An Evaluation of Meteorological Data Needs for Urban Pollution modelling

This thesis is submitted for the degree
of
Doctor of Philosophy

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September 2001

ABSTRACT

Atmospheric dispersion models are being increasingly used by local authorities in the United Kingdom as part of their urban air quality management programmes. Output from dispersion models now forms a vital part of any environmental impact assessment, road improvement or traffic management scheme or environmental health study. This study is centred primarily on Northampton, a county town in Southern England and is concerned with the limitations of meteorological data available to local authorities.
The first part of the study investigates the variation in certain key meteorological parameters both within Northampton and between synoptic stations up to 70 kilometres away. The second part examines modelling outcomes using different sets of meteorological data and evaluates the performance of an urban dispersion model in relation to monitored air quality data. Special emphasis is placed on the use of cloud cover as a meteorological input variable. A small case study of monitoring and modelling work carried out in the London Borough of Richmond is also presented.

Word count - approximately 40,000

ACKNOWLEDGEMENTS

I would like to thank:
My supervisors, Professors John McClatchey, University College Northampton and Dr Roy Colvile, Imperial College, London.
Tina Fairless, John Gulliver, Margaret Brudenell and Paul Stroud, University College Northampton, for technical support and advice.
Joe Alfano, Environmental Health Department, Northampton Borough Council.
Keith Jinks, Horiba Instruments Ltd.
John Coates and others in the Environmental Health Department, London Borough of Richmond.
Alastair Manning, Meteorological Office.
Northampton Borough Council, Lowther Primary School, Holy Trinity Church and the London Borough of Richmond for allowing the siting of monitoring equipment on their premises.
Northampton Borough Council, London Borough of Richmond, Viasala Ltd and the British Atmospheric Data Centre for access to various databases.

This work was supported by an internal grant from University College Northampton and by Horiba Instruments Ltd.

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ABBREVIATIONS AND ACRONYMS

a^{-1}	Per annum
ADMS	Atmospheric Dispersion Modelling System
AMS	American Meteorological Society
AQMA	Air Quality Management Area
AWS	Automatic Weather Station
BADC	British Atmospheric Data Centre
BST	British Summer Time
CERC	Cambridge Environmental Research Consultants
CITAIR	Science and Research for better air in European Cities
CO	Carbon monoxide
COST	Co-operation in the field of Scientific and Technical Research
c_p	Specific heat capacity
DETR	Department of Environment, Transport and the Regions
DoT	Department of Transport
DMRB	Design Manual for Roads and Bridges
EEC	European Economic Community (forerunner of the EU)
EPA	Environmental Protection Agency
EU	European Union
$F_{\theta 0}$	Surface heat flux
g	Acceleration due to gravity
h	Boundary layer height
κ	Von Karman constant
km	Kilometre
L_{MO}	Monin Obukhov length
m	Metre
ms^{-1}	Metres per second
NAME	Nuclear Accident Response Model
NAPS	Northampton Air Pollution Study
NETCEN	National Environmental Technology Centre
NO_2	Nitrogen dioxide
NO_x	Oxides of nitrogen
NWP	Numerical Weather Prediction
O_3	Ozone
ppm	Parts per million
ppb	Parts per billion
ρ	Air density
r	Correlation coefficient
SO_2	Sulphur dioxide
T_o	Surface temperature
TVEF	Time Varying Emission Factors
u_*	Friction velocity
U	Horizontal wind speed
UCN	University College Northampton
UK	United Kingdom

UKMO	United Kingdom Meteorological Office
URL	Uniform Resource Locator
UTC	Urban Traffic Control
VOC	Volatile Organic Compounds
Wm^{-2}	Watts per square metre
Z	Universal standard time
z	Vertical height

Frontispiece – The Horiba Mobile Pollution Laboratory parked at Kingsthorpe, Northampton

CHAPTER ONE

INTRODUCTION

1.1 Background to project

This project investigates the ability of an atmospheric dispersion model to accurately predict pollution levels in an urban environment where the principal pollutant sources are traffic related. The importance of good quality data for air pollution modelling has been stressed by Oettl *et al.* (2001) and the principal concern here is the performance in relation to routinely available meteorological data. The type and quality of data used by model developers is often less readily available to model users. The increased use of dispersion modelling by local authorities in the United Kingdom is due to the growth in legislation placing control of pollution at a local level. A brief resume of how the legislation has developed is given to put this work in context.

Over the past 100 years or so there has been a shift in emphasis in the strategies for controlling air pollution and the scale on which these strategies operate. Air pollution caused by large single point sources, such as power stations, has been much reduced in the UK since the introduction of the Alkali Acts of the last century, the Clean Air Acts of 1956 and 1968 and EU directives such as The Large Combustion Plant Directive 88/609/EEC. In the UK and elsewhere national policies for controlling pollution frequently resulted in or even encouraged the dispersion of pollutants away from their point of production, often across national boundaries. These policies failed to recognise the possible adverse effect of air pollution on a global scale. During the late 1970's and 1980's the transboundary effects of air pollution, particularly acid rain, led to concern from multi-and international organisations such as the European Union, the United Nations and the World Health Organisation and resulted in the setting up of various international conventions and protocols.

Once pollution from domestic and large-scale industrial sources had been brought under legislative control, road transport became the greatest source of many air pollutants (Table 1.1). In the UK, where meteorological conditions do not lend themselves to the formation of photochemical smog, pollution from road traffic was ignored by the legislative procedure until the Road Traffic Act 1972. Subsequently there has been much European Union and national legislation relating to vehicle emissions and although vehicle numbers and the relative percentage contribution have risen, the total net contribution of pollutants from road traffic has been falling since the early 1990's. With the prediction that traffic volumes will continue to rise (Figure 1.1 and 1.2), it is expected that net emissions from this source will rise again by the year 2025 (URL1).

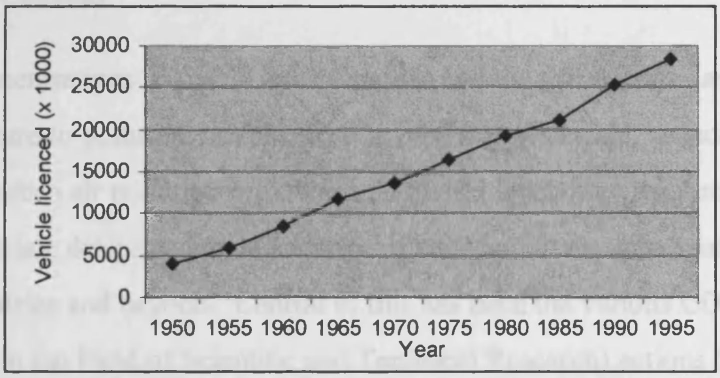


Figure 1.1 Number of vehicles holding a current licence in the UK. (DETR, 2000a)

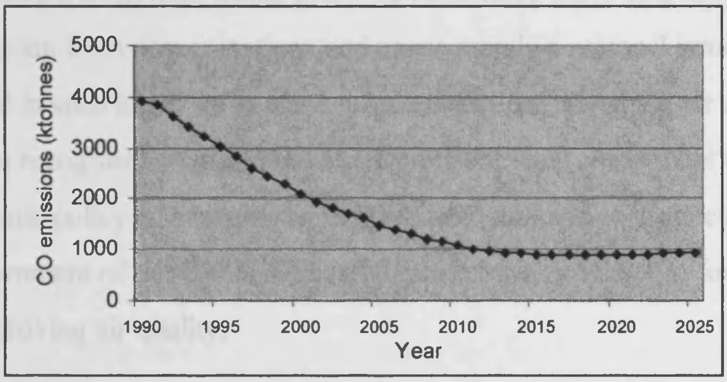


Figure 1.2 Urban UK emissions of CO per annum in kilotonnes. (URL1)

Pollutant	Percentage contribution from road transport
NO_x	47
CO	73
VOCs	38
Black smoke	55
Particulates	26

Table 1.1 Percentage contribution of five pollutants from all road transport sources - 1998. (DETR, 2000a)

The high concentrations of traffic in urban areas and the greater risks associated with human exposure to pollution has also lead, over the past decade, to increased research interest into urban air pollution problems. At an EU level there has been a concerted effort to stimulate discussion and exchange of information on these matters between member countries and beyond. Central to this has been the various COST (European Cooperation in the Field of Scientific and Technical Research) actions (see Appendix One) and the CITAIR (Science and research for better air in European cities) programme.

Although pollution from road traffic could be considered more of a ubiquitous problem than say pollution from power stations and consequently a national issue, the legislation has developed in such a way as to place responsibility for managing air quality firmly at a local level. In recognising that there was a significant local component to poor air quality and that national policy instruments are neither adequate nor cost effective in dealing with this, the government of the 1990s believed that there was a vital role for local authorities to play in improving air quality.

Building on the legislation of the Environmental Protection Act 1990, the Environment Act 1995 aimed to put in place new regulatory requirements on local authorities with regard to local air quality management. This led to the development of the UK National

Air Quality Strategy. The main aim of the strategy is to set out ambient air quality policy until the year 2005 based on a system of standards and objectives, these are given in Table 1.2

Compound	EU directive level	UK strategy levels (2000)
Benzene	5 ppb	5 ppb – running annual mean to be achieved by 1.12.2003
1,3 butadiene	-	1 ppp – running annual mean to be achieved by 1.12.2003
Carbon monoxide	10 ppm – running eight hour mean to be achieved by Jan 2005	10 ppm – running eight hour mean t.b.a. by Dec 2005
Lead	0.5 μgm^{-3} - annual mean to be achieved by Jan 2005	0.5 μgm^{-3} - annual mean to be achieved by Dec 2005
Nitrogen dioxide	105 ppb - one hour mean * 21 ppb - annual mean t.b.a. by Jan 2010	105 ppb - one hour mean * 21 ppb - annual mean t.b.a. by Dec 2005
Particulates	50 μgm^{-3} – 24 hour mean * 40 μgm^{-3} – annual mean t.b.a. by Jan 2010	50 μgm^{-3} – 24 hour mean * 40 μgm^{-3} – annual mean t.b.a. by Dec 2004
Sulphur dioxide	132 ppb - one hour mean* 47ppb – 24 hour mean t.b.a. by Dec 2004	132 ppb -one hour mean* 47ppb – 24 hour mean t.b.a. by Dec 2004

*Table 1.2 Summary of 2000 air quality strategy requirements – (DETR, 2000b) - * a certain number of exceedences are allowed per year for these pollutants.*

The strategy requires local authorities to carry out a periodic review and assessment of air quality and to set up Air Quality Management Areas (AQMA) where quality objectives are unlikely to be met.

In order to carry out their duties of assessing air quality a local authority has two main courses of action open to it, either monitoring or mathematical modelling. In reality, a combination of the two is most likely. Generally monitoring is carried out to provide the local authority with a picture of long-term trends and to cover specific known problem areas, but it provides poor spatial coverage and is inflexible. For local authorities to comply with the National Air Quality Strategy there are many aspects of their air quality assessment/management programmes for which there is no alternative to modelling. This is particularly the case when forecasting is required or new traffic management scenarios are to be investigated.

Modelling provides greater flexibility, but is not without its own particular set of problems. Let alone the question of model reliability, accuracy and quality of the predicted data, there are issues concerning model selection, the cost of model purchase or consultant fees and the problems of acquiring suitable data to run the model. These data consist principally of an emission source inventory that includes traffic flow data and meteorological data. Although most local authorities already gather traffic data for other purposes, local urban meteorological data are rarely available. The network of synoptic weather stations that exists in the UK was principally set up to aid aviation not to support air quality management. The few weather stations that are to be found near large urban area are usually at airports. The impact on modelling outcomes of using meteorological data gathered at some distance and over different terrain is the subject of this thesis.

The discussion so far has concentrated on how models may be used by local authorities to comply with legislation. Although this is the focus of this particular study, it is useful to also consider some of the other uses to which models are put and why the use of local meteorological data may be important. Other uses for dispersion modelling include the production of baseline and predicted concentration fields as part of an Environmental Impact Assessment, epidemiological studies, investigations into pollution episodes, source apportionment, general air quality forecasting and real-time traffic management schemes. In some of these cases it may be possible to use long-term meteorological data that has been classified by pre-defined sets of conditions into statistical frequencies. From this it is possible to calculate average annual mean and percentile concentration levels and to derive likely concentrations for shorter time periods.

For some forms of air quality assessment where 'worst case' scenarios are required, the use of statistical data may be appropriate. Statistical data may average out some of the natural variation that would be found between the modelling and the meteorological data site and lack of local data may present less of a problem. Where a definite bias is known to exist, for example between a coastal and inland site or between an urban and rural site, this can be taken into account. Interpretation of past pollution episodes or epidemiological studies investigating high incidences of respiratory disease are more likely to require the model to produce short-term average concentrations and as such require sequential meteorological data gathered on an hourly basis. It is more crucial in these cases for the data to be appropriate to the modelling domain. Predicted short-term averages often correlate less well with observed values than do long term averages. Uncertainties in both the meteorological and the emission inputs frequently result in errors in model output with the same magnitude as the predicted value (Häggkvist, 1997).

Although statistical data can be used in some circumstances, this is declining for a number of reasons. Increase in computing power has made it easier to run long-term sequential data to obtain the same output parameters. Davies and Thomson (1997) have shown that statistical data and long runs of sequential data produce very little difference in model output, but in their case the hypothetical modelling exercise was carried out in a rural area with the emission source close to the meteorological station. The increased complexity of models and nature of modelling problems make statistical data less appropriate. The number of meteorological parameters used by complex models result in a rapid increase in the number of categories used in the statistical classification.

The spatial and temporal variability of the meteorology is important in many modelling scenarios and this is not taken into account when statistical data are used (URL 2). For these reasons a study into the use of meteorological data in urban modelling using hourly sequential data is particularly appropriate.

1.2 Aims of the project

The main aim of the project is to investigate the performance of an atmospheric dispersion model in an urban environment and with particular reference to quality of the meteorological input data. A subsidiary aim is to investigate the spatial variability in key meteorological variables.

1.3 Objectives

The broad aims of the project will be achieved through the following objectives. These are to

- determine the sensitivity of a urban atmospheric dispersion model to key meteorological variables
- determine the spatial variability of meteorological variables within an urban environment and within an area of approximately 110 x 210 km in southern England.
- assess the performance of the model in relation to the meteorological input data.
- determine the degree of error likely to be encountered if inappropriate data are used.
- investigate the relationship between cloud cover and global radiation and determine its importance to modelling outcomes.
- to investigate different sources of background concentration data
- test the performance of a model under the conditions in which it would be routinely used.

CHAPTER TWO

LITERATURE REVIEW

2.1 Introduction

An increasing number of local authorities are using dispersion modelling to predict pollution levels in urban areas as part of their urban air quality management programmes. Although dispersion models are used in conjunction with pollution monitoring as a means of assessing air quality, for reasons of cost and flexibility, there is a trend towards using modelling as an alternative to monitoring. When model predictions are used to aid decision making processes and in the support of laws and regulations designed to protect air quality, confidence in these predictions is vital because of the large cost of implementing policy decisions based on them (Russell, 1988). Although models have evolved as the most practical and scientifically reliable means of relating source emissions to pollutant concentrations, there are some fundamental problems that need to be addressed before they are adopted for widespread use. Some of these relate to modelling *per se,* but others are particular to atmospheric dispersion modelling alone. These two issues will be dealt with in turn. The first part of this chapter reviews work relating to some of the more general problems although confined to the field of air quality modelling, whilst the later part reviews work specifically related to the urban environment, meteorology and one particular model – ADMS.

2.2 Modelling capabilities

Dispersion models can be used in a variety of ways and there is now a vast range, each designed to fulfil a specific role. In discussing the problems of modelling many workers in the field of atmospheric pollution (Zanetti, 1990; Olesen, 1995a; Dennis *et al.,* 1996) make the distinction between the 'scientific' use of models, where they are used as an aid in the understanding of the physical and chemical processes pertinent to an environmental problem and the use of models for regulatory purposes.

The relative merits of using monitoring or modelling to determine the state of the environment is a separate issue and not covered here, but it is to some extent fundamental to the question of how good or reliable a particular model is. Zanetti (1990) stresses the importance of remembering that both are only tools that can be used to provide useful information for developing control strategies and that they do not in themselves present solutions to environmental problems. Monitoring and modelling can be used to provide useful information in a relatively inexpensive way to guide the implementation of more costly emission reduction and control programmes. Some (for example, Jones *et al.*, 2000) would suggest that a combination of the two is the best way to tackle poor air quality. However Zanetti (1990) also suggests that only a 'well tested and well calibrated' model with good spatial and temporal resolution can give a good representation of the 'real world'. Monitored data can have good temporal resolution, but will only be representative of a very narrowly defined location, this in itself creates problems for model evaluation.

The inherent problems of mathematical modelling can be broken down broadly into the following topics; the complexity versus the simplicity of the model and how this affects the quality of its output, the resolution of the model in both temporal and spatial scales, the sources of error and uncertainty in model output and hence its accuracy and reliability, and finally the problem of recognising and defining the limits of applicability. Many of these more theoretical problems apply to modelling in general, but this analysis is confined to work relating to atmospheric dispersion.

2.2.1 Modelling frameworks

Models come in a variety of forms, conceptual, physical or mathematical. Physical models consisting of small-scale laboratory representations of the real world, usually in the form of a wind tunnel, they have their uses in investigating dispersion phenomena often at a small scale. Flow through street canyons, street intersections or around individual buildings is often studied, but this tends to be more related to research rather than regulation. Research of this nature has been carried out by Kastner-Klein *et al.* (1997), Wichmann-Fiebig *et al.* (1997) and Leitl and Meroney (1997).

Mathematical models consist of a set of analytical or numerical algorithms that describe the physical and chemical aspects of the real world. They can be either empirical or deterministic. Empirical or statistical models relate observed air quality to the accompanying emission patterns and environmental variables, but they only include chemistry and meteorology implicitly. Examples of empirical models that are designed for use with the urban environment include the CAR (Eerens *et al.,* 1994), CAR-SMOG (den Tonkelaar, 1996 and den Boeft *et al.,* 1996) and the SAVIAH (Briggs *et al.,* 1997) model. Although they have their advocates and are still used for some air quality assessments, deterministic models based on fundamental mathematical descriptions of atmospheric processes have a much more practical use. They provide a much clearer relationship between source and receptor, and give much greater flexibility and transferability.

There are three different approaches used for deterministic dispersion modelling, Eulerian, Lagrangian and Gaussian. Unless photochemical modelling is involved, Guassian plume models are mostly widely used mesoscale models for regulatory purposes. They are easy to use and have modest input requirements - easily measurable meteorological parameters. They were initially developed using simple formulae describing the three-dimensional concentration field (Gaussian distribution) generated from a point source under stationary meteorological and emission conditions. The formulae take into account emission rate, downwind receptor distance and horizontal and vertical turbulence conditions. Even deterministic models use coefficients in these formulae that are determined by statistical optimisation techniques and therefore have a stochastic component. The initial limitations led to research and development of models that could cope with more complex situations. They can now deal with line, area and volume sources and nonstationary sources in nonhomogenous conditions. (Zanetti, 1990; Milford, 1993). Early Gaussian models used Pasquill-Gifford stability classes (Table 2.1), six discrete categories based on insolation and wind speed, as a means of determining atmospheric turbulence. Increased understanding of atmospheric physics and dispersion has led to the development of 'new generation' dispersion models such as ADMS (Carruthers *et al.,* 1994) and OML (Olesen, 1995a).

		Surface Wind Speed (ms^{-1})				
Time	Insolation/Cloud Cover	<2.0	2 - <3	3 - <5	5 - <6	≥6
Day	Strong insolation	A	A - B	B	C	C
Day	Moderate insolation	A - B	B	B - C	C - D	D
Day	Slight insolation	B	C	C	D	D
Day or Night	Overcast	D	D	D	D	D
Night	≥ 0.5 cloud cover	-	E	D	D	D
Night	≤0.4 cloud cover	-	F	E	D	D

Table 2.1 Pasquill dispersion classes.

Notes: 1. A, very unstable; B, unstable; C, slightly unstable; D, neutral; E, slightly unstable; F, stable; G, very stable.

2. Strong insolation correspond to a solar elevation of 60° or more, slight insolation corresponds to a solar elevation of 15° to 35°.

3. Pollutants emitted under clear night time skies with winds less than 2.0 ms^{-1} have subsequently been defined as class G. (Zanetti 1990)

In these models vertical profiles of mean velocity, temperature and turbulence are taken into account by parameterising the boundary layer in terms of Monin-Obukhov length (L_{MO}) and boundary layer height. Monin-Obukhov length represents the depth of boundary layer in which turbulence is predominately due to mechanical mixing. It is essentially the ratio between friction velocity and buoyancy, the former increases with increasing wind speed and surface roughness and the latter with increasing surface heat flux. A full definition is given by the formula below, but it can be calculated in the field by a simple method based on wind speed and temperature gradients.

$$L_{MO} = -u_*^{3} / ((\kappa\, g\, F\theta_0) / (\rho\, c_p\, T_0))$$

In unstable conditions when convective turbulence dominates, the Monin-Obukhov length is small (typically less than 10m) and negative. Under stable conditions, the Monin-Obukhov length is small and positive, it now represents the height at which stable stratification suppresses mechanical turbulence caused by friction at the earth's surface. In neutral conditions, mechanical turbulence dominates the boundary layer and the Monin-Obukhov length is large whether negative or positive (CERC, 1999). The non-dimensional parameter h/L_{MO} is used by ADMS in dispersion calculations. The relationships between Monin-Obukhov length, wind speed, boundary layer height and the Pasquill-Gifford stability categories are given in Table 2.2.

By giving a more accurate representation of changing dispersion characteristics with height and by using measurable physical parameters to define the boundary layer, these newer models are generally more accurate and more transferable. ADMS-Urban is widely used in the UK and was therefore the model of choice for this study.

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

Table 2.2 Values of wind speed (U), Monin Obukhov length (L_{MO}) and boundary layer height (h) that may be used to represent Pasquill-Gifford categories.

Note: There is no exact correspondence between boundary layer height and Monin Obukhov length, and Pasquill-Gifford categories as many different h and L_{MO} values may be found in one category. (CERC, 1999)

Despite their limitations many workers have compared the performance of Gaussian with other types of model and found there to be little difference. For example Nanni *et al.* (1996) compared the performance of a Lagrangian model (SPRAY) with a Gaussian one (HIWAY2) in complex terrain and found the Gaussian model to be only marginally out performed. Oettl *et al.* (2001) also compared a Lagrangian and a Gaussian model, GRAL and CAR-FMI respectively and only found the latter outperformed under either low wind speed or parallel wind conditions. Häggkvist (1997) found only small differences between a Gaussian and a grid model under general urban conditions.

Recent developments in dispersion modelling include the use of Artificial Neural Networks (Nunnari *et al.*, 1998; Gardner and Dorling, 1998) and Computational Fluid Dynamics

(Riain *et al.*, 1998). Although both these techniques are primarily still at the research stage, CFD models are commercially available, but as yet are not widely used.

2.2.2 Model complexity

Dennis *et al.* (1996) in describing the latest developments in modelling technology, carried out the in the USA under the auspices of the EPA, suggest that the solution to many modelling problems lies in the development of complex models. With improved computer technology and greater understanding of atmospheric processes it is now possible to model natural phenomena at higher resolution and with increasingly complex algorithms (Dennis et *al.*, 1996).

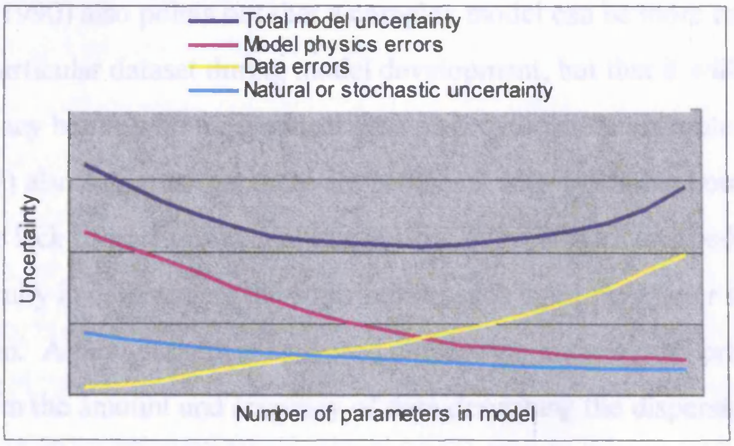


Figure 2.1 *Optimal model application. (after Hanna, cited Zanetti, 1990)*

Zanetti (1990) suggests that the extensive data requirements of complex models are rarely satisfied, therefore they do not necessarily perform better than simple models. Although with a greater number of parameters, the model may provide a better representation of the real world and there may be less uncertainty attached to the model output, the stochastic nature of the processes involved means that parameters cannot all be described deterministically. There is also an increased likelihood of data error associated with each input variable. The uncertainty associated with each variable may also be propagated throughout the model. In commenting on the limits of analytical dispersion modelling, Benarie (1987) also implicitly suggests that increasing the complexity of a model does not necessarily improve performance. Benarie describes how the chain between cause and effect in an atmospheric process is multi-nodal. The more sophisticated the model, the more nodes there are present and consequently more scope for error propagation. There are an optimum number of parameters that minimises total model uncertainty and anything over this may ultimately lead to an increase in error (Figure 2.1).

Lewellen and Sykes (1989) compared the performance of two models; a standard model and one with 'improved' resolution. They found no consistent advantage with the more complex model, concluding that meteorological uncertainty had an overriding effect and that until this could be more precisely determined, improvements in the understanding and modelling of plume dynamics would have little effect.

Zanetti (1990) also points out that a complex model can be more easily tuned or calibrated to fit a particular dataset during model development, but that it will not necessarily perform any better with independent data or provide more accurate predictions. Beck *et al.* (1997) also suggests that there are problems with validating complex models, one being the lack of peer group analysts capable of such work and perhaps more importantly the difficulty in determining the contribution each input parameter makes to the final prediction. Although there is increased confidence in the model prediction with an increase in the amount and accuracy of data describing the dispersion process, Beck *et al.* (1997) see this as an unjustified reliance on model complexity. The assumptions being that if all relevant processes are included in the model, the prediction must be correct. Beck *et al.* (1997) conclude that,

"Intuitively, a 'good' model would contain relatively few parameters yet be able to predict behaviour accurately over a wide range of conditions."

2.2.3 Model error and uncertainty

The problems of error and uncertainty have long been recognised in model evaluation studies, but it was not until the early 1980's that methods for formally quantifying error and uncertainty were developed.

Fox (1984) made the distinction between uncertainty as being something inherent in a stochastic system and error that 'represents in some sense inadequacy in either the measuring procedure' or the model prediction. The error can, of course, come from a number of different sources and in practice it is difficult to distinguish between them (Fox, 1984). Other workers in the field have gone on to define sources of error more specifically. Error in input data can arise from analytical technique, sampler bias, lack of sampling representativeness caused by spatial and temporal limitations and miscellaneous sources such as sample degradation or data entry mistakes (Batterman, 1992; Zanetti, 1990). Model errors can also result from incorrect computer coding (which is more likely to be significant the more complex the model becomes), from incorrect or simplified representation of physical processes or by changing continuous variables into discrete values as occurs with the use of Pasquill-Gifford stability classes (Mole *et al.*, 1993; Fox, 1984; Russell, 1988). Some sources of error can be reduced by using more sophisticated monitoring equipment or by incorporating recent improvements in the understanding of atmospheric processes into computer algorithms. This may reduce bias, but will not reduce inherent uncertainty (Fox, 1984).

Model output always has some uncertainty associated with it, as the predicted concentration only represents an estimate within a distribution of possible values. This particularly apparent in air pollution models where there is the 'natural variability' resulting from the random nature of turbulent diffusion. The atmospheric dispersion process itself is composed of a deterministic part and random element. The deterministic part can be modelled to a degree of precision allowed by input data that is itself derived from a stochastic process, but the random element is unpredictable. Uncertainty in model output not only results from the inherent uncertainty attached to the input data, but also from the physical parameterisations of, for example, turbulent processes within the model.

(Beck *et al.,* 1997; Benarie, 1987; Fox, 1984; Wotawa *et al.,* 1997). Uncertainty cannot be significantly reduced by improving model physics or by making more accurate meteorological measurements. It is not possible to completely describe turbulent flow with a finite amount of data, so it maybe more productive to quantifying the possible range of fluctuations and incorporate this into the modelling framework than to attempt to eliminate uncertainty from a particular dataset. This allows the determination of confidence limits on model predictions (Lewellen and Sykes, 1989). Pielke (1998) has recently suggested that the opposite approach should be taken; that one should start by looking at the impacts of various pollutant concentrations and assess what level of uncertainty can be tolerated in the model inputs. What spread of concentration estimates is acceptable for the data to still be useful to air quality policy?

Moussiopoulos *et al.* (1999) in a review of uncertainty analysis studies suggested that uncertainty associated with emission data may have more impact than uncertainty in either meteorological or boundary condition input data. Where models are used for air quality assessments, the prime interest has tended to be in the uncertainty in meteorological input data. Although it is acknowledged that error in emission data plays an important role, data are already generated at a local level and do follow clearer daily, weekly or seasonal patterns than do meteorological data. It is the problem associated with unrepresentative meteorological data that is of interest here.

2.2.4 Defining and recognising the limits of applicability

Providing a measure of uncertainty associated with model output may give some idea of the accuracy of the results, but will not define the model limitations. Confidence in air pollution dispersion model predictions is vital for a number of reasons, principally because of the large cost of implementing policy decisions based on them and the health risks that may occur if policies are formulated on the basis of erroneous predictions. For these reasons Russell (1988) suggested that model evaluation should not only identify and quantify the likely differences between predicted and observed values, but identify the required accuracy of the model inputs and determine the range of circumstances over which the model will perform adequately. Beck *et al.* (1997) considered the manner in which capabilities and limitations of models are communicated to be an ethical problem.

Weil *et al.* (1992) point out that the operational performance of models is often site dependent and that models should be examined for systematic error before they are applied to new source locations. They also comment, in the context of evaluation studies, that most model users are only concerned with model performance and that model physics is of secondary importance. However one must have faith in the basic structure of the model if one is to have confidence in model prediction when it is used beyond the range of existing data and when modelling new situations with different dispersion characteristics (Weil *et al.,* 1992). Although models are designed with specific uses in mind and there are many evaluation studies published, of those studies, some identify the sensitivity of the model to different inputs but few have determined the range over which the model performs well.

Zanetti (1990) claims that 'since ideal model application conditions are seldom found, air quality models are often used beyond their theoretical and practical limits of applicability'. Beck *et al.* (1997) also acknowledge that because models are cheap and readily available, there is a temptation to use them beyond their limits, but ask how these limits can be defined. In considering the regulatory use of models in Denmark, the modellers themselves recognised the tendency of environmental managers to use their model beyond its limitations. They have not found a solution except to indicate the limits and to suggest models that are more suitable where appropriate (Olesen, 1995a).

Apart from cases where models are clearly used in situations for which they are unsuitable, it is difficult to resolve the issue of recognising and defining limitations. Models are by their very nature expected to predict pollutant concentrations in hypothetical situations, situations in which it is impossible to test out model performance. Beck *et al.* (1997) referring to exposure assessment uses the example of a novel chemical released into the environment, clearly it is not possible to carry out validation based on experience or observation. Validation relies on comparison with past performance, but the next purpose to which a model is put is unlikely to be identical and predictions always contain some element of the unknown. The conundrum is that model validation becomes more difficult as the degree of extrapolation from known conditions increases, but this is exactly when use of a model is most needed and validity most critical. An important question is whether it is enough to rely on measures of uncertainty to determine if a model is accurate in unknown situations, it may be that validation and verification can give false ideas about a model (Beck *et al.*, 1997).

2.3 The Urban Environment and dispersion modelling

Modelling in urban areas presents many complex problems, not only because of the nature of the emission sources, emissions from point, area and line sources all overlay each other resulting in a composite mix of pollutants, but due to the complex topography in urban environments. The main features that differentiate urban and rural modelling are the highly variable nature of the urban surface, in terms of vertical height and the fabric of the surface. Both can influence local weather systems and create different dispersion characteristics. This modification is manifest at two levels; beneath roof level processes operate at a street canyon level (urban canopy layer) and above roof level where processes operate on the local or meso-scale to modify the atmosphere in a region defined as the urban boundary layer. This in itself does not necessarily present a problem if highly specialised models are used, but for routine modelling requirements there are two factors that need to be considered; firstly the sourcing of meteorological data that is representative of the urban environment and secondly how the meteorological pre-processors in commercially available models parameterise the urban boundary layer.

2.3.1 Urban meteorology

Meteorological conditions are largely governed by large-scale synoptic weather patterns, however there will always be a degree of modification caused by local topography. The extent to which synoptic or local features predominate generally depends on the strength of the synoptic flow patterns. During periods of high cloud cover and strong winds, local influences will be suppressed. When the sky is clear and wind speeds are low, the potential for weather modification is greatest. These conditions, particularly with clear skies at night, are incidentally those most likely to result in a build up of pollutants in the lower atmosphere (Landsberg, 1981).

It is difficult to exactly quantify the extent to which modification is the result of topography or the urban fabric. Human settlements have often developed, for a number of reasons, in areas of complex topography; for example defensive vantage points or natural harbours. Unless monitoring has been carried out at a site prior to urbanisation it is difficult to exactly quantify topography and urban fabric effects. Lowry (cited in Landsberg, 1981) developed a model to describe what he termed the metropolitan variable M, a measured meteorological parameter. This is made up of three components C - the basic climate of the region, L - a difference produced by location and U -the effect of urbanisation.

$$M = C + L + U$$

The modification can be determined to some degree by creating a profile across the urban area in which measurements are taken by wind sector, upwind of the urban area, within and on the lee side. If this is carried out in reasonably flat terrain useful information may be obtained.

One reason why little work has been done on the dispersion of pollutants and airflow in urban environment is due to difficulties with both tracer experiments and the use of aircraft for taking measurements. The increased use of wind tunnels has done much to understand flow at the micro-scale level, for example the work of Kastner-Klein *et al.* (1997), Wichmann-Fiebig *et al.* (1997) and Leitl and Meroney (1997).

Even allowing for these difficulties it is possible to discern many factors that can be directly attributed to the urban environment and these have been well documented by Oke (1987) and Barry and Chorley (1992). The principal effects of urbanisation can be seen to result from changes in the land surface; which is now relatively impermeable, has greater heat absorbing capacity, has less vegetation cover and is rougher. Additional effects can also be attributed to modification of the lower atmosphere from pollution and anthropogenic heat input.

The degree to which these factors can bring about changes in the synoptic weather patterns is complex and depends not only on the strength of the synoptic system, but also on the size and density of the urban area. Leicester, a city slightly larger than Northampton, has exhibited warming comparable to central London over small areas suggesting density may be more crucial than size (Chandler, 1961). However the synoptic wind speed required to breakdown these thermal differences increases with settlement size. In large urban areas mixing height, stability, turbulence, wind speed and direction, and even cloud cover and rainfall may be different from the surrounding area. It is worth considering how some of these influences affect the dispersion of pollutants.

The basic properties of the boundary layer that are of importance for air pollution studies are wind profile (speed and direction) which determines transport, the level of turbulence which is responsible for the spread and dilution of plumes and the height of the boundary layer which determines the volume of air in which mixing can occur.

A decrease in wind speed is one of the most import features and Landsberg (1981) quotes many studies in which expansion of urban areas around already existing meteorological stations have recorded a progressive decrease in either average wind speeds or the frequency of periods with high winds. Wind speeds are usually 5% lower in the city centre than the suburbs, but can reach 30% lower than surrounding areas if synoptic winds are light. The presence of trees when in leaf can further lower the urban wind speed. However the channelling of wind in street canyons and drawing in of cooler air at night when an urban heat island forms can result in slightly higher wind speeds. The development of the urban heat island may also lead to a change in wind direction when rural/urban breeze patterns develop. The wind in urban areas is also gustier. In rural areas large turbulent eddies tend to dominate whereas the increase in surface roughness in the city results in smaller eddies (Landsberg, 1981; Moran and Morgan, 1997).

Urban environments are rougher and it is often difficult to measure the roughness length (z_0). The height at which wind speed is effectively zero and the lower boundary for dispersion modelling. Wind profile equations in dispersion calculations need to include a zero displacement height. It is possible to estimate this using $z_0 = 0.5h\ (A^* / A)$ where h is the canopy height, A^* is the average silhouette area and A is the unit area of ground occupied by each element. The increased roughness results in a deeper zone of frictional resistance and reduced wind speeds over a greater depth compared to the surrounding rural area (Oke, 1987). Even in convective and stable atmospheric conditions, the vertical wind speed profile should still follow the characteristic logarithmic profile of neutral conditions. However Rotach (1997) has shown that above the urban surface, the boundary layer is divided into two layers, a mixed layer and a surface layer. The surface layer is further sub-divided into a roughness sublayer that extends from the surface up to 2 to 5 times the average height of the roughness elements. In these layers the logarithmic profile is no longer valid. Reduced wind speed results in a 'piling up' of wind, this and the urban heat island cause the daytime urban boundary layer to dome up by approximately 250m.

