines and are metabolized in the human body. It is estimated that 9 0f 100,000 people exposed to 1 $\mu g/m^3$ of benzo[a]pyrene, a PAH, over a lifetime, would develop cancer (Onursal and Gautam, 1997)
Ozone	Adverse health effects of ozone can occur at exposure periods as short as 5min. Ozone can cause severe damage to lung tissues and impair defences against bacteria and viruses (Onursal and Gautam, 1997). Short-term effects of ozone begin at hourly averages of 200 $\mu g/m^3$ include: eye, nose and throat irritation, coughing, throat dryness, thoracic pain and chest tightness. A decrease in pulmonary functions in children and young adults has been reported at hourly average ozone concentrations between 160 and 300 $\mu g/m^3$. Long term exposure to ozone may reduce pulmonary function (Romieu, 1992)
CO	CO interferes with the absorption of oxygen by heamoglobin (Hb) (CO binds with Hb to form carboxyhaemoglobin (COHb) which can occupy oxygen-binding sites), increases cardiovascular disease and can affect the nerves. Threshold for effect is ~2% COHb, is equivalent to 8hr exposure at moderate activity to 15-20 ppm CO (Elsom, 1996).
Lead	Lead causes impairment of brain development and function in infants and children, even at seemingly low blood lead levels (Committee of the Environmental and Occupational Health Assembly of the American Thoracic Society, 1996). Blood lead levels of less than 10 $\mu g/dl$ have been reported causing the effects (Romieu, 1990).
SO_2	SO_2 is associated with reduced lung function and increased risk of mortality and morbidity. Adverse health effects include coughing, phlegm, chest discomfort, and bronchitis. SO_2 exacerbates the effects of PM, and vice versa (Onursal and Gautam, 1997)

Source: de Hoogh (1999)

Indeed, the clearest associations have been shown in short-term studies. Short-term studies are typically time series studies that are either population or cohort (group) based. Time-series studies are essentially regression analyses that look at aggregated data on counts of mortality or morbidity against monitored data on air pollution. The other type of study (panel) follows specific 'cases' over time, looking at their symptoms in relation to exposure to PM_{10}.

2.1.2 Population-based time-series studies

A number of population-based studies have reported a positive association between increased mortality and increased levels of PM_{10}. The results from most of these studies show that increased mortality occurred within 1 to 5 days following an increase in PM_{10} concentration. The increases in mortality lie in the range 0.6% to 3.5%, for a 10 $\mu g/m^3$ increase in PM_{10}, with the average increase in mortality at just over 1%. Several of these results are from the U.S six cities study (Schwartz *et al.*, 1991; Pope *et al.*, 1992; Schwartz and Dockery, 1992; Dockery *et al.*, 1993; Schwartz, 1993b), which was the first large-scale study to show a significant association between increased daily all-cause mortality and elevated levels of PM_{10}. In the six cities studies, the strongest associations were seen for respiratory and cardiovascular causes (Pope *et al.*, 1992; Schwartz and Dockery, 1992; Schwartz, 1993b).

A number of studies in other countries have reported similar associations. Ostro *et al.* (1996), for example, reported an increase of 0.6% in total mortality for a 10 $\mu g/m^3$ increase in PM_{10} in Santiago, Chile. In Europe, Verhoeff *et al.* (1996) and Pönka *et al.* (1998) found increases of 1.2% and 3.5%, respectively, in daily mortality with a 10 $\mu g/m^3$ increase in PM_{10}. Interestingly, Verhoeff also found no association between increased mortality and increases in levels of either CO or SO_2. Other studies have also shown weak associations between mortality and other pollutants, though the effects of PM_{10} seem to be independent of other pollutants (Katsouyanni *et al.*, 1997; Samet *et al.*, 2000). Also worthy of note is that, some studies have observed associations between increased mortality and increased PM_{10} at relatively low levels of PM_{10} (Schwartz, 1991; Hong *et al.*, 1999;

Pönkä *et al.*, 1998). In the study by Hong *et al.* (1999), for example, mortality was raised at levels of PM_{10} well below the current Korean air quality standard.
Two recent papers have reviewed and analysed results from studies of PM_{10}. Levy *et al.* (2000) compared mortality estimates from over twenty daily time series studies. They found that, on average, mortality increases by about 0.7% per 10 $\mu g/m^3$ increase in PM_{10} concentrations. Samet *et al.* (2000) pooled data from 20 US cities (1987 to 1994) on all cause, respiratory, and cardiovascular mortality, alongside pollution data on CO, SO_2, O_3, NO_2 and PM_{10}. After controlling for confounding by other pollutants, they found that the increase in rate of death for a 10 $\mu g/m^3$ increase in PM_{10} was 0.51% for all causes, and 0.68% for respiratory and cardiovascular causes. A weak association was found between elevated O_3 and increased mortality, but there was no association for increased death with any of the other pollutants.

Other studies have considered associations between particulate air pollution and morbidity using a variety of health endpoints. Several of the population-based, time-series studies on mortality have also analysed hospital admissions. For example, Pope *et al.* (1991), using hospital admissions data from Utah, USA, found strong, statistically significant associations between PM_{10} and respiratory symptoms. Schwartz *et al.* (1993) found that a 10 $\mu g/m^3$ increase in PM_{10} was associated with a 3.4% increase in asthma visits. A number of other studies have found associations between hospital admissions for asthma and elevated levels of PM_{10} (Thurston *et al.*, 1994; Burnett *et al.*, 1994; Burnett *et al.*, 1995; Delfino, 1997). Significant associations have also been found between air pollution and admissions to hospital for COPD (chronic obstructive pulmonary disease) (Burnett *et al.*, 1994; Schwartz, 1994a) and pneumonia (Schwartz, 1994a).

The overall picture from population-based time-series studies is a convincing pattern between particulate air pollution and acute effects on health. There are, however, two basic problems with this type of study. Firstly, there is doubt that the associations between particulate matter and health are entirely causal. The main problem in this respect is inadequate control for ecological confounding (e.g. weather conditions). Secondly, and of fundamental importance, is that time-series studies are essentially dimensionless in time. In essence, each event is a once-

only event. These events do not explain the factors that contribute to adverse effects of particulate air pollution. Little is known about the characteristics of persons that put them at increased risk of adverse events related to particulate air pollution (Zanobetti *et al.,* 2000), or how this varies between susceptible groups (e.g. children, the elderly). These problems are exacerbated by the crude measures of exposure used in many of these studies. Typically, time series studies have used sparse networks of monitoring stations as the sole measure of exposure. Künzli *et al.* (2001) suggested that because of the complexity of relating exposure to health outcomes, the impact of air pollution on mortality should be based on panels rather than time series analysis.

2.1.3 Panel studies

A relatively small but growing number of studies have attempted to link PM_{10} and acute morbidity using panels of individuals. These studies tend to use either measured lung function or rely on self-reporting of respiratory symptoms as health end-points. Asthma, for example, has been used in a number of panel studies of acute effects of particulate air pollution. Studies by Roemer *et al.* (1993) in The Netherlands and Ostro *et al.* (1991) in Denver, Colorado, both reported increased asthma attacks associated with an increase in PM_{10}. For a panel of asthmatic schoolchildren, Roemer *et al.* (1993) reported a 2.3% increase in bronchodilator use and a 1.1% increase in asthmatic attacks per 10 $\mu g/m^3$ increase in PM_{10}. Ostro *et al.* (1991) reported an 11.5% increase in asthma attacks in adults per 10 $\mu g/m^3$ increase in PM_{10}. Smaller and less significant associations have been found in studies of lung function. Across the studies by Pope *et al.* (1991), Pope and Dockery (1992), Neas *et al.* (1995 and 1996), Roemer *et al.* (1993), and Peters *et al.* (1997), there was an average decrease in lung function of between 0.04 and 0.25% for an increase of 10 $\mu g/m^3$ in PM_{10}.

In a more recent study in The Netherlands, by van der Zee *et al.* (1999), PEF (peak expiratory flow) was measured, and respiratory symptoms were reported on a daily basis, for children aged 7 to 11 years living in urban areas with high traffic intensity. Simultaneously, panels of children living in non-urban areas were measured. For children with symptoms, significant associations were found

between PM_{10}, black smoke (BS) and sulphate (SO_4) and the presence of symptoms in the lower respiratory tract (LRS). Particle concentrations were associated with the use of bronchodilators in the urban areas but not in the non-urban areas. An association was also found between respiratory symptoms and PM_{10} in children without symptoms, but this was only small. The results of this study suggest that children with respiratory symptoms are more susceptible to the effects of particulate air pollution than those children without symptoms. Furthermore, in a cohort of 2,200 school children, from Vancouver, Vedal *et al.* (1998) found that declines in PEF were associated with increased respiratory symptoms such as cough and sore throat. Children with asthma were found to be more susceptible to these effects than other children.

Many panel studies also utilise diaries, coupled with information on time-activity, to provide self-reporting of respiratory symptoms. The use of time-activity diaries allows for interpretation of associations between air pollution and specific health outcomes in terms of the activities and locations where exposures occur. Neas *et al.* (1996), for example, in a study of 108 children, found that symptoms of cough, cold, and wheeze were significantly associated with exposure to particles less than 2.5μm in diameter. One of the largest reported panel studies, as part of the six cities study in the USA, tracked respiratory symptoms of 1844 (about 300 in each city) schoolchildren with the aid of diaries (Schwartz *et al.,* 1994b). A highly significant association was found between incidence of cough and particles concentrations.

2.1.4 Measures of particles

There is emerging evidence that health effects of particles are due principally to fine particles ($PM_{2.5}$). As part of the six cities study in the US, for example, Schwartz *et al.* (1996) found fine particles to be more strongly associated with daily mortality than coarse particles (PM_{10} - $PM_{2.5}$). This study also recognised that sulphate did not account for all the effects of $PM_{2.5}$, suggesting that particles from primary combustion were most important. A similar study was commissioned by EPAQS (Expert Panel on Air Quality Standards) in Birmingham, UK, to analyse daily mortality against PM_{10}, $PM_{2.5}$, sulphate, and black smoke for the period 1994-

1996. None of the individual measures showed a significant effect on mortality, but the effect of the coarse fraction ($PM_{2.5-10}$) was found to be negative (DETR, 2000b). In other words, health was seen to get better with higher concentrations. This analysis was extended to look separately at cardiovascular and respiratory mortality, and hospital admissions. The only clear outcome was a strong negative effect of the coarse fraction on daily respiratory mortality (Anderson, 2000).

Results from other studies, however, suggest that there is insufficient evidence of health effects being attributable solely to $PM_{2.5}$. Burnett *et al.* (1997), for example, looked at the effects of different particle fractions on both respiratory and cardiac admissions in Toronto, Canada between 1992 and 1994. For respiratory admissions, a stronger association was observed in the fine fraction, but for cardiac admissions the outcome was reversed, with the coarse fraction marginally more significant. Overall, the study showed that health effects of the fine and coarse fraction were about the same. Furthermore, a study in Seattle, WA, by Sheppard *et al.* (1999) showed that both the fine and coarse particle fractions were positively and equally associated with increased incidence of asthma attacks.

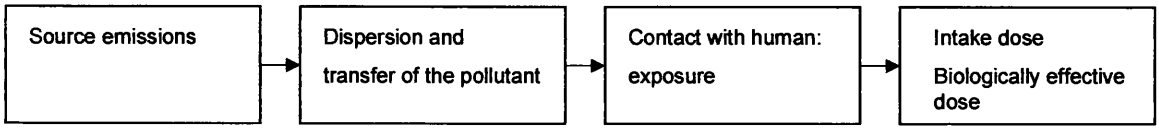
In summary, there is inconclusive evidence to suggest that the toxic component of PM_{10} lies solely in the $PM_{2.5}$ fraction. In a recent review on *'what is the appropriate measure on which to base a standard for particulates'*, EPAQS recommends that:

> "the control of particulate air pollution by measurement of PM_{10} provides adequate protection of the health of the public"
> (DETR, 2000b).

2.2 THE EXPOSURE PROCESS

Exposure is an event that occurs when a person comes into contact with a pollutant (Ott, 1982). There is a sequence of stages in the exposure process from pollutant source through to exposure and dose, as shown in *Figure 2.1* (after Lioy, 1990). The links between each stage in the exposure process are often referred to as pathways or routes of exposure.

Figure 2.1: The sequence of exposure



Source: Lioy (1990)

Ideally, a comprehensive approach would exist that considers each stage of the exposure process. The exposure process is, however, intrinsically complex.

There are a wide range of traffic-related pollutant species that are emitted into the atmosphere from a number of source types (petrol / diesel cars, lorries, buses, trains). Vehicle fleet behaviour (e.g. vehicle speed, number of vehicles) also affects the rate of emissions emitted from source. The transfer and dispersion of pollutants into the atmosphere is governed by the relationship between source orientation and surrounding vegetation, terrain, and buildings, and meteorological conditions (e.g. wind speed, wind direction, temperature). Contact between air pollution and humans will depend on the microenvironment which they occupy: outdoors (at home, at work, shopping), indoors (kitchen, bedroom, living room), and in transit (walking, in-car, cycling, bus). Finally, the *intake dose* depends on the duration of the exposure and rate of breathing associated with the activity of the exposed. The *target organ dose* is the amount that reaches the human organ where the relevant health effect occurs (Corvalán *et al.*, 1996). Important physical characteristics that affect *target organ dose* are volume of lungs, and metabolism.

2.3 METHODS OF EXPOSURE ASSESSMENT

2.3.1 Introduction

Estimates of human exposure to air pollution can be made using two approaches: 1) the direct approach, using measurements of air pollution in each occupied

microenvironment, or 2) the indirect approach by intersecting models of population distribution with models of exposure (Ott, 1990). A number of direct and indirect methods have been developed to facilitate exposure assessment.

Direct methods include:

- fixed-site monitoring
- personal monitoring
- micro-environmental monitoring

Indirect methods include:

- indicators
- bio-markers
- GIS-based techniques
- dispersion modelling

2.3.2 Direct methods of exposure assessment

a) Fixed-site monitoring

Fixed-site monitors are the most common source of data on air pollution in exposure assessment and epidemiological studies. Typically, exposure estimates are made on the basis of air quality data from the nearest site in a national network of monitoring stations. For example, as part of the six US cities study, Pope *et al.* (1992) measured concentrations of pollutants at three sites in Utah City. Similarly, van der Zee *et al.* (1999) took measurements of PM_{10} and BS at a single location in four Dutch communities. Examples of studies using routine monitoring data are shown in *Table 2.2.* The table shows that even within highly populated urban areas there are relatively few routine sites available. Routine monitoring networks are typically sparse and do not necessarily represent the levels of pollution across urban areas.

This pattern is also reflected in the UK. The UK national monitoring network is run by DEFRA, through AEA Technology, providing continuous measurement of a range of pollutants, including PM_{10}, CO, NO_2, NO_X, O_3, SO_2. There are currently 52 urban, and 3 rural sites monitoring PM_{10}. A summary of the monitoring network by pollutant is shown in *Table 2.3.*

Table 2.2: Exposure assessment in acute studies: examples from studies based on routine monitoring data

Authors	***Area/city***	***Study population***	***Pollutants***	***No. of sites***
Katsouyanni *et al.* 1990	Athens	ca. 700,000	SO_2 BS	5 5
Pope *et al.* 1991	Utah City	188,000	PM_{10}	3
Mackenbach *et al.* 1992	Netherlands	ca. 15 million	SO_2	6
Saldiva *et al.* 1994	Sao Paulo	16 million	SO_2 CO NO_x O_3 PM_{10}	8 3 3 4 8
Schwarz 1994a	Philadelphia	ca. 1.4 million	TSP	2+
Kieding *et al.* 1995	Copenhagen	ca.1 million	SO_2, NO, NO_2, O_3, CO	5
Pope *et al.* 1999	Salt Lake City Ogden Provo/Orem	965,000 165,000 275,000	PM_{10} PM_{10} PM_{10}	4 1 3
van der Zee *et al.* 1999	Rotterdam Amsterdam	600,000 720,000	PM_{10}, BS, SO_4, SO_2, NO_2	4 4
Norris *et al.*, 2000	Spokane, Washington, USA	300,000	PM_{10}, CO, SO_2, NH_4, EC, O_3	3

As research has progressed, it has become clear that in many instances ambient monitoring at fixed locations does not adequately or accurately reflect the exposure of the human population (Ozkaynak *et al.*, 1996). A number of studies have shown high levels of variation in particulate, and other types of, air pollution within relatively small areas (e.g. sub-city level). Blanchard *et al.* (1999), for example, showed that concentrations of PM_{10} varied by up to 20% in the San Joaquin Valley in California. Burton (1996) also showed spatial variation

Table 2.3: Automatic monitoring (AUN) sites in operation in the UK

Pollutant	***Urban sites***	***Rural sites***	***Total***
Total number of sites	93	19	112
Ozone	52	19	73
Nitrogen Oxides	80	7	87
Carbon Monoxide	65	0	65
Sulphur Dioxide	60	7	67
Particles as PM_{10}	52	3	55
Hydrocarbons	12	1	13

Source: AEA technology

of fine particulates in Philadelphia (USA) to be large. Furthermore, Janssen *et al.* (1997,1998) showed that, although strongly correlated, there were large differences in the magnitude of PM_{10} concentrations between ambient monitoring and personal exposure for non-smoking adults and children. A study in Holland by Roorda-Knape (1998) showed that concentrations of NO_2 declined rapidly from major roads, but, converse to many studies, there was little spatial variation in fine particles. The reason for this was attributed to the high proportion of a fairly uniform background concentration of coarse particles. A study of spatial variation in concentrations of indoor and outdoor pollutants in Huddersfield (UK), by Kingham *et al.* (2000), showed that there was only a small decay in levels of outdoor pollution with distance from main roads, thus supporting studies that have only used one or two monitoring stations. However, the Huddersfield study represented only part of the town along a single corridor and may not be representative of other parts of the same town, or situations in other areas.

In essence, the appropriateness of ambient monitors as a measure of exposure depends on the specific characteristics of a study area. The magnitude of variation in air pollution within an urban area will depend on the size of urban areas, diversity of traffic levels, and other sources of pollution. Another important consideration is that, for most people, exposure is not only a function of the levels of air pollution at outdoor locations but also indoor locations. Levels of air pollution indoors have been shown to have a strong association with the immediate outdoor

vicinity but a weak association with fixed-site monitors at distance (Mark *et al.*, 1999).

The use of fixed-site monitors can, therefore, lead to misclassification of exposure in areas where there is significant variation in air pollution. Harrison and Deacon (1998) suggested that the number of monitors must be large to cover all the spatial variation within cities. One possible improvement over current methods is to compliment automatic monitoring with additional monitoring equipment.

Boudet *et al.* (2001), as part of the European EXPOLIS programme, proposed a method to enhance ambient concentration data used to estimate personal and population exposure to particles. In Grenoble, France, 40 adult volunteers were equipped with $PM_{2.5}$ monitors and time-activity diaries for a 48h period. Individuals' exposure was categorised by either a roadside (proximity) or a background monitoring site according to the location of their places of work and residence. The study concluded that differentiation between proximity and background locations was an improvement on classifying exposure by a single ambient monitoring station, but further investigation was needed in contrasting urban areas.

An alternative to automatic monitoring is low cost methods such as passive samplers (e.g. Palmes tubes). With this approach, a far denser network of monitoring can be achieved at relatively low cost. Passive samplers have been shown to provide a good estimate of annual concentrations (de Hoogh, 1999). In the UK, a national passive sampler network for NO_2 has been established at over 1000 sites. This has vastly increased the information available on levels of NO_2 around the UK. The main limitation of passive techniques, however, is that they only provide reliable measurements over periods of about two weeks. Thus, they do not meet the requirements for short-term (i.e. daily or hourly) exposure assessment. More crucial here is that there is no tube device that can be used on mass for particles.

b) Personal and microenvironment monitoring

Ideally, human exposure to air pollution would be obtained directly from measurements of personal (i.e. individual) exposure to air pollution. This method tends, however, to be too costly and time-consuming to use in most exposure studies. Personal monitoring nevertheless provides both important information in exploring exposures in different microenvironments and inputs to exposure models. Those studies that have used personal monitors have tended to use low-cost, low-precision passive devices such as tubes (Linaker *et al.*, 1996) or badges (Raachou-Nielsen, 1996), which (as mentioned in the previous section) suffer from poor temporal resolution and are not available for particles.

A number of studies have, however, used other small, lightweight devices for measuring personal exposure to particles. A pump is generally used to draw air continuously through a sampling head onto a filter. The filter is then weighed in a laboratory and a mass concentration is determined. *Table 2.4* shows a summary of those studies where personal exposure to PM_{10} has been measured, giving the average PM_{10} exposure for each study. In all of these studies exposure is only represented by an average measure over 24 hours. The sampling period is constrained by the time needed to collect sufficient material to be weighed accurately, and the need to do this under laboratory conditions. These measures give a total or average exposure on a daily basis, but do not provide information on exposure in specific microenvironments. A measure of exposure on a daily basis does not reflect the short-term (i.e. hourly) peak exposures that may be the important factor in terms of health effects of particulate air pollution.

In these studies of personal exposure, the range of exposures is from 42 to 114 $\mu g/m^3$. These levels are generally higher than recorded at fixed-site locations, or in locations of close proximity to the individuals. The average ratio of mean personal exposure to the corresponding indoor or outdoor concentration is in the range of 1.2 to 3.7. It is not known what proportion of the elevated levels of PM_{10} are from localised sources or merely a result of the *personal cloud* effect. The *personal cloud* effect is where dead skin and fibres from clothing fall in the wake of the monitoring equipment.

Table 2.4: Studies where personal exposure to particles has been measured

Author	***Study area***	***Study group***	***No. of subjects***	***No. of measurements***	***Average PM_{10} exposure ($\mu g/m^3$)***
Lioy *et al.*, 1990	Philipsburg, NJ, USA	Non-smoking adults	14	189	64
Pellizzari *et al.*, 1993	Riverside, CA, USA	Non-smoking adults	178	171	114
Linn *et al.*, 1996	Los Angeles	Adult COPD patients	45	45	42
Janssen *et al.*, 1997	Amsterdam, Netherlands	Healthy children	45	301	62
Mark *et al.*, 1997	Birmingham, UK	Healthy adults	30	178	54
Janssen *et al.*, 1998	Amsterdam & Wageningen, Netherlands	Non-smoking adults	37	262	103
Seaton *et al.*, 1999	Edinburgh & Belfast, UK	Non-smoking adults (over 60 years)	111	111	55

Source (adapted from DETR, 2000b)

These particles are considered to be relatively large in size and would be quickly removed from the atmosphere; thus, they are not contributing to the ambient particle concentration.

More recently, active portable analysers have become available that have capabilities more in line with automatic fixed-site monitors. Such devices can measure, and store, results against a time reference, or provide continuous measurement in real-time. Even though many of these instruments were designed for occupational use, they can be used in a range of indoor and outdoor environments for personal exposure monitoring. For example, as part of the Northampton Air Pollution Study (NAPS), Ashmore *et al.* (2000) used small, portable, occupational monitors (resolution = 1ppm), by Drager Ltd, to measure personal exposure to CO at 10 second intervals. The advantage of this device is that it can be carried by individuals during their normal daily activities.

There are also a growing number of devices for particles, though many of these are not suitable for personal monitoring as the weight and bulk of equipment would restrict normal activity. Nonetheless, these portable devices offer great potential in microenvironment monitoring. The basic idea of microenvironment monitoring is that instead of measuring the exposure at the individual, the measurement is taken in the immediate vicinity of, or within a representative hull of, the area that they occupy (Otto, 1982). In this way, portable devices do not restrict normal activity and can be transferred between locations as the individual moves from one location to another.

c) Monitoring in transport microenvironments

It may be short, peak exposures during certain activities that are most important in terms of health effects of PM_{10}. It is thought that a large proportion of peak exposures occurs during journeys. Only a limited number of studies have, however, investigated the exposure to particles during journeys. The majority of these studies have dealt solely with exposure inside motorised transport (Akbar and Ashmore, 1996; Praml and Schierl, 2000; Zagury *et al.* 2000), with only a few studies looking at the exposure of cyclists (Bevan *et al.*, 1991; van Wijnen *et al.*, 1995, Adams *et al.* 2001) and pedestrians (van Wijnen *et al.*, 1995).

Table 2.5 shows a summary of particulate exposure assessment studies in transport microenvironments. Comparison between the results of these studies is difficult because many of them use different fractions of particulate matter. Nevertheless, there are some clear patterns in these results in terms of both the magnitude of concentrations in different types of transport, and the magnitude of difference between concentrations in-transit and those measured by fixed-site monitors. On the whole, concentrations in buses and cars are about the same, but concentrations for cyclists tend to be somewhat lower than those found in motorised road transport. Several studies have shown in-transit concentrations to be markedly higher than those from fixed-site monitors (Zagury, 2000; Adams *et al.*, 2001).

Table 2.5: Summary of studies of personal monitoring in transport microenvironments

Study	Location	Particulate measure	Total number of transport samples	Transport modes	Concentration means and variability ($\mu g/m^3$)	Sampling time	Fixed-site monitor (FSM) ratio / other comments
Morandi *et al.* (1988)	Houston, US	$PM_{3.5}$	94	Car	35 (S.D. =23)	Various, single journeys	Car / indoor/ outdoor 35:27:22
Bevan *et al.* (1991)	Southampton, UK	$PM_{3.5}$	16	Bicycle	135 (range: 13-253)	36 min	Spot measurement: 541 behind car exhaust, 215 on High street, 23 in park
van Wijnen *et al.* (1995)	Amsterdam, NL	PM_{10}	22	Car	Urban routes: Summer 90-194 Winter 17-62	1 h	No difference in concentration of PM_{10} between inner city and urban routes, but lead and PAH content were 7 times higher on the inner city route.
Akbar and Ashmore (1996)	Delhi, India	PM_4	14 10 17	Bus Car Open-vehicle	354 (S.D. = 187) 389 (S.D. = 249) 782 (S.D. = 554)	1 h	FSM 20% less than bus and car.
Bizjack and Tursic (1998)	Ljubljana, Slovenia	Elemental Carbon (EC)	Not stated	Bus	Summer 10-40 Winter 20-50	Various	Much lower EC in newer buses than old ones
Kingham *et al.* (1998)	Huddersfield, UK	** Absorbance values of inhalable dust	6 6 5 6 6	Bus Car Train Bicycle Bicycle	7.6 (S.D. = 4.4) 5.7 (S.D. = 2.0) 5.3 (S.D. = 3.0) 6.3 (S.D. = 4.6) 2.7 (S.D. = 2.0)	Not stated	** Absorbance values, not PM

Study	Location	Particulate measure	Total number of transport sample	Transport modes	Concentration means and variability ($\mu g/m^3$)	Sampling time	Fixed-site monitor (FSM) ratio / other comments
Priest et al. (1999)	London, UK	PM_9	7	Underground rail (Tube)	500-1120	Approx. 3h	Tube concentrations higher than surface concentrations.
Sitzmann *et al.* (1999)	London, UK	PM_5	4 2	Bicycle Underground rail (Tube)	14,16,16,89 709, 893	7.5 h	
Pfeifer *et al.* (1999)	London, UK	$PM_{2.5}$	20 4	Taxi* Underground rail (Tube)	33 (S.D. = 21) 246 (S.D. = 52)	7 days in total, 16h	*not just journeys - integrated over 7 full days
Gee et al (1999)	Manchester, UK	PM_4	31	Car	42 (19-65)	Averaged over 10 journeys	
Gee and Raper (1999)	Manchester, UK	PM_4	34 8	Bus Bicycle	338 (S.D. = 300) 54 (S.D = 34)	3 h	Very high concentrations of PM inside bus
Praml and Schierl (2000)	Munich, Germany	PM_{10}	117	Bus	153	4 h	Bus / FSM 4:1
Zagury *et al.* (2000)	Paris, France	Black Smoke	28	Taxi (diesel)	168 (S.D. = 53)	8 h	Concentrations in taxis were, on average, twice those at the urban centre monitor, and four times those at suburban sites.

Study	Location	Particulate measure	Total number of transport sample	Transport modes	Concentration means and variability ($\mu g/m^3$)	Sampling time	Fixed-site monitor (FSM) ratio / other comments
Adams et al. (2001)	London, Uk	$PM_{2.5}$	36	Bus	Summer = 39.0 (G.S.D = 1.8)	Various, between several minutes and 1 h per sample	Urban Kerbside (Marylebone):
			32		Winter = 38.9 (G.S.D = 2.1)		Summer = 24
			42	Car	Summer = 37.7 (G.S.D = 1.5)		Winter = 37
			12		Winter = 33.7 (G.S.D = 2.4)		Urban centre (Bloomsbury):
			40	Bicycle	Summer = 34.5 (G.S.D = 1.7)		Summer = 15
			56		Winter = 23.5 (G.S.D = 1.8)		Winter = 13
			10	Underground rail (Tube)	Summer = 247.2 (G.S.D = 1.3)		Road transport elevated by about 100% compared to levels at urban centre
			12		Winter = 157.3 (G.S.D = 3.3)		

Source: modified version from Adams et al. (2001)

Three of these studies are particularly worthy of further mention. Firstly, the study by van Wijen *et al.* (1995) is one of the few studies that compares exposure to a range of pollutants across the full range of transport (walking, cycling, and in-car), whilst also considering breathing rates of subjects. The second is a study of Paris taxi drivers by Zagury (2000) because it deals with a group of users that spend considerable time in transit, and it makes comparisons between journey exposures and exposures from ambient monitoring. Finally, the study by Adams *et al.* (2001) will be described because it is the first comprehensive study of multi-modal particulate exposures to be carried out in the UK.

The study by van Wijen *et al.* (1995) was conducted on three routes (two inner city, one rural) in Amsterdam, The Netherlands. Sampling took place on four days each week during morning and evening rush hours, over two-week periods in January and May of 1993. Concentrations of CO, NO_2 and benzene were measured for both cyclists and car drivers, on all routes, with PM_{10}, PAH (Polycyclic Aromatic Hydrocarbons), and lead measured only for car drivers on one inner city route and the rural route. Where PM_{10} was not monitored, CO was regarded as a good marker for traffic-related particles. Ventilation rates and heart rates were measured for a sample of car drivers and cyclists during journeys. Results showed that, on all routes, the concentrations of CO and benzene were much higher for car drivers than cyclists. This difference was accentuated on the inner city routes, particularly in periods of very busy or queuing traffic. In a similar study, Rudolf (1986) reported that car drivers were exposed to twice the level of CO that was experienced by cyclists.

In the Dutch study, concentrations of NO_2 were slightly higher for cyclists on the inner city route, which was attributed to the rapid formation of NO_2 from oxidisation of NO emitted in the wake of the road. There was far less difference in concentrations of all pollutants between modes on the rural route. Of particular interest is that cyclists exhaled, on average, 2.3 times more air than car drivers exhaled; therefore, the suggestion is of a higher intake of pollution for cyclists. Pedestrians are also likely to have increased breathing over that of car drivers, albeit somewhat lower than breathing rates of cyclists. Therefore, exposure assessment needs to be considered not only in terms of personal concentration

but also breathing rates. There is, however, currently no other data relating concentrations of particles to breathing rates in different modes of transport. Finally, there were some results for PM_{10} and related compounds. The range of PM_{10} measurements did not differ between the busy inner city route and the quiet rural route. However, on busy routes, lead and PAH were, on average, about 7 times higher than on the rural route.

The second example comes from a recent study by Zagury *et al.* (2000) of Paris taxi drivers' exposure to CO, NO, NO_2, and black smoke (BS) within their vehicles. Black smoke is seen as a surrogate for particles. The study took place over a two-month period in early 1997, using 29 randomly selected drivers who worked during the day. CO was measured using a portable monitor (Drager PAC II), with a sampling period of three seconds, and black smoke was measured using a portable pump monitor with a filter. The results for black smoke were expressed in terms of the French BS index. NO_2 and NO were measured with passive samplers. Ambient air pollution measurements were taken from three automatic sites (one in the centre, and two suburban background sites) that form part of the AIRPARIF monitoring network. Results show that, for all pollutants, the in-car measurements were, on average, twofold higher than those at the central monitoring site, and about four times the concentrations found at background sites. Although concentrations for all pollutants were generally high, only BS was in excess of the threshold recommended by WHO. Correlations between in-car measurements and background sites were on the whole good, but only moderate correlation was found between in-car measurements and the central site. Only CO gave good correlation across all sites ($r > 0.57$). Overall, these results show that taxi drivers were exposed to relatively high levels of exposure for all pollutants, and that ambient pollution monitors severely underestimate in-car concentrations of black smoke and other pollutants.

The final example comes from a multi-modal study of personal and microenvironment exposure to $PM_{2.5}$ in London, UK (Adams *et al.*, 2001). For the purpose of the study, a gravimetric monitor was developed in-house with a sufficiently high flow rate to give short-term exposure estimates. A total of 465 journeys were completed by 61 volunteers during two, 3-week campaigns (one in

summer and one in winter). Four different modes of transport were used (bicycle, bus, car, London underground Tube), at four different times of the day, over three contrasting routes (one central and two outside the centre). In summary, levels of exposure to $PM_{2.5}$ were about the same inside the bus and car in both seasons, whereas exposure for cyclists was significantly lower than the exposure inside both the bus and car in summer, and marginally lower than both bus and car exposure in winter. Lower levels of exposure for cyclists were explained by the position of the cyclist in the road, and the ability of cyclists to avoid sitting in traffic jams. It is important to note, once again, that the intake dose for cyclists may be equivalent to, or even exceed, the levels of exposure for people travelling by bus and car due to higher breathing rates associated with cycling. To a lesser degree the same can be said for pedestrians' exposure. Pedestrians are, however, likely to be exposed for longer journey periods than cyclists are; thus, the target dose for pedestrians and cyclists may be similar over an equivalent route and distance.

In complete contrast, concentrations in the Tube were between 5 and 8 times those found on any other mode of transport above ground. In addition to between-mode variation, between-route and intra-day variability were also analysed. A large amount of between-route variation was seen in both winter and summer. Concentrations averaged across all modes of transport except Tube were 1.5 times greater in the central route than those found in quieter areas. There was no significant intra-day variability in mean exposure levels. This was explained by the lack of peak traffic and fairly even levels of traffic in London across the day.

The study also made comparisons between concentrations levels in each transport mode and those found at two fixed-site monitors (Bloomsbury and Marylebone). Neither of the fixed-site monitors reflected peak exposure levels. Concentrations of $PM_{2.5}$ at Bloomsbury (urban centre) were on the whole about 50% of concentrations found in buses and cars, and about 75% of the levels of exposure experienced by cyclists. During the summer, concentration levels at Marylebone (kerbside) were about two-thirds of the level of concentrations found in buses and cars, but similar to concentrations recorded for cyclists. In the winter, levels of $PM_{2.5}$ concentrations were about the same between Marylebone and motorised transport. This study has shown that there is considerable variation in

$PM_{2.5}$ exposures between transport microenvironments and fixed-site monitors. There are also differences in exposure between modes of transport, and by route. Even though this study measured exposure to $PM_{2.5}$, similar patterns ought to be observed for PM_{10}, albeit with a lower magnitude in variation across microenvironments due to the presence of a significant contribution of a coarse component in the PM_{10} measure.

Notwithstanding the considerable amount of new information that these monitoring studies have provided on transport microenvironment exposures, there remain a number of problems that inhibit their use as a primary means of estimating exposure. These problems relate to all types of microenvironment but are most severe in transport. Firstly, there are problems in using many of these portable devices in transit, particularly where the individual is walking or cycling. Secondly, extensive sampling would be required to collect enough data to represent the range of exposures that occur in a potentially large number of contrasting microenvironments. Other fundamental issues are the cost of equipment and the time taken to collect large numbers of samples. These techniques are, therefore, best suited to studies exploring exposures in different microenvironments, or assessing exposures for a small sample of subjects. Where exposure estimates are required for a large number of individuals, alternative methods such as exposure modelling should be considered. Portable devices are, nonetheless, a potentially useful means of testing and validating models of exposure.

2.3.3 Indirect methods of exposure assessment

a) Indicators

Indicators are indirect measures of exposure based on the relationship between source and surrounding locations. Commonly used indicators include distance to busy roads, and traffic density or traffic volume, and self-reporting of proximity to traffic. A number of recent studies have reported associations between respiratory symptoms and both distance of residence to roads (Livingstone, 1996; Nitta *et al.*, 1993; Murakami, 1990), and traffic volume (Wjst, 1993; Brunekreef *et al.,* 1997).

Other studies have seen associations with respiratory symptoms and self-reporting of proximity to traffic (Weiland, 1994; Duhme, 1996). The major limitation of indicator-based techniques, however, is that they cannot be easily related to levels of air pollution; therefore, they cannot be directly used as a quantitative measure of exposure. The other main limitation is that they cannot be related to specific pollutants. Indicators are therefore only suitable in the context of chronic effects of exposure.

b) Bio-markers

Biological markers (or bio-markers) can be grouped into markers of exposure and markers of effects (Monn, 2001). A marker of an effect is essentially an indicator, such as lung function, which can be used as a basis for diagnosis. Wallace *et al.* (1989), for example, used VOCs in exhaled breath to assess personal exposure to VOCs. A marker of exposure reflects the concentration of a substance that is found on or within the human body. Bio-markers have been collected from skin, hair, and blood as proxies for exposures to air pollution. Yanagisawa *et al.* (1988), for example, used hydroxy-proline as a biological marker for exposure to NO_2 and tobacco smoke in urine sample. Higher levels of hydroxy-proline were shown in persons living near main roads. Although bio-markers are excellent in terms of giving an integrated exposure measure over time, they are not suited to short-term exposure assessments because changes in many bio-markers are difficult to detect over short periods of time (i.e. a few hours). There is also the obvious logistical problem of taking repeated measurements for large samples of subjects.

c) GIS-based techniques

A number of GIS-based techniques have been developed for exposure assessment. These fall into two main categories: spatial interpolation and regression mapping. Spatial interpolation is the process of making estimates at unsampled locations based on monitored data. Geostatistical techniques, such as Kriging, have often been used in modelling air pollution (Liu ad Rossini, 1996; Collins 1998, de Hoogh 1999) as they inherently attempt to model local variation rather than fit smooth, global surfaces (e.g. trend surface analysis) to the field of

pollution. Co-Kriging, where one or more additional variables are used with data on monitored concentrations, has been shown to have some potential in modelling air pollution. Both de Hoogh (1999) and Collins (1998) have used monitored NO_2 concentration in conjunction with measures of road and traffic density to predict concentrations of NO_2. Both of these studies showed that co-Kriging gave reasonable estimates of NO_2 concentrations, but better results came from regression-based techniques.

Regression mapping involves establishing a relationship between predictor variables (e.g. traffic volume, road density, land use) and monitored data, and applying the model at independent locations. Regression mapping has been used by Briggs *et al.* (1997, 2000), as part of the SAVIAH study, in modelling NO_2 in four contrasting urban environments. Results show that the method performs well, but requires monitored data for local calibration and validation of the model. Both regression and geostatistical techniques can be used as an indicator of chronic effects but are not suitable for assessing short-term effects, as, essentially, they do not reflect the influence of meteorology.

d) Dispersion modelling

Dispersion modelling is an attempt to simulate the movement of air pollution from source to a target (location) using mathematical models. Dispersion models can model both temporal and spatial variations in air pollution at high resolution (e.g. hourly); thus, they are a potentially useful tool in assessing the acute effects of air pollution on health. In terms of the scale of transportation in which they operate, dispersion models are generally categorised as either short-range (<100km) or long-range (greater than 100km). Commonly, short-range dispersion models use a Gaussian distribution to model the plume of pollution, whereas long-range transport models use either Lagrangian or Eulerian methods. Lagrangian and Eulerian models are similar in that both look at chemical and physical process taking place within an air mass as it moves through a given trajectory. They only differ in terms of the way that the mass of air is represented in space and time. The Lagrangian approach follows the air mass continuously over a trajectory, whereas the Eulerian approach looks at fixed points over specific time intervals

(Harrison, 1990). Studies at the local scale (i.e. city-wide) are, therefore, more suited to the use of Gaussian models.

In the Gaussian plume model, the plume has a Gaussian, or normal, distribution of concentration in the vertical (z) and lateral (y) directions. The concentration *C* at any point (*x,y,z*) is then given by:

$$C(x,y,z) = \frac{Q}{2\pi\sigma_y\sigma_z U}\exp\left[-\frac{y^2}{2\sigma_y{}^2}\right]\left\{\exp\left[-\frac{(z-H_e)^2}{2\sigma_z{}^2}\right]+\exp\left[-\frac{(z+H_e)^2}{2\sigma_z{}^2}\right]\right\}$$

C concentration ($\mu g\ m^{-3}$)

Q pollutant mass emission rate ($\mu g\ s^{-1}$)

U wind speed ($m\ s^{-1}$)

x,y,z the along wind, crosswind and vertical distances (m)

H_e effective stack height

σ_y standard deviation of horizontal concentration in the plume (m)

σ_z standard deviation of vertical concentration in the plume (m)

When y = z = 0 the concentration calculated will be at ground level and the equation reduces to:

$$C(x) = \frac{Q}{\pi\sigma_y\sigma_z U}\exp\left[-\frac{H_e{}^2}{2\sigma_z{}^2}\right]$$

A number of chronic exposure studies have used dispersion modelling techniques. Collins (1998), for example, used an adaptation of CALINE3 within a GIS to estimate concentrations of NO_2 and compare results with those from other GIS-based techniques (e.g. Kriging). Oosterlee *et al.* (1996), used the Dutch CAR model (Eerens *et al.*, 1993) to model NO_2 exposure of 673 adults and 106 children living along heavily-trafficked streets in the city of Haarlem, with a control sample of 812 adults and 152 children along quiet streets. Questions relating to respiratory symptoms were asked by postal questionnaire. The results of the study showed that children living along heavily-trafficked streets had a higher prevalence of respiratory symptoms than children did living in quiet streets.

A recent detailed study of exposure using dispersion modelling, by deHoogh (1999), compared the performance of several models (ADMS, DMRB, CALINE-4) against monitored NO_2 from Sheffield, UK. The study incorporated data on respiratory symptoms from 1800 children within the Sheffield area, but no significant associations were found between exposure measures and health outcomes. The study did show, however, that all of the dispersion models were generally reliable in estimating long-term average concentrations. A study of children's exposure in Copenhagen (Raaschou-Nielson *et al.*, 2000) used the Danish model OSPM (Operation Street Pollution Model) to provide estimates of NO_2 outside the front door of 100 children in urban areas, and 100 children in rural areas. Personal monitoring was undertaken using NO_2 badges. Palmes tubes were used to monitor NO_2 outside the home as a basis for model validation. The correlation coefficient (r) between predicted and monitored half-year concentrations of NO_2 was between 0.75-0.8. In contrast to assumptions made by other research, this study found that NO_2 outside the home was of limited use as a marker for the exposure of children.

Notwithstanding the wide range of models that are available, examples of using dispersion modelling in studies of short-term exposure are limited to only a few. There are a number of possible reasons that restrain their use in this context. Primarily, there is concern about the validity of the output, particularly in the ability of dispersion models to predict short-term exposures. This is, in part, manifested in the data requirements of models. To make short-term estimates, models need very specific and detailed data on meteorological conditions (e.g. wind speed, wind direction, boundary layer conditions), road source geometry, and traffic composition and emission rates. Dispersion models are also self-limiting. It is not possible to embark on any modelling where some of these data are not available, or where data do not provide sufficient coverage. In the UK, for example, there is a relatively sparse coverage of meteorological stations that hold sufficient information to feed a dispersion model. The nearest Meteorological Office site would typically be within tens of kilometres. Furthermore, meteorological conditions tend to be recorded at exposed, rural locations, which can have vastly different conditions from those found in urban areas. Equally, good quality traffic data is difficult to procure. Many local authorities, for example, have databases of

traffic composition on main roads, which are either from traffic modelling, or based on extrapolation from limited monitoring. Models of traffic are typically static in that they represent long-term situations but do not reflect day-to-day changes.

Nevertheless, where there are sufficient inputs, dispersion modelling is a potentially fast and resourceful way of making estimates of exposure. In view of the problems with personal and fixed-site monitoring data, dispersion modelling is potentially the most feasible way of making short-term (i.e. hourly) estimates of air pollution at a large number of locations. With the advent of GIS, some dispersion models now have the ability to produce maps of air pollution, from which large numbers of exposure estimates can, potentially, be made. Indeed, as the next section will demonstrate, GIS have increasingly become an important tool, alongside dispersion models, in modelling exposures.

2.4 EXPOSURE MODELLING

2.4.1 Characteristics of exposure models

Some of the methods described above are used in developing and providing inputs to exposure models. A growing number of exposure models exist, including those that deal exclusively with indoor exposure, those that account for both indoor and outdoor exposure, and those which deal with exposure to air as part of a range of multi-media (e.g. food, water, soil). In essence, there are two components in an exposure model: 1) a model that estimates exposure in different microenvironments and, 2) a model that describes the distribution of the population across these microenvironments. Models of microenvironment exposures can be characterised as either physical or statistical models (Ott, 1990).

Typically, statistical models determine exposure in each microenvironment as a function of air pollution levels at outdoor, fixed-site monitors. Exposure in each microenvironment will then either be assumed to be the same as the outdoor monitor, or be derived from empirical data on ratios of pollution levels at the outdoor site and those found different microenvironments. Physical exposure

models, on the other hand, attempt to model the physical processes that determine levels of air pollution in different microenvironments. They may include information on a number of factors that influence microenvironment exposures: indoor and / or outdoor sources, atmospheric and meteorological conditions, building fabric, and ventilation rates. Physical models tend, however, to use generalised descriptions of each microenvironment due to the inherent complexity in attempting to describe the potentially vast number of conditions present in each microenvironment at any one time.

Population activity data come from one of two sources: 1) purpose-designed time activity surveys, typically consisting of several thousand individuals, or, 2) demographic groups, from a census, characterised by their activity. Some models simply use deterministic methods to generate exposure profiles directly from a finite body of time-activity data. Others use stochastic methods, such as Monte-Carlo techniques, to simulate exposure profiles for a desired study population.

The standard method of estimating exposure for an individual, or population, is to express exposure as an integrated measure over time. This method is based on the equation by Duan (1982):

$$E_i = \sum_{j=1}^{J} c_j * t_{ij}$$

Where;

E_i is the integrated exposure of person i
c_j is the concentration in microenvironment j
t_{ij} is the time spent by person i in micro-environment j
J is the number of microenvironments visited

Additionally, some models estimate intake dose on the basis of microenvironment activities by multiplying inhalation rates (or breathing rates) with air pollutant concentration. Some models use even more sophisticated techniques that require information on body mass, age, gender and the health status of each individual to determine measures of target dose.

2.4.2 Overview of models

The lead in developing exposure models has been provided by researchers in the USA. One of the first formal exposure models was NEM (National Exposure Model) developed by the US Environmental Protection Agency (EPA) (Johnson, 1995). NEM is a multi-source, pollutant-specific model for estimating population or cohort exposures to CO, NO_2, O_3 and particulate matter, on an hourly basis. The Hazardous Air Pollution Exposure Model (HAPEM) uses a similar methodology to NEM but differs in the averaging time used for exposure estimates (Johnson, 1995). Instead of making hourly estimates of exposure, HAPEM has a minimum resolution of 3 months. HAPEM is, thus, suited to modelling the long-term effects of exposure to pollutants. HAPEM simply represents the exposure history of each population group as a fraction of time spent multiplied by the pollutant concentration in each microenvironment, whereas the NEM series of models express exposure in terms of dose as well as exposure concentration. The early versions of NEM were deterministic, generating profiles of exposure for a population cohort from the activity of a finite number of individuals. Later versions of NEM, known as *probabilistic*NEM (*p*NEM) (McCurdy, 1995), use stochastic methods to select the values for variables that are entered into the model. The variance of exposures within each cohort is computed by Monte Carlo Simulation. Using random numbers, samples are repeatedly taken from distributions of activity and used to generate frequency distributions of exposures. The variance of the frequency distribution is taken to describe the uncertainty in exposure estimates. This is the standard approach taken by many existing models.

The other main exposure model from the US is SHAPE (Simulation of Human Activity and Pollutant Exposure), first described by Ott (1982). SHAPE is similar to the NEM series of models, but has been designed to accept real activity data as well as simulated activity data, so it can be used to make exposure estimates for individuals as well as groups of the population. Several other models have used the NEM / *p*NEM approach. Noteworthy models are REHEX (Fruin, 2001) from California Environmental Protection Agency, AirPEX (Feijer *et al.*, 1998) the Dutch exposure model, and BEADS (Benzene Exposure and Absorbed Dose Simulation)

(MacIntosh *et al.*, 1995). BEADS is functionally the same as SHAPE but only uses probabilistic methods to model activity patterns.

Many existing models rely on air pollution data from fixed-site monitors as the basic input. As already mentioned, networks of monitors are sparsely populated; therefore, they do not adequately represent the variation in levels of air pollution in many urban areas. Furthermore, indoor air has been shown to be a function of outdoor air in close vicinity, but the relationship between indoor and outdoor air deteriorates with distance (Mark, 1997). An important development in this context is the use of exposure models together with Geographical Information Systems (GIS). It is possible to couple dispersion modelling with GIS techniques, for example, to produce point estimates of pollutants at, potentially, any number of locations. In this way, microenvironment concentrations can be considered in terms of the ambient outdoor concentration in the immediate vicinity of the microenvironment, rather than using the nearest fixed-site monitor. A number of new models exist that take this approach, including AirGIS (Jensen *et al.*, 2001) from Denmark, and the Norwegian model AIRQUIS (Clench-Ass *et al.*, 1999). A summary of selected exposure models is given in *Table 2.6.* A number of models will now be described in more detail.

2.4.3 Exposure models

a) NEM / pNEM (National Exposure Model)

The NEM / *p*NEM group of models were developed by the US EPA to simulate the movement of specific population sub-groups (cohorts) through 'zones' of varying air pollution. These 'zones' are defined both by geographical location and microenvironment. By the mid-1990s, thirteen versions of NEM / *p*NEM existed covering five pollutants (CO, Pb, NO_2, O_3, PM_{10}). In *p*NEM, the exposure history of each cohort is represented by a series of 'exposure events'. Each exposure event is characterised by a start-time, duration, geographic location, microenvironment, and breathing rate of a cohort. Therefore, *p*NEM is particularly suited to assessing short-term exposures. The output of each exposure event is a dose estimate.

Table 2.6: Summary of exposure models

Model	Developer	Micro-environment			Time-activity		Geographical basis			Resolution		Comments	Reference
		Outdoor	indoor	travel	Stochastic	survey	point	line	area	Spatial	Temporal		
AirPEx	RIVM, NL	✓	✓			✓	✓			nearest monitoring station	hourly	Daily activity patterns of 4985 individuals. Location and level of activity from a 24h period. Individual and population exposures.	Freijer *et al.* (1998)
TRIM.expo	US EPA	✓	✓		✓		✓		✓	Population -level	hourly	Under development. The basic model is pNEM, with enhancements such as modelling outdoor concentrations using GIS.	Efryonson *et al.* (2001)
SHAPE	US EPA	✓	✓	✓	✓	✓	✓			nearest monitoring station	24hour average	Activities can be generated from a population based profile of activity, using Monte Carlo methods, or from activity at the individual level	Ott (1982, 1984); Ott *et al.* (1988)
AirGIS	Denmark	✓	✓				✓	✓		1m	hourly	Estimates number of people at address locations and outdoors in streets (by road segment) from Census, instead of time-activity data. Uses OSPM dispersion model for multi-pollutant modelling.	Jensen (1998), Jensen *et al.* (2001)
INTAIR	UK	✓	✓	✓	✓	✓	✓			Selected urban and rural sites	Annual mean	Models the physical processes in and between microenvironments.	Dimitroupolou *et al.* (2001)

Model	Developer	Micro-environment			Time-activity		Geographical basis			Resolution		Comments	Reference
		Outdoor	indoor	travel	stochastic	survey	Point	line	area	Spatial	Temporal		
SHEDS-PM	US	✓	✓		✓		✓			Region	n/a	Models exposure to fine particulates.	Zartarian *et al.* (2000)
pNEM	US EPA	✓	✓	✓	✓		✓			Population -level	hourly	Single-pollutant models for CO NO_2, O_3 and particulate matter. Simulated time-activity using Monte-Carlo methods.	McCurdy (1995); Johnston (1995) Law (1997)
BEADS	US EPA	✓	✓		✓		✓			County	24-hour average	Population-level exposure-dose model. Functionally similar to SHAPE. Single pollutant -benzene	MacIntosh *et al.* (1995)
AirQUIS	NIAR, Norway	✓				✓	✓			Address - level	1 hour	Loose-couples a traffic model, emissions inventory, dispersion model within a GIS. Links individual exposures to time activity diaries.	Clench-Aas *et al.*. (1999)
HEM	US EPA	✓					✓		✓	ED centroid	Annual average	Estimates from dispersion modelling at each ED centroid represent the population at ED-level (US Census). Default dispersion model is ISCLT	www.chemicalawarenees.org/toolkit//php/onepagers/hem.html www.nap.edu/openbook/030904894X/html/375.html
CalTOX	US EPA	✓			✓		✓			Nearest monitoring station	n/a	Multi-media pathway model.	Loranger *et al.* (1997)

Model	Developer	Micro-environment			Time-activity		Geographical basis			Resolution		Comments	Reference
		Outdoor	indoor	travel	Stochastic	survey	Point	line	area	Spatial	Temporal		
HAPEM	US EPA	✓	✓			✓	✓			50km circular buffer (from city centre) split into exposure districts	Annual mean	CO exposures used as a surrogate for all other air pollutants.	www.nescaum.org/committees/MSPeerReview/Sec_5_HAP.html
GRAM	Bombay, India	✓							✓	Intra-urban zones	Annual mean	Intersects population districts of Bombay, India, with pollution maps (from dispersion modelling) to derive population exposure estimates	Sengupta *et al.* (1997)
REHEX	US EPA	✓	✓	✓	✓		✓			Region	15-minute	Based on pNEM. Exposures modelled by demographic group. Uses monitoring stations to predict micro-environmental exposures.	Fruin *et al.* (2001)
RCS	US EPA/ Stanford University	✓	✓		✓		✓			nearest monitoring station	24-hour average	Statistical model used to extrapolate population exposure distributions to cities where monitored exposure data are not available	Ott *et al.* (2000)

In *p*NEM, indoor air is a function of outdoor air. Early versions of *p*NEM used either I/O ratios from empirical data, or regression-based relationships derived from personal exposure monitoring, to estimate indoor concentrations based on concentrations outdoors. Later versions of *p*NEM use a mass-balance approach, that considers the movement of air from outdoors to indoors in relation to the air exchange rate (ventilation), indoor emissions sources, and the indoor decay factor representing pollutant removal mechanisms. This is, essentially, a simplified version of the model used in INTAIR (see below). *p*NEM does not consider the movement of air between adjacent indoor microenvironments. In all cases, the nearest fixed-site monitor would be used as the basic input for indoor microenvironment modelling.

In addition to providing exposure estimates as pollutant concentrations, *p*NEM also characterises exposure by equivalent ventilation rate (EVR). In essence, EVR is a function of breathing rate (or heart rate) and body surface area. If both pollutant concentration and EVR are known it is possible to make an estimation of intake dose. Intake dose (D_{liT}) for any individual is defined as:

$$D_{liT} = \frac{1}{T} \sum (\overline{C}_{jt} * t_{iaj} * V_{Eiat})$$

Where;

$\overline{C}_{jt}$ = average concentration $\overline{C}$ in ME *j* during time period *t*

t_{iaj} = time *t* spent by individual *i* undertaking activity *a* in ME *j*

V_{Eiat} = ventilation rate (V_E) for individual *i* associated with activity *a* during time period *t*

$T = \sum t_{iaj}$

At first, NEM used aggregated data on where people, from a specific cohort, spent time during a typical day. Time-activity data was represented by hourly-block averages, and different daily sequences were developed for weekday and weekend. The one-hour time resolution was thought to be crude, leading to "lumpy" exposures. Therefore, the one-hour block was resolved to a new

averaging time of ten minutes, which was facilitated by time-activity data from a study by Pope (1986).

A significant improvement in the database of time-activity came as a result of three large surveys in Cincinnati, Washington D.C and Denver in the late 1980s. (Note, the same data from Denver became the basis for time-activity modelling in SHAPE (Ott *et al.*, 1988)). These surveys provided variable length data, with the shortest time period being one minute. A further eight databases have been added to NEM as a basis for generating activity profiles. At this time, a new stratified random sampling approach was also incorporated that uses an individual's sampled diary day to represent the activities of a person's age/occupational cohort group (McCurdy, 1995). This was the first version of *p*NEM.

Notwithstanding the wealth of time-activity databases and detailed information on EVR that underpin the NEM / *p*NEM models, there is still a fundamental shortcoming in that NEM / *p*NEM, like many other exposure models, uses fixed-site, outdoor monitors as the basis for all microenvironment concentrations. To overcome this weakness, some studies with NEM / *p*NEM have used adjusted monitored data, with the aid of data from personal monitoring, to reflect the disparity between microenvironment concentrations and concentrations at fixed-site monitors. This is a minor improvement to a significant problem. Despite this inherent weakness, it would be possible to incorporate a dispersion model into *p*NEM because of its modular structure.

b) SHAPE (Simulation of Human Activity and Pollutant Exposure)

The US EPA developed SHAPE to model the frequency distribution of population exposures to CO by computer simulation of microenvironment concentrations and human time-activity patterns. In essence, SHAPE simulates the activity patterns and exposures of a sample of urban dwellers during their daily routines. Exposure estimates are made at the individual level and are given as an integrated measure over a 24-hour period. SHAPE uses the equation developed by Duan (1982), shown above. As previously mentioned, the basic functionality of SHAPE is similar to the NEM series of models. In the first version of SHAPE (Ott, 1982), Monte

Carlo simulation techniques were used to generate estimates of exposure based on field studies of CO in various microenvironments (e.g. inside moving vehicles, houses, offices etc). Microenvironment concentrations were treated as a random variable, whose mean and variance for each microenvironment were either derived from the empirical field studies, or specified by the user. Information on time-activity came from the US Census and an independent study on national (US) personal transportation patterns. Activity patterns were then simulated using sampling from probability distributions of the chance of entry and time of entry into a particular microenvironment. Daily profiles of activity were generated for each individual.

SHAPE has a number of important assumptions that must be noted. A fundamental assumption about the microenvironment model in SHAPE is 'the superposition hypothesis' (Ott *et al.*, 1988). This is where the concentration in any microenvironment, over any given time period, is the sum of two independent components: 1) a concentration resulting from CO generated within the microenvironment, and 2) an ambient (background) component concentration. Ideally, the ambient background concentration would be based on the concentration in the immediate outdoor vicinity, adjusted by some lag in time to take account of air exchanges rates between outdoors to indoors. In most examples of using SHAPE (Ott, 1982, 1984, 1988) there has been no data available on exchange rates, so the lag has been set to zero. This is acceptable for averaging periods of, say, at least one day, but not realistic for concentrations over a period of one hour. The background contribution to the microenvironment concentration is also independent of location within the urban area. All background concentrations - both indoor and outdoor – are based on ambient monitoring from fixed-site stations within the urban area.

SHAPE has three ways of estimating the ambient concentration: 1) composite fixed-site stations – the average hourly reading from all fixed-site monitors, 2) nearest fixed-site station - the hourly reading from the fixed-site monitor nearest to a given location, and 3) no-source environments - the average of readings from personal monitoring in locations without sources. The last of these is used where no fixed-site monitoring is available. Some studies using SHAPE (e.g. Ott *et al.*,

1988) have improved the estimates from fixed-site monitoring by populating the study area with additional monitoring equipment. For example, in a study in Denver, Colorado, designed to validate SHAPE, six monitors were deployed in addition to the nine within the Denver network. This was necessary for validation purposes, but would not normally be possible because it would be costly and time-consuming to deploy a large number of monitors.

A modified version of SHAPE was implemented in the mid-1980s that accepted real time activity data as well as simulated activity data. The main purpose of the new version was to validate the SHAPE concept and exposure models (Ott *et al.*, 1988). As a basis for validation, micro-environmental data on concentrations of CO and time-activity were collected in Denver, Colorado. Personal exposure monitors were carried by individuals over two, consecutive days (Day 1 and Day 2), giving hourly average concentrations of CO. During the winter of 1982, after allowing for problems with monitoring equipment and unusable activity data, 722 valid 24-hour CO exposure profiles were collected from 336 participants. The diary data consisted of 34 activity codes and 33 locations, which were combined and dissolved into 22 microenvironments. Exposure estimates were made for each person's Day 2 profile by Monte Carlo sampling from the Day 1 microenvironment CO concentration distributions, and added to the estimated ambient concentration for the time and date. The predicted distributions of the 1 hour average exposure and the maximum moving 8 hour average were compared with the exposure frequency distributions actually observed on Day 2 in Denver. Overall, the modelled distribution of exposures agreed well with the observed frequency distribution.

There are, however, a number of inherent limitations in the modelled results that must be noted. Firstly, using a surrogate (fixed-site monitor) to represent background concentrations means that the model estimates may exceed measurements from personal monitors. This is most likely in situations where the fixed-site monitor is located at roadside locations in heavily-trafficked areas. Where negative exposures did occur, estimates were set to zero. Incidentally, the Denver study found that the three definitions of ambient concentrations gave broadly similar results in terms of the frequency distribution of exposure, with the

composite fixed site giving the best results. As expected, the 'no-source environment' data gave lower estimates of exposure.

Another limitation of SHAPE, as with NEM, is not recognising the intrinsic 'autocorrelation' of the concentrations observed in the microenvironment that a person visits. Each microenvironment concentration is a random hourly sample (with replacement) from a distribution of microenvironment exposures, but, in reality, the microenvironment concentration from one hour may contribute to the concentration in the following hour in the same microenvironment. Ott *et al.* (1988) use the example of cooking activity to demonstrate how there will potentially be a partial contribution from gas stove activity across periods of greater than one sampling period (i.e. 1 hour). In essence, the limitation of SHAPE, as with many other models, is in treating each microenvironment and time period as discrete entities. Clearly, concentration auto-correlation will also occur from day-to-day because individuals tend to travel by the same means and occupy the same space at home and work.

c) INTAIR

Most exposure models use empirical data on I/O ratios to predict indoor concentrations for a range of microenvironments. INTAIR (Dimitroulopoulou *et al.*, 2001) is a new modelling approach that attempts to incorporate the physical processes that determine both the contribution of outdoor air to indoor air and the exposure resulting from indoor sources, with the movement of typical individuals through a series of microenvironments.

INTAIR comprises two model components:

1) A physical model, used to calculate hourly indoor air pollution concentrations for different microenvironments, as a function of outdoor concentrations, building characteristics, and indoor sources
2) An exposure model used to calculate personal exposure to NO_2 for typical individuals through a series of microenvironments.

Both the physical model and exposure model were developed in EXCEL using Visual Basic. Each model component will now be described in more detail.

The physical model is a simple, dynamic-compartment model which calculates the concentration C_i in the indoor microenvironment *i* over time *t*, by considering the air exchange rate between indoors and outdoors (λ_r), the exchange rate between indoor microenvironments (λ_t), the volume (V_i) and surface area (A_i) of each microenvironment, building fabric (*f*), the deposition velocity of the pollutant (v_d), and the emission rate of the pollutant in each microenvironment *i*. C_O is the outdoor concentration at the urban background monitoring site. The model solves the following differential equation to consider each of the physical processes as C_i changes over time *t*:

$$dC_i/dt = [-v_d(A_i/V_i)C_i] + [\lambda_r f C_O - \lambda_r C_I] + [\lambda_t(C_j - C_i)] + [Q_i/V_I]$$

The first term of the equation is the pollutant mass deposited on the internal surfaces in the room. The second term determines the net pollutant mass entering the microenvironment compartment from outdoors, with the third term explaining the pollutant mass leaving the same microenvironment compartment. The fourth term explains the mass of pollutant moving between microenvironment compartments (e.g. kitchen to living room). The physical model considers several different types of microenvironment compartment: kitchen and living room at home, office, classroom and outdoors. In terms of time spent in different microenvironments, the other important compartment is bedroom. The bedroom is not included in the physical model, but is assumed to be 2 ppb lower than the concentration in the living room, based on UK field data. Two other microenvironments are considered outside the physical model. The concentration in shops is assumed to be the same as that found in a well-ventilated office. For transport, the concentration is assumed to be double that found at the urban background monitoring site. The transport factor is based on the ratio of roadside to urban background concentrations in London (QUARG, 1993).

At present, INTAIR simulates personal exposures to NO_2 for three population sub-groups: 1) a representative homemaker who moves mainly between the home and

outdoors, 2) a school child who moves mainly between home and school, and 3) an office worker who moves mainly between the home and office. The model simulates all individuals in a typical home at night and then the groups differ in their time activity during the day. The time-activity data profiles for each population group are based on a BBC survey of daily life from 1984 (BBC, 1989), that interviewed over 6000 people and recorded the percentage of people engaged in defined activities at given times of the day. Activity patterns and pollutant concentrations are entered into the model at fifteen-minute intervals, from hourly monitoring data. INTAIR can calculate exposures on an hourly basis, or provide an integrated exposure over a period of 24 hours. The integrated exposure uses the standard method by Duan (1982).

INTAIR has been tested in Leeds (UK) to simulate personal exposure to NO_2 based on data from a single, outdoor monitoring station in an urban background location. Dimitroulopoulou *et al.* (2001) showed that the results from INTAIR vary substantially from those that would be achieved using simple I/O ratios. Preliminary results suggest that INTAIR provides exposure estimates consistent with those from empirical I/O ratios reported in the UK, but the model has not yet undergone rigorous testing, or been validated against data from microenvironment monitoring. The value of INTAIR is that it allows the effects of different physical features of the indoor microenvironment and different time activity patterns to be quantitatively compared (Dimitroulopoulou *et al.*, 2001).

However, in its current form, INTAIR has a number of weaknesses, mainly due to its rather generalised inputs. Firstly, the compartment model has single, fixed parameters for compartment (room) size (e.g. volume and area), whereas compartment size can vary substantially between buildings. The building fabric, exchange and ventilation rates are also fixed, and the doors between rooms are assumed to be open at all times. The pollutant factors used in those microenvironments not included in the physical model are also crude. For example, the concentration for transport is simply the urban background concentration multiplied by a factor of two. This assumes that exposures during transport are always the same, for each individual, regardless of the type of transport and location within the urban environment. The time activity data is also

very weak. Each population sub-group is represented by indicative individuals, with very simplified activity patterns. Moreover, the activity data is based on a survey that is now over 15 years old, so it is not known to what extent this data reflects the activity of the population today. Finally, and perhaps most importantly, given the variation in outdoor concentrations of pollutants, INTAIR, like most exposure models, is limited in that it uses data from fixed site monitors as the basic input. Clearly, it is relatively easy to provide the model with rather more detailed data on the physical features of different microenvironments, and more detailed and updated time activity data. Of greater difficulty, however, is to improve on using data from sparsely populated, fixed-site monitors. Despite these shortcomings, INTAIR is currently the only model that truly attempts to simulate indoor exposures, considering both the physical processes and physical features in different microenvironments.

d) REHEX (REgional Human EXposure) model

The REHEX (REgional Human EXposure) model was designed with the primary goal of assessing the risks of living and working in specific areas of a city. The REHEX model calculates cumulative or integrated exposures using the same basic model by Duan (1982). REHEX is designed to sample from distributions of activity in order to simulate the range of variability in microenvironments, and works at a minimum resolution of 15 minute intervals. Each cumulative exposure is a 24-h exposure. REHEX is essentially a modified version of *p*NEM with some features of SHAPE.

REHEX has been used to calculate human exposure to benzene, O_3, PM_{10}, and NO_2 and related dose effects of these pollutants. A more recent study, in Southern California, has used the REHEX model to calculate the reduction in exposure to benzene between the years 1989 and 1997 (Fruin *et al.*, 2001). This study had three aims: 1) to study the difference in exposures between demographic groups, 2) to determine the contributions microenvironment sources made towards these exposures, and 3) to quantify the reduction of individual and microenvironment sources in relation to overall reductions in benzene. For this purpose, 11 demographic groups (age, gender, workforce, smoking status) were defined, and

time activity data were sought from a survey of 1100 24-h time-activity profiles, collected by Wiley *et al.* (1991) in 1987 and 1988. Each activity / location was described by one of 14 microenvironments. These 14 microenvironments were variants of home, work, transit, bars, any other, by presence of smoking, windows opened / closed. To simulate 1989 activity, the activity data were used unaltered. For 1997, the activity data were modified to reflect the known reductions in exposure to smoking at home and work.

The main limitation in this study, however, was that these measurements of benzene were 24h average concentrations, taken at 12 hourly intervals, whereas REHEX ideally runs at a maximum resolution of one hour. For indoor microenvironments with no sources, it was assumed that the concentrations matched that at the nearest fixed-site monitor. This was a reasonable assumption because of the chemical stability of benzene. All residential microenvironments were assumed to have the same concentration as the nearest fixed-site monitor. Seven benzene profiles were used to calculate the concentration of benzene in microenvironments that were not assumed to equal the fixed-site. These profiles were mainly based on empirical measurements of benzene (in-transit, presence of smoking in the workplace, active smoking) from other studies, at an hourly resolution. These hourly data were used in REHEX and divided into 15-minute intervals.

e) AirPEX (Air Pollution Exposure)

In The Netherlands, The National Institute of Public Health and Environment have developed an inhalation exposure model, based on the *p*NEM approach, which can estimate individual and population exposures to a range of pollutants (Freijer *et al.*, 1998). The basic inputs for AirPEX are air quality time-series data from fixed-site monitoring stations and time activity data. AirPEX employs a microenvironment approach, which, like many other models, simply divides space into a finite number of compartments and assumes linear relationships between monitoring stations and each microenvironment using I/O ratios. The I/O ratios are based on empirical data from other studies. The model relies on a single time-activity database from a large activity pattern survey of the Dutch population in

1994. The database contains daily activity patterns for 4985 individuals, who completed activity diaries over a single 24-hour period. For each individual, the location, time spent in each microenvironment, and the level of activity was recorded.

As the basic output, AirPEX expresses exposure in terms of a concentration of a pollutant over time. Population exposures are determined using the *p*NEM approach (i.e. sampling from groups of the population with similar demographic and activity characteristics). In addition to the basic exposure concentration, calculation of individual exposures also employs mathematical models that consider the ventilation rates of individuals. The level of activity and body mass are used to derive ventilation rates, as a function of time. The rate of intake is then calculated by multiplying the ventilation rate by the average concentration over a given time period. A number of data from clinical studies are used to correct the relatively higher ventilation rates of smokers and subjects with compressed airways. In addition to the basic dose model, effective air-tissue interface areas in the airways of the subjects are estimated from their age and health status. This additional breathing parameter is important in understanding the dose-response relationships of subjects in relation to specific health outcomes. Friejer *et al.* (1998), describe the use of the AirPEX in calculating the exposure of the Dutch population (over 1000 subjects) to ozone. There is currently no reporting on validation of the model.

The AirPEX model has been implemented in a Windows PC environment, and is fully transportable. The system includes a user interface that allows the user to model air pollution scenarios and estimate exposure distributions in different populations. AirPEX also has the possibility of analysing the socio-demographic characteristics of individuals with high exposures.

f) AirGIS

Jensen (1998) has coupled a GIS (ArcView) and dispersion model (OSPM) to model exposure. A prototype of this system, named AirGIS (Jensen *et al.*, 2001), has been developed to assist Danish local authorities in air quality management

and exposure assessment. AirGIS uses inputs on traffic emissions, street configuration, and meteorological conditions, together with information on background levels of pollution, to provide hourly estimates of NO_2, O_3, and CO. AirGIS uses a microenvironment approach to model exposure in a range of locations (e.g. workplace, home). The number of people present at residential locations is taken from the Central Person Database (CPD) and the number of people at workplaces is taken from the Central Business Database (CBD) by work address. From these data, it is only possible to estimate exposure for the number of people at any location (cohorts), rather than at the individual level. The system does, however, have the scope to model personal exposure at the individual level if time-activity data were collected. Exposure estimates are generated through using point-based techniques to intersect microenvironment locations with pollution surfaces generated in ArcView. Indoor concentrations are estimated from those in the immediate vicinity outdoors using empirical data on indoor/outdoor ratios. Uniquely, the system can also provide exposure estimates for individual line segments (streets), as a basis for assessing the exposure of street-workers and pedestrians.

g) AirQUIS

The Air Quality Information System (AirQUIS) (Clench-Aas *et al.*, 1999) is the air pollution component of the Norwegian Environmental Surveillance and Information System (ENSIS), developed by the Norwegian Institute for Air Research (NIAR) in 1994. AirQUIS was principally developed as an air quality assessment and air quality planning tool, but has been enhanced to include sophisticated exposure assessment methods, with components for outdoor and microenvironment modelling. AIRQUIS is a modular system that integrates an air pollution measurement system, emissions database, a dispersion model (EPISODE), a GIS, and an exposure model. Thus, AirQUIS is very similar to AirGIS in terms of both its components and structure.

The main sources of emissions (industry, traffic) are geographically referenced in a central inventory. The traffic database, for example, includes detailed information on the number of vehicles, traffic speed, road type, and road geometry. From the

traffic database, time varying emission factors can thus be derived as a basis for modelling the dispersion of pollutants from source. EPISODE was developed in-house for the purposes of modelling the dispersion of SO_2, NO_X, NO_2, and suspended particles. EPISODE is scale-free; it can model traffic within a street, traffic and gridded pollution from households and industry, or aggregated emissions on a regional scale. Information on air quality is collected from a network of fixed-site monitors by an automated logger, and fed to a central database. These monitors are used to validate output from EPISODE.

Exposures can be modelled for a population or at the individual level. Population exposures can be calculated in two ways: 1) the exposure of people living in each km^2 can be related to hourly concentrations of pollutants; 2) point estimates can be made simultaneously at a large number of receptors, and subsequently displayed in the GIS. At the individual level, a more dynamic approach is used: DINEX (Dynamic INdividual EXposure estimate). DINEX utilises time-activity diaries in which information about location, time and health are recorded. Microenvironmental exposures are determined by adjusting modelled outdoor concentrations using ratios of indoor-outdoor concentrations. Exposure estimates are made at the address level from EPISODE and related to activity records.

2.4.4 Summary

In this section, a number of exposure models have been described with contrasting functionality. In essence, there are currently two types of exposure model. There are those exposure models that model microenvironment exposures on the basis of concentrations at fixed site, outdoor monitors. Microenvironment exposures are then calculated using either empirical data on I/O ratios, or using a mass-balance approach. These type of models use time-activity data to model individual (deterministic) exposure, or population (stochastic) exposures from profiles of population cohort activity. In addition to providing exposure concentrations, several of these models have the ability to model dose. The second type of model use a combination of dispersion modelling and GIS to model concentrations at outdoor locations.

Clearly, a combination of features from both types of model would be a big improvement on current methods. As noted previously, the NEM and SHAPE types of model have the capability of being adapted to include GIS and dispersion modelling techniques. Equally, models such as AirGIS have been designed to accept time-activity data and could be adapted to model microenvironment exposures. Models of exposure in indoor microenvironments are well developed, but there remains little scope for modelling journey exposures. GIS, however, offer the potential for modelling journey exposures using network analysis techniques. Ideally, therefore, an exposure modelling approach would exist that incorporated models of time-activity, dispersion modelling, and GIS techniques, with the capability to model exposures across the full range of microenvironments.

2.5 METHODOLOGY

The design and structure of the methodology in this research is shown in *Figure 2.2.* The overall approach is called STEMS (Space-Time Exposure Modelling Systems) (Gulliver and Briggs, 2001). As this indicates, STEMS involves linking four models: a) a traffic assignment model (SATURN); b) a dispersion model (ADMS); c) a model of background pollution; and d) a purposely designed exposure model (TOTEM). These models are loose-coupled, by the definition of Goodchild *et al.* (1991), within a GIS (ArcInfo/ArcView).

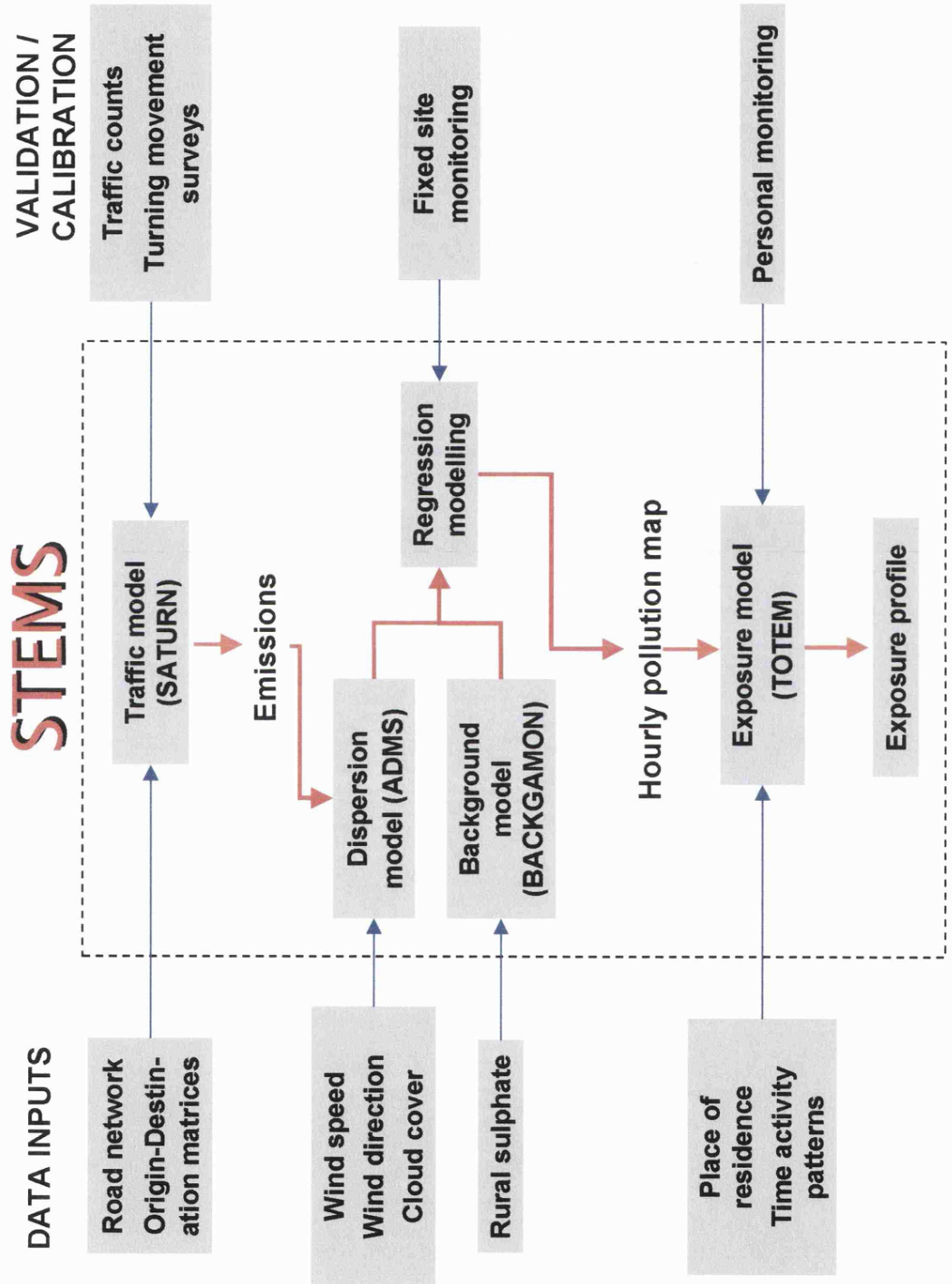


Figure 2.2: The STEMS (Space-Time Exposure Modelling System) approach

2.6 STUDY AREA

This research grew out of work undertaken in Northampton as part of an EPSRC project on 'Effects of traffic management on patterns of air pollution and human exposure' (Briggs *et al.*, 1998). Northampton is a large provincial town located in the East Midlands, about 90km north-north-east of London and 60km east-south-east of Birmingham. Northampton lies on gently undulating slopes in the Nene valley. The river Nene intersects Northampton to the south of the town centre, running west to east. The total population is about 180,000 and the main industries are retailing, finance and logistics. The main residential areas lie to the east and north of the town centre, interspersed with small areas of parkland and light industry, with smaller residential developments to the south and west, close to the M1.

The study area (*Figure 2.3*) is based on Northamptonshire County Council's 'Northern Package' (Transport Policies and Programme, 1997-98), which defines a number of strategic and policy measures for traffic and public transport management. Located to the north of the town centre it covers about 8km^2, stretching along one of the main access routes (A508/A5199) to Northampton, passing through densely inhabited residential areas, including a number of schools, University College Northampton, a shopping area, and light industry. The eastern areas of Northampton are reached from the north and west by a circular route running through the study area from the M1. The study area is bounded to the north by open countryside, to the west by a tributary of the River Nene and a branch line of the 'West Coast Mainline' railway, and to the east by the University College and Moulton Park industrial estate.

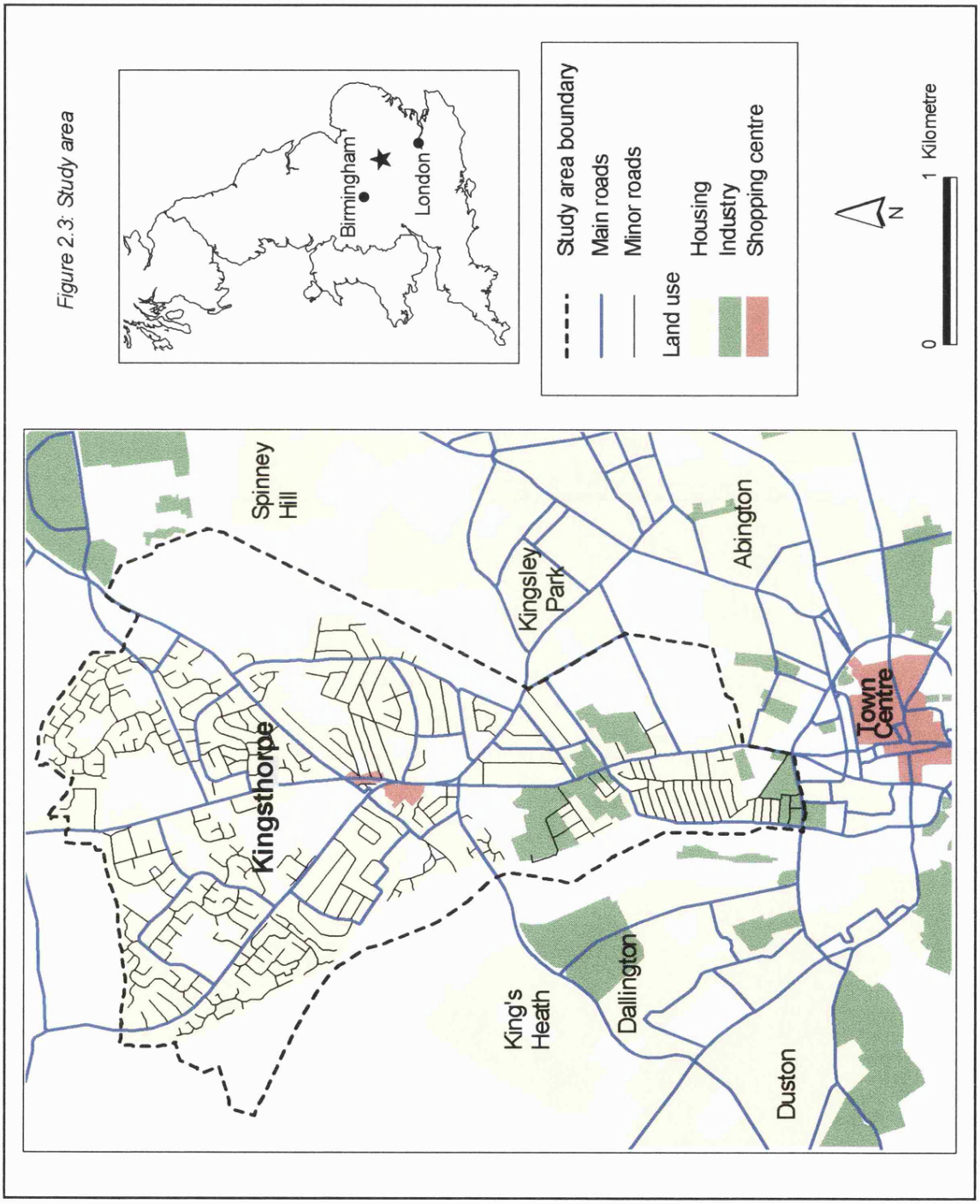


Figure 2.3: Study area

2.7 SUMMARY

This chapter has reviewed the literature on the health effects of air pollution and methods and models for assessing exposure. The development of STEMS is described in Chapters 3 and 4. The focus of Chapter 3 is the development of the outdoor pollution model. Chapter 4 deals with the development of the exposure modelling.

3 DEVELOPMENT OF OUTDOOR PARTICULATE POLLUTION MODELLING

3.1 INTRODUCTION

This chapter begins by examining the sources and characteristics of PM_{10} (*Section 3.2*). It then looks at existing methods of quantifying and modelling PM_{10} concentrations (*Section 3.3*), before describing the methods used to develop and implement an outdoor pollution model for PM_{10} (*Section 3.4 to 3.6*).

3.2 SOURCE PROFILE AND CHARACTERISTICS OF PM_{10} IN THE UK

3.2.1 Components of PM_{10}

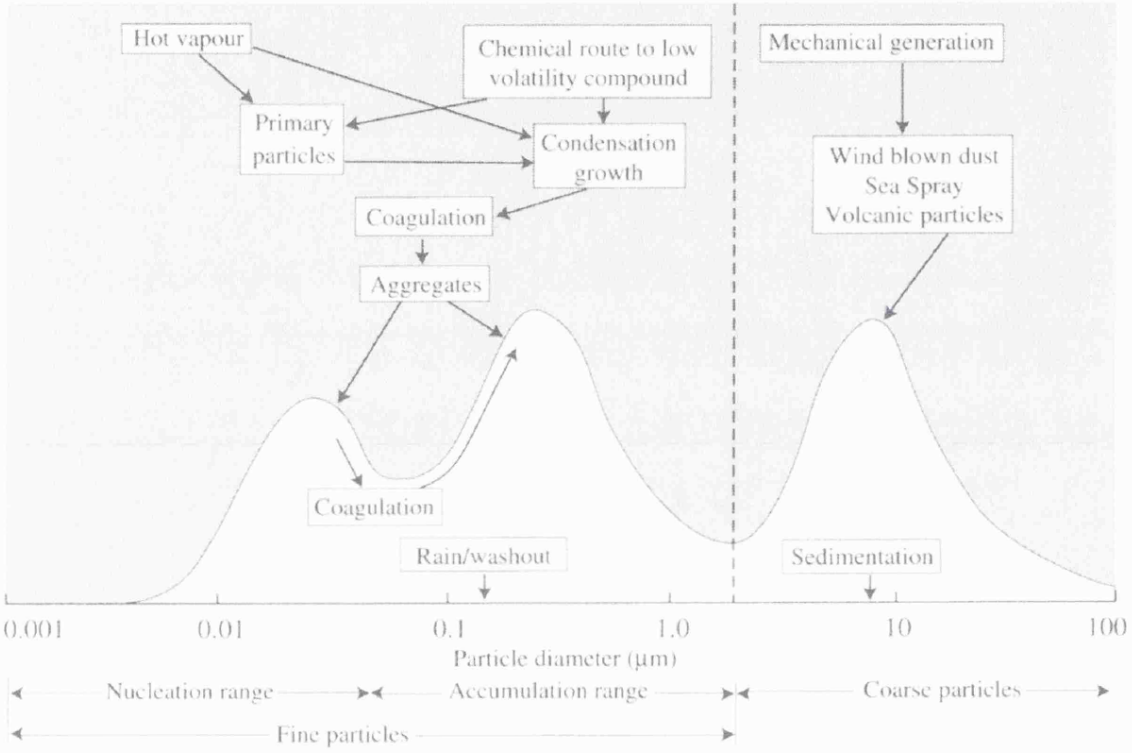
As detailed in *Section 2.1*, particulates of interest in terms of human health are generally those with an aerodynamic diameter of less than 10 μm. PM_{10} is broadly made up of three components: primary, secondary, and other particles. Primary particles are those directly emitted into the atmosphere from sources such as road transport and industry. Secondary particles are from chemical reactions and condensation of gases in the atmosphere, whilst particles such as wind blown dust, soil, sea spray (marine particles), and non-exhaust traffic sources make up 'other' particles. Other particles are also primary, but are specifically those coming indirectly from traffic exhaust, industrial and geological sources. In other words, they are primary particles not related to combustion processes.

3.2.2 Particle size distribution and chemical composition

In general, it is thought that traffic-related primary particles are predominantly found in the fine fraction of $PM_{2.5}$ (i.e. with a median aerodynamic diameter less than 2.5 μm). Indeed, a recent study by QUARG (1996) in Birmingham, UK, showed that during winter periods road traffic is, on average, only responsible for 3% of the coarse particle mass (PM_{10}-$PM_{2.5}$). The same study also deduced that about 80% of sulphates (SO_4) and nitrates (NH_4) are attributed to the $PM_{2.5}$ fraction. Thus, the coarse fraction is almost entirely made up of particles from non-

traffic sources such as soil and wind-blown dust. These findings agree with those of Whitby (1978) who showed that the coarse fraction is almost entirely found within the range PM_{10}-$PM_{2.5}$ and primary and secondary particles are generally in the fine fraction (i.e. <2.5 µm). *Figure 3.1* shows the typical size distribution of particles and pathways of particle formation (COMEAP, 1995).

Figure 3.1 Typical size distribution of particles (COMEAP, 1995)



A number of key chemical and atmospheric processes contribute to the formation of particles, across a wide range of particle sizes. For fine particles (i.e. <2.5 µm), there are two main categories of process leading to particle formation: nucleation and accumulation. In broad terms, nucleation mode is where primary particles - mainly in the form of elemental carbon – are emitted from combustion sources. Nucleation particles can also be formed during, or directly following, combustion as hot vapour cools and condenses to form a condensation nucleus. Once a particle is formed it can increase in size through Brownian motion by merging or colliding with other particles, or by direct contact with other condensation nuclei. Brownian motion is where particles are constantly moved by the bombardment of molecules and other particles (Hinds, 1982).

When a particle becomes airborne (i.e. emitted into the atmosphere) it enters the accumulation (or aggregation) phase, where particles change size, shape, and composition by a range of chemical and mechanical processes. Particles formed at this stage are generally known as secondary matter. The principal mechanism for creation of secondary matter is by particles acting as the target area for particles formed by chemical oxidisation of atmospheric gases. There are two main products of this process: sulphates and nitrates.

Sulphates are formed from sulphuric acid vapour, which itself comes from the oxidisation of sulphur dioxide. Similarly, oxidisation of NO_2 forms nitric acid, which, in turn, forms nitrates by further oxidisation. Moreover, where nitric and sulphuric acid react with ammonia (mainly from agriculture), ammonium nitrate and ammonium sulphate are formed. Sulphates and nitrates can load on existing primary particles, or directly create new, secondary sulphate and nitrate particles (QUARG, 1996). Nitrates and sulphates are the main elements of secondary particles in the UK, with a small proportion comprising secondary organic matter. Secondary organic compounds are carbons and hydrocarbons that are emitted as primary particles but form in the atmosphere. They are particularly prevalent in summer months when photochemical oxidisation of volatile organic compounds leads to the formation of less volatile compounds which associate with airborne particles (DETR, 2000b). Small amounts of chloride, sodium, and magnesium are also found in the secondary component. These matter are mainly from sea-spray carried inland by the wind. However, only a very small amount of this matter is found in the fine fraction, with the majority in the coarse particle range.

3.2.3 The life-time of particles

Most particles have a relatively short lifetime in the atmosphere. Very small particles (i.e. <0.1 μm) will be either directly deposited on the surface of the earth or intercepted by obstacles such as vegetation or buildings, or be quickly lost - probably within a few minutes - to the accumulation phase. Particles formed in the accumulation phase may be suspended in the atmosphere for several days before being washed out by rainfall (i.e. wet deposition). Nonetheless, it is possible during this period for particles to be moved over hundreds of kilometres, depending on

the height of the boundary layer (i.e. mixing height) and wind speed. These particles are generally those in the range $PM_{0.1}$-$PM_{2.5}$; thus they represent the secondary component. Larger, coarse particles (i.e. PM_{10}-$PM_{2.5}$) will be airborne for, say, one day before they reach the surface by dry deposition.

3.3 SOURCE APPORTIONMENT AND QUANTIFICATION OF PM_{10}

3.3.1 Introduction

In terms of health effects, and policy, there is an important requirement to quantify the relative contribution from primary and secondary sources in the different urban areas and with time of year (QUARG, 1996). Despite concerns over adverse health effects, knowledge of sources of airborne particulate matter in the United Kingdom is incomplete (Harrison *et al.*, 1997). To date, there remain very few attempts to quantify the sources of PM_{10} in the UK. A detailed study in Birmingham, UK, gave new insights into the sources and quantity of the components of PM_{10} (Harrison *et al.*, 1997; part published by QUARG, 1996). Two recent studies have developed regression techniques to quantify, separately, the components of PM_{10} (Stedman *et al.*, 1998; Turnbull and Harrison, 1999). These approaches are generally known as 'receptor modelling' techniques (QUARG, 1996). This next section discusses each of these approaches in detail.

3.3.2 The Birmingham study

The Birmingham study involved analysing the relative contribution of traffic-related particles to monitored PM_{10} and $PM_{2.5}$ by looking at the relationship of each fraction with roadside measurements of NO_X, on an hourly basis. Both fractions of PM_{10} correlated very strongly with NO_X – which can be taken at heavily-trafficked roadside locations to be wholly traffic-related (QUARG, 1996). Monitoring took place at Birmingham Hodge Hill (close to a busy section on the M6) for the period October 1994 to March 1995 [n = 4000 hours]. During this period, an average concentration of 60.9ppb was recorded for NO_X, and 16.51 $\mu g/m^3$ and 11.04 $\mu g/m^3$

for PM_{10} and $PM_{2.5}$, respectively. Using these data, regression techniques were applied to predict separately hourly PM_{10} and $PM_{2.5}$ from NO_X:

$PM_{2.5} = 0.114\ NO_X\ (ppb) + 6.54$ $[r^2=0.49]$

$PM_{10} = 0.134\ NO_X\ (ppb) + 11.31$ $[r^2=0.49]$

In each case, the constant can be taken to include both secondary and other particles. Based on these equations, there is an average background of 11.31 $\mu g/m^3$ of PM_{10} and an average traffic-related component of 5.2 $\mu g/m^3$ of PM_{10} (=16.51-11.31) at this site. In the same manner for $PM_{2.5}$, there is an average traffic-related component of 4.5 $\mu g/m^3$ (=11.04 – 6.54) and a background of 6.54 $\mu g/m^3$.

Interpretation of the above regression equations means that 32% of PM_{10} is attributable to local, traffic-related primary particles. The other major component of PM_{10} in the UK is non-traffic, secondary particles. According to Harrison *et al.* (1997), the $PM_{2.5}$ fraction consists only traffic-related primary particles and non-traffic, secondary particles. Harrison *et al.* (1997) states that 80% of non-traffic, secondary particles lie in the $PM_{2.5}$ fraction. In the Birmingham study, this means that 8.2 $\mu g/m^3$ of PM_{10} is non-traffic secondary particles (0.8 * 8.2 = 6.54), which is approximately 50% of total PM_{10}. This figure is consistent with that obtained from a separate study of chemical components of PM_{10} by Clarke (1984). On average, sulphates make up just over half of the non-traffic, secondary component, with nitrates making up the majority of the remainder. The other main component of PM_{10} is traffic-related secondary particles. This is mainly in the form of nitrates and represents about 2 $\mu g/m^3$ of typical urban PM_{10}. In the Birmingham study this equates to about 12% of total PM_{10}. Thus, for heavily-trafficked urban locations, about 44% (=32% + 12%) of PM_{10} is derived from road transport. This leaves, by default, 1.1 $\mu g/m^3$ as 'other' particles, which are either from non-exhaust, traffic sources (e.g. tyre and break wear), or are wind-blown dust.

Table 3.1 shows a summary of the source apportionment of average PM_{10} from the Birmingham study. These figures represent the average situation based on a six-month winter period.

Table 3.1: Source apportionment of PM_{10} from a study in Birmingham, UK (Harrison et al., 1997)

PM_{10} (µg/m3)	source	component	local / remote	% of total
5.2	traffic	primary	local	32
2.0	traffic	secondary	remote	12
8.2	non-traffic	secondary	remote	50
1.1	traffic, non-exhaust	other	local	6
	non-traffic	other	local + remote	
16.5				100

(compiled from Birmingham study by Harrison et al., 1997)

Clearly, there will be hourly variation due to changes in traffic volume, both locally and at the regional scale, affecting the local-primary, traffic-related secondary and other components. The non-traffic, secondary contribution will be dependent on air mass trajectory and meteorological conditions (Turnbull and Harrison, 2000), with daily or weekly variation. There will also be seasonal variation in the proportional contribution of each component. Indeed, analysis of summer data in the Birmingham study, using the same approach, showed a higher magnitude of the coarse component in summer months, which is attributable to increased dust due to drier conditions.

The results are thought to be representative of busy, roadside locations within urban areas. Birmingham is a large city in the centre of the West Midlands conurbation with a relatively high proportion of its PM_{10} coming from local, traffic-related sources. Smaller urban areas, however, may not exhibit the same source profile. This can be tested in Northampton using three months of 'winter' data (Sept-Dec 1999) where PM_{10} was co-monitored with NO_X at a relatively busy roadside location. Using the same approach as the Birmingham study gives:

$$PM_{10} = 0.126\ NO_X\ (ppb) + 10.40 \qquad [r^2=0.53]$$

$$N=1727$$

The average monitored NO_X for this period was 62.2ppb and the average PM_{10} was 18.21 ug/m3. Incidentally, both NO_X and PM_{10} are higher in Northampton than at Birmingham Hodge Hill, but cannot be directly compared because they are not from the same year.