The increase in turbulence intensity bought about by both the increased surface roughness and the modified radiation balance prevent extreme conditions of either stability or instability. As a result urban effects are greatest when stable upwind rural conditions occur and especially at night (McElroy, 1997). This has a significant effect on dispersion and is taken into account in models such as ADMS-Urban. In ADMS, a minimum L_{MO} value can be set (see Appendix Three). This prevents the model determining a stability equivalent to Pasquill-Gifford stability categories of F or G (CERC, 1999)

It is difficult to predict the degree to which the synoptic weather conditions will be modified. Avissar (1996) has shown how the degree of vegetation cover is crucial to the gradients of sensible and latent heat flux across the urban area and to the setting up of local meso-scale circulations. The linear increase in vegetation cover produces a non-linear response in turbulent fluxes and other dynamic processes. The effects will be different if the urban area is highly vegetated in an arid area or if there is sparse vegetation in highly vegetated surroundings. Avissar suggests that the regulation of vegetation in the urban area can be used to mitigate some of these urban effects. Because the horizontal wind flow was found to be quite irregular, the work also highlighted the importance of carefully siting of meteorological monitoring stations (Avissar 1996)

2.3.2 Urban Modelling

There are many factors that can lead to inaccuracy or uncertainty in model prediction not least because the formulation of the models themselves are based on certain assumptions about physical processes in the atmosphere and how these influence the dispersion of pollutants. All the early work carried out to characterise the nature of atmospheric dispersion, and which was later used to develop dispersion models, was done in flat open terrain. This in itself leads to problems for modelling in the urban environment, where many factors serve to increase complexity and leads researchers to question model applicability (McElroy, 1996; Karpinnen *et al.*, 2000b).

In any situation, but particularly in highly variable urban environments it is often simply the lack of appropriate meteorological data that hinders the modelling process. One solution is the use of meteorological models. The use of coupled meteorological and photochemical models in an urban setting has been successfully demonstrated by Lu *et al.* (1997) in Los Angeles and by Svenson (1996) in Athens. Pielke and Uliasz (1998), although referring to meso-scale and regional air quality models, suggest that meteorological models are not used to their full advantage, but question the value of using such advanced models when there is still much uncertainty associated with emission data. Dabberdt (1999) also reports that the US weather research programme recommends the coupling of meso-scale air quality forecasts with meteorological and chemical reaction models.

As urban air pollution modelling is now an important part of local air quality management it forms part of one of the EU COST actions (see Appendix One). It is recognised that modelling and even monitoring in the urban environment present certain unique problems. Emissions occur either within or slightly above the canopy layer, receptors are often close to the sources and pollution plumes from many source may be superimposed on one another making estimation of background concentration difficult. One particular aspect of the COST 615 action was to consider the types and quality of urban dispersion models and to attempt to achieve some harmonisation of the modelling process (Schatzmann, 2000; URL3).

Some statistical models (Hecq *et al.*, 1994: den Tonkelaar, 1996; Ziomas *et al.*, 1995; Briggs *et al.*, 1997) or neural network models take account of the urban nature of the modelling domain implicitly. Middleton (1998) also describes how one particular model (BOXURB) takes into account urban effects when modelling pollution. With deterministic, particularly Gaussian, models this is more complex. Determination of heat flux and boundary layer height are fundamental to any complex dispersion modelling, but this is especially difficult in the urban domain. This is partly due to lack of appropriate monitoring, but also due to the inhomogeneity of the surface and how this effects the urban boundary layer.

The recognition that there is generally a lack, in both the nature and the spatial resolution, of suitable meteorological data for modelling needs, led to the setting up of the European COST action 715 - Meteorology applied to Urban air pollution problems (URL4). The aim of this Action is to advise modellers on the most appropriate meteorological data to use in urban dispersion modelling. An investigation of current modelling practice across Europe suggests that there is still a reliance on using meteorological data obtained from outside the urban area. Furthermore modifications to the data generally only involved taking into account the increased surface roughness and an assumption that extreme atmospheric stability will not occur. However work such as that carried out by Avissar (1996) shows how sensitive local scale meteorology can be even to the amount of vegetation present. The Cost 715 report concluded that there is a need for more meteorological monitoring in urban areas, but acknowledged the problems of siting equipment and deciding on the type of instruments to use (URL4).

The fact that current dispersion models often require meteorological parameters that are not routinely recorded, even at synoptic stations, led to the setting up of COST action 710. This looked at the harmonisation of pre-processing of meteorological data for dispersion modelling (URL 2). One important consideration is that possible error in, and the differences between, pre-processors can lead to model output errors of the same magnitude as the dispersion modelling output itself. Stübi *et al.* (1997) have demonstrated how different parameterisation schemes can produce different values for Monin Obukhov length, friction velocity, sensible heat flux and boundary layer height. They have also shown how sensitive these parameters are to height at which wind speed measurements are taken and to the estimation of roughness length z_0.

Surface heat flux and mixing depth are the two parameters that provide the greatest challenge to pre-processing. Direct measurement of both surface heat flux and mixing height require instrumentation that is unlikely to be available to most model users. The determination of latent heat flux requires a measure of soil moisture and water loss by transpiration; these were identified by COST 710 as being the least widely available data. One solution to provide this missing data has been suggested by Mensink and de Ridder (2000). Using satellite remote sensing, data are gathered on vegetation cover, cloud type and amount, and soil moisture with daily global coverage. Access to these databases is relatively inexpensive and allows the calculation of sensible surface and latent heat flux without the need for any ground based meteorological data. This approach goes some way to removing the need for locally monitored data, but may not have the required temporal resolution.

The use of satellite data has not yet been widely adopted and most urban modelling still relies on pre-processing. One of the assumptions often made in urban modelling is that the internal boundary layer height is constant over the whole urban area. This is not the case and it needs to be taken into account. One outcome of COST 710 was the development of Local scale Urban Meteorological Pre-processing Scheme (LUMPS). This takes data from the nearest synoptic station and uses knowledge of the urban surface to derive urban surface heat fluxes. LUMPS takes into account the degree of vegetation cover, allowing the partitioning between sensible and latent heat fluxes to vary across the urban area (URL2).

Some work has also gone into modifying the earlier pre-processing schemes to take into account the different characteristics of the urban boundary layer. Karpinnen *et al.* (2000b) has incorporated the new approach to defining the boundary layer proposed by Rotach (1997), and described above, into a refined meteorological pre-processor. Under this scheme, depending on the degree of stability, dispersion parameters are modified with respect to the rural values. When this is translated into pollution predictions, the high concentrations that are normally associated with stable conditions were found to be lower and in fact more realistic. Similar modification of the OML model were carried out by De Haan *et al.* (2001)

Another problem of urban pre-processing, demonstrated by Stübi *et al.* (1997), is the estimation of roughness length z_0. Two different methods can be used; morphometric which relates aerodynamic parameters to measures of surface morphometry and anemometric which uses field observations of wind or turbulence. Grimmond *et al.* (1998) have demonstrated the difficulties of using anemometric methods and how sensitive the estimate of surface roughness is to time of year, wind direction, height of sensors and type of instrumentation. Morphometric methods are in some ways easier as there is no need for tall towers and instrumentation, and many local authorities may already have suitable GIS based databases that can be used. However a disadvantage is that estimates are based on empirical formulae based on work carried out in wind tunnels.

Grimmond and Oke (1999) evaluated several different morphometric methods and again found that although most methods gave reasonable estimates it was by no means obvious which method performed best. There was poor agreement between the morphometric methods and high quality field measurements. Part of this discrepancy could be explained by irreducible errors in the observation, analysis of wind flows and in the unavoidable simplification that occurs in the description of geometric forms. There are difficulties in determining roughness length and zero-plane displacement and this does have implications for dispersion modelling. However morphometric methods are relatively simple and cost-effective. In many cases z_0 will be estimated from tables of typical values, as is the case with ADMS-Urban, however Grimmond and Oke (1999) suggest that these should be based on a visual depiction rather than function or land use.

There is undoubtedly a lack of appropriate meteorological data in many parts of Europe. This problem has also been considered in the USA where part of the Weather Research Program was similarly given the task of identifying research needs related to short-term prediction of weather and air quality in urban forecast zones. Not only was it recognised that large urban area impact on the weather, but that there are different forecasting needs in urban areas. The accurate forecasting of urban air quality for flexible pollution management strategies and public health were seen as crucial urban forecast needs. Although mostly concerned with providing better forecasting tools in relation to long term air quality management, in the context of emergency response planning it was recognised that there is a need for improved observational systems, particularly a well sited dense network of anemometers. Coupled Modelling-monitoring systems were also recognised as an important part of modern urban planning (Dabberdt *et al.* 2000).

2.3.3 Urban air pollution

Urban pollution is characterised by the short distances travelled, the inhomogeneous surface and short dispersion times that minimise chemical transformations. There are particular pollution problems associated with urban areas including London type cold weather smog and photochemical smog.

Although long term trends in pollution concentrations are dominated by changing emission characteristics, short-term trends are influence by both human activity patterns and meteorology. Diurnal trends tend to be fairly constant from one day to the next and any additional variation in pollution concentration is therefore largely due to meteorology. Climate also produces strong seasonal trends. These trends occur regardless of location but in the urban environment there is the additional factor of how the urban fabric interacts with local meteorology.

Conditions that lead to high pollution levels are often due to the inability of the atmosphere to transport pollutants away from their source. In the case of pollutants that have long residence times in the atmosphere high concentrations can be due to movement of air masses across continents. Certain meteorological conditions correlate well with pollution episodes and in specific circumstances these conditions can be exacerbated by the urban area. In the general urban area, wind speed and direction have the greatest influence on air quality. A study in Dublin found the strongest correlation was with wind speed followed by wind direction. Air pressure also correlated significantly with NO, NO_2 and NO_x concentration. This study also found that high levels of pollutant occurred on days with high pressure (anticyclones) and with low wind speeds. In spring and summer pollution episodes corresponded with higher temperatures and in autumn and winter on days when temperatures were lower than the seasonal mean (Delaney and Dowding, 1998). This is a feature of anticyclonic weather, it tends to have lighter winds and is warmer in summer, but colder in winter.

Using two-week average NO_2 concentrations as measured by Palmes tubes, a study in Cambridge also demonstrated the seasonal variation in concentrations at background sites. Similar results were not found in street canyons. Although roadside sites generally showed more consistent values throughout the year, pollution 'episodes' did tend to be more discernable in these and street canyon sites. Episodes were associated with stable periods with wind speeds less than $5ms^{-1}$. (Kirby et al. 1998). In a Copenhagen study looking at pollution concentrations in street canyon, Berkowicz *et al.* (1996) found no notable dependence on stability.

In studying the pollution characteristics of two cities (Copenhagen and Milan) which exhibit quite different synoptic weather conditions, Vignati *et al.* (1996) found that wind speed clearly explained most of the difference in pollution levels. Atmospheric pressure did not reveal any clear separation of the data at street level, although it did influence background levels.

Results from the few studies reviewed here are conflicting. Clearly the interplay between meteorology, the urban environment and the emission and subsequent dispersion of pollutants is complex and a challenge to model developers.

2.4 Atmospheric Dispersion Modelling System, ADMS-Urban

This section provides a review of the published papers relating to ADMS. Although all versions of the ADMS model are based on the same principles for parameterising the boundary layer conditions, most papers relate to the point source versions of the model and as such are not entirely relevant to the present study. Consequently only those deemed most appropriate are reviewed.

ADMS was first released in the early 1990s. It soon became clear that it was likely to become widely used for regulatory purposes in the UK. It had many features that had only previously been available in research models. It was PC based and was fully integrated with a Geographical Information System. This allowed easy set up of modelling scenarios and improved data presentation. The main difference between ADMS and other models available in the UK was that by use of a meteorological pre-processor, the model was able to characterise boundary layer conditions in precise terms. The model could be directly linked to an emissions inventory database; urban versions also contained a street canyon sub-module and could model a variety of source types (CERC, 1999).

Version 2.0 of the model was released in November 1995 and was the subject of a validation study using meteorological and particulate data (as measured by Rapid-Scanning Lidar RASCAL) obtained in the vicinity of High Marnham power station during August 1994. The meteorological pre-processor within ADMS can derive parameters such as boundary layer height, Monin Obhukov length and friction velocity from different sets of meteorological input. The effect this has on model output was a central feature of this investigation. Meteorological data consisting of different parameters was obtained from three different sites. Wind measurements at two heights, temperature readings at four heights and insolation at ground level were obtained from a TV mast situated 45km ENE from High Marnham. Standard hourly synoptic data were available from a meteorological station 40km ESE of the power station. Wind speed and direction were also monitored on a 10m mast at the power station itself.

The model was run using the 'surface' synoptic data and the 'elevated' data from the Television mast along with Lidar estimates of boundary layer depth (h) and surface heat flux ($F_{\theta0}$) derived from the insolation values. ADMS was used to calculate ensemble plume centreline concentrations for the power station site at a nominal height of 40m. The most significant finding was that predictions of C_{max} using the surface data could be anything up to a factor of two greater than those using the elevated data. The discrepancy arose because estimated values of $F_{\theta0}$ and h were consistently greater when derived from the synoptic measurements. Using the 'elevated' meteorology, the predicted maximum 10 minute mean ground level concentration was found to be a factor of three less than the mean concentration measured by RASCAL. With either set of meteorological data ADMS was able to correctly predict or slightly under estimate the distance of the maximum value (C_{max}) from the source. Overall it seemed that ADMS performed reasonably well, but the sensitivity to the meteorological data did give cause for concern. (Bennett and Hunter, 1997)

Other workers have been concerned with sensitivity of ADMS to different format meteorological data i.e. whether it is statistical or sequential, and to the location from which the data is gathered. Davies and Thomson (1997) ran ADMS version 1.5 with meteorological data from Wattisham, a synoptic site in SE England, but with emission characteristics based on hypothetical sources, a power station and a factory. The sequential data consisted of standard synoptic readings of temperature, wind speed, wind direction, cloud cover and precipitation amount. The statistically analysed dataset contains data in the form of a multi-dimensional frequency table. Meteorological conditions are described by five parameters each divided into a number of classes as shown in Table 2.3

Parameter	No of classes
Hourly average wind speed	5
Hourly average wind direction	12
Surface sensible heat flux	7
Boundary layer depth	7
Hourly precipitation amount	3

Table 2.3 Statistical data – meteorology classification.

Both datasets were for a ten-year period. ADMS was run to produce long-term average concentrations. Although sequential data might be expected to give a more accurate prediction, model output was very similar. Contour plots showed the location of the peak concentration to be very close and peak value only varied by 1% for the factory site and by 3% for the power station. More variation was found for 98th percentile values. Statistical data was also prepared using shorter time periods. Long-term averages produced by the three-year and five-year datasets differed from the ten-year set by no more than 1% for the power station, but by as much as 6% for the factory site. Statistical data for individual years showed more variation for the power station and could vary by up to 23%.

The effect of selecting different meteorological parameters for ADMS was also investigated. Statistical data was produced using seven categories of reciprocal Monin Obukhov length (L_{MO}) instead of surface sensible heat flux. This did not improve the long-term concentration when compared to the sequential data output and the model was found to be sensitive to how L_{MO} categories were defined. Finally the model was run using statistical data from three other sites within 100km of Wattisham with similar terrain, with data from a more distant site with similar terrain and from two sites with quite different topographies. Again the power station and the factory gave different results. For the factory, peak values varied by no more than 7% and the 98th percentile of the peak value by no more than 10 %, if data came from within 70km of Wattisham. The power station was found to be more sensitive to meteorological data with equivalent figures of 6% and 13% using data from a site only 20km away. In both cases sites from further afield gave significantly different results. It is also likely that these differences would have been amplified if short-term average concentrations had been used instead.

ADMS-Urban, released in 1996, is one of the more recent ADMS developments. This model builds on the original model physics of the point source version, but includes line, area and volume sources. ADMS-Urban also includes a fully integrated street canyon model, an atmospheric chemistry sub-module and a traffic emissions database. Owen *et al.* (1999) carried out a study using ADMS-Urban for the first time, to model NO_x and SO_2, in a large urban area. These pollutants were modelled for the whole of the Greater London area using the Greater London emissions inventory for a summer and a winter period in 1995. Predicted concentrations were compared with monitored data from four sites. This paper is of interest as it highlights some of the problem of using meteorology from a distant site and some of the sensitivities of the model to various meteorological parameters. Meteorological data used in the modelling was obtained from a meteorological synoptic station at Heathrow airport. Although Heathrow is situated nearly 30km to the west of central London, it is still within the Greater London area and was considered to adequately represent weather conditions for the whole domain. However Owen *et al.* (1999) did acknowledge a certain degree of accuracy would be lost and this may have led to problems modelling some of the larger point sources situated far to the east. Correlation of paired time-series data for monitored and modelled Sulphur dioxide was poor. Although the model could accurately predict the value of peak SO_2 concentrations the exact timing was not precise. This was considered to be due to the distance between the meteorological site and the pollutant source, and to the assumption made by the model that pollutants will arrive at a receptor in the same hour during which they are emitted. Sensitivity to wind direction was also noted.

ADMS performed better predicting oxides of nitrogen, presumably because emissions are less dependant on large point sources and consequently less sensitive to wind direction. However the model performance for NO_x did highlight another modelling problem. Predicted NO_x values did not show much variation between summer and winter, but observed values show an increase in winter. Model error could be attributed to problems with either the emissions or the meteorological data. Domestic and industrial NO_x emissions are known to have significant seasonal variations as are 'cold start' emissions, but the biggest contribution to ambient NO_x is from 'hot' emissions which are more constant throughout the year. Lack of seasonal variation in emission inventories could contribute to under-prediction in winter, but other factors were obviously involved. The distribution of wind speed, wind direction and cloud cover was similar for winter and summer, but temperature obviously shows seasonal change. Using the given data the meteorological pre-processor calculated a higher frequency of stable atmospheric episodes in winter than in summer and it was thought that the tendency to under-predict may be linked to this (Owen *et al.,* 1999).

Two groups of workers describe the use of ADMS by local authorities in the UK as part of their air quality assessment activities. In both cases ADMS is used in conjunction with other models. In the first example Chatterton *et al.* (2000) used ADMS in conjunction with the Meteorological Office NAME model in a source apportionment exercise and used NAME to produce regional background concentrations. In the second example Crabbe *et al.* (2000) used ADMS to provide in-depth modelling in the third stage of their Review and Assessment of local air quality after two screening models had been used. In this study, details were given of meteorological data. Although the modelling was carried out in the London Borough of Barnet, an area some 10 kilometres north of the city centre, the meteorological data was obtained from the London Weather Centre, situated in the centre of the metropolitan area. Although Crabbe *et al.* (2000) draw attention to the limitations imposed on modelling accuracy through the use of emission inventories compiled with annual average data and suggest the need for more detailed information for Stage 3 assessments, the problems of using unrepresentative meteorological data were not highlighted.

In carrying out the National Air Quality Strategy in the UK, 14 local authorities were designated as pilot areas, were a combination of monitoring and modelling work was carried out. ADMS-Urban was used in half of these and the work reported on by Carruthers *et al.* (2000). Of interest here are the findings that relate to the meteorological input data. Primarily that it is important to use local data, especially for wind direction. They state this in relation to making comparisons between predicted and measured concentrations, but by implication it is equally important if using the model as a predictive tool. They also found that modelling traffic sources in low wind conditions could lead to over-prediction and acknowledged that in running the model no allowance can be made for the variation in meteorological conditions that exist over large conurbations such as London.

2.5 Concluding comments

As models become more complex their input requirements become more exacting, it becomes harder for non-specialised users to satisfy these needs. With more widespread use and also with a greater reliance placed on model output in the field of air pollution control, their inappropriate use becomes more likely.

It is clear that the influence of errors in the meteorological input to air pollution models is not fully understood. Although the use of meteorological observations from sites some distance from the modelling domain is common, it may present significant problems. The influence of errors arising from the meteorological input is examined in this thesis.

CHAPTER THREE

RESEARCH DESIGN AND ACTIVITY

3.1 Introduction

As noted in the previous chapters, the principal aim of this investigation is to examine the meteorological data requirements for atmospheric dispersion modelling in an urban environment. The extent to which data from different meteorological sites or even databases can affect model output has already been demonstrated to some degree by Bennett and Hunter (1997), Davies and Thomson (1997) and Hall *et al.* (1999). This project intends to take this work further by the investigation of the performance of a dispersion model using meteorological and air quality data that has been gathered within an urban environment. The extent to which the urban environment can itself influence local meteorology is well documented (Oke, 1987; Barry and Chorley, 1992). It is also an acknowledged problem that models designed for use within this urban environment setting were initially developed using empirically formulae derived from experimental work carried out in homogenous rural terrain (McElroy, 1996; Karppinen *et al.*, 2000b).

Before models can be used successfully in urban areas there needs to be an examination of the quality and the nature of the data available to model users. No detailed assessment of how representative the available meteorological data are and the effect this is likely to have on air pollution dispersion modelling outcomes in the general urban environment has been reported in the scientific literature although Manning *et al.* (2000) have evaluated the performance of a street canyon model in these terms. To perform this particular investigation some specific objectives were outlined. Firstly the variation in certain key meteorology parameters that occurs between the urban environment and the nearest meteorological recording stations must be determined. Secondly the sensitivity of the chosen model must be assessed. This shows which parameters have the greatest influence on model output and gives an indication of the error that is likely to occur when using unrepresentative data. Thirdly a modelling exercise needs to be carried out using different sets of real data to assess the affect on model output. Comparisons need to be made between predicted and monitored air quality data. Finally, the findings of this study must be considered within the context of regulatory air quality management.

The project therefore has three clearly defined practical stages; data collection, modelling and data analysis. The sensitivity study is purely a theoretical exercise and as such requires no raw data. However in order to carry out the first and third objectives certain specific data requirements must be met. Categories of data can be defined under the following four broad headings meteorological, topographical, traffic flow and air quality; the first three being required for input into the dispersion model. Although many relevant data are already available from other agencies, the first stage of the project is necessarily one of data collection. The sources of available data and the process that has been undertaken to generate data specifically for this project will be described below. The rationale behind the selection of study areas and the particular model used will be described first followed by a description of the sourcing and use of data.

3.2 Choice of study area

Although a larger urban area could have been used where the modification of the meteorology by the urban surface is substantially greater, medium-sized (150,000 to 300,000 populations) urban areas are still subject to air quality problems and are as such required to carry out local air quality assessments. The cost and inflexibility of physical monitoring often means that they often rely on modelling to achieve this.

Northampton has a population of around 200,000 and has the advantages of locally available meteorological data and the close links already established with the local authority through previous pollution studies (NAPS –Northampton Air Pollution Study). University College Northampton has its own automatic weather station and in addition readings of climatological conditions are taken daily at 0900Z. Northampton Borough Council carry out air quality monitoring at a site close to the town centre (Cliftonville) and data were made available to the project. Northampton is typical of many urban environments in the Southeast and Midlands of England where concerns over air quality are more likely to arise from high traffic densities rather than from industrial sources.

In considering site selection for meteorological and air quality monitoring the following requirements have to be taken into account. Air quality monitoring sites need to be well within the urban area, on roads with high traffic flows so that pollution levels are likely to be relatively high, where traffic flows can easily be recorded, with suitable space to locate the monitoring equipment and with the co-operation of site owners. Only a few sites are able to satisfy all the criteria.

UCN does not have the facility to collect traffic flow data and the sub-contracting out of this task is expensive. A solution to this is to use data that are already gathered from traffic management purposes by the local authority. Traffic sampling is undertaken at a number of sites in Northampton, the majority being on roads with large traffic volume. Three different types of survey are carried out each providing data in a different format. They were all evaluated for inclusion in the study, but only UTC sites were considered worth investigating.

UTC (Urban traffic Control) sites provide continuous two-way hourly traffic counts. Negotiations were made with the Borough Council in order to find council owned property in the vicinity of UTC sites. For air quality monitoring purposes, the site requires a suitable area (2.5m x 5m) to park a mobile air pollution laboratory, proximity to the road without tall building that would create highly localised wind patterns and access to a 32 amp power supply. Of the five sites that were initially identified only three proved to be suitable; the car park on Upper Mounts, the council depot at Westbridge and Kingsthorpe housing office on Harborough Road. The monitor was placed at each of these sites for at least two months. However the UTC traffic data proved to be of poor quality and only the Kingsthorpe site (See Figure 3.1) was used in subsequent modelling exercises, even then using traffic data obtained from a special survey.

Meteorological and air quality monitoring was also carried out in conjunction with Imperial College, London in Barnes, part of the London Borough of Richmond. Hammersmith Bridge which crosses the River Thames linking Barnes with Hammersmith was closed for nearly three years with traffic diverted to other river crossings. Imperial College, Department of Epidemiology and Public Health intend to use air quality data

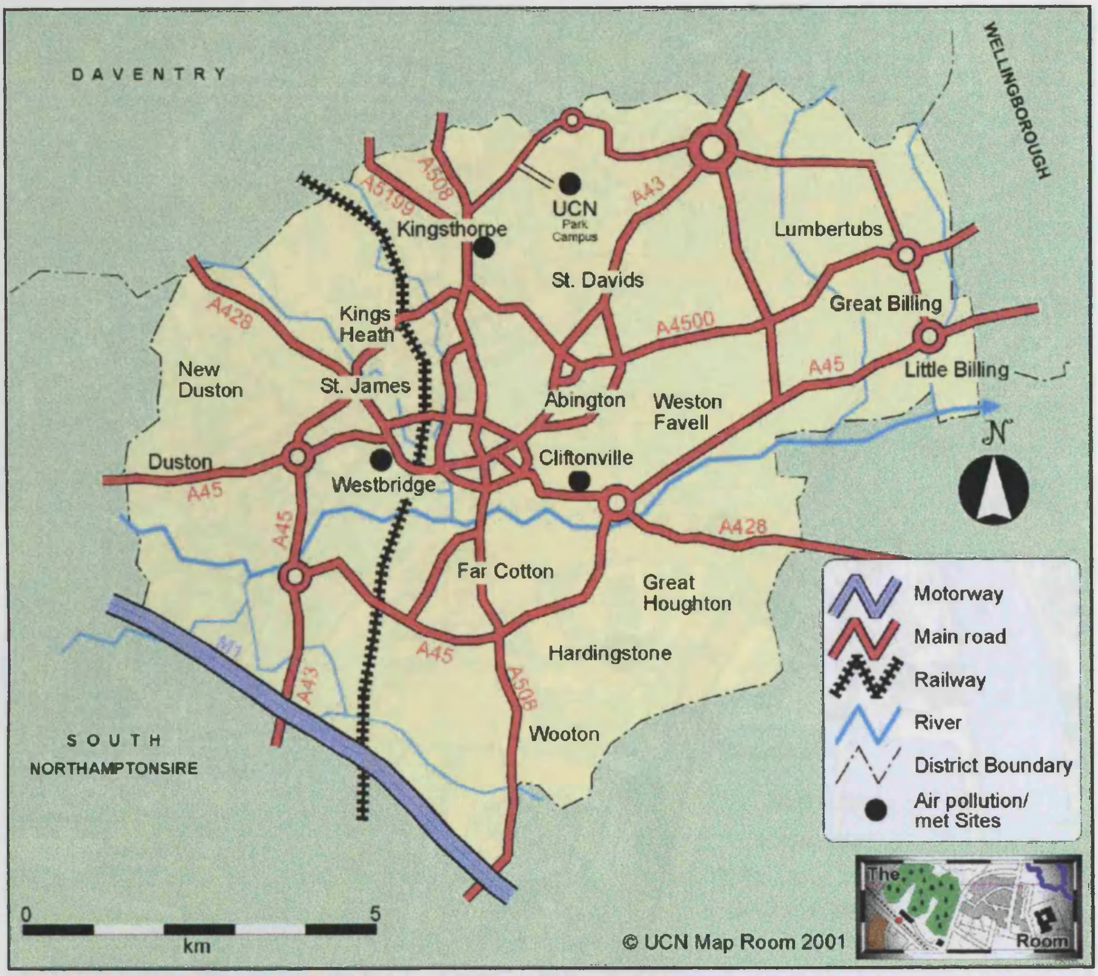


Figure 3.1 Map of Northampton showing monitoring sites

Figure 3.2 Map of Barnes showing monitoring sites.

1. - Holy Trinity Church

2. - Lowther Primary School

3. - Traffic monitoring sites

4. - Barnes Library

gathered in the area to form part of an investigation into the epidemiological effects of a traffic management scheme.

The reopening of Hammersmith Bridge was planned for late 1999 or early 2000. It was decided in the summer of 1999 to locate two mobile air pollution laboratories in the area, one at a background and one at a roadside site. A suitable site also had to be found for a meteorological mast. In this case access to traffic flow data was not a problem. The closure of Hammersmith Bridge was the subject of much local interest and the local authority were willing to undertake traffic monitoring for their own use and to make the data freely available. A suitable background and meteorological site was found at Lowther Primary School, Stillingfleet road and a roadside site at Holy Trinity Church on Castlenau. See Figure 3.2.

Both study areas are described in greater detail in Appendix Two.

3.3 Choice of model

The intention of this study is not to evaluate the performance of any particular model. Any readily available urban dispersion model could have been used to illustrate the problems that may be encountered through the use of inaccurate or inappropriate data. However certain criteria needed to be met for the findings to be of practical use. The model used had to be representative of a type in common use in the UK and must use 'up to date' modelling techniques. ADMS-Urban, a Gaussian dispersion model developed by Cambridge Environmental Research Consultants Ltd (CERC) was already available at UCN and does satisfy the above criteria. ADMS-Urban is one of the 'new generation' type of models and is also currently one of the most popular with local authorities in the UK for air quality assessment. The urban form of the model has been developed from an earlier point source model and provides the additional facility to model line and area sources as well as containing a street canyon sub-module. It can be used to model complex scenarios with multiple industrial, domestic and traffic emissions covering a large area. Large urban areas, in particular, still present certain difficulties to modelling with ADMS-Urban and these are readily acknowledged by the model developers (Carruthers *et al.*, 2000; Owen *et al.*, 1999).

Although a fairly complex model was selected its use was confined to simple and straightforward modelling scenarios. The effect of one particular type of input was the focus of this study and the outcomes would have been more difficult to analyse if more complex scenarios had been set up. Although ADMS contains an integrated street canyon sub-module model, based on the Danish OSPM, it was felt that street canyon modelling is a specialised areas and beyond the scope of this project. The performance of OSPM has been evaluated elsewhere (Berkowicz *et al.,* 1996). In fact no data collection was carried out in what could truly be described as a 'street canyon'. The complex terrain sub-module was similarly disregarded. Including this sub-module can effect pollutant concentration significantly under certain conditions. However, Carruthers *et al.* (2000) claim that the model is less sensitive to terrain when modelling low level sources such as traffic.

One of the most important features of ADMS is how the model parameterises the boundary layer structure. ADMS uses a physical parameter, Monin Obukhov length, to define atmospheric stability. This makes it particularly sensitive to meteorological data and allows some flexibility in the type of input data used to run the model. It also, in theory, allows for a more accurate prediction of pollutant concentrations than the use of Pasquill-Gifford stability classes, which characterise atmospheric stability into 6 broad bands based on time of day, degree of cloud cover and wind speed.

ADMS contains a number of sub-modules embedded into its structure, in addition to those already mentioned; these include a meteorological pre-processor and a choice of chemical reaction modules. The model is also integrated with a Geographical Information System - ArcView. This enables modelling scenarios to be created using digital mapping data or aerial photography and the output to be presented visually by way of contour plots.

A number of different inputs are required to run ADMS. Some of these are mandatory, but others may be entered to refine the quality of the output. Details of the input data used for this study are given below. A description of setting up and running of ADMS concentrating on those parts relevant to the present study is given in Appendix Three. Further details on ADMS can be found in the user's manual (CERC, 1999).

3.4 Meteorology

Atmospheric dispersion models are generally run with meteorological data in one of two formats; either 'statistical' where the data are presented as the frequency at which different meteorological conditions occur or as sequential runs of hourly data. Statistical data are generally derived from 10 years worth of data recorded at one site. There only are a limited number of sites in the UK that have data runs of sufficient length. Sequential hourly data for shorter time periods are more widely available. Whichever type of data are used, unless the modelling domain is close to a meteorological recording site, the distance between the two inevitably introduces some error in the model output. Data are rarely recorded at the location required for the modelling nor are they recorded in the specific format required by complex models. Furthermore most meteorological data recording sites are in rural locations whereas by necessity most of the air pollution modelling activity carried out in the UK is in highly urbanised settings. If statistical data are used some of the variation due to distance between sites may be 'averaged' out or if a consistent bias is know to occur this can be taken into account (Royal Meteorological Society, 1995). With the facilities available to take local meteorological readings over the study period, it was decided for the purposes of this project to only use sequential data.

One possible solution to the lack of appropriate data lies in the use of meteorological models. However there are cost implications involved in obtaining data and questions could still be raised regarding their accuracy. It is unlikely that regulatory users of models would have the necessary software or expertise to run such models themselves. For the most part routine use of dispersion models still requires the use of data gathered at a site some distance from the modelling domain.

Although data from a meteorological model were used for part of this study, data were mainly either collected specifically for the project or obtained from databases collated by other agencies. Apart from data from the AWS on the college campus, meteorological

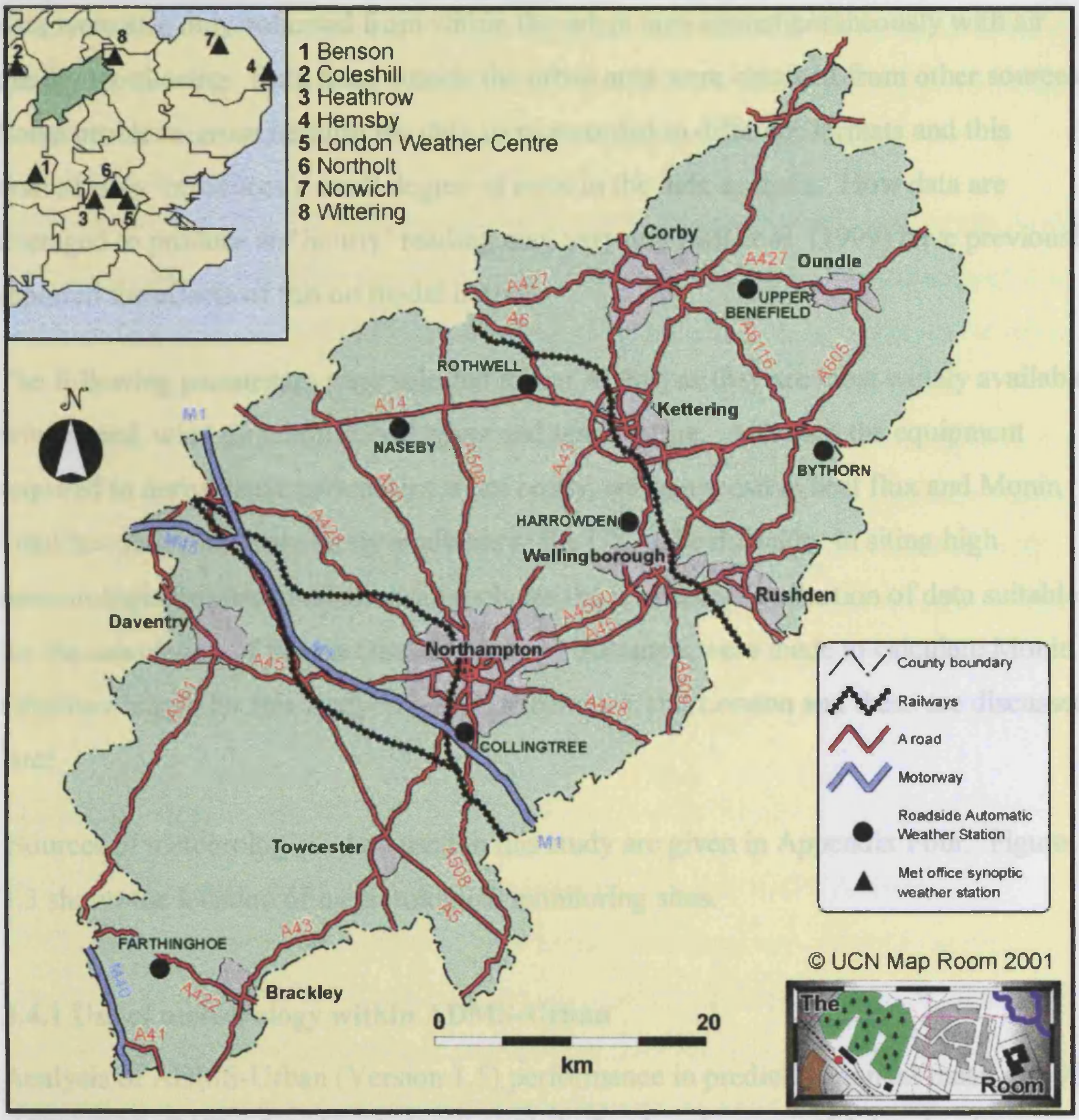


Figure 3.3 Meteorological monitoring sites in Northamptonshire and (Inset) SE England.

data were also only collected from within the urban area contemporaneously with air quality monitoring. Data from outside the urban area were obtained from other sources. Some problems arose because the data were recorded in different formats and this unavoidably introduces a small degree of error in the data analysis. How data are averaged to produce an 'hourly' reading may vary and Hall *et al.* (1999) have previously reported the effects of this on model output.

The following parameters were selected to run ADMS as they are most widely available; wind speed, wind direction, cloud cover and temperature. Although the equipment required to derive these parameters is not costly, surface sensible heat flux and Monin Obukhov length data are rarely available in the UK. The difficulty in siting high meteorological masts in urban area precludes the widespread collection of data suitable for the calculation of Monin Obukhov length. Attempts were made to calculate Monin Obukhov length for this study in both Northampton and London and these are discussed later.