Only PM_{10} was monitored in Northampton so it was not possible to replicate the same analysis in Northampton as in Birmingham. Nevertheless, some interesting similarities are exhibited between the two areas. Both sets of results show similar magnitude in the background (i.e. secondary + other) component, and a similar coefficient for the primary component. In using the same interpretation as the Birmingham study, 43% of the total PM_{10} mass is attributable to traffic-related sources, compared with 44% in Birmingham. In Northampton this approach explains 53% of the variation, compared with 49% in Birmingham.

3.3.3 The Stedman method

The first of two receptor modelling studies was developed by Stedman (1999) to estimate PM_{10} at background sites. Stedman experimented separately with black smoke (BS) and NO_X as a marker for the primary component, whilst using sulphate as the secondary component, and other particles were the constant. The general form of the Stedman approach is as follows:

PM_{10} = A(primary) + B(secondary) + C

Thus, this becomes:

$PM_{10} = A(NO_X) + B(SO_4) + C$ or;
$PM_{10} = A(BS) + B(SO_4) + C$

In each case, NO_X and black smoke monitors are co-located with measured PM_{10}, and an implicit assumption in the model is of common sources of PM_{10}, black smoke and NO_X (Stedman *et al.*, 1998). In this way, primary and secondary particles are independent variables in the regression and 'other' particles (C) are simply the difference between measured PM_{10} and the total of primary and secondary component. As described in *Section 3.2.2*, the majority of secondary particles are either sulphates or nitrates, with a smaller, yet significant, coastal contribution in the form of sodium chloride. Nitrates and sodium chloride are, however, only measured at two sites within the UK, whereas sulphate is measured

at eight sites. Stedman recognised that sulphates may only represent about 50% of the secondary component, but, due to the lack of data on the other main components, sulphate is assumed to be a good marker for total secondary particulates (Stedman *et al.*, 1998). Moreover, the volatility of ammonium nitrate and secondary organic particles means that these species are unlikely to be detected by the TEOM PM_{10} and $PM_{2.5}$ monitors that are widely used in the UK. Both ammonium sulphate and sodium nitrate should, however, be quantitatively detected (APEG, 1999).

Stedman applied this approach to model daily average PM_{10} for urban background sites from the NETCEN network for 1996. Eight sites were chosen where NO_X (ppb) and black smoke monitors were co-located with PM_{10}. A long-term interpolated map of SO_4 ($\mu g/m^3$), developed by QUARG (1996), was used to derive sulphate concentrations for each site (*Figure 3.2*). *Table 3.2* and *Table 3.3* show the performance of the different equations across these sites for black smoke and NO_X, respectively.

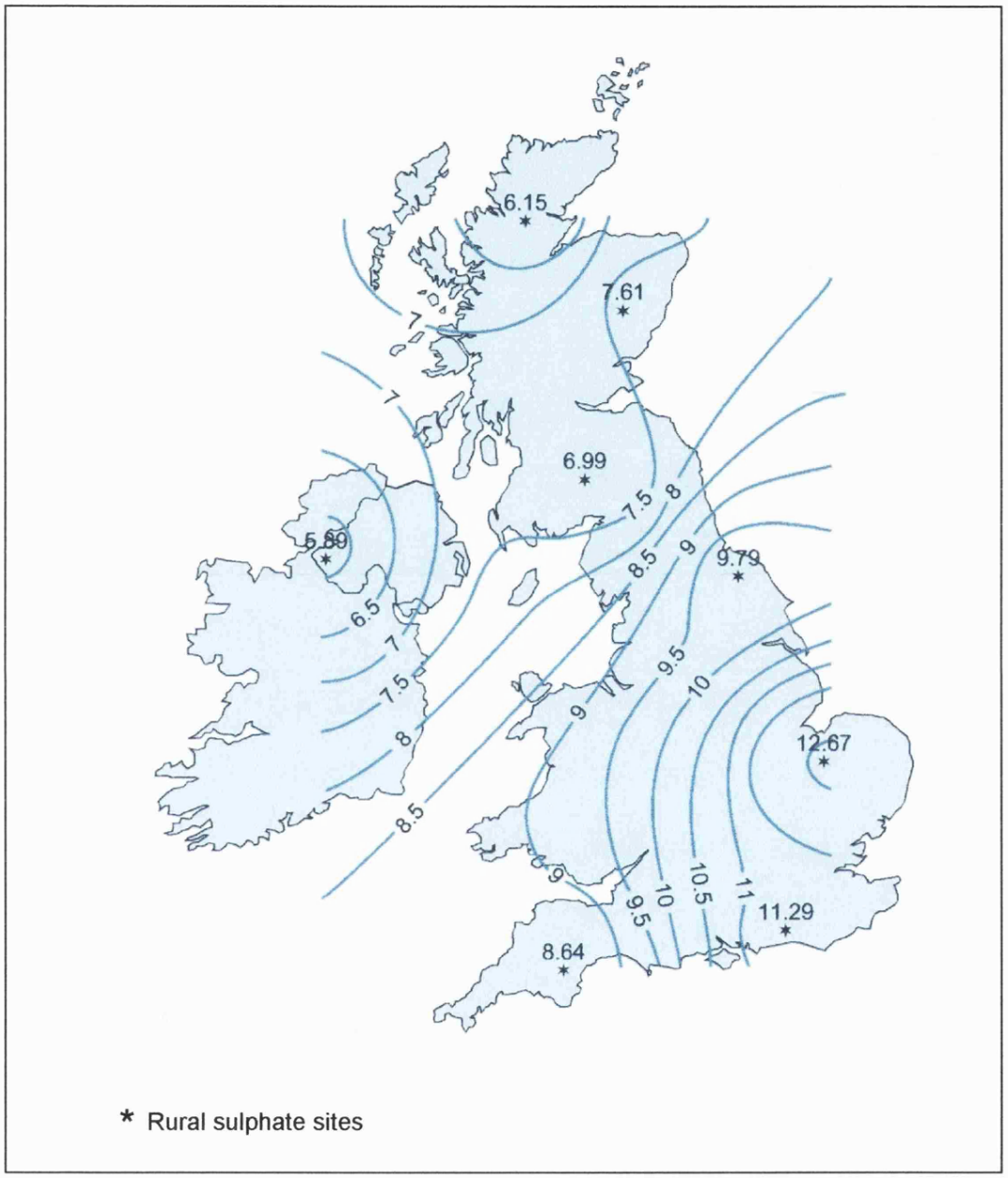
Table 3.2: Regression coefficients for Black Smoke (BS) and Sulphate (SO_4) receptor modelling of 1996 daily PM_{10}

	BS	SO_4	Intercept, C	r^2
London Bloomsbury	0.64	2.26	10.96	0.78
Birmingham Centre	0.59	2.41	8.30	0.71
Bristol Centre	1.03	2.35	10.83	0.70
Manchester Picadilly	0.60	2.46	9.77	0.74
Newcastle Centre	0.66	3.13	7.73	0.84
Belfast Centre	0.71	2.30	9.21	0.79
Edinburgh Centre	0.59	2.46	9.85	0.61
Liverpool Centre	0.92	2.46	9.79	0.76

(source: Stedman et al., 1998)

Both models clearly perform well across all sites. One noticeable difference, however, is that the constant for BS is higher in all cases. This reflects the circumstance that, even at background sites, BS only represents particulate matter, whereas NO_X cannot be entirely attributed to road sources at background locations. In this situation, NO_X also explains a proportion of 'other' particles. Stedman *et al.* (1998) applied this approach to a further 24 NETCEN sites, for

Figure 3.2: Interpolated long-term average concentration of secondary particulate matter across the UK (μg/m^3).



Source: QUARG, 1996

Table 3.3: Regression coefficients for NO_X and Sulphate (SO_4) receptor modelling of 1996 daily PM_{10}

	NO_X	SO_4	Intercept, C	r^2
London Bloomsbury	0.11	2.55	9.29	0.74
Birmingham Centre	0.21	2.29	6.52	0.72
Bristol Centre	0.15	2.62	7.97	0.76
Manchester Picadilly	0.16	2.95	7.69	0.74
Newcastle Centre	0.16	3.48	5.66	0.83
Belfast Centre	0.30	2.65	5.91	0.85
Edinburgh Centre	0.12	2.99	6.42	0.67
Liverpool Centre	0.19	2.56	7.57	0.76

(source: Stedman, 1998)

1997. In this case, only NO_X was used because BS was only available at the eight sites used in the original experiment. Results were broadly comparable to the initial study in terms of the magnitude of the coefficients. Values for r^2 were in the range 0.27 to 0.78, with 80% of sites returning a value greater than or equal to 0.6. The weakest performances were for those sites either in the west of the UK, or coastal sites (e.g. Derry, Narberth, Cardiff, and Hull). This can be explained by the dominance of background and secondary matter in westerly locations, coupled with a relatively small contribution from local traffic sources. The composition of PM_{10} at coastal sources has, however, been shown to be quite different from those inland (Turnbull and Harrison, 2000). Nonetheless, the model was considered broadly acceptable.

The receptor modelling approach for urban background sites has also been extended to model at roadside locations (Stedman *et al.*, 1998; Stedman *et al.*, 2001). This approach requires at least one background site for each roadside location. The roadside 'enhancement' contribution is simply calculated as the difference between the monitored roadside concentration and background concentration of NO_X:

Roadside enhancement = roadside – background

The regression analysis thus becomes:

$PM_{10} = A(Background\ NO_X) + B(SO_4) + R(Roadside\ enhancement\ NO_X) + C$

Stedman *et al.* (1998) applied this approach to nine NETCEN sites using 1997 PM_{10} data. The results of this analysis are shown in *Table 3.4*. Clearly, this approach gives a strong fit with monitored daily PM_{10} at all sites. By comparing the results of this analysis with *Table 3.3*, the values for other particles (C) are seen to be broadly consistent with those using only background NO_X, and the magnitude of the intercept and sulphates is similar for sites in roughly the same location (e.g. London).

Table3.4: Regression coefficients for receptor modelling at roadside sites using NO_X and SO_4 to predict daily PM_{10} (1997)

	Roadside NO_X	Background NO_X	SO_4	Intercept	r^2
London A3	0.042	0.177	2.74	6.43	0.74
Bury	0.066	0.138	2.64	6.90	0.71
Camden	0.069	0.120	2.33	11.20	0.77
Haringey	0.078	0.125	2.53	7.90	0.80
Glasgow	0.123	0.069	3.11	6.92	0.79
London Hillingdon	0.075	0.135	2.42	5.60	0.78
Marylebone Road	0.090	0.122	3.26	6.12	0.79
Sutton	0.096	0.110	2.40	7.04	0.74

(source: Stedman et al., 1999)

In a follow-up study, Stedman *et al.*, (2001) applied this receptor modelling technique to London Bloomsbury, this time only using NO_X as the primary component, because only NO_X was co-located with the PM_{10} monitor at this site. As this study involved just one site, this looked at the nearest sulphate sites to London to represent the secondary component, instead of using the long-term interpolated map (QUARG, 1996). Of the eight sulphate sites in the UK, two lie within the south east of England: Stoke Ferry (in Norfolk) and Barcombe Mills (in East Sussex). Analysis of daily sulphate (N.B. the highest temporal resolution of sulphate is a daily average) showed a strong correlation between these sites for the period April 1995 to March 1996 (APEG, 1999).

Therefore, a daily average of Stoke Ferry and Barcombe Mills was taken to represent regional sulphate in the south east of England (Stedman *et al.*, 2001). Stedman applied his regression technique (Stedman *et al.*, 1998), with this new secondary element, to predict daily average PM_{10} for London Bloomsbury, for the period 1996-1998. Results were comparable (r^2 0.53~0.74) to those cited in previous applications of this model (Stedman, 1997; Stedman *et al.*, 1998).

3.3.4 The Turnbull and Harrison method

Turnbull and Harrison *et al.* (2000) used a similar approach to that of Stedman, but with three main improvements. Firstly, all parameters were measured directly at each site and co-located with PM_{10}, rather than either using interpolation from the long-term map of SO_4 or the nearest monitoring site(s). Secondly, the temporal resolution was increased from 24 hours to 6 hours, allowing for some assessment of diurnal variation. Finally, in addition to sulphate, nitrates and sodium chloride (NaCl) were measured, representing the other main components of secondary matter. Only four sites were used, but these covered a mixture of rural, urban, and coastal locations, whereas Stedman only used roadside or background urban sites. In this approach, only black smoke was used to represent the primary component, sulphate and nitrate were combined as one unit in the secondary, and sodium chloride was used to represent marine sources within the secondary component. Like Stedman, Turnbull and Harrison assumed that other particles were represented by the constant.

Multiple regression techniques were used to predict PM_{10} at each of the four sites during 1995 and 1996 [n=672 hours]. The form of this equation is:

$$PM_{10} = A(BS) + B(NH_4 + SO_4) + C(NaCl) + \text{constant}$$

This approach yielded r^2 values in the range 0.55 to 0.94, depending on season, and r^2 of 0.76 from regressing data from all years. These results are mostly consistent with those of Stedman *et al.* (1998, 2001), but also give insight into the variation between sites with contrasting characteristics, and at higher temporal resolution. The Turnbull and Harrison approach showed that the secondary

component was nearly always uniform between sites and, on average, contributed about 32% to the total mass of PM_{10}. Only nitrates showed a diurnal variation, and this was a small day-to-night variation. As expected, marked differences were seen in the primary component between urban and non-urban sites. However, this effect was reduced when expressed as a proportion of the total, with the primary contributing in the range 10 to 40 percent of total PM_{10}. Chloride was uniform at inland sites, contributing about 12% to total PM_{10}, but showed a marked increase in contribution (34%) at the coastal site. These findings support results obtained by Harrison *et al.* (1997) in the study of source apportionment in Birmingham, cited in *Section 3.3.2*. They also further demonstrate the dominance of secondary and other particles, even at busy roadside locations.

3.3.5 Summary

The Stedman approach has shown that NO_X (or black smoke) and sulphate are good markers for the primary and secondary components of PM_{10}, respectively. Furthermore, regression techniques are an effective way of using these variables to predict total PM_{10} at locations where a marker for the primary component is monitored. Harrison has validated and extended the findings of Stedman *et al.*, (1998), and provided more detail in the site-to-site, seasonal, and diurnal variation for a wider range of sources of particulate matter. By using linear regression techniques, both approaches have managed to explain most of the variation in PM_{10} at a number of contrasting sites within the UK. These studies not only stand as important new information for understanding health effects of airborne particulate matter, but also provide a useful, theoretical basis for modelling source contribution to PM_{10} at locations where monitored data is not available.

3.4 DEVELOPMENT OF A BACKGROUND PM_{10} MODEL (BACKGAMON)

3.4.1 Monitoring equipment

An important tool in developing and testing models was state-of-the-art monitoring equipment. As part of NAPS, an automatic pollution-monitoring unit was made available to this research, provided by HORIBA instruments (*Figure 3.3*).

Figure 3.3: Fixed-site air pollution monitoring unit at the roadside location in Kingsthorpe, Northampton.

This unit monitors PM_{10}, NO_X, NO_2, CO, O_3 and SO_2. It is suitable for deployment at fixed-site locations and can easily be moved from one site to another. The unit offers equipment comparable to that used in the UK automatic monitoring network (i.e. NETCEN). For example, PM_{10} is measured using the TEOM system, providing measurements as PM_{10} mass ($\mu g/m^3$). In addition, the unit has a set of meteorological instruments to electronically measure wind speed, wind direction and temperature. All of this data can be downloaded using purpose-built software - supplied by HORIBA instruments - over a modem. In this case, the software was programmed to automatically retrieve the data every six hours. This unit was first

deployed at a background site, in the grounds of UCN (Sept 1997 to May 1998), and then at a busy, roadside location in the centre of Kingsthorpe (July-Dec 1997 and Feb-Apr 2000). This unit was the main reference point used in this study. Other, portable monitors were used to develop exposure models and therefore are discussed in the following chapter.

3.4.2 Modelling background PM_{10}

The initial, and most logical approach for modelling PM_{10} comprised: modelled PM_{10} from a dispersion model to represent local sources, and a background site, devoid of any direct influence from local sources, to represent the regional / far-travelled secondary and other components. Sources of background PM_{10} were sought on this basis. The general model for this approach is:

PM_{10} = local modelled + monitored background

During the course of this study, a PM_{10} measurement site (OSIRIS) was established at Guilsborough, 15km to the north of Northampton, which is managed by Daventry District Council. Initially, this site appeared to be a suitable source of the background PM_{10}; however, a number of problems were encountered. The main restriction with this site was that only 33 days [n=792 hours] of data were available that coincided with the monitoring at Kingsthorpe [n=3329 hours] - about 15% of the total. Furthermore, for those periods where data was available from both sites, simultaneously, marginally higher concentrations at Guilsborough were seen for hours with winds from a general Northampton direction. This could not be attributed to particular hours of the day, so it was deemed not possible to remove this effect prior to modelling. Due to this problem, and the limited amount of simultaneous data, this approach was therefore abandoned.

This led to consideration of a more comprehensive data set from a rural site further afield. There are, however, only four rural NETCEN sites measuring PM_{10} in the UK: Harwell (Oxfordshire), Rochester (Kent), Narberth (Pembrokeshire), and Lough Navar (Northern Ireland). Harwell is the nearest of these to Northampton, at a distance of about 70km. The nearest town to Harwell is the small town of

Didcot - about 5 km to the east. It would be expected that only a small contribution to the particle concentration recorded at Harwell would come from traffic-related sources in Didcot, although Didcot power station is a potentially significant source of particles.

Unfortunately, however, data was only available at Harwell until the end of August 1999, whereas monitoring at Kingsthorpe commenced in July 1999. This meant that there were only 55 days [n=1261] when data from Harwell coincided with the monitoring in Northampton. It is understood that further data from Harwell exist but it was not possible to obtain these in the course of this study. Although a greater coverage of data was available from Rochester, this site was not considered a suitable alternative due to high levels of PM_{10} relative to the Northampton site. This is probably explained by its close, windward location relative to Greater London. The remaining sites are not only at great distance (>250km) from Northampton, but their westerly position also means that they generally experience lower concentrations of secondary particulate matter than central and eastern UK (QUARG, 1996). In view of the lack of suitable data on background PM_{10}, an alternative approach was sought.

3.4.3 Rationale for the approach adopted in this study

Receptor modelling for PM_{10} is a potentially useful technique when a proxy can be found to represent each component, but is not possible if there is no locally monitored data. As shown in *Section 3.3.3*, there is clearly a strong body of evidence to suggest that the secondary component is fairly uniform over distances of tens and even hundreds of kilometres, at least on a day-to-day basis. It is, therefore, reasonable to assume that mean daily sulphate can be allocated to each hour within any day to represent secondary PM_{10}, at a particular location. Turnbull and Harrison (2000) also showed that 'other' particles, such as wind blown dust and soil, are fairly uniform between site and time-of day. Moreover, in urban areas, other particles have been shown to correlate strongly with traffic volume (Harrison *et al.*, 1997), which reflects a contribution to this component from non-combustion sources, such as brake and tyre wear. Therefore, in a regression model, this portion of other particles can be represented by a modelled primary

component, as the primary should account for variation in all traffic-related particles, including those from non-combustion sources. Secondary, and non-traffic, other components are assumed to be uniform at the 'region' level (Tarrason and Tsyro, 1998); therefore, the nearest regional site for sulphate ought to be a good marker for secondary particles. On the basis of this argument; a general form for this approach is proposed:

$PM_{10} = A(\text{modelled primary}) + B(SO_4) + \text{other}$

3.4.4 Analysis of rural sulphate data

As stated in *Section 3.3.3*, rural sulphate is only available at eight sites in the UK. Only three of these of these sites are within the south of England. The nearest site to Northampton is Stoke Ferry, in Norfolk, at a distance of about 80km. Other sites are at Barcombe Mills in Sussex and Yarner Wood in Devon. Sulphate measurements were obtained from NETCEN for each of these sites for the period 1995 to 2000, inclusive, for the purpose of assessing the uniformity between sites over space and time. A statistical summary of measurements at these sites was compiled (*Table 3.5*).

Table 3.5: Summary of monitored SO_4 (ug/m^3) at three sites in southern England

		1995	1996	1997	1998	1999	2000
Stoke	Mean	1.19	1.47	1.08	.89	.82	.83
Ferry	Stdev	.84	1.45	.86	.61	.55	.67
Barcombe	Mean	1.43	1.63	1.23	.97	.87	.78
Mills	Stdev	.81	1.41	.94	.59	.55	.59
Yarner	Mean	1.47	.83	.81	.64	.55	.50
Wood	Stdev	1.23	.74	.73	.60	.50	.48

With the exception of 1996, mean sulphate levels have continuously declined over this period at all sites, possibly due to improved emissions controls and lower sulphur fuels. It is well documented that the meteorological conditions were

particularly unusual in 1996, with a predominantly easterly wind. This is reflected in significantly lower sulphate at Yarner Wood for this year, and generally lower levels between Yarner Wood and sites in the east of England. Yarner Wood is a coastal location in the west of the UK, so it will have a direct contribution to the secondary component from sea-salt spray in the form of sodium chloride (NaCl) and sulphates. However, it is leeward of the prevailing wind so is usually not effected by the accumulation of secondary and far-travelled particles as seen across central and eastern UK. The long-term interpolated map of secondary particulate matter across the UK (*Figure 3.2*), generated by QUARG (1986), reflects this pattern: low concentrations of secondary particles to the west and north, higher concentrations towards the east and south, with the highest levels in East Anglia.

Comparisons were also made between these three sites using Pearson correlation, for the years 1995 to 2000. Site-to-site comparisons are presented in *Table 3.6*. These figures show a reasonably strong correlation between Stoke Ferry and Barcombe Mills, apart from 1999. With the exception of 1996, there is also a moderate to good correlation between Barcombe Mills and Yarner Wood. Both of these sites are in close proximity to the coast, so they will share some basic similarities in source characteristics. A much weaker correlation, however, is shown between Stoke Ferry and Yarner Wood. These results are logical

Table3.6: Pearson correlation coefficients (r) for SO_4 sites in southern England

	1995	1996	1997	1998	1999	2000
Stoke Ferry v Barcombe Mills	.58 *(p<0.001)*	.83 *(p<0.001)*	.76 *(p<0.001)*	.77 *(p<0.001)*	.49 *(p<0.001)*	.72 *(p<0.001)*
Stoke Ferry v Yarner Wood	.28 *(p<0.001)*	.10 *(p<0.098)*	.46 *(p<0.001)*	.60 *(p<0.001)*	.42 *(p<0.001)*	.42 *(p<0.001)*
Barcombe Mills v Yarner Wood	.53 *(p<0.001)*	.10 *(p<0.088)*	.55 *(p<0.001)*	.69 *(p<0.001)*	.66 *(p<0.001)*	.65 *(p<0.001)*
N	316	296	304	313	292	292

considering the position of these sites relative to one another: Yarner Wood in the west, Barcombe Mills in a central location on the south coast, and Stoke Ferry in the far east of the UK. This data also affirms the 1996 situation, where sites to the east are strongly correlated with each other, whereas Yarner Wood is very poorly correlated with both Stoke Ferry and Barcombe Mills. Combining data from all years [N=1813] gives r=0.75 for Stoke Ferry versus Barcombe Mills, r= 0.31 for Stoke Ferry versus Yarner Wood, and r=0.45 for Barcombe Mills versus Yarner Wood.

This investigation has shown that sulphate levels in the west of England are generally lower in magnitude than those in the east, and Yarner Wood does not exhibit the same day-to-day pattern of sulphate as Stoke Ferry. Some similarities are, however, shown between Barcombe Mills and Yarner Wood, both of which are in close proximity to the coast. Generally, Barcombe Mills and Stoke Ferry show a very similar pattern on a day-to-day basis, which confirms that sulphate levels are fairly uniform over the east and south of England.

3.4.5 Modelling of hourly sulphate

For the purposes of modelling, rather than taking a mean of sulphate from the average of two sites (i.e. Stoke Ferry and Barcombe Mills), like Stedman in London, it was felt that Stoke Ferry should be used as the sole input due to its much closer proximity to the study area and inland location.

A mathematical smoothing technique was applied to the daily sulphate from Stoke Ferry to model hourly sulphate and remove the stepwise nature of the daily values. For this purpose, the exponential smoothing function in SPSS was used. Daily values of sulphate were firstly attributed to each hour within the same day. A 'simple' model was then applied with linear trend and seasonal variation. The level of linear trend and seasonal variation are controlled by values of Alpha (α) and Gamma (γ), respectively, on a continuous scale ranging from zero to 1. Values close to zero mean that trend and seasonal variation are a function of short time periods (e.g. a few hours), whereas values close to 1 consider trend and seasonal effects to occur over long time-periods (e.g. several days). Initially, a *Grid Search*

function was used to observe the effects of using different values of α and γ. The best-fitting model gave $\alpha = 0.1$ and $\gamma = 0.1$, reflecting that levels of sulphate at any hour are influenced by levels found within a number of hours, up to one or two days. The smoothing function was applied to several groups of sequential hourly data because of breaks in the monitoring. These modelled sulphate values were used to represent background concentrations in all subsequent modelling of PM_{10}. *Figure 3.4* shows monitored daily sulphate from Stoke Ferry and modelled hourly sulphate from July 1999 to March 2000.

3.5 MODELLING LOCAL SOURCES OF PM_{10}

3.5.1 Types of dispersion model

A large number of dispersion models have been developed for modelling emissions from road (line) sources. These models vary in functionality and complexity, from simple 'screening' models, through intermediate models, to 'advanced' models. Screening models are typically used in making a quick and general assessment of air pollution; they are often applied in the early stages of an air quality management study to assess the potential for a pollution problem as a precursor to more complex modelling. For example, in the UK, the Design Manual for Roads and Bridges (DMRB) was developed in this context for assessing the impact of existing or proposed road schemes (DoT, 1994). DMRB is essentially a set of look-up tables, derived from a Gaussian dispersion model, for making broad estimates of pollution concentrations. The model only requires information on average traffic speed, peak hour traffic volume, and distance from the source to the receptor. No information on meteorological conditions is used, so the model can only make an assessment of the worst case or general (i.e. long-term) situation. Another type of screening model is CAR (Eerens *et al.*, 1993), developed in The Netherlands, for estimating average concentrations of CO, NO_2 and NO_X at kerbside locations, on an annual basis. The basic principle of CAR is that the concentration at any point is simply a function of the relationship between distance from source to receptor, and magnitude of the emission at source. CAR

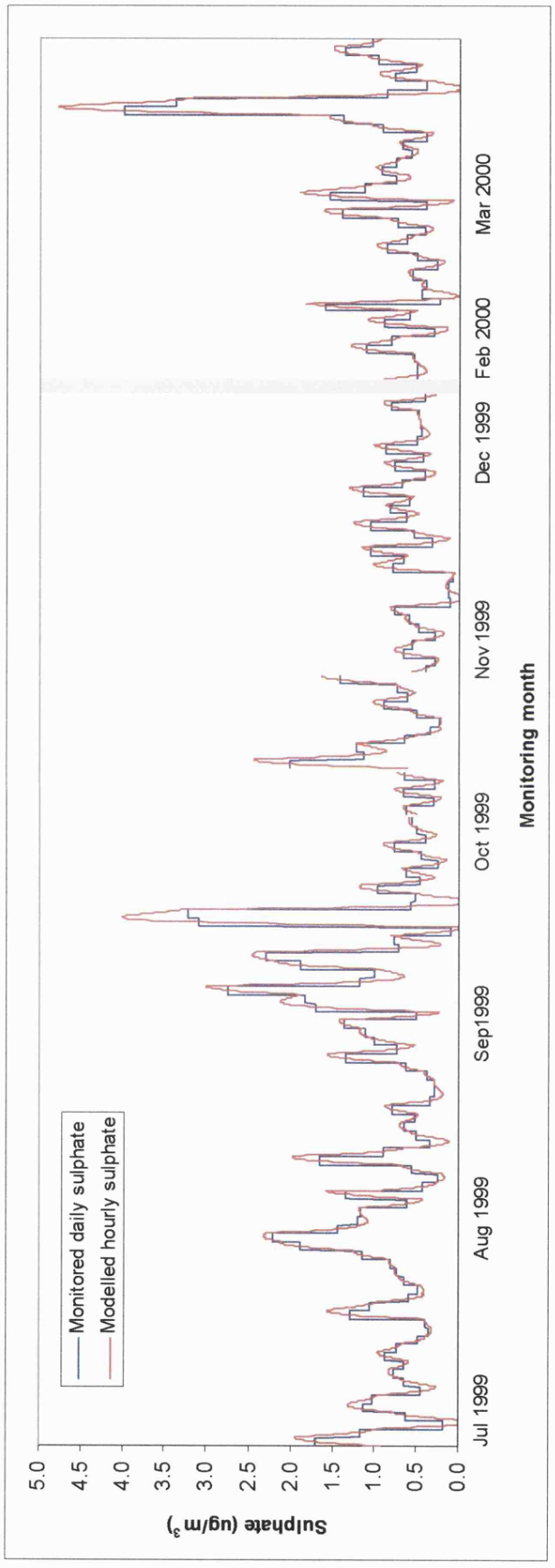


Figure 3.4: Monitored daily sulphate and modelled hourly sulphate (July 1999 to March 2000)

can successfully be used to provide a general survey of air quality in busy streets (Eerens *et al.*, 1993).

Intermediate dispersion models are those models that are capable of providing estimates over both space and time, whilst having relatively simple parameterisation and capabilities. These dispersion models apply Gaussian, or other, mathematical models of distribution, and use some meteorological data to achieve results over time intervals (e.g. hourly). An example of an intermediate model is AEOLIUS (Assessing the Environment Of Locations In Urban Streets), developed by DETR and the Meteorological Office. AEOLIUS can model concentrations for a number of pollutants (e.g. PM_{10}, NO_X, NO_2, CO) at a single receptor within a street canyon, on an hourly basis. AEOLIUS uses a standard Gaussian distribution but also allows for re-circulation within the canyon. The program determines the level of re-circulation at a point, by means of wind direction (either windward or leeward), receptor bearing, and distance across the street. The main inputs are weekly average traffic flow and speeds, and hourly meteorological data.

Advanced models can accommodate relatively large numbers of sources and provide concentrations at high spatial and temporal (i.e. hourly) resolution, for a number of grid or point receptors. They can use detailed information on traffic emissions and meteorology, varying by time of day and week. A distinguishing feature of advanced models is that they incorporate the effects that boundary layer conditions have on dispersion. Traditionally, the boundary layer has been characterised using Pasquil-Gifford (P-G) stability categories. These categories describe stability in the atmosphere in terms of wind speed, cloud cover, and mixing height (Pasquil, 1974). Where mixing height is not directly available, models often use cloud cover, wind speed, and temperature to make an estimate of mixing height. Examples of advanced models are CALINE4 (Benson *et al.*, 1992), AERMOD (Cimorelli *et al.*, 1998), and ADMS-Urban (Carruthers *et al.*, 1994). CALINE4, developed by the US-EPA, is a traditional advanced model in the sense that it uses a Gaussian-type method and describes the boundary layer by P-G stability category. AERMOD and ADMS-Urban are so-called 'new generation' models in that they model the vertical dispersion profile on a

continuous scale with greater understanding of turbulence within the boundary layer.

ADMS-Urban is part of the Advanced Dispersion Modelling System from CERC (Cambridge Environment Research Council), UK. The urban software is a model of dispersion of pollutants from road traffic, industrial, and domestic sources. ADMS contains a street canyon model, building effects model, photo-chemistry model, and complex terrain model. ADMS-Urban can be used as a stand-alone program or be used in conjunction with ArcView, ESRI's desktop GIS. The integrated GIS provides tools for creating point, line, and areas sources / receptors, and adopts some spatial analysis capabilities of ArcView for creating gridded pollution maps. An emissions inventory, in Microsoft Access, is available to store sources, traffic volumes and the emission rates of the various pollutants, and can be accessed directly from ADMS.

AERMOD is the equivalent development to ADMS in the US. It has been developed to supersede the ISC point-source model and extend the range of sources to lines and areas. Both ISC and AERMOD are part of the 'Breeze' suite of dispersion models. AERMOD is similar to ADMS in that it attempts to represent the vertical profile of the boundary layer on a continuous scale and provide a greater understanding of turbulence. However, AERMOD has not yet been endorsed by the US-EPA and is currently still being tested.

3.5.2 Performance measures

a) Introduction

The reliability of dispersion models has commonly been assessed by comparing modelled and monitored concentrations. A number of performance measures have been used in making these comparisons, including the coefficient of determination (r^2), Pearson's correlation coefficient (r), mean bias, factor of two (Fa_2) analysis, Fractional Bias (FB), Normalised Mean Squared Error (NMSE) or Root Mean Squared Error (RMSE). The coefficient of determination and factor of two have, however, been the most commonly used methods for quantifying model

performance. Factor of two describes the percentage of predicted values that are within the range of one-half and twice the observed values.

b) Studies reporting on model performance

Table 3.7 shows a summary of those studies reporting model performance in terms of r_2 and Fa_2, by pollutant and type of model. For Fa_2, the performance of models reported is in the range 25% to 99%, with the majority above 50%. There appears to be no clear pattern of model performance, however, when described by r^2. Only a few of these studies have evaluated the reliability of ADMS. Results by Owen *et al.* (1999), for modelling SO_2 and NO_X in London, are on the whole poor. The reasons for this are likely to be the lack of detailed knowledge of the temporal variations in emissions, and, secondly, the use of a steady-state model, which assumes that all pollutants emitted in one-hour will arrive at the receptor during the same hour. The model may predict well the levels of these pollutants but with a time shift (Owen *et al.*, 1999). Further results have been reported by CERC for modelling PM_{10}, NO_X and NO_2 for 1997 monitoring data, on any hourly basis (Carruthers *et al.*, 2000). For each pollutant, a nearby background concentration (Harwell, Oxfordshire) was added to the modelled value from ADMS-Urban.

In terms of r^2, results are poor for NO_2, generally poor for NO_X, but somewhat better for PM_{10}. For PM_{10}, Fa_2 is reported as 80%, which is comparable to results from other studies. One other study (de Hoogh, 1999) has reported on the performance of ADMS, but because this dealt with long-term averages it is not suitable to compare results with those from hourly studies.
In addition to r^2 and Fa_2, two other performance measures are often used: Fractional Bias (FB) and Normalised Mean Square Error (NMSE). Fractional bias (FB) deals with arithmetic mean bias, is non-linear, and bounded by +/- 2. Negative values of FB indicate under-prediction and positive values indicate over-prediction. An over prediction of exactly 2 times is 0.67. Ideally, FB will be zero. NMSE is a measure of the overall scatter in the data. Small values of NMSE denote better model performance.

Table 3.7: Performance of dispersion models - summaries from selected studies

Source	Pollutant	Averaging time	Site characteristics	Model	Factor of 2 (%)	r^2 (%)
Rodden *et al.*. 1982	CO	15 minute	Test conditions in Houston, Dallas, El Paso and San Antonio, USA	TRAPS IIM	80-97	17 - 48
				CALINE-3	80-95	17 - 37
				CALINE-2	83-97	21 - 30
				HIWAY	46-92	3 - 36
				AIRPOL-4	80-95	2 - 35
Rao *et al.* 1989	SF_6	1 hour	Artificial vehicle proving ground	GM	87	83
				AIRPOL-4	61	49
				HIWAY	56	26
				CALINE-2	52	24
				DANARD	51	14
				MROAD2	28	20
				ROADS	27	20
Yamartino & Wiegand 1986	CO	Half-hour	Street canyon, Cologne	MAPS	N/A	44
				STREET	N/A	44 - 58
				CPB	N/A	58 - 62
Benson 1992	SF_6	Half-hour	Vehicle test track	CALINE-4	N/A	61
	CO	1 hour	Intersection		N/A	77
	CO	1 hour	Freeway		N/A	53
	NO_2	1 hour	Freeway		N/A	59 - 64
	CO	1 hour	Intersection		N/A	73
	CO	1 hour	Highway		N/A	53
	SF_6	Half-hour	4 lane highway		N/A	31 - 35
Namdeo & Colls 1996	CO	1 hour	Urban street, Leicester	SBLINE	95	68
Reynolds and Broderick 1999	CO	1 hour	Street junction, Dublin	CALINE-4	71	64
	NO_X				71	74
	PM_{10}				71	66
	HC				75	79
	C_4H_6				29	71
	C_6H_6				42	71
de Hoogh 1999 (unpublished)	NO_2	Mean annual	28 monitoring sites across city, Sheffield	ADMS	N/A	42 - 53
				DMRB	N/A	86
				CALINE-3	N/A	69
Owen *et al.*, 1999	SO_2	1 hour	1 urban street, 1 urban background, 2 suburban, London	ADMS	0-1	13-47
	NO_X				3-30	24-67
Carruthers *et al.*, (2000)	NO_X	1 hour	Roadside on M4 and M25 motorways, UK	ADMS	28-59	2-34
	NO_2				60-72	4-7
	PM_{10}				80	24-25
Kukkonen *et al.*, 2001	CO	1 hour	Urban street (street canyon), Helsinki	OSPM	N/A	66
	NO_X				N/A	77
	NO_2				N/A	83
Kukkonen *et al.*, 2001	NO	1 hour	34m from centre of a major road, Southern Finland	CAR-FMI	67	67
	NO_2				71	72
	O_3				92	86

All of the measures form part of a 'tool-kit' that appears to be a standard for assessing the performance of atmospheric models (Owen *et al.*, 1999; Carruthers

et al., 2000; Kukkonen, 2001). This has been formalised as the 'Model Validation Kit' and is part of the BOOT package, developed by Hanna *et al.* (1999). In addition to the performance measures described above, BOOT can also calculate other statistics such as Mean Bias, Correlation, and Standard Deviation.

Another commonly used measure of model error is Root Mean Square Error (RMSE). NMSE is, essentially, RMSE normalised by the product of the means from the observed and predicted populations. Thus, NMSE better reflects the error between pairs of individual measurements. It is possible, however, to compute any of these performance measures in a standard spreadsheet environment. Equations for NMSE, FB, and FA_2 are shown in *Figure 3.5.*

Figure 3.5: Equations for performance measures

$$NMSE = \frac{\overline{(C_P - C_O)^2}}{\overline{C_P}\,\overline{C_O}}$$

$$FB = \frac{\overline{C_P} - \overline{C_O}}{0.5\left(\overline{C_P} + \overline{C_O}\right)}$$

$$FA_2 = 0.5 \leq \frac{C_P}{C_O} \leq 2$$

Where;

C_O is the observed concentration
C_P is the predicted concentration

3.5.3 Choice of dispersion model

ADMS-Urban is widely used in the UK by local authorities, government agencies, and for private consultancy through CERC. The Environment Agency has recommended ADMS-Urban as the standard platform for dispersion modelling. ADMS-Urban has been validated and tested in a number of contrasting urban locations within the UK (Carruthers *et al.*, 1999; Owen *et al.*, 1999) and in the US (Bennet and Hunter, 1997). Results suggest that the performance of ADMS-Urban

is consistent with other dispersion models, and ADMS-Urban performs better in some situations (Bennet and Hunter, 1997).

In terms of modelling personal exposure, one particular benefit of ADMS-Urban is that it can directly produce maps of pollution at both high spatial and temporal resolution (or provide a fine grid of point receptors from which to create pollution maps). ADMS-Urban was also a natural choice because it was extensively used as part of NAPS. In view of this evidence, ADMS-Urban was adopted as the system for all dispersion modelling in this research.

The following sections describe the data and methods used to implement ADMS-Urban. A fundamental stage to this implementation - and to the methods used in the following chapter - was the GIS. The remaining sections of this chapter start with a description of the GIS development.

3.6 GIS DEVELOPMENT

Development of the GIS and related data was performed with ArcView GIS. Aerial photography (1:10000 scale) was used as the primary base for data capture, consisting of ½ by ½ km TIFF images at 50cm pixel resolution (supplied by Cities Revealed). To cover the study area (7.4 km^2), and a surrounding area for control, 110 tiles were obtained to provide a 5 by 5½ km (27.5km^2) base. The TIFF images were geo-referenced in ARCVIEW by creating 'world' files containing British National Grid (BNG) references for each image. The 'world' files are used both to register the top-left corner of each image and define the X-Y extent of the image (i.e. by pixel size) in geographical space. Thus, when applied to a series of images, a seamless mosaic is formed. In addition, a 1:50k raster (source: OS) was used to digitise features from the surrounding area.

A number of themes were created within ArcView from the aerial photography, with the OS raster being used to create line sources from the surrounding area.

3.7 IMPLEMENTATION OF ADMS-URBAN

In broad terms, ADMS requires five groups of basic information to model from line sources:

- Traffic composition (type, count, speed)
- Traffic emissions (hourly)
- Meteorological parameters
- Road geometry (coordinates, width)
- Receptors (point, grid)

There are also a number of advanced options such as the street canyon model, complex terrain, building effects and chemistry for use in modelling NO_2. A discussion of the methods used to establish the basic inputs now follows.

3.7.1 Traffic data

SATURN was used to model traffic flows in the study area. SATURN (van Vliet, 1982) is a vehicle assignment model that can model vehicle movement, and the distribution of vehicles for a network of roads. In determining the flow and distribution of vehicles, SATURN simulates vehicle movement characteristics such as acceleration, idling and cruising by considering the geometry of roads, road capacity, vehicle speeds and vehicle control (e.g. position and function of traffic lights).

An existing SATURN model of roads for Northampton, and the surrounding area, was obtained from Northamptonshire County Council (NCC) and used as a basis for producing a model of traffic flows and vehicle speeds within the study area. The existing model represented the 1995 situation and therefore required calibration to create the 1997 scenario. Data on traffic flows, turning movements and journey times were required to calibrate the traffic assignment model (SATURN). To develop an understanding of the traffic movements in the study area, automatic traffic counters were installed at two sites. Pneumatic tube detectors were laid across the carriageways and intelligent data loggers recorded

vehicle movements according to vehicle classification and speed. These surveys were completed in two phases: 11/8/97-18/9/97 and 10/10/97-20/10/97. The data were used to identify any extraordinary events in the study area (e.g. the effect of market days) and to collect comparison data for the main survey date (15/10/97). The turning movement surveys were completed between 0700-0900 (am peak), 1200-1400 (midday peak) and 1630-1830 (pm peak). The turning movement surveys involved 15 observers, classifying and counting the traffic making specific turning movements at three of the main signalised intersections in the study corridor; turning movements at the fourth junction were monitored using the CCTV system based at Northamptonshire County Council Area Traffic Control Centre.

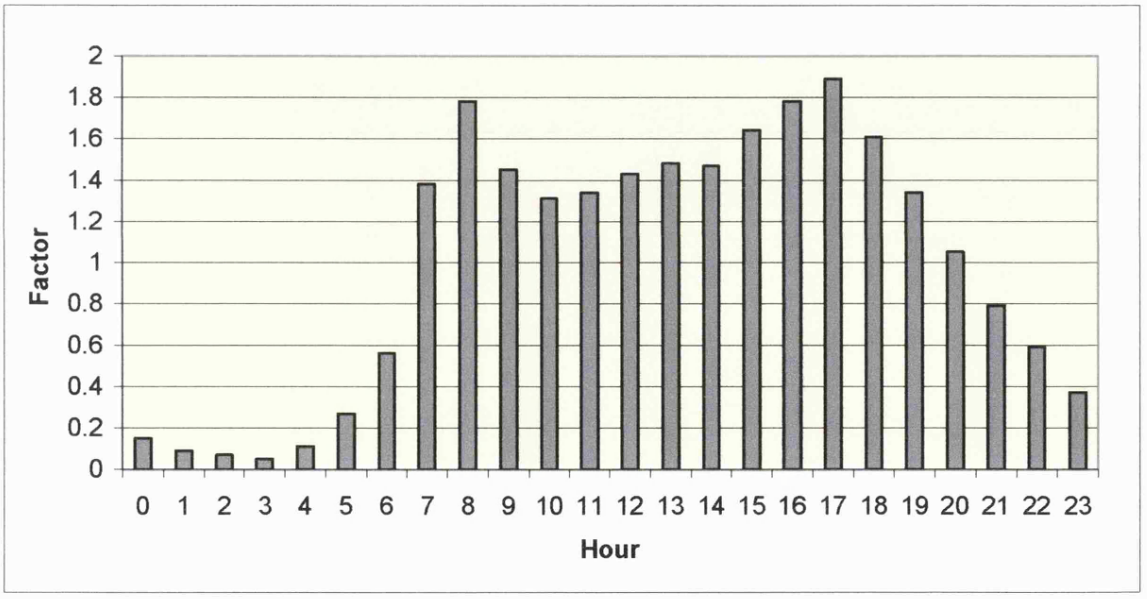
The amount of delay experienced along the major routes in the study area was assessed through journey time surveys. A driver and an observer pair drove repeatedly along designated routes, recording the time at which specified landmarks were passed. Timing vehicles completed 12-16 routes during each time interval and drivers were requested to drive in a representative manner. Local police and traffic control personnel were consulted before and after the surveys to ensure that no unusual events, incidents or roadworks were occurring, which might lead to the collection of unrepresentative data. Modelled flows from SATURN are shown in *Figure 3.6* as a daily total for each road.

Further monitoring of hourly traffic flows was undertaken in June and September 1999 to validate the 1997 SATURN flows for application to 1999 modelling scenarios. The same method of pneumatic tube detection was used at a single location on the main route through the study area, just to the south of busy intersection of the A5199/A508. The two campaigns provided about four weeks of hourly traffic data in total [n=670 hours], from which an average flow was calculated for each hour of the day, separated into weekdays and weekends. These values were then compared with the 1997 flows for the same road sections in SATURN. Comparisons of the 1997 (modelled) and 1999 (monitored) flows showed a high level of consistency for each hour of the day.

Figure 3.6
Modelled traffic flows from SATURN

The 1999 data were therefore used to estimate time varying emissions factors (*Figure 3.7*) in ADMS and these were applied to all roads with traffic data in the study area. ADMS only allows one set of factors for the whole study area. Nevertheless, the traffic in the study area is a function of the main route, so it is reasonable to assume the factors derived from monitoring data on the main route can be applied to the rest of the network. As this shows, the peak traffic falls within the hours 0800-0900 and 1700-1800. Traffic flows are lower during the day until 1500hrs when they increase towards the evening peak. Traffic flows rise sharply in the morning before the AM peak hour and fall more steadily in the evening. The lowest traffic flow is seen between 0300 and 0400 hours.

Figure 3.7: Time varying emission factors applied to average hourly traffic data



3.7.2 Road Geometry

Two methods were available for entering road sources into ADMS-urban: 1) interactively, by digitising each line source, in turn, from a base-map in ARCVIEW, from which each vertex is read into the ADMS interface, or, 2) importing a list of pre-defined vertices, representing the whole road network, through the emissions

inventory. The latter method requires extraction of the road geometry from a GIS into a format suitable for the emissions inventory. Although this method is time consuming - particularly for city-wide data sets - it is preferable in situations where the road network is to be used in other GIS-based analysis, in order to keep the spatial data consistent.

In this study, for example, it would not be possible directly to use road geometry created within the ADMS interface for subsequent network analysis, because in ADMS there is no connectivity (topology) between the road sources. To use the ADMS sources for network analysis would require manipulation in a GIS that may change the geometry of the road network, thus creating secondary, inconsistent data.

The road network was therefore digitised in ARCVIEW, as road centre-lines, to provide single arcs for all types of carriageway (i.e. essentially, the same as Ordnance Survey's OSCAR family of products). It should be noted that ADMS treats all roads in this way regardless of the number of lanes and direction of flow. In this process, roundabouts and other complex junctions were reduced to simple crossings. Two groups of sources were digitised corresponding to SATURN: 1) roads contained within the study area, and 2) significant road sources further afield, covering the whole of Northampton. There are no significant sources of particles from industry, within the study area. The two sets of road sources were joined using the APPEND function in ArcInfo. These joins were checked manually using the aerial photography as a back-drop.

The road geometry was then exported and edited for transfer to ADMS-Urban. The road width is specified for each source in the inventory before final transfer. Ideally, it would be possible for ADMS to read the road sources directly from existing data (e.g. OSCAR from Ordnance Survey), but this facility is not available at the present time.

3.7.3 Linking SATURN and the GIS

Modelled traffic counts and vehicle speeds were attributed to the road sources in ADMS via Arc/Info GIS by applying SATURN naming conventions to each road section. To define the flow of traffic in SATURN, each road link is delimited by a pair of nodes (e.g. 1001 to 1002); thus a single direction carriageway is represented by a pair of nodes (e.g. 1001 to 1002), and a bi-directional carriageway has two pairs of nodes (e.g.1001 to 1002 & 1002 to 1001). ADMS, however, does not refer to the direction of flow but merely treats traffic on each road as a total volume. In the GIS, therefore, each road was coded in order to collapse directional flow to a single value that represents the total volume. SATURN flows were then automatically transferred to each road section in the GIS using the JOINITEM command.

3.7.4 Emissions database

Model parameters can either be entered directly into the ADMS screen or imported from an emissions inventory (i.e. Microsoft ACCESS database), with the exception of meteorological data which is held in a 'text' file system. The advantage of using the inventory is that a database of sources and emissions can be developed, modified, and easily transferred to and from ADMS and other software (e.g. GIS, spreadsheets). In the inventory each source is defined by its road geometry, traffic data and other source characteristics such as road width, road height, canyon width etc. There is also the facility to change the type and number of pollutants for which emissions will be calculated. The inventory is particularly useful when combining data from disparate sources, and is necessary where the number of inputs is large.

In this case a separate inventory was developed for each group of road sources: a) study area only and b) study area plus wider area. The inventory held three of the five main inputs: road geometry, road width, and traffic composition.

3.7.5 Emissions

Following transfer to ADMS-Urban, road-source emissions were calculated from the traffic data held in the emissions inventory. The current version of ADMS-Urban uses the 1999 model of DMRB emissions. These are essentially a set of factors that convert traffic flow to emission rates, depending on traffic type (e.g. light or heavy vehicles) and vehicle speed. These emission correction factors also vary between pollutant. The emissions factors are automatically applied to the traffic data on transfer to ADMS-Urban. It is possible, however, to supply pre-defined emissions for each source. The traffic data, and therefore the emissions, were thus adjusted for each hour of the day by specifying the time varying emissions factors (as shown in *Figure 3.7)* in the ADMS interface, thereby giving an hourly weighted flow.

3.7.6 Meteorological parameters

In ADMS-Urban, the boundary layer is characterised by the boundary layer height *h* and the Monin-Obukhov length L_{MO}, providing continuous variation through the boundary layer, rather than a Pasquill-Gifford category, which corresponds to ranges of h/L_{MO}. Monin-Obukhov length is defined as:

$$L_{MO} = \frac{-u_*^3}{KgF_{\theta_0}/(\rho c_p T_0)}$$

where;

u_* is the friction velocity at the Earth's surface
κ von Karman constant (0.4)
g is the acceleration due to gravity
$F_{\theta 0}$ surface sensible heat flux
ρ density of air
c_p specific heat capacity of air
T_0 the surface temperature

Table 3.8 shows median values of wind speed, Monin-Obukhov length (L_{MO}) and boundary layer height (h) corresponding to each P-G category. A is the most

unstable category and G is the most stable. Category D refers to neutral conditions ($h/L_{MO} = 0$), commonly associated with overcast conditions.

Table 3.8: Corresponding Monin-Obukhov lengths for P-G categories

$U(ms^{-1})$	$L_{MO}(m)$	$1/L_{MO}(m)$	h(m)	h/L_{MO}	P-G category	Condition
1	-2	-0.5	1300	-650	A	Unstable
2	-10	-0.1	900	-90	B	↓
5	-100	-0.01	850	-8.5	C	↓
5	∞	0	800	0	D	Neutral
3	100	0.01	400	4	E	↓
2	20	0.05	100	5	F	↓
1	5	0.2	100	20	G	Stable

(source: adaptation of ADMS-Urban 2000 user guide)

Translation of P-G categories gives the following stability classes for L_{MO}:

Stable $h/L_{MO} \geq 1$
Neutral $(-0.3 \leq h/L_{MO} < 1)$
Convective $h/L_{MO} \leq -0.3$

In unstable conditions, the Monin-Obukhov length is negative. In the UK, during winter months, h/L_{MO} is typically small and negative, which reflects either moderate convective activity and / or overcast conditions. For stable and neutral conditions, ADMS uses a standard Gaussian approach in the vertical profile, and a non-Gaussian approach in convective conditions. This follows evidence from field experiments (Briggs, 1985) showing that the vertical profile in convective conditions is skewed and significantly non-Gaussian.

ADMS can make an estimate of h/L_{MO} from a number of possible inputs. Meteorological data are entered either as hourly sequential (one line of data for each hourly condition) or statistical (long-term frequency analysis) data. It is appropriate to use hourly data for short-term calculations and either hourly or statistical data for long-term calculations. The minimum data requirements are:

- wind speed
- wind direction

plus one of the following:

- sensible surface heat flux $F_{\theta 0}$
- cloud cover, time of day, and time of year
- reciprocal of the Monin-Obukhov length $(1/L_{MO})$

In practice, values of L_{MO} are difficult to obtain because calculations require measurement of wind speed and temperature at two heights, simultaneously. ADMS claims to make a good estimate of boundary layer height - when the site is in mid-latitudes (e.g. UK) - using cloud cover, time of day, year and temperature. These data are readily available from most climatological stations in the UK and thus are the most common inputs for applications using sequential data. In this case, wind speed, wind direction, cloud cover and temperature were used to estimate all boundary layer characteristics.

The following meteorological parameters were used: year, Julian day, wind speed, wind direction, surface temperature, all measured at a purpose-deployed local site within the study area, and cloud cover from Wittering, the nearest national Meteorological Office monitoring site (at Wittering, Cambridgeshire, about 60 km to the east). Meteorological parameters were entered into MET files which are called-in by ADMS during each model run. An example MET file, using hourly parameters, can be seen in *Appendix 3A*. ADMS processes each line of meteorological data, sequentially, estimating the boundary layer conditions for each hour. In sequential mode, the model will recognise the state and contribution of boundary layer conditions from the previous hour.

3.7.7 Other parameters

A number of other parameters are required by ADMS-Urban. These include surface roughness, height of recorded wind speed, and latitude of the study area. Surface roughness is classified by type of location (e.g. city, woodland, grassland etc). In this case, a default value of 0.5m was used for urban and sub-urban locations. The height of recorded wind speed was measured at the meteorological

site in Kingsthorpe, Northampton as 6m. The latitude of the study area is approximately 52° N.

3.7.8 Sensitivity analysis

Sensitivity of ADMS-Urban to data inputs was assessed using a number of parameters from the same two-month period from winter 1999. Sensitivity tests were carried out by varying independently five model parameters: meteorology, surface roughness, vehicle speed, traffic flows, and terrain (with or without). *Table 3.9* shows the parameters, tests, and differences between the results under each variation and those obtained using the default parameters.

Table 3.9: Sensitivity of ADMS-Urban to alternative model parameters

Parameter	Test	Mean percentage difference	Median percentage difference	St. Dev. of difference	percentage within a factor of 2
Meteorology	Equivalent variables from Wittering (Cambs.)	62.85	61.48	68.80	32
Surface Roughness	1.0m	12.23	7.87	11.65	100
	0.25m	10.34	4.80	11.39	100
Vehicle speed	+ 5 km/hr	4.10	3.77	1.20	100
	+10km/hr	7.55	7.34	1.96	100
	- 5 km/hr	6.74	5.41	3.39	100
	-10km/hr	15.77	13.64	7.30	100
Traffic flows	+10%	10.80	11.02	0.78	100
	-10%	10.60	9.92	1.41	100
Terrain	Included	33.60	20.61	37.16	92

Two alternative values were picked for surface roughness, which were the adjacent categories to the default (0.5) in the ADMS-Urban screen. Vehicle speeds were altered by both separately increasing and decreasing the average speed for each road section by 5 km/h and then 10km/hr. A change in vehicle speeds of +/- 10km/hr ought to represent a significant error in the traffic modelling. In a similar way, traffic flows were reduced and increased by 10% of modelled traffic flows. Meteorological inputs were assessed by comparing modelled

concentrations using local data on wind speed and wind direction from within the study area against the equivalent data obtained from Wittering.

As *Table 3.9* shows, of all the tests, meteorological data from a remote site has by far the greatest impact on modelled concentrations. It is also worthy of note that the use of meteorological data from the remote site led to severe underestimation of monitored concentrations. This is to be expected as Wittering, relative to Northampton, is a flat exposed site, which, in general, experiences wind speeds of a higher magnitude.

Cloud cover is also available from Northampton but only as a daily spot-measurement, at 9am, taken from the University College grounds. Direct comparisons of cloud cover between Wittering and Northampton using three years' of data, for 0900 hours, show that on 35% of occasions there is no difference between sites and 35% observations are within one Okta. Sensitivity tests show that varying cloud cover by +/- 1 Okta from the observed value can vary modelled concentrations up to four-fold (pers.comm. C.Arciszewska).

The inclusion of terrain also has a comparatively big impact on modelled concentrations. It was not feasible, however, to consider including terrain in the modelling, as this led to greatly increased model run-times. The effect of all other parameters was shown to be moderate.

3.8 IMPLEMENTATION OF OUTDOOR POLLUTION MODEL

On the basis of this investigation, the following model was developed to predict outdoor concentrations of PM_{10}. The outdoor pollution model comprised two main components: 1) local, primary sources modelled in ADMS-Urban, and 2) the nearest rural sulphate site to represent the regional and far-travelled secondary. Thus, for hourly estimates, the contribution from local sources varies by space and time, but the secondary / far-travelled matter is assumed to vary only with time. Rather than allowing for other particles to be explained by the constant - which imposes a minimum threshold for PM_{10} concentrations - regression through the

origin was used. In this approach, traffic-related, other particles are represented by local modelled PM_{10}, as discussed in *Section 3.2.1*, and all other particles are represented by the secondary component. Therefore, the model assumes that non-traffic, other particles and secondary particles are uniformly distributed.

A single receptor was included in ADMS-Urban to represent the fixed-site monitor in Kingsthorpe. The receptor was located at a distance of 9m from the centre-line of the nearest carriageway, and at a height of 3m. ADMS-Urban was executed for all hours of available monitoring data that had corresponding meteorological data from both the Kingsthorpe site (wind speed, wind direction, and temperature) and Wittering (cloud cover). An example of an ADMS-Urban input file is shown in *Appendix 3B*. A wind rose for the period used in outdoor air pollution modelling is shown in *Appendix 3C*. On completion of the run, each hour was allocated a modelled hourly value for SO_4 based on data from Stoke Ferry. Model output from ADMS-Urban was removed for those hours without SO_4 data.

The final model was developed by regressing local modelled PM_{10} from ADMS and the modelled hourly rural sulphate from Stoke Ferry against the monitored hourly PM_{10} concentration from the fixed-site monitor. The model was built by randomly selecting 70% [n=2651] of the total available monitoring data [n=3908], with the remaining data kept for model validation [n = 1257]. Linear regression through the origin (the no-intercept model) was used to allow for PM_{10} levels to be zero. This gave the following model:

PM_{10} = (1.322 * local modelled PM_{10}) + (14.228 * Sulphate)

Table 3.10 shows the performance on the model developed to predict hourly PM_{10} at the fixed-site monitor. For regression through the origin, r^2 measures the proportion of the variability in the dependent variable about the origin explained by the regression. It must be remembered that this cannot be compared to r^2 for models that include the intercept. As the table shows, the model gives a highly significant fit to the monitored concentrations. The model was subsequently used to provide all estimates of outdoor pollution within the study area.

Table 3.10: Performance of the outdoor pollution model using 70% of monitoring data

r	r^2	*Adj.* r^2	*SEE* ($\mu g/m^3$)	*Sig.*	*N*
.884	.781	.780	10.40	.000	2651

3.9 SUMMARY

This chapter has described the sources and characteristics of PM_{10} in the UK. A discussion on existing methods of quantifying and modelling air pollution has then been provided, before describing the methods used to develop and implement a model to estimate outdoor air pollution.

4 DEVELOPMENT OF EXPOSURE MODELLING

4.1 INTRODUCTION

The previous chapter has demonstrated the method used in developing a model for estimating PM_{10} at outdoor locations. In this chapter, that method is used as a basis for developing a Travel and Occupancy Time Exposure Model (TOTEM) as part of STEMS. The overall framework for TOTEM is explained in Section 4.2, before describing the methods used to develop and calibrate the individual components of the model (section 4.3 to Section 4.6).

4.2 TOTEM (Travel and Occupancy Time Exposure Model)

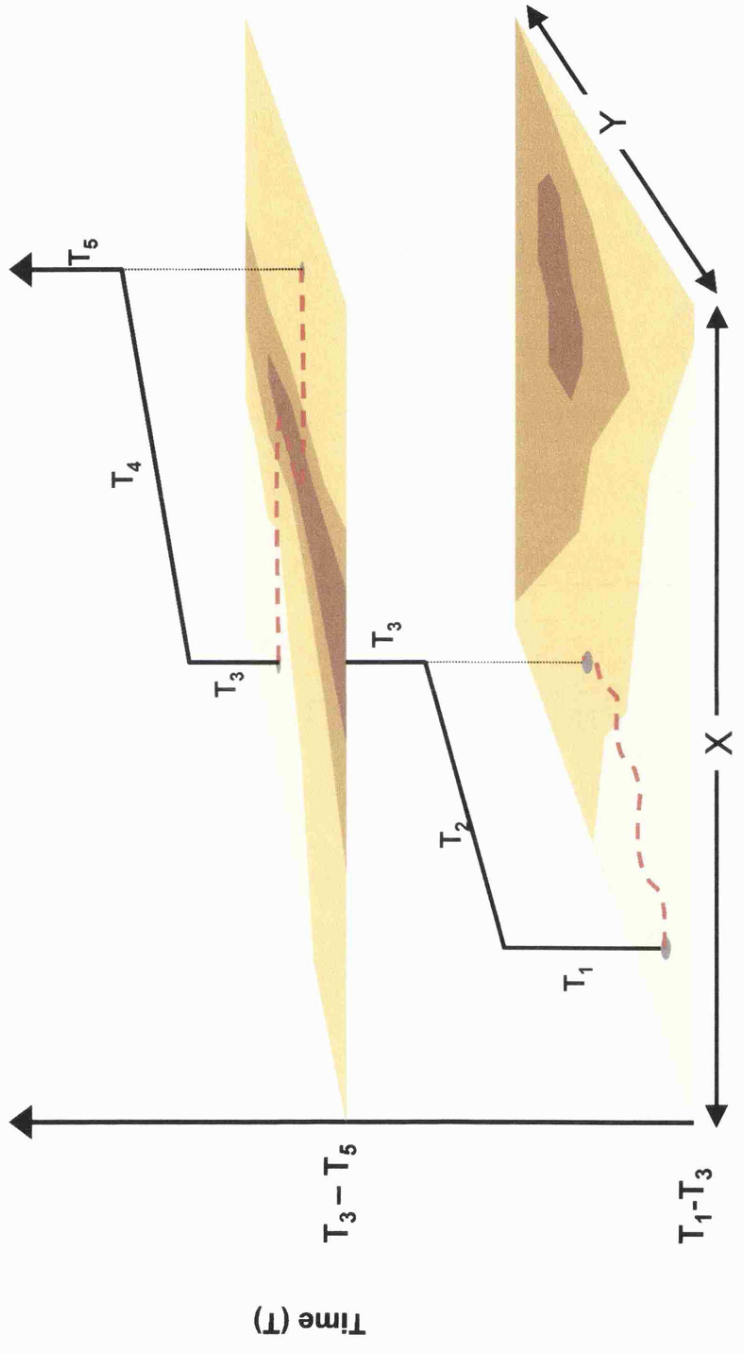
4.2.1 Outline of the model

Any individual, or population, can be exposed to air pollution in one of the following microenvironments: indoor at a fixed location, outdoors at a fixed location, or moving between outdoor locations. Exposure in these environments can potentially occur at any time during the day. The aim of TOTEM is to be able to provide exposure estimates for these micro-environments, in order to calculate total (or average) exposure for any given time period, for an individual or population. A conceptual framework for TOTEM is illustrated in *Figure 4.1*. This shows a diary of activity in both space and time, for an individual at fixed site locations (T_1, T_3, T_5) and moving through a field of pollution (T_2, T_4). The estimate of total exposure *ET* for this activity is a function of the sum of concentration *C* at location *j* (x, y, z) in micro-environment *m* over time *T* at location *j*. This model is based on the general exposure model by Duan (1982):

$$ET = \sum_{j=1}^{m} CjTj$$

Where;

C = concentration at location *j* (x, y, z, c)
T= time spent at location *j* (x, y, z, t)
j = location (x, y, z) of micro-environment *m*



Activity	Start location X	Start location Y	End location X	End location Y	Time	Exposure
T_1	200	100	200	100	30	10
T_2	200	100	300	500	15	13
T_3	300	500	300	500	35	15
T_4	300	500	800	900	10	25
T_5	800	900	800	900	30	20

Figure 4.1: Conceptual model for TOTEM

The following sections detail the methods employed to calculate exposure in each microenvironment and how each component feeds into TOTEM to estimate an integrated (total) exposure measure.

4.2.2 Pollution maps and point estimates

The basic input for TOTEM is maps of pollution. ADMS can be set to produce both a gridded contour plot and the concentration at individual receptor locations, for the range of pollutants specified, for a range of time intervals (e.g. hourly, daily etc). ADMS can run in either receptor or gridded output mode, or do both at the same time. The extent of the gridded surface can either be manually entered into the ADMS interface, or automatically read by using a click-and-drag box in ArcView. ADMS then produces a single *regular* tessellation of 32 by 32 points, which is used as a basis for creating a gridded map. The spacing between these points is dependent on the extent of the gridded area; thus the grid has variable resolution. In addition to this default, the *intelligent gridding option* can be selected to add hundreds of further points where the steepest changes in the pollution surface are seen. This option is usually required for complex or large numbers of road sources. For either *regular* or *intelligent* gridding a *z* value is used to specify the height above ground (0 to 3000m) at which the grid calculations are made.

The gridded and additional (i.e. intelligent) point data are then read into ArcView with the Spatial Analyst extension. Inverse Distance Weighted (IDW) interpolation is used to produce a gridded surface from the point data. IDW determines cell values using a linearly weighted set of sample points. The weight is a function of inverse distance. The influence of surrounding points is expressed using a power function. ADMS uses 2 as the default power function. The best results from IDW are obtained when sampling is sufficiently dense with regard to the local variation we are attempting to simulate (Watson and Philips, 1985).

In the Northampton study area, using a single grid (with the intelligent option) to cover the study area provided a minimum distance between points of about 32m. As a basis for interpolation this does not provide enough detail in areas of steepest

pollution gradient. To obtain a higher resolution, six separate grid areas were used to tile the study area. Each grid measured 1.5 x 1.5 km with a minimum distance between points of about 8m. Unfortunately, however, this method significantly increases the number of model runs – one model run per grid. The output from each model run was exported as a series of point-based concentrations and then merged into a single dBASE file. Finally, the dBASE file was used to create a point coverage as a basis for interpolation in ArcView. Parameters for IDW interpolation were taken from the ADMS default. Each pollution map covered a total area of 14 km^2 (i.e. 4.5 x 3 km) with a resolution of 5m grid cells.

In much the same way, the co-ordinates of receptors can either be entered manually in the ADMS interface, or by using the *Add Receptor* tool in ArcView and clicking a location on the map base, from which the co-ordinates are read into ADMS. Up to fifty points are allowed in any single model run. Therefore, for a large number of receptors it is advisable to produce gridded output and use a point-in-cell operation in the GIS to assign a concentration to each location. In order to model exposure for any number of individuals, the latter method was therefore adopted.

Point-based exposure estimates can be obtained by simply intersecting a coverage of point locations with any pollution surface. Therefore, estimates can be obtained on an hour-to-hour basis, for any number of individuals.

4.2.3 Travel-time exposures using network modelling

A network system, by definition, is a system of interconnected linear features through which energy, matter, or information is transported. The routes and pathways taken by people moving through the environment are themselves real-world network systems. Network analysis, or path-finding tools, are offered by most proprietary GIS software. Many GIS offer the capability to examine networks in both raster and vector environments.

The earliest network models tended towards raster because of the ease of processing. With recent improvements in computer technology, however, network

applications have tended towards a vector environment. One of the best known examples of the use of network analysis in a vector environment is Microsoft AutoRoute©. Indeed, these techniques are well established in the fields of transportation / distribution and communication to applications of route finding and service provision (Goodchild, 1998).

Some of the more recent developments include, for example, a system for modelling demand for freight, within Belgium, using travel-time matrices (Beuthe, 2001), and a system for the routing of ambulances to incident sites and then to the closest hospital (Derekanaris *et al.*, 2001). Network techniques have also been used to manage the flow and distribution of water systems (Burrows, 2000), and other utilities (Mahoney, 1991). These GIS are traditionally referred to as Automated Mapping / Facilities Management (AM/FM). Despite the analysis of routes and pathways being well established in GIS, only a few examples exist that use network techniques for modelling exposure. Lovett *et al.* (1997) and Brainard *et al.* (1996) have used GIS-based network tools in developing a model to assess the exposure of the resident population to transportation of hazardous waste by road. To select transportation routes, the model employs algorithms to compute the shortest travel time between the producer and disposer of waste.

A purpose-built network analysis tool for calculating outdoor exposure to air pollution was created using AML (Arc Macro Language). The routine runs within Arc/Info GIS. Two separate networks are available: a network of road centre-lines for car journeys, and a network of footpath centre-lines for walking. Both of these networks were created from aerial photography. In simple terms, the network routine performs a line-in-polygon overlay to extract the concentration from each grid-cell in a given pollution map, and assigns the concentration to the segmented vector, which represents part of either the network of roads or walkways (*Figure 4.2*). A route is then selected, based on start and end locations of a journey, and the exposure along the route is calculated.

It must be borne in mind, however, that although exposure is calculated along two separate networks, both networks are given the same concentration where pathways follow the road network. This limitation is intrinsic to the way that ADMS

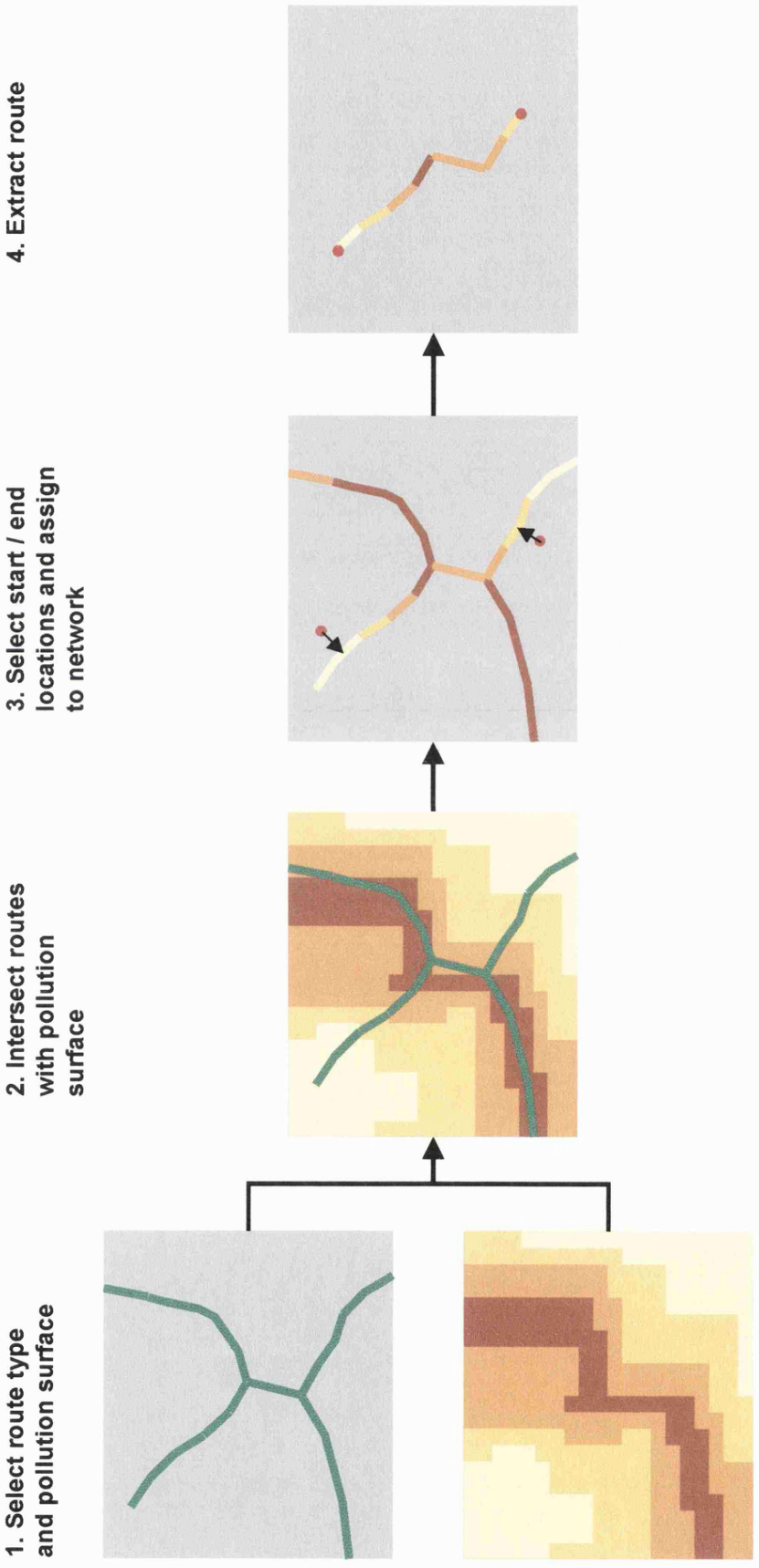


Figure 4.2: The STEMS network analysis procedure

places receptors in production of pollution maps. ADMS will place receptors at a minimum, perpendicular, distance from the road centre-line of 0.45 times the width of the road; thus, for a road of 10m width the receptor will be 4.5m from the centre of the road and 0.5m from roadside. The next receptor will be placed a further 8m from the first at 12.5m from the centre of the road and about 8.5m from roadside. Where pathways follow roads, pathways are nearly always within about 2m of the edge of the road; therefore, pathway and road concentrations are effectively from the same, roadside receptor points. It is, however, essential to use separate networks for walking and car journeys to reflect the differential rates of movement along each network. A full description of the network analysis algorithm follows. The program script is shown in *Appendix 4A*.

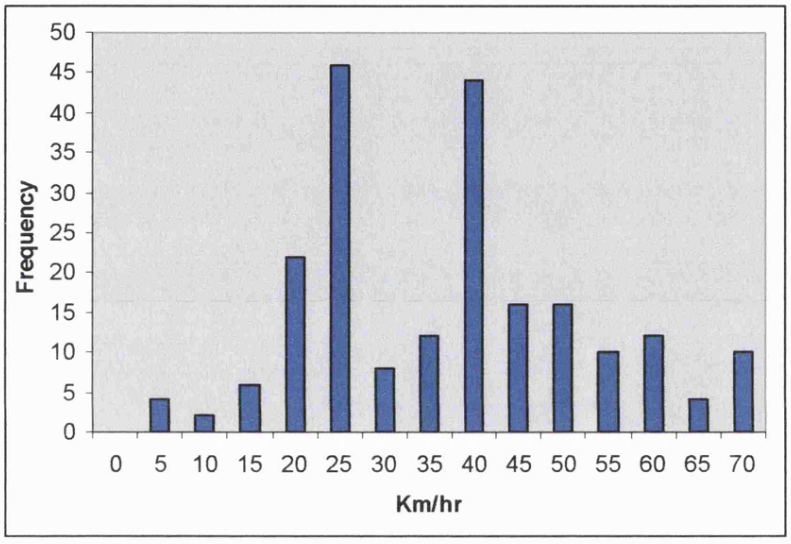
Before the network analysis program is executed, a sub-program (GRIDCONV) is called (*Appendix 4B*) that transforms the pollution maps from a raster data structure (N.B. output from ADMS) to an equal-area tessellation of polygons, by using the GRIDPOLY command. Each polygon is therefore the equivalent of a grid cell in the raster map. Each polygon is 5 x 5m.

The program then requires the following input: mode of transport (i.e. walk or car), vehicle or other speed of movement, pair(s) of co-ordinates to define the start and end locations for each journey (e.g. home and school), and a unique identity for each pair of locations. From the list of locations, a point coverage is created as a basis for route selection. The points are assigned to the nearest node (point) on the network (i.e. roads, walkway) by the minimum distance rule, using the NEAR command. Because the route is broken up into very small sections - as a result of the line-in-polygon overlay - the point allocation will always be near to perpendicular from the road segment, unless the point is nearest to a node (i.e. dangle) at the end of the network. These points become temporary nodes in the network and are used to delimit the subsequent routes. In this case, the aim of route selection is to determine the quickest path between the start and end points.

For each network, the speed of movement is defined in the Arc Attribute Table (AAT). For roads, two categories are used to describe vehicle speeds along the network: 1) Major roads – vehicle speeds from the SATURN network, 2) Minor

roads – a fixed speed of 30km/hr for all other roads in the study area. The frequency distribution of vehicle speeds within the road network can be seen in *Figure 4.3*. As this shows, there are two peaks in the distribution of vehicle speed. There are a large number of roads with speeds of 40km/hr and 25km/hr. Roads with a speed of 25km/hr are generally those that represent either congested sections of the network or small lengths of road at junctions. A fixed speed of 5km/hr is used for pedestrians.

Figure 4.3: Frequency of vehicle speeds on roads within the study area



Routes are selected based on the shortest-time to travel between two locations. The speed (or rate) of movement in the network is described by the *impedance* (or resistance). For each *arc*, a single impedance value is stored in seconds to describe the amount of time it takes to travel that part of the network. The following equation is applied to calculate the network *impedance* for each *arc*:

$$I = \frac{L}{S/3.6}$$

Where;

I = Impedance (s)

L = Length of each arc (m)

S = Speed of movement along each arc (km/hr)

The suite of commands in NETWORK is then utilised to perform route selection. The PATH command is used to calculate the shortest route using *impedance* values attached to each arc. (It is also possible to use alternative selection rules; for example, shortest route.) A new coverage is created containing only the selected route. This route is then the basis for exposure analysis. The network routine is designed to return average exposure along each route, but can also return cumulative and peak exposure, or the length of route above an exposure threshold. There is also the scope to display a profile of the pollution along a route, from which the above measures are made. These exposure measures are subsequently exported from the GIS to a text file. The output file records the *ID* of the start and end locations of the route, the start and end coordinates of these locations, the time taken to travel the route (in seconds), distance, and exposure calculations (e.g. average, total etc.). There are no limits to the number of routes than can be analysed in any single model run.

4.3 MODEL CALIBRATION

4.3.1 Time-activity

The population as a whole spends much of its time indoors. It is estimated in temperate regions that about 80% of time is spent in indoor locations (Farrow, 1997). A number of studies have shown that occupancy time can, however, vary significantly by population subgroup. For example, a study in Hong Kong (Tso and Leung, 2000) reported that home occupancy for babies was 83.3%, while primary school children spent 66.7 % of their time at home and secondary school children spent 54.2%. A study of 26 elderly US residents by Williams (2000) found that the average time spent indoors to be 97%. A study in California used a telephone questionnaire to gather time activity data over a 24-hour period for 1780 people. The average occupancy of the complete sample was found to be 61.9%, with toddlers and elderly having occupancy of 73% and 74%, respectively. Apart from Farrow (1997), there have been very few studies in the UK on time spent indoors by different population subgroups.

In this study, data on time activity were required to provide information on patterns of movement and places of residence of people in the area. Three main target groups were identified in this context: primary school children, College students and adult residents. Time activity surveys were undertaken for each group, as shown in *Appendix 4C.*

In each case a time activity diary was completed, describing activities over a 24-hour weekday period, in hourly intervals. The survey of adult residents was conducted by scripted telephone questionnaire during December 1997. In order to minimise bias, a strict protocol was applied. Names were selected from the telephone directory using random numbers, and these were contacted by telephone between 18:00 and 20:00. If the call was answered and the named individual was present, he/she was asked to undertake an interview. If the call was answered, but the named individual was absent, an appointment to call again was made. If there was no answer, two further attempts were made. If the third call was answered and the named individual was absent, another adult present was asked to answer the questionnaire.

Written questionnaires, including a time activity diary, were distributed to undergraduates from University College Northampton, resident in term time in the study area. Students were asked to report their activities over a single 24-hour weekday, including the time spent in all main rooms at home. Time spent indoors at home was recoded into two categories: living rooms (e.g. kitchen, lounge) and bedroom (including bathroom). In total, 992 questionnaires were sent out via the teaching staff during February 1998, and 247 (25%) were returned and analysed. Of the 247 analysed, 153 (15%) were deemed usable.

The third sample comprised schoolchildren, between the ages of 9 to 13 years, attending Middle Schools in the study area. These were surveyed during October and November 1998. In order to reduce recall bias, and to maximise completeness of the data, children were asked to fill in the activity diary on three occasions during the day - twice at school, supervised by the class teacher, and once in the evening, supervised by a parent. Again, time spent at home was classified by room and subsequently recoded to living rooms and bedrooms.

1,037 questionnaires were initially distributed, but only those from children living within the study area (n = 700) were used. Of these, 447 (64%) provided sufficiently complete responses to be used for the final analysis.

A summary of the data collected can be seen in *Table 4.1*. These data were used firstly to provide simple weighting of time spent in the home and at school/work as a basis for assessing population-based exposures. The data for school children were also used to define detailed time activity patterns (including travel modes and routes) for a sub-sample of individuals, for whom detailed exposure modelling could be undertaken.

Table 4.1: Summary of Time activity data

Target group	Methodology	No. surveyed	No. of usable responses (%)
School children	24 hour time activity diary completed by child (at school and home)	1037	447 (43)
College students	24 hour time activity diary completed by student (distributed through lecture class)	992	153 (15)
Adult residents	Telephone survey	325	286 (88)

Time activity data from Kingsthorpe, a Northamptonshire-wide study (NRPB, 1992), and a UK study (Francis, 1986), based on information collated by the BBC, are summarised in *Table 4.2*. Time at home makes up the single largest portion of the day. In Kingsthorpe, for example, this accounts for an average of 16 hours per day (66.7%) for the combined study group. Variations occur between the three groups studied in Kingsthorpe, however, with averages of only 14.8 hours (61.7%) for students (though the low response rate for this group should be noted), 14.9 hours (62.1% for schoolchildren) and 17.7 hours (73.8%) for adult residents. Broadly similar occupancy times are shown by the county-wide survey (overall mean = 16.2 hours) though, notably, males have slightly higher home occupancy times (17.0 hours) than females (15.7 hours).

Table 4.2: Time spent at home, other locations and travelling (hours): results from the Northamptonshire and Kingsthorpe surveys compared with those reported by Francis (1986)

Study area	Survey methodology	Population group	Home		Other		Travel	
			Mean	St. Dev	Mean	St. Dev	Mean	St. Dev
Northants	Questionnaire	All	16.2	4.3	-	-	-	-
Northants	Questionnaire	Females	15.7	3.5	-	-	-	-
Northants	Questionnaire	Males	17.0	4.9	-	-	-	-
Kingsthorpe	Telephone/ time activity diary	All	16.0	3.8	6.7	3.3	1.1	1.1
Kingsthorpe	Telephone	Adults	17.7	4.6	5.1	4.2	1.2	1.6
Kingsthorpe	Time activity diary	Students	14.8	3.7	7.5	3.2	1.3	0.8
Kingsthorpe	Time activity diary	School-children	14.9	1.8	7.8	1.3	1.0	0.6
UK	BBC survey (1976)	All	18.5	-	4.6	-	1.0	-
UK	BBC survey (1976)	Females	21.1	-	2.4	-	0.5	-

Notes: - = data not reported

Home occupancy times of both males and females surveyed across Northamptonshire, for example, are strongly polymodal, with a double peak for occupancy times of 12 and 14 hours, and a second double peak for occupancy times of 18 and 20 hours. Study groups surveyed in Kingsthorpe show less marked variation: for students, home occupancy peaks at 13 and 15 hours/day and for schoolchildren at 15 hours per day, while for adult residents there is a strong peak at 16 hours per day and a series of secondary peaks between 21 and 24 hours per day.

Data on other activities are available from the Kingsthorpe survey only. These show that time at other locations (e.g. work, school, college, leisure) accounts for about 6.7 hours per day (27.9%) on average across the study group, schoolchildren (7.8 $\pm$1.3 hours) and students (7.5 $\pm$3.2 hours) having higher occupancy times at other locations compared to the adult residents (5.1$\pm$4.2

hours). Travel time makes up only a relatively small proportion of the average day for all groups (1.0-1.3 hours).

4.3.2 Portable monitoring equipment

For personal and microenvironmental monitoring, two OSIRIS units were acquired from Turnkey Instruments Ltd. The OSIRIS is a light-weight, portable monitor for measuring airborne particles. A technical description of how OSIRIS works is given in *Appendix 4D.* Co-location of OSIRIS and TEOM units was undertaken to study the difference in concentrations between the different monitoring equipment. Details of this procedure and results are shown in *Appendix 4E.* A strong correlation can be seen between the two instruments, so a factor of 1.03 resulting from linear regression analysis was used to adjust all data from OSIRIS.

4.3.3 Indoor versus outdoor concentrations

There is evidence to show that indoor air is a function of the air outdoors, albeit when there are no significant indoor sources (Jones *et al.*, 2000; Kopenen *et al.*, 2001; Kingham *et al.*, 2000). The relationship between indoor and outdoor air depends on the conditions that allow the transfer of air from outdoors to indoors, and the magnitude of indoor sources relative to those found outdoors. Kingham *et al.* (2000) showed that homes (without indoor sources) close to road traffic have marginally higher indoor-outdoor (I/O) ratios than homes in background locations, but most of the variation in I/O ratios is associated with indoor conditions. The exchange of air between indoors and outdoors is dependent on a number of factors. Air can enter the indoor environment through ventilation points, cracks and air pockets in the building, or through open doors and windows. Where the air exchange rate is known, indoor concentrations can be calculated from outdoor concentrations (Morawska *et al.*, 2001).

Research has shown that indoor levels of PM_{10} are generally lower than those found in the immediate vicinity outdoors. *Table 4.3* shows mean I/O ratios from a number of recent studies. These results, however, represent situations without the presence of significant indoor sources. In a review of indoor / outdoor studies,

Table 4.3: Summary of studies on indoor-outdoor particle concentrations

Author	***Year***	***Study Area***	***No. of locations***	***No. of days***	***Sampling period (hours)***	***Environment***	***Mean I/O Ratio***
Fischer *et al.*	2000	Amsterdam, Nethelands	29	19	24	Home	0.61-0.86
Funasaka *et al.*	2000	Osaka, Japan	10	40	48-72	Home	0.62-0.96
Giess	1998	Wolverhampton, UK	4	10	1	Work	0.45
Janssen *et al.*	1997	Amsterdam / Vagningen, Netherlands	45	7	8-24	School	0.46-0.75
Jones *et al.*	2000	Birmingham, UK	7	5	0.5	Home	0.6
Kingham *et al.*	2000	Huddersfield, UK	27	28	24	Home	0.81
Monn *et al.*	1995	Zurich, Switzerland	2	7	48-72	Home	0.8
	1997		17	12-30	48-72	Home	0.70
Morawska *et al.*	2001	Brisbane, Australia	16	1	1	Home	0.78-1.07
Ozkaynak	1996	Riverside, CA, USA	?	?	?	Home	0.81

Wallace (1996) concluded that if there were no indoor sources then the ratio would be between 0.4 and 0.6. Indeed, Janssen *et al.* (1997) reported very similar results (*Table 4.3*) from a study in schools that were said to have no significant indoor sources. The range of values taken from studies on indoor PM_{10} (*Table 4.3*) shows that, in general, I/O ratios intersect the upper part of this range, with several studies reporting higher ratios. Studies in homes report I/O ratios between 0.6 and 1.07, which indicates that some indoor sources are present. Elevated concentrations in the home are dependent on the presence of indoor sources such as gas cooking, tobacco smoke, movement of people and type of furnishings. The highest levels of indoor particle concentrations have been associated with smoking in the home (Spengler *et al.*, 1981; Wallace, 1996; Ozkaynak *et al.*, 1996), although levels of the same magnitude have been seen from gas cooking (Jones *et al.*, 2000; Ozkaynak *et al.*, 1996). Where significant indoor sources are present, there is weak correlation between indoor and outdoor concentrations (Jones *et al.*, 2000).

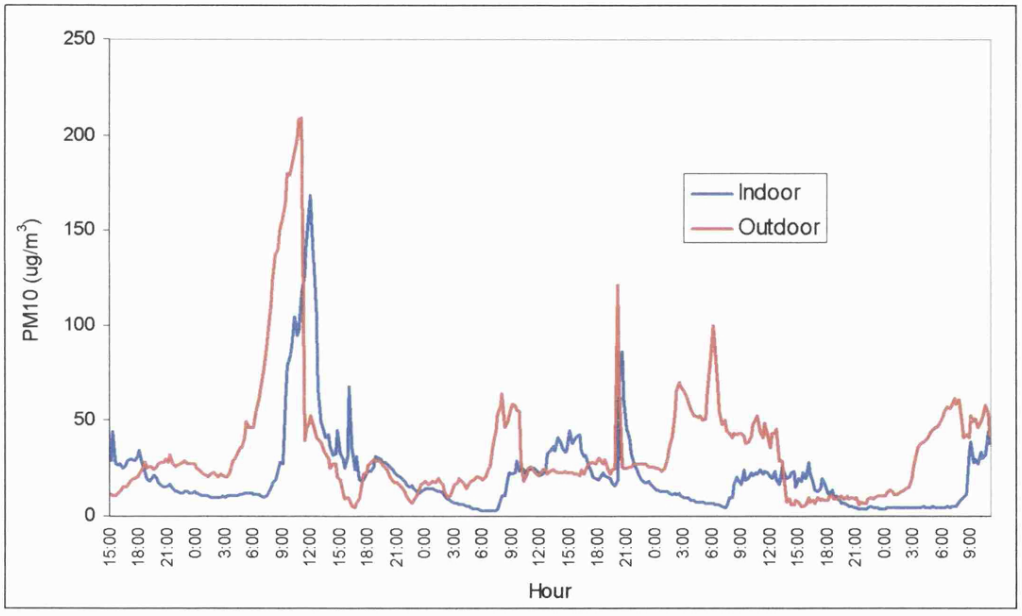
In a study of Birmingham homes, with the aid of time activity diaries, Jones *et al.* (2000) showed that markedly higher concentrations were recorded during periods of cooking (independent of fuel type) and housework, by a factor between 3 and 10 and by a factor of 2, respectively. In the same study, movement of people within the home did not lead to higher concentrations, except for those involved in housework. Indoor levels exceeded those recorded outdoors during all periods of activity attributed to indoor sources. Similarly, Monn *et al.*, (1997) reported an I/O ratio of 0.7 for homes without indoor sources, an I/O ratio of about 1.2 for homes with gas cooking, and 1.9 for homes with smoking. Clearly, therefore, the presence of indoor sources is critical in making estimates of indoor exposure.

The majority of I/O ratios are based on an integrated measure of indoor concentrations over periods typically lasting one to two days. To investigate I/O ratios at higher temporal resolution in this research, four days of simultaneous indoor-outdoor monitoring was undertaken at UCN, using OSIRIS, from 02/10/99 to 06/10/99. One OSIRIS was deployed in an office at the Nene Centre for Research (NCR), away from the movement of people, and the other OSIRIS was mounted to a lamppost, at a height of 2m above ground, about 10m from the front

of the building. Both units recorded at 15-minute intervals. The office was occupied by up to ten people at any time and the NCR building would typically accommodate thirty people at any time. Windows remained closed in the office during this period, but there was no control through the rest of the building.

A profile of the monitoring can be seen in *Figure 4.4*. The I/O ratio for the whole period is 0.6. The most notable feature of these data are periods of high outdoor concentrations followed, with a lag, by periods of high indoor concentrations. Analysis of this time-series shows moderate correlation between indoor and outdoor particles (r=0.37).

Figure 4.4: Indoor-outdoor monitoring at NCR (UCN) from 02/10/99 to 06/10/99



These findings are consistent with those of Morawska *et al.* (2001) who reported weak correlation between indoor and outdoor concentrations using 15-minute and hourly data. Morawska *et al.* (2001) did, however, demonstrate that the correlation between indoor and outdoor concentrations improved if a delay was introduced to the indoor concentration based on a function of the ventilation rate. It was not possible to apply this method to the data collected in this study because exchange rates between indoor and outdoor air at the NCR were not known. Nevertheless,

further investigation of the relationship between indoor and outdoor concentrations was undertaken by varying the lag in the outdoor concentrations by increments of 15-minute intervals up to a lag of one hour. This, however, merely resulted in increasingly poor correlation between indoor and outdoor concentrations with each lag increment, thus supporting the theory that the lag is dependent on ventilation rates and not a constant. The poor correlation between indoor and outdoor concentrations may also be partly dependent on activity patterns inside the building. Other studies have shown much better correlation between indoor and outdoor air but have used much longer averaging periods (e.g. 24 hours).

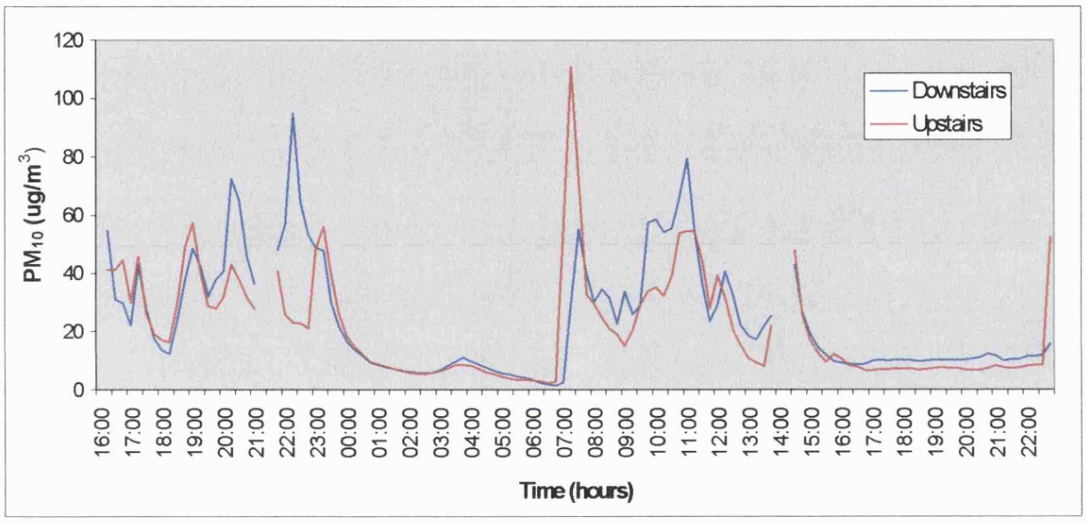
It was not within the scope of this study to develop a model to predict indoor-outdoor concentrations, for this requires detailed, heavily-resourced monitoring of different types of building, occupancy patterns, indoor sources, and ventilation rates. Therefore, the I/O ratio of 0.6 from the monitoring campaign was used in all ground-floor locations (e.g. living room, kitchen), which corresponds with the average ratio from the studies shown in *Table 4.3*.

4.3.4 Vertical profile of PM_{10} concentrations

To investigate the relationship between PM_{10} concentrations in upstairs and downstairs locations, monitoring was undertaken inside a home located in the study area using OSIRIS. A modern semi-detached house was chosen with an 'open-plan' design on the ground floor, with only the kitchen in a separate room. The property has three upstairs rooms and a bathroom. The property was selected for convenience as it was occupied by one of the members of the NAPS study team. The home is situated in a relatively low-trafficked area in the northeast of the study area. One monitor was placed downstairs in the main living room area and the other was placed upstairs in a room functioning as a study. Monitoring took place in each room, simultaneously, away from areas directly affected by the movement of people, from 1600 hours to 0800 hours over a two-day period between 3rd and 5th April 2001, at 15-minute intervals. A record was kept of the number of people occupying each room during each 15-minute interval. The monitors were unavailable for two 30-minute periods during this time whilst data was being downloaded.

The average PM_{10} concentration over the monitoring period was 24.01 ug/m^3 downstairs and 21.01 ug/m3 in the upstairs room [n= 1005 15-minute periods]. This gives a mean downstairs to upstairs ratio of 1.23. A strong, linear correlation [r= 0.84, p<0.001] was found between PM_{10} concentrations in the upstairs and downstairs rooms. As *Figure 4.5* shows, levels of PM_{10} are significantly higher during the evening and in the morning (when there is movement in the house) than during the night.

Figure 4.5: PM_{10} ($\mu g/m^3$) concentrations in upstairs and downstairs locations within a Northampton home



During the night, when there was no movement in the house, the average concentration over the monitoring period was 9.92 ug/m^3 downstairs and 8.13 ug/m3 upstairs [n= 58], giving a ratio of 1.22 between upstairs and downstairs concentrations.

For all other periods, the average PM_{10} concentration was 37.62 $\mu g/m^3$ downstairs and 30.34 $\mu g/m^3$ upstairs, which gives an average ratio of 1.24 between upstairs and downstairs concentrations [n=61]. Thus, there is very little difference in the ratio of concentrations between upstairs and downstairs rooms between night-time and day-time periods.