Sources of meteorological data used in this study are given in Appendix Four. Figure 3.3 shows the location of meteorological monitoring sites.

3.4.1 Use of meteorology within ADMS-Urban

Analysis of ADMS-Urban (Version 1.5) performance in predicting annual mean and 98th percentile concentrations in relation to statistical or hourly sequential data has been reported elsewhere (Davies and Thomson, 1997). This study was only concerned with the performance of the model in predicting hourly mean concentrations on a short-term basis; therefore only hourly sequential meteorological data of at least one-month duration were used. Of the possible range of meteorological data that can be used to run ADMS, apart from wind speed and direction that are obligatory, Julian day, hour and cloud cover was selected. Temperature, as it is readily available, was also used to improve the quality of the output. Although hourly readings of cloud cover are only available from a few sites across the country, these data can be purchased by regulatory users of the model from the Meteorological Office or other agencies and can more easily accessed than other parameters such as surface sensible heat flux, Monin Obukhov length or boundary layer height. The intention of the study was to replicate typical model use so cloud cover from Wittering, the nearest synoptic site to Northampton, was used. Wittering meteorological data were also used as a baseline with which to compare other datasets.

Although this was not a stated objective of the study, attempts were made to run ADMS with some with some of the parameters that CERC recommend should be entered if more accurate estimates are available than can be predicted by the model's own meteorological pre-processor. Meteorological monitoring was carried out in both Barnes and Northampton in order to calculate local Monin Obukhov lengths. Technical problems meant that only data from Barnes could be used in the final analysis. However ADMS was run with some of these other parameters using output produced by the Met. Office NWP model. As ADMS's own pre-processor predicts boundary layer conditions based on data from only one site, one might assume that the NWP model would produce a better estimate of boundary layer height.

3.5 Traffic Data

In the UK the major atmospheric pollutants are now largely traffic derived. Consequently the focus of this study is the ability of a dispersion model to accurately predict pollution levels using traffic as the principal emission source. In Northampton there are few industrial atmospheric pollution sources and any industrial component was only included in the modelling process implicitly as a background pollution reading. ADMS contains a database of emission factors from which emission rates are calculated depending on the traffic speeds and flows for each road in the modelling domain. The emission factors, calculated by the Transport Research Laboratory, are representative of the UK vehicle fleet.

In order to improve the accuracy of the modelling it was decided to use as far as possible actual traffic counts taken concurrently with the air quality monitoring rather than use historical local authority data or traffic model output. The financial cost of the monitoring meant that in Northampton it could only be collected at two sites and for short time periods. However the data proved to be remarkably consistent from one weekday to the next and from week to week. Hourly averages were calculated for weekdays, Saturdays and Sundays. The small variations that do occur (less than 5%) are unlikely to have a significant effect on model output. Entering traffic flow figures into the model on a daily basis would have been extremely time consuming and was unwarranted in terms of the increased accuracy.

Three sources of traffic data were used for this study. A description of these is given in Appendix Five.

3.5.1 Use of traffic data within ADMS

ADMS requires traffic data in the form of a vehicle count per hour for each road section. This can be broken down into a light duty vehicle and a heavy-duty vehicle component with an associated average speed. To allow for variation in flow throughout the day and over weekends, time varying emission factors can be entered to represent weekdays, Saturdays and Sundays. The model assumes vehicle speeds to be constant at all times.

Setting up modelling scenarios within ADMS required a mixture of data obtained from the sources mentioned in Appendix Five and local knowledge. For example, in Kingsthorpe, the actual hourly average flow was used for Harborough Road, but all other roads were assigned the SATURN flow figures (see Appendix Five for details). Although the fluctuations in flow throughout the day on Harborough Road data may be representative of the main roads in the area, they may not be applicable for side roads, especially those near schools or local shopping centres. Some inaccuracy was unavoidable, as it is only possible to enter one set of time varying emission values. At Cliftonville two-week traffic counts were available from Bedford Road West and Cliftonville Road. These two sites and two additional sites on the Nene Valley Way also had one-day sustainable transport policy counts. SATURN data were used for the other roads in this area. Although local variation may be present the data from Cliftonville Road were used to calculate time varying emission factors.

Modelling for Barnes relied on traffic counts taken on Castlenau and Lonsdale Road. As no SATURN data were available all other roads in this study area, which were small back streets, had flows and speed estimated from local knowledge. Hammersmith Bridge was closed to all traffic except buses so numbers were estimated from the number of bus routes using the bridge and frequency of buses per hour. Modelling from the 22nd December, when Hammersmith Bridge reopened, until the end of the year was carried out on a daily basis. Traffic varied greatly from day to day over the holiday period, as schools were closed for the Christmas and as people got used to using the bridge again. For this period the number of vehicles, apart from buses, crossing Hammersmith Bridge was estimated from the difference in numbers on Castlenau and Lonsdale Road. Although pollution monitoring carried on until mid February no further modelling was carried out after the end of 1999. Traffic counts were not available for the period when St Paul's School, a major contributor to traffic in the area, reopened after the holidays.

ADMS requires an estimate of average vehicle speed over the whole day and there is no facility for taking into account how speeds may very during rush hours. As vehicle speed clearly has an effect on emission rates this may again lead to some error in the model output. It was considered more important to have a realistic vehicle speed during peak flows rather than at other times of days. As a result vehicle speeds on some road sections that are subject to very heavy flows during rush hour were reduced from the SATURN estimates on the basis of local knowledge. Traffic counts from Barnes included an estimate of average speed and these were used in setting up the modelling scenarios.

3.6 Air quality data

The third stage of the study involves running ADMS with different sets of meteorological data and assessing the performance of the model against air quality data. The modelling was generally confined to predicting carbon monoxide concentrations as in the urban environment it is estimated that a large percentage is derived from transport. Background levels are generally below 0.1ppm, but levels can rise rapidly near emission sources. Its detection and behaviour in the atmosphere are relatively straightforward which makes it the ideal pollutant to use where confounding factors were to be avoided. As it has a residence time of $0.4a^{-1}$ in the atmosphere background levels are not subject to the temporal and spatial variation found with oxides of nitrogen, it does not travel great distances or come from a variety of sources as do particulates. Carbon monoxide concentrations are considered to be a good indicator of other traffic derived pollutants levels. In an air quality assessment programme, once the model predictions of carbon monoxide have been shown to be accurate and correlations between it and other pollutants have been established, continued measurements of these may no longer be necessary (Jones *et al.*, 2000).

Although NO_2 and NOx were used in this study, photochemical effects and the model's ability to cope with these were not investigated. It is acknowledged that nitrogen dioxide may have been a more appropriate pollutant to study in terms of health effects and air quality objectives. It has an EU air quality objective based on hourly concentrations whereas the objective for carbon monoxide is based on an eight hour running mean. Using a computer model to predict eight-hour running mean is not subject to the same degree of error as the prediction of single hour concentrations. Although in this study, the performance of the model was evaluated on the ability to predict one-hour carbon monoxide concentrations it is unlikely that local authorities would use a model in this way to satisfy air quality management criteria. Sulphur dioxide also has a one-hour mean objective, but is not emitted by road traffic to the same degree as carbon monoxide or oxides of nitrogen.

Air quality data were collected using a Horiba Mobile Air Pollution Laboratory, and two Learian Streetboxes. Similar fixed site Horiba analysers are installed at Northampton Borough Council offices on Bedford Road, Northampton. Data from these were made available for this project. Technical information relating to this equipment is given in Appendix Six.

3.7 Background air quality

In order to assess the performance of ADMS and make a comparison between model output and monitored data, it is necessary to add a background value to the predicted concentration value. This takes into account that component of a particular pollutant not directly produced by the emission sources included in the modelling scenario. Ideally monitored background data should be available for every modelling domain, but this is rarely the case. Without such data there are four possible options

- To extend the geographical size modelling domain and to include a greater range of emission sources.

- To use nested models with a coarse gridded model for transboundary pollutants and a finer gridded model for the area containing the receptor points.

- To use proxy data gathered at a background site elsewhere, but sharing a closely as possible the same physical features.

- To use the estimated annual mean concentration for each pollutant calculated for the whole of the UK on a $1Km^2$ basis by the DETR and NETCEN.

Some local authorities do carry out background monitoring, but ideally there should be several monitoring sites within each area to allow for the different values to be used depending on wind direction. Financial costs are likely to prevent this. However most local authorities already have emission inventories for the industrial sources in their area and these could be included in modelling scenarios. The sources of background data used in this study are given in Appendix Seven.

3.8 Monitoring Programmes

The monitoring work for this project started in December 1998 and continued until May 2000. During this period monitoring was carried out at four separate sites, three in Northampton and one in London. It was decided that more information could be gained from sampling at a variety of urban sites rather obtaining one long run of data at an individual site. Ideally a winter and a summer sampling programme would have been carried out at each site to gain a representative range of meteorological conditions, but this was only possible at Kingsthorpe.

Further details of the monitoring programme are given in Appendix Eight.

3.9 Modelling

Initially a model sensitivity study was carried out using a hypothetical scenario with a single road source. Input parameters were altered one at a time by pre-defined increments to assess their influence on model output.

The penultimate stage of the project involved running the ADMS model using the meteorological and traffic data collected during earlier stages of the project. Three modelling scenarios were set up based on the Kingsthorpe, Cliftonville and Barnes monitoring sites. Each scenario was run using meteorological data obtained from difference sources. The model output was in the form of predicted hourly average concentrations of a particular pollutant at a receptor site. The receptor site was defined by the location of the air quality monitor. This enabled direct comparison to be made between predicted and observed concentrations of pollutants.

3.10 Data analysis

The analysis of the meteorological data is largely qualitative and descriptive, although correlation coefficients are calculated where appropriate. For the data to be useful in later modelling exercises it was necessary to quantify the variation that is likely to occur over the small spatial scales used in the study area. This was achieved by comparing data on an hourly basis and recording the degree of disparity and its frequency.

Methods for assessing the performance of dispersion models have been well documented in the scientific literature and these have been broadly followed here. Both measures of difference and measures of correlation have been used in this study to assess to performance of ADMS-Urban. A brief review of model evaluation and details of the statistical tests used in this study is given in Appendix Nine.

CHAPTER FOUR

ANALYSIS OF METEOROLOGICAL DATA

4.1 Introduction

In order to run any type of model successfully access to reliable input data is required. For atmospheric dispersion models this will generally consist of an emission inventory and a set of meteorological data. When the pollutant of interest is principally derived from traffic, the emissions inventory will at the simplest level, consist of average traffic flows, vehicle speeds and some rudimentary analysis of fleet composition. This is commonly a breakdown into heavy or light duty vehicles, or the number of petrol and diesel engined vehicles. In urban areas traffic data are routinely recorded for traffic management and planning purposes, but if not, it is relatively inexpensive to collect, and monitoring programmes can be carried out with a high degree of flexibility. The same cannot be said for meteorological data.

The siting of meteorological equipment in urban areas is far from straightforward. This means that there is often a reliance on data recorded at synoptic stations, sited at rural locations and often many tens of kilometres from the modelling domain. If the data to be used is statistical rather than sequential this presents less of a problem as, unless the underlying topography is vastly different or the site is very distant, any differences in the data will tend to be averaged out over time (Davies and Thomson, 1997). The Royal Meteorological Society (1995) suggest that geographic proximity alone is not sufficient to confer representativeness and recommend that local measurements be made for a reasonable period, such as a year, in order that long term data from elsewhere with any systematic variation taken into account. This may not always be possible nor a feasible solution and the possibility that local variation in meteorology can have a substantial effect on modelling outcomes should always be borne in mind.

For some parameters needed to run ADMS there is a great scarcity of data. Even for the more widely recorded parameters such as wind speed and direction, certain areas of the country are better served than others. Cloud cover, in particular, is not frequently recorded, except at the many climatological sites that take daily readings at 0900Z. In Northamptonshire itself there are no synoptic sites and only two climatological sites that record cloud cover. Table 4.1 gives an analysis of meteorological monitoring carried out during 1999, in Northamptonshire and in neighbouring counties.

County	Number of Synoptic sites	Sites that record cloud cover at 0900Z
Northamptonshire	0	2
Cambridgeshire	1	6 (1 only intermittently)
Bedfordshire	0	1 (until June)
Buckinghamshire	1	1
Warwickshire	2	3
Leicestershire	1 (no cloud cover)	1

Table 4.1 Availability of meteorological data during 1999.

The UK Meteorological Office recommend Wittering, the synoptic site in Cambridgeshire, as being most representative for Northampton. It is situated approximately 50 kilometres to the northeast on the edge of the fens.

This chapter examines some of the differences in meteorology that are to be found on a local scale, within Northampton itself and within the county, and on a regional scale by comparing data from Wittering with other synoptic sites. The differences in cloud clover between all the sites listed in Table 4.1 are also examined. Finally the significance of variability between sites on dispersion modelling is considered.

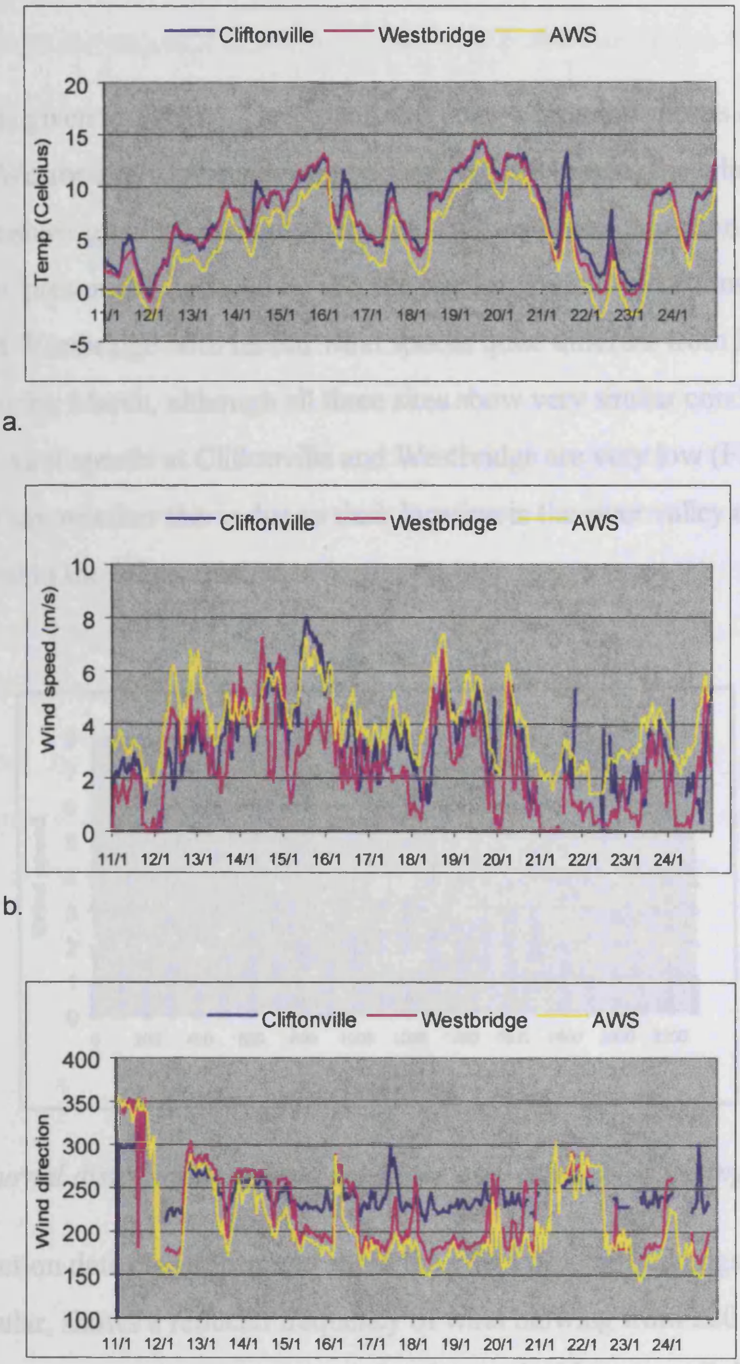
4.2 Differences within Northampton

Meteorological monitoring was carried out in conjunction with the pollution monitoring at two sites within Northampton using the mast that is integral to the Horiba mobile pollution laboratory. This was placed at Westbridge from January to April 1999, at Kingsthorpe from July to November 1999 and at Kingsthorpe again from February to May 2000. Data were also available from the automatic weather station (AWS) at Moulton Park and from the Horiba equipment at Cliftonville, though both these later two sites had technical problems during 1999.

Wind speed, although it follows the same trends over the two-week period shown in Figure 4.1, does show distinct differences between the three sites. Westbridge and Cliftonville appear to generally have lower wind speeds than Moulton Park, though the AWS and the Westbridge site are both sheltered by buildings to the south-west, which is the direction of the prevailing wind. The higher wind speeds at the AWS may be due to its increased altitude (60m) relative to the other two sites. It is not possible to make direct comparison as the AWS has a 1.25ms^{-1} cut off point below which actual wind speeds are not recorded.

Sites	r^2 value (Wind speed)	r^2 value (Temperature)
Cliftonville/Westbridge	0.66	0.91
Cliftonville/AWS	0.68	0.91
AWS/Westbridge	0.63	0.96
Cliftonville/Kingsthorpe	0.83	0.92
Kingsthorpe/AWS	0.66	0.99

Table 4.2 Wind speed and temperature correlation coefficients - Northampton.



c.

Figure 4.1 Temperature (a), wind speed (b), and wind direction (c) recorded at Cliftonville, Westbridge and the automatic weather station at Moulton Park over a two-week period during January 1999.

The correlations given in Table 4.2 are calculated using only wind speeds over $1.25ms^{-1}$ except for the Westbridge/Cliftonville comparison, which contains the whole dataset. Although these show greatest similarity between Cliftonville and Moulton Park, there are definite periods, presumably induced by specific meteorological conditions, during which Cliftonville and Westbridge both record wind speeds quite different from Moulton Park. For example during March, although all three sites show very similar conditions during the day, nighttime wind speeds at Cliftonville and Westbridge are very low (Figure 4.2). It is not possible to say whether this is due to their location in the river valley or because they are more central to the urban area.

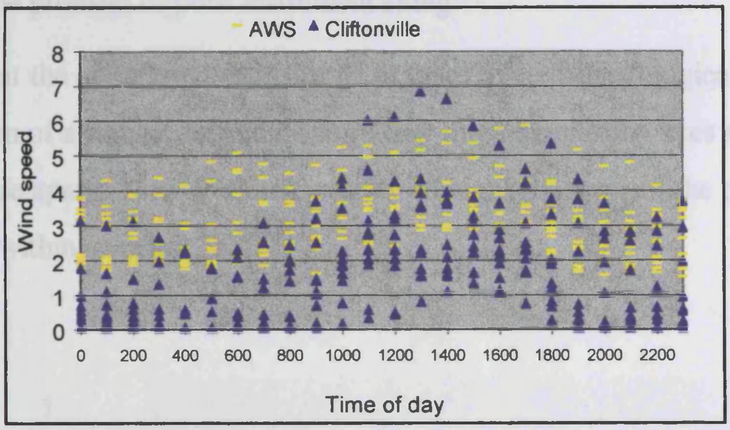


Figure 4.2 Diurnal distribution of wind speed for a 10 day period during March 1999.

The wind direction data also appears to show the effect of local buildings. Westbridge data, in particular, shows a reduced frequency of wind blowing from 220 degrees. The wind direction for some of the time is very closely matched between all three sites, but generally tends to be most similar between Westbridge and the AWS at Moulton Park.

As might be expected, there is very little variation in temperature across Northampton except on occasions it appears to be much warmer at Cliftonville. This is entirely due to the poor siting of the temperature probe. The AWS readings tends to be cooler. Again this may be an effect of increased altitude or because it is nearer to the edge of the urban area. Temperature shows much less influence of local conditions and so r^2 values are all over 0.9 (Table 4.2).

During summer the Horiba mobile was moved to another site within the town, nearer to the Moulton Park (Kingsthorpe). Data from a two-week period are shown in Figure 4.3. The wind vane had ceased to work at Cliftonville so wind direction comparisons could no longer be made, but there was close correlation between Kingsthorpe and the AWS on the majority of occasions. The wind speed showed more agreement during this period than it did in the winter. The AWS still tended to record higher wind speeds, but the $1.25ms^{-1}$ cut off point meant that the low winds that tend to occur overnight in summer were not being recorded. Again temperature readings were very similar, with correlations of over 0.9 between all sites. The higher day time temperatures recorded at Cliftonville again demonstrates the problem of poor instrument siting.

It is possible that the variation demonstrated in these three meteorological parameters, would have been of a similar magnitude using any three monitoring sites spaced an identical distance apart. However these results do highlight some of the problems of recording data within an urban area.

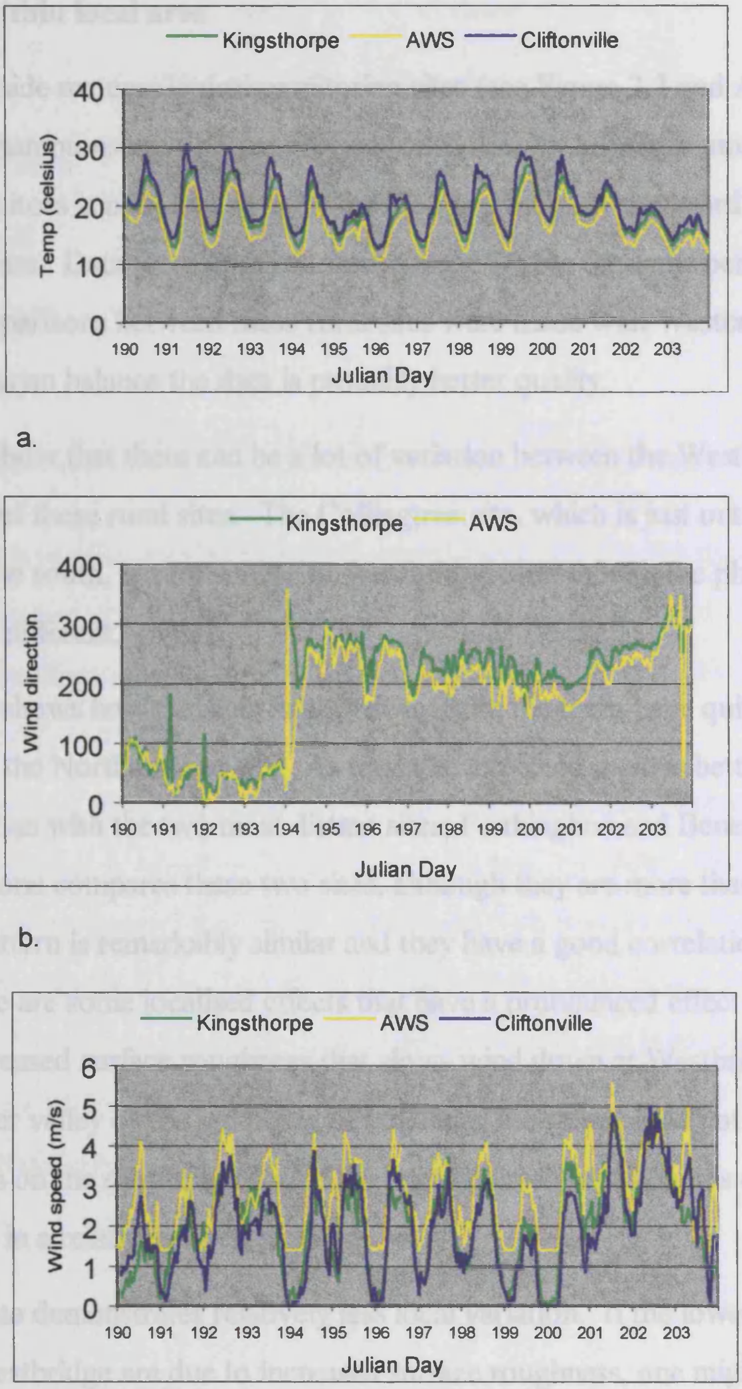


Figure 4.3 Temperature (a), wind direction (b), and wind speed (c) recorded at Cliftonville, Kingsthorpe and the automatic weather station at Moulton Park over a two-week period during July 1999.

4.3 Differences within local area

There are six roadside meteorological monitoring sites (see Figure 3.3 and Appendix Four) within Northamptonshire that are only operated during the winter months. The siting of these monitors is not ideal, as their function is principally to record data for road de-icing programmes. Data were obtained from these sites for the same period as from Westbridge. Comparisons between these rural sites were made with Westbridge rather than Cliftonville, as on balance the data is probably better quality.

Wind speed data show that there can be a lot of variation between the Westbridge site and the furthest away of these rural sites. The Collingtree site, which is just outside Northampton to the south, is very similar to Westbridge even though the physical nature of the site is quite different.

Figure 4.4 clearly shows how the sites further away from the town have quite different wind speeds from the Northampton site. As might be expected there is better correlation with Collingtree than with the two most distant sites; Farthinghoe and Benefield (Table 4.3). However if one compares these two sites, although they are more than 60km apart, the wind speed pattern is remarkably similar and they have a good correlation. This suggests that there are some localised effects that have a pronounced effect on wind speed. If it is increased surface roughness that slows wind down at Westbridge, its location in the river valley or the proximity of buildings, the same could not be said of Collingtree, which on the southerly edge of the built up area and when this data was recorded was still in a relatively open position.

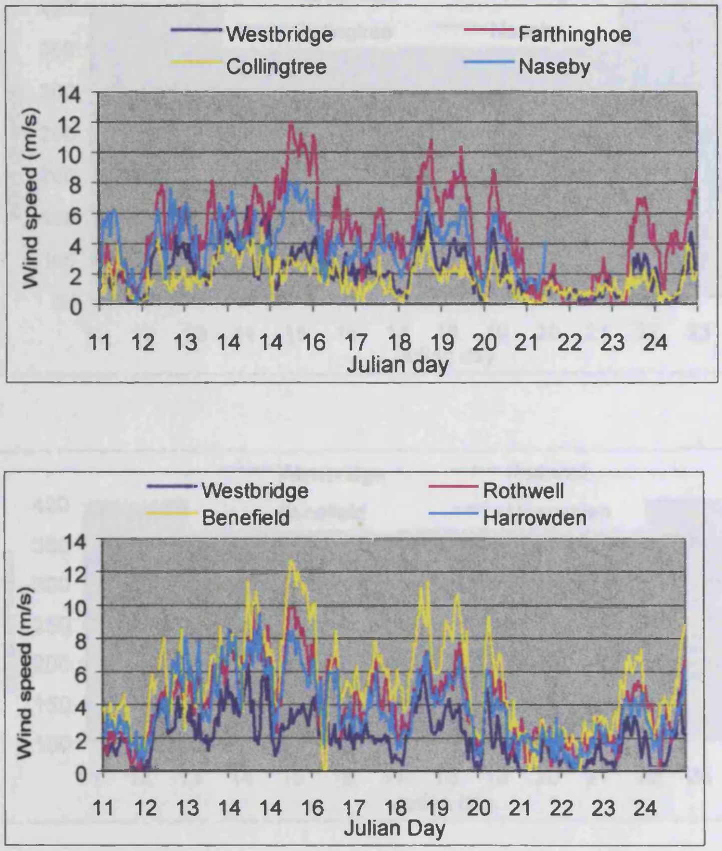
Wind direction data demonstrates relatively less local variation. If the lower wind speeds experienced at Westbridge are due to increased surface roughness, one might also expect a backing of the wind direction. This does not always appear to be the case. Comparison of the two most distant sites, Farthinghoe and Benefield, again shows very little underlying variation across the county.

Sites	r^2 value (Wind speed)	r^2 value (Temperature)
Westbridge/Farthinghoe	0.48	0.97
Westbridge/Benefield	0.54	0.96
Westbridge/Collingtree	0.66	0.98
Benefield/Farthinghoe	0.77	0.95

Table 4.3 Wind speed and temperature correlation coefficients- Northamptonshire.

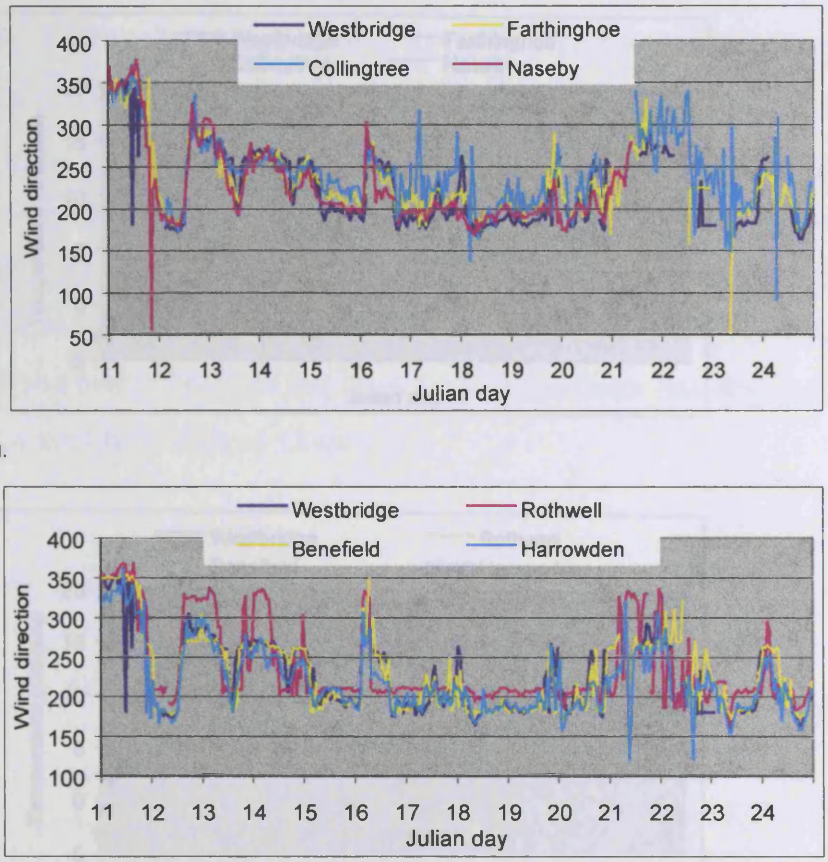
Temperature data shows Westbridge to be consistently warmer than the rural sites (Figure 4.6). All these sites are very similar. The AWS data produces values that closely correspond to the rural sites, showing that there is as much variation within the urban area as there is within a radius of 50 km.

A line running Northeast from Northampton to Wittering, the nearest Meteorological Office synoptic site, takes in Moulton Park, Harrowden and Benefield. Data from these sites show that there is a general increase in wind speed with distance from Northampton. Wind direction is frequently 10 to 20 degrees less at Moulton Park than it is at Wittering. The combination of these two factors does indicate that, in this particular location, increasing the distance between the modelling and the monitoring sites will increase the uncertainty in the input data.



b.

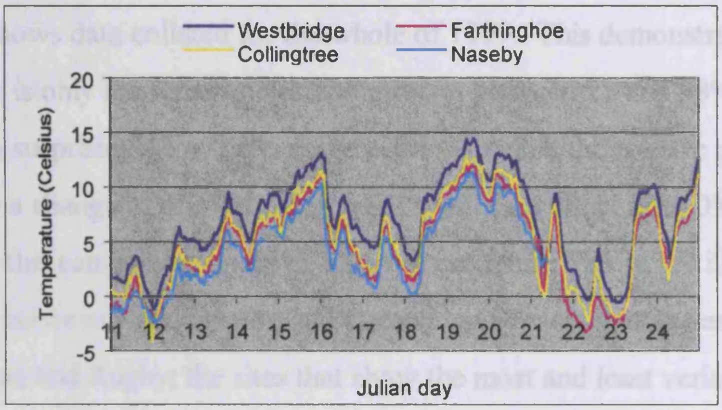
Figure 4.4 Wind speed recorded at Westbridge and six roadside sites over a two-week period during January 1999.



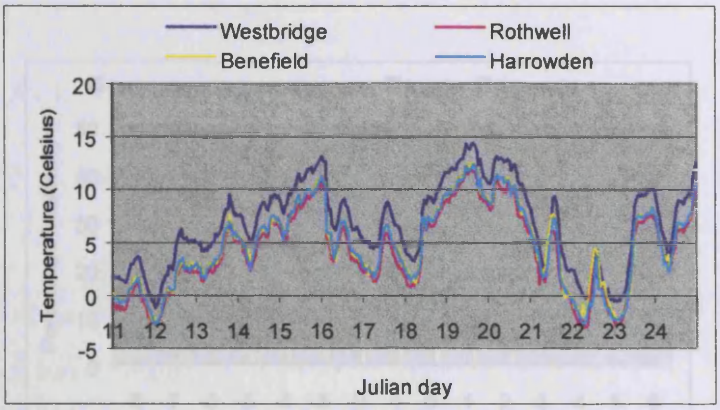
a.

b.

Figure 4.5 Wind direction recorded at Westbridge and six roadside sites over a two-week period during January 1999.



a.



b.

Figure 4.6 Temperature recorded at Westbridge and six roadside sites over a two-week period during January 1999.

4.4 Regional differences

4.4.1 0900Z Cloud cover

Climatological readings are taken daily at 0900Z at Moulton Park and at 12 other sites in Northamptonshire and neighbouring counties (see Appendix Four).

Table 4.4 shows data collated for the whole of 1999. This demonstrates that recorded cloud cover is only the same as Northampton on between 25 and 38% of occasions. This is perhaps a surprisingly low percentage bearing in mind the relative proximity of these sites. Even a change of one oktas can has a significant effect on ADMS output, as the data shows this can happen on up to 33% of occasions. Figure 4.7 illustrates the differences between Northampton and Cambridge Botanic gardens and between Northampton and Rugby; the sites that show the most and least variation respectively. The average is calculated from all 12 sites.

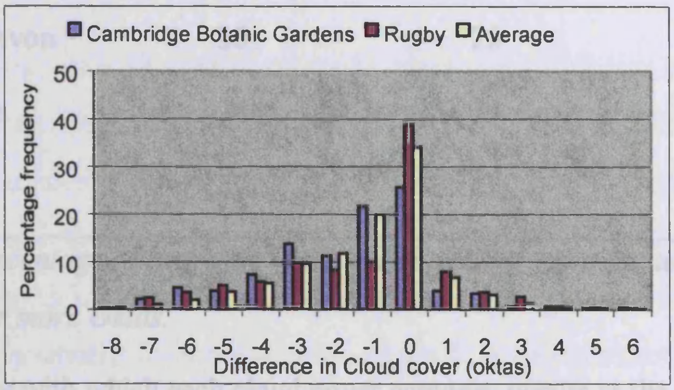


Figure 4.7 Difference in cloud cover between Northampton and other climatological sites - 1999.

Some of the more distant sites show greater similarity to Northampton than near sites. The Mann Whitney test shows some of the close sites to come from the same statistically significant population (p=0.05) such as Rockingham/Rugby, Rugby/Newtown Linsford, Cambridge/Mepal, and Stratford/Wellesbourne.

Climatological site	Cloud cover identical to Northampton	Cloud cover is 1 oktas different	Cloud cover >1 oktas different
Cambridge BG	26	26	48
Cambridge NIAB	34	30	36
Grendon U'wood	37	25	38
Marholm	37	23	40
Mepal	27	28	45
Monk's Wood	27	30	38
Newtown Linsford	35	27	38
Rockingham	34	28	43
Rugby	39	18	36
Stratford u. Avon	36	28	29
Wellesbourne	38	33	29
Woburn	37	25	38

Table 4.4 Percentage of hours on which cloud cover is identical to Northampton or varies by 1 or more Oktas.

The frequency with which each cloud cover category occurs at the different sites also varies considerably. The data suggest that it is nearly always cloudier in Northampton. As cloud cover is assessed visually, it could be that this is a systematic recording error and not a genuine climatological feature, however the observer at Northampton has been trained by the UK Meteorological Office. It has been shown that urban environments are cloudier than surrounding areas (Oke, 1987), but Northampton is unlikely to be large enough to show this effect. Two other sites are also urban; Cambridge Botanic Gardens and Rugby. Cambridge in particular is much less cloudy than Northampton and is the least similar. Northampton and Rugby (the site that shows the greatest similarity to

Northampton) both have a much higher frequency of 8 oktas than any other cloud cover category. At all other sites even though 8 is the most frequent category, there is not such great separation between 7 and 8 oktas. It is not possible to distinguish any systematic trends in this data. The frequency of cloud amounts for three stations are shown in Figure 4.8.

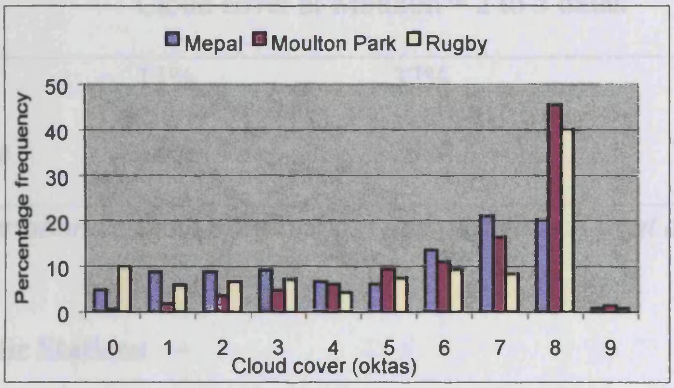


Figure 4.8 Cloud cover distribution at three sites - 1999.