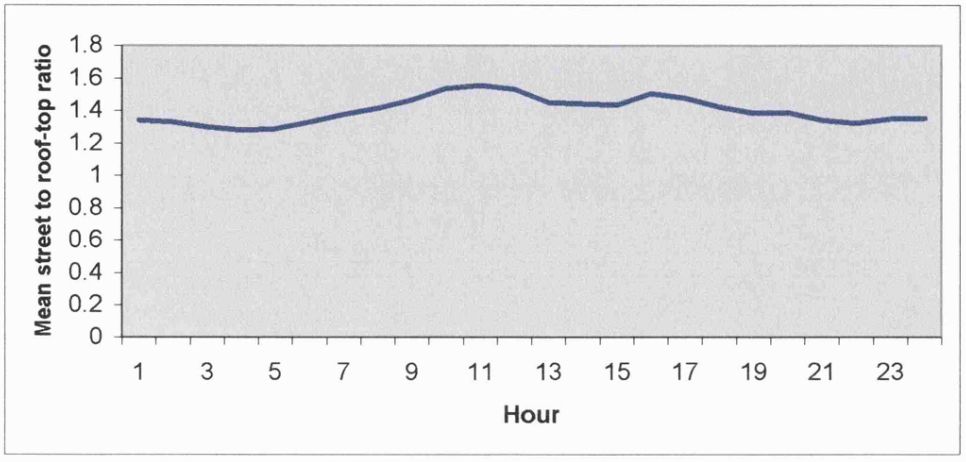
The only potentially significant indoor sources of PM_{10} during this time were cooking and the movement of people. Two periods of cooking took place - one each evening – but they did not have an apparent effect on the levels of PM_{10} recorded in either downstairs or upstairs locations. The level of occupancy in each location did, however, have a significant effect on the level of PM_{10} concentrations.

Raised levels of PM_{10} concentrations tended to coincide with periods when monitoring rooms were occupied. During other periods when neither room was occupied, the ratio of PM_{10} concentrations between downstairs and upstairs monitors was 1.22 [n= 68]. During periods when the downstairs room had at least one occupant but the upstairs room was vacant, the average ratio of PM_{10} concentration was 1.43 [n=38]. Conversely, when the upstairs room was occupied and the downstairs room was vacant, the average ratio of PM_{10} concentrations between the two rooms was 0.76 [n=13]. There were no periods when both rooms were occupied at the same time. These results show that there is an effect on the ratio of upstairs to downstairs concentrations when only one of the monitoring rooms is occupied. However, levels of occupancy were not recorded in the time-activity survey, so factors of 1.24 and 1.22 were applied to day-time (0600 to 0800 hours and 1600 to 0000 hours) and night-time concentrations downstairs, respectively, (0000 to 0600 hours), to give a concentration for upstairs locations.

A second data set was utilised to investigate diurnal variations in the vertical profile of PM_{10} concentrations. As part of an air pollution monitoring campaign at Gower Place, London, on behalf of the Wellcome Trust, monitoring of PM_{10} using OSIRIS was carried out for a period of six weeks (September 28th - November 11th 1999) at two sites at Gower Place: one at street-level and one at roof-level. The average PM_{10} concentration over the monitoring period is seen to be 23.8 $\mu g/m^3$ at street level and 17.0 $\mu g/m^3$ at rooftop [n= 1005 hours]. This gives a street-level to rooftop ratio of 1.40. To investigate diurnal variation between street-level and rooftop concentrations, the average ratio was determined for each hour of the day. The minimum ratio of 1.28 occurred at 0300 hrs and the maximum ratio of 1.56 occurred at 1000 hrs. As shown in *Figure 4.6*, the ratio of street-level to rooftop concentrations is higher during the day than at night. There is, however, some variation in the ratio during the day, with peaks in the morning and early

evening. These variations are, however, considered to be small and the ratio is reasonably consistent across the day.

Figure 4.6: Average ratio of street-level to roof-top concentrations by hour of the day, at Gower Place, London, 28th September to 11th November 1999



The difference in height between street-level and rooftop was estimated to be about 25m. Assuming that the level of PM_{10} concentrations decline linearly with height, the ratio of PM_{10} concentrations between street-level and the second floor (equivalent to upstairs in a two storey house) is 1.1 by extrapolation.

These results differ from those obtained in similar studies of the vertical profile of particles. Väkevä *et al.* (1999), for example, found a rooftop to street level ratio of 5:1 for particles with an aerodynamic diameter less than 0.3 μm. In contrast, Morawska *et al.* (1999) found no overall difference in PM_{10} concentration (range: 0.012 - 0.63 $\mu g/m^3$) levels in respect to height. The differences in results between these studies is possibly due to a number of interrelated factors, including particle size and composition, building design and building density - which may affect the airflow and recirculation of particles - local traffic conditions, and weather conditions. These results suggest therefore that models of the vertical profile of particles require local calibration.

4.3.5 Personal monitoring during journeys: car versus walking

The aim of personal monitoring during journeys was twofold. The first aim was to establish the relationship between measurements of PM_{10} taken simultaneously during walking and in-car journeys. A strong correlation between pairs of walk and in-car measurements would allow a factor to be used to adjust the estimated concentration at the roadside, from network modelling, to an in-car concentration. The second purpose of personal monitoring during journeys was to validate model predictions of PM_{10} made by the network modelling routine.

a) Personal monitoring routes

Two OSIRIS units were used in personal monitoring. A picture of OSIRIS during personal monitoring is shown below (*Figure 4.7*).

Figure 4.7: OSIRIS being used during a personal monitoring campaign

Initially, one route was selected to develop and test simultaneous personal monitoring of walk and in-car exposures using OSIRIS. The route covers part of the main thoroughfare in the centre of Kingsthorpe and crosses the junction of the A508/A5199. The first two-thirds of the journey is in a heavily trafficked area, with the last third along a side road leading to Bective Middle School. The following

method was piloted in July 1999 and applied in monitoring campaigns during November 1999 and March 2000.

Each campaign consisted of two iterations of the same route both in the morning (0800-0900 hrs) and afternoon (1500-1600 hrs) to coincide with the school-run. Each walk and car trip commenced at the same time, departing at 0810 and 0830 in the morning and 1510 and 1530 in the afternoon. The journeys were timed to span the period with the highest frequency of use for the school-run, within each hour. On average, the walk took 12 minutes and the car 4 minutes; thus the car usually arrived ahead of the walk. The car (with walker as passenger) then returned to the start of the route for a repeat journey. This yielded four samples of the route (two in either direction) for each mode, per day.

A second route was chosen to provide contrasting traffic and meteorological conditions. A single campaign took place during April 2000. This covered suburban roads in the north-east of the study area, which are, in general, less heavily trafficked than the route through the centre of Kingsthorpe. However, a section of the route (on Boughton Green Road) would often experience congestion and queuing during the morning school-run. The operation of monitoring on this route was somewhat different from the first route. This route was circular, so the walk would return to the start point without requiring assistance from the car, with the walking taking about 20 minutes and the car about 10 minutes, for each iteration. Two iterations of the route were completed within each hour and for each mode. *Figure 4.8* shows the routes used in personal monitoring campaigns.

b) *Personal monitoring campaigns*

Personal monitoring was undertaken for a total of 38 hours over the three monitoring campaigns. A single pair of measurements was lost – due to operational problems with the monitoring equipment - during the morning campaign on November 18th; thus, 73 pairs of simultaneous in-car and walk measurements were obtained in total. *Table 4.4* shows a summary of the personal monitoring campaigns, including the number of measurements recorded by route, season, and time of day.

Figure 4.8
Personal monitoring routes

Table 4.4: Summary of personal monitoring campaigns

Season	*Route*	*Number of measurements by hour of day*		
		0800	*1200*	*1500*
November '99	1	9	-	10
March '00	1	14	-	16
April '00	2	8	8	8
Total		31	8	34

4.3.6 Analysis of data from personal monitoring

A method was devised to analyse the difference between each pair of in-car and walk measurements. A 'standardised difference' approach was adopted that expresses the difference between a pair of measurements relative to the magnitude of concentrations for any given pair. The difference is therefore standardised by the average of the in-car and walk measurements. This is defined as follows:

$$D_S = \frac{W_O - C_O}{(W_O + C_O)/2}$$

Where;

D_S = Standardised difference between walk and in-car

W_O = Observed walk

C_O = Observed in-car

This measure was used in a number of tests to compare means and examine the main sources of variation between campaign, monitoring route, and time of day. The independent t-test and ANOVA techniques were used for the two and three way tests, respectively. Monitoring data from 1200 hours, exclusive to route 2, only represents a total of 8 data points; therefore, data from this hour was excluded from this analysis as it was deemed too small a data set for these statistical tests.

Thus, the total number of pairs of in-car and walk measurements in this analysis was 65 [n = 73-8]. *Table 4.5* shows the results of comparisons between walk and in-car concentrations (expressed as D_S) by route, survey, and hour of day. For each test, the null hypothesis is: no difference between the means of each population.

Table 4.5: Comparison of walk against in-car exposures expressed by standardised difference (D_S)

Test	Survey	Route	Hour	N	D_S Mean	D_S St. Dev	t	Sig	df
t-test 1	all	all	8	31	.174	.353	.901	.371	63
			15	34	.250	.334			
	all	1	8	23	.171	.344	.318	.752	47
			15	26	.145	.281			
t-test 2	1	1	all	19	.155	.432	-0.16	.988	47
	2	1		30	.157	.206			
	1	1	all	19	.155	.432	-1.67	.105	33
	3	2		16	.390	.388			
	2	1	all	30	.157	.206	-2.66	.110	44
	3	2		16	.390	.388			
ANOVA	1	all	8	9	.206	.483	.069	.934	30
	2			14	.149	.236			
	3			8	.179	.401			
	1	all	15	10	.110	.401	8.38	.001	33
	2			16	.164	.184			
	3			8	.599	.248			

As *Table 4.5* shows, there is no significant difference between mean D_S in every case except in the ANOVA test between all three surveys and time of day. The main source of variation is between the two monitoring routes during the afternoon monitoring period.

4.3.7 Between-trip variation

Two iterations of each route were completed within each hour, for both walk and in-car monitoring. If only a small part of the total variance in D_S is explained by individual trips, then it would be possible to take the average of both trips to represent a single value for that hour. Variance Components Analysis (VCA) techniques were, therefore, used in SPSS to determine the contribution of each factor to the variance of D_S. In this case, the factors were day, route, survey, hour of day, and whether the trip was the first or second within each hour. *Table 4.6* shows the contribution of each of these factors to main effects, expressed as a percentage of the total variance.

Table 4.6: Variance components from personal monitoring

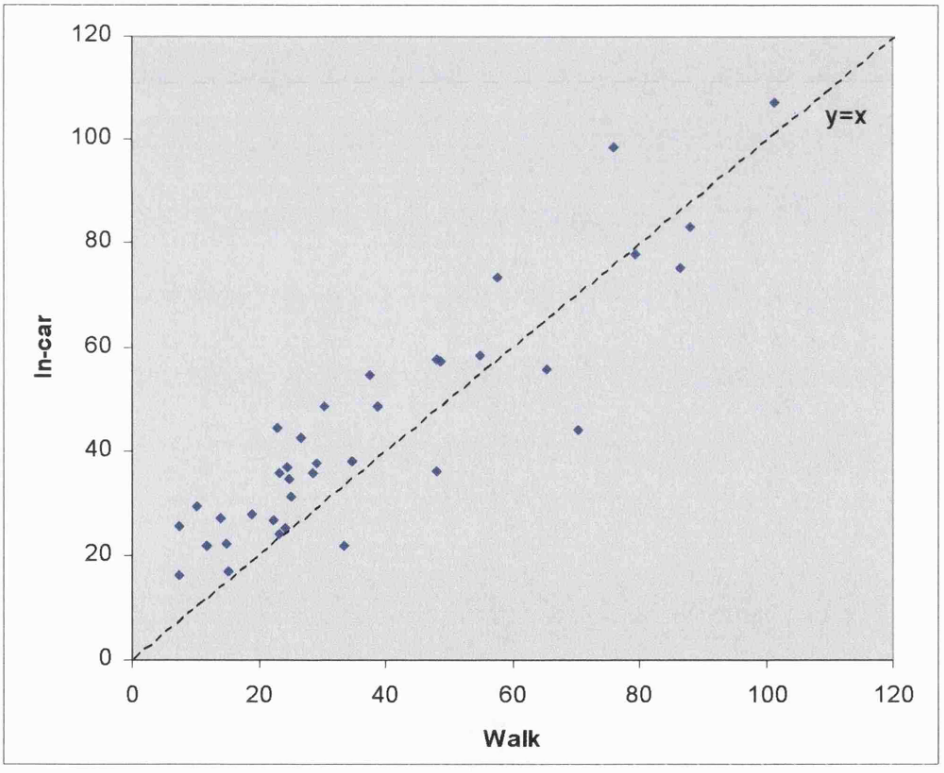
Factor	*Type of factor*	*Sum of squares*	*% contribution*	*Factor level*	*N*
day	random	22728.44	49.2	1-17	72
route	random	12065.01	26.1	1	48
				2	24
survey	random	3699.23	8	1	18
				2	30
				3	24
hour	fixed	705.81	1.53	1	30
				2	8
				3	34
trip	random	379.593	0.82	1	36
				2	36
error		6593.95	14.3		
total		46172.04	100		

These results show that there is a large day (49.2%) and route (26.1%) effect, but only a small effect from trip (0.82%). The effect of hour is, however, only small (1.53%). Error in the variance components is attributed to about half a dozen 2, 3, and 4-way interactions between factors. The main error term is an interaction between day and hour, which contributes 11% to the total variance.

4.3.8 Development of in-car model

Based on the results in *Table 4.6*, it was considered appropriate to combine the two monitoring trips within each hour. Only one pair of measurements was made during the morning session (i.e. 0800 hrs) on 18th November 1999, so this pair was removed (73 –1) from the data pool prior to combining trips. Thus, a total of 72 pairs of in-car and walk observations were reduced to 36 pairs. These 36 measurements were subsequently used to develop a model for predicting in-car concentrations based on the monitored concentration during walking. *Figure 4.9* shows the relationship between the in-car and walk measurements using a single value for each hour [N=36].

Figure 4.9: Personal Monitoring of PM_{10} ($\mu g/m^3$): in-car versus walk



As *Figure 4.9* shows, there is a strong, linear association between monitored concentrations during walking and in-car (r=0.91). Where concentrations during walking are less than 20 $\mu g/m^3$, in-car concentrations are higher than those recorded during walking by about a factor between 1.5 and 2. The magnitude of

difference between in-car and walk concentrations gradually diminishes with increased levels of concentrations. Values for in-car and walk concentrations reach equivalence at 70µg/m^3. *Figure 4.10* shows the relationship between walk and in-car concentrations by monitoring survey: November 1999, March 2000, and April 2000. Regression models for each survey are summarised in *Table 4.7*.

Figure 4.10: Personal monitoring of PM_{10} (µg/m^3) by survey: in-car versus walk

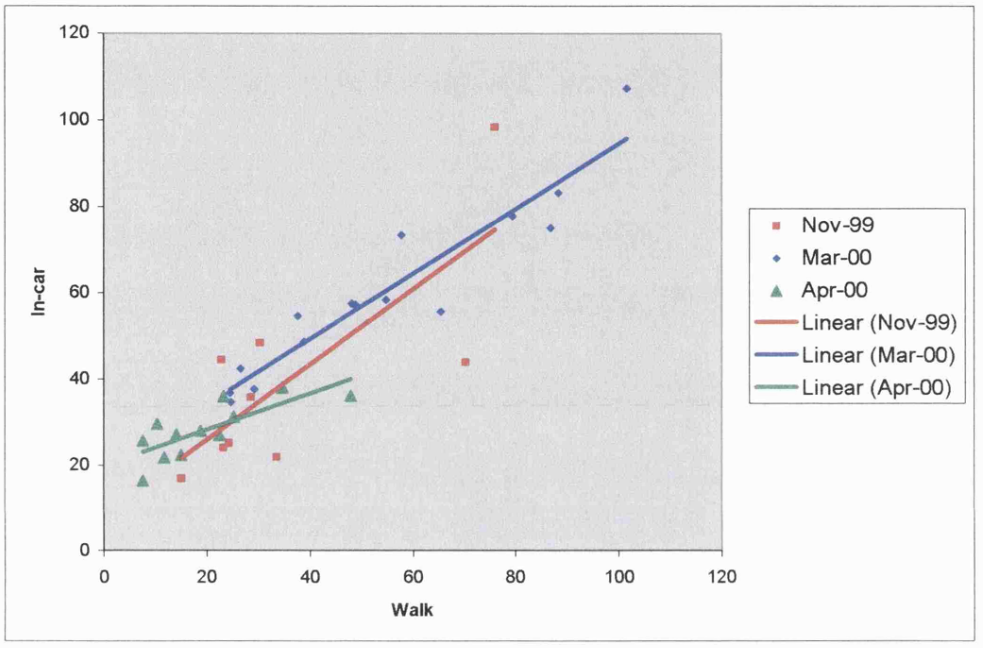


Table 4.7: Modelling to predict in-car concentrations from monitored concentrations during walking, by survey

Survey	Slope	constant	r	r^2	Adj. r^2	SEE	sig.	N	95% CI
Nov '99	.88	8.35	.77	.59	.53	16.92	.019	9	.22, 1.53
Mar '00	.76	19.14	.95	.89	.89	6.83	.000	15	.59, .91
Apr '00	.42	19.86	.78	.61	.57	4.24	.003	12	.18, .66

For individual surveys the slope of the regression equation varies somewhat, but r^2 remains high. The small number of observations mean that all results for individual surveys are sensitive to outliers; for example, a single pair of observations from the April survey (walk = 45 $\mu g/m^3$) has a big effect on the slope of the regression line. The April survey has lower monitored concentrations reflecting the location of the route – a less heavily-trafficked area. Overall, there is no significant difference between the surveys, so the data have been pooled to predict in-car concentrations from monitored concentrations during walking. The resulting model is shown in *Table 4.8.* This gives a ratio between 1.5:1 and 2:1 for in-car versus walking at low concentrations (walk < 20 $\mu g/m^3$), falling to 1:1 at a walking concentration above 80 $\mu g/m^3$.

Table 4.8: Summary of model to predict in-car from monitored walk

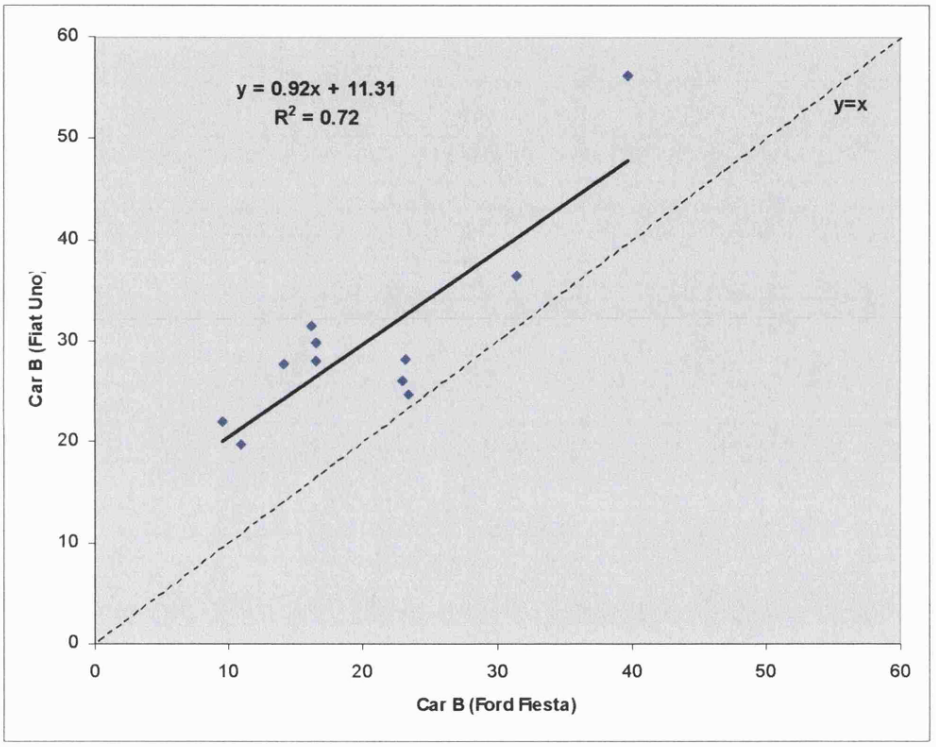
Slope	Constant	r	r^2	Adj. r^2	SEE	Sig.	N
0.82	13.21	0.91	0.82	0.81	9.81	.000	36

4.3.9 Dual-car monitoring

Only one car was used for all personal monitoring campaigns. In order to assess the effects of choice of car on the car-to-walk ratio, a campaign was carried out to compare in-car concentrations in two cars. For this purpose, the original car (Ford Fiesta) was driven in convoy with a second car - Fiat Uno (built 1992) - along a survey route. The Fiat had undergone regular service checks (six-monthly). A route was chosen to give both good coverage of the study area and contrasting traffic levels. In each car, an OSIRIS monitor was placed on the front passenger seat and windows remained closed throughout each journey, and between journeys. On the first journey, Car A was directly in front of Car B, and on the second journey Car B took the lead, and so on. Twelve separate iterations of the same route were undertaken, over two days. A single pair of measurements were lost, due to operating error, giving a total of 11 pairs of in-car measurements.

Figure 4.11 shows a scatter-plot of the dual-car monitoring. Clearly, there is a strong relationship overall between the cars. Concentrations in the Fiat are, however, consistently higher than in the Fiesta, possibly due to variation in the re-circulation of the two vehicles, or because of differences in the compaction and state of cleanliness of the upholstery - which can act as a significant source of re-circulated particles.

Figure 4.11: Dual-car monitoring of PM_{10} ($\mu g/m^3$)



If the model to predict in-car concentrations (*Table 4.8*) is adjusted for the Fiat then the car to walk relationship would be:

$C_C = (0.75 * W_C) + 23.46$

Where;

C_C = in-car concentration

W_C = concentration during walking

This model gives a ratio about 2:1 for in-car versus walking at low concentrations (walk < 20 $\mu g/m^3$), falling to 1:1 at a walking concentration above 95 $\mu g/m^3$. Further monitoring is therefore required to establish a definitive relationship between concentrations during walking and in-car. A strong, linear relationship is nevertheless implied by these results.

4.4 SUMMARY

This chapter has described the methods used to develop and calibrate each component of TOTEM. TOTEM can calculate exposures both indoors and outdoors at fixed-site locations, and during journeys. Indoor concentrations are based on empirical studies of the relationship between indoor and outdoor concentrations. Different factors are used to adjust outdoor concentrations to indoor on the ground floor and upstairs. Exposure during journeys can be calculated for two transport microenvironments: walking and in-car. Outdoor exposure along pathways and roads is modelled by a GIS-based network analysis tool. Exposures during walking are taken directly from the model, whereas in-car exposure is determined from a regression analysis derived by simultaneously monitoring concentrations during walking and in-car journeys. A database of time-activity is available for three population sub-groups (students, adults and children). The activity data can be used on an individual level or to characterise the activity of a population group.

5 VALIDATION AND APPLICATION

5.1 INTRODUCTION

This chapter is split into two distinct sections. The first part describes the performance and validation of the models. The second part demonstrates the capability of the STEMS methodology and applies it to examine the effects of a walk-to-school policy within the study area.

5.2 VALIDATION

5.2.1 Introduction

The following three sections describe the testing and validation of three sub-models used in STEMS: outdoor pollution modelling at fixed-sites, the model used to estimate in-car exposures, and the network modelling used to estimates journey-time exposures. Validation of these sub-models provides the basis for modelling exposure profiles for individuals or groups. It was not possible in this study, however, to test and validate all model elements used in STEMS. The estimates made indoors, for example, could not be validated because this would have involved heavily-resourced and time-consuming monitoring. It was also only possible to validate fixed-site, outdoor monitoring at a single-site within the study area because only one monitoring site was available to give a sufficiently long period of monitored data.

5.2.2 Outdoor pollution modelling at fixed-sites

Validation of the outdoor pollution modelling was performed with monitored data on hourly PM_{10}, CO, and NO_X at a single, fixed-site in Kingsthorpe.

As described in chapter 4, PM_{10} is modelled using the following equation:

$$PM_{10} = [A] + [B+C]$$

Where;

A = primary

B = secondary

C = other

The model was developed by regressing local modelled PM_{10} from ADMS (A), and rural sulphate (B+C), to predict hourly PM_{10} at the fixed-site in Kingsthorpe. As described in *Section 3.8*, the model was built by randomly selecting 70% [n=2651] of the total available monitoring data [n=3908]. Linear regression through the origin was used to allow for PM_{10} levels to be zero.

This gave the following model:

$$PM_{10} = (14.228 * Sulphate) + (1.322 * local\ modelled\ PM_{10})$$

The coefficient for sulphate reflects the dominance of background PM_{10} in this area. The coefficient for the local modelled component is slightly above 1, at 1.322. This is appropriate, as the local component ought to include a small proportion of traffic-related matter from non-combustion sources.

Validation was performed by application of this model to the remaining 30% [n=1257] of the monitored data. The results of the validation are shown in *Table 5.1*.

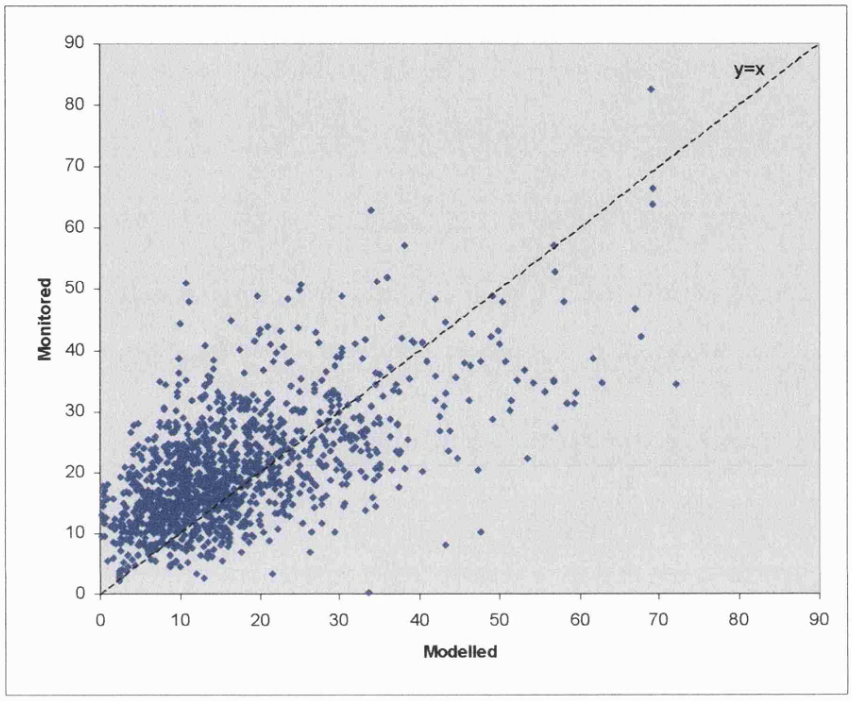
Table 5.1: Performance of the outdoor pollution model for PM_{10}

r	r^2	*Adj.* r^2	*SEE* ($\mu g/m^3$)	*Sig.*	*N*
.60	.36	.36	7.57	.000	1257

This gave a moderate (r^2=0.36), but highly significant model fit ($p < 0.001$). The observed against predicted concentrations can be seen in *Figure 5.1*. As this

shows, the model has a slight tendency to over-predict at higher concentrations, whereas the majority of low and mid-range concentrations show an under-prediction.

Figure 5.1: Application of PM_{10} model to 30% of monitored data from the fixed-site



As markers for model performance, and further testing of ADMS, CO and NO_X were also modelled at the Kingsthorpe site, for the same period as PM_{10}. At most locations, CO is almost entirely attributable to road traffic, and, as stated in *Section 3.3.2*, NO_X can be taken as wholly from road traffic at busy, roadside locations. For these pollutants, a greater proportion of the total comes from local sources, but there is still a regional contribution for both pollutants, which can be represented by background (i.e. rural) monitoring sites. For CO, there were no nearby rural sites, so an alternative for the regional component was sought. The report on Airborne Particulate Matter in the UK (QUARG, 1996) suggests that there is a fairly stable, continuous background for CO in the North Atlantic of about 0.15ppm. This value was therefore added as a constant to the modelled CO. For NO_X, the

average daily value from the nearest rural site (Wicken Fen, Cambridgeshire) was added to the modelled NO_X from ADMS.

The results of modelling CO and NO_X are shown alongside those for PM_{10} in *Table 5.2.* This shows that the performance of the model for PM_{10} (r^2=0.36) is reasonably consistent with that for CO (r^2 = 0.41) and NO_X (r^2=0.43). For PM_{10}, 74% of modelled values lie within a factor of 2. This result compares with the best of the results from other studies. Modelled CO and NO_X gave Fa_2 values of 75% and 71%, respectively, which also rank highly amongst other results. For PM_{10}, 50% are within a factor of 1.5 (i.e. $Fa_{1.5}$). This measure of performance is rarely used in reporting of this type, but is useful because it is a much more stringent test of model reliability. A result of this magnitude for $Fa_{1.5}$ should be considered encouraging.

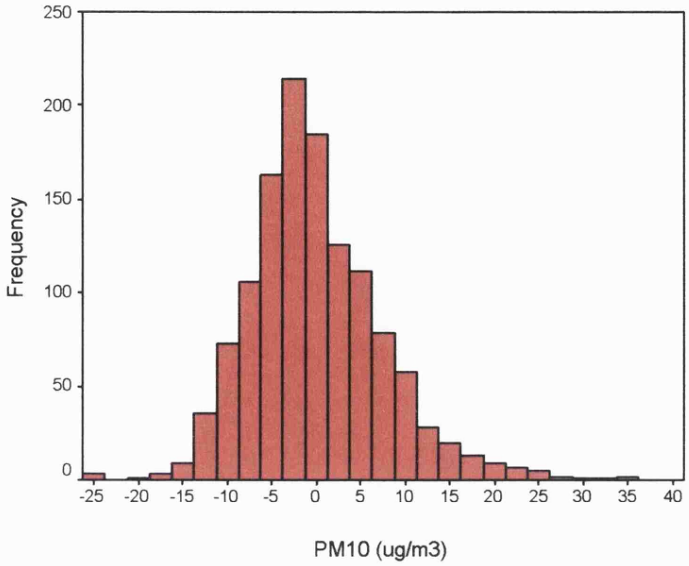
Table 5.2: Comparison of performance measures for different pollutants

	PM_{10}	NO_X	CO
r	0.60	0.66	0.64
r^2	0.36	0.43	0.41
Fa_2 (%)	74	71	75
FB	-0.21	0.04	-0.43
NMSE	0.26	0.43	0.72

Figure 5.2 shows the frequency distribution of unstandardised residuals. From a total of 1257 hours, 570 (45%) model predictions lie within +/- 5 $\mu g/m^3$ of observed values.

Computation of FB for PM_{10} returned a value of -0.21, which overall indicates a slight under-prediction. CO, also, shows a slight under-prediction (FB = -0.43), while NO_X shows a very slight tendency for over-prediction (FB = 0.04). The somewhat poorer FB for CO can probably be explained by the lack of detailed information on background concentrations. Due to the fixed and moderate value for background concentrations, modelled CO tends greatly to under-predict at high concentrations.

Figure 5.2: Frequency distribution of unstandardised residuals from hourly modelling of PM_{10}



NMSE is 0.26 for PM_{10}, compared with values of 0.43 and 0.72 for NO_X and CO, respectively.

The value of FB for PM_{10} is comparable with results obtained by Kukkonen *et al.* (2001), using OPSM (n.b. uses same dispersion model as ADMS), who obtained FB in the range -0.28 to 0.18 for NO, NO_2, and O_3, and an average FB of 0.36 from Owen *et al.* (1999) using ADMS. Kukkonen *et al.* (2001) obtained NMSE in the range 0.04 to 0.32 for NO, NO_2, and O_3, whereas the average NMSE from Owen *et al.* (1999) was 1.92. Results from Carruthers *et al.* (2000) quote NMSE of 0.34 for modelling PM_{10} alongside the M4.

The results from this study can thus be considered reasonable compared to the performance of the range of models used in other studies. There are limited data from other studies using ADMS-Urban for comparative purposes, but the results in this study are broadly consistent with those cited in research using the same averaging period. The results for CO and NO_X show that ADMS-Urban is making reasonable estimates because a high proportion of CO and NO_X would be expected to come from local traffic sources, particularly at locations near to busy

roads. On this basis, this model can be considered suitable for general application. Improved results for PM_{10}, and the other pollutants, may be achieved by providing better models of background concentrations.

5.2.3 In-car exposure modelling

Due to the large amount of time, and effort, that was required simultaneously to collect data on concentrations during walking and in-car, no independent data were collected for validation of the in-car model. Rather than using, for example, two of the campaigns to build the model, and the third for independent testing, it was felt that a more robust model would be achieved by using all the monitoring data in model building. This approach allowed samples to be drawn over a greater number of routes, seasons, and times of day, potentially giving a wider range of meteorological conditions and traffic-flow behaviour. Alternative methods were therefore sought for model testing. Resampling methods offer an attractive alternative where it is not possible to collect independent data for model validation (Efron, 1983). This assumes that the range of values in the measured population is representative of the range that could be collected, as part of an ongoing process, from a potentially infinite number of measurements. The 'Resampling add-in' for Microsoft Excel© (Blank *et al.*, 1999) was obtained for this purpose.

The jacknife is a commonly used technique where there is no, or insufficient, data available for model validation. In jacknife, the same test is repeated by leaving one subject out each time until all subjects have been used. A jacknife approach was used to build 36 separate regression equations to predict in-car concentrations by leaving out one pair of walk and in-car measurements in each iteration. The regression equation from each iteration was then used to predict the in-car concentration to correspond with the pair of measurements excluded from the jacknife. The jacknife predictions were then regressed against the monitored in-car concentration from each pair of measurements. Linear regression analysis gave an r^2 of 0.79 (r=0.89), which corresponds well with r^2 of 0.81 from the original in-car model. Results from the jacknife indicate that the in-car model will make reasonable predictions of in-car concentrations from monitored data.

A second resampling method was employed to test the likelihood that the pairs of in-car and walk measurements were obtained by chance. The following question was posed: is there still a relationship between in-car and walk concentrations when randomly pairing any walk concentration with any in-car concentration? To test this, a shuffling method from the 'Resampling add-in' for Microsoft Excel© was used. In each iteration, the walk was fixed whilst randomly shuffling the in-car concentration and assigning each to a new position alongside a walk concentration. Thus, a new pool of walk and in-car pairs was created each time. This exercise was repeated 1000 times, returning a Pearson correlation coefficient for each random population (N = 36). A maximum Pearson r of 0.35 and minimum of – 0.32 were observed, with a mean of zero (0). Given that $r = 0.91$ ($p<0.001$) for the monitored pool, this affirms that the relationship between the monitored walk and in-car measurements is dependent on simultaneous monitoring and not merely a series of events from a randomly distributed population.

5.2.4 Network modelling

a) Overview

The network model, described in *Section 4.2.3*, was used to model each route from the personal monitoring campaigns. Each modelled route was extracted from an hourly (N.B. the highest resolution attainable) pollution map generated by ADMS via ArcView (*Section 4.2.2*). Thirty-six pollution maps were therefore produced to correspond with the 36 simultaneous in-car and walk trips, over two personal monitoring routes.

b) Validation

The bench-mark for model performance is how well the network model performs against the monitored data from the fixed-site in predicting both monitored walking and in-car concentrations. The first step, therefore, was to establish a measure of performance based on the relationship between the fixed site monitor and the monitored walk and in-car concentrations, separately. The start and end times of each journey were intersected with the 15-minute monitoring periods for the fixed-

site. *Figure 5.3a* and *Figure 5.3b* show the performance of the fixed-site monitor in predicting monitored concentrations from walking and in-car, respectively, using least squares linear regression.

As the regression equations and coefficients of determination show, the concentrations recorded at the fixed-site give a good prediction of concentrations monitored during both journeys by walking and in-car. It is, however, notable that the slope coefficient is well above 1 in both cases, so the monitored fixed site tends to under-predict the in-car and walk concentrations at higher levels of PM_{10}, by a factor of about 1.6 in both cases. Using the fixed-site monitor to predict concentrations during walking and in-car gives 67% and 72%, respectively, of predictions within a factor of 2.

The performance of the network model in predicting concentrations both in-car and during walking is shown in *Figures 5.4a* and *5.4b*. For predictions of concentrations during walking, the performance of the model (r^2=0.54) is the same as the fixed-site (r^2=0.54). Using the model to predict concentrations during walking gives 61% of predictions within a factor of 2. The slope of the regression equations is, however, somewhat closer to unity. The main source of error is thus in the intercept in the regression model. At low concentrations, the model tends to under-predict concentrations during walking by about a factor of 2; at high concentrations it under-predicts by a factor of about 1.5.

Although there is some difference in the slopes of the regression lines, these results show that the fixed-site and the model are performing equally in predicting monitored concentrations during walking. In contrast, in terms of r^2, the model (r^2=0.36) does not perform as well as the fixed-site (r^2=0.62) in making predictions of in-car concentrations. However, using the model to predict concentrations during in-car journeys gives 82% of predictions within a factor of 2, whereas using the fixed-site gives 72% within a factor of 2.

In interpreting the performance of these models, it must be borne in mind that the fixed-site is very close to the route, and a distant site would possibly not work as well because it would not be measuring the contribution from the local traffic where

Figure 5.3a: Monitored fixed-site PM_{10} (ug/m^3) concentration versus monitored walk PM_{10} (ug/m^3) concentration

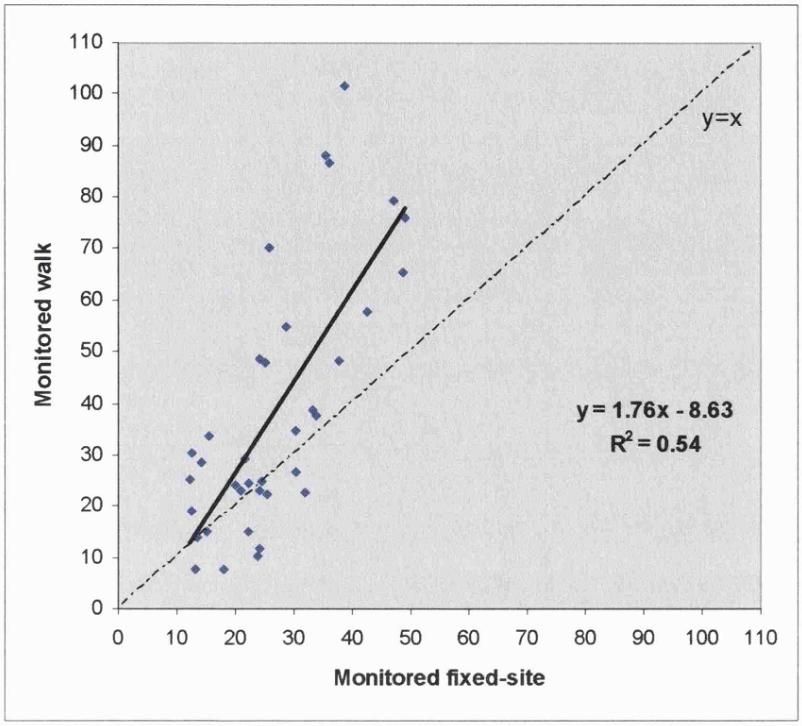


Figure 5.3b: Monitored fixed-site PM_{10} (ug/m^3) concentration versus monitored in-car PM_{10} (ug/m^3) concentration

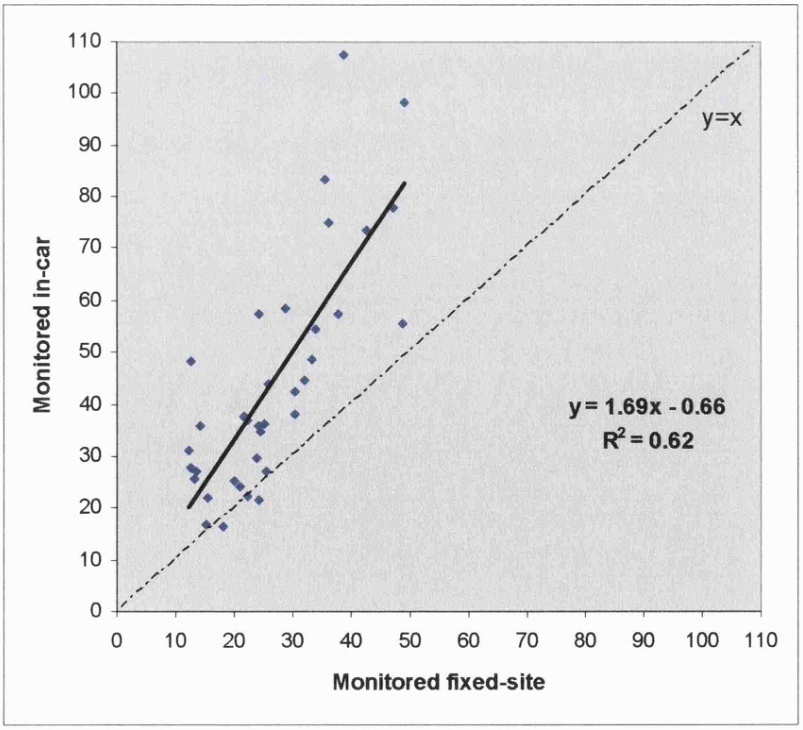


Figure 5.4a: Modelled walk PM_{10} (ug/m^3) concentration versus monitored walk PM_{10} (ug/m^3) concentration

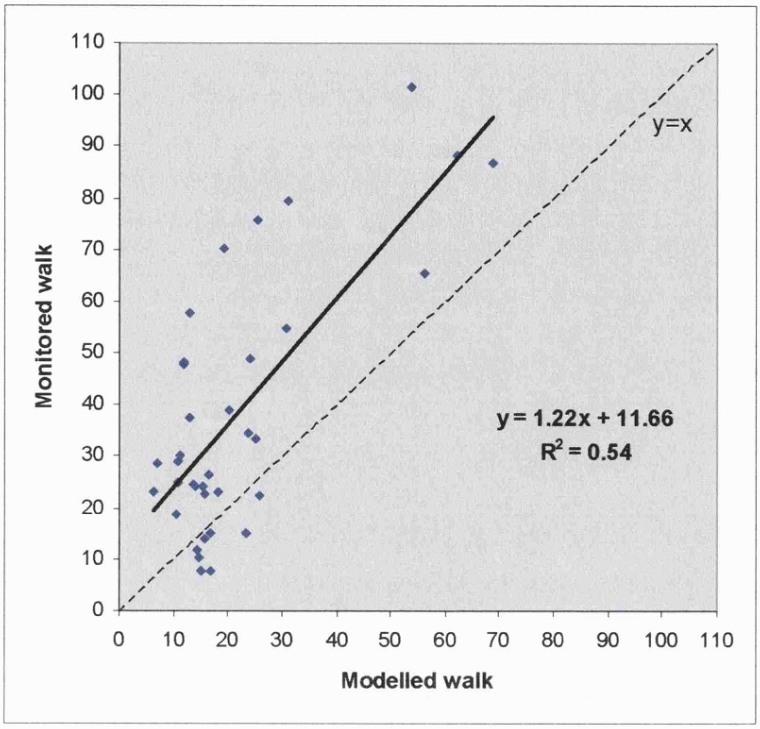
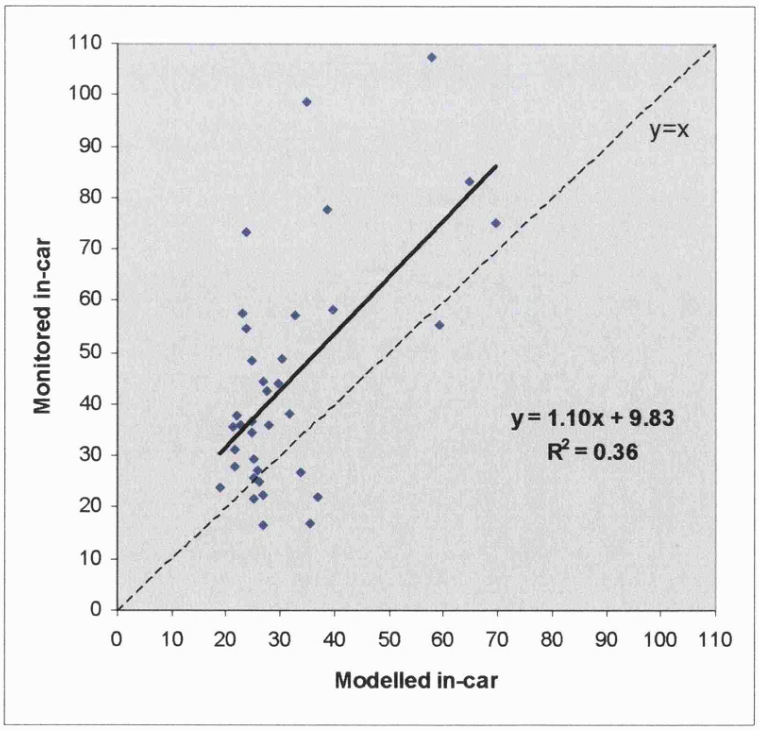


Figure 5.4b Modelled in-car PM_{10} (ug/m^3) concentrations versus monitored in-car PM_{10} (ug/m^3) concentrations



the journey is being made. Unfortunately, there was only one fixed-site monitor available within Northampton during the course of this study so it was not possible to test this theory.

It must be noted that the fixed-site monitor has better temporal resolution than the modelling. Therefore, this analysis was repeated using hourly average PM_{10} concentrations from the fixed-site against the hourly estimates from modelling. This merely resulted in the hourly average and 15-minute concentrations from the fixed-site monitor performing equally in predicting concentrations during walking and in-car.

5.2.5 Summary

Sections 5.1 and 5.2 have shown that a reasonable model of outdoor pollution has been obtained using regression techniques to combine models of background and local PM_{10}. The performance of the model is comparable to results from other studies. The model has been applied to make predictions of concentrations during journeys by walking and in-car. Modelled concentration during walking and the monitored concentration from the fixed-site perform equally in predicting the monitored concentration during walking. The model does not, however, work as well as the fixed-site in predicting in-car concentrations.

Ideally, modelling would have been extended to compare modelled exposures with data collected from continuous personal monitoring over a period of at least one day. In this way it would be possible to validate each component in STEMS separately. This was not within the scope of this study, as it would have involved considerable additional work.

5.3 APPLICATION

5.3.1 Introduction

The STEMS approach provides both new methods for studying the effects of traffic management and policy interventions on human exposure to air pollution. To

demonstrate how the methodology works and its potential for linking traffic planning with epidemiological studies, TOTEM has been applied to two traffic management scenarios:

1) A 'no-change' scenario to assess hourly exposures under current conditions and quantify the contribution to exposures from different micro-environments.

2) A walk-to-school policy to assess the potential health benefits of reducing traffic during the 'school run'.

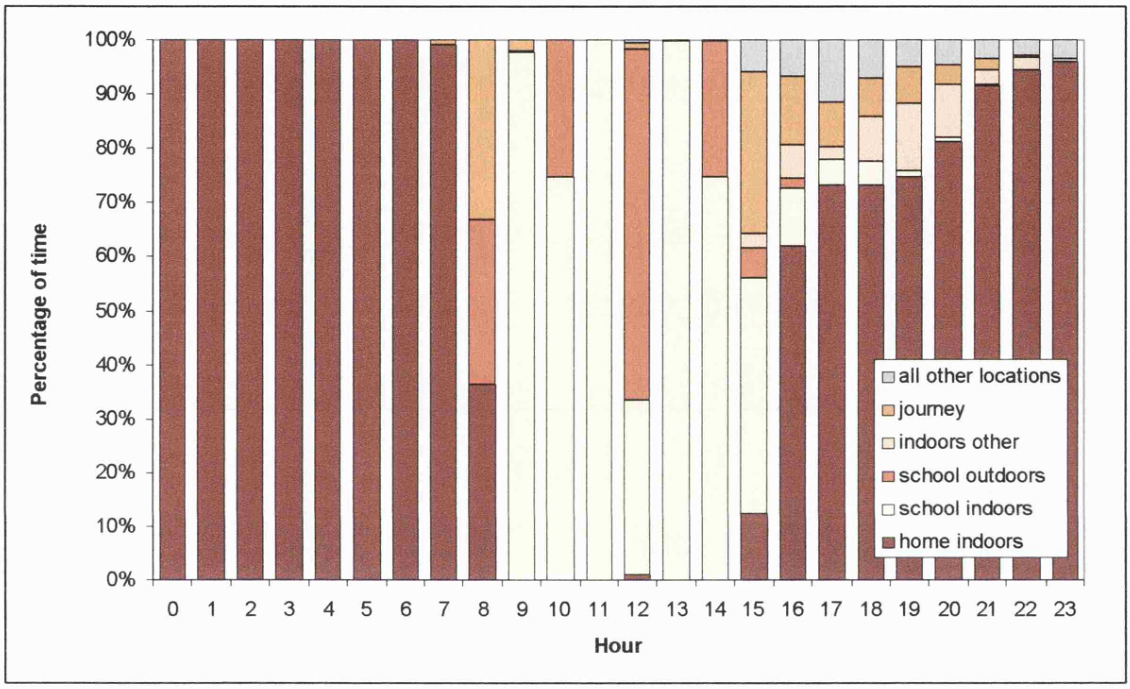
Exposure modelling for both scenarios was conducted for 5 weekdays between 12^{tth} and 18th November 1999. This period was chosen for convenience as it coincided with one of the periods where models had already been prepared for use in testing and validation of personal monitoring. The period is also thought to cover a range of pollution levels typically found during winter in the study area. The mean hourly PM_{10} concentration recorded at the monitoring site in Kingsthorpe between July 1999 and March 2000 was 20.90 ug/m^3, compared with 18.90 ug/m^3 for the exposure modelling period between 12th and 18th November (excluding weekend days). About 25% of monitored values from the modelling period are either below the 10th or above the 90th percentile from the full run of monitored data.