Spearmann rank correlation gives values of over 0.8 for sites that are close together such as Cambridge Botanic Gardens/ Cambridge NIAB and Stratford/Wellesbourne. Obviously lower values are found between distance sites such as Stratford and Monk's wood (0.43) and Grendon and Newtown Linsford (0.55). Moulton Park correlates with all sites in a range between 0.55 (Stratford) and 0.68 (Newtown Linsford). Rockingham, Mepal and Woburn were excluded from the analysis, as the data was too sparse.

The most significant effect this would have on modelling would be if the error was associated with a particular cloud amount, 7 or 8 oktas in summer and 6 or 7 oktas in winter. Although analysis was only carried out between Moulton and two other sites, Wellesbourne and Cambridge Botanic Gardens, it does not appear that there is any increased difference in cloud amounts between stations on days with 7 oktas at Moulton Park (Table 4.5). The figures suggest that large differences are more frequent at low cloud covers.

	Cloud cover identical	Cloud cover differs by up to 2 oktas	Number of records
	Cloud cover at Moulton = 7		
Cambridge	11%	27%	56
Wellesbourne	42%	29%	55
	Cloud cover at Moulton = 2 to 5 oktas		
Cambridge	11%	32%	79
Wellesbourne	18%	29%	79

Table 4.5 Variation between Northampton and other two sites at different cloud covers.

4.4.2 Synoptic Stations

The differences between three synoptic sites in the region were examined. Wittering is the nearest synoptic site to Northampton. Comparisons were made with Coleshill in Warwickshire and Benson in Oxfordshire. Hourly data covering two years (1998 and 1999) for wind speed, wind direction and cloud cover were analysed. Wittering was chosen as the principal site and other sites compared with it.

Cloud cover

There is a difference in the distribution of the cloud cover categories at all three sites and the frequency distribution for the full 24hours is different from that found at 0900Z.

The biggest difference between the synoptic sites in is the occurrence of completely cloudless skies. The data also suggest that there is an increased tendency for obscured skies at 0900Z. This has implications for modelling; ADMS is most sensitive to cloud cover at high oktas values and during early morning or late afternoon.

Cloud cover (Oktas)	Wittering	Benson	Coleshill	Average 0900Z cloud cover (13sites)
0	2.5	10.8	4.8	3.2
1	9.9	8.5	7.1	8.1
2	5.1	4.6	5.1	5.9
3	4.9	4.7	5.1	6.9
4	4.3	3.2	3.9	7.6
5	5.4	5.5	6.2	7.7
6	9.1	6.7	10.5	11.2
7	32.9	33.6	29.7	19.0
8	25.6	22.3	27.5	29.6
9	0.3	0.2	0.1	0.6

Table 4.6 Distribution of cloud cover over a two-year period (1998+1999) at three synoptic stations.

The difference in cloud covers between both Benson and Coleshill, and Wittering was also determined for each individual hour. There are no clear trends that would enable one to distinguish any systematic differences in cloud cover. This might only be expected with more distant sites exhibiting different climatologies; hence a comparison with Aviemore is also included.

To summarise Table 4.7, cloud cover is on average the same on 39% of occasions, it is identical or differs by no more than 1 Oktas 69% of the time, by no more than 2 oktas during 79% or by 3 oktas during 86% of the time. Although there are differences, the data does not show Benson or Coleshill to be significantly more or less cloudy than Wittering. The ADMS sensitivity study (see Chapter Five) shows that even one oktas difference can produce a large variation in predicted CO, but it very much depends on time of day and where the change occurs on the cloud cover scale.

Difference in cloud cover	Benson-Wittering	Coleshill-Wittering	Aviemore-Wittering
-8	0.6	0.2	0.4
-7	2.1	0.8	1.5
-6	2.2	1.2	2.5
-5	2.4	1.5	3.0
-4	2.9	2.2	2.9
-3	3.7	3.0	3.2
-2	5.3	5.0	4.6
-1	16.0	14.5	14.6
0	37.5	40.6	25.9
1	13.6	16.7	14.0
2	4.7	5.3	6.4
3	2.8	3.4	4.2
4	2.3	2.6	4.3
5	1.8	1.6	4.2
6	1.5	1.0	5.5
7	0.4	0.3	2.5
8	0.1	0.1	0.4

Table 4.7 Percentage differences in cloud cover between Wittering and three other sites, 1998 and 1999.

Greatest change in model output occur in summer when cloud cover varies between 7 and 8 oktas over the hours 7, 8 and 9am, and in winter when it varies between 6 and 7 oktas during the middle of the day. Data was further sub-divided to look at these specific hours. A summary of the results is given in Table 4.8.

Cloud cover difference	All data points	Summer 7,8 9 am	Winter 11,12,1pm	All hours when cloud =7 at Wittering
0	39	40	44	48
+/- 1	69	74	77	80
+/- 2	79	85	84	*
+/- 3	86	91	89	*
No of records	33775	1087	1075	11149

Table 4.8 Average differences in cloud cover(combined data) as cumulative percentages (Benson and Coleshill, 1998+1999).

** not reported as it is not possible to have +2 or +3 oktas values when cloud cover is 7 at Wittering.*

These results show that on average there is no more variation in cloud cover at the times of day or year when cloud differences could be most crucial to ADMS output. Sensitivity studies also show that model output is only affected by variation in cloud amounts at 6 oktas or more. Therefore using data from a distant site would be more crucial if greater deviations occurred at these high cloud covers values. This does not appear to be the case. As it appears that when cloud cover is 7 oktas at Wittering difference with other sites is less.

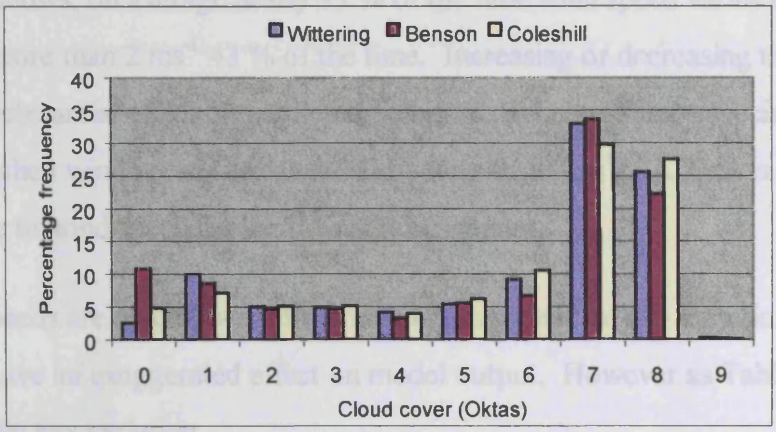


Figure 4.9 Distribution of cloud cover over a two year period at three synoptic stations - 1998 and 1999.

Wind speed

Unlike cloud cover there does seem to be a systematic difference in wind speed. This is not surprising, as it is much more likely to be affected by local topography. Neither Benson nor Coleshill show the high wind speeds found at Wittering. The quality of the data is suspect at Benson because of the large number of times when wind speed is recorded as zero.

	Wittering	Benson	Coleshill
Mean	4.9	3.6	3.8
Median	4.6	3.6	3.6
Maximum	22.6	17.5	16.5

Table 4.9 Wind speed data - 1998 and 1999.

Differences in wind speed were assessed by subtracting the Wittering value from the wind speed at the other two sites, a summary of the differences are given in Table 4.10.

As this shows, on average nearly 73 % of the time wind speed varies by more than 1 ms^{-1} and by more than 2 ms^{-1} 43 % of the time. Increasing or decreasing the wind speed can have a substantial effect on predicted pollutant concentrations, especially if this variation occurs when wind speeds are already low (less than 3ms^{1}). ADMS is found to be more sensitive to wind speed during the night in summer.

Wind speeds are much lower at night which suggests that any variation between sites would have an exaggerated effect on model output. However as Table 4.12 shows there is actually less variation.

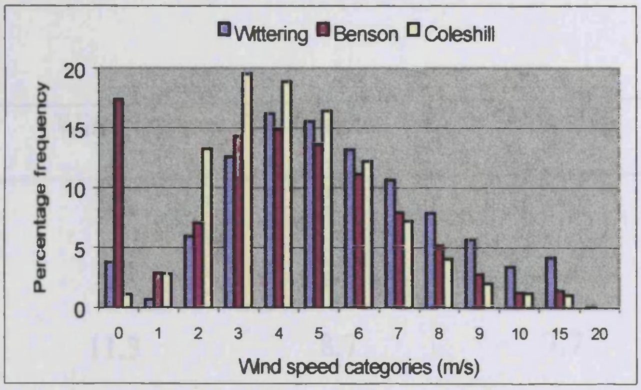


Figure 4.10 Wind speed distribution at three synoptic stations - 1998 and 1999 (Note change in scale).

	Benson	Coleshill	Ave
+/- $1ms^{-1}$	23.9	29.3	26.6
+/- $2ms^{-1}$	51.5	62.7	57.1
+/- $3ms^{-1}$	73.9	83.6	78.6
+/- $5ms^{-1}$	95.4	97.7	96.5
+/- $7ms^{-1}$	99.3	99.7	99.5
+/- $10ms^{-1}$	99.9	99.9	99.9

Table 4.10 Wind speed differences - 1998 and 1999(cumulative percentages).

	Wittering	Benson	Coleshill
Mean	3.6	2.5	2.8
Median	3.6	2.6	2.6
Maximum	11.3	8.7	7.7

Table 4.11 Wind speed characteristics during for night-time, May June July 1998 and 1999.

	Benson	Coleshill	Ave
+/- $1ms^{-1}$	28.3	29.9	29.1
+/- $2ms^{-1}$	58.6	67.4	63.0
+/- $3ms^{-1}$	79.6	89.8	84.7
+/- $5ms^{-1}$	97.8	99.7	98.8
+/- $7ms^{-1}$	99.9	100	100

Table 4.12 Wind speed differences for night-time, May, June and July 1998 and 1999 (cumulative percentages).

Wind direction

As might be expected for sites that are relatively close together and are not affected by any unusual topographical features such as mountain ranges or coastline, the wind direction distribution is very similar. All show a more or less bimodal distribution. Wittering has modal frequency groups between 0 to 20 degrees and from 220 to 240 degrees. Both Benson and Coleshill seem to show a relative backing of the wind as both have 180 to 200 degrees as the main modal group. Figure 4.12 shows the distribution of wind direction differences and from this it is clearer that there are differences between the two sites even though they are both to the west of Wittering and have similar terrain. A summary of these differences shown as cumulative percentages is given in Table 4.13.

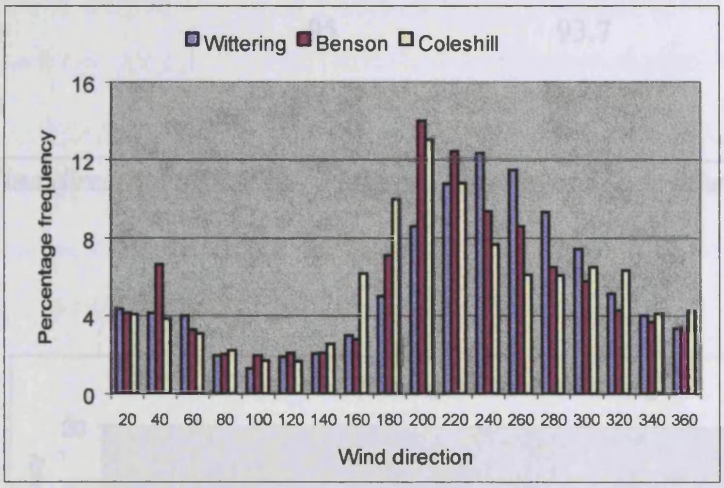


Figure 4.11 Wind direction distribution for three synoptic stations - 1998 and 1999.

In terms of absolute variation there is very little difference between Benson and Colehill and in fact relatively little difference between these sites and Wittering. The accuracy of the monitoring equipment is to within 10 degrees and from this data, on nearly 50% of occasions the difference is +/- 10 degrees. This is within the range of acceptable error. Wind direction only varies by more than 90 degrees during less than 4% of hours. However it is the direction that the wind is actually blowing from that has a crucial effect on air pollution modelling and at certain angles to the line source even difference of 10 degrees in wind direction could have a significant effect on predicted pollutant concentrations.

Difference in Wind direction	Benson - Wittering	Coleshill- Wittering	Average
0 degrees	18.4	18.7	18.5
+/-10 degrees	48.8	48.4	48.6
+/-20 degrees	68.6	67.5	68.1
+/-30 degrees	79.5	78.7	79.1
+/-50 degrees	90.6	89.8	90.2
+/-70 degrees	95	93.7	94.3
+/-90 degrees	97.3	95.9	96.5

Table 4.13 Wind direction differences between Wittering and two other synoptic stations.

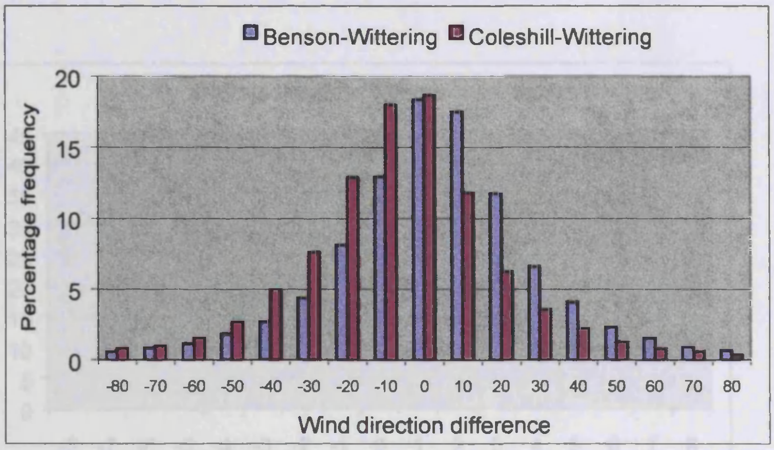


Figure 4.12 Wind direction differences between Wittering and two other synoptic stations - 1998 and 1999.

Differences at low wind speed

Pollutant concentrations at the side of a busy road will be higher when wind speeds are low. Therefore it is useful to determine whether or not there is more variability in the cloud cover and wind direction when speeds are low. When pollutant levels are already high, there is potentially more error introduced into the model output with the use of inaccurate meteorological data.

The data for Wittering, Benson and Coleshill were sorted into three wind speed categories; 0 to 3ms^{-1}, >3 to 6ms^{-1} and >6 to 9ms^{-1}. The difference in cloud cover and wind direction between Wittering and the two other sites were then determined for each of these categories.

Cloud cover shows only slightly more variation at low wind speed; the percentage of hours within each category is given in Table 4.14 and presented graphically in Figure 4.13.

Wind direction differences do show more sensitivity to wind speed. At low wind speeds on over 50% of occasions does the wind direction differs by up to 30 degrees. At very low wind speeds wind direction will have little influence, but at speeds of 2-3ms^{-1}, this may potentially have greater significance than cloud cover variation.

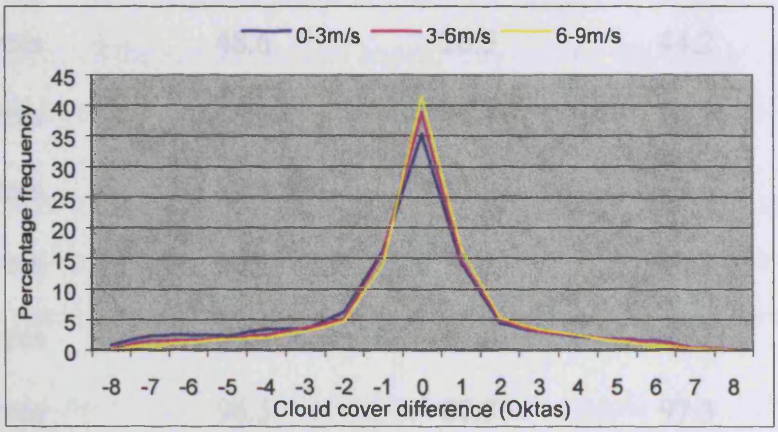


Figure 4.13 Difference in cloud cover by wind speed category.

	Wind speed category			
Cloud cover difference	All data points	0-3ms^{-1}	3-6ms^{-1}	6-9ms^{-1}
0	39	35	39	42
+/- 1	69	66	69	72
+/- 2	79	76	79	82
+/- 3	86	82	86	89
Total no. in category	33775	7775	15228	8119

Table 4.14 Difference in cloud cover by wind speed category (cumulative percentage).

	Wind speed category			
Difference in Wind direction	All data points	0-3ms^{-1}	3-6ms^{-1}	6-9ms^{-1}
0 degrees	18.5	9.4	16.1	24.4
+/-10 degrees	48.6	26.5	44.2	62.1
+/-20 degrees	68.1	41.4	64.7	83.4
+/-30 degrees	79.1	53.3	77.6	92.6
+/-50 degrees	90.2	71.2	90.7	98.1
+/-70 degrees	94.3	81.4	95.1	99.4
+/-90 degrees	96.5	87.8	97.3	99.7
Total no. of records in category	29929	5289	14011	8018

Table 4.15 Difference in wind direction by wind speed category (cumulative percentage).

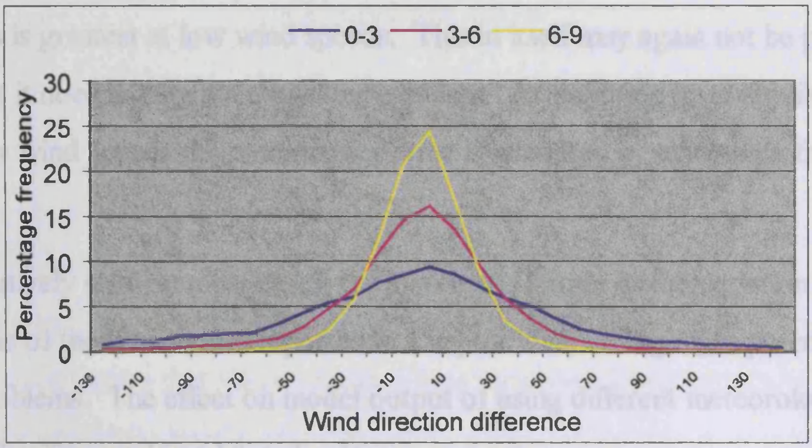


Figure 4.14 Difference in wind direction by wind speed category.

4.5 Conclusions

Clear systematic differences are apparent between the four urban monitoring sites. There are also clear differences between data from the Westbridge site in Northampton and data obtained from roadside sites in the county. It is not possible to say whether this is due to the urban environment or the topography. However the two most distant sites show remarkable similarity with each other. Variation seems to increase with distance from Northampton. The Westbridge site is consistently warmer than the rural sites. It should be noted that of all the meteorological inputs, temperature has least effect on modelling outcome and its use is only recommended as a refinement.

The difference in cloud cover taken at 0900Z is perhaps more surprising bearing mind the relatively small geographical area that the data is collected from. Some of these differences may be attributable to the manner in which the data is collected.

Some obvious differences are also noted between the synoptic sites. Although the meteorological parameters may show diurnal trends this is not reflected in the variation between sites. A feature that could further increase model unreliability if error was greatest at times of times of enhanced sensitivity.

Another key point raised by this data, however, is that the variation found between synoptic sites is greatest at low wind speeds. This in itself may again not be particularly surprising but it does highlight a modelling problem. As pollution level are likely to be highest at low wind speeds the potential for error is increased by using data from a remote site.

Although relatively small in most cases, the systematic nature of the variation between sites and some of the sensitivities reported in Chapter Five, do highlight potential modelling problems. The effect on model output of using different meteorological datasets will be discussed in a later chapters.

CHAPTER FIVE

MODEL SENSITIVITY

5.1 Introduction

Although many papers relating to ADMS have been published, as yet there have been no scientific research papers that relate specifically to sensitivity in ADMS-Urban. The function of this sensitivity testing is to determine which input parameters model output is most sensitive to. The study was restricted to those meteorological parameters that would be used in ADMS during the course of its normal use within a regulatory framework. The model response varies at different times of day and users of the model need to be aware of this and its effect on model uncertainty. The results of the sensitivity study were then used to determine further, more specific analysis of ADMS performance.

Sensitivity analysis can vary in its design from a simple approach of changing selected input parameters one at a time to a complex multi-dimensional framework. A brief review of sensitivity testing is given in Appendix Nine.

5.2 Model Scenario

A simple model scenario was defined within ADMS consisting of a single 500m long, line source running from east to west. Four receptor points were placed mid-way, to the north of the line, at distances of 5m, 10m, 15m and 20m. Vehicle flows were set at 2000 light-duty vehicle and 100 heavy-duty vehicles per hour, travelling at 60km per hour. These data were chosen as representative of a fairly busy road in an average sized town. The surface roughness was set at 0.5m. No minimum Monin Obukhov length was set as it was found in preliminary model runs that it makes very little difference to model output except at wind speeds of less than $2ms^{-1}$ and even then the differences are only seen at night. However a minimum value of 30m was used for the model runs in which wind speed was varied. The surface roughness value and the minimum Monin Obukhov length are the values recommended by CERC for the type of environment found in Northampton.

Model runs were carried out for a representative summer and a winter day (Julian day 173 and 356 respectively), with and without time varying emissions. The time varying emission factors used were determined by traffic surveys carried out in Northampton during 1999. How the time varying emissions vary over the course of a weekday is represented graphically in Figure 5.1. Model runs without time varying emission factors give a much clearer picture of response to the meteorological input parameters alone, but using them shows the model response in terms of predicted pollutant in a more realistic way.

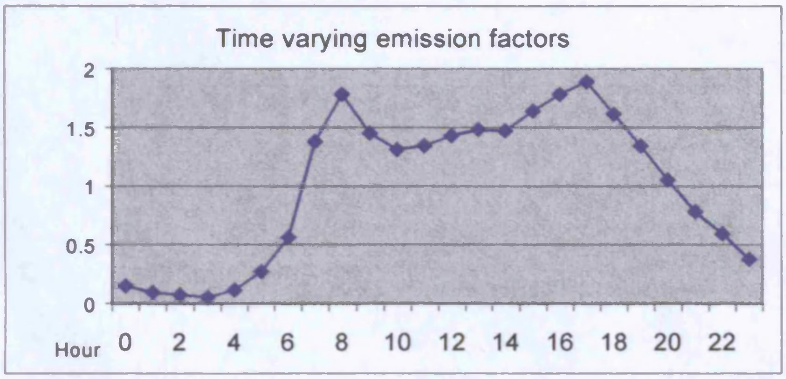


Figure 5.1 Time varying emission factors.

The model was always run with meteorological parameters held constant for twenty-four hours in order that ADMS could effectively predict changing boundary layer characteristics during the day. The carbon monoxide concentration at each of the four receptor points was predicted for each hour as an hourly average. A baseline set of meteorological parameters was defined; each was then varied in turn, within a pre-defined range. The baseline values and the ranges chosen for these variables is considered representative of any site in Southeast England. Details are given in Table 5.1. The increments with which the variables were changed also reflect the type of variation found between modelling sites and their nearest synoptic station. Real life situations are invariably more complex as, even if trends are discernable, deviation between sites will change in magnitude and direction over the course of time. Surface roughness and albedo were included to demonstrate the effects of wrongly categorising the urban area and the possible error resulting from the use of one value as a blanket classification for the entire urban area.

Parameter	Baseline value	Range	Increments
Wind speed	$2ms^{-1}$	$0.75 - 6ms^{-1}$	$0.5ms^{-1}$
Wind direction	210°	180 - 330°	10°
Temp (summer)	20°C	8 - 26°C	2°C
Temp (winter	5°C	-3 - 13°C	2°C
Cloud cover	7 oktas	0 – 8 oktas	1 okta
Surface roughness	0.5m	0.005 – 10m	variable
Albedo	1.0	0 - 1.0	0.2

Table 5.1 Meteorological parameters used in the sensitivity study.

5.3 Results

Although it is unrealistic to model constant conditions, it is useful to see how in this situation, ADMS predicts changing boundary layer characteristics and hence CO concentrations over the course of 24 hours. The model was run with the baseline set of condition and the initial output analysed, subsequently each parameter was changed in turn and the effect on model output examined. Data is presented graphically in two ways. Firstly with two or more levels of the input parameter under investigation as a function of time and secondly to show how the ADMS internal parameters change at two or more specific hours as a function of changing input. These times were selected as representative of periods of greatest and least sensitivity.

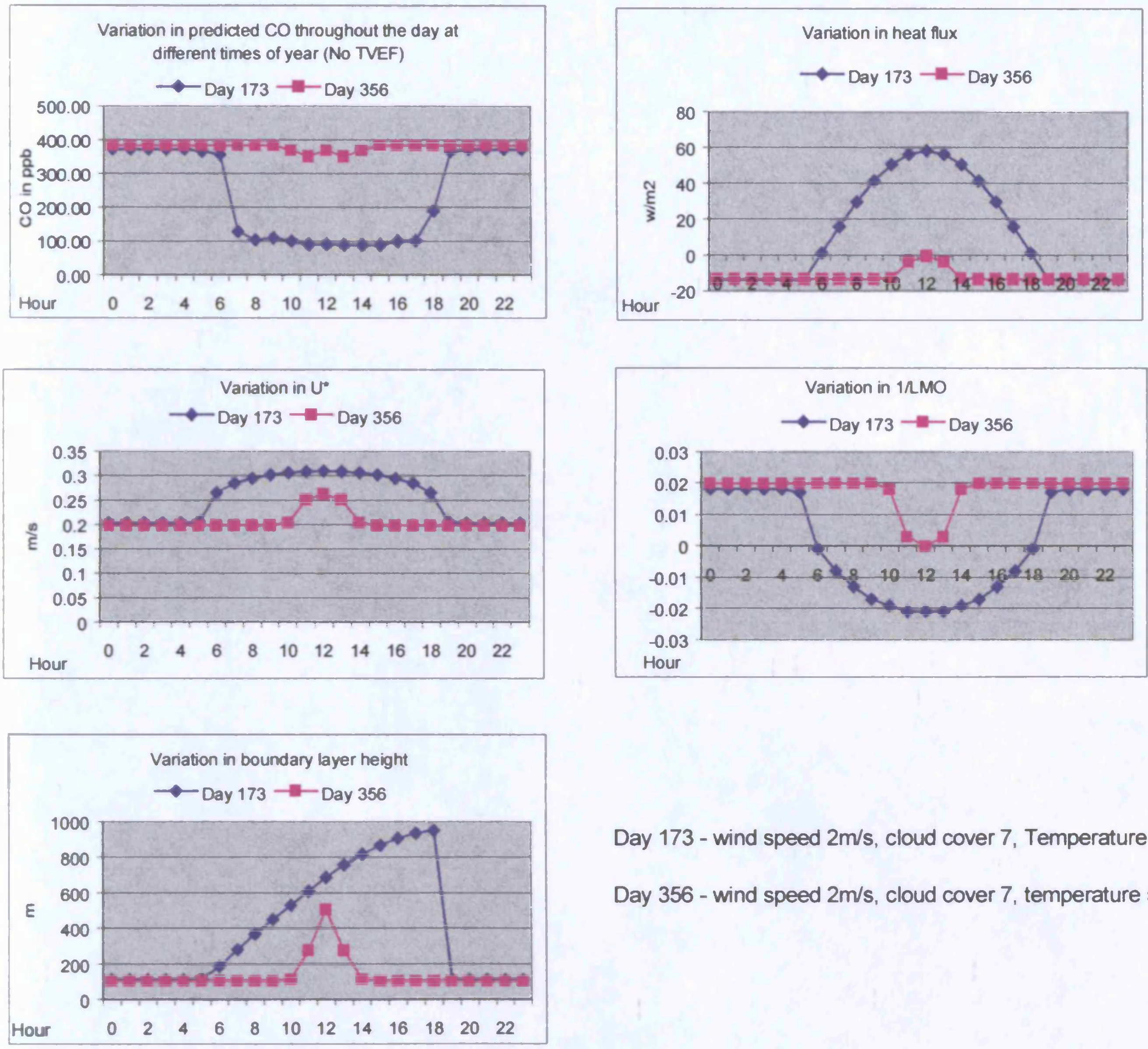
5.3.1 Baseline Conditions

Figure 5.2 shows how the model responds to the baseline conditions for Julian day 173 and day 356.

The reciprocal of the Monin Obukhov length shows how with this data ADMS determines the atmosphere to be stable between 1900Z and 0500Z and unstable between 0700Z and 1700Z, during summer. The boundary layer becomes slightly unstable during the day even with a large amount of cloud cover. Although the boundary layer height develops slowly during the day, the change from unstable to stable conditions in the evening results in a sudden drop in height. In winter, the atmosphere is stable for most of the time and only becomes neutral in the middle of the day. The changes in stability, friction velocity and heat flux are mirrored exactly by the rise and fall in the boundary layer height.

If emissions are constant, with this particular set of conditions the model predicts falling CO concentrations as the atmosphere becomes either unstable or neutral. The most notable difference between winter and summer is the degree to which this happens. In summer, although the predicted output falls rapidly in the morning the rise again during late afternoon is more gradual and more closely reflects the changes in heat flux and stability rather than boundary layer height. In fact CO concentrations start to increase again whilst the boundary layer is still rising. Even with a large amount of atmosphere in which dilution of pollutant can occur, it appears that dispersion is stifled.

In winter, although the general trend is for CO concentrations to fall during the middle of the day, there is an anomaly that is not easily explained by how the pre-processor is working. Predicted CO does not follow the trends exhibited by boundary height, heat flux etc. and rises slightly at 1200Z.



Day 173 - wind speed 2m/s, cloud cover 7, Temperature 20

Day 356 - wind speed 2m/s, cloud cover 7, temperature 5

Figure 5.2 Response of ADMS-Urban to constant input variables over a twenty four-hour period – summer and winter

5.3.2 Sensitivity to wind speed

The principal effect of increasing wind speed is to bring about a reduction in predicted CO. In summer, this effect is most noticeable at night when increased wind speed has greatest effect on increasing the negativity of the heat flux and on the reciprocal of the Monin Obukhov length. Friction velocity responds to increasing wind speed in a similar fashion regardless of time of day. The reduced effect of wind speed during the day can in part be explained by the fact that the heat flux remains constant regardless of increasing speed. Boundary layer height although it increases with increasing wind speed at all times is most sensitive at 0600Z. This is a period of transition when the heat flux is changing from negative to positive and is already completely unaffected by wind speed. See Figure 5.4.

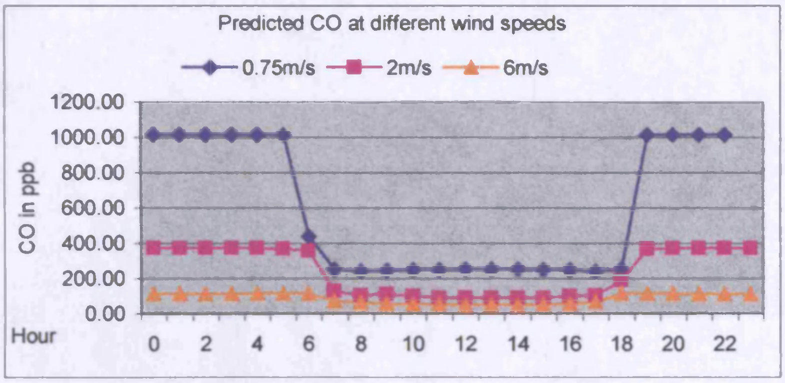
The trend from unstable towards neutral that occurs during the day with increasing wind speed has less effect on predicted CO than the decline in stability during the night. Treating the $2ms^{-1}$ wind speed value as a standard and relating the predicted CO to that with either lesser or greater wind speeds shows that ADMS is more sensitive to increased wind speeds at night than during the day. An increase of $1.5ms^{-1}$ over and above $2ms^{-1}$ is needed to halve predicted CO at night, but it is not until it reaches $6ms^{-1}$ that it is halved during the day. A reduction in wind speed to $1ms^{-1}$ can effectively double the CO prediction and a reduction to $0.75ms^{-1}$ can almost triple it during the night and early afternoon (Figure 5.3). This seems to be driven largely by the effect of wind speed on heat flux and to a lesser degree by how the friction velocity responds at different times of day. In terms of CO output, the model is actually least sensitive to wind speed at 6am and 6pm (see Figure 5.4), even though these are times when boundary layer height is greatly affected.

During the day, in particular, there is very little further reduction on CO once the wind speed has increased to $4.5ms^{-1}$. As hourly wind speeds recorded at Wittering are over $5ms^{-1}$ during approximately 45% of hours, only if frequent very large discrepancies occurred would this data cause a problem for modelling in Northampton.

The model output described above was produced with a cloud cover of 7 oktas, this amount of cloud would not be expected to produce particularly strong convective conditions during the day. Although in summer there could be occasions when convective activity is strong. The model runs were repeated with a cloud cover of 0 oktas (Figure 5.5). With clear skies ADMS is even more sensitive to wind speed at night. At 3am and with a wind speed of $0.75ms^{-1}$ ADMS predicts 50% higher CO than with a cloudy sky. This is almost certainly because the stability is much lower with cloudy skies giving greater mixing in the lowest layers. Once wind speeds have reached $3ms^{-1}$ the cloudiness of the sky makes no difference to model output. The atmosphere manages to remain stable for longer and it takes higher wind speeds to bring about neutral conditions. Higher wind speeds also bring about a negative heat flux of greater magnitude when skies are clear, but this has no effect on model output. The cloud cover only has a minimal effect on CO during the day even though boundary layer depth and heat flux are larger. Cloud cover appears to have no direct effect on the nature of the response of ADMS to changing wind speed.

In winter the effect of wind speed is more or less constant over the course of the day and there is only a slightly different response seen at midday, at low wind speeds (Figure 5.6and 5.7a). The data for 0300Z shown in Figure 5.7 is representative of conditions from 1500Z to 0900Z and shows that the model responds in a similar manner in winter as it does in summer between 1900Z and 0500Z. However in winter the atmosphere is assumed to be stable at midday with low wind speeds and it only requires a small increase in speed to bring about neutral conditions. At night a greater increase in speed is required to achieve this.

In winter CO levels are generally higher and the model shows greater sensitivity. Reducing the wind speed from $2ms^{-1}$ to $0.75ms^{-1}$ results in a quadrupling of the CO regardless of time of day (Figure 5.6). Even a reduction to $1ms^{-1}$ will more than double model output.



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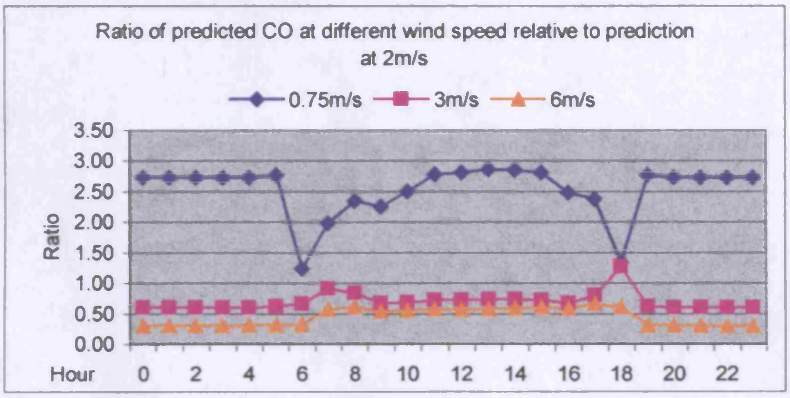

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Figure 5.3 Variation in predicted CO at different wind speeds (7 oktas cloud cover) - day173.

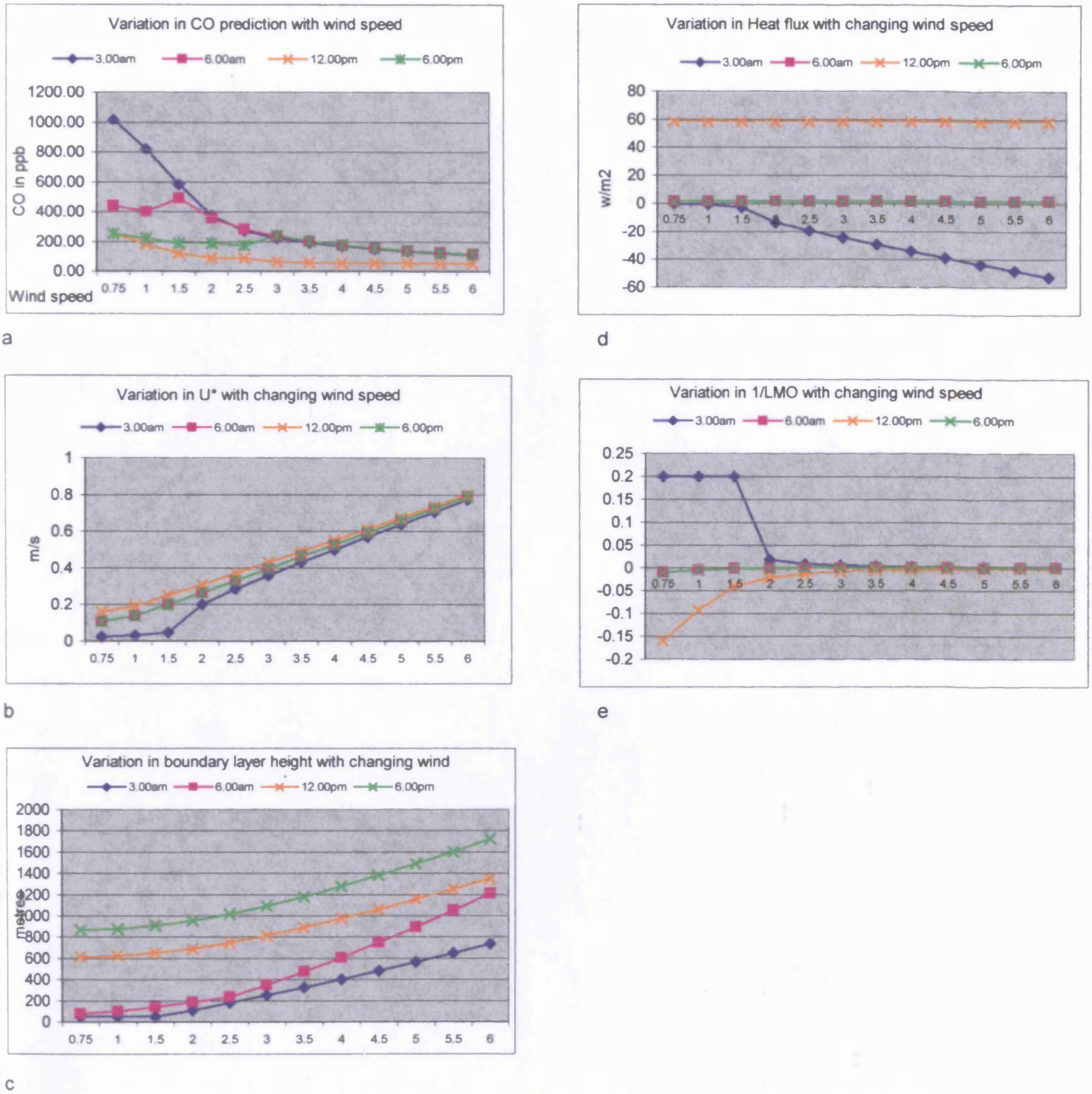


Figure 5.4 Model sensitivity to changing wind speed (7 oktas cloud cover) - day 173.

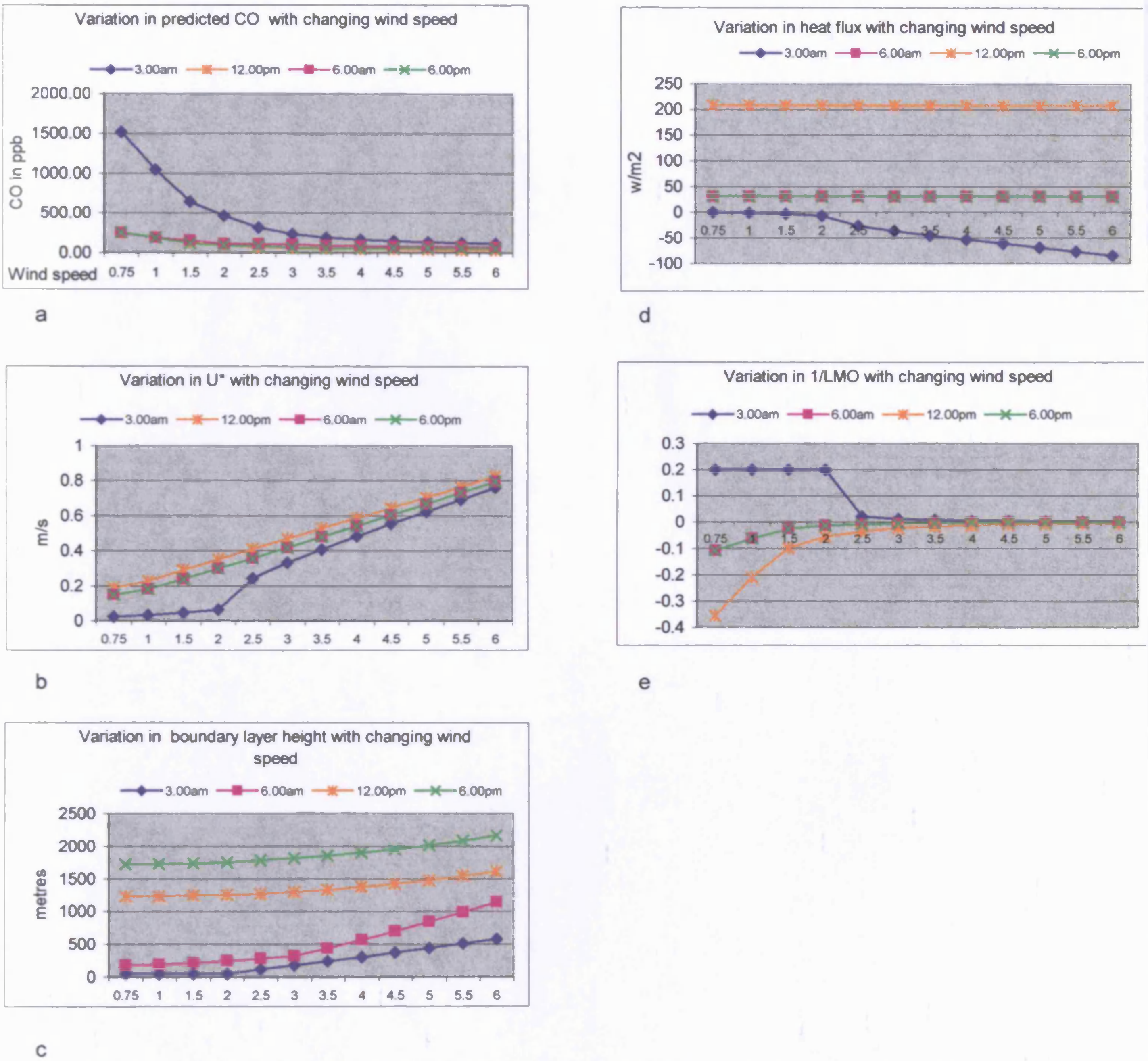
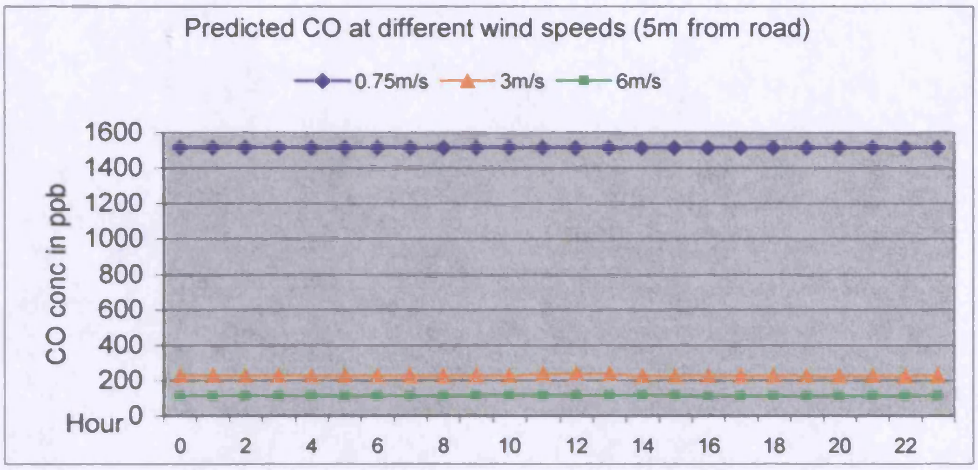


Figure 5.5 Model sensitivity to changing wind speed (0 oktas cloud cover) - day 173.



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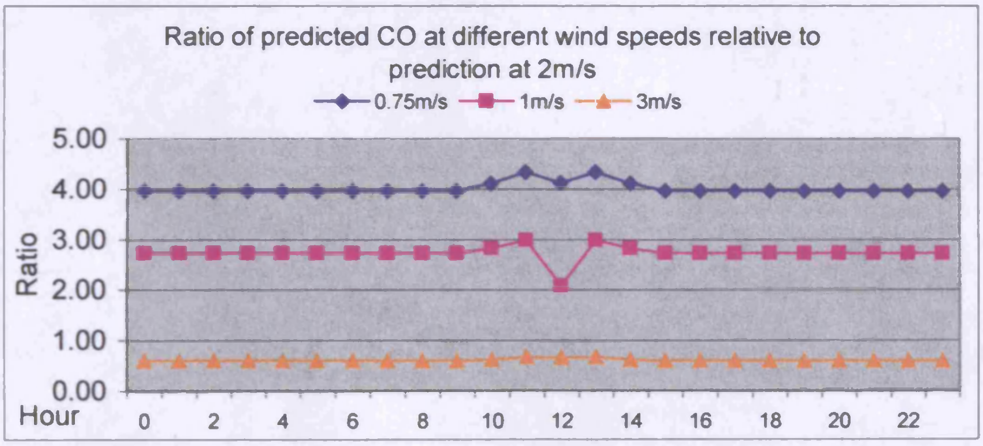

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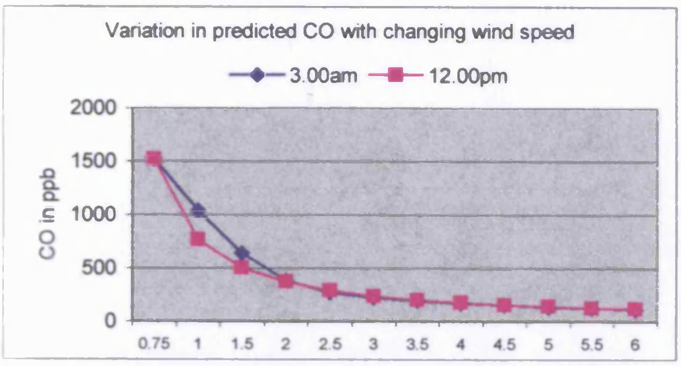
Figure 5.6 Variation in predicted CO at different wind speeds - day 356.



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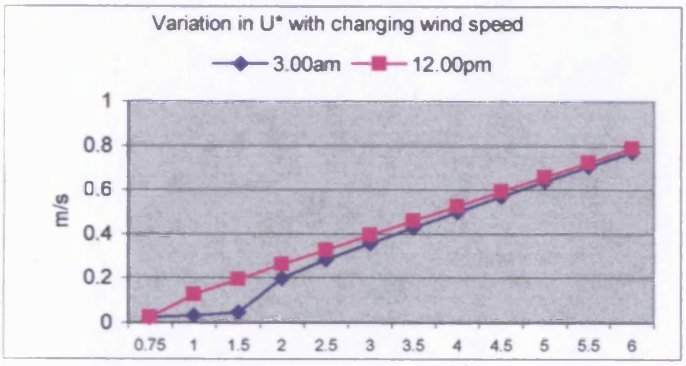

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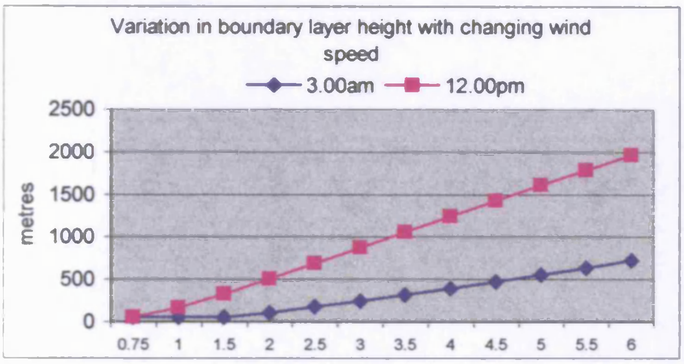

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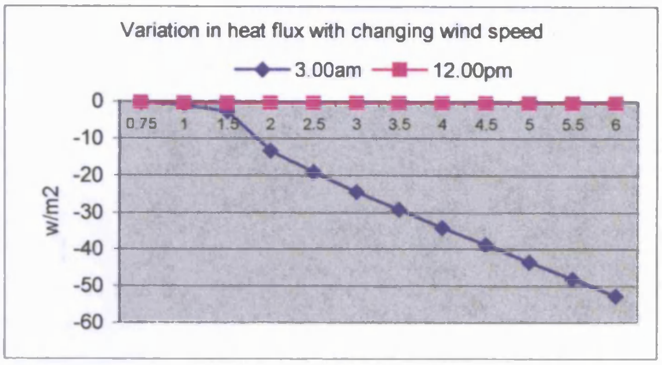

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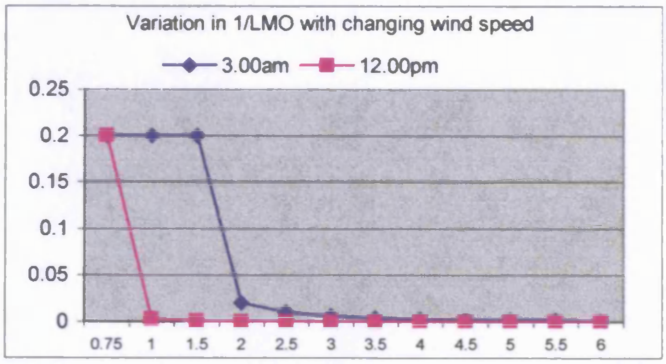


e

Figure 5.7 Model sensitivity to changing wind speed - day 356.

5.3.3 Sensitivity to cloud cover

In summer, at night, cloud cover affects parameters such as friction velocity over the complete range of cloud covers and still does even during sunset and sunrise. However during the day friction velocity, boundary layer height, and 1/LMO are virtually unaffected by cloud cover until it reaches 5 oktas, then as cloud cover increases they all show a decrease in values. As a result, the 1/LMO shows a change from unstable to almost neutral.

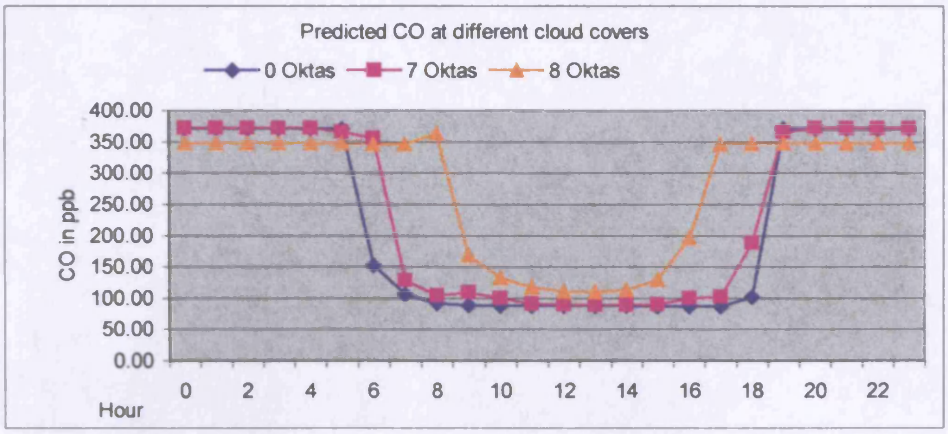
The effect on predicted CO is only really noticeable when cloud cover changes between 7 to 8 Oktas (Figures 5.8 and 5.9). Not surprisingly changing cloud cover on its own has little effect at night. Although a small amount of cloud (3 to 5 oktas) does result in higher CO levels than completely clear or completely obscured skies. At 5am, when the sun is low in the sky, the opposite effect is shown with a dip in CO levels from 2 to 6 oktas. This is presumably caused by increased radiation levels, over all wavelengths, due to reflection and scatter from the base of clouds and the ADMS meteorological pre-processor seems to be taking this into account when it calculates the various boundary layer parameters. The reflection is not so pronounced when the sun is high in the sky and there a slight increase in predicted CO as cloud cover increases. However ADMS is most sensitive to cloud cover between the hours of 0500Z and 0900Z, and between 1600Z and 1900Z. These are periods of transition when the sun is either rising or setting and the amount of radiation reaching the earth's surface appears to be most crucial. Using the input data given ADMS predicts a similar response in boundary layer conditions to changing cloud cover at 0800Z and 1200Z; this does not explain why the predicted CO concentrations rise rapidly at 8 oktas at the earlier time. In the morning and later afternoon increasing cloud cover from 7 to 8 oktas can bring about a four-fold increase in CO. These are incidentally times when the model is least sensitive to wind speed (compare Figure 5.3 and 5.8).

Some of these responses to changing cloud cover can be explained by the findings of Kasten and Czeplak (1980), and Haurwitz (1945). In summer global radiation levels increase slightly at one okta relative to clear skies and then decline as cloud cover increases (regardless of solar elevation). Diffuse radiation, however, increases up to 6 oktas before declining sharply. In this case the effect is more pronounced at higher solar elevations

In winter, ADMS is particularly sensitive to cloud cover during the middle of the day (Figure 5.10 and 5.11). The effect of cloud cover on boundary layer parameters as determined by the ADMS pre-processor is complex (see Figure 5.11). For example heat flux increases and then decreases steadily from 5 oktas, becoming negative at 7 and 8 oktas during the middle of the day. During mid-morning the boundary layer height increases up to 4 oktas cloud cover and then drops again whereas at midday it is more or less constant until it rises rapidly at 7 oktas, then falls again at 8 oktas. The atmosphere is only marginally unstable at midday with low cloud covers, is neutral at 7 and stable at 8 oktas.

Some of these features can again be explained by the balance in the different types of incoming radiation as shown by Kasten and Czeplak (1980). In winter global radiation increases at one oktas but then decreases rapidly, diffuse radiation reaches a peak at 6 oktas and long wave radiation increases rapidly as cloud cover increases. Although this explains the behaviour of the model at 1000Z, it does not explain why boundary layer height has a peak at 7 oktas at 1200Z. This is contrary to the increase in CO exhibited between 6 and 7 oktas (Figure 5.11a). Again one might expect a greater depth of boundary layer to result in greater dispersion of pollutants.

From 1400Z to 1000Z predicted CO levels decrease as cloud cover increase. The response in winter at 2200Z is similar to summer at 0500Z, as they have similar solar elevations at mid-latitudes. Between 1100Z and 1300Z increasing cloud cover initially has the effect of decreasing CO and then the transition between 6 and 7 oktas results in a large increase in CO. The model shows greatest sensitivity to cloud cover at midday. Overall the effect of cloud cover on the model is less than that of wind speed.



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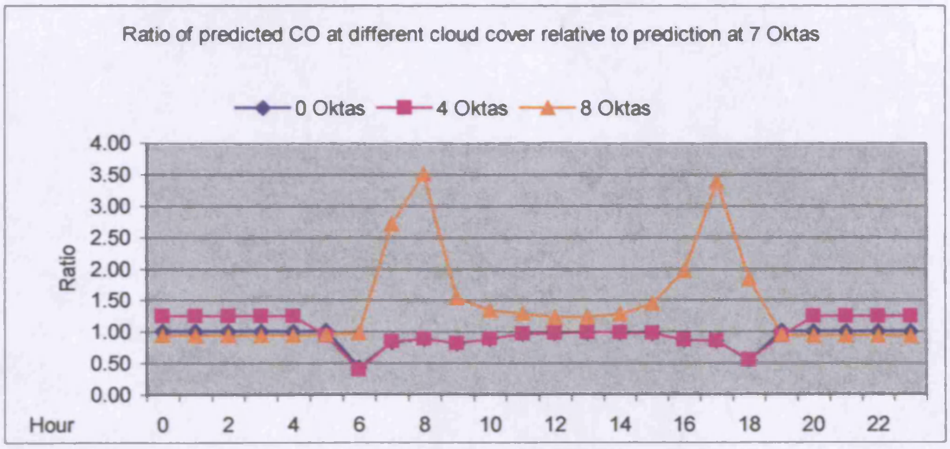

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Figure 5.8 Variation in predicted CO at different cloud covers - day 173.

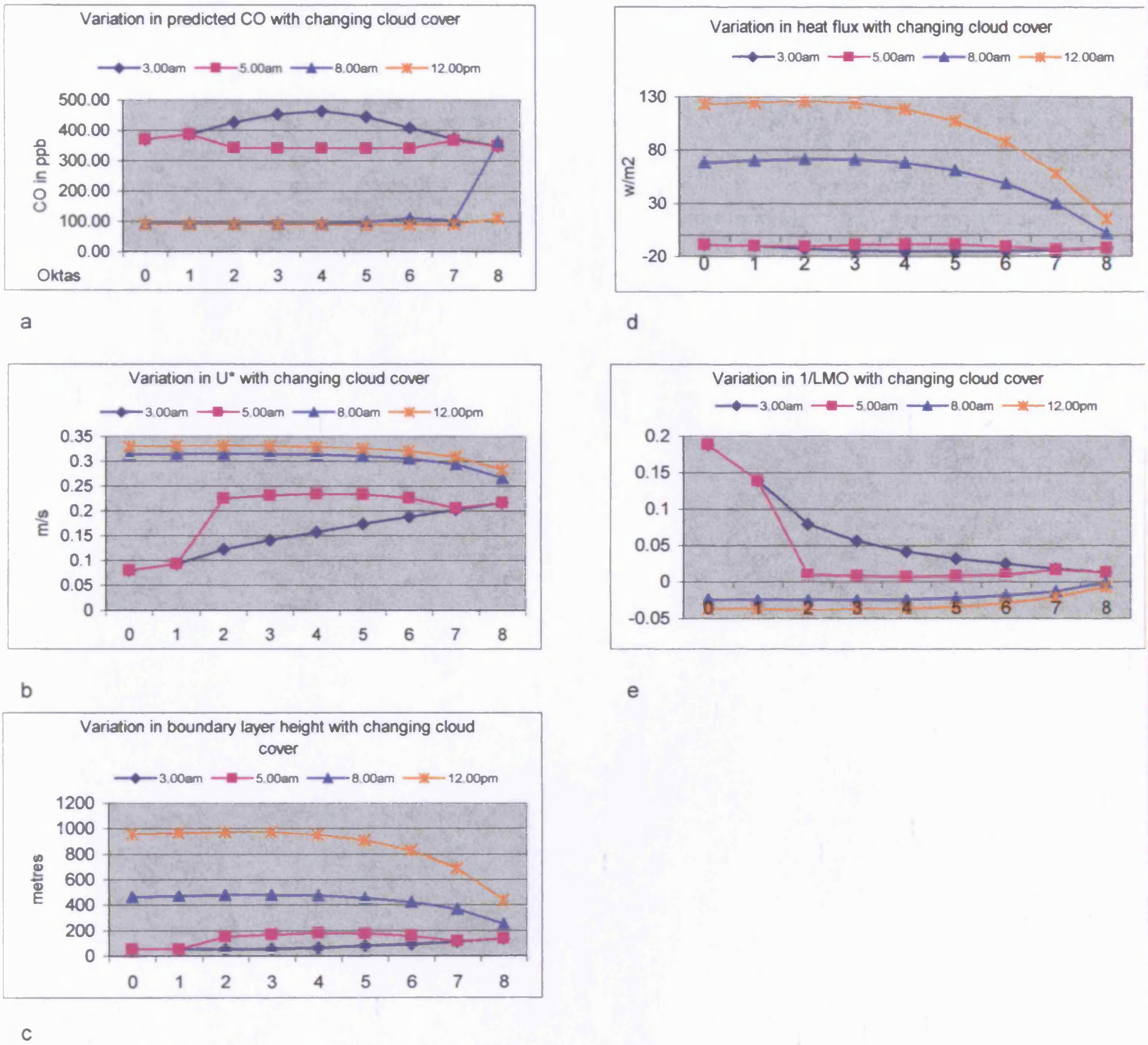
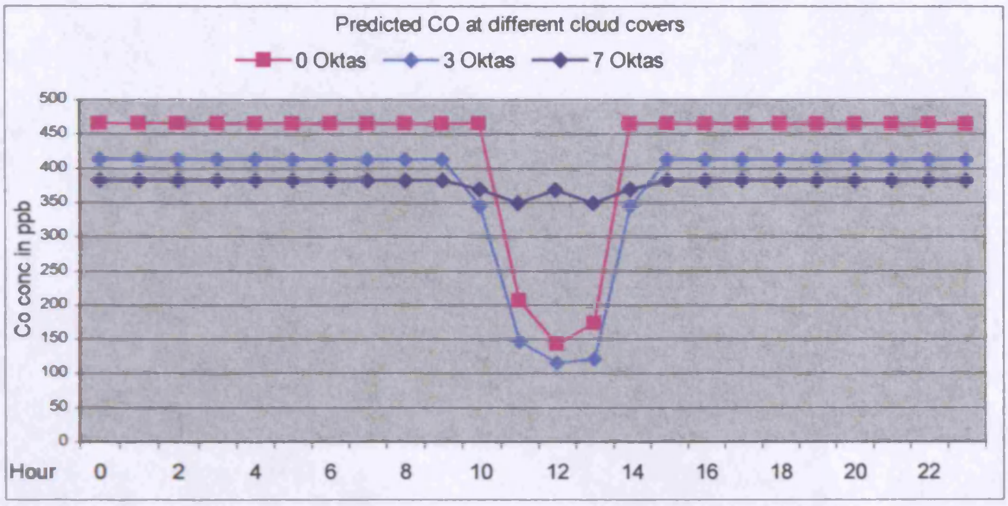


Figure 5.9 Model sensitivity to cloud cover - day 173.



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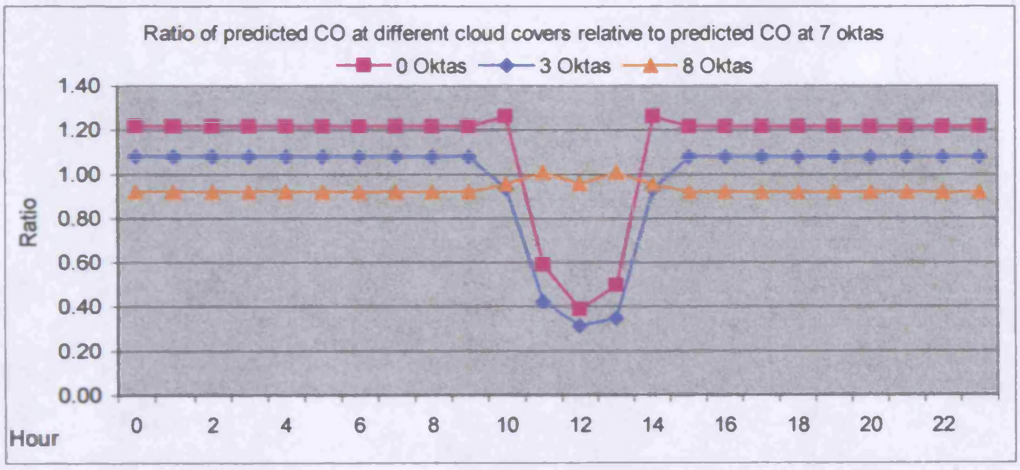

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Figure 5.10 Variation in predicted CO at different cloud covers - day 356.

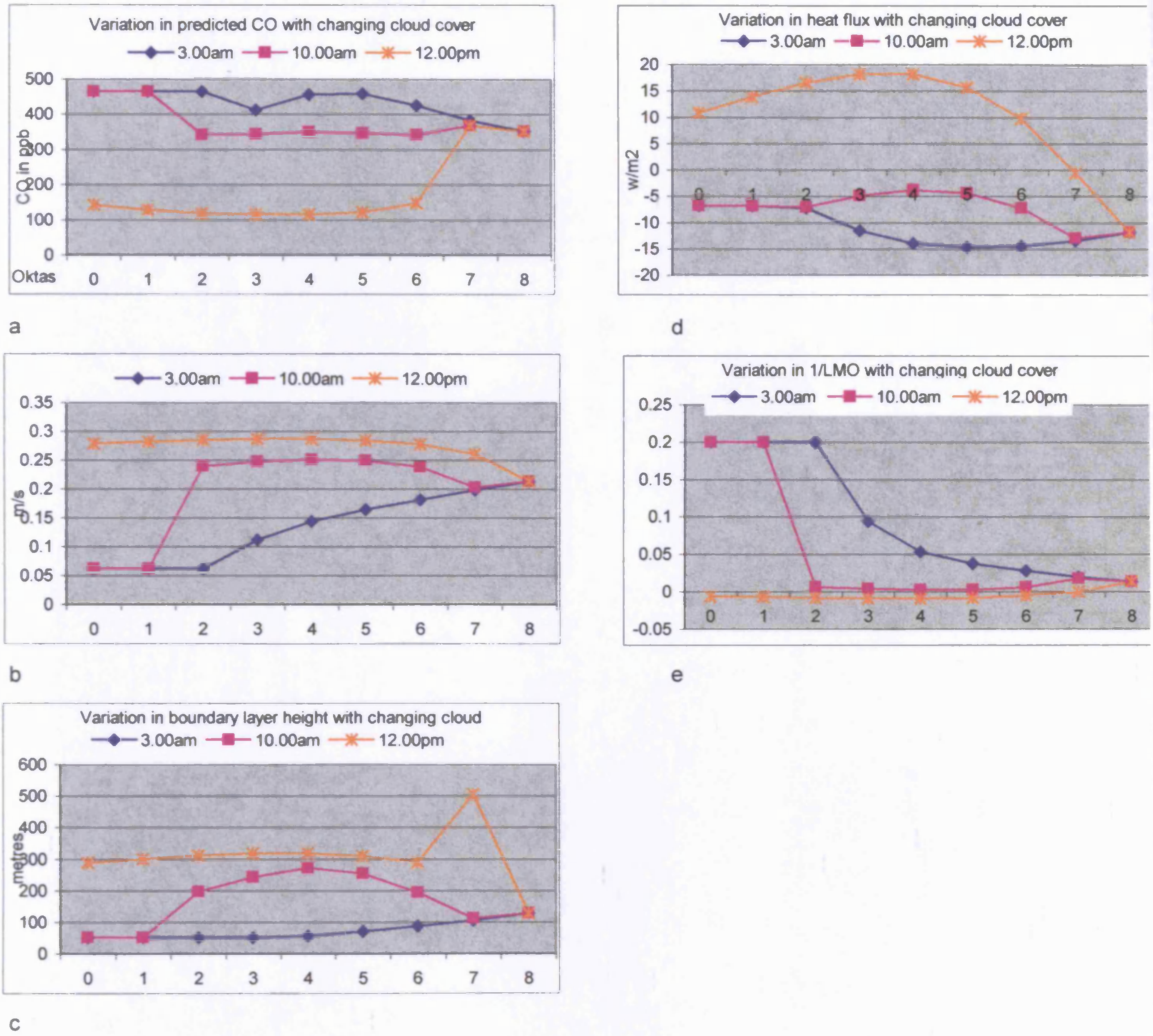


Figure 5.11 Model sensitivity to cloud cover - day 356.

5.3.4 Sensitivity to changing wind direction

Changing wind direction has no effect on the meteorological pre-processor, but affects the absolute concentration of CO reaching the receptor points and on the pattern of concentration with distance from road. The response also varies with time of day (Figure 5.12). In the case investigated, CO is highest when the wind is blowing from 260° i.e. at an angle of 10 degrees to the road. The closer the wind blows parallel to the road the greater the difference between the 5m and 10m receptor points. When the wind is blowing on to the road or at an obtuse angle the concentration is only higher at 5m from 0700Z until early evening. Once the wind is blowing off the road absolute values are obviously much lower, but concentrations are always much higher at 5m and there are three peaks concentration during the day at 0600Z, 1300Z and 1900Z (Figure 5.13). These results are important for two reasons. Firstly if modelling is used to provide information regarding the siting of monitors and secondly in model evaluation exercises as it shows how important it is to accurately match monitor and receptor points.

The greatest diurnal variation in CO is found when the wind is blowing more or less parallel to the road. Additionally greatest sensitivity to wind angle occurs during the night, but it depends on the angle to the road. An error of +/- 20 degrees will have little effect if the wind is parallel or at right angles to the road, but will have most effect when it is blowing at an angle between 20 to 50 degrees.

5.3.5 Sensitivity to temperature

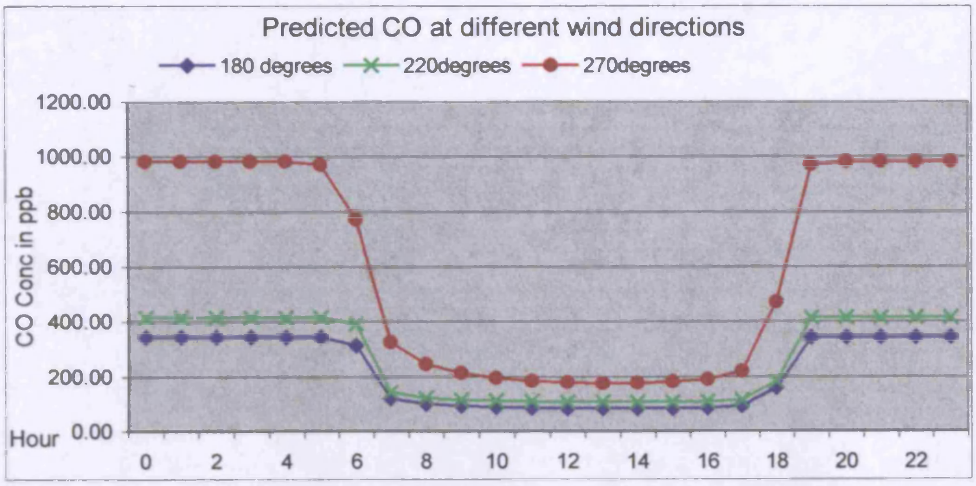
The temperature range chosen for Julian day 173 extended from 8°C to 26°C. In southeast England, nighttime temperatures do drop as low as 8°C on occasions during the summer months, but they are more usually in the range of 10°C to 15°C. Daytime temperature are unlikely to ever be as low as 8°C, but do rise occasionally to 24 or 26°C during the afternoon.

In terms of friction velocity, heat flux and reciprocal of Monin Obukhov length, the model responds identically to changing temperature at 0600Z and 1800Z. However the effect on boundary layer height is quite different, at 1800Z rising temperature has the effect of lowering the height with quite a dramatic change over 22°C. At 0600Z the boundary layer height is unaffected until 22°C is reached when there is a dramatic rise in height. The boundary layer appears to be most sensitive to the change between 22°C and 24°C, but these are not the temperatures at which model output is most sensitive (Figure 5.15a and 5.15c).

Although at midday, the model determines heat flux and reciprocal of Monin Obukhov length to be particularly sensitive to temperature, this has little resultant effect on model output. Heat flux and boundary layer height both decrease with rising temperature during the day but remain more or less constant during the night, at neither time is model output affected.

In summer increasing temperature has generally very little effect on predicted CO. However the model does appears to be sensitive to different temperature changes at different time of day. At 1800Z an increase of 2 degrees between 20°C and 22°C can cause a doubling of CO, where as in the morning (0600Z) the change between 16°C and 18°C will have nearly the same effect (Figure 5.15a). These responses do not seem to be explained by the output from the meteorological pre-processor.

The model is even less sensitive to temperature in winter (Figure 5.16). Although the internal meteorological parameters are affected by temperature at 1200Z, these responses are not translated into any variation in model output. As the temperature is increased the model determines that the atmosphere changes from unstable to stable, being neutral at 3°C and 5°C. It can be seen (Figure 5.16c) that this transition has a marked effect on boundary layer height, but this does not effect predicted CO concentration. Only when the temperature drops below zero is the predicted CO concentration halved.



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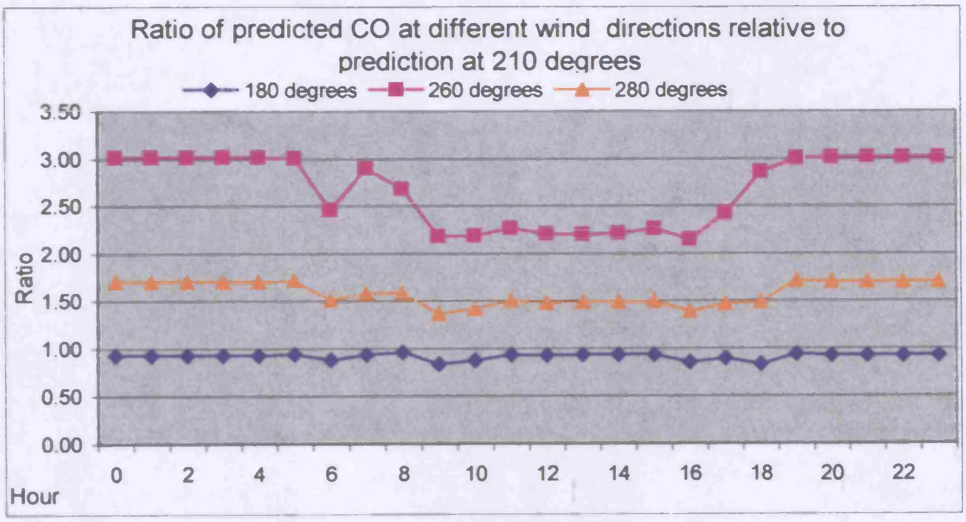

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Figure 5.12 Variation in predicted CO at different wind angles - day 173.