The diurnal pattern of activities for schoolchildren [n = 447] is shown in Figure 5.5. As this shows, the majority of time is split between home and indoors at school. The percentage of time spent by schoolchildren in the home gradually rises throughout the evening, from 60% at 1600 hours to about 80% by 2000 hours, and reaching more than 96% by the end of the day. As expected, the two significant journey periods are between 0800 and 0900 hours and 1500 and 1600 hours. Journey-times make up about 30% of the total time within these hours. During the evening (i.e. 1700 – 0000 hours), time spent making journeys gradually decreases from about 10% at 1700 hours to 3% by 2100 hours.

The first stage in exposure modelling was to prepare the time-activity data for use in the exposure model. Those children who made one or more journeys in the

evening after returning home from school were regarded as 'active', whereas children who did not make any further journeys in the evening were regarded as 'inactive'. It was not possible to model other journeys made outside the school day, as the geographical location of these destinations was not known; therefore, only children from the 'inactive' database were considered in this case study for

Figure 5.5: Activity patterns for schoolchildren



exposure modelling. This provided a population of 92 (21% of total) children, from which 50 were randomly selected for exposure modelling. A total of six micro-environments were identified for modelling exposures: downstairs inside the home, upstairs inside the home, outdoors at home, journeys to and from school, indoors at school, and outdoors at school.

The following time-activity model was applied to the sample. Each child would spend time at home in the morning before leaving for school, travel directly to school by the quickest route, stay at school for the entire duration of the school day, then return home at the end of the school day and remain at the home in the

evening. The waking-up time and amount of time spent in the home before leaving for school was taken from the time-activity database. Time at home in the morning was split equally between upstairs and downstairs in the home. Journey-times were modelled using the network analysis method described in *Section 4.2.3.*

For each child, journeys were modelled both as walking and by car. Separate exposures profiles were constructed for each mode of transport during the 'school-run' hours. For the selected group of children, all journeys to school were made between 8 and 9am, with no modelled journey-times exceeding forty five minutes. The time taken for each journey was used as a basis for proportioning the amount of time spent in other microenvironments during this hour. During time at school, it was assumed that children remained indoors except for breaks and at lunchtime. Two outdoor breaks of 15 minutes were assumed: one in the morning and one in the afternoon. A lunchtime break of one hour was assumed that comprised twenty minutes inside the school when school meals would be served, and forty minutes outdoors at school for play-time. The school day ends at 3:15pm.

The same method of using journey times to proportion the amount of time spent in each microenvironment was applied between 3 and 4pm for the journey home. For the evening period (4pm to midnight), information was available in the time-activity database on the micro-environments occupied by each individual on an hourly basis. Where two, or more, microenvironments were recorded within any hour the time was split equally between each microenvironment. During the night, it was assumed that all children would be upstairs in the bedroom.

5.3.2 Exposure profiles

Exposures were, therefore, modelled for a total of 55 locations, comprising 50 homes and 5 schools. Journey exposures were modelled for 50 routes, twice a day, over 5 days, under two traffic management scenarios; thus, a total of 1000 journey-time exposures were calculated over the 5 days. The routes selected for journeys to school are shown in *Figure 5.6.* For both walkways and roads, these 50 routes represent about 30% of the total length of available routes within the

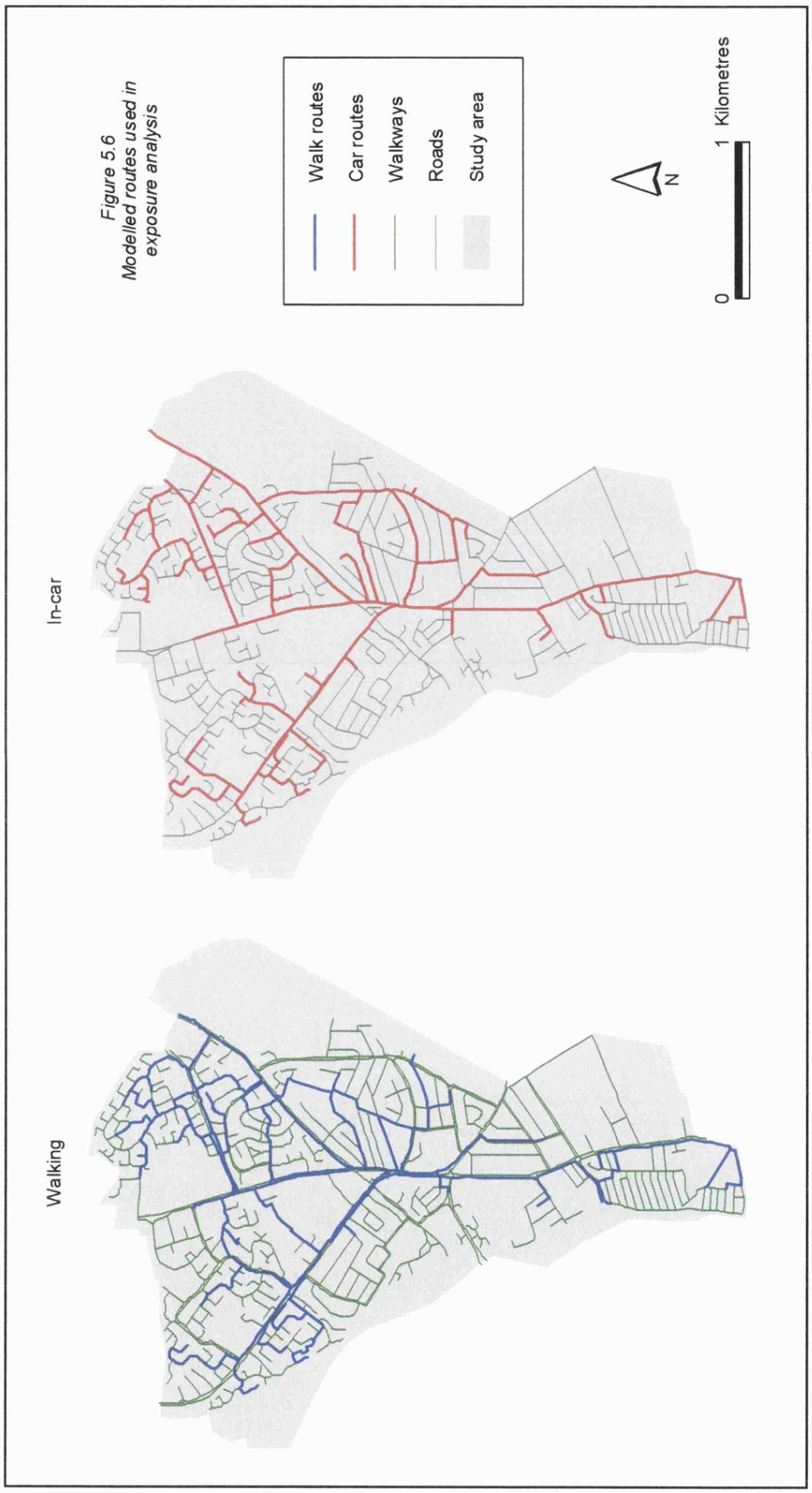


Figure 5.6
Modelled routes used in exposure analysis

study area. Exposure profiles were constructed for each of the 50 individuals at hourly resolution in Microsoft Excel. Indoor exposures were determined by applying the relevant I/O ratio for each microenvironment. In the home, downstairs exposures were converted to an upstairs exposure using the upstairs-downstairs ratio as described in *section 4.3.4.*

Figure 5.7 shows modelled hourly exposure profiles for the 50 schoolchildren, by day, with the journey component modelled as walking. *Figure 5.8* shows modelled hourly exposures for a sample of individual profiles.

The profiles in *Figure 5.7* clearly show that elevated exposures occur during hours when journeys to and from school are being made. In fact, 'school-run' hours provide the highest mean hourly modelled exposures across the whole period. Elevated exposures also occur during some lunchtime periods, largely because a significant amount of time is spent outdoors at school during the lunchtime period, and three of the five schools are in close proximity to busy roads. This effect is particularly marked during the lunchtime periods on 15th and 16th November.

Another significant feature of the exposure profiles is that a number of hours on separate days have a range in exposures in excess of 15 ug/m^3. Strong differential effects on exposure can be seen during either the 'school-run' or the evening when the group of children experience contrasting exposure environments. Logically, the lowest range in exposures occurs during the day when the children are shared between five locations, with the majority of the sample being shared between only two schools. Exposures are generally lowest at night, with the exception of 15th and 17th November which were dominated by a high regional / far-travelled component in the PM_{10} concentration.

Table 5.3 shows a statistical summary of modelled journey-time exposures for both walking and car, over 5 days during November 1999. For the whole period, modelled in-car exposures are, on average, about twice modelled exposures from walking. Journey-times on foot are, of course, considerably greater than those by car. For the sample of schoolchildren, the average modelled journey-time by walk is 15 minutes compared with 3 minutes by car; thus, car journey times are about

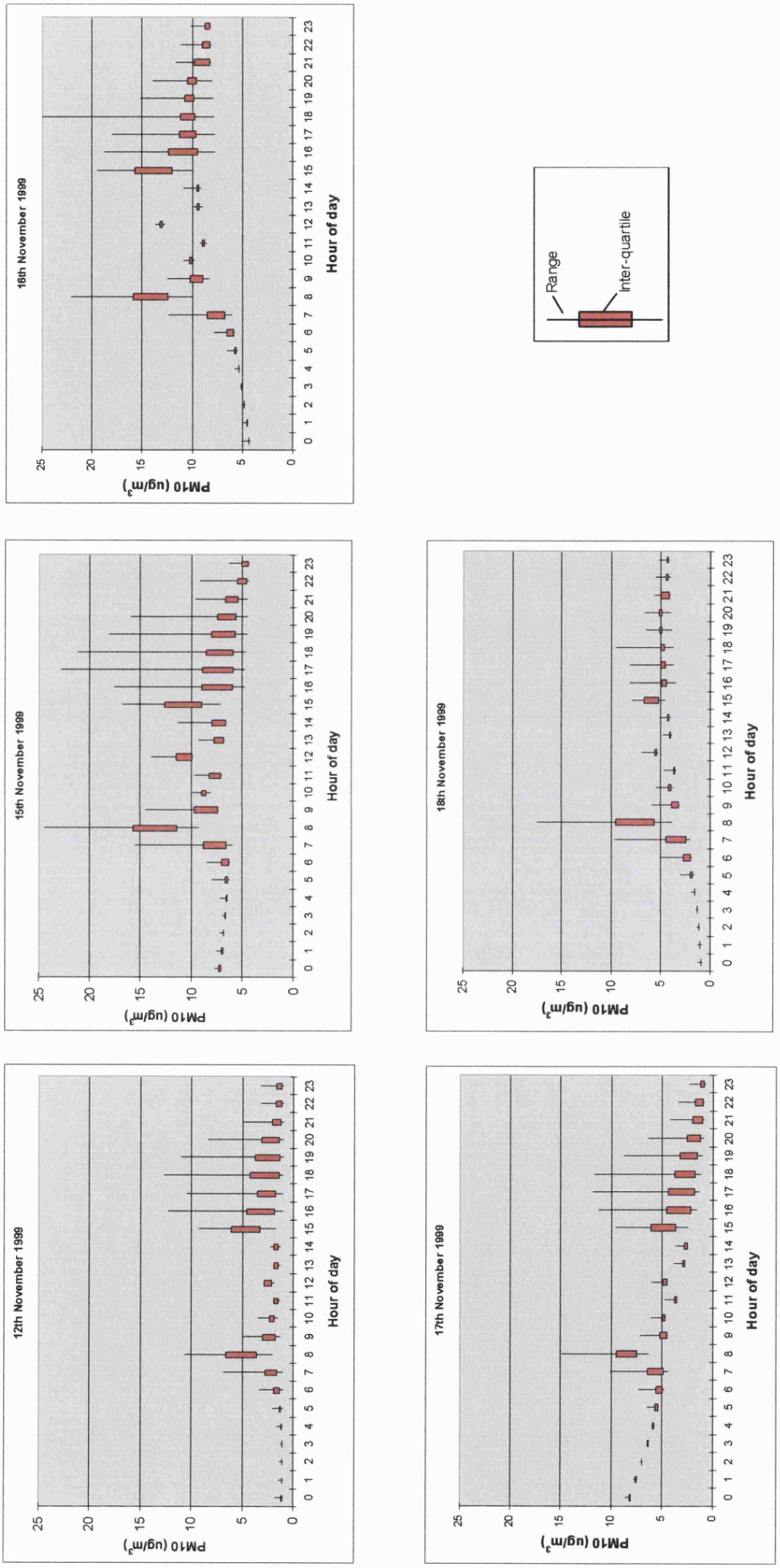


Figure 5.7: Modelled exposure profiles for a population (n=50) of schoolchildren for 5 days during November 1999

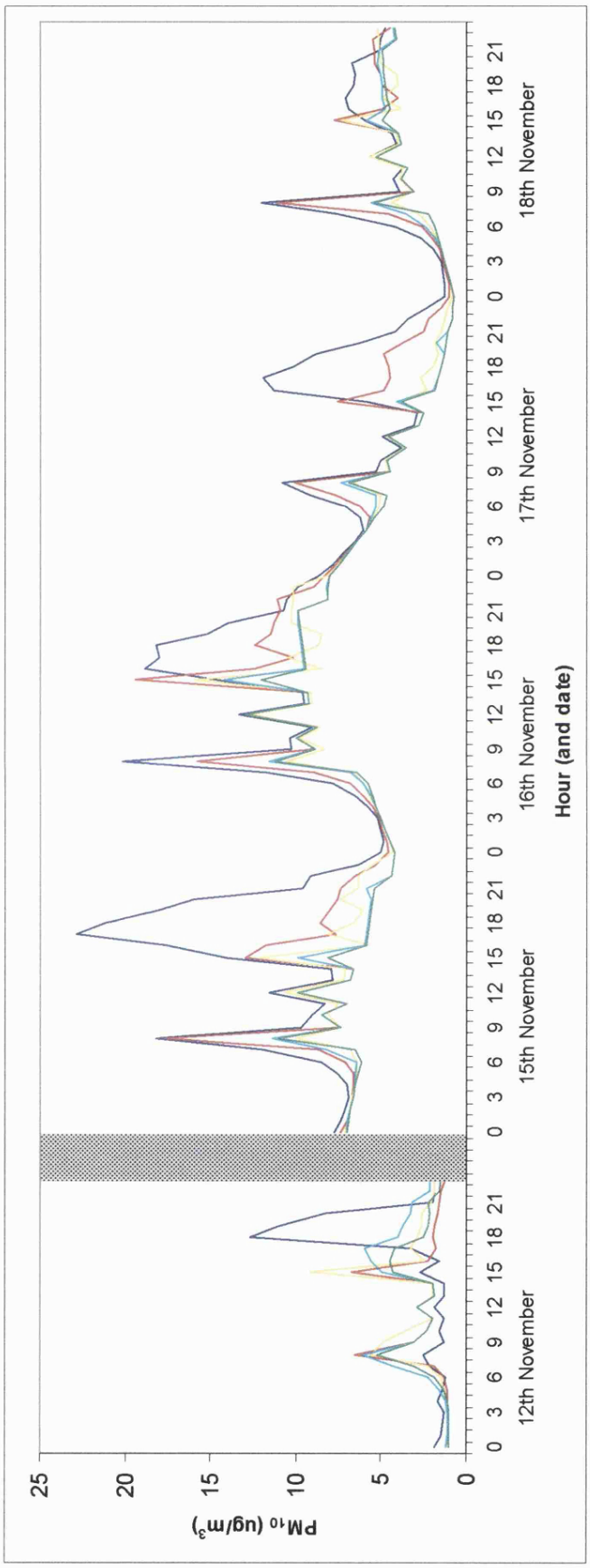


Figure5.8: Selected modelled exposure profiles

five times the length of journey-times by walking. By integrating the journey-time exposure into an hourly average exposure, allowance is made for both travel time and the relative contribution of exposures from other micro-environments (e.g. home, school). As *Table 5.4* shows, in contrast to journey-time exposures, hourly average exposures that include the journey component modelled as walking are consistently higher than average hourly exposures that include the equivalent journey modelled as in-car, by a factor of about 1.1.

Table 5.3: Modelled journey-time exposures by journey mode

		Nov 12		Nov 15		Nov 16		Nov 17		Nov 18	
		0800	1500	0800	1500	0800	1500	0800	1500	0800	1500
Mean	car	20.13	19.85	29.93	25.74	28.97	30.28	22.97	20.38	23.87	20.19
	walk	8.62	8.33	19.87	15.46	19.19	20.57	11.80	8.50	12.48	8.61
Min	Car	16.09	15.99	23.83	22.34	24.46	26.29	20.18	16.48	17.72	18.86
	Walk	3.24	3.00	13.19	11.39	13.95	16.18	8.81	4.22	5.73	7.20
Max	Car	26.03	24.37	42.90	31.34	37.37	37.44	30.33	27.80	36.20	24.23
	Walk	17.40	15.51	36.29	24.40	31.50	30.14	20.45	18.20	28.31	13.85

Table 5.4: Modelled hourly exposures by journey mode

		Nov 12		Nov 15		Nov 16		Nov 17		Nov 18	
		0800	1500	0800	1500	0800	1500	0800	1500	0800	1500
Mean	car	4.85	4.02	12.85	10.25	13.72	12.63	8.40	4.61	7.11	5.70
	walk	5.36	4.64	14.02	11.02	14.51	13.98	8.74	5.05	7.89	5.91
Min	Car	2.15	1.83	7.43	6.81	7.78	9.12	5.18	2.59	3.29	4.43
	Walk	1.98	1.77	9.30	7.24	9.94	10.00	6.34	2.52	3.87	4.59
Max	Car	10.27	9.60	23.26	16.85	21.06	19.51	14.20	8.55	16.78	8.44
	Walk	10.60	9.32	24.49	16.76	22.08	19.48	14.96	9.63	17.49	7.91

5.3.3 Effects of a walk-to-school policy

a) Introduction

The methodology developed in this study has been applied to a walk-to-school policy, aimed at reducing the 'school run'. The choice of this example needs some

explanation. In recent years, there has been a marked tendency for children to be driven to school in the UK. This trend derives partly from a general increase in affluence and personal mobility, and accompanying change in lifestyle, and partly out of fear for the children's safety, either from road traffic or from crime (Whitelegg, 1992). Nationally, therefore, by 1995 about 50% of children regularly went to school by car, compared with fewer than 10% in 1980 (DETR, 1998). In the study area of Northamptonshire, a survey conducted by the local authority in 1999 showed that about 60% of journeys to and from school were made by car (Shortland pers. comm.). A recent government survey has suggested that up to 40% of morning peak hour vehicles in urban areas in the UK are engaged, at least in part, on the school run (DETR, 2000a).

Reducing reliance on the motor car for travel to school was thus prioritised in the Transport White Paper as a potentially effective way of reducing rush-hour traffic, whilst at the same time offering potential health benefits, in terms of increased exercise and socialisation, for the children concerned (DETR, 1998). For most children the 'school-run' is likely to represent the single largest period of outdoor exposure because out-of-school activities tend to occur in areas where pollution levels are lower (e.g. around the home). For those who live in homes with limited indoor sources (e.g. smoking, gas cooking) the trip to school may also represent the peak exposure of any significant duration encountered either indoors or outdoors. As such, it represents a valid and politically topical example through which to demonstrate the methodology.

The effects of a walk-to-school policy were modelled by assessing journey-time exposures for a sample of children living within the study area. The walk-to-school policy is implemented as a local management scenario; therefore, the policy will only affect the local, traffic-related component of PM_{10} concentrations. As a basis for modelling, it was assumed that 20% of peak hour trips by car were eliminated by introduction of this policy (about half the maximum theoretically possible) during the hours when the majority of school journeys take place (i.e. 0800 to 0900 hrs & 1500-1600 hrs). A 20% decrease in traffic flows was modelled in the study area using SATURN, as part of NAPS, using the same technique as described in *Section 3.7.1.*

b) Results

Figure 5.9 shows a map of the modelled mean hourly reduction in PM_{10} exposure for 'school-run' hours across the study area. As is to be expected, the greatest absolute savings in exposures are close to main roads.

The absolute change and percentage change in modelled exposures, as a result of the walk-to-school policy, are shown in *Figure 5.10* and *Figure 5.11*, respectively, by travel mode and in terms of both average journey-time and average hourly exposure. As is to be expected, the reduction experienced depends to a great extent on the mode of travel both before and after introduction of the walk-to-school policy.

For children walking both before and after introduction of the policy, the saving in journey-time exposure varies between 0.4 and 2.38 ug/m^3 (mean = 0.97 $\mu g/m^3$), as shown in *Figure 5.10a*. These changes represent a reduction of between about 4% and 11% (mean = 7.3%) in journey-time exposures. For children travelling by car both before and afterwards (*Figure 5.10b*), the reduction in journey-time exposures varies between 0.32 and 2.09 ug/m^3 (mean = 0.85 $\mu g/m^3$), equivalent to between about 2% and 7% (mean = 3.5%) of the average journey-time PM_{10} exposure. Much greater reductions in exposure are achieved by those who switch from car-travel to walking as a result of introduction of the policy, as shown in *Figure 5.10c*. The mean individual saving in journey-time exposure ranges from 9.09 to 16.99 ug/m^3 (mean = 11.86 ug/m3), representing a 33-64% (mean = 49%) reduction in journey-time exposure.

The effect of the policy on changes in exposure has considerably different meaning when the journey component is aggregated into an hourly exposure. As *Figures 5.10d* and *5.10e* show, using hourly exposures results in slightly different exposure savings for those keeping the same mode before and after introduction of the policy, but markedly different effects are seen for those shifting from car to walking. This is because those who walk spend less time at home and school than those who travel by car. Again, taking the average effect for all hours for each

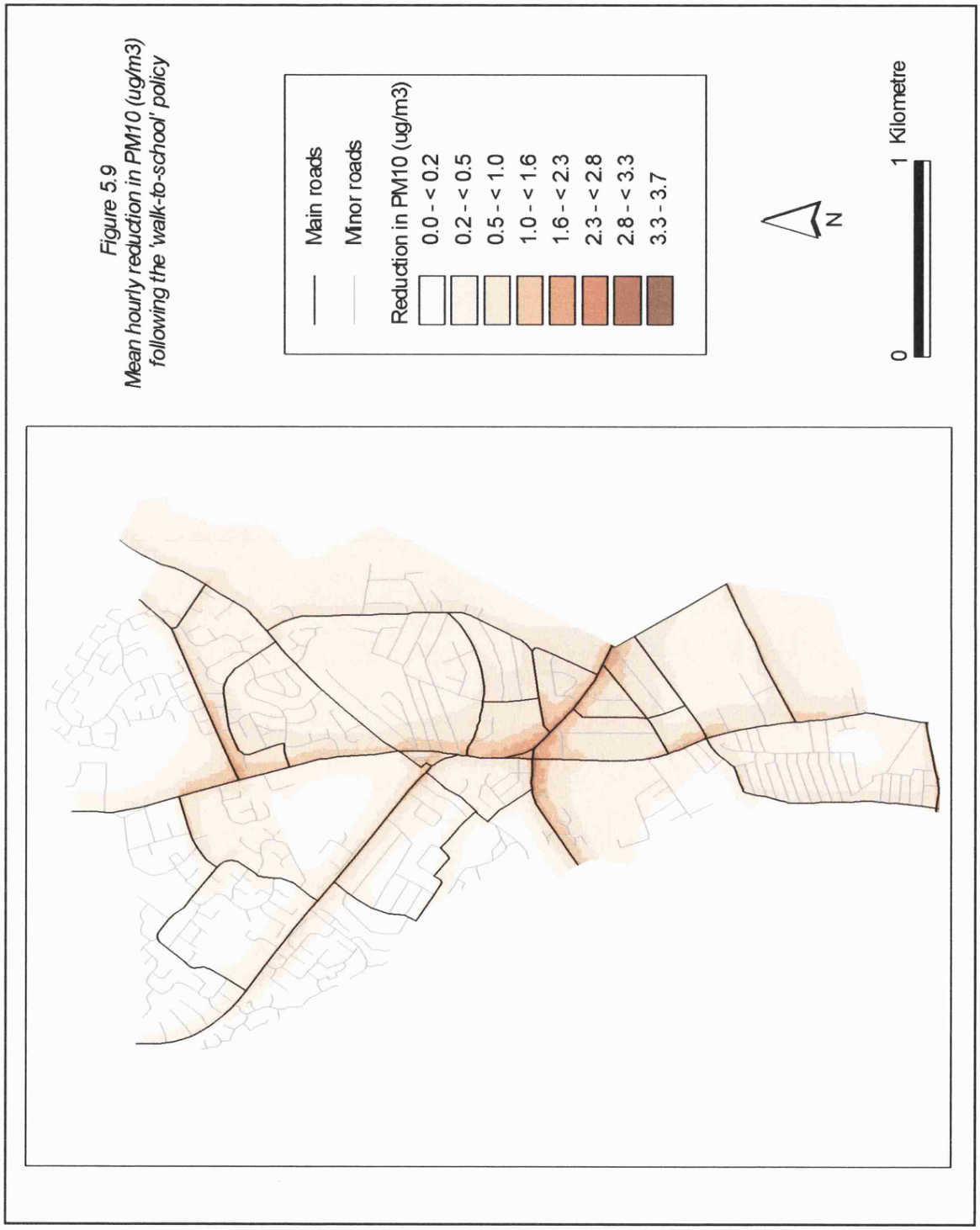


Figure 5.9
Mean hourly reduction in PM10 (ug/m3) following the 'walk-to-school' policy

Figure 5.10: Frequency distribution of absolute change in exposures as a result of the walk-to-school policy: comparison of journey-time and hourly exposures by mode of transport

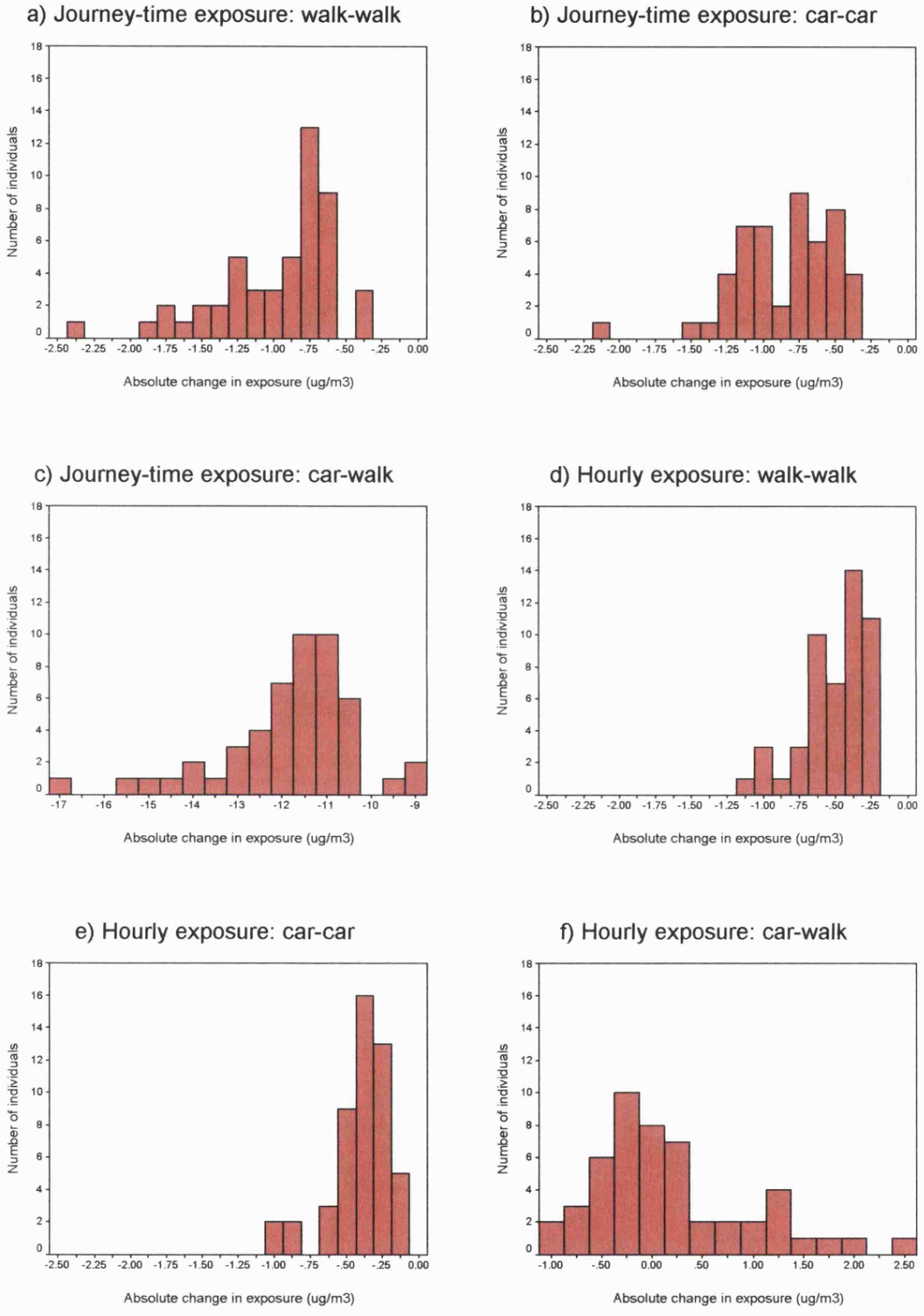
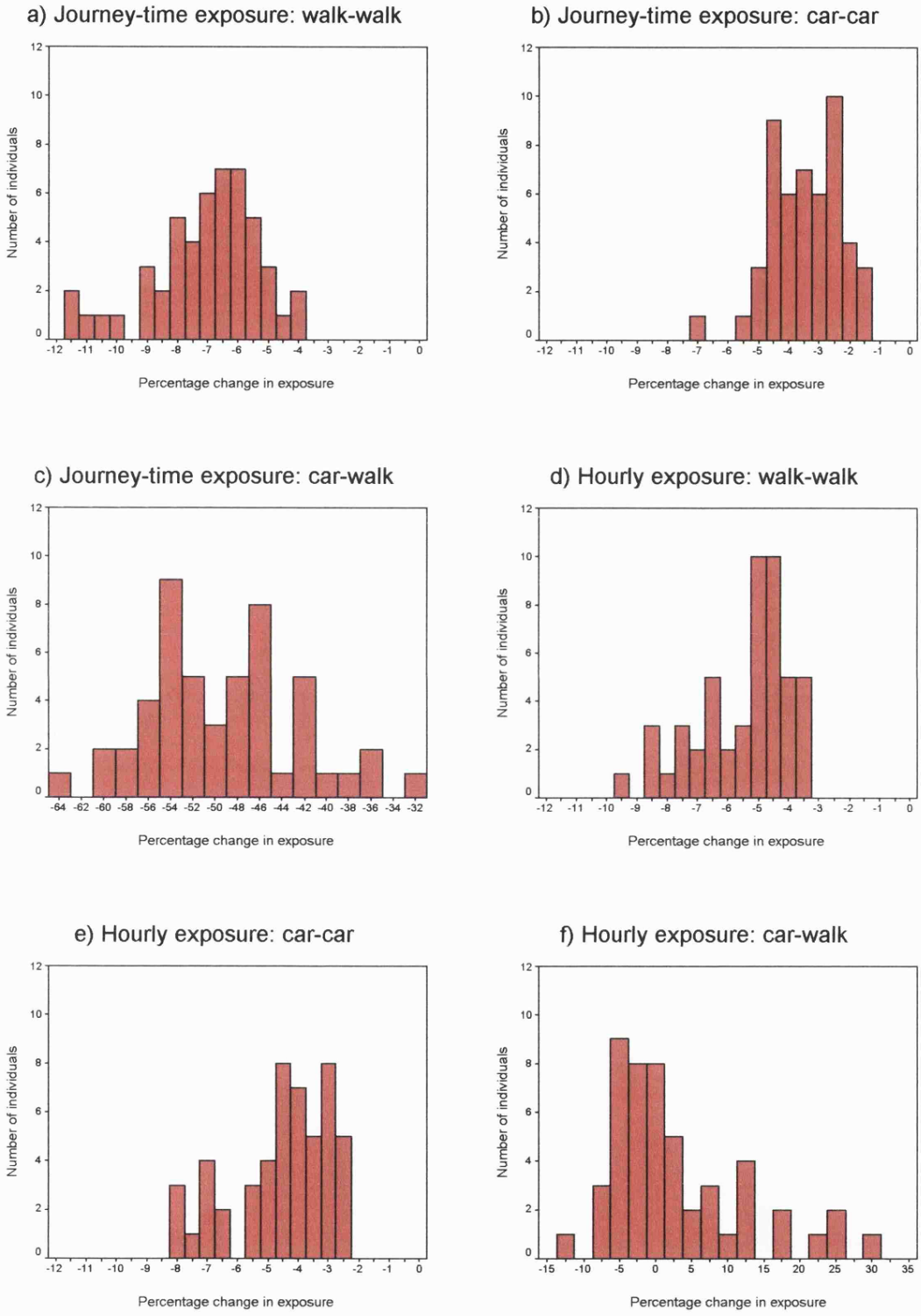


Figure 5.11: Frequency distribution of percentage change in exposures as a result of the walk-to-school policy: comparison of journey-time and hourly exposures by mode of transport



individual, for children walking before and after implementation of the policy the saving in exposure is between 0.27 and 1.18 ug/m^3 (mean = 0.51 μg/m^3). These changes represent a reduction of between about 3% and 9% (mean = 5.6%) in average hourly exposures. For children travelling by car both before and afterwards, the reduction in hourly exposures varies between 0.16 and 1.01 ug/m^3 (mean = 0.40 μg/m^3), equivalent to between about 2 and 8% (mean = 4.7%) of the average hourly PM_{10} exposure.

For those who switch from car to walking, as a result of the policy (*Figure 5.10f*), there is considerable contrast between the savings made in journey-time exposures and those savings made in hourly exposures. The mean individual saving ranges from –2.41 to 0.96 ug/m^3 (mean = 0.96 ug/m^3). A negative value indicates that exposures decrease as a result of switching from car to walking after implementation of the policy. From the 50 children modelled, 26 children experience an increase in exposure under this scenario.

As *Figure 5.11f* shows, although there is a fairly even split in the distribution of negative and positive effects from switching from car to walking, the distribution is skewed towards negative effects of the policy on hourly exposures. There are, in fact, 8 (16% of the total) individuals with more than a 10% increase in average hourly exposures, whereas there is only one individual whose exposure is reduced by more than 10% as a result of the policy. These results can be attributed to the fact that, under the base-case situation, hourly average exposures are higher, on average, for children who walk to school, as noted earlier. The reduction in ambient concentrations under the walk-to-school policy does not sufficiently compensate for the increased exposure, due to longer journey times, for those switching from car to walking when expressed in terms of an average hourly exposure.

As noted, all these estimates relate to average pollution days. For more extreme pollution events, when concentrations in both the ambient and in-car environment are higher, the absolute reduction in modelled exposures would be expected to be higher. If the policy is implemented both locally and nationally, the savings in exposure could also be considerably greater, since the far-travelled and secondary

components of PM_{10} concentrations may also be expected to decline. Indeed, because the far-travelled and secondary component make up more than 80% PM_{10} concentrations in most urban areas, the need is for national policy interventions. This would result in the policy demonstrated here having a substantially larger effect on exposures.

c) Summary

Application of this approach to the example of a walk-to-school policy has shown both the capability of the methodology and the complex effects of such policy interventions. Benefits in terms of reduced journey-time exposure are not spread evenly across the population, but vary substantially depending on the specific routes taken by individuals and the travel mode. The case study has also shown that the benefits in terms of exposure savings are highly dependent on whether the exposure is expressed against journey-time or as an average hourly exposure. The study has shown, in particular, that for those children switching from car to walking there are considerable benefits of this policy in terms of savings in journey-time exposures, but smaller and more varied (often negative) effects in terms of the savings in hourly exposures. This type of policy would have substantially larger effects if implemented nationally.

5.4 SUMMARY

This chapter has tested and validated the three main components of STEMS: 1) a model of outdoor pollution against fixed-site monitored data, 2) a model to estimate in-car exposures using data on monitored concentrations of PM_{10} from the nearest walkway, and 3) a model for estimating outdoor exposure during journeys by walk and in-car. Results may be summarised as follows:

- A reasonable model (r^2 =0.36) of outdoor pollution has been obtained by using regression techniques to combine models of background and local PM_{10}
- The performance of the outdoor model is comparable to results from a number of other studies
- A strong, linear relationship has been observed between concentrations of PM_{10} during simultaneous monitoring of walking and in-car. It is therefore possible to make predictions of in-car concentrations based on monitored data from the nearest walkway
- The modelled concentration during walking and the monitored concentration from the fixed-site perform equally in predicting the monitored concentration during walking. The model does not perform as well as the nearby fixed-site in making prediction of in-car concentrations. It would be expected, however, that a distant monitoring site would not work as well because it would not be measuring the contribution from traffic where the journey takes place.

The STEMS methodology has been demonstrated to show the capability of the model and to assess the effects of a walk-to-school policy aimed at reducing traffic during the 'school-run'. The main finding is that for those switching between car and walk there may be positive or negative effects of the policy in terms of savings in average hourly exposures, depending on their specific journey and time activity patterns.

6 SUMMARY AND CONCLUSIONS

6.1 SUMMARY OF MAIN FINDINGS

This main aim of this research was to develop, test and apply space-time modelling of exposure to air pollution using GIS, as a basis for epidemiological and policy applications. To this end, a mixture of monitoring and modelling techniques were used both to develop new methods and enhance existing methods. This approach involves the linkage within a GIS of five main sub-models: a traffic model, a model of urban air pollution, a 'background' pollution model, a network analysis tool for modelling exposure during journeys, and a time-activity model. Traffic and air pollution monitoring techniques were used as a basis for calibrating models to local conditions and as a basis for model validation. The results and findings of this research have general importance in epidemiological studies and traffic management and policy applications. This section gives a summary of the main results and findings.

A method for modelling outdoor concentrations of PM_{10} was developed as a basis for exposure modelling. This used regression techniques to combine a model of background PM_{10} concentrations with estimates of locally-derived PM_{10} from dispersion modelling. Establishing reliable estimates of PM_{10} concentrations was, however, not an easy task. The main difficulty was in modelling the regional and far-travelled components (i.e. secondary and other). Ideally, monitored PM_{10} from a nearby rural site would have been used, but a suitable site could not be found. Rural sulphate was used as the best available proxy for the secondary component, but was only expected to represent, on average, about 50% of secondary and far-travelled particles. The other main secondary component is nitrates, but again no suitable monitoring data were available to use as a proxy for PM_{10}. Difficulties in modelling the background PM_{10} component were further compounded by the fact that rural sulphate was only measured as a daily average value. This problem was overcome by applying a smooth function across the series of daily values to interpolate hourly estimates. Improved results for modelling PM_{10} concentrations

may be achieved by either using a local rural monitoring site or providing more detailed proxies for background components.

Indoor PM_{10} concentrations were determined from modelled outdoor concentrations. Estimates of indoor concentrations were made by applying an I/O ratio of 0.6 (based on results from empirical studies) to modelled outdoor concentrations. The I/O ratio was determined from the results of a number of empirical studies and a purpose-designed study applied here. Conversion factors to adjust indoor, ground floor concentrations to first floor (i.e. upstairs) concentrations were determined from a purpose-designed survey. Factors of 1.24 and 1.22 were applied to daytime (0600 to 0800 hours and 1600 to 0000 hours) and nighttime (0000 to 0600 hours) concentrations downstairs, respectively.

Personal monitoring during journeys was undertaken to establish a relationship between measurements of PM_{10} taken simultaneously during walking and in-car journeys, as a basis for developing an in-car exposure model. A strongly linear ($r=0.91$) and highly significant relationship ($p<0.0001$) was seen between measurements of PM_{10} taken simultaneously during walking and in-car journeys. At low concentrations (i.e. < 20 $\mu g/m^3$), in-car concentrations were between 1.5 and 2 times those obtained during walking, but the magnitude of difference between in-car and walk concentrations was seen to gradually diminish with increased levels of concentrations, reaching equivalence at 70$\mu g/m^3$. As was to be expected, the two routes used in personal monitoring gave different levels of PM_{10} concentrations. The route used in the April 2000 survey gave generally lower monitored concentrations, reflecting the location of the route – a less heavily trafficked area. The largest effect on the variation in standardised difference (D_S) between walking and in-car measurements was day, with route having a smaller but significant impact. A strong route effect was also seen in comparing mean D_S between walking and in-car measurements. The effect of route was not significant so a single model was built to adjust modelled outdoor exposures at the roadside to give an in-car exposure.

Resampling techniques were used as an alternative to validating the in-car model. Predictions made from jacknifing gave an r^2 of 0.79 (r=0.89), which corresponds well with the r^2 of 0.81 from the original in-car model.

The outdoor pollution model was validated against monitored PM_{10} concentrations from a fixed-site monitor located in the centre of the study area. The model used to estimate PM_{10} concentrations gave a moderate (r^2=0.36) but highly significant fit ($p<0.001$) to observed data, with 74% of modelling predictions within a factor of two of monitored concentrations at the fixed-site. The model has a slight tendency to over-predict at higher concentrations, whereas the majority of low and mid-range concentrations show an under-prediction. These results were shown to be comparable with the best results from other studies.

A network analysis routine was developed to model journey-time exposures during trips made by walking or car. As a measure of performance, modelled exposures from network analysis were compared with monitored concentrations from the fixed-site. The network model and fixed-site monitor performed equally (r^2=0.54) in predicting concentrations during walking, across a five-day period in November 1999. In terms of predicted values within a factor of 2, the performance of the model (61%) and the fixed-site (67%) was similar. In contrast, in terms of r^2, the model (r^2=0.36) did not perform as well as the fixed-site (r^2=0.62) in making predictions of in-car concentrations. However, using the model to predict concentrations during in-car journeys gives 82% of predictions within a factor of 2, whereas using the fixed-site gives 72% within a factor of 2. It must be remembered, however, that the fixed-site is very close to the route, and a distant site may not work as well because it would not be measuring the contribution from the traffic in close proximity to where the journey is being made. The same can be said, of course, for all other microenvironments.

These results can be considered remarkably good given that they are based on somewhat generalised data inputs, which work at a highest resolution of one hour. There were a number of inherent limitations with individual data inputs. The traffic data, for example, is a series of average traffic flows for each hour of the day, which were based on limited monitoring to calibrate a traffic survey made two

years before this study was carried out. Traffic patterns can, however, show day-to-day variation and can show considerable variation between seasons. Even more limiting was the data on vehicle speeds. In the version of ADMS-Urban used in this study, the same average vehicle speed for each road had to be applied to every hour of the day when modelling in 'sequential' mode. It is possible to model each hour separately with vehicle speeds unique to each hour, but this would be to the detriment of the meteorological pre-processor which would then not be able to account for the cumulative effects (over several hours) of meteorological conditions on boundary layer conditions. It is equally important to use meteorological data that reflects local conditions. With the exception of cloud cover, all meteorological variables came from the purpose-deployed meteorological site in the study area. Cloud cover had to be obtained from a distant site (Wittering). The use of remote cloud cover could lead to some instances of ADMS-Urban wrongly characterising boundary layer characteristics. Finally, it is worth noting again that the problems with local data sources are, of course, compounded by the use of a remote proxy to represent background PM_{10} concentrations.

To demonstrate the capability of the exposure model, hourly exposure modelling was conducted for a sample of fifty schoolchildren for 5 weekdays between 12^{tth} and 18th November 1999. Exposure modelling showed that substantial variations in exposure occur depending on time-activity patterns, with journeys most often constituting the highest exposures. Exposure profiles for several individual hours returned a range in exposures in excess of 15 ug/m^3.

Application of STEMS to a walk-to-school policy, in which modelled local traffic levels were reduced by 20%, showed that children walking before and after the policy would save between 0.4 and 2.38 ug/m^3 (mean = 0.97 μg/m^3) in journey-time exposure. Children travelling to school by car would save between 0.32 and 2.09 ug/m^3 (mean = 0.85 μg/m^3) in exposure as a results of the policy. These benefits were reduced when expressed in terms of average hourly exposures because the elevated level of exposures seen during journeys became a smaller proportion of the averaging period. Those children who walked before and after implementation of the policy had an exposure saving between 0.27 and 1.18

ug/m^3 (mean = 0.51 μg/m^3), whereas the saving was between 0.16 and 1.01 ug/m^3 (mean = 0.40 μg/m^3) for children travelling by car both before and after the policy. These benefits are not spread evenly across the sample of schoolchildren, but vary depending on the route used to school and the location of homes and schools.

The main finding from the case study was that the exposure period had a major effect on the measured benefits for those changing from car to walking as a result of the policy. For those shifting from car to walk, the saving in journey-time exposures was between 9.09 and 16.99 ug/m^3 (mean = 11.86 ug/m3), representing a 33-64% (mean = 49%) reduction in journey-time exposure. In contrast, the mean individual hourly saving ranged from –2.41 to 0.96 ug/m^3 (mean = 0.96 ug/m^3), with about the same number of children having negative benefits as having positive benefits.

The results of the case study present a predicament for policy makers and epidemiologists. In terms of the benefits in journey-time exposures there is a good case for encouraging the switch from car to walk. On the other hand, there is little basis for pursuing the policy in terms of reducing hourly exposures for the population as a whole. It is not possible to make inferences about the health benefits of such a policy without some aetiological understanding of traffic-related health problems. There is, however, currently very little understanding of the way that the magnitude, frequency and duration of exposures interact to affect health. In other words, dose-response relationships are not yet properly understood. The problems in understanding the links between exposure and health are further complicated because toxicity of traffic-related particles also remains largely misunderstood. It is not yet clear exactly which elements of traffic-related particles are causing health problems. There is a growing body of evidence to suggest that the causes of traffic-related health problems are associated with minute (ultrafine) particles (less than 0.1 μm in diameter) of elemental carbon and PAHs (Wichmann, 2000). However, as discussed in chapter 2, associations between health and coarse particles are still being found (Levy *et al.*, 2000).

These gaps in understanding the link between exposure and health are, of course, in part due to the fairly crude methods used by many studies for assessing exposure. GIS alongside the methods developed here clearly have an important role to play in the endeavour to establish links between exposure and health. The STEMS approach not only can be used to model exposures for policy applications but also provides a basis for studying the relationships between exposure and health. Due to the complexity of relating exposures to health outcomes, air pollution studies should be based on panels or cohorts rather than time-series studies (Künzli *et al.*, 2001). The STEMS approach offers a new level of detail in exposure assessment that is particularly geared towards the needs of panel and cohort studies. STEMS could be used, for example, to monitor the health of individuals or groups of people against their exposure to air pollution, at both high spatial and temporal resolution. Ultimately, the real value of STEMS is that it is a significant departure from static methods of exposure assessment towards a more dynamic approach that reflects the complex patterns of human movement through a continually changing field of pollution.

6.2 IMPLICATIONS FOR FUTURE RESEARCH

As this research has shown, there is still a need for considerable improvements on the methods used here. Firstly, modelled exposures would ideally be compared with data collected from personal monitoring lasting periods of at least one day. In this way, it would be possible to validate each component in STEMS separately. This was not within the scope of this study, as it would have involved considerable additional work, but would nevertheless provide a more rigorous testing of the methods applied here.

It would be also be useful to collect additional data to study in more detail the factors that may influence the relationship between measurements of PM_{10} made while walking and in-car. Clearly, monitoring on routes in locations with different levels of traffic flow would add rigour to the development of models used in estimating in-car exposures. It would also be useful to collect data, and build it into the model, that reflected the range of different in-car conditions and driver

behaviours. In this study, the same driver was used throughout personal monitoring and the ventilation conditions and location of the monitor were fixed. The type of car, number of passengers, cleanliness of furnishings, and ventilation could affect the level of in-car concentrations. The exposure of individuals could even be affected by which seat they occupy. A dual-car monitoring campaign showed a difference in PM_{10} concentrations between two cars, of similar age and specification, when the number of people inside the car and ventilation conditions were the same. The cumulative effect of type of car, difference in numbers of passengers, ventilation, and driver behaviour could therefore be considerable.

Another improvement would be in considering exposure in terms of intake dose. This could be achieved by monitoring breathing rates of subjects under the different levels of exercise and apply dose factors for different modes of transport to modelled exposures obtained from STEMS. The use of breathing rates may have important implications for modelling exposures. Using the example of the walk-to-school policy, there may be stronger negative effects for those switching from car to walk if exposure was considered in terms of dose. People who walk are more likely to have higher breathing rates over car passengers (van Wijnen *et al.*, 1995), which would presumably lead to a higher rate of intake. In reality, however, higher levels of breathing may not lead to an increased health risk by the very fact that people who walk may be fitter and less sensitive to the effects of air pollution.

A number of practical implications of implementing STEMS also need to be addressed. It was difficult to produce large numbers of high resolution maps, as a basis for exposure modelling, because of the considerable amount of processing time involved. One possible solution to this problem is to use alternative methods of obtaining concentration estimates such as a purpose-built dispersion model, or add-on software for existing models that can enhance the map production process. There is also an urgent need for better models of background (i.e. regional and far-travelled) PM_{10} concentrations. One immediate improvement would be to populate the existing national network of monitors with PM_{10} monitors in rural locations. A second possibility would be to deploy monitors as part of

exposure modelling studies. The latter option would, however, probably be costly to implement for individual studies.

A useful extension of STEMS would be a population-level exposure model that uses probabilistic methods to simulate the time-activity of large numbers of the population. This could add significant value to assessing the effects of traffic management policies that have potentially widespread implications for the geography of road traffic pollution. This would also overcome a problem inherent in time-activity data, in that it is does not reflect the complex changeable behaviour of an individuals' activity. For example, children might vary the mode of transport used to make journeys to school depending on weather conditions and the timing of their out-of-school activities.

Despite the need for further innovation and improved methods of exposure assessment, this research has nonetheless clearly demonstrated the potential of GIS-based techniques for space-time modelling of human exposure to air pollution. The outcome of this work is an exposure model (STEMS) that loose-couples a mixture of existing technologies and new methods of exposure assessment, in order to provide exposure estimates for a group, or at the individual level. The model has been tested, validated and applied in Northampton to model exposure to PM_{10} on an hourly basis. The STEMS method is to be further developed as part of two EU-funded studies: 1) a study in several European cities of acute effects of exposure to traffic-related air pollution, and 2) a study developing and linking a number of environmental models to provide an integrated basis for modelling exposure to a range of environmental pollutants.

APPENDIX 3A

EXAMPLE OF AN ADMS-Urban MET. FILE

VARIABLES:
7
YEAR
TDAY
THOUR
T0C
U
PHI
CL
DATA:
2000,101,5,3.9,1.2,356.5,0
2000,101,6,3.8,1.5,186.1,0
2000,101,7,4.5,2.0,30.2,1
2000,101,8,5.8,1.9,23.2,6
2000,101,9,7.5,2.0,38.2,1
2000,101,10,9.0,2.0,42.3,4
2000,101,11,10.2,2.0,48.7,3
2000,101,12,10.7,1.9,48.4,2
2000,101,13,11.2,1.7,37.5,2
2000,101,14,11.9,1.5,34.3,2
2000,101,15,12.3,1.7,107.6,2
2000,101,16,12.7,1.8,100.6,2
2000,101,17,12.6,1.5,178.2,2
2000,101,18,12.0,1.5,239.6,3
2000,101,19,11.1,1.0,264.6,4
2000,101,20,9.4,1.0,83.9,6
2000,101,21,8.1,1.0,100.6,6
2000,101,22,7.1,1.0,55.6,6
2000,101,23,6.7,1.0,83.5,7
2000,102,0,6.6,1.0,178.5,7
2000,102,1,6.1,1.0,168.7,7
2000,102,2,6.2,1.0,174.8,8
2000,102,3,6.5,1.0,192.1,8
2000,102,4,6.6,1.0,175.6,8
2000,102,5,6.6,1.0,178.3,8
2000,102,6,6.4,1.0,179.6,8
2000,102,7,6.6,1.2,194.6,8
2000,102,8,6.5,1.0,199.7,8
2000,102,9,6.2,1.1,180.2,8
2000,102,10,6.2,1.3,221.1,8
2000,102,11,5.9,1.0,236.8,8
2000,102,12,6.5,1.0,255.2,8
2000,102,13,7.2,1.0,289.0,8
2000,102,14,7.7,1.0,236.9,8
2000,102,15,8.4,1.0,232.4,7
2000,102,16,8.7,1.0,81.3,7
2000,102,17,8.6,1.0,75.1,7
2000,102,18,8.4,1.0,87.0,6
2000,102,19,7.8,1.0,74.5,7
2000,102,20,7.1,1.0,143.2,7
2000,102,21,6.3,1.0,266.5,7
2000,102,22,6.3,1.0,173.9,8
2000,102,23,6.1,1.4,259.2,8

APPENDIX 3B

EXAMPLE OF AN ADMS-Urban INPUT FILE

```
&ADMS_HEADER
Comment = 'This is an ADMS Urban Model parameter file                    '
Model = 'ADMS-Urban'
Version = 1.53
Complete = 1
/

&ADMS_PARAMETERS_SUP
SupSiteName      = 'Kingsthorpe pollution mapping: 12-18th Nov 1999      '
SupProjectName   = 'NAPS                                                 '
SupReleaseType   = 0
SupComplexEffects = 0
SupOther         = 0
SupRoughness     = 5.000000000000000e-001
SupLatitude      = 5.200000000000000e+001
SupUseMinLmo     = 1
SupMinLmo        = 3.000000000000000e+001
SupPufType       = 0
SupCalcChm       = 1
SupCalcDryDep    = 0
SupCalcWetDep    = 0
SupUseHourlyEmissionFactors = 1
SupHourlyEmissionFactorWeekday =
    1.500000e-001  9.000000e-002  7.000000e-002  5.000000e-002
    1.100000e-001  2.700000e-001  5.600000e-001  1.380000e+000
    1.780000e+000  1.450000e+000  1.310000e+000  1.340000e+000
    1.430000e+000  1.480000e+000  1.470000e+000  1.640000e+000
    1.780000e+000  1.890000e+000  1.610000e+000  1.340000e+000
    1.050000e+000  7.900000e-001  5.900000e-001  3.800000e-001
SupHourlyEmissionFactorSaturday =
    2.900000e-001  1.900000e-001  1.200000e-001  8.000000e-002
    1.100000e-001  1.800000e-001  3.300000e-001  5.600000e-001
    9.800000e-001  1.360000e+000  1.580000e+000  1.600000e+000
    1.670000e+000  1.540000e+000  1.420000e+000  1.300000e+000
    1.400000e+000  1.380000e+000  1.230000e+000  1.160000e+000
    9.200000e-001  6.000000e-001  4.800000e-001  5.000000e-001
SupHourlyEmissionFactorSunday =
    3.400000e-001  2.200000e-001  1.600000e-001  1.100000e-001
    6.000000e-002  1.000000e-001  1.500000e-001  2.600000e-001
    3.700000e-001  6.900000e-001  1.200000e+000  1.320000e+000
    1.450000e+000  1.170000e+000  1.150000e+000  1.110000e+000
    1.140000e+000  1.110000e+000  1.130000e+000  9.800000e-001
    8.500000e-001  6.900000e-001  4.700000e-001  3.000000e-001
SupUseBackgroundParFile = 1
SupBackgroundParPath = 'c:\data\modell~1\nov1999\nov12.par'
SupUseGravimetricParFactor = 0
SupGravimetricParFactor   = 1.400000000000000e+000
SupTrafficEmissionsDataset = 'DMRB 1999'
/
&ADMS_PARAMETERS_MET
MetDataSource            = 0
MetDataFileWellFormedPath = 'c:\data\modell~1\nov1999\nov12.met'
MetFileWindHeight        = 6.000000000000000e+000
MetWindSectorSize        = 0
MetLateralSpreadType     = 0
MetLateralSpreadStdDev   = 0.000000000000000e+000
MetDataIsSequential      = 1
MetDataIsSequential_Subordinate = 1
MetSiteIsRepr            = 1
MetUsePrecipFactor       = 0
MetPrecipFactor          = 0.000000000000000e+000
MetUseRoughChanges       = 0
MetSurfRough             = 2.000000000000000e-001
MetHandWindHeight        = 1.000000000000000e+001
MetHeatFluxType          = 0
MetInclBoundaryLyrHt     = 1
MetInclSurfaceTemp       = 0
```

```
MetInclLateralSpread     = 0
MetHandNumEntries        = 0
/
&ADMS_PARAMETERS_BLD
BldNumBuildings = 0
/
&ADMS_PARAMETERS_HIL
HilGridSize   = 1
HilRoughInput  = 0
HilTerrainPath = 'c:\adms-urb\data\terrain.ter'
HilRoughPath   = 'c:\adms-urb\data\roughnes.ruf'
/
&ADMS_PARAMETERS_FLC
FlcAvgTime          = 0.000000000000000e+000
FlcCalcToxicResponse = 0
FlcToxicExp         = 1.000000000000000e+000
FlcCalcPercentiles   = 0
FlcNumPercentiles    = 0
FlcCalcPDF          = 0
FlcPDFMode           = 0
FlcNumPDF            = 0
/
&ADMS_PARAMETERS_GRD
GrdType             = 2
GrdSpacingType       = 2
GrdRegularMin =
 4.735000e+005
 2.640000e+005
GrdRegularMax =
 4.750000e+005
 2.655000e+005
GrdVarSpaceNumPointsX  = 0
GrdVarSpaceNumPointsY  = 0
GrdGriddedZ          = 0.000000e+000
GrdPtsNumPoints       = 1
GrdPtsPointNames =
    'HORIBA          '
GrdPtsPointsX =
    4.751750e+005
GrdPtsPointsY =
    2.632350e+005
GrdPtsPointsZ =
    3.000000e+000
/
&ADMS_PARAMETERS_PUF
PufStart          = 1.000000000000000e+002
PufStep           = 1.000000000000000e+002
PufNumSteps        = 10
/
&ADMS_PARAMETERS_GAM
GamCalcDose          = 0
GamNumOutputPoints    = 0
/
&ADMS_PARAMETERS_OPT
OptNumOutputs        = 4
OptPolName =
    'PM              ' 'CO              ' 'NO2             ' 'NOx             '
OptInclude =
    1 1 1 1
OptShortOrLong =
    0 0 0 0
OptSamplingTime =
    3.600000e+003 3.600000e+003 3.600000e+003 3.600000e+003
OptUseRollingAvg =
    0 0 0 0
OptPercentile1 =
    0.000000e+000 0.000000e+000 0.000000e+000 0.000000e+000
```

```
OptPercentile2 =
    0.000000e+000  0.000000e+000  0.000000e+000  0.000000e+000
OptUnits =
    'ug/m³' 'ppm ' 'ug/m³' 'ug/m³'
OptGroupsOrSource    = 0
OptNumGroups         = 1
OptIncludedGroups =
    'All sources        '
OptIncludedSource    = '_____'
/
&ADMS_PARAMETERS_CHM
ChmScheme          = 0
ChmBackgroundPath    = 'c:\adms-urb\data\backgnd.bkg'
ChmManualBackgrounds = 1
ChmNumLevels         = 6
ChmPolNames =
    'NOx             ' 'NO2             ' 'SO2             ' 'VOC             '
    'CO              ' 'O3              '
ChmLevels =
    0.000000e+000  0.000000e+000  0.000000e+000  0.000000e+000
    0.000000e+000  0.000000e+000
/
&ADMS_PARAMETERS_ETC
SrcNumSources   = 106
PolNumPollutants = 7
PolNumIsotopes   = 1
/

  &ADMS_POLLUTANT_DETAILS
  PolName              = 'PM              '
  PolPollutantType       = 1
  PolGasDepVelocityKnown   = 1
  PolGasDepositionVelocity = 0.000000000000000e+000
  PolGasType             = 0
  PolParDepVelocityKnown   = 1
  PolParTermVelocityKnown  = 1
  PolParNumDepositionData  = 1
  PolParDepositionVelocity =
   0.000000e+000
  PolParTerminalVelocity =
   0.000000e+000
  PolParDiameter =
   1.000000e-005
  PolParDensity =
   1.000000e+003
  PolParMassFraction =
   1.000000e+000
  PolWetWashoutKnown = 1
  PolWetWashout      = 0.000000000000000e+000
  PolWetWashoutA     = 1.000000000000000e-004
  PolWetWashoutB     = 6.400000000000000e-001
  PolConvFactor      = 0.000000000000000e+000
  /

  &ADMS_ISOTOPE_DETAILS
  IsoName              = '(unnamed-isotope)  '
  IsoPollutantType       = 0
  IsoGasDepVelocityKnown   = 1
  IsoGasDepositionVelocity = 0.000000000000000e+000
  IsoGasType             = 0
  IsoParDepVelocityKnown   = 1
  IsoParTermVelocityKnown  = 1
  IsoParNumDepositionData  = 1
  IsoParDepositionVelocity =
   0.000000e+000
  IsoParTerminalVelocity =
   0.000000e+000
```

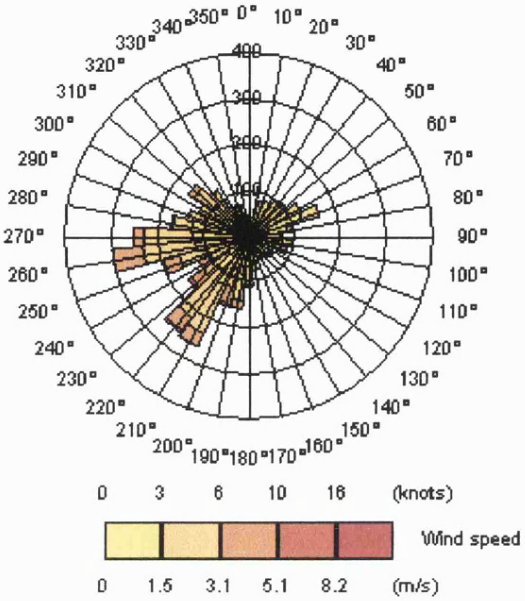
```
 IsoParDiameter =
  1.000000e-006
 IsoParDensity =
  1.000000e+003
 IsoParMassFraction =
  1.000000e+000
 IsoWetWashoutKnown = 1
 IsoWetWashout     = 0.000000000000000e+000
 IsoWetWashoutA    = 1.000000000000000e-004
 IsoWetWashoutB    = 6.400000000000000e-001
 IsoConvFactor     = 1.000000000000000e+000
 /

&ADMS_SOURCE_DETAILS
SrcName         = '10081009          '
SrcHeight       = 0.000000000000000e+000
SrcDiameter     = 1.000000000000000e+000
SrcVolFlowRate  = 0.000000000000000e+000
SrcVertVeloc    = 0.000000000000000e+000
SrcTemperature  = 1.500000000000000e+001
SrcMolWeight    = 2.896000000000000e+001
SrcDensity      = 1.225000000000000e+000
SrcSpecHeatCap  = 1.012000000000000e+003
SrcSourceType   = 4
SrcReleaseAtNTP  = 0
SrcVolFlowKnown  = 0
SrcDensityKnown  = 0
SrcX1           = 0.000000000000000e+000
SrcY1           = 0.000000000000000e+000
SrcL1           = 1.000000000000000e+001
SrcL2           = 0.000000000000000e+000
SrcNumGroups    = 1
SrcGroup =
  'All sources       '
SrcTraEmissionsMode = 0
SrcTraYear        = 1999
SrcNumVertices = 5
SrcTraNumTrafficFlows = 2
SrcNumPollutants     = 4
SrcPollutants =
  'VOC             ' 'PM              ' 'CO              ' 'NOx             '
SrcPolEmissionRate =
  2.040380e-001  1.967110e-002  1.177790e+000  4.291870e-001
SrcPolTotalemission =
  1.000000e+000  1.000000e+000  1.000000e+000  1.000000e+000
SrcPolStartTime =
  0.000000e+000  0.000000e+000  0.000000e+000  0.000000e+000
SrcPolDuration =
  0.000000e+000  0.000000e+000  0.000000e+000  0.000000e+000
SrcNumIsotopes      = 0
/

 &ADMS_SOURCE_VERTEX
 SourceVertexX = 4.752110000000000e+005
 SourceVertexY = 2.607760000000000e+005
 /

 &ADMS_SOURCE_VERTEX
 SourceVertexX = 4.752310000000000e+005
 SourceVertexY = 2.608300000000000e+005
 /

 &ADMS_SOURCE_VERTEX
 SourceVertexX = 4.752730000000000e+005
 SourceVertexY = 2.609210000000000e+005
 /
```

```
&ADMS_SOURCE_VERTEX
SourceVertexX = 4.753160000000000e+005
SourceVertexY = 2.610030000000000e+005
/

&ADMS_SOURCE_VERTEX
SourceVertexX = 4.753370000000000e+005
SourceVertexY = 2.610690000000000e+005
/

&ADMS_TRAFFICFLOW_DETAILS
TraVehicleCategory    = 'light duty vehicle '
TraAverageSpeed       = 3.000000000000000e+001
TraVehicleCount       = 886
TraNumRoadPollutants  = 4
TraPolName =
 'NOx             ' 'VOC             ' 'PM              ' 'CO              '
TraEmissionFactor =
 7.560000e-001 7.400000e-001 3.830000e-002 4.631400e+000
/

&ADMS_TRAFFICFLOW_DETAILS
TraVehicleCategory    = 'heavy duty vehicle '
TraAverageSpeed       = 3.000000000000000e+001
TraVehicleCount       = 54
TraNumRoadPollutants  = 4
TraPolName =
 'NOx             ' 'VOC             ' 'PM              ' 'CO              '
TraEmissionFactor =
 1.880000e+001 1.461043e+000 6.830000e-001 2.530000e+000
/
```

(N.B. This input file only represents 1 source. There were 106 sources in total)

APPENDIX 3C

WIND ROSE FOR OUTDOOR AIR POLLUTION MODELLING


July 1999 - March 2000
0°
10°
20°
30°
40°
50°
60°
70°
80°
90°
100°
110°
120°
130°
140°
150°
160°
170°
180°
190°
200°
210°
220°
230°
240°
250°
260°
270°
280°
290°
300°
310°
320°
330°
340°
350°
0 3 6 10 16 (knots)
Wind speed
0 1.5 3.1 5.1 8.2 (m/s)

APPENDIX 4A

NETWORK ANALYSIS ROUTINE

```
/*-------------------------------------------------------------------------------------------------------
/*
/*                    NETWORK ANALYSIS ROUTINE FOR TOTEM
/*
/*      1. Drops points, representing start and end locations of a journey, onto a network of
/*         roads or pathways.
/*      2. Determines the quickest path between the points.
/*      3. Extracts this route from the network to create a new coverage.
/*      4. Intersects the route with a pollution map to determine exposure along the route
/*      5. Plots time-series graph of exposure for given route
/*      6. Calculates exposure measures for each route
/*
/*      J. Gulliver     Created:        13th May 1999
/*                      Last modified:  08th Oct 2001
/*
/*-------------------------------------------------------------------------------------------------------


/* Reads the start and end locations of each journey and opens a file for exporting exposure
/* measures for each route

  &s file_id [open outward.txt openstat -r]

  &if %openstat% ne 0 &then
  &return &error Error in creating file
  &else &type File is created

  &s line [read %file_id% readstat]

  &s file [open temp.txt openstat -w]

  &s writestat [write %file% [quote StartID,EndID,AvExp,SumPol,SumImp,SumTime]]


/* Loop to read start and end locations of each journey and call main routine

/*-----------------------------------------------------------------

  &do &while %readstat% ne 102
      &s id1 [extract 1 %line%]
      &type Start ID = %id1%
      &s xstart [extract 2 %line%]
      &type Start X-COORD = %xstart%
      &s ystart [extract 3 %line%]
      &type Start Y-COORD = %ystart%
      &type
      &s id2 [extract 4 %line%]
      &type End ID = %id2%
      &s xend [extract 5 %line%]
      &type End X-COORD = %xend%
      &s yend [extract 6 %line%]
      &type End Y-COORD = %yend%
      &s stime [extract 7 %line%]
      &s ftime [extract 8 %line%]
      &type
      &type Journey period is %stime% to %ftime%
      &call pointdrop
  &end

/*-----------------------------------------------------------------
```

```
    &s closestat [close %file%]
    &s closestat [close %file_id%]

    &call total

  &return


/*-----------------------------------------------------------------
  &routine pointdrop
/*-----------------------------------------------------------------

    generate resloc
    points
    %id1%,%xstart%,%ystart%
    end
    quit

    generate destloc
    points
    %id2%,%xend%,%yend%
    end
    quit

    build resloc point

    build destloc point


  /* overlay selected route with pollution map

    intersect allroads f:\projects\stems\exposure\mask temp2 line

    &data arc info
      ARC
      SELECT TEMP2.AAT
      CALCULATE TEMP2-ID = TEMP2#
      Q STOP
    &end

    build temp2 line
    build temp2 node
    build temp2 point


  /* select the node nearest to both the start and end locations

    near resloc temp2 node 5000
    near destloc temp2 node 5000


  /* return the ID of the node to a text string

    &data arc info
      ARC
      SELECT RESLOC.PAT
      EXPORT F:\PROJECTS\STEMS\EXPOSURE\START.TXT SDF TEMP2#
      Q STOP
    &end
```

```
  &data arc info
    ARC
    SELECT DESTLOC.PAT
    EXPORT F:\PROJECTS\STEMS\EXPOSURE\END.TXT SDF TEMP2#
    Q STOP
  &end


/* create variables to receive the node Ids


  &s ndstart [open start.txt openstat -r]
  &s start [read %ndstart% readstat]
  &s closestat [close %ndstart%]


  &s ndend [open end.txt openstat -r]
  &s end [read %ndend% readstat]
  &s closestat [close %ndend%]



/*-----------------------------------------------------------------
/* call all other routines
/*-----------------------------------------------------------------


&call route

&call nodedist

&call graph

&call stats



/* kills temporary files and coverages at end of each iteration


  &data arc info
    ARC
    SELECT ROUTE.STP
    ERASE ROUTE.STP
    Y
    Q STOP
  &end


  kill rte%id1% all
  kill route2 all

  kill resloc all
  kill destloc all

  kill temp2 all

  &s closestat [close %start%]
  &s closestat [close %end%]
```

```
    &delvar %start%
    &delvar %end%
    &delvar %stime%
    &delvar %ftime%

    &sys del start.txt
    &sys del end.txt

    &s line [read %file_id% readstat]

  &return


/*-------------------------------------------------------------
  &routine route
/*-------------------------------------------------------------


  /* create a 'stops file', for use by PATH, containing start and end locations

    &data arc info
      ARC
      DEFINE ROUTE.STP
      TEMP2-ID,8,8,I
      IN_ORDER,2,2,I
      ROUTE_ID,2,2,I
      STOP_IMP,3,3,N,1
      ~
      SELECT ROUTE.STP
      ADD
      %start%,1,1,1.0
      %end%,2,1,1.0
      ~
      Q STOP
    &end


  /* calculate impedance values for each arc

    additem temp2.aat temp2.aat imp 8 8 n 3

    &data arc info
      ARC
      SELECT TEMP2.AAT
      CALCULATE IMP = LENGTH / ( SPEED / 3.6 )
      Q STOP
    &end


  /* Use PATH to determined the shortest route between the start and end locations (STOPS)

    ap
    mapex temp2
    disp 9999
    netcover temp2 path1
    arcs temp2
    impedance imp imp
    stops route.stp in_order route_id stop_imp
    path stops
    arcs temp2
```

```
  reselect temp2 nodes keyfile route.stp temp2-id
  nodes temp2 ids
  routelines temp2 path1 2
  points destloc ids
  points resloc ids
  q


/* reselect arcs fot the derived route and create a new coverage

  reselect temp2 rte%id1% section.path1
  res path1-id gt 0
  ~
  n
  n

  build rte%id1% line


  ap
  disp 9999 4
  mapex rte%id1%
  shadeset color
  polygonshades polmap grid-code
  linecolor 6
  linesize 0.03
  arcs rte%id1%
  q

&return


/*-----------------------------------------------------------------
  &routine nodedist
/*-----------------------------------------------------------------


/* find the coordinates of the network start point

  nodepoint temp2 temp3
  addxy temp3 point
  build temp3 point

  reselect temp3 temp4 point
  res temp3# = %start%
  ~
  n
  n

  &data arc info
    ARC
    SELECT TEMP4.PAT
    EXPORT F:\PROJECTS\STEMS\EXPOSURE\NXSTART.TXT SDF X-COORD
    Q STOP
  &end

  &data arc info
    ARC
    SELECT TEMP4.PAT
    EXPORT F:\PROJECTS\STEMS\EXPOSURE\NYSTART.TXT SDF Y-COORD
```

```
  Q STOP
&end

kill temp3 all
kill temp4 all


&s nxtemp1 [open nxstart.txt openstat -r]
&s nxstart [read %nxtemp1% readstat]
&s closestat [close %nxtemp1%]
&delvar nxtemp1

&s nxtemp2 [open nystart.txt openstat -r]
&s nystart [read %nxtemp2% readstat]
&s closestat [close %nxtemp2%]
&delvar nxtemp2

ae
mapex rte%id1%
editcov rte%id1%
editf section.path1
sel all
remeasure length %nxstart% %nystart%
q
y
y


/* create a second STOPS file for NODEDISTANCE calculation

&data arc info
  ARC
  DEFINE NODE.STP
  rte%id1%-ID,8,8,I
  IN_ORDER,2,2,I
  ROUTE_ID,2,2,I
  STOP_IMP,3,3,N,1
  ~
  SELECT NODE.STP
  ADD
  %start%,1,2,1.0
  %end%,2,2,1.0
  ~
  Q STOP
&end


/* calculate distances between locations and each node along the network

ap
mapex rte%id1%
netcover rte%id1% path2
stops node.stp in_order route_id stop_imp
impedance length length
path stops
nodedistance stops nodes nodedist.dat 10000 network ids
q
```

```
/* join accumulative lengths to the Node Attribute Table

  &data arc info
    ARC
    SELECT NODEDIST.DAT
    ALTER
    rte%id1%-IDB
    rte%id1%-ID
    ~
    ~
    ~
    ~
    ~
    ~
    RESELECT FOR rte%id1%-IDA = %end%
    PURGE
    Y
    Q STOP
  &end

  joinitem rte%id1%.nat nodedist.dat rte%id1%.nat rte%id1%-id rte%id1%-id


/* display the result of the route selection

  ae
  mapex rte%id1%
  editcov rte%id1%
  backcov temp2 2
  backenv arcs
  drawe arcs
  draw
  &pause
  q


/* remove files created by PATH etc.

  &data arc info
    ARC
    SELECT ROUTE.STP
    ERASE ROUTE.STP
    Y
    SELECT NODE.STP
    ERASE NODE.STP
    Y
    SELECT NODEDIST.DAT
    ERASE NODEDIST.DAT
    Y
    Q STOP
  &end

&return


/*-----------------------------------------------------------------
  &routine graph
/*-----------------------------------------------------------------

    &data arc info
```

```
    ARC
    SELECT rte%id1%.NAT
    CALCULATE rte%id1%-ID = rte%id1%#
    SELECT rte%id1%.AAT
    CALCULATE rte%id1%-ID = FNODE#
    Q STOP
  &end

  joinitem rte%id1%.nat rte%id1%.aat graph.dat rte%id1%-id rte%id1%-id

  &data arc info
    ARC
    SELECT GRAPH.DAT
    RESELECT FOR LENGTH = 0
    PURGE
    Y
    SORT ON NETWORK
    Q STOP
  &end

  ap
  disp 9999 4
  mapex rte%id1%
  pageunits cm
  pagesize 30 21
  maplimits 2 2 12 12
  box 1.75 1.75 12.25 12.25
  arclines rte%id1% 4
  graphextent graph.dat info network grid-code2
  graphlimits 14 6 28 20
  units graph
  linecolor 2
  graphline graph.dat info network grid-code2
  linecolor 5
  axis horizontal
  axisruler Metres
  axis vertical
  axisruler PM10(ug/m^3)
  q

  &data arc info
    ARC
    SELECT GRAPH.DAT
    ERASE GRAPH.DAT
    Y
    Q STOP
  &end

&return

/*-----------------------------------------------------------------------
&routine stats
/*-----------------------------------------------------------------------

  statistics rte%id1%.aat sum_imp.dat
  sum imp
  ~
  n
  n
```

```
/* sum of weighted arcs (time * pollution)

  statistics rte%id1%.aat sum_pol.dat
  sum imp grid-code2
  ~
  n
  n

  &data arc info
    ARC
    SELECT SUM_IMP.DAT
    EXPORT F:\PROJECTS\STEMS\EXPOSURE\SUM_IMP.TXT SDF SUM-IMP
    DELETE SUM_IMP.DAT
    y
    Q STOP
  &end

  &data arc info
    ARC
    SELECT SUM_POL.DAT
    EXPORT F:\PROJECTS\STEMS\EXPOSURE\SUM_POL.TXT SDF SUM-W-IMP
    DELETE SUM_POL.DAT
    y
    Q STOP
  &end

  &s ndtemp1 [open sum_imp.txt openstat -r]
  &s sumimp [read %ndtemp1% readstat]
  &s closestat [close %ndtemp1%]
  &delvar ndtemp1

  &s ndtemp2 [open sum_pol.txt openstat -r]
  &s sumpol [read %ndtemp2% readstat]
  &s closestat [close %ndtemp2%]
  &delvar ndtemp2

  &s avexp [calc %sumpol% / %sumimp%]


/* find the proportion of the journey where exposure exceeds a specified threshold

  reselect rte%id1% route2 line
  res grid-code2 gt 10
  ~
  n
  n

  build route2 line
  build route2 node

  statistics route2.aat sum_time.dat
  sum imp
  ~
  n
  n

  &data arc info
  ARC
  SELECT SUM_TIME.DAT
  EXPORT F:\PROJECTS\STEMS\EXPOSURE\SUMTIME.TXT SDF SUM-IMP
```

```
    DELETE SUM_TIME.DAT
    y
    Q STOP
    &end

    &s ndtemp1 [open sumtime.txt openstat -r]
    &s sumtime [read %ndtemp1% readstat]
    &s closestat [close %ndtemp1%]

    &s exprop [calc %sumtime% / %sumimp%]
    &s exprop2 [calc %exprop% * 100]


    &type ------------------------------------------------------------------------------------------------------------
    &type
    &type - The total time for this journey is %sumimp% seconds
    &type
    &type - The average exposure for this journey is %avexp% ug/m^3
    &type
    &type ------------------------------------------------------------------------------------------------------------


    &s writestat [write %file% [quote
    %id1%,%id2%,%avexp%,%sumpol%,%sumimp%,%sumtime%]]

    del sumtime.txt
    del sum_imp.txt
    del sum_pol.txt

&return
```

APPENDIX 4B

GRIDCONV
(SUB-ROUTINE FOR NETWORK ANALYSIS)

```
/*-------------------------------------------------------------------------------------------------------------
/*
/*                              GRIDCONV
/*
/*
/*      Converts each gridded output from ADMS-Urban into a
/*      regular grid of 5 x 5m polygons
/*
/*      J. Gulliver     Created:        24th Feb 2000
/*                      Last modified:  21st Mar 2000
/*
/*-------------------------------------------------------------------------------------------------------------


/* create 5 x 5m polygon mask for grid overaly

    generate mask
    fishnet
    473500,261000
    473500,265500
    5,5
    900,600
    q

    build mask poly


/* Opens file reading in list of grid for conversion


    &s file_id [open gridlist.txt openstat -r]

    &if %openstat% ne 0 &then
      &return &error Error in creating file

    &else &type File is created

    &s line [read %file_id% readstat]

    &do &while %readstat% ne 102
        &s grid [extract 1 %line%]
        &type GRID = %grid%

        &call gridconv

        &s line [read %file_id% readstat]
    &end


    &s closestat [close %file_id%]

    kill mask all

  &return
```

```
/*-------------------------------------------------------------
  &routine gridconv
/*-------------------------------------------------------------

  grid
  gridout1 = %grid% * 10
  gridout2 = int(gridout1)
  q

  gridpoly gridout2 polygrid
  additem polygrid.pat polygrid.pat grid-code2 8 8 f 1

  &data arc info
  ARC
  SELECT POLYGRID.PAT
  CALCULATE GRID-CODE2 = GRID-CODE / 10
  Q STOP
  &end

  identity polygrid mask %grid%x poly # join

  kill gridout1 all
  kill gridout2 all
  kill polygrid all

&return
```

APPENDIX 4C

TIME-ACTIVITY SURVEYS

Northampton Air Pollution Study - NAPS	Name:

Schoolchildrens' activity questionnaire

Background:

As part of the Northampton Air Pollution Study we would like to find out about how long you spend each day in different sorts of places, and how you travel to school and other places in the town.

There are three parts to this form. The first part asks about how you spent this morning and how you travelled to school. We would like you to **complete this now.** The second part asks about how you spend the school day today. We would like you to **complete this before you go home.** The third part asks about how you spent the rest of the day, and any journeys you made during it. It will be easiest for you to complete this as you go through the day, but if you cannot do this, or forget, we would like you to **complete it at the end of the day.**

You will find that we are asking a lot of questions about when different things happened in the course of the day. If you are not quite sure about when you did a particular activity, just try to estimate the time as closely as you can.

Please answer the questions by ticking one of the boxes or writing in the spaces provided. If you do not know the answer to a question please leave the boxes blank.

To the parent or guardian:

Information in this questionnaire will be treated as strictly confidential and will be used for statistical purposes only. In the event of any problems or questions in this study, please contact Emma Livesley on (01604) 735500 ext. 2510.

Please return your completed questionnaire to your class teacher tomorrow morning.

THANK YOU FOR YOUR HELP

Part 1: This Morning

1 What time did you get up this morning? _____

2 Did anyone cook breakfast in your house this morning?

Yes ☐
No ☐

3 What time did you leave for school? _____

4 What time did you reach school? _____

5 How did you travel to school this morning?

Walk ☐
Cycle ☐
Car ☐
Bus ☐
School bus ☐
Other ☐

6 If you ticked '**other**', what form of transport did you use? _____

7 Do you **usually** travel to school in the morning in the same way?

Yes ☐
No ☐

8 If **NO**, what is your usual way of getting to school?

Walk ☐
Cycle ☐
Car ☐

Bus ☐
School bus ☐
Other ☐

Part 2: Your School Day

9 Were the windows of your classrooms open today?

Yes ☐
No ☐

10 Did you have any lessons out of doors today?

Yes ☐
No ☐

11 If **yes**, when were those lessons? _____

12 Did you leave the school at lunchtime?

Yes ☐
No ☐

13 If you did leave school:

where did you go ? _____

how did you get there ? _____

14 If you stayed at school, where did you spend the lunchtime?

Outside ☐
Inside ☐
Both ☐

15 Where did you mainly spend the morning break?

Outside ☐
Inside ☐

16 Where did you mainly spend the afternoon break?

Outside ☐
Inside ☐

Part 3: After School

Think about how you spent your time after school. On this page, we have drawn a chart, showing the hours of the day in sequence, from 3 O'clock to 12 O' clock midnight. For each hour, we would like you to mark where you spent **most** of the time in that hour, using the code below. If you spent a similar amount of time in two places, mark both in that hour of the chart, divided by a /. Here is a list of the places you may have spent time, with their code:-

B: In the bedroom at home.
L: In the living room at home.
K: In the kitchen at home.
G: In the garden at home.
S: Indoors at school.
P: Outdoors at school.
IH: Inside another home.
OH: Outdoors at another home.
I: In another indoor place (e.g. a shop, cafe, church or sports centre).
O: In another outdoor place (e.g. in a park or sports ground).

Don't include travelling from one place to another, as this will be recorded in the journey record on the next page.

Chart:

Time:

3→	4→	5→	6→	7→	8→	9→	10→	11→

Now we would like you to think about **ALL** the journeys you make today after you leave school. For **EACH** journey in turn, try to record:

- where the journey started
- where the journey ended
- when the journey started
- when the journey ended
- how you made the journey (e.g. in a car, on a bus, by walking, on a bicycle)

Then write down these details in table below, starting with the first journey from school, and ending with the last. We would like you to write down each journey separately; for example, if you went to the shops *before going home, record two separate journeys (from school to shops, from shops to home).* Even if you just went from home to the shop and back, you should record two journeys. Try to indicate where in the town you went; for example, write 'shops in the town centre' or 'local shops', rather than just 'shops'. Finally, if you used two types of travel to make a journey, record both. For example, if you got a lift in a car to the centre of town, and then took a bus, record both car and bus.

Journey number	Where you started	Where you finished	When it started	When it ended	How you made the journey
1	School				
2					
3					
4					
5					

6					
7					
8					
9					
10					

Reference:

Residents' activity questionnaire

Good evening, may I speak to [.....*person's name*.....] .

My name is [.....*interviewer's name*.....] and I'm from Nene College. I'm phoning you this evening to ask you if you could take part in a survey concemed with air pollution problems within your area of Northampton. You may have received a card notifying you of this study within the last few days.

Can you spare 5 minutes of your time to answer some questions?

Yes ☐

No ☐

Anything you tell me will be kept completely confidential and will not be passed on to any other person other than for any analysis of the data we collect.

The purpose of my asking you a number of questions is to find out where people living in the northern corridor of Northampton spend most of their time and what type of activity they are doing. That is, how long you spend in different locations each day. Also, we are trying to find out how many journeys you make within a typical day, and by what method of transport.

This information will be used to help us draw up a profile of how much air pollution people in Northampton are exposed to.

Questions will fall into two parts. The first asks for some information about your home; the second asks about all the journeys you made yesterday.

So, starting with the first section, I'm now going to ask you some questions about your home:

1 What type of house do you live in ? Is it a . . . . . [*offer the following choices*]

- Detached house ☐
- Semi-detached house ☐
- Terraced house ☐
- Flat ☐

2 Do you smoke ? [*allow them to answer*]

- Yes ☐
- No ☐

3 [*If 'yes'*]. How many cigarettes do you smoke on an average weekday inside the home ? [*allow them to answer*] _____

4 Does anyone else smoke in the house ? [*allow them to answer*]

- Yes ☐
- No ☐

5 [*If 'yes'*]. How many cigarettes do they smoke on an average weekday inside the home ? [*allow them to answer*] _____

6 Do you have any of the following cooking appliances at home ? [*offer the following choices*]

	Yes
Electric hob/rings	☐
Electric oven	☐
Microwave	☐
Gas hobs/rings	☐
Gas oven	☐

7 How many hours a week do you use them, approximately: [*Offer the choice of cooking appliance according to the answer to the previous question, offer assistance in calculating the approximate number of hours per week by working on a daily basis and multiplying by 7, if required*]

	Hours per week
Electric hob/rings	_____
Electric oven	_____
Microwave	_____
Gas hobs/rings	_____
Gas oven	_____

8 Do you open the windows or have an extractor fan on when you cook? [*allow them to answer*]

- Yes ☐
- No ☐
- Sometimes ☐

9 What type of heating do you have in your house ? [*offer the following choices*]

- None ☐
- Gas central heating ☐
- Other central heating ☐
- Gas heater ☐
- Electric heater ☐
- Other ☐

10 Do you have a garden at home?

- Yes ☐
- No ☐

Thank you.

Now I am going to ask you about the journeys you made yesterday.

I would like you to think about **all** the journeys you made yesterday. For **each** journey in turn, try to recall:

- where the journey started
- where the journey ended
- what time the journey started
- what time the journey ended
- how you made the journey

For example, possible places you may have gone to could include work, the supermarket, to you son's or daughter's school or a sports centre or cinema [*be specific about what part of town they visited or made the journey in*]. You may have walked, cycled, caught a bus or driven by car.

[*If no journeys were made then state so on the journey planner*]

Starting with your first journey . . . . . [*fill out the table below]*

Journey	Where you started	Where you ended	When it started	When it ended	How you made the journey
1					
2					
3					
4					
5					
6					
7					
8					
9					
10					
11					
12					
13					
14					

15					
16					
17					
18					
19					
20					

That completes the questions for both sections. May I remind you that this information will be kept confidential and in no way will you be able to be identified as an individual within this work.

It only remains for me to thank you for your time and I wish you a very good evening. Goodbye.

Example of journey record

Journey	Where you started	Where you ended	When you started	When it ended	How you made the journey
1	Home	Work	8.00	8.45	Walk/ bus
2	Work	Cafe	12.30	12.35	Walk
3	Cafe	Work	13.15	13.20	Walk
4	Work	Sports centre	17.30	17.50	Car
5	Sports centre	Pub	19.00	19.05	Walk
6	Pub	Home	20.00	20.45	Walk/ bus
7					
8					
9					
10					
11					

Northampton Air Pollution Study - NAPS	Reference:

Students' activity questionnaire

This questionnaire forms part of a study on sources of exposure to air pollution of residents in the northern part of Northampton being carried out by the Nene Centre for Research and other collaborating partners. Its purpose is to determine the time-activity pattern of a typical group of residents (i.e. how long you spend each day in different locations), and to determine how many journeys are made, and by what transport mode. There are three parts to the questionnaire. The first asks for some information about the characteristics of your accommodation; the second asks you to describe how you spent yesterday; and the third asks for more detailed information about all the journeys you made yesterday. All questions relate to your term-time activities in Northampton. Each question is concerned with you as an individual. It is therefore important that you answer questions in terms of your own activity pattern and not in terms of those of the people around you. If you live in a multiple occupancy residence (e.g. hall of residence) please answer the questions in part 1 in relation to your own room and any communal areas that you may use.

Information in this questionnaire will be treated as strictly confidential and will be used for statistical purposes only. Please answer the questions by ticking appropriate boxes or writing in the spaces provided. If you do not know the answer to a question please leave the boxes blank.

In the event of any problems or questions in this study, please contact Frances Machin on (01604) 735500 ext. 2512.

PLEASE RETURN THE COMPLETED QUESTIONNAIRE TO THE ASSIGNMENT OFFICE, EITHER AT PARK OR AVENUE CAMPUS, NO LATER THAN WEDNESDAY FEBRUARY 11TH. (Note: Park Campus Assignment Office is preferred and is open for longer hours.)

THANK YOU FOR YOUR HELP

Part 1 : About your accommodation.

1. What type of accommodation do you live in ?

Hall of residence	☐
Detached house	☐
Semi-detached house	☐
Terraced house	☐
Flat	☐
Other	☐

2. What is the name of the street and the full post-code of the accommodation that you live in?

_____

3. Please tick if you have any of the following cooking appliances in your accommodation and indicate approximately how many hours per week **you** use them :

	Yes	Hours per week
Electric hob/rings	☐	_____
Electric oven	☐	_____
Microwave	☐	_____
Gas hobs/rings	☐	_____
Gas oven	☐	_____

4. Do you have a living room or communal area in your accommodation?

Yes	☐
No	☐

5. If 'no', do you use your kitchen as the communal area?

Yes	☐
No	☐

6. If 'yes', on average, how many hours per week do you spend in the kitchen while other occupants are using the cooking appliances ? _____

7. Do you open the windows or have an extractor fan (venting off to the outside) on when you cook ?

Yes	☐
No	☐
Sometimes	☐

8. Do you smoke ?

Yes ☐
No ☐

9. If yes, how many cigarettes do you smoke in total on an average weekday inside your accommodation ? _____

10. How many other smokers are there in your accommodation ?
0 : ☐ 1: ☐ 2: ☐ 3: ☐ 4: ☐ more than 4: ☐

11. If yes, how many cigarettes do they smoke in total on an average weekday inside your accommodation? (please give the total for all smokers) _____

12. What types of heating do you have in your accommodation?

None ☐
Gas central heating ☐
Other central heating ☐
Gas heater ☐
Electric heater ☐
Other ☐

13. If other, please specify _____

14. Do you have a gas heater in your room?

Yes ☐
No ☐

15. If yes, is the heater on at night ?

Never ☐
Sometimes ☐

16. Do you travel between Park Campus and Avenue Campus?

Yes ☐
No ☐

17. State how many journeys you make between the two sites in the course of an average week (count an outward and return journey as two separate journeys) _____

18. Outline the normal route you would take between the two sites giving street names of outward and return journeys (if appropriate). _____

_____

19. What is the normal mode of transport you use between the two sites?

Walk
Cycle
Car
Bus
College bus
Other

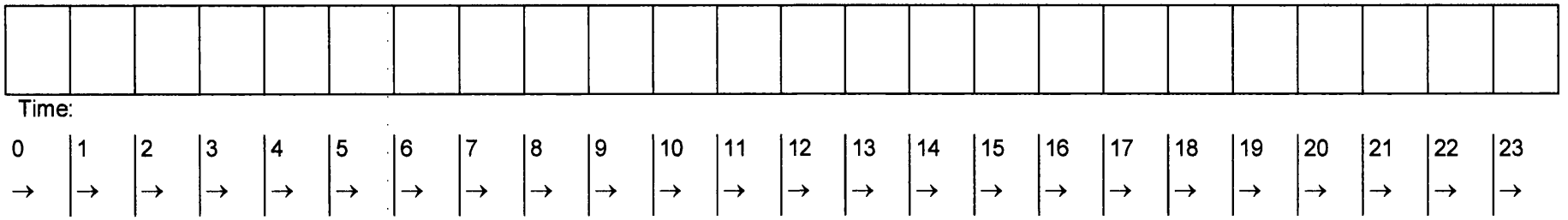
20. If 'other', please specify. _____

Part 2 : What you did yesterday.

Think about how you spend your time **yesterday**. Below we have drawn a chart showing the hours of the day in sequence. For each hour we would like you to indicate where you spent **MOST** of the time in that hour. If you spent a similar amount of time in two locations mark both in that hour of the chart divided by a '/'. We have divided locations into the following categories:-

B: In your room
L: In the living room
K: In the kitchen
G: In the garden
C: At college whilst involved in your studies (e.g. lectures, library, practical classes)
P: In a pub or college bar
H: In someone else's accommodation
I: In another indoor location (e.g. a shop, cafe, church or sports centre) outside college
O: In another outdoor location (e.g. in a park or sports ground) outside college

Chart:



Part 3 : Your journeys yesterday.

Now we would like you to think about **all** the journeys you made yesterday. For **each** journey in turn, try to recall:

- where the journey started
- where the journey ended
- when the journey started
- how long the journey took (or when it ended?)
- how you made the journey (e.g. in a car, on a bus, by walking, on a bicycle, on a motorbike)

Then write down these details in table below, starting with the first journey of the day, and ending with the last. We appreciate that you may not wish to tell us exactly where you went, but we would like enough information to locate it on a map of the town (e.g. the street, or at least the general area). You may also not be sure exactly when you started the journey, but we would like you to estimate it at least to the nearest 15 minutes. We would like you to record each journey separately; for example, if you went from college to the supermarket and then went to the pub before going home, record three separate journeys - from college to the supermarket, from the supermarket to the pub and from the pub to home. Even if you just went from home to college and back, you should record two journeys. It is important to remember to record every journey that you may have done during the day, especially if you travel between the two campus sites. Finally, if you used two travel 'modes' to make the journey, record both. For example, if you walked to the bus stop, and then took a bus, record both.

Many thanks for your time.

Once again, we remind you that the information given here is completely confidential and in no way will you be able to be identified as an individual within this study by any outside body.

Yesterday's date: _____

Journey	**Where you started**	**Where you ended**	**When you started**	**When you ended**	**How you made the journey**
1					
2					
3					
4					
5					
6					
7					
8					
9					
10					
11					
12					
13					
14					
15					
16					
17					
18					
19					
20					

Example of journey record

Journey	Where you started	Where you ended	When you started	When it ended	How you made the journey
1	Home	Work	8.00	8.45	Walk/ bus
2	Work	Cafe	12.30	12.35	Walk
3	Cafe	Work	13.15	13.20	Walk
4	Work	Sports centre	17.30	17.50	Car
5	Sports centre	Pub	19.00	19.05	Walk
6	Pub	Home	20.00	20.45	Walk/ bus
7					
8					

Example of time-activity record

Chart:

B	B	B	B	B	B	B	B/K	O	W	W	W	W	I/O	W	W	W	W	P	K/L	L	L	L	B

Time:

0 →	1 →	2 →	3 →	4 →	5 →	6 →	7 →	8 →	9 →	10 →	11 →	12 →	13 →	14 →	15 →	16 →	17 →	18 →	19 →	20 →	21 →	22 →	23 →

APPENDIX 4D

TECHNICAL DESCRIPTION OF OSIRIS

The OSIRIS (shown below) is a light -weight, portable monitor for measuring airborne particles.