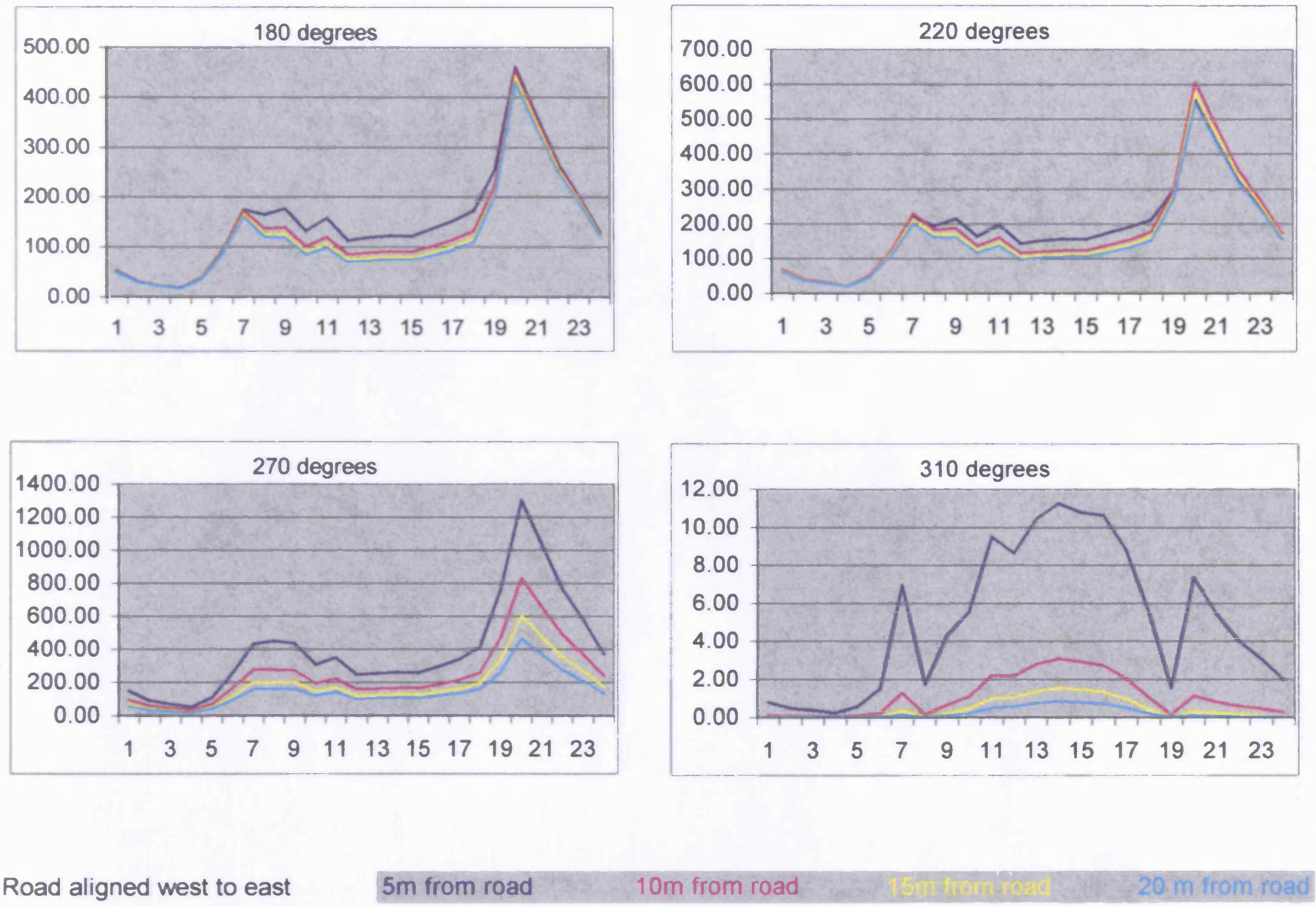
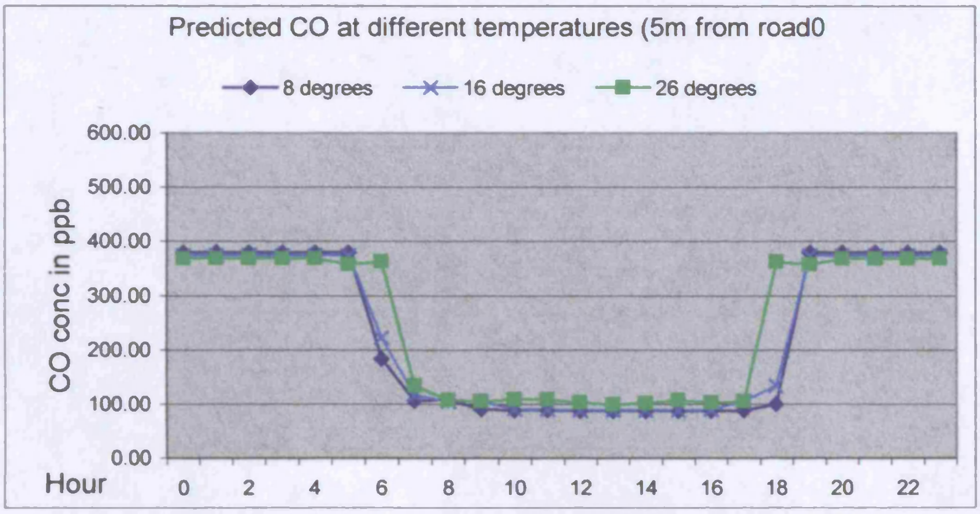


Figure 5.13 Variation in predicted CO with wind angle and distance from road -day173 (note varying CO concentration scale).



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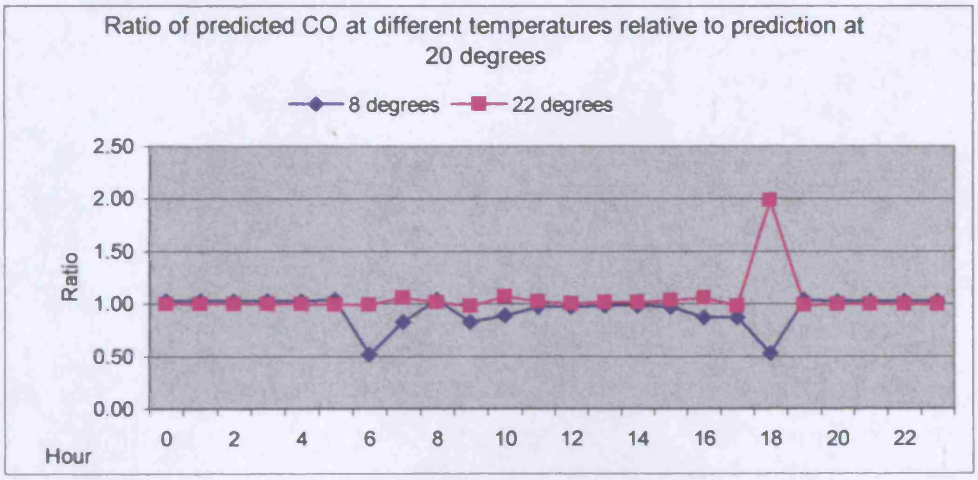

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Figure 5.14 Variation in predicted CO at different temperatures - day 173.

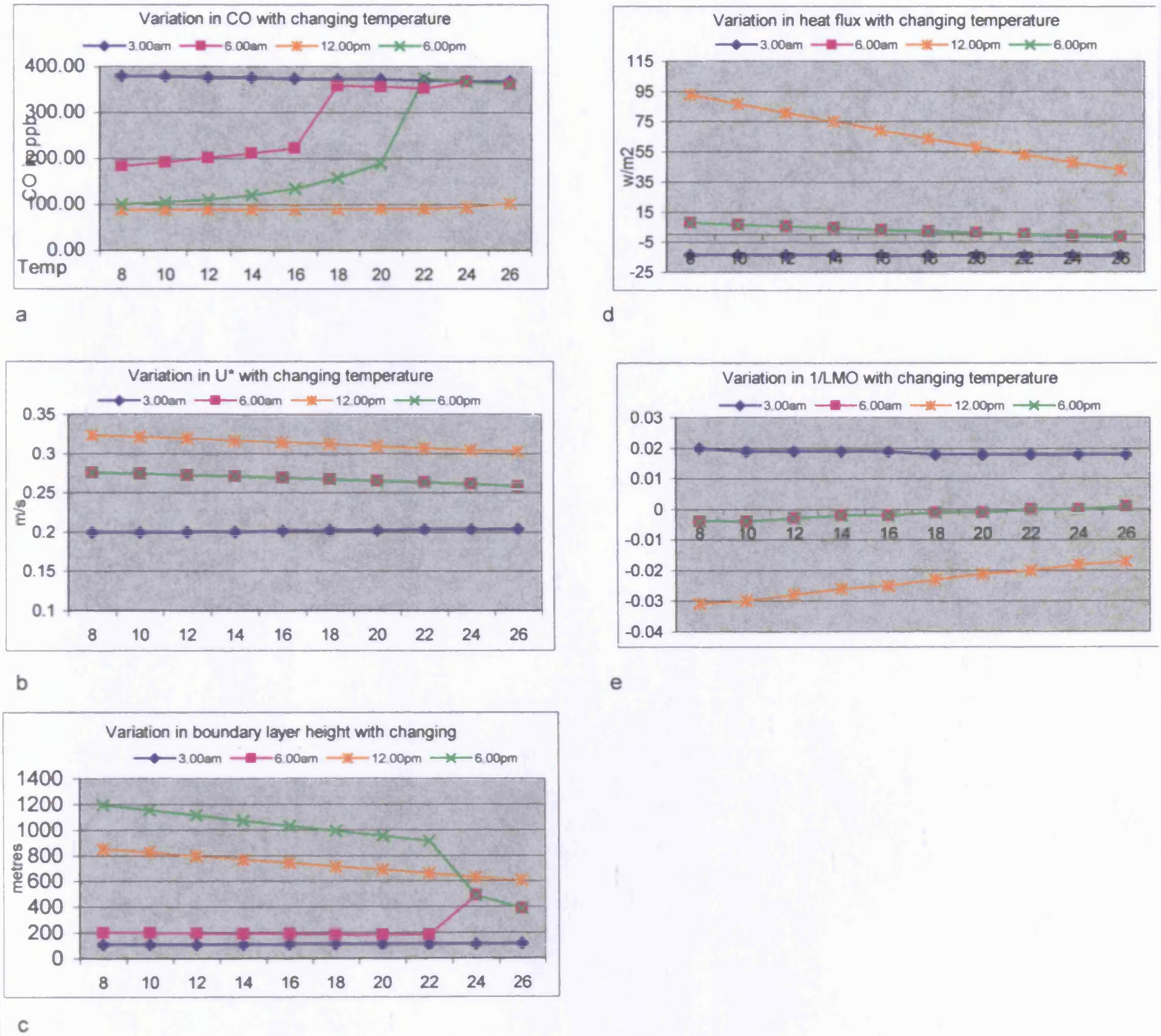


Figure 5.15 Model sensitivity to changing temperature - day 173.

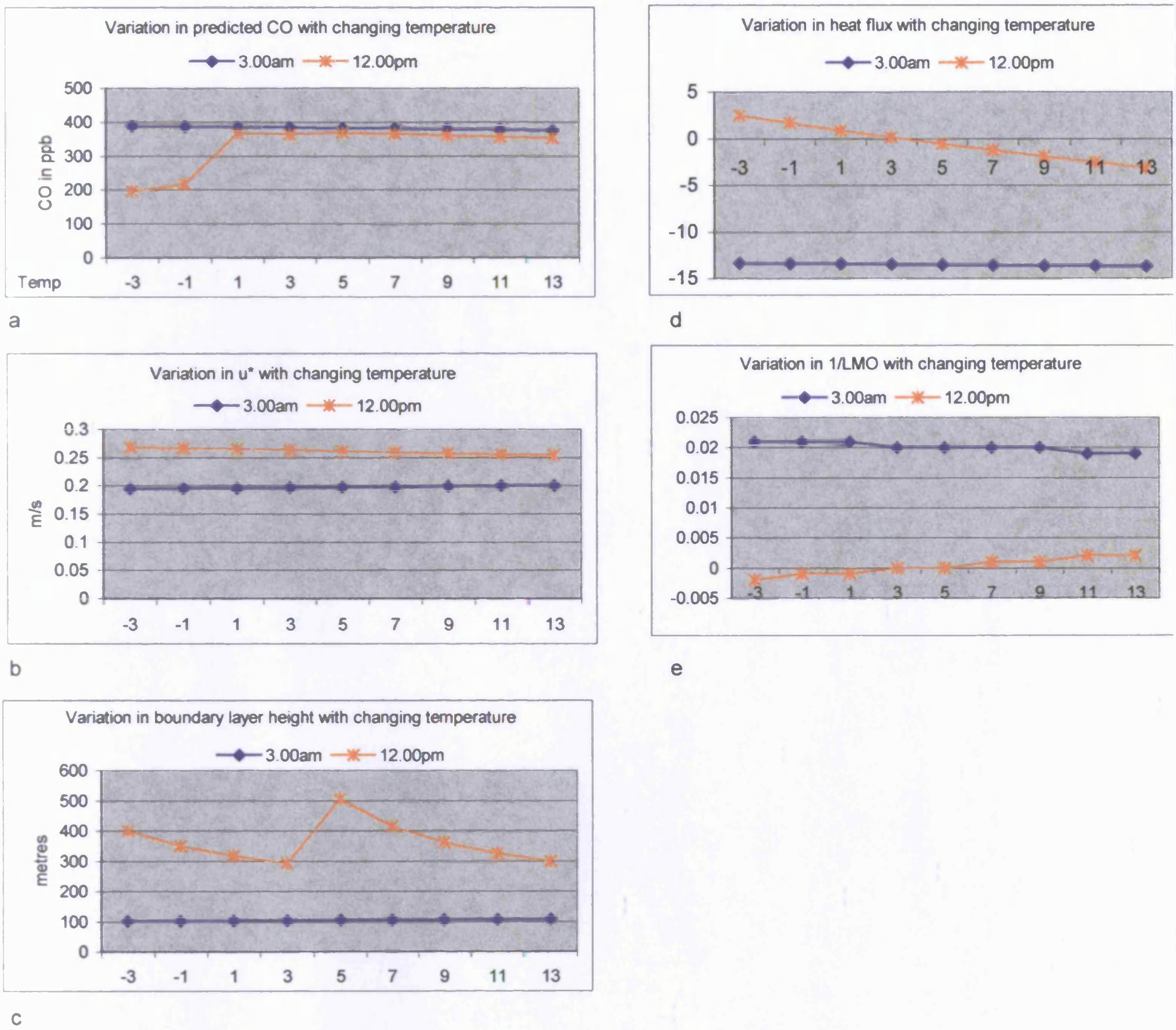


Figure 5.16 Model sensitivity to changing temperature - day 356.

5.3.6 Sensitivity to surface roughness

Surface roughness is difficult to estimate within an urban area and it is also likely to be highly variable. It is perhaps the only model input parameter that users of the model will have no actual value for and will rely on the values recommended by CERC. Surface roughnesses of up to 10m were included in the model runs although roughness of such a magnitude is unlikely to be found at any location in the UK. In summer, surface roughness makes little difference during the day when conditions in the boundary layer are driven by convective turbulence. As with some of the other parameters the model shows greatest sensitivity between 0600Z and 0800Z and at 1800Z. At night, surface roughnesses of 0.5 or 1m produce the same predicted CO, but a value of 0.3m results in higher predicted concentrations as mechanical mixing is only acting over a shallow layer. However even lower surface roughness results in lower predicted concentrations (Figure 5.17 and 5.18).

In winter, as there is little or no convective turbulence, the model is sensitive to surface roughness over the whole 24 hours. The nature of the response is different at midday when energy input into the atmosphere is greatest.

Under-estimating surface roughness is most likely to result in model error, but it not easy to say whether this will result in over- or under-prediction. The error caused by wrongly estimating surface roughness will be small compared with error resulting from other input variables.

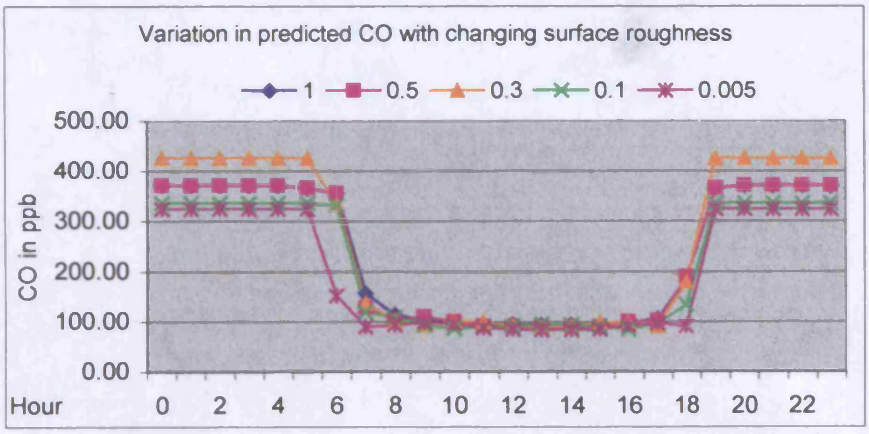
5.3.7 Sensitivity to albedo

The use of albedo by ADMS is discussed in chapter 8. Albedo is an optional input parameter and if it is not entered, ADMS uses a default of 0.23.

The model is sensitive to albedo across all daylight hours (Figures 5.19 and 5.20). The change in model output is perceptible over the whole range of albedo values for a period in the morning and a period in the late afternoon, but during the middle of the day the model is only sensitive in the 0.6 to 0.8 range. This is outside the range of values in which urban environments are most likely to lie. The model predicts a gradual decrease in boundary layer height, friction velocity and heat flux as albedo increases, but it is predominately the stability of the atmosphere ($1/L_{MO}$) that seems to determine model output. At both 0700Z and 1200Z, it is when the model determines increasing albedo to cause a change from unstable to stable atmospheres that correlates precisely with an increase in predicted CO.

The model behaves in a similar fashion in winter, but the response to changing the albedo value is confined to the middle of the day between 1100Z and 1300Z. During these hours the model behaves as it does at 0700Z in the summer. (Figure 5.21)

Within a large urban area there will be much variation in surface material and hence albedo. Even an average value, wrongly estimated could have a significant effect on model output. A difference of 0.2, which is easily possible within the range of values quoted for most surface materials, can result in a factor of two change in model output.



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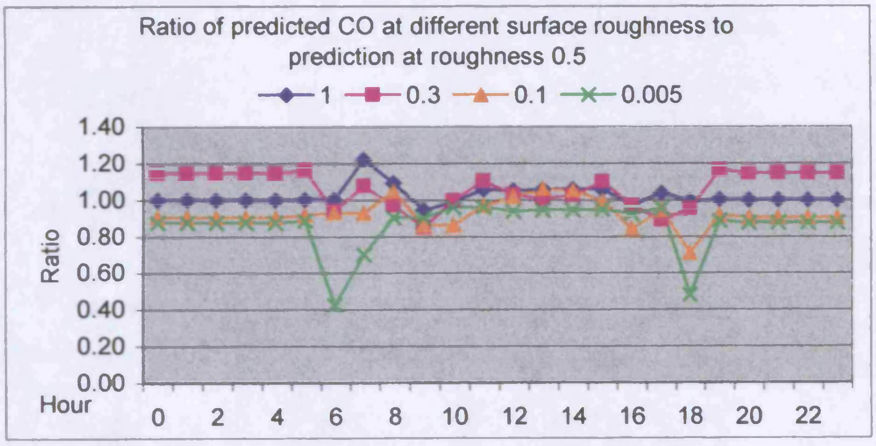

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Figure 5.17 Variation in predicted CO at different surface roughness - day173.

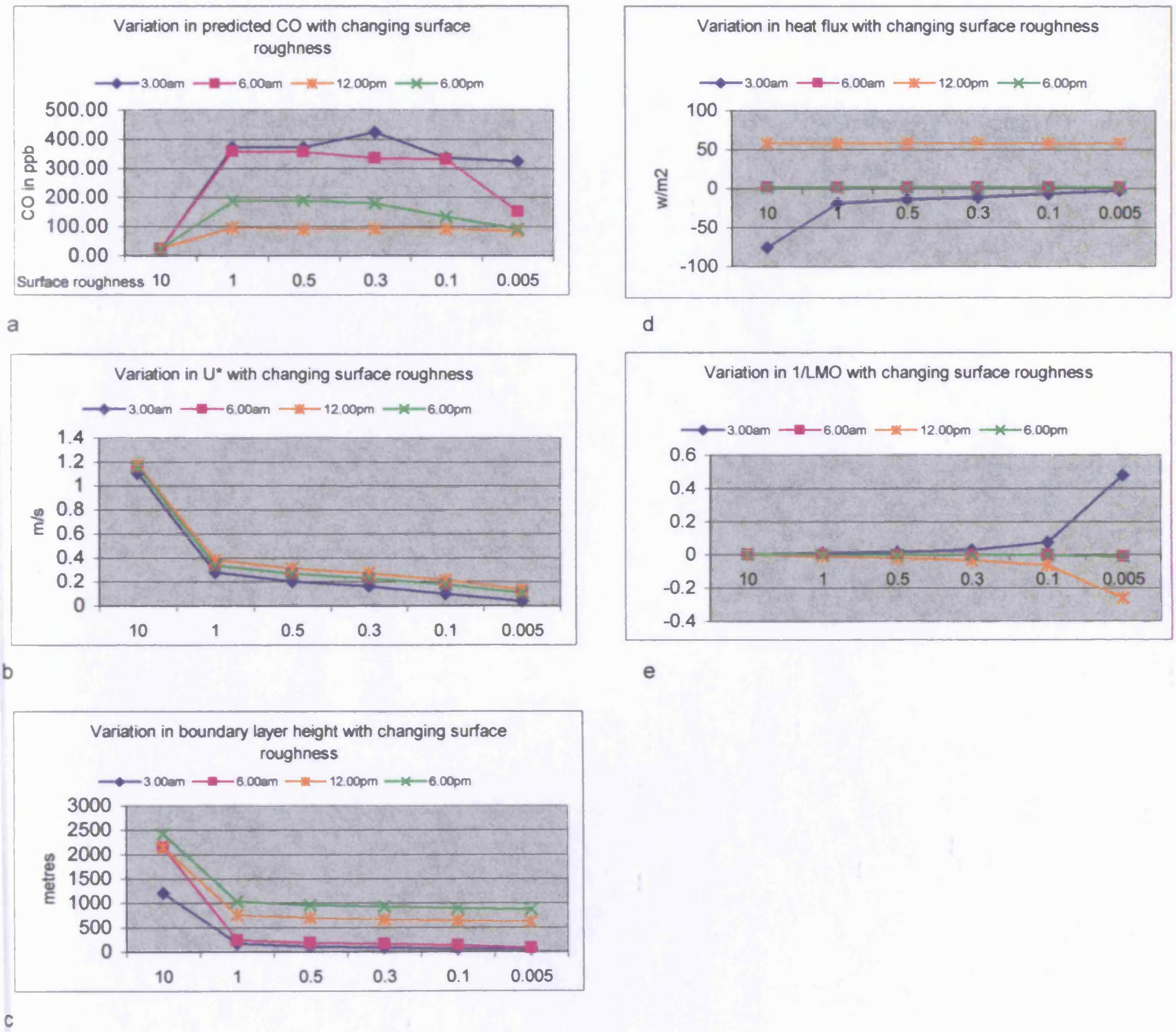
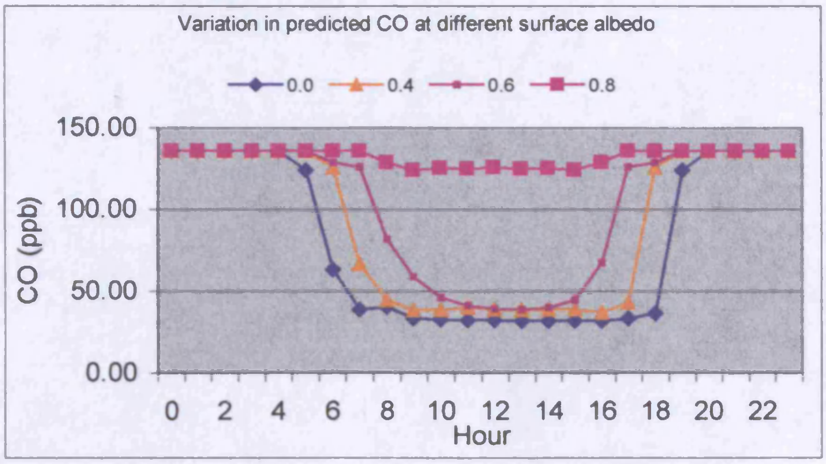


Figure 5.18 Model sensitivity to surface roughness - day173.



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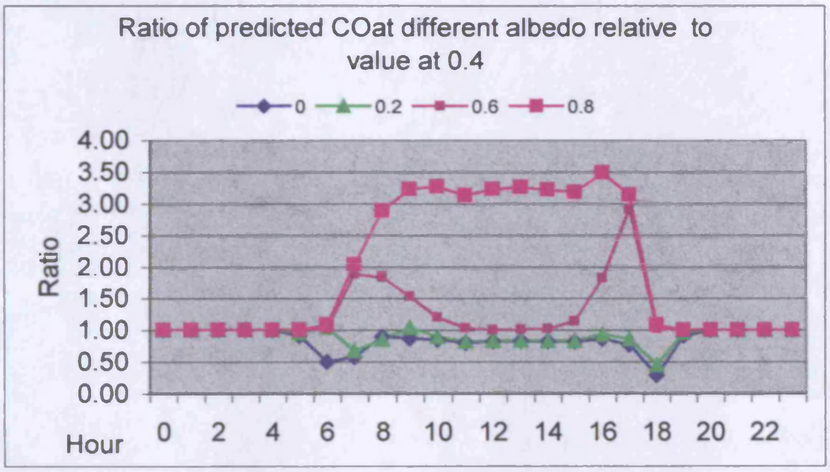

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Figure 5.19 Variation in predicted CO at different albedo settings- day 173.

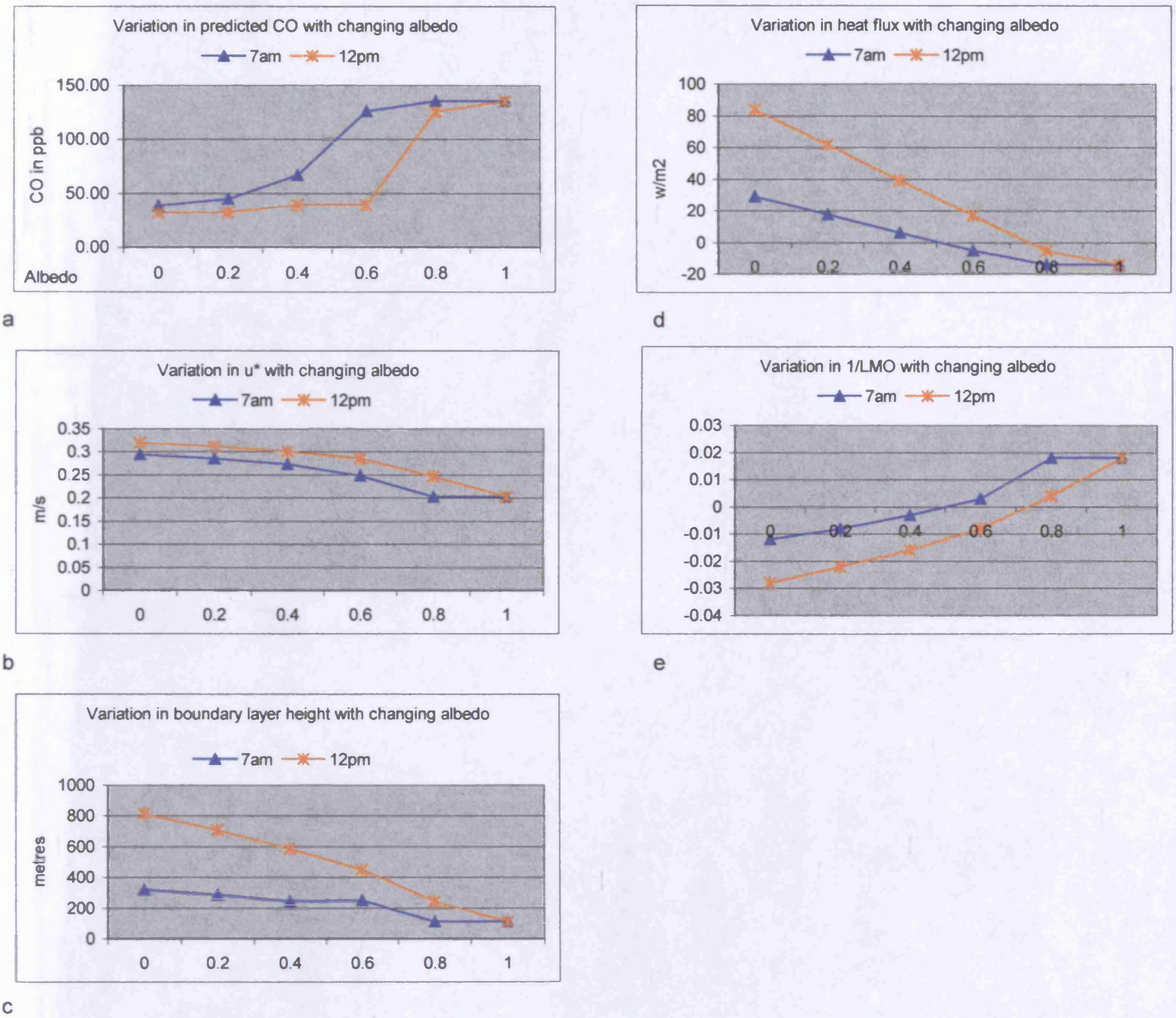


Figure 5.20 Model sensitivity to albedo - day 173.

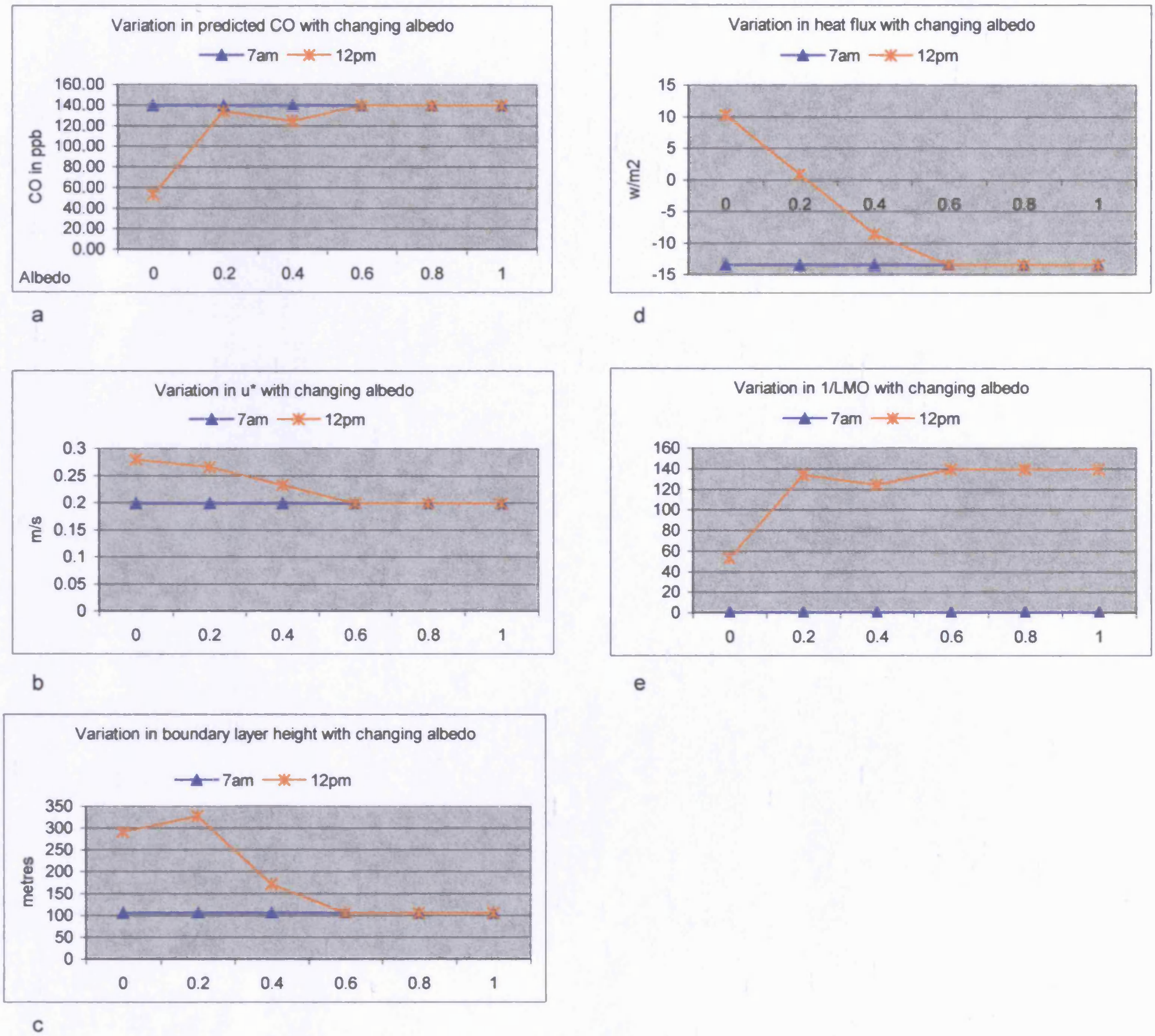


Figure 5.21 Model sensitivity to albedo - day 356.

5.3.8 Time varying emission factors

Once time varying emissions are introduced into the model runs a clearer impression is given of how crucial an understanding of model sensitivity is. The time varying emission factors used are shown graphically in figure 5.1. They show clearly morning and afternoon traffic peak flows. Air quality monitoring, regardless of meteorological conditions, invariably picks up high levels of traffic related pollutants during both these rush hours. At certain times of day, particularly at rush hour and when changes in stability are occurring, it is quite possible for the model to fail to predict high pollutant levels. These features of the model could have serious implications for regulatory compliance.

In summer when the model is run with the baseline conditions i.e. 7 oktas cloud cover, no morning pollution peak is predicted and the afternoon peak is only predicted at low wind speeds (Figure 5.22). It is only with 8 oktas cloud cover that a morning peak is predicted of the same magnitude as that occurring in the afternoon. This extra one oktas cloud cover also has the effect of bringing the afternoon peak two hours forward. As 7 and 8 oktas are the most frequently occurring cloud covers in the UK, a slight change in their relative frequency has the potential to produce large variation in predicted pollutants at the times of day when levels are already high (Figure 5.23).

It is only once the model has been run with an unrealistically high albedo that CO predictions coincide with the time varying emissions at lower cloud covers.

In winter, when time varying emissions are modelled, ADMS does predict a pollutant peak in the morning at low wind speeds. The model output now shows a very similar response to what would be expected. However at low cloud covers there is a dip in the predicted CO during the middle of the day, so again there appears to be a danger of the model underestimating pollution (Figure 5.24).

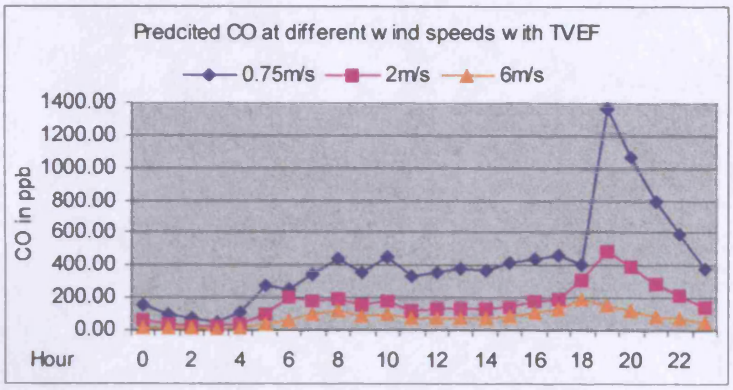


Figure 5.22 Variation in predicted CO at different wind speeds (7 oktas cloud cover) with time varying emission factors - day173.

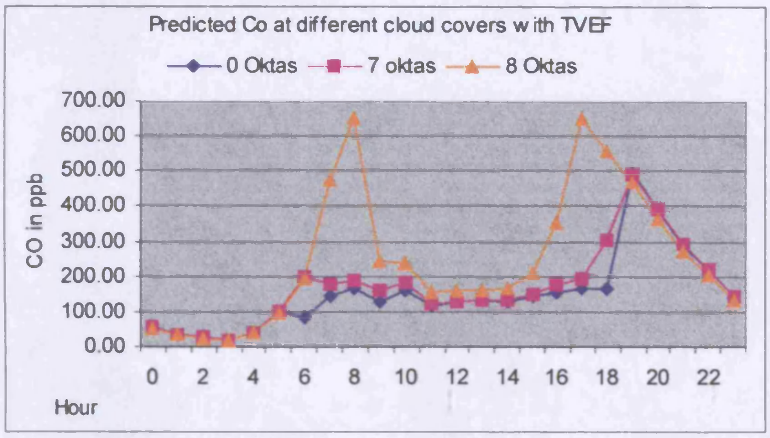


Figure 5.23 Variation in predicted CO at different cloud covers with time varying emission factors - day173.

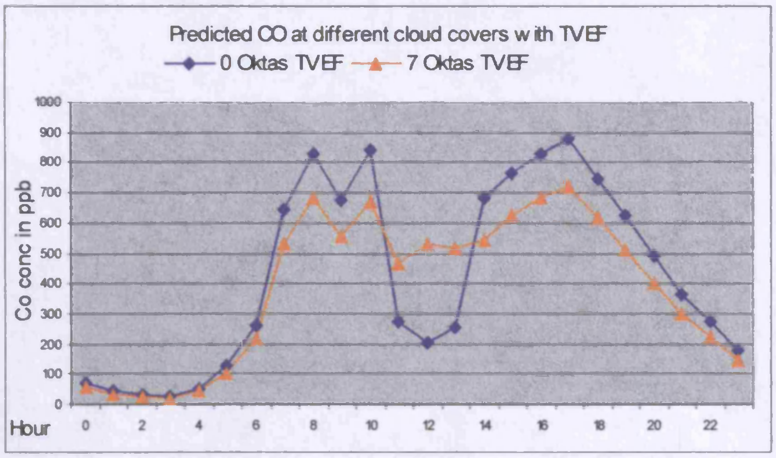


Figure 5.24 Variation in predicted CO at different cloud covers with time varying emission factors - day356.

5.4 Multi-factorial sensitivity analysis

Using the same scenario, model runs were carried out for a single hour only. Four meteorological parameters were changed in turn. Three temperature settings were used, 10°, 15° and 20°C, four different wind speeds from $1ms^{-1}$ to $7ms^{-1}$ and four different wind directions, 190°, 210°, 230° and 280°. Cloud cover was varied from 0 to 8 oktas. This resulted in 432 combinations.

Modelling was carried out for hours beginning 0900Z and 1200Z for a representative summer day and at 1200Z for a winter day. In winter 1200Z is the hour that shows maximum sensitivity when parameters are changed individually. In summer maximum sensitivity is shown between 0600Z and 0800Z and again during late afternoon. Little variation would be expected at 1200Z. The model does still show some sensitivity, particularly to cloud cover, at 0900Z. However using a slightly earlier hour might have given a greater spread of results.

Taking cloud cover 7, wind speed $3ms^{-1}$, wind direction 210° and temperature 15° as a baseline value, the percentage change in model output produced by deviating from these conditions was calculated. From this baseline value one would generally expect an increase in model output for wind speeds of $1ms^{-1}$, cloud cover of eight oktas, wind directions of 230 and 280° and temperature of 20°C. A decrease would be expected with wind speeds of $5ms^{-1}$ and $7ms^{-1}$, cloud covers less than7 oktas, wind direction of 190° and temperatures of 10°C. When all four factors are in the right combination to bring out a maximum increase or a decrease quite large percentage changes in output can occur. Depending of which of these parameters has the greatest effect of model output there may be situations where anomalies occur. These results should not be considered as entirely representative of a normal modelling situation as the model is determining boundary layer conditions from only a single hour of data.

Results for the summer model runs are given in Table 5.2. This only shows what happens at 15°C. In summer, temperature generally has little effect. The higher temperature only serves to exaggerate increases when these are already brought about by changes in other parameters, similarly the lower temperature has the same effect on marginally decreasing output in the extreme cases. The results are broadly as one might expect. The point worth noting however is that at certain wind directions the effect of increased cloud cover can overide the effect of higher wind speed and bring about an increase in predicted CO level.

The same pattern of results is shown at 0900Z and 1200Z. As expected more sensitivity is demonstrated at the earlier time, especially in situations that lead to a reduction in model output.

In winter, the pattern of results is slightly different. This is largely due to sensitivity to cloud cover, which produces a different response in winter than in summer (Figure 5.9a and 5.11a). If 6 oktas had been used as the baseline value results would have been more similar to summer but it was left at 7 oktas. Analysis of synoptic cloud cover data has shown this to be the most frequently value at the majority of sites in the UK.

Overall the model is far more sensitive to cloud cover in winter. When all other parameters are held constant, the lowest output occurs at 3 or 4 oktas. There is a much greater separation between the results at these cloud covers and the maximum values that occur at 7 or 8 oktas. Results for model runs carried at 8°C only are given in Table 5.3. In winter temperature has no effect when there is 7 or 8 oktas cloud cover. It is only at lower cloud covers that increasing temperature has a significant effect on increasing model output.

Wind speed	Cloud cover	Wind direction 190deg	210deg	230deg	280deg
$1ms^{-1}$	0	120.9	166.0	222.2	155.6
	1	121.1	166.3	221.1	154.4
	2	121.3	166.6	220.3	153.6
	3	121.2	166.5	220.6	153.8
	4	120.8	165.9	222.5	155.9
	5	121.0	164.4	227.0	161.0
	6	120.8	161.8	235.5	171.2
	7	122.7	158.2	250.3	193.1
	8	157.9	191.6	310.7	287.6
$3ms^{-1}$	0	-20.9	-9.3	23.8	18.0
	1	-21.1	-9.4	23.5	17.6
	2	-21.2	-11.5	23.3	17.4
	3	-21.2	-11.5	23.4	17.5
	4	-20.8	-9.3	23.8	18.2
	5	-20.0	-8.1	25.0	19.9
	6	-17.8	-6.0	28.1	24.1
	7	-11.6	0.0	36.9	34.9
	8	19.4	34.6	85.4	90.3
$5ms^{-1}$	0	-44.9	-37.7	-15.6	-15.7
	1	-45.1	-38.0	-16.0	-16.1
	2	-45.4	-38.5	-16.5	-16.6
	3	-45.8	-38.9	-17.1	-17.3
	4	-45.7	-38.8	-16.9	-17.1
	5	-44.6	-37.5	-15.3	-15.3
	6	-41.8	-34.5	-10.9	-10.3
	7	-34.7	-26.6	0.3	2.0
	8	-16.4	-4.8	29.8	35.3
$7ms^{-1}$	0	-56.0	-50.7	-33.0	-32.1
	1	-56.6	-51.3	-33.9	-33.1
	2	-57.1	-51.9	-34.7	-34.0
	3	-57.4	-52.2	-35.2	-34.5
	4	-57.2	-52.0	-34.9	-34.2
	5	-56.2	-50.9	-33.3	-32.4
	6	-53.8	-48.2	-29.4	-28.1
	7	-48.1	-40.9	-20.3	-18.1
	8	-34.9	-26.0	1.3	6.2

Table 5.2. Percentage changes in model output with varying input parameters – summer 0900Z.

Wind speed	Cloud cover	Wind direction 190deg	210deg	230deg	280deg
1ms^{-1}	0	-11	1	42	33
	1	-14	-3	37	27
	2	-16	-5	34	23
	3	-17	-6	33	21
	4	-17	-6	33	21
	5	-15	-4	35	24
	6	-9	2	44	37
	7	159	205	363	339
	8	305	364	556	701
3ms^{-1}	0	-55	-49	-31	-28
	1	-58	-53	-35	-34
	2	-61	-56	-40	-39
	3	-63	-58	-43	-42
	4	-64	-59	-44	-43
	5	-62	-57	-41	-40
	6	-56	-50	-31	-29
	7	-15	0	51	39
	8	-13	2	53	45
5ms^{-1}	0	-70	-66	-53	-51
	1	-71	-67	-55	-54
	2	-73	-69	-58	-56
	3	-74	-70	-59	-58
	4	-74	-70	-60	-59
	5	-73	-69	-58	-57
	6	-70	-66	-53	-51
	7	-49	-40	-10	-17
	8	-49	-40	-10	-16
7ms^{-1}	0	-64	-57	-36	-41
	1	-78	-75	-65	-64
	2	-79	-76	-67	-65
	3	-79	-76	-68	-66
	4	-79	-76	-68	-66
	5	-79	-76	-67	-65
	6	-64	-57	-36	-41
	7	-64	-57	-36	-41
	8	-64	-57	-36	-40

Table 5.3 Percentage change in model output with varying input parameters – winter 1200Z.

5.5 Conclusions

The results presented in the chapter show several features of the ADMS-Urban model. They highlight when the model is most sensitive and to which parameters. They also indicate the magnitude of the error that is likely to be found when using inaccurate input data and show the importance of carrying out multi-factorial analysis. Input parameters can cause unexpected results under certain conditions.

The sensitivity analysis shows that large changes can occur in model output with only small changes in certain parameters and that this is dependent on time of day and time of year. Generally these changes can be explained by how the model parameterises the boundary layer, but this is not always the case. Even without fully understanding the behaviour of the model, model users should be aware of conditions under which the model demonstrates particular sensitivities.

There is an enhanced risk of error in summer at the times of day when emissions are greatest i.e. morning and late afternoon. Also under certain conditions it is quite likely that the model will fail to predict any morning pollutant peak. Practically any examination of monitored data from an urban area will show peaks in traffic derived pollutants that coincide with the morning rush hour. There are very few days on which meteorological conditions result in rapid dispersion of pollutants. In winter the model is most sensitive at midday. Although emissions not will be as high as at rush hours, the overall enhanced sensitivity at this time of year, when stable conditions are more likely to leads to a build up of pollutants means that modelling errors are to be expected.

At all times of year cloud cover can have quite a powerful effect. This parameter is unlikely to be monitored locally, its assessment is subject to human error and the most frequent value is in the range of greatest model sensitivity. As such it seems most likely to be the parameter that could cause real problems for model uncertainty.

Cloud cover is also the parameter that is most likely to show systematic error. Land based observations are most likely to result in an overestimate of cloud. Error for the other parameters is likely to be averaged out when predicting long- term averages or if statistical data is used. When modelling large urban areas errors in wind direction, in particular, will be balanced out simply because the road network will contain roads going is many different directions. If modelling is carried out on a single section of road depending on the alignment of that road in relation to the prevailing wind direction, even small changes in wind direction could have a significant effect.

Although this averaging effect is true for constantly changing meteorological parameters the same can not be said for surface roughness or albedo. Even though both may be variable across an urban area, the latter can to some extent change over the course of a year, particularly in areas likely to experience long periods of snow cover in winter. Variation in surface roughness within the range that could be found in an urban area, say 0.3 to 0.5, is unlikely to result in more than 15% variation in model output. However, albedo, if it is only 0.2 more than the default, can result in a 50% increase in output at certain times of day.

Modelling is sometimes used to provide information with regard to the siting of monitoring equipment. Again care must be taken when interpreting model output. The 5m from road receptor point only has the highest CO concentrations between 0700Z and 1800Z, during the night it is actually the lowest and highest values are found 10 m from the road. This same pattern is seen regardless of wind speed.

These finding do highlight some of the problems faced by local authorities when using models to carry out air quality management assessments. If hourly sequential data are used, the model is clearly sensitive to small changes in certain meteorological parameters just at times at which pollution levels are likely to be highest.

CHAPTER SIX

ADMS PERFORMANCE IN RELATION TO METEOROLOGICAL DATA

STUDY ONE - NORTHAMPTON

6.1 Introduction

As dispersion models become more complex, provision of the necessary input data becomes increasingly difficult. Simple Gaussian models often require nothing more than a simple parameterisation of the wind field in terms of speed and direction, whereas complex photochemical models often require hourly vertically and horizontally resolved wind fields and other data such as temperature, humidity, and global radiation. The 'new generation' models also require more complex inputs to enable the calculation of stability parameters such as the Monin Obukhov length. The inputs required by older style models that use Pasquill-Gifford stability categories are less demanding.

The uncertainty in atmospheric dispersion modelling resulting from the stochastic nature of the meteorological input data has already been mentioned in Chapter 2. There are three further problems associated with meteorological data that will be considered here. Firstly meteorological data are often recorded at sites outside the modelling domain. Secondly, in the case of urban dispersion modelling, the physical nature of the meteorological recording site is often very different from the location to which the data will be applied and thirdly, only a few of the parameters required for modelling are routinely recorded so some form of pre-processing is required.

The model, whose performance is considered in this chapter, does to some extent allow for physical differences between the meteorological recording site and the modelling site. It is possible to specify whether the meteorological data are representative of the source site by giving a precipitation factor that relates the rainfall between the two sites and by stating the surface roughness of the meteorological site. The latter is particularly important for urban modelling where the combination of a high surface roughness value and the heat island effect can prevent the boundary layer from ever becoming truly stable. To allow for this ADMS-Urban also gives the option of setting a minimum Monin Obukhov length (see Appendix Three).

The problems of modelling in areas that lack raw meteorological data and the errors that this is likely to cause have been discussed by many workers, notably Milford and Russell in 1993. They considered two solutions, interpolation and the use of meteorological models. Like dispersion models, meteorological models can either be diagnostic or deterministic in approach. The use of deterministic models that describe the physics and thermodynamics of the atmosphere with fundamental equations is intended to improve the accuracy of the inputs over the modelling domain. However, in both cases, if they rely on sparse input data there is an intrinsic uncertainty that will in turn lead to uncertainty in the dispersion model output. As meteorological models are also used for weather forecasting there is a great deal of experience with their use. However Dennis (1996) cautions that they have not been developed to support air quality modelling, they have different parameterisation schemes for the various atmospheric processes and are based on different assumptions and theories. Further consideration of one particular meteorological model is given in Chapter Nine.

The paucity of meteorological data in general and the problem that some parameters required by dispersion models are not routinely recorded has recently been under investigation as part of the European Union COST 710 Action programme. The purpose of this was not to address the accuracy of dispersion models, but to ensure that they use the most appropriate data in their calculations. A second concern was to ensure that the meteorological pre-processing required to calculate parameters not routinely measured, but used to characterise dispersion conditions, is adequate. The errors and differences between methods used in the pre-processing can be as great as the errors occurring elsewhere in the modelling itself (Cosemans *et al.*, 1997).

In the UK, neither the Government nor the Environment Agency specifies the use of any particular dispersion model, although there are models available that have been development by government agencies such as R91 (National Radiological Protection Board), NAME, AEOLIUS (UK Meteorological Office) and DMRB (Transport and Road Research Laboratory). However the Royal Meteorological Society (1995) has produced a policy statement giving guidelines on the choice and use of models. In this they suggest that a more pro-active approach should be taken to obtaining meteorological data for modelling purposes. It should not be simply accepted that site specific data are not available. They consider that this particularly applies where local topography is likely to have a significant and characteristic effect on dispersion. Even if local data are only available for a short period this can be related to measurements obtained from the most representative nearby site for which longer periods of data are available. Where local data are not available they advise that careful consideration should be given to how representative nearby sites are. This cannot be assumed on the basis of geographical proximity. They do not however specifically consider the problem of modelling in urban areas.

The Danish government is more proscriptive with regard to dispersion modelling and recommends the use of one particular point source model - OML, for regulatory work. The Danish Environmental Protection Agency has formulated a set of guidelines for model users. They recommend that for administrative and regulatory purposes data from Karstrup airport should be used for the whole country. Although it is acknowledged that this is at the expense of accuracy, they felt it was administratively more simple. Modellers are however are free to provide their own data where available (Olesen, 1995a).

When modelling is used to support planning application in the case of environmental impact assessments or air quality legislation as part of an air quality impact assessment, there may be something to be said for standardisation in the approach to gathering meteorological data. If it is obtained from recognised agencies such as the UK Meteorological Office this implies a degree of quality control. This argument is also put forward in the United States where the EPA considers it unreasonable to expect each regulated point source to have on-site meteorological measurements, consequently no such requirements are placed on *any* source. The use of only routinely recorded data avoids disputes concerning the representativeness of the data, the response of the instrumentation, etc. (Turner, 1997).

The problem remains that routinely recorded meteorological data are not necessarily representative of the modelling site and monitoring sites are not evenly spread across the country. Statistical data used to run ADMS-Urban can be obtained from the UK Meteorological Office. The data are provided as the frequency with which certain conditions exist. Several years' data are required to obtain a statistically satisfactory dataset and there are only 60 meteorological stations across the UK with sufficient data to provide this (Carruthers *et al.*, 1994). A larger number of sites can supply sequential meteorological data of suitable quality. However they are not evenly spread across the country or representative of all topographies.

In an ADMS-Urban evaluation study (Carruthers *et al.*, 2000) some of the problems of using and obtaining representative meteorological data were highlighted, although there was no elaboration of actual effects on model output. Data were available from two synoptic sites within 50 km of the modelling site. Although close to each other the wind roses for the two sites were quite different and a combination of data were used for the modelling. They are both coastal sites are as such are likely to be subject to local topographic effects.

Meteorological data are also available from different agencies and even here differences can occur. Data for the UK are available from the UK Meteorological. Office and from the US based Trinity Consultants. The latter provide by far the most comprehensive set of data, at a lower cost and with fewer restrictions placed on use. Hall *et al.*, (1999) compared hourly data from both sources for four sites from 1994. The best agreement was for cloud cover with 83% of hours having identical values, but the worst case was for temperature (resolution +/- 0.5°C) with, at one site, only 18% of hours identical. The majority of the data for all parameters (temperature, cloud cover, wind speed and direction) does agree to within twice the resolution of the parameter. Some of the variation is undoubtedly due to how the data are averaged over the course of an hour. There are differences between the annual averages for wind speed and for the distribution of wind direction, but these differences are smaller than those that occur from year to year with data from the same source. Both these sets of data were used in a hypothetical modelling situation using two different versions of the point source model; ADMS 2.2 and ADMS 3. Annual mean and various percentile concentrations were calculated. The results showed differences in concentration of the same order and in some cases greater between the two models than between the meteorological datasets. Hall *et al.* (1999) concluded that these differences were due to the inherent uncertainty in meteorological data and the year to year variation meant that the differences due to the source of the meteorological data were not significant.

The differences in output obtained from the two versions of ADMS discussed above are due to how the models used the meteorological data to calculate boundary layer conditions rather than how they calculated the actual dispersion parameters (Hall *et al.*, 1999). A similar situation was found by Stübi *et al.* (1997), where as part of the COST 710 programme, different parameterisation schemes for calculating atmospheric stability and boundary layer height were compared (See chapter 8 for discussion of sensible heat flux calculations). They found that estimation of z_0 is sensitive to the height of the two anemometers from which the wind speed profile is calculated. Determination of the Monin Obukhov length is also sensitive to the height at which the measurements are taken and to the surface roughness. This work not only highlights the problem of using different parameterisation schemes, but also the need to ensure standardisation of data recording. This supports the case made by Turner (1997).

As part of a larger study in Finland, Karpinnen *et al.* (1999) investigated how data recorded at rural sites relates to conditions in an urban area and the effects of using different meteorological pre-processing schemes. Potential temperature gradient data were taken from meteorological masts in Helsinki and used to calculate mixing height using a scheme developed by the Finnish Meteorological Institute (FMI). Comparisons were made with mixing heights calculated using three different schemes and using radiosonde data from a rural site. Mixing heights calculated with the FMI scheme did not correlate well with the other schemes, although these did correlate well with each other. The rural site is less than 100km from Helsinki, but the two sets of data only poorly correlated. The data were further divided into 12 wind sectors. The two wind sectors giving the poorest correlation were when wind was from the south (from the sea) and from the east (wind blowing over the urban area towards the monitoring site). The rural site was 100km inland. These results showed how sensitive the mixing height is to the local topography. The mean values were identical when wind was from the northwest, but varied by up to 100 metres with wind from the south.

Several modelling studies have compared the use of meteorological data from different sites. Some relating specifically to ADMS have been reviewed elsewhere, but the work of Manning *et al.* (2000) is particularly relevant as it compared modelling outcomes using data gathered within the urban area with data gathered at a distant synoptic site. Manning *et al.* (2000) investigated the performance of a street canyon model (AEOLIUS) using data recorded at the modelling site in Leek and from Manchester airport, which is 40km away. AEOLIUS is based on the Danish OSPM street canyon model which is incidentally also used as a sub-module within ADMS-Urban. Street canyon modelling requires the 'roof top' wind, that may be measured above the canyon or from a distant site, to be transformed into a 'street level' wind. There are several different methods for doing this. OSPM assumes a logarithmic relationship between rooftop and street level wind.

$$u_{street} = u_{roof} \text{ x } (\ln \ (/z_0) / \ln (H/ z_0)) \text{ x } (1- 0.2 \sin \phi)$$

Where h_{street} is the emission height, H is the canyon height, z_0 is the surface roughness and $\sin \phi$ is the angle between the wind and the street axis.

Although Manning *et al.* (2000) collected roof top wind speed data, it is most unlikely that such data would be routinely available to the prospective model users and instead they would have to rely on wind data recorded at 10m height from synoptic stations. In this case the model must first transform the synoptic wind speed to a roof top wind speed at the appropriate height using a logarithmic relationship before the street level wind is calculated. Manning *et al.* (2000) found that on average wind speeds were higher at Manchester even though Leek wind speeds were recorded at 16.2m. The correlation coefficient (r) was 0.77. The wind direction data showed better agreement. Using the Manchester airport data to run AEOLIUS inevitably produced lower output concentrations. The calculated pollutant data obeyed the following linear relationship with an r value of 0.68;

Manchester concentration = 0.73 x Leek concentration - 0.05.

This further supports the need to harmonise how data are prepared for regulatory modelling and to ensure that it is representative of urban area.

The variation found in meteorological data gathered over quite small distances has been demonstrated in Chapter 4 and the sensitivity of ADMS-Urban to meteorological parameters has been discussed in Chapter 5. This chapter shows how meteorological data from different sources affects the model output. Model runs were carried out using data obtained from a variety of sources; the pollution monitoring site, an automatic weather station within the urban area (Moulton Park) and only 1km to the northeast of the pollution monitoring site and from synoptic sites at Wittering, Norwich, Hemsby, Coleshill and Benson. The location of the synoptic sites is described in Appendix Four and shown in Figure 3.3. The two urban sites both used Wittering cloud cover. A further modelling exercise was carried out with three receptor sites in Kingsthorpe. Model output using four sets of meteorological data was compared with monitored data.

6.2 Results

6.2.1 Exercise 1 – Modelling using 'transect' meteorological data

Model runs were carried out for July and August 1999 using meteorological data from the Kingsthorpe monitoring site, the Moulton Park AWS and the following synoptic sites;

Wittering, Norwich and Hemsby. These sites are at increasing distance to the northeast from the air quality monitoring site.

A comparison of model output with monitored carbon monoxide from Kingsthorpe shows that whichever source of meteorological data is used, the model generally under-predicts. Obviously the choice of a background value is crucial in assessing model performance. The background data used here were determined by local monitoring and the method used to derive it is described in Appendix Seven. The under-prediction might suggest that background concentrations should be increased, however under-prediction is generally due to the inability of the model to accurately predict peak concentrations, so the solution is not straightforward. The background data do mirror the bi-modal pattern of traffic flow with morning and evening concentration peaks (Figure 6.1) and without having monitored background data concurrent with the roadside pollution data, the scheme adopted here seems a reasonable compromise.

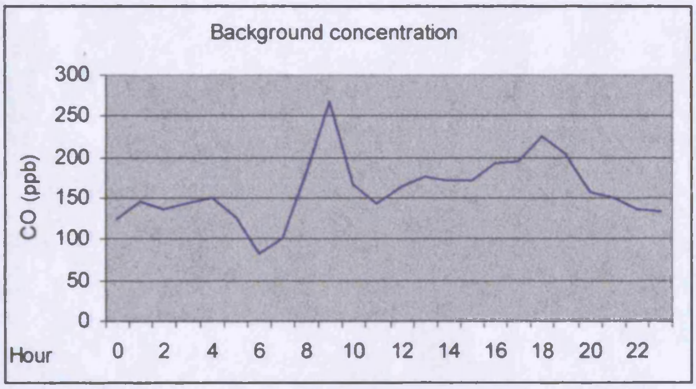


Figure 6.1 Diurnal background carbon monoxide concentration for Kingsthorpe.

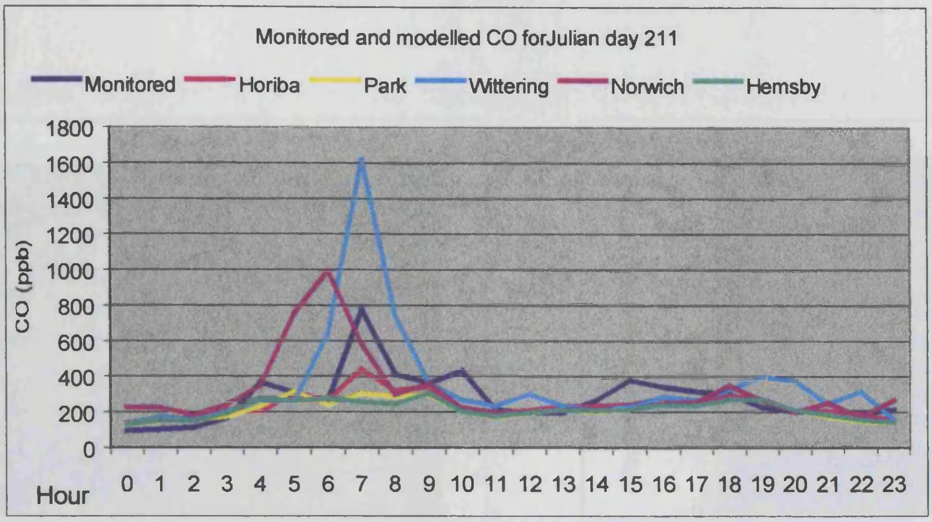
Model performance statistics for exercise 1 are given in Table 6.1. An explanation of the statistical tests used is given in Appendix Nine. Plots of model output and scatter diagrams showing the relationship between monitored and modelled carbon monoxide are shown in Figures 6.2 and 6.3.

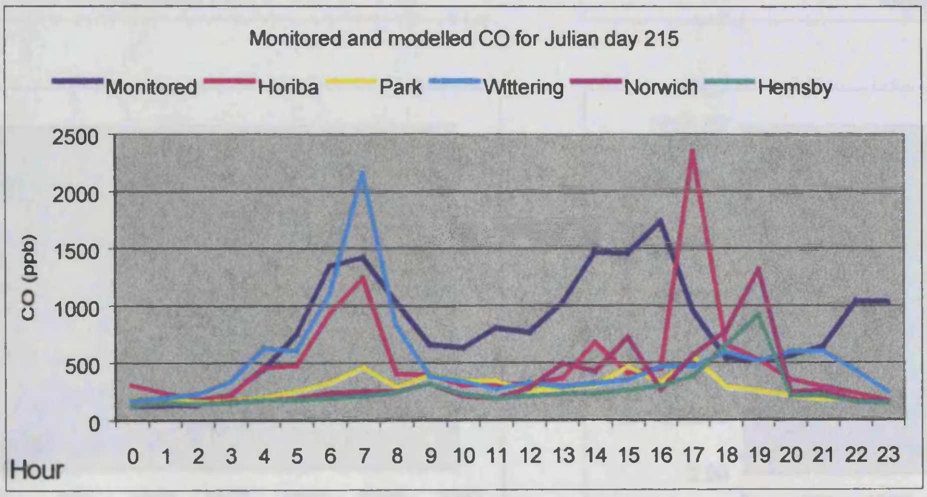
Meteorological data	FB	NMSE	r^2	I of A	FAC2
Kingsthorpe	-0.13	0.69	0.29	0.69	0.73
Moulton Park	-0.48	0.95	0.37	0.54	0.67
Wittering	-0.34	0.70	0.27	0.60	0.70
Norwich	-0.21	0.68	0.18	0.56	0.68
Hemsby	-0.36	0.82	0.18	0.50	0.65

Table 6.1 Performance measures – transect data.

The local meteorological data gives the best overall performance. The low wind speeds recorded at Kingsthorpe result in some very large over-predictions at certain times of day, these can be up to ten times the monitored values, but generally the model still under-predicts. Even so the normalised mean square error is relatively low and Kingsthorpe data produces the highest number of results within a factor of two. Although the Moulton Park meteorological data are obtained from within the urban area and predicted CO correlates well with monitored data, the tendency to under-predict still leads to a poorer performance. Hemsby is the most distant site and performs worst by all performance measures, but the results for Norwich are conflicting. The fact that Norwich performs well by some measures cannot easily be explained. Norwich is over 150 km from Northampton, but data comes from the Norwich Weather Centre that is situated in the centre of the urban area and as a result it may provide more appropriate data for urban modelling. The built environment may be affecting the mesoscale meteorology in a similar way at both locations.

The fact that Wittering data does perform reasonably well and is better than the Moulton Park data, may suggest that this better depicts the general meteorological conditions over Northampton. This is corroborated by the close agreement in meteorological data found amongst the rural sites in Northamptonshire (see Chapter 4) and how closely they match Wittering. The model may be able to predict boundary layer conditions more accurately with Wittering data than with data from the urban area, which only represents a narrowly





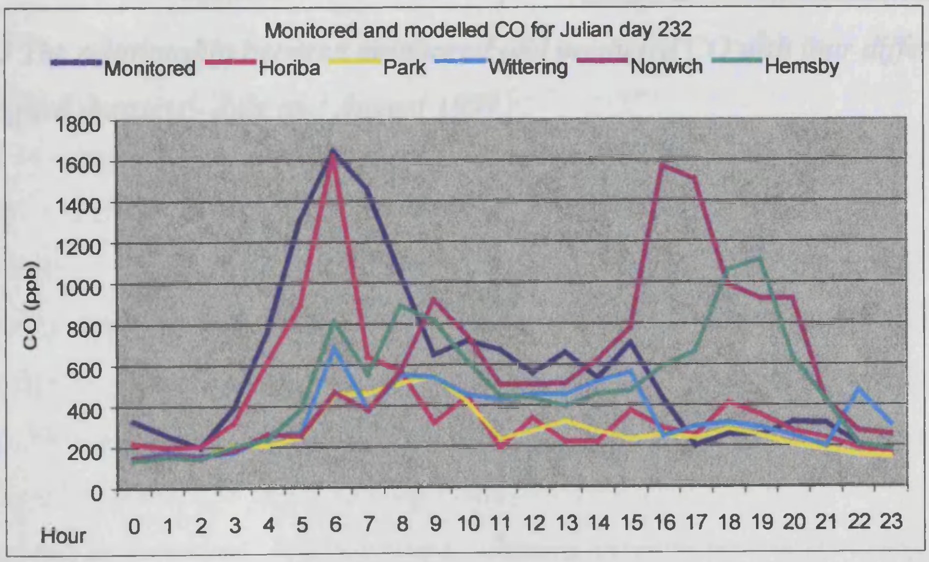


Figure 6.2 Monitored and modelled carbon monoxide for three separate days during July and August 1999.

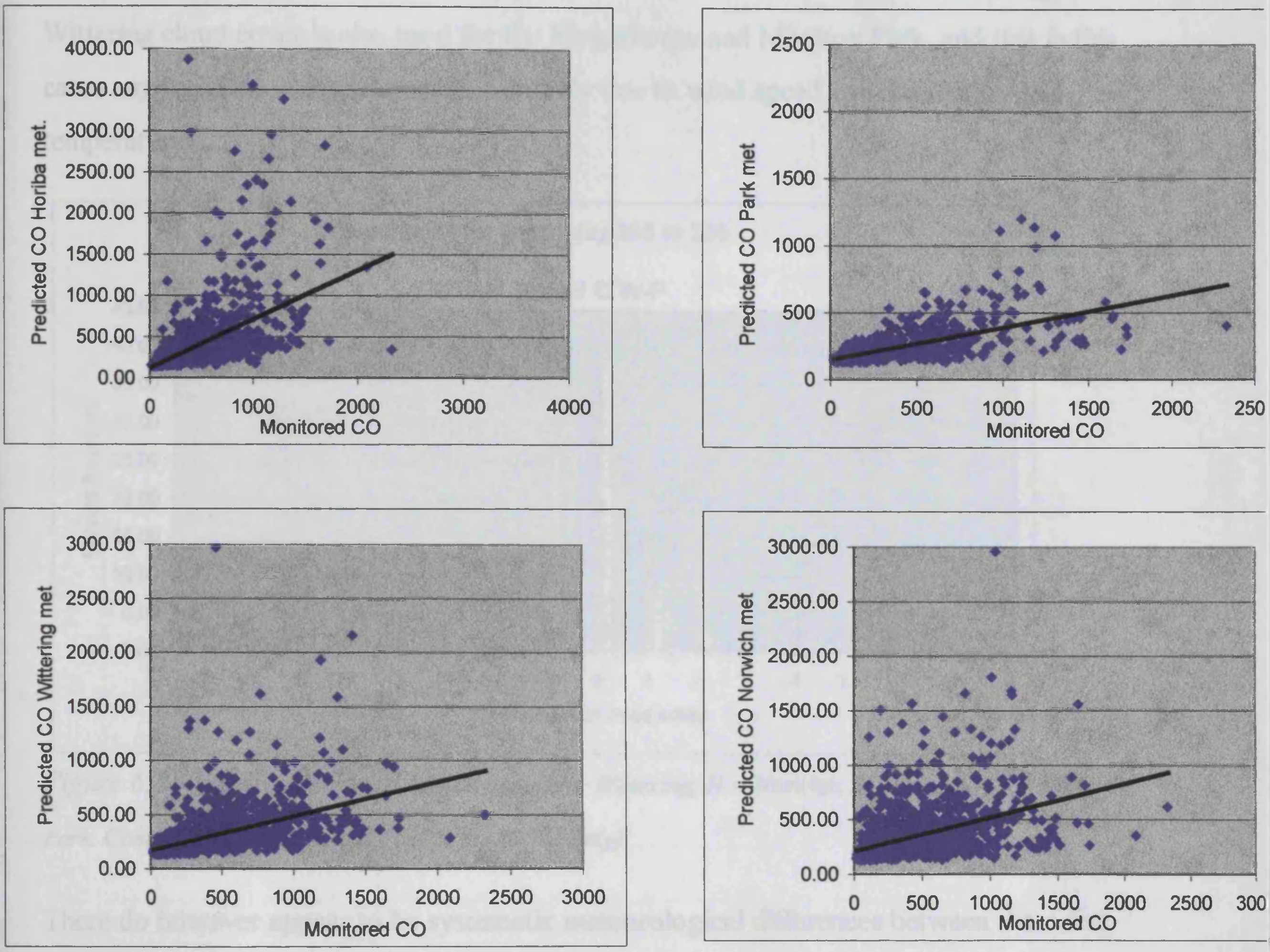


Figure 6.3 The relationship between monitored and predicted CO with four different meteorological datasets - July and August 1999.

defined domain. It should be pointed out that as cloud cover is not recorded locally, Wittering cloud cover is also used for the Kingsthorpe and Moulton Park, and that in this cases any variation in model output is entirely due to wind speed, wind direction and temperature.

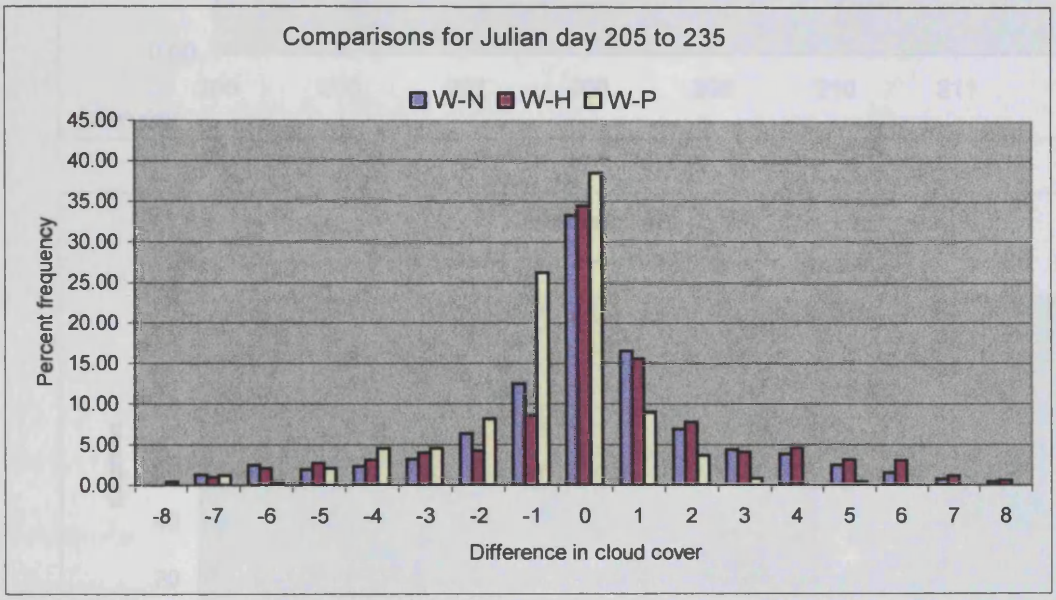


Figure 6.4 Variation in cloud cover (note W = Wittering, N = Norwich, H = Hemsby, P = Moulton Park. Comparison with Moulton Park is for 0900Z only).