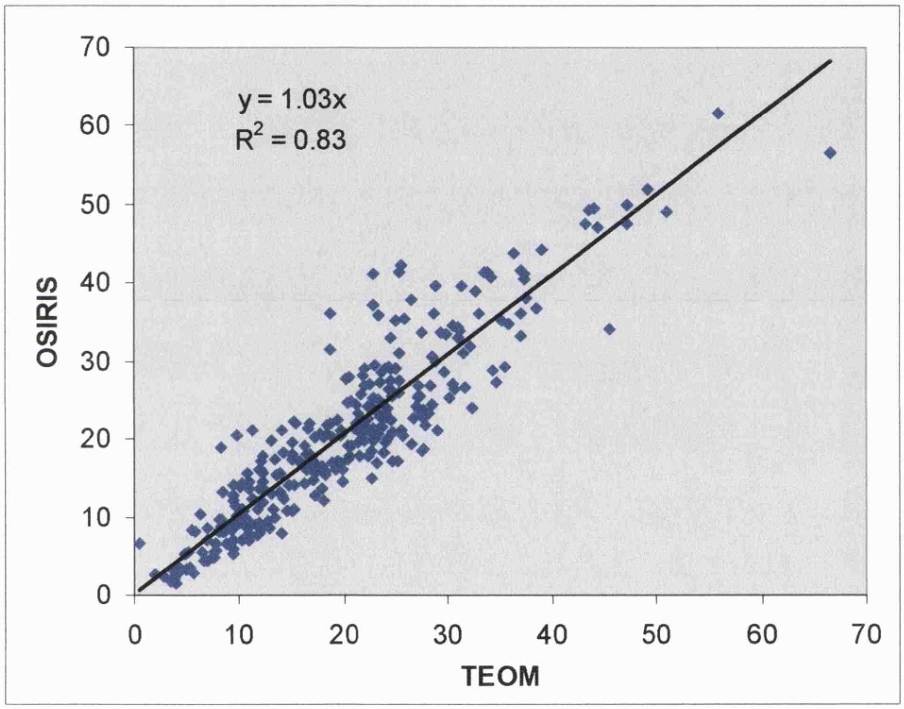
The OSIRIS uses light scattering techniques to determine the concentration of airborne particles and dust in the size range from about 0.4 microns to about 20 microns in diameter. Above 20 microns, all particles are sized as 20 microns. The air is continuously drawn into the instrument by a pump at 10 cc per second and the instrument sizes individual particles as they pass through a laser beam. Over 20,000 particles a second can be sized before coincidence occurs (two particles in the beam at once). This is approximately equivalent to a particle concentration of 6000 $\mu g/m^3$. The light scattered by the individual particles of dust is converted into an electrical pulse, which is proportional to the size of the particle. The intensity of the light pulse is therefore an indicator of particle size and from this the microprocessor calculates the expected mass of the particle. The instrument is fitted with a TSP (total suspended particles) inlet and therefore records the particle concentration during sampling based on all the particles. In addition, the instrument can be set to measure and record PM_{10}, $PM_{2.5}$ and PM_1 particles electronically selected from the those particles entering the inlet, thus they meet the sensitivity requirements of DETR guidelines and European Directives. The instrument stores a concentration in $\mu g/m^3$ for averaging periods in the range of 1 second to 60 minutes. OSIRIS can be deployed both as an indoor or outdoor

monitor to study short, or long-term. For outdoor applications, the *Turnkey Lamp Post Box* is used to house the OSIRIS, mainly as a weather shield. For the purposes of this study, a special leather carry case was created to allow the unit to be used as a personal monitor. For personal monitoring applications, the unit is powered by a 12 volt power cell, carried in a pouch attached to a trouser belt. Data from the OSIRIS can be downloaded using Turnkeys' own *AirQ for Windows* software. Data can be viewed and analysed in *AirQ,* or exported for processing in *Microsoft Excel.*

APPENDIX 4E

CO-LOCATION OF OSIRIS AND TEOM

Co-location of the OSIRIS and TEOM units was undertaken to study the difference in concentration between the different monitoring techniques. For practical reasons, the OSIRIS was mounted to the cage surrounding the TEOM inlet on the roof of the mobile. This was deemed to be at sufficient distance not to interfere with the normal volume of air drawn by the TEOM pump. Co-location was carried out between 18th February and 2nd March 2000 [n = 302]. An averaging period of fifteen minutes was used. The relationship between OSIRIS and TEOM is almost 1:1, and, as the scatter-plot below shows, TEOM and OSIRIS are broadly yielding the same results.



Based on this analysis, the following equation was applied to adjust the TEOM concentration to an OSIRIS eqiuvalent:

OSIRIS = 1.03 * TEOM

REFERENCES

Adams, H.S., Nieuwenhuijsen, M.J., Colvile, R.N., McMullen, M.A.S., Khandelwal, P., 2001, Fine particle ($PM_{2.5}$) exposure levels in transport microenvironments, London, UK. *The Science of the Total Environment* 279, 29-44.

AEA technology: http://www.aeat.co.uk/netcen/airqual/

Akbar, S., Ashmore, M.R., 1996, Particulate air pollution and respiratory morbidity: personal exposure in Dehli and its implications. *Paper presented at the World Congress on Air Pollution in Developing Countries, San Jose, Costa Rica, 21-26 October, 1996.*

Anderson, H.R., 2000, Differential epidemiology of ambient aerosols. In: Brown, L.M., Collings, N., Harrison, R., Maynard, A.D., Maynard, R.L., *Ultrafine particles in the atmosphere, Philosophical Transactions of the Royal Society* 358, 2771-2785.

APEG, UK Airborne Particles Expert Group, 1999, Source apportionment of airborne particulate matter in the UK. Report for the Department of Transport, Environment and the Regions in partnership with the Welsh Office and the Department of the Environment (Northern Ireland).

Ashmore, M. R., Batty, K., Machin, F., Gulliver, J., Grossinho, A., Elliot, P., Tate, J., Bell, M., Livesley, E., Briggs, D.J., 2000, Effects of traffic management and transport mode on the exposure of schoolchildren to carbon monoxide. *Environmental Monitoring and Assessment* 65, 49-57.

Baek, S.-O., Kim, Y.-S., Perry, R, 1997, Indoor air quality in homes, offices and restaurants in Korean urban areas - indoor/outdoor relationships. *Atmospheric Environment* 31 (4), 529-544.

BBC 1978 The people's activities and use of time. *London: BBC Audience Research Department.*

BBC Survey, 1989, Daily life in the 1980s. *General activities* Vol. 2.

Bennet, M., Hunter, G.C., 1997, Some comparisons of lidar estimates of peak, ground-level concentrations with predictions of UK-ADMS. *Atmospheric Environment* 31(3), 429-439.

Benson, P.E., 1992, Review of the development and application of the CALINE3 and CALINE4 Models. *Atmospheric Environment* 26B(3), 379-90.

Beuthe, M., Jourquin, B., Geerts, J.F., Koul à Ndjang Ha, C., 2001, Freight transportation demand elasticities: a geographical multimodal transportation network analysis. *Transport Research Part E: Logistics and Transportation Review* 37(4), 253-266.

Bevan, M.A.J., Proctor, C.J., Baker-Rogers, J., Warren, N.D., 1991, Exposure to carbon monoxide, respirable suspended particulates, and volatile organic compounds whilst commuting be bicycle. *Environmental Science and Technology* 25, 788-791.

Blanchard Ch.L., Carr, E.L., Collins, J.F., Smith T.B., Lehrman, D.E., Michaels, H.M., 1999, Spatial representativeness and scales of transport during the 1995 integrated monitoring study in California's San Joaquin Valley. *Atmospheric Environment* 33, (29), 4775-4786.

Blank, S., Seiter, C., Bruce, P., 1999, *Resampling Stats in Excel*. Resampling Stats Inc, Virginia, USA.

Boudet, C., Zmirou, D., Vestri, V., 2000, Can one use ambient air concentration data to estimate personal and population exposures to particles? An approach within the European EXPOLIS study. *The Science of the Total Environment* 267, 141-150.

Brainard, J.S., Lovett, A.A, Parfitt, J.P., 1996, Assessing hazardous waste risks using GIS. *International Journal of GIS* 10 (7), 831-849.

Briggs, D.J., Ashmore, M., Bell, M., Elliott, P, 1998, Coming up for air. *The Surveyor*, October 1, 22-24.

Briggs, D.J., Collins, S., Elliott, P., Kingham, S., Fisher, P., Lebret, W., van Reeuwijk, H., van der Veen, A., Pryl, K., Smallbone, K. 1997 Mapping urban air pollution using GIS: a regression-based approach. *International Journal of Geographical Information Science* 11 (7), 699-718.

Briggs, D.J., de Hoogh, C., Gulliver, J., Wills, J., Elliott, P., Kingham, S., Smallbone, K., 2000, A regression-based method for mapping traffic-related air pollution: application and testing in four contrasting urban environments. *Science of the Total Environment* 253, 151-67.

Briggs, D, J., Elliott, P., 1995, The use of geographical information systems in studies on environment and health. *World health statistics quarterly* 48, 85-94.

Brunekreef, B., Janssen, N.A.H., de Hartog, J., Harssema, H., Knape, M., van Vliet, P. 1997 Air pollution from truck traffic and lung function in children living near motorways. *Epidemiology* 8(3), 298-303.

Burnett, R.T., Dales, R.E., Krewski, D., Vincent, R., Dann, T., Brook, J., 1995, Associations between ambient particulate sulfate and admissions to Ontario hospitals for cardiac and respiratory diseases. *American Journal of Epidemiology* 142, 15-22.

Burnett, R.T., Dales, R.E., Raizenne, M.E., Krewski, D., Summers, P.W., Roberts, G.R., Raad-Young, M., Dann, T., Brook, J., 1994, Effects of low ambient levels of ozone and sulfates on the frequency of respiratory admissions to Ontario hospitals. *Environmental Research* 65, 172-194.

Burnett, R.T., Cakmak, S., Brook, J.R., Krewski, D., 1997, The role of particulate size and chemistry in the association between summertime ambient air pollution and hospitalization for cardiorespiratory diseases. *Environmental Health Perspectiveds* 105, 614-620.

Burrows, R., Crowder, G.S., Zhang, J., 2000, Utilisation of network modelling in the operation of water distribution systems. *Urban Water* 2(2), 83-95.

Burton, R.M., Suh, H.H., Koutrakis, P., 1996, Spatial variation in particulate concentrations within metropolitan Philadelphia. *Environmental Science and Technology* 30 (2), 400-407.

Carruthers, D.J., Holroyd, R.L., Hunt, J.R.C., 1994, UK-ADMS: A new appraoch for modelling dispersion in earth's boundary layer. *J Wind Eng* 52, 139-153

Carruthers, D.J., Dixon, P., McHugh, C.A., Nixon, S.G., Oates, W., 1999, Determination of compliance with UK and EU air quality objectives from high resolution pollutant concentration maps calculated using ADMS-Urban. In *Proceedings of the Sixth International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes. October 11-14th 1999, INSA de Rouen, France* (on CDRom).

Carruthers, D.J., Edmunds, H.A., Lester A.E., McHugh, C.A., Singles, R.A., 2000, Use and validation of ADMS-Urban in contrasting urban and industrial environments. *International Journal of Environment and Polllution* 14, 1-6.

CERC Ltd, 1999, ADMS-Urban. An urban air quality management system. User guide. Cambridge: CERC Ltd.

Chan, C.C. and Wu, H.W.Y. 1993 A study of bus commuter and pedestrian exposure to traffic-related air pollution in Hong Kong. *Environment International* 19, 121-132.

Ciccone, G., Forastiere, F., Agabati, N., Biggeri, A., Bisanti, L., Chellini, E., Corbo, G., Dell'Orco, V., Dalmasso, P., Volante, T.F., Galassi, C., Piffer, G., Renzoni, E., Rusconi, F., Sestini, P., Viegi, G. and the SIDRIA collaborative group, 1998, Road traffic and adverse respiratory effects in children. *Occupational Environmental Medicine*, 55, 771-8.

Cimorelli, A.J., Perry, S.G., Venkatram, A., Weil, J.C., Paine, R.J., Wilson, R.B., Lee, R.F., Peters, W.D., 1998, AERMOD: Description of Model Formulation. (12/15/98 Draft Document) Prepared for Environmental Protection Agency, Research Triangle Park, NC. 113pp. (Docket No. A-99-05; II-A-1).

Clarke, A.G., Willison, M.J., Zeki, E.M., 1984, A comparison of urban and rural aerosol composition using dichotomous samplers. *Atmospheric Environment* 18, 1767-1775.

Clench-Aas, J., Bartonova, A., Bøhler, T., Grønskei, K.E., Sivertsen, B., Larssen, S., 1999, Air pollution monitoring and estimating. Part 1. Integrated air quality monitoring system. *Journal of Environmental Monitoring* 1, 313-319.

Collins, S., 1998, Modelling spatial variations in air quality using GIS, in: Gatrell, A. C., Löytönen, M. (Eds), *GIS and Health.* (London, Taylor & Francis Ltd) 81-95.

Colome, S.D., Kado, N.Y., Jaques, P., Kleinmann, M, 1990, Indoor-outdoor relationships of particles less than 10μm in aerodynamic diameter (PM_{10}) in homes of asthmatics. In: *Indoor Air '90: Proceedings of the 5th International Conference on Indoor Air Quality and Climate*, 2, 275-280. Ottawa, Canada Mortgage and Housing Corp.

COMEAP (Committee on the Medical Effects of Air Pollutants), Department of Health, 1995, *Non- Biological Particles and Health.* London: HMSO.

COMEAP (Committee on the Medical Effects of Air Pollutants, 1998, Quantification of the effects of air pollution on health in Great Britain. Department of Health Committee on the Medical Effects of Air Pollutants. The Stationary Office. ISBN 0-11-322102-9

Corvalán, C., Briggs, D., Kjellström, T., 1996, Development of environmental health indicators, in: Briggs, D. J., Corvalán, C., Nurminen M. (Eds) *Linkage methods for environment and health analysis: general guidelines. A report of the Health and Environment Analysis for Decision-making Project (HEADLAMP).* UNEP, ESEPA & WHO. Geneva: Office of Global and Integrated Environmental Health, WHO. 19-54.

de Hoogh, C.,1999, Estimating exposure to traffic-related pollution within a GIS environment. Unpublished PhD thesis, University of Leicester.

Delfino, R.J., Murphy-Moulton, A.M., Burnett, R.T., Brook, J.R., Backlake, M.R., 1997, Effects of air pollution on emergency room visits for respiratory illness in Montreal, Quebec. *American Journal of Respiratory and Critical Care Medicine* 155, 568-576.

Department of Transport, 1994, *Design manual for roads and bridges, Vol 11: Environmental assessment, Section 3, Part 1 – Air Quality*, HMSO, London.

Department of Environment, Transport and the Regions (DETR), 1997, The UK national air quality strategy. HMSO, London.

Department of Environment, Transport and the Regions (DETR), 1998, *The Transport White Paper (A new deal for transport: better for everyone).* HMSO, London

Department of Environment, Transport and the Regions (DETR), 1999, *Review of the United Kingdom national air quality strategy.* HMSO, London.

Department of Environment, Transport and the Regions (DETR), 2000a, Transport trends: 2000 edition. HMSO. London.

Department of Environment, Transport and the Regions (DETR), Expert Panel on Air Quality Standards, 2000b, *Airborne particles: what is the appropriate measurement on which to base a standard?* HMSO, London.

Department for Transport, Local Government and the Regions (DTLR), 2001, Key transport facts 1980-1989: http://www.dtlr.gov.uk/itwp/.

Derekanaris, G., Garofalakis, J., Makris, C., Prentas, J., Sioutas, S., Tsakalidis, A., 2001, Integrating GIS, GPS and GSm technologies for the effective management of ambulances. *Computers, Environment and Urban Systems* 25(3), 267-278.

Dimitroulopoulou, C., Ashmore. M.R., Byrne, M.A., Kinnersley, R.P., 2001, Modelling of indoor exposure to nitrogen dioxide in the UK. *Atmospheric Environment* 35, 269-271.

Dockery, D.W., Pope, C.A.III, Xu, X., Spengler, J.D., Ware, J.H., Fay, M.E., Ferris, B.G.Jr., Speizer, F.E, 1993, An association between air pollution and mortality in six U.S. cities. *Journal of Medicine* 329(24), 1753-1759.

Dörre, W.H, 1997, Time-activity-patterns of some selected small groups as a basis for exposure estimation : a methodological study. *Journal of Exposure Analysis and Environmental Epidemiology* 7(4), 471-91.

Duan, N., 1982, Models for human exposure to air pollution. *Environment International* 8, 305-309.

Duhme, H., Weiland, S.K., Keil, U., Kraemer, B., Schmid, M., Stender, M., Chambless, L, 1996, The association between self-reported symptoms of asthma and allergic rhinitis and self-reported traffic density on street of residence in adolescents. *Epidemiology* 7 (6), 578-82.

Edwards, J., Walters, S., Griffiths, R.K, 1994, Hospital Admissions for Asthma in Preschool Children: Relationship to Major Roads in Birmingham, United Kingdom. *Archives of Environmental Health* 49 (4), 223-227.

Efron, B. 1983, Bootstrap Methods: another look at the jacknife. *The Annals of Statistics* 7, 1-26.

Efroymson, R.A., Murphy, D.L., 2001, Ecological risk assessment of multi-media hazardous air pollutants: estimating exposure and effects. Science of the Total Environment 274, 219-230.

Eerens, H.C., Sliggers, C.J., van den Hout, K.D., 1993, The CAR model: the Dutch method to determine city street technology. *Atmospheric Environment* 27B (4), 389-399.

Farrow, A., Taylor, H., Golding, J, 1997, Time spent in the home by different family members. *Environmental* Technology 18, 605-14.

Field, R.W., Smith, B.J., Brus, C.P., Lynch, C.F. Neuberger, J.S., Steck, D.J., 1998, Retrospective temporal and spatial mobility of adult Iowa women. *Risk Analysis* 18 (5), 575-584.

Fischer, P.H., Heok, G., van Reeuwijk, H., Briggs, D.J., Lebret, E., van Wijnen, J.H., Kingham, S., Elliott, P.E., 2000, Traffic-related differences in outdoor and indoor concentrations of particles and volatile organic compounds in Amsterdam. *Atmospheric Environment* 34 3713-3722.

Francis, E.A. 1986 Patterns of building occupancy for the general public. *NRPB Memorandum M-129.*

Freijer, J.I., Bloemen, H.J.Th., de Loos, S., Marra, M., Rombout, P.J.A., Steentjes, G.M., van Veen, M.P., 1998, Modelling exposure of the Dutch population to air pollution. *Journal of Hazardous Materials* 61, 107-114.

Fruin, S.A., St. Denis, M.J., Winer, A.M., Colome, S.D., Lurman, F, 2001, Reductions in human benzene exposure in the California South Coast Air Basin. *Atmospheric Environment* 35, 1069-1077.

Funasaka, K., Miyazaki, T., Tsuruho, K., Tamura, K., Mizuno, T., Kuroda, K., 2000, Relationship between indoor and outdoor carbonaceous particulates in roadside households. *Environmental Pollution* 110, 127-134.

Giess, P., 1998, Respirable particulates and oxides of nitrogen measured inside a building alongside a busy road. *Indoor Built Environment* 7, 308-314.

Goodchild, M.F., 1998, Geographic information systems and disaggregate transportation modeling. *Geographical Systems* 5(1–2), 19–44 [294].

Goodchild, M., Haining, R., Wise, S., 1991, Integrating GIS and spatial data analysis: problems and possibilities. *International Journal of Geographical Information Systems* 6, 407-23.

Gulliver, J., Briggs, D.J., 2001, STEMS: a time-activity based exposure model for traffic-related pollution. *Thirteenth Conference of the International Society for Environmental Epidemiology*, Sept 2-8th, 2001, Garmisch-Paternkirchen.

Hall, J., 1996, Assessing the health effects of air pollution. *Atmospheric Environment* 30 (5), 743-746.

Hanna, S.R., Egan, B.A., Purdum, J., Wagler, J, 1999, Evaluation of the ADMS, AERMOD, and ISC3 dispersion models with the Optex, Duke Forest, Kincaid, Indianapolis and Lovett field data sets. In *Proceedings of the Sixth International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes. October 11-14th 1999,* INSA de Rouen, France, 1999 (on CDRom).

Hanna, S.R., 1988, Air quality model evaluation and uncertainty. *Journal of Air Pollution Control and Assessment* 38, 406-412.

HAPEM: www.nescaum.org/committees/MSPeerReview/Sec_5_HAP.html

Harrison, R. M. (ed), 1990, Pollution: causes, effects and control. (Cambridge, Royal Society of Chemistry) 201-219.

Harrison, R.M., Deacon, A.R., 1998, Spatial correlation of automatic air quality monitoring at urban background sites: implications for network design. *Environmental Technology* 19 (2), 121-132.

Harrison, R.M, Deacon, A.R., Jones, M., Appleby, R.S., 1997, Sources affecting concentrations of PM_{10} and $PM_{2.5}$ particulate matter in Birmingham (U.K.). *Atmospheric Environment* 31 (24), 4103-4117.

Hinds, W. C., 1982, *Aerosol Technology: Properties, Behaviour and Measurement of Airborne Particles.* John Wiley & Sons, New York.

Hoek, G., Brunekreef, B., 1994, Effects of low level winter pollution concentrations on respiratory health of Dutch children. *Environmental Research* 48, 328-335.

Hong, Y.C., Leem, J.H., Ha, E.H., 1999, Air pollution and daily mortality in Ichon, Korea. *Journal of Korean Medical Science* 14(3), 239-244.

Human Exposure Model (HEM):
www.chemicalawarenees.org/toolkit//php/onepagers/hem.html
www.nap.edu/openbook/030904894X/html/375.html

Janssen, N.A.H., Hoek, G., Brunekreef, B., Harssema, H., Mendink, I., Zuidhof, A., 1998, Personal sampling of particles in adults: relation among personal, indoor and outdoor air concentrations. *American Journal of Epidemiology* 147 (6), 537-547.

Janssen, N.A.H., Hoek, G., Harssema, H., Brunekreef, B, 1997, Childhood exposure to PM_{10}: relation between personal, classroom and outdoor concentrations. *Occupational and Environmental Medicine* 54, 888-894.

Jantunen, M.J., Hanninen, O., Katsoutanni, K., Knoppel, H., Keunzli, N., Lebret, E., Maroni, M., Saarela, K., Sram, R., Zmirou, D. 1998 Air pollution in European cities: the EXPOLIS study. *Journal of Exposure Assessment and Environmental Epidemiology* 8 (4), 495-518.

Jensen, S.S., Berkowicz, R., Hansen, H.S., Hertel, O., 2001, A Danish decision-support GIS tool for management of air quality and human exposures. *Transportation Research Part D* 6, 229-241

Jensen, S.S., 1998, Mapping human exposure to traffic air pollution using GIS. *Journal of Hazardous Materials* 61, 385-392

Johnson, E., 1995, Recent advances in the estimation of population exposure to mobile source pollutants. *Journal of Exposure Analysis and Environmental Epidemiology* 5 (4), 551-571.

Jones, N.C., Thornton, C.A., Mark, D., Harrison, R.M., 2000, Indoor/outdoor relationships of particulate matter in domestic homes with roadside, urban and rural locations. *Atmospheric Environment* 34, 2603-2612.

Katsouyanni, K., Karakatsani, A., Messari, I., Touloumi, G., Hatzakis, A., Kalandidi, A., Trichopoulos, D, 1990, Air pollution and cause specific mortality in Athens. *Journal of Epidemiology and Community Health* 44, 321-4.

Katsouyanni, K., Touloumi, G., Spix, C., Schwartz, J., Balducci, F., Medina, S., Rossi, G., Wojtyniak, B., Sunyer, J., Bacharova, L., Schouten, J.P., Ponka, A., Anderson, H.R., 1997, Short-term effects of ambient sulphur dioxide and particulate matter on mortality in 12 European cities: results from time series data from the APHEA project. *Air pollution and health: a European approach. British Medical Journal* 314(7095), 1658-1663.

Kieding, L.M., Rindel, A.K., Kronborg, D, 1995, Respiratory illness in children and air pollution in Copenhagen. *Archives of Environmental Health* 200-6.

Kingham, S., Briggs, D.J., Elliott, P., Lebret, E., Fischer, P., 2000, Spatial variations in concentrations of traffic-related pollutants in indoor and outdoor air in Huddersfield, England. *Atmospheric Environment* 34 (6), 905-16.

Klepeis, N.E., Nelson, W.C., Ott, W.R., Robinson, J., Tsang, A.M., Switzer, P., Bhar, J.V., Hern, S.C., Engelmann, W.H., 2001, The National Human Activity Pattern Survey (NHAPS) : a resource for assessing exposure to environmental pollutants. *Journal of Exposure Analysis and Environmental Epidemiology* 11(3), 231-252.

Kopenen, I.K., Asmi, A., Keronen, P., Puhto, K., Kulmala, M., 2001, Indoor air measurement campaign in Helsinki, Finalnd 1999 – the effect of outdoor air pollution on indoor air. *Atmospheric Environment* 35, 1465-1477.

Kukkonen, J., Härkönen, J., Walden, J., Karppinen, A., Lusa, K., 2001, Evaluation of the CAR-FMI model against measurements near a major road. *Atmospheric Environment* 35, 949-960.

Kukkonen, J., Valkonen, E., Walden, J., Koskentalo, T., Aarnio, P., Karppinen, A., Berkowicz, Kartastenpää, R, 2001, A measurement campaign in a street canyon in Helsinki and comparison of results with predictions of the OSPM model. *Atmospheric Environment* 35, 231-243.

Künzli, N., Kaiser, R., Rapp, R., Sommer, H., Wanner, H.U., Seethaler, R., Ackermann-Liebrich, U., 1997, Air pollution in Switzerland – quantification of health effects using epidemiological data. *Schweizerische Medizinische Wochenshrift*, 127(34), 1361-1370.

Künzli, N., Medina, S., Kaiser, R., Quénel, P., Horak. Jr, F., Studnicka, M., 2001, Assessment of deaths attributable to air pollution: should we use risk estimates based on time series or on cohort studies? *American Journal of Epidemiology*, 153(11), 1050-1055.

Law, P.L., Lioy, P.J., Zelenka, M.P., Huber, A.H., McCurdy, T., 1997, Evaluation of a probabilistic exposure model applied to carbon monoxide (*p*NEM/CO) using Denver personal exposure monitoring data. *Journal of the Air and Waste Management Association* 47, 491-500.

Lee, J.T., Hong Y.C., Schwartz, J., Christiani, D.C., 2000, Air pollution and daily mortality in seven major cities of Korea, 1991-1997. *Environmental Research* 84 (3), 247-254.

Leung, P-L., Harrison, R.M. 1999, Roadside and in-vehicle concentrations of monoaromatic hydrocarbons. *Atmospheric Environment* 33, 191-204.

Levy, J.I., Hammitt, J.K., Spengler, J.D., 2000, Estimating the mortality impacts of particulate matter: what can be learned from between-study variability? *Environmental Health Perspectives* 108(2), 109-117.

Linaker, C.H., Chauhan, A.J., Inskip, H., Frew, A.J., Sillence, A., Coggon, D., Holgate, T., 1996, Distribution and determinants of personal exposure to nitrogen oxide in school children. *Occupational and Environmental Medicine* 53, 200-203.

Linn, W.S., Clark, K.W., Anderson, K.R., Gong, H., 1996, Short-term particulate exposures and health changes in Los Angeles residents with chronic obstructive pulmonary disorder (COPD): in *Proceedings of the Colloquium on particulate air pollution and human health,* Park City, Utah, 3 (15).

Lioy, P.J., 1990, Assessing total human exposure to contaminants. *Environmental Science and Technology*, 24 (7), 938-945.

Lioy, P.J., Waldman, J.M., Buckley, T., Butler, J., Pierarinen, C., 1990, The personal, indoor and outdoor concentrations of PM_{10} measured in an industrial community during winter. *Atmospheric Environment*, 24B, 57-66.

Liu, L.J.S., Rossini, A.J., 1996, Use of kriging models to predict 12-hour mean ozone concentrations in Metropolitan Toronto – a pilot study. *Environment International* 22, 677-92.

Livingstone, A.E., Shaddick, G., Grundy, C., Elliott, P. 1996 Do people living near city main roads have more asthma needing treatment? Case control study. *British Medical Journal* 312, 676-677.

Loranger, S., Courchesne, Y., 1997, Health risk assessment of an industrial site contaminated with polycyclic aromatic hydrobarbons using CalTOx, an environmental fate, exposure model. *SAR and QSAR in Environmental Research*, 6(2), 81-104

Lovett, A., Parfitt, J.P., Brainard, J.S., 1997, Using GIS in risk analysis: a case study of hazardous waste transport , in *Risk Analysis: an Official Publication of the Society for Risk Analysis* 17 (5).

Mackenbach, J.P., Looman, C.W.N., Kunst, A.E, 1993, Air pollution, lagged effects of temperature, and mortality: the Netherlands. *Journal of Epidemiology and Community* Health 47, 121-6.

MacIntosh, D.L., Özkaynak, J.X.H., Spengler, J.D., Ryan, P.B., A population-based exposure model for benzene. 1995, *Journal of Exposure Analysis and Environmental Epidemiology* 5 (3), 375-403.

Mahoney, R.P., GIS and utilities. eds. Maguire, D.J., Goodchild, F., Rhind, D.W., 1991, *Geographical Information Systems: principles and applications.* Longman Scientific and Technical 2, 101-114.

Mark, D., Upton, S.L., Lyons, C.P., Appleby, R., Dyment, E.J., Griffith W.D., Fox, A.A., 1997, Personal exposure measurements of the general public to atmospheric particles. *Annals of Occupational Hygiene* 41 (1), 700-706.

McHugh, C.A., Carruthers, D.J., Higson, H., Dyster, S.J. 1999 Comparison of model evaluation methodologies with application to ADMS 3 and US models. In *Proceedings of the Sixth International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes. October 11-14th 1999,* INSA de Rouen, France, 1999 (on CDRom).

McHugh, C.A., Carruther, D.J., Edmunds, H.A., 1997, ADMS and ADMS-Urban. *International Journal of Environment and Pollution* 8, 3-6, 438-440.

McCurdy, T., 1995, Estimating human exposure to selected motor vehicle pollutants using the NEM series of models: lessons to be learned. *Journal of Exposure Analysis and Environmental Epidemiology* 5 (4), 533-551

Moolgavkar, S.H., 2000, Air pollution and daily mortality in three US cities. *Environmental Health Perspectives* 108, 777-84.

Monn, C., 2000, Exposure assessment of air pollutants: a review on spatial heterogeneity and indoor/outdoor/personal exposure to suspended particulate matter, nitrogen dioxide and ozone. *Atmospheric Environment* 35, 1-32.

Monn, C., Brandli, O., Schindler, C., Ackermann-Liebrich, U., Leuenberger, P., 1995, Personal exposure to nitrogen dioxide in Switzerland. *The Science of the Total Environment* 215, 243-251.

Monn, C., Fuchs, A., Högger, D., Junker, M., Kogelschatz, D., Roth, N. and Wanner, H.-U. 1997 Particulate matter less than 10μm (PM_{10}) and fine particles less than 2.5μm ($PM_{2.5}$): relationships between indoor, outdoor and personal concentrations. *The Science of the Total* Environment 208, 15-21.

Morandi, M.T., Stock, T.H., Contant, C.F., 1988, A comparative study of respirable particulate microenvironmental concentrations and personal exposures. *Environmental Monitoring and Assessment* 10 (2), 105-122.

Morawska, L., Congrong, H., Hitchins, J., Gilbert, D., Parappukkaran, S., 2001, The relationship between indoor and outdoor particles in the residential environment. *Atmospheric Environment* 35 3463-3473.

Morawska, L., Thomas, S., Gilbert, D., Greenaway, C., Rijnders., E., 1999, A study of the horizontal and vertical profile of submicrometer particles in relation to a busy road. *Atmospheric Environment* 33, 1261-1274.

Murakami, M., Ono, M., Tamura, K., 1990, Health problems of residents along heavy-traffic roads. *Journal of Human Ergology* 19, 101-106.

Namdeo, A.K., Colls, J.J., 1996, Development and evaluation of SBLINE, a suite of models for the prediction of pollution concentrations from vehicles in urban areas. *The Science of the Total Environment* 189/190, 311-320.

National Radiological Protection Board (NRPB), 1992, Radon effected areas, Derbyshire, Northamptonshire and Somerset. *Documents of the NRPB* 3(4), Chilton, NRPB.

Neas, L.M., Dockery, D.W., Koutrakis, P., Tollerud, D.J., Speizer, F.E., 1995, The association of ambient air pollution with twice daily peak expiratory flow rate measurements in children. *American Journal of Epidemiology* 141, 111-122.

Neas, L.M., Dockery, D.W., Burge, H., Koutrakis, P., Speizer, F.E., 1996, Fungus spores, air pollutants, and other determinants of peak expiratory flow rate in children. *American Journal of Epidemiology* 143, 797-807.

Nitta, H., Nakai, S., Maeda, K., Aoki, S., Ono, M., 1993, Respiratory health associated with exposure to automobile exhaust. I. Results of cross-sectional studies in 1979, 1982 and 1983. *Archives of Environmental Health* 48, 1, 53-58.

Norris, G., Larson, T., Koening, J., Clairborn, C., Sheppard, L., Finn, D., 2000, Asthma aggrivation, combustion and stagnant air. *Thorax* 55, 466-470.

Northampton County Council (NCC), 1997, *Transport Policy and Programme: Volume 2 – The Northampton Package*, July 1996.

Oosterlee, A., Drijver, M., Lebret, E., Brunekreef, B., 1996, Chronic respiratory symptoms in children and adults living along streets with high traffic density. *Occupational and Environmental Medicine* 53, 241-247.

Ostro, B.D., Lipsett, M.J., Wiener, M.B., Selner, J.C., 1991, Asthmatic response to airborne acid aerosols. *American Journal of Public Health* 81, 694-702.

Ostro, B.D., Sanchez, J.M, Aranda, C., Eskeland, G.S, 1996, Air pollution and mortality: results from a study of Santiago, Chile. *Journal of Exposure Analysis and Environmental Epidemiology* 6, 97-114.

Ostro, B.D., Eskeland, G.S., Sanchez, J.M, Feyzioglu, T., 1999, Air pollution and health effects: a study of medical among children in Santiago, Chile. *Environmental Health Perspectives* 107(1), 69-73.

Ott, W., 1982, Concepts of human exposure to air pollution. *Environmental International* 7, 179-196.

Ott, W., 1984, Exposure estimates based on computer generated activity patterns. *Journal of Toxicology* 21: 97

Ott, W., 1990, Total Human Exposure: Basic concepts, EPA field studies, and future research needs. *Journal of the Air and Waste Management Association.* 40 (7), 966-975

Ott, W., Thomas, J., Mage, D., Wallace, L., 1988, Validation of the simulation of human activity and pollutant exposure (SHAPE) model using paired data from the Denver, Colorado, Carbon Monoxide Field Study. *Atmospheric Environment*, 22, 2101

Ott, W., Thomas, J., Mage, D., Wallace, L., 2000, Predicting particulate exposure distributions using a random component superposition statistical model. *Journal of the Air and Waste Management Association* 50, 1390-1406.

Owen, B., Edmunds, H.A., Carruthers, D.J., Raper, D.W., 1999, Use of a new generation urban scale dispersion model to estimate the concentration of oxides of nitrogen and sulphur dioxide in a large urban area. *The Science of the Total Environment* 235, 277-291.

Ozkaynak, H., Xue, J., Spengler, J., Wallace, L., Pellizzari, E., Jenkins, P., 1996, Personal exposure to airborne particles and metals: results from the Particle Team Study in Riverside, C.A.. *Journal of Exposure Analysis and Environmental Epidemiology*, 6, 57-78.

Pasquill, F., 1974, Atmospheric diffusion. 2nd ed.(New York, Halstead Press).

Pellizari, E., Thomas, K.W., Clayton, C.A., Witmore, R.W., Shores R.C., Zelon, H.S., Peritt, R.L., *Particle total exposure assessment methodology (PTEAM):*

Riverside California Pilot Study – Volume 1 EPA/600/SR-93/050. Research Triangle Park: EPA.

Peters, A., Winchmann, H.E., Tuch, T., Heinrich, J., Heyder, J., 1997, Respiratory effects are associated with the number of ultrafine particles. *American Journal of Respiratory and Critical Care Medicine*159, 768-775.

Pönkä, A., Savela, M., Virtanen, M., 1998, Mortality and sir pollution in Helsinki. *Archives of Environmental Health* 53(4), 281-286.

Pope, A., 1986, Development of activity patterns for population exposure to ozone. *Report for PEI associates, Durham, NC.*

Pope, C.A. III, Dockery, D.W., 1992, Acute health effects of PM_{10} pollution on symptomatic and asymptomatic children. *American Review of Respiratory Disease* 145, 1123-1128.

Pope, C.A.III, Bates, D.V., Raizenne, M.E, 1995, Health effects of particulate air pollution: time for reassessment? *Environmental Health Perspectives* 103(5), 472-480.

Pope, C.A. III, Dockery, D.W., Spengler, J.D., Raizenne, M.E, 1991, Respiratory health and PM_{10} pollution: a daily time series analysis. *American Review of Respiratory Disease* 144, 668-74.

Pope, C.A. III, Hill, R.W., Villegas, G.M, 1999, Particulate air pollution and daily mortality on Utah's Wasatch Front. *Environmental Health Perspectives* 107, 567-73.

Pope, C.A.III, Schwartz, J., Ransom, M.R., 1992, Daily mortality and PM10 pollution in Utah Valley. *Archives of Environmental Health* 47, 211-217.

Praml, G., Schierl, R., 2000, Dust exposure in Munich public transportation: a comprehensive 4 year survey in buses and trams. *International Archives of Occupational and Environmental Health* 73 (3), 209-214.

Quackenboss, J.J., Lebowitz, M.D., Crutchfield, C.D, 1989, Indoor-outdoor relationships for particulate matter: exposure classifications and health effects. *Environment International* 15, 353-360.

Quality of Urban Air Review Group (QUARG), 1993, Urban air quality in the United Kingdom. *First report, Air Quality Division, Great Britain, Department of the Environment, London.*

Quality of Urban Air Review Group (QUARG), 1996, Airborne particulate matter in the United Kingdom. *Third Report of the Quality of Urban Air Review Group, QUARG, Birmingham.*

Raaschou-Nielson, O., Hertel, O., Vignati E., Berkowicz, R, Jensen, S.S., Larson, V.B., Lohse, C., Olsen, J.H., 2000, An air pollution model for use in epidemiological studies: evaluation with measured levels of nitrogen dioxide and benzene. *Journal of Exposure Analysis and Environmental Epidemiology* 10 (1), 4-14.

Raaschou-Nielson, O., Olsen, J.H., Hertel, O., Berkowicz, R., Skov, H., Hansen, A.M., Lohse, C., 1996, Exposure of Danish children to exhaust fumes. *The Science of the Total Environment* 189/190, 51-55

Reynolds, A.W., Broderick, B.M, 1999, Development of a GIS-based urban air quality modelling system for transport-related pollution. In: *Proceedings of the Sixth International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes, October 11-14 1999.* France, INSA de Rouen (on CDRom).

Rodden, J.B., Green, N.J., Messina, A.D., Bullin, J.A. 1982 Comparison of roadway pollutant dispersion models using the Texas data. *Journal of the Air Pollution Control Association* 32, 1226-1228.

Roemer, W., Hoek, G., Brunekreef, B., 1993, Effects of ambient winter air pollution on respiratory health of children with chronic respiratory symptoms. *American Review of Respiratory Disease* 147, 118-124.

Roemer, W., Hoek, G., Brunekreef, B., Clench-Aas, J., Pekkanen, J., Schutz, A., 2000, PM_{10} elemental composition and acute respiratory effects in European children (PEACE project). Pollution Effects on Asthmatic Children In Europe. *The European Respiratory Journal: Official Journal for Clinical Respiratory* Physiology, 15(3), 553-559.

Roorda-Knape, M.C., Janssen, N., De-Hartog, J.H., van-Vliet, P., Harssema, H., Brunekreef, B., 1998, Air pollution from traffic in city districts near major motorways. *Atmospheric Environment* 32 (11), 1921-1930.

Rudolf, W., 1986, Horizontal gradients in city streets. In Muller H (ed), Cooperation between The Netherlands and the Federal Republic of Germany on air pollution problems. *Measurements of automative emissions,* RIVM/UBA, 101-107

Salcedo, R.L.R., Alvim Ferraz, M.C.M., Alves, C.A., Martins, F.G., 1999, Time-series analysis of air pollution data. *Atmospheric Environment* 33, 2361-72.

Saldiva, P.H.N., Lichetenfels, A.J.F.C., Paiva, P.S.O., Barone, I.A., Martins, M.A., Massad, E., Pereira, J.C.R., Xavier, V.P., Singer, J.M., Bohm, G.M., 1994, Association between air pollution and mortality due to respiratory diseases in children in Sao Paulo, Brazil: a preliminary report. *Environmental Research* 65, 218-25.

Samet, J.M., Dominici, F., Curriero, F.C., Coursac, I., Zeger, S.L., 2000, Fine particulate air pollution in 20 US cities, 1987-1994. *The New England Journal of Medicine* 343(24), 1742-1749.

Schwartz, J., 1991, Particulate air pollution and daily mortality in Detroit. *Environmental Research* 56, 204-213.

Schwartz, J., 1993a, Particulate air pollution and chronic respiratory disease. *Environmental Research* 63, 7.

Schwartz, J., 1993b, Air pollution and daily mortality in Birmingham, Alabama. *American Journal of Epidemiology* 137, 204-213.

Schwartz, J., 1994a, What are people dying of on high air pollution days? *Environmental Research* 64, 26-35.

Schwartz, J., 1994b, Acute effects of summer air pollution on respiratory symptom reporting in children. *American Journal of Respiratory and Critical Care Medicine* 150 (5), 1234-1242.

Schwartz, J., 1999, Air pollution and hospital admissions for heart disease in eight U.S. counties. *Epidemiology* 10, 17-22.

Schwartz, J., 2000, Is there harvesting in the association of airborne particles with daily deaths and hospital admissions? *Epidemiology* 12(1), 55-61.

Schwartz, J., Dockery D.W., 1992, Particulate air pollution and daily mortality in Steubenville, Ohio. *American Journal of Epidemiology* 135, 12-19.

Schwartz, J., Dockery, D.W., Neas, L.M., 1996, Is daily mortality associated specifically with fine particles? *Journal of Air and Waste Management Association* 46, 927-939.

Schwartz, J., Slater, D., Larson, T.V., Pierson, W.E., Koenig, J.Q., 1993, Particulate air pollution and hospital emergency visits in Seattle. *American Rev. Respiratory Disease*,147, 826-831.

Schwartz, J., Spix, C., Wichmann, H.E., Malin, E., 1991, Air pollution and acute respiratory illness in five German communities. *Environmental Research* 56, 1-14.

Seaton, A., Soutar, A., Crawford, V., Elton R., McNerlan, S., Cherrie, J.W., Watt, M., Agius, R., Stout, R., 1999, Particulate air pollution and the blood. *Thorax* 54, 1027-1032.

Seika, M. 1996 Evaluation of control strategies for improving urban air quality with London and Berlin as examples. Unpublished PhD thesis, University of Birmingham.

Sengupta, S., Patil, R.S., Venkatachalam, P., 1996, Assessment of population exposure and risk zones due to air pollution using the geographical information system. *Computers, Environment and Urban Systems* 20 (3), 191-199.

Sheppard, L., Levy, D., Norris, G., Larson, T.V., Koenig, J.Q., 1999, Effects of ambient air pollution on nonelderly asthma hospital admissions in Seattle, Washington, 1987-1994. *Epidemiology* 10, 23-30.

Spengler, J.D., Dockery, D.W., Turner, W.A., Wolfson, J.M, Ferris, B.G., 1981, Long-term measurements of respirable sulfates and particles inside and outside home. *Atmospheric Environment* 15, 23-30.

Stedman, J R., 1997, The secondary particle contribution to elevated PM_{10} concentrations in the UK. AEA Technology, National Environmental Technology Centre. AEAT – 4630.

Stedmand, J.R., Linehan, E., Espanhahn, S., Conlan, B., Bush, A., Davies, T., 1998, Prediciting PM_{10} concentrations in the UK. AEA Technology, National Environmental Technology Centre. Report 20008001/007. AEAT – 2959.

Stedman, J.R., Lineham, E., Conlan, B., 2001, Receptor modelling of PM_{10} concentrations at a United Kingdom national network monitoring site in central London. *Atmospheric Environment* 35, 297-304.

Tarrason, L., Tsyro, S., 1998, Long-range transport of fine secondary particles as presently estimated by the EMEP Lagrangian model. Norwegian Meteorological Institute. Research Report no. 69. EMEP MSC-W Note 2/98.

Thurston, G.D., Ito, K., Hayes, C.G., 1994, Respiratory hospital admissions and summer haze air pollution in Toronto, Ontario. Consideration of the role of acid aerosols. *Environmental Research* 65, 271-290.

Tso, M.Y.W., Leung, J.K.C., 2000, Population dose due to natural radiation in Hong Kong. *Health Physics* 78(5), 555-558.

Turnbull, A.B., Harrison, R.M., 2000, Major component contributions to PM_{10} composition in the UK atmosphere. *Atmospheric Environment* 34, 3129-3137.

Väkevä, M., Hämeri, K., Kulmala, M., Lahdes, R., Ruuskanen, J., Litinen, T., 1999, Street level versus rooftop concentrations of submicron aerosol particles and gaseous pollutants in an urban street canyon. *Atmospheric Environment* 33, 1385-1397.

van der Zee, S.C, Hoek, G., Boezen, H.M., Schouten, J.P., van Wijnen, J.H., Brunekreef, B, 1999, Acute effects of urban air pollution on respiratory health of children with and without chronic respiratory symptoms. *Occupational Environmental Medicine* 56, 802-13.

van Wijnen, J.H., Verhoeff, A.P., Jans, H.W.A., Vanbruggen, M., 1995,The exposure of cyclists, car drivers and pedestrians to traffic-related air pollutants. *International Archives of Occupational and Environmental Health* 67 (3), 187-193.

van Vliet, D. 1982 SATURN – a modern assignment model. *Traffic Engineering and Control* 23, 578-581.

Vedal, S., Petkau, J., White, R., Blair, J., 1998, Acute effects of ambient inhalable particles in asthmatic and nonasthmatic children. *American Journal of Respiratory and Critical Care Medicine* 157 (4,1), 1234-1242.

Verhoeff, A.P., Hoek, G., Schwartz, J., van Wijnen, J.H., 1996, Air pollution and daily mortality in Amsterdam. *Epidemiology* 7(3), 225-230.

Wallace, L.A., 1996, Indoor particles: a review. *Journal of the Air and waste Management Association* 46, 98-126.

Wallace, L.A., Pellizzari, E.D., Hartwell, T.D., Davis, V., Michael, L.C., Whitman, R.W., 1989, The influence of personal activities on exposure to volatile organic compounds. *Environmental Research* 50, 37-55.

Watson, D. F., Philips, G. M., 1985, A Refinement of Inverse Distance Weighted Interpolation. *Geo-Processing* 2, 315 – 327.

Whitby, K.,1978, The physical characteristics of sulphur aerosols. *Atmospheric Environment* 12, 135.

Whitelegg, J. (ed), 1992, Traffic congestion: is there a way out?, Leading Edge Publishers.

World Health Organisation (WHO), 1999, WHO's 1999 Guidelines for Air Pollution Control. Fact sheet 187. WHO.

Wieland, S.K., Mundt, K.A., Ruckmann, A., Keil, U., 1994, Reported wheezing and allergic rhinitis in children and traffic density on streets of residents. *AIR* 4, 79-84.

Wiley, J.A., Robinson, J.P., Piazza, T., Garrett, K., Cirksena, K., Cheng, Y.T., Martin, G., 1991, Activity patterns of California residents. *Final report to the California Air Resources Board*, Sacramento, CA. Number ARB/ R-93/487.

Williams, R., Creason, J., Zweidinger, R., Watts, R., Sheldon, L., Shy, C., 2000, Indoor, outdoor, and personal exposure monitoring of particulate air pollution: the Baltimore elderly epidemiology-exposure pilot study. *Atmospheric Environment* 34, 4193-4204.

Wichmann, H., Peters, A., 2000, Epidemiological evidence of the effects of ultrafine particle exposure. In: Brown, L.M., Collings, N., Harrison, R., Maynard, A.D., Maynard, R.L., *Ultrafine particles in the atmosphere, Philosophical Transactions of the Royal Society* 358, 2751, 2769.

Wjst, M., Reitmeir, P., Dold, S., Wulff, A., Nicolai, T., von Leoffelholz-Colburg, E., von Mutius. E., 1993, Road traffic and adverse effects on respiratory health in children. *British Medical Journal* 307, 596-60.

Wordley, J., Walters, S., Ayres, J.G., 1996, Short term variations in hospital admissions and mortality and particulate air pollution. *Occupational Environmental Medicine* 54, 108-16.

Yamartino, R.J., Weigand, G. 1986. Development and evaluation of simple models for the flow, turbulence and pollutant concentration fields within an urban street canyon. *Atmospheric Environment* 20(11), 2137-56.

Yanagisawa, Y., Nishimura, H., Matsuki, H., Osaka, F., Ksuga, H., 1988, Urinary hydroproxyproline to creatinine ratio as a biological effect marker of exposure to NO_2 and tobacco smoke. *Atmospheric Environment* 22 (10), 2195-2203.

Zagury, E., Le Moullec, Y., Momas, I., 2000, Exposure of Paris taxi drivers to automobile air pollutants within their vehicles. *Occupational Environmental Medicine* 57 (6), 406-410.

Zanobetti, A., Schwartz, J., Gold, D., 2000, Are there sensitive subgroups for the effects of airbone particles? *Environmental Health Perspectives* 108(9), 841-845.

Zartarian, V.G., 2000, A modelling framework for estimating childrens' residential exsposure and dose to chlorpyrifos via dermal residue contact and non-dietary ingestion. *Environmental Health Perspectives* 108, pp 505-514