There do however appear to be systematic meteorological differences between these five sites and this is reflected in how the model responds to the data. Analysis of cloud cover data at Northampton is given elsewhere, but Figure 6.4 shows that there is more similarity between Northampton and Wittering than there is between Wittering and either Norwich or Hemsby. On average these two sites are less cloudy than Wittering, whereas Northampton is cloudier. Figure 6.5 shows how the wind speed and the temperature vary over a 7-day period and how wind direction varies over 3 days. Figure 6.6 shows how data from this same time period is used by ADMS-Urban to determine boundary layer conditions. Hemsby data are not shown, but there are some notable differences between this and the other sites. The most obvious features is that there is less diurnal variation in wind speed and temperature. Wind speeds at Hemsby are as high as at Wittering during the day, but they do not fall as low at night. The proximity to the sea also results in cooler

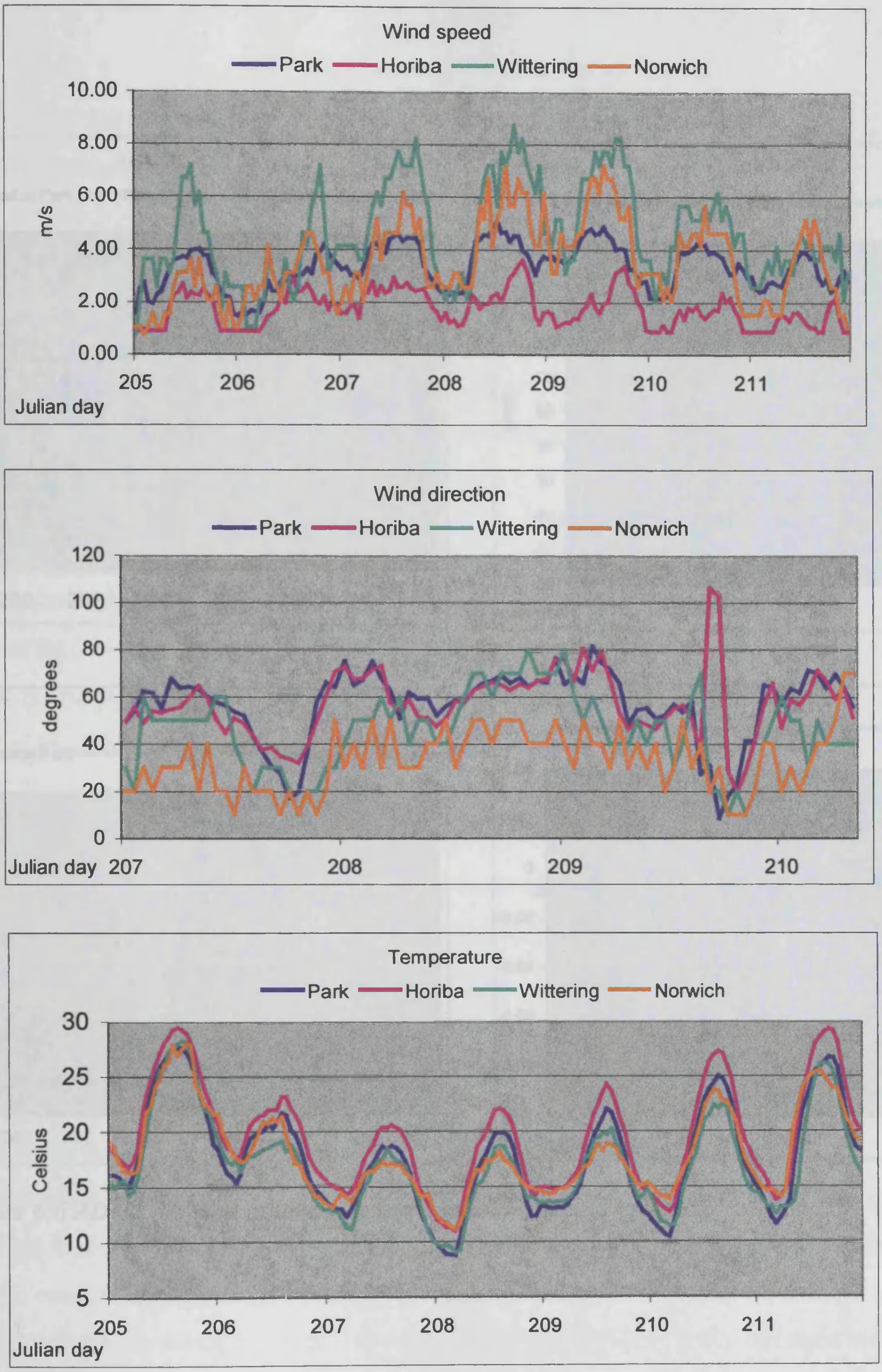


Figure 6.5 Variation in meteorological parameters. (note Park = Moulton Park, Horiba = Kingsthorpe).

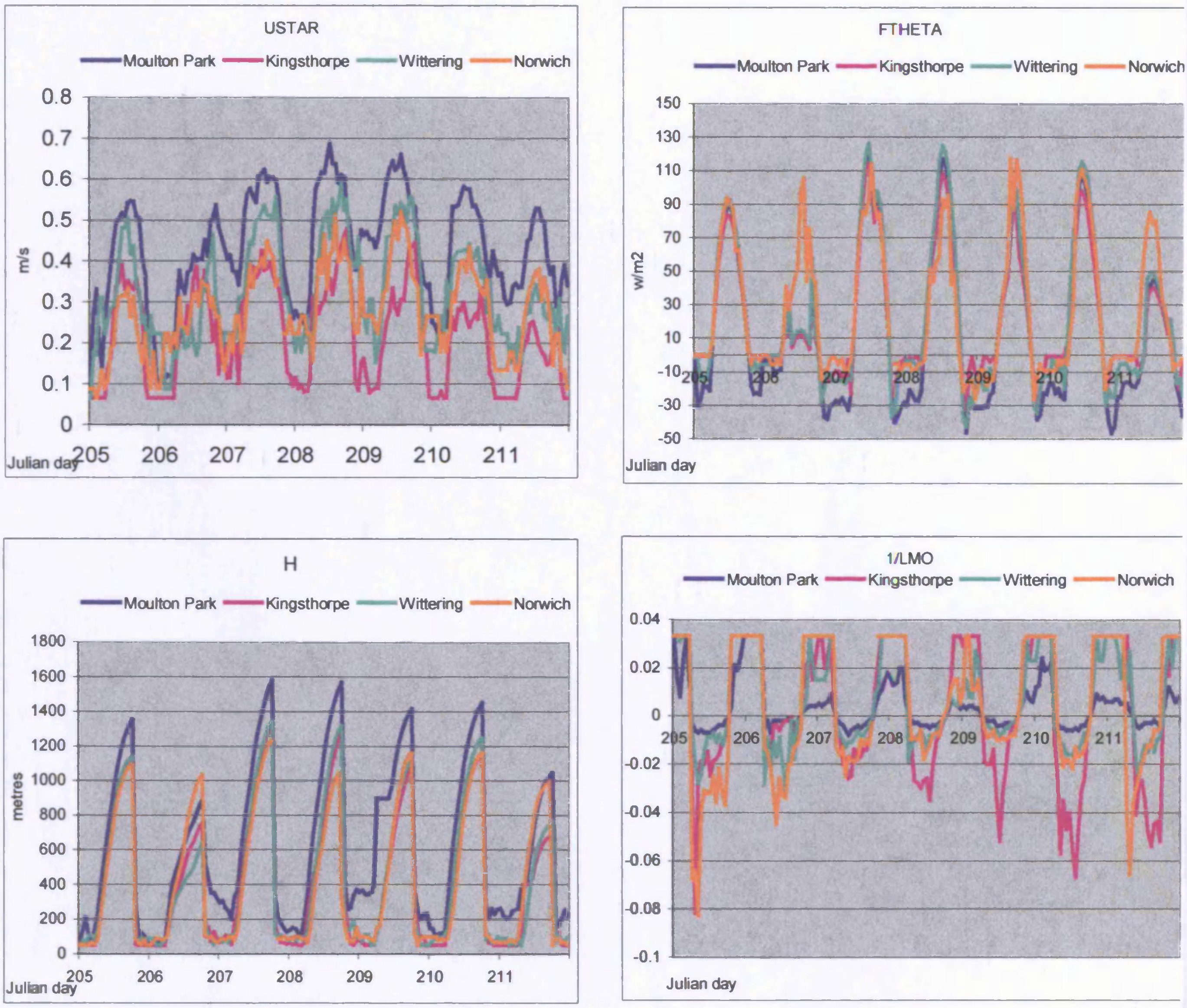


Figure 6.6 ADMS - Urban boundary layer conditions.

daytime and warmer nighttime temperatures, but the wind direction data shows no evidence of land-sea breezes affecting this site.

It was suggested above that the urban nature of the Norwich site might have some bearing on how the model performs using this data. Night-time temperatures are similar to Northampton and are higher than at the rural sites, however day time temperatures do not show any enhanced warming effect of the urban environment. Norwich experiences lower midday temperatures than Wittering. The higher wind speeds are recorded in Norwich than in Northampton.

Bearing in mind the comments made above about the suitability of the background data and the knowledge that ADMS shows increased sensitivity to input parameters at certain time of day, the percentage of modelled data within a factor of two of the monitored data (FAC2) was recalculated for four separate periods during the day. Night-time data (2000Z to 0500Z) shows that all sites still achieve an FAC2 statistic of between 0.65 and 0.75. However there is an increased tendency to over-predict rather than under-predict, this may be entirely due to an over-estimation of the background data. The only two sites to give a relative improvement in performance at night are Moulton Park and Hemsby. All sites show an increased tendency to under-predict during the morning (0600Z to 0900Z) and during the middle of the day (1000Z to 1500Z). This is undoubtedly due to the difficulty that the model has in accurately predicting peaks in pollutant levels that coincide with the morning rush hour.

Peak traffic flows tend to coincide with times of day when the model is most sensitive to meteorological input data, particularly cloud cover. It is at these times of day that using the Kingsthorpe meteorological data gives the largest improvement in the FAC2 statistic (0.77), even though Wittering cloud cover was used. Moulton Park data performs particularly poorly during morning and middle of the day (FAC2 = 0.58). At night and during late afternoon the performance with Moulton Park data is almost a good as that attained using the Kingsthorpe data, so this poor performance at other times is hard to explain.

Two other statistical tests were applied to the data. The Spearmann rank correlation was used as a non- parametric measure of relationship between data sets. Although this is more appropriate as the data are not normally distributed, it does not test the ability of the model to match the monitored data hour by hour.

Spearmann rank correlation with monitored data	r_s
Kingsthorpe	0.72
Moulton Park	0.68
Wittering	0.68
Norwich	0.56
Hemsby	0.60

Table 6.2 Spearmann rank correlation - all significant at the 0.01 level (two tailed).

The predicted data from the different sources was also compared. The highest correlation was between Norwich and Hemsby data (r = 0.80) and the worst between Norwich and Moulton Park (r = 0.60).

The Mann Whitney test is a non-parametric test for significant differences between two samples. The test was carried out comparing the monitored data with each set of predicted data and between the predicted datasets. At the 0.001 (two tailed) significance level there is statistically no difference between the monitored data and any of the predicted datasets. The only data that are significantly different are the predicted data derived from Kingsthorpe and Norwich meteorological data.

6.2.2 Exercise 2 - Modelling using closest synoptic site meteorological data

Data from the three synoptic sites that were analysed in Chapter 3 were used to run ADMS-Urban, for a five week period during November and December 1999. Plots of model output and scatter diagrams showing the relationship between monitored and

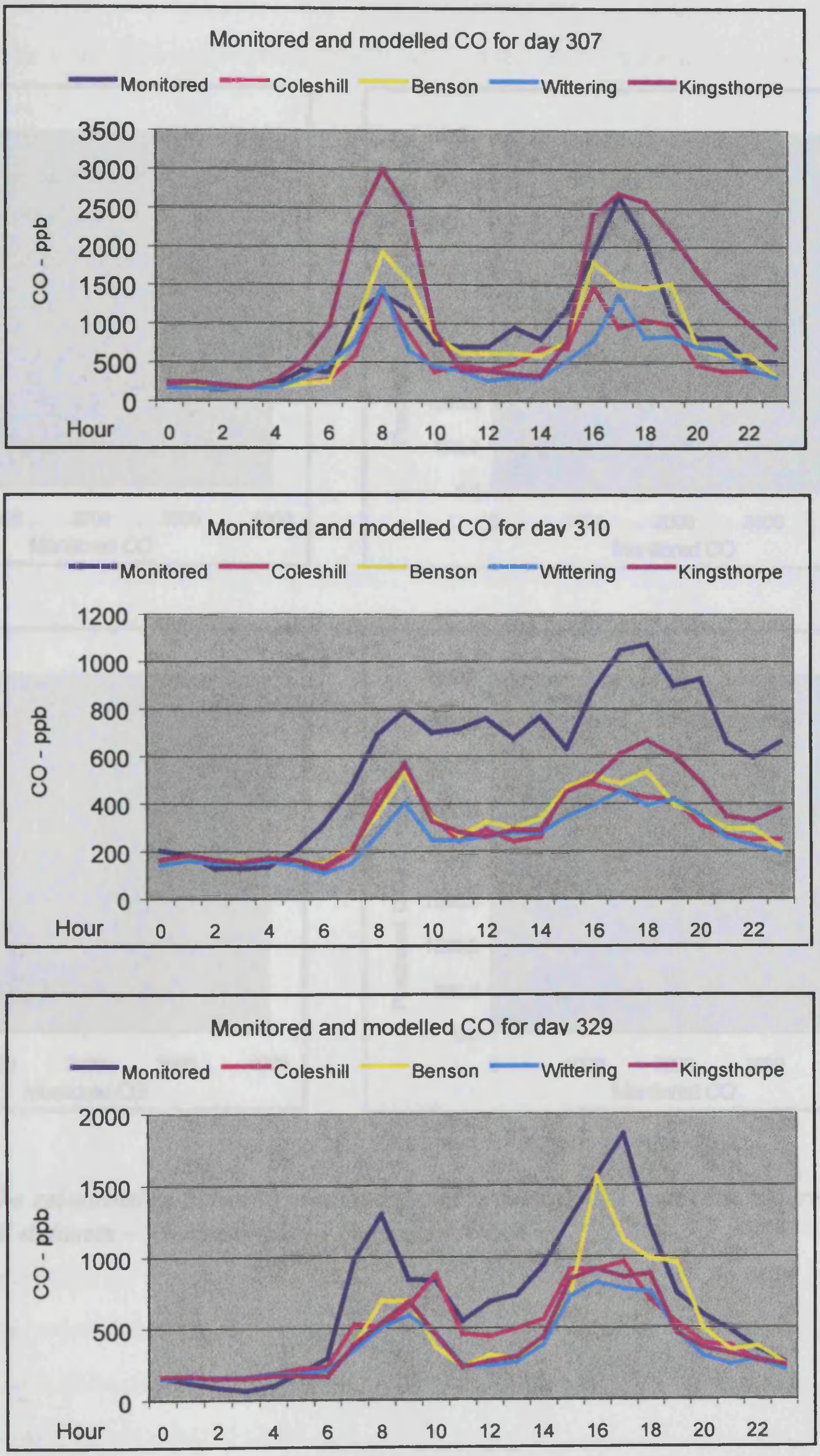


Figure 6.7 Monitored and modelled carbon monoxide for three separate days during November 1999.

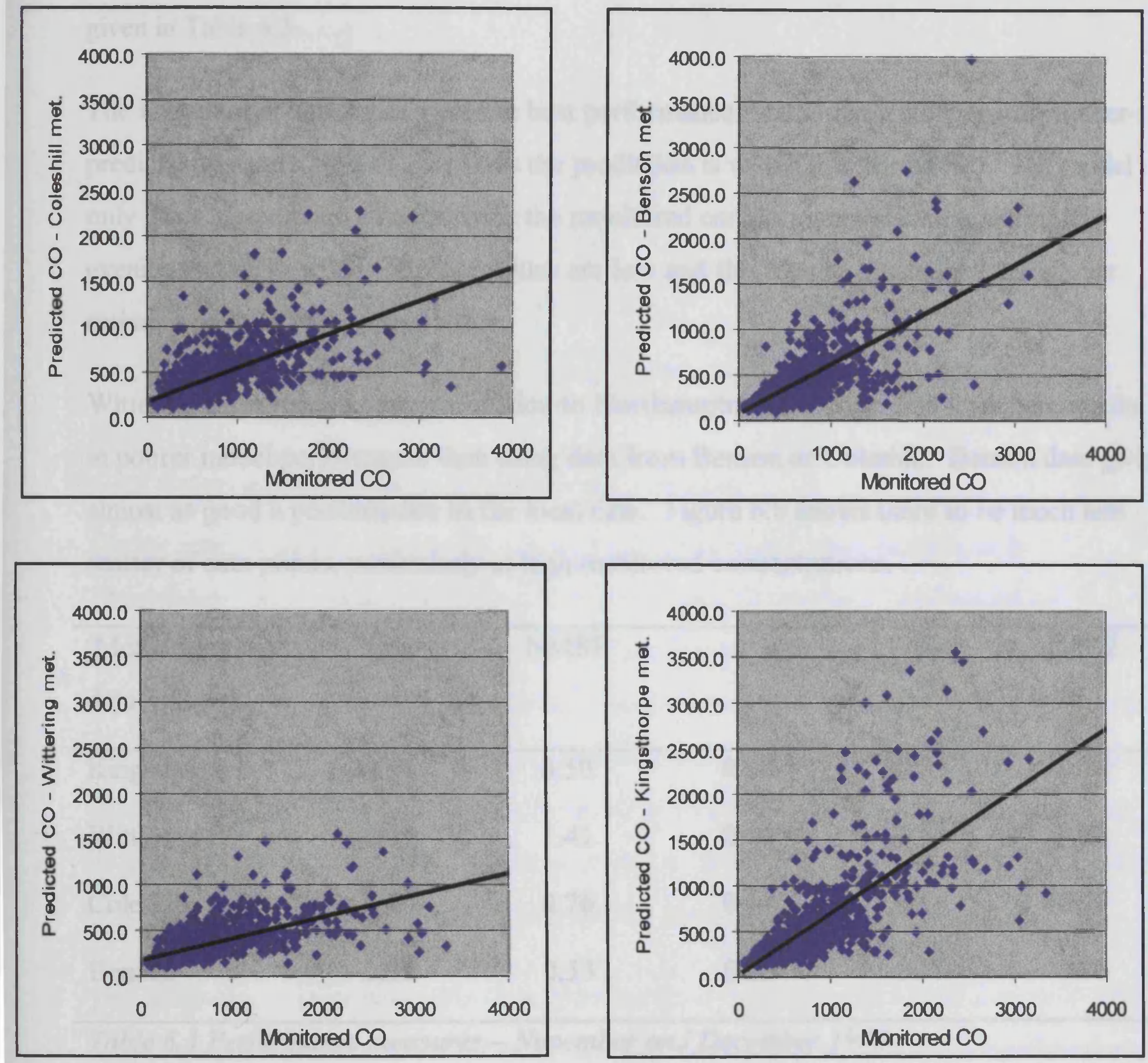


Figure 6.8 The relationship between monitored and predicted CO with four different meterological datasets – November and December 1999.

modelled carbon monoxide are shown in Figures 6.7 and 6.8. Performance measures are given in Table 6.3.

The Kingsthorpe data again gave the best performance. Although it still generally under-predicts, on nearly 80% of occasions the prediction is within a factor of two. The model only gives a prediction of over twice the monitored carbon monoxide value during the evening and night when pollutant values are low and this may be attributable to an over estimation of the background value.

Wittering is the nearest synoptic station to Northampton, but using data from here results in poorer model performance than using data from Benson or Coleshill. Benson data gives almost as good a performance as the local data. Figure 6.8 shows there to be much less scatter of data points, particularly at high monitored concentrations.

Meteorological data	FB	NMSE	r^2	I of A	FAC2
Kingsthorpe	-0.34	0.50	0.58	0.83	0.78
Wittering	-0.69	1.41	0.44	0.48	0.58
Coleshill	-0.46	0.76	0.47	0.66	0.73
Benson	-0.38	0.53	0.53	0.79	0.75

Table 6.3 Performance measures – November and December 1999.

Spearmann rank correlations were also carried out on this data. Results show good correlation with the monitored data for all four datasets. There is less discrimination with Spearmann than with the Pearson correlation, this might lead model users to take less care in selecting the most suitable input datasets. The Mann Whitney test also shows that there are no significant differences either between the different sets of model output and between these and the monitored data.

Spearmann rank correlation with monitored data	r_s
Kingsthorpe	0.87
Wittering	0.78
Coleshill	0.80
Benson	0.81

Table 6.4 Spearmann rank correlation - all significant at the 0.01 level (two tailed).

6.2.3 Exercise 3 – Modelling for three receptor sites

During spring 2000 the Horiba mobile air quality laboratory was placed back in Kingsthorpe. For a four-week period two Streetbox CO monitors were placed on lampposts along the same stretch of road. The location of these is shown in Figure 6.9. It can be seen from this that the monitor near the old tram stop is on a traffic island with an open aspect and as such has traffic on all three sides. The buildings on the western side of the road, adjacent to the travel agent site are three storeys high. The Horiba site is open to the north and northeast and has single storey buildings adjacent with two story buildings on the other side of the main road. None of the roads was defined as a street canyon.

The monitored carbon monoxide data gathered from these three sites was compared with ADMS model output. The model was run using local meteorology from Kingsthorpe and data from synoptic sites at Wittering, Norwich and Hemsby. The same background data was used as in exercise 1 and 2. Only the FAC2 statistic and the Pearson correlation are quoted here - see Table 6.5.

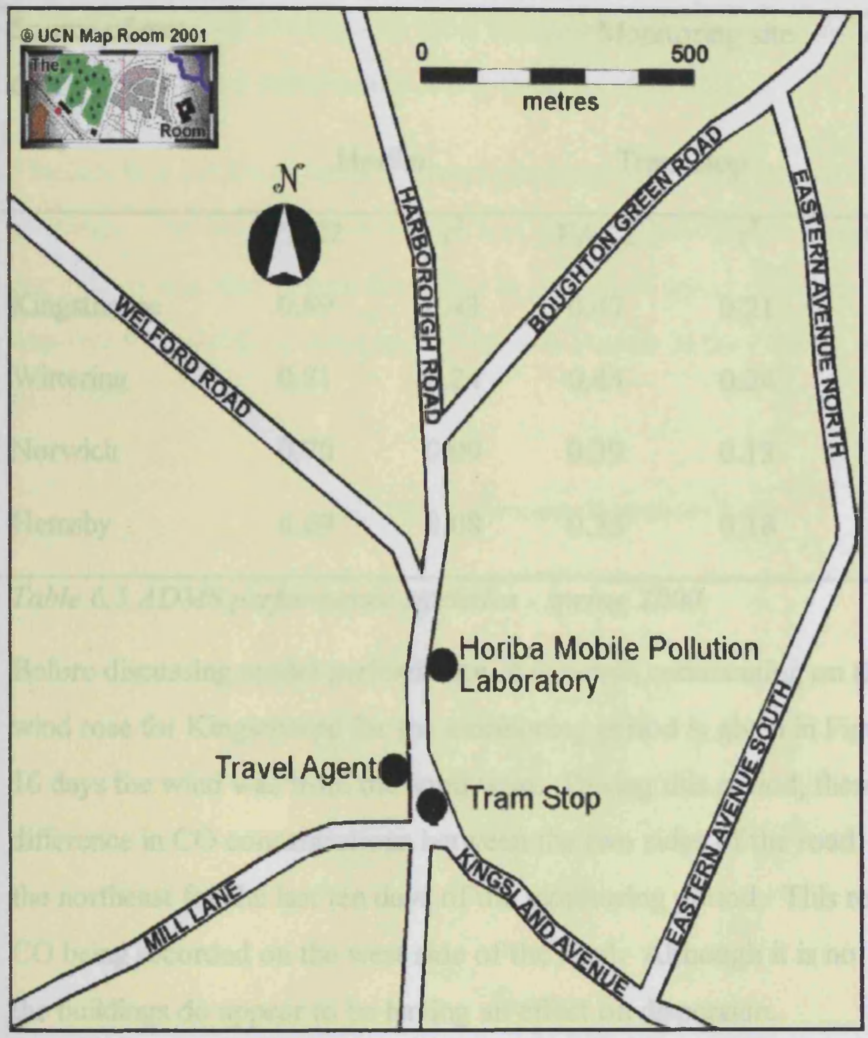


Figure 6.9 Map of Kingsthorpe showing location of pollution monitors.

Source of met. data	Monitoring site					
	Horiba		Tram stop		Travel agent	
	FAC2	r^2	FAC2	r^2	FAC2	r^2
Kingsthorpe	0.89	0.43	0.47	0.21	0.40	0.10
Wittering	0.81	0.24	0.44	0.24	0.37	0.22
Norwich	0.70	0.09	0.39	0.13	0.27	0.06
Hemsby	0.69	0.08	0.35	0.18	0.23	0.16

Table 6.5 ADMS performance statistics - spring 2000.

Before discussing model performance, it is worth commenting on the monitored data. The wind rose for Kingsthorpe for the monitoring period is given in Figure 6.10. For the first 16 days the wind was from the southwest. During this period, there is very little difference in CO concentrations between the two sides of the road. The wind was from the northeast for the last ten days of the monitoring period. This resulted in much higher CO being recorded on the west side of the road. Although it is not a true street canyon the buildings do appear to be having an effect on dispersion.

With all data there was a tendency to for the model to under-predict. All sites performed best with the local meteorological data, but this was most noticeable at the Horiba site. In a rapidly changing urban environment even the data recorded here is unlikely to be truly representative of conditions a few 100 m away. The poor performance of two sites nearer the road may reflect either; the increased difficulty of modelling accurately close to the source where vehicle induced turbulence will have a notable effect, the unrepresentiveness of the meteorological data or inaccuracies in data recording. These two roadside sites used a different type of monitor. An evaluation of Streetbox performance is given in Appendix Six and although they do not give identical readings to the Horiba no assumption was made as to which type of monitor is superior. Performance measures were calculated on uncorrected data, this could contribute a small source of error, but was thought unlikely to significantly affect the overall results.

The fact that the travel agent site performed so badly may be due to the proximity of tall buildings. The street canyon option was not selected for this section of road as buildings are only on one side. More flexibility in ADMS to select different the types of canyon may improve the modelling outcomes. This is a feature of the Dutch CAR model (Eerens *et al.*, 1994).

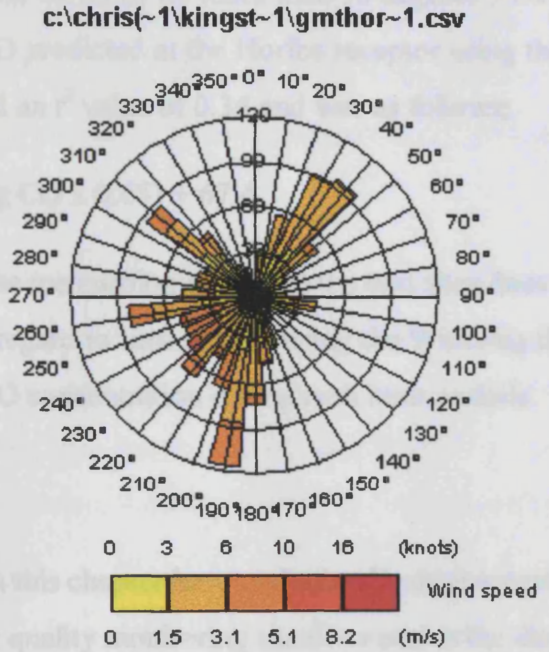


Figure 6.10 Kingsthorpe wind rose for Julian day 53 to 113, 2000.

The differences between the local and Wittering meteorology were analysed. Wittering cloud cover data were used with both datasets. Over the period from Julian day 53 to day 131, the cloud cover recorded at 0090Z was identical on 47% of days. This is better agreement than was found between Northampton and other climatological recording sites in the area during 1999. A further 24% of hours only vary by 1oktas. There is still during this period, a tendency for it to be cloudier at Northampton and using Wittering data can cause significant errors in model output. As already demonstrated in Chapter 4, wind speeds are generally higher at Wittering. When the wind is converted to 10m at

Kingsthorpe the following linear relationship was found for data gathered between February and May 2000;

Kingsthorpe wind speed = (Wittering wind speed x 0.46) – 0.34

The correlation has an r^2 value of 0.73. This comparable to the level of agreement found by Manning *et al.* (1999) between 'roof top' wind in Leek and Manchester Airport. A comparison with the 6m value shows that the wind is actually faster by 2.5 to 3.5 ms^{-1} on 22% of occasions and is more than 3.5 ms^{-1} faster 45% of the time. The wind direction is relatively more stable and varies by no more than 10 degrees 55% of the time. The relationship between CO predicted at the Horiba receptor using the two sets of meteorological data had an r^2 value of 0.34 and was as follows;

Horiba CO = (Wittering CO x 0.65) + 67.4.

Although variation in the meteorology between the two sites does not show any systematic trends with regard to time of day, using the Wittering data is more likely to lead to over-prediction in CO concentration during rush hour periods.

6.3 Conclusions

The results presented in this chapter have undoubtedly demonstrated that meteorological data collected at the air quality monitoring site does enable the dispersion model to predict pollutant concentration more accurately at that site. However it also shows that data collected from elsewhere within the urban area may not give any improvement over data collected from a nearby synoptic site and that even data collected from within the modelling domain may not be appropriate for other sites distant from the meteorological monitoring. The data also highlights some of the question raised by the Royal Meteorological Society (1995) regarding the appropriateness of meteorological data gathered at some distance. For example although Benson is further away from Northampton than Wittering, data from Benson gave better model performance. For the prevailing wind direction in Southeast England, Benson is upwind of Northampton and this may be a relevant factor. The topography of Oxfordshire is also more similar to that of Northamptonshire than the topography at Wittering. However Benson data would not comply with the recommendation by Trinity consultants for use with their own models of 90% yearly data coverage (Hall *et al.*, 1999). Meteorological data from Norwich works well by some model performance measures and although it is over 150 km distant, it is an urban monitoring site.

It has been stated earlier how the model generally fails to predict the high levels of pollutants that occur during the morning rush hour and model performance varies throughout the day with a tendency to under-predict at some times and to over-predict at others. Although sensitivity to changes in environmental conditions would be desirable in a well formulated model, that sensitivity is greatest at the times of days when pollutant levels are already highest should make model users wary of placing too great a reliance on model output.

This work also shows how complex the situation is and that the interplay of the different parameters can cause unexpected results. The Royal Meteorological Society (1995) suggests making short term comparisons of local data with synoptic data, where long term datasets are not available. This may not be as straightforward solution as it initially seems. Unless the relationship between the two sites is clear and consistent, it could lead to more problems than it solves.

CHAPTER SEVEN

ADMS PERFORMANCE IN RELATION TO METEOROLOGICAL DATA

STUDY TWO - BARNES

7.1 Introduction

In February 1997 Hammersmith Bridge was closed to all traffic, except motorcycles and buses, in order that structural repairs could be carried out. Hammersmith Bridge crosses the river Thames connecting Barnes, south of the river, with Hammersmith. Closure of the bridge resulted in a significant drop in traffic levels in the Barnes area as traffic was diverted to Putney and Chiswick bridges. Table 7.1 shows flows across seven bridges.

Bridge	Autumn 1994	March 1997	October 1997
Battersea	25,087	36,034	31,581
Wandsworth	56,840	55,001	52,501
Putney	55,003	70,754	57,103
Hammersmith	30,678	3000#	3,092
Chiswick	49,715	51,352	40,760
Kew	44,587	63,742	60,115
Twickenham	49,595	50,191	48,440

Table 7.1 24-Hour traffic flows on London's bridges. # = estimated flow (URL6).

From Table 7.1 it is possible to see that initially traffic flows increased on all bridges, except Wandsworth, but then fell again, some to even pre-1994 levels. The local authorities concerned could give no explanation for this. Richmond Borough Council also carried out traffic counts in the local area, but the effect of the bridge closure was hard to quantify (Pers comm., John Coates). Some roads in the area showed very little change, however there was an increase in congestion on Clifford Avenue, as this is used as an access road to Chiswick and Kew bridges. Bus travel over the bridge increased by between 14% and 27% depending on route. The bridge re-opened with a weight restriction in place on 21.12.1999. By May 2000 traffic flows over the bridge had returned to 14,563 cars and light goods vehicle, 900 buses and 173 heavy good vehicle per 24 hours (URL6).

As part of a wider epidemiological study Imperial College, Department of Epidemiology and Public Health were interested in obtaining air quality data from the Barnes area whilst the bridge was closed and again after it reopened. Monitoring at Holy Trinity Church, Castlenau (see Figure 3.2), was carried out from November 1999 to February 2000 for a range of air quality parameters. This data formed the basis for a modelling case study. As with the study in Northampton, meteorological data from a variety of sources were used to run the model. Data from three synoptic stations were compared in a similar manner as in Chapter Four.

Traffic related emissions were very low in the Barnes area during this period and as a result were swamped by background pollutants from the Greater London area. Modelling effects of a traffic management scheme on a small part of a large conurbation is difficult without extending the modelling domain to also cover a larger area and without using extensive emission inventories. Neither of which were possible with this study. Although the effects of the bridge closure were not of prime interest, it was decided to still carry out a modelling exercise to investigate the effect of using different meteorological datasets within a larger urban area than Northampton. Local background data were required to add to the model output in order to assess performance. An initial investigation was carried out using background data from different sources and following the recommendations of McHugh (2000), data were selected from the most appropriate source depending on wind direction.

7.2 Results

7.2.1 Meteorological data

Differences between three synoptic stations within the Greater London area were examined using hourly data from 1999 (Appendix Four). All are between 10 to15 kilometres from Barnes; Heathrow to the west, Northolt to the northwest and the London Weather Centre to the northeast. Although Heathrow is in the direction of the prevailing winds and has been used for modelling studies using ADMS in the past (Owen *et al.*, 1999), the London Weather Centre is the nearest site to Barnes and cloud cover data from here were used in the modelling exercise. It was chosen as the main site to which others were compared.

Cloud cover

Figure 7.1 shows the frequency distribution of cloud cover at the different sites. The very high incidence of completely clear skies is questionable but generally the distribution is similar to that found at the three rural sites (compare with Figure 4.9). The London sites are closer together geographically than the rural sites previously studied and this is reflected in the higher percentage of hours at which cloud cover is identical. This is summarised in Table 7.2.

Cloud cover difference	Northolt – London WC	Heathrow – London WC
0 oktas	42.5	51.2
up to +/- 1 oktas	74.9	82.3
up to +/- 2 oktas	84.5	92.1
up to +/- 3 oktas	90.1	96.3
No of records	8989	8744

Table 7.2 Differences in cloud cover as cumulative percentages – 1999.

This shows that cloud cover is identical for approximately fifty percent of the time, however even the thirty percent of occasions when it varies by one okta can be significant if the variation occurs between 6, 7 or 8 oktas.

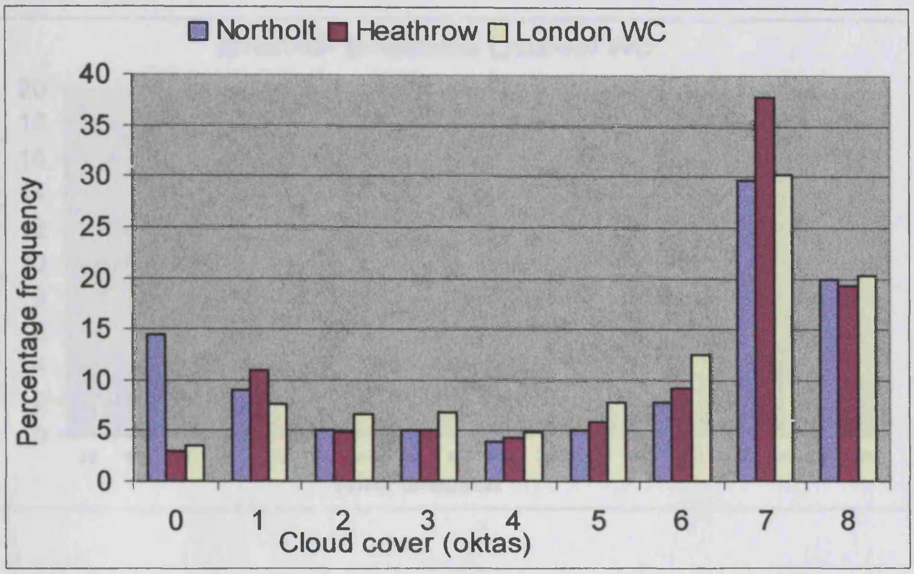


Figure 7.1 Cloud cover distribution at three synoptic stations -1999.

Wind direction

The wind direction distribution shows the similar bimodal distribution to the rural sites, but is particularly exaggerated in the case of the London Weather Centre (Figure 7.2). Its position in central London may result in more influence from surrounding buildings than at the two airfield sites. Although the data for the rural sites were for 1998 and 1999 so no direct comparisons can be made, there does appear to be a veering of the wind over London. This may be due the topographical effect of the Thames valley as much as anything else.

Difference in hourly wind direction readings again shows greater similarity than seen with Wittering, Coleshill and Benson. The disparity in the wind direction is greatest when comparisons are made between the London Weather Centre and Heathrow (Figure 7.3), however it is often Heathrow data that are used to represent the whole of London. Heathrow wind direction can frequently be 30 to 50 degrees backed in relation to the

London Weather Centre suggesting some degree of systematic variation and significant enough to cause model error.

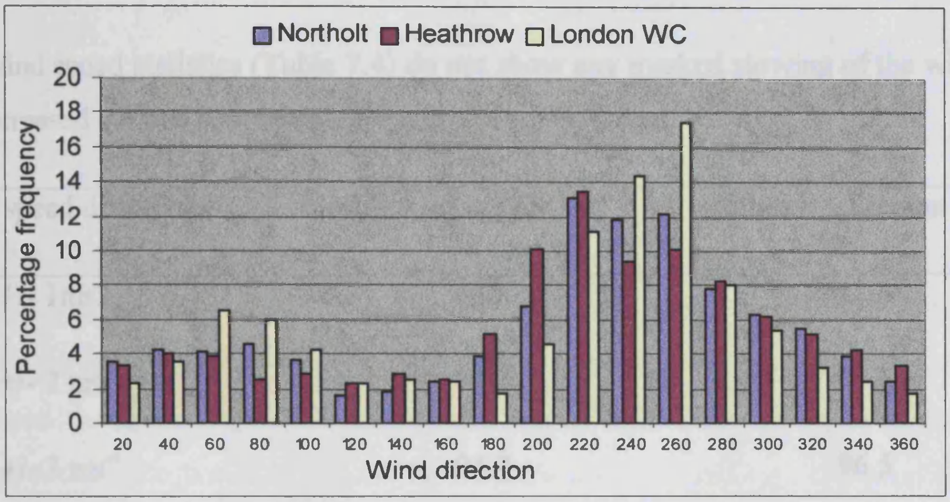


Figure 7.2 Wind direction distribution at three synoptic stations - 1999.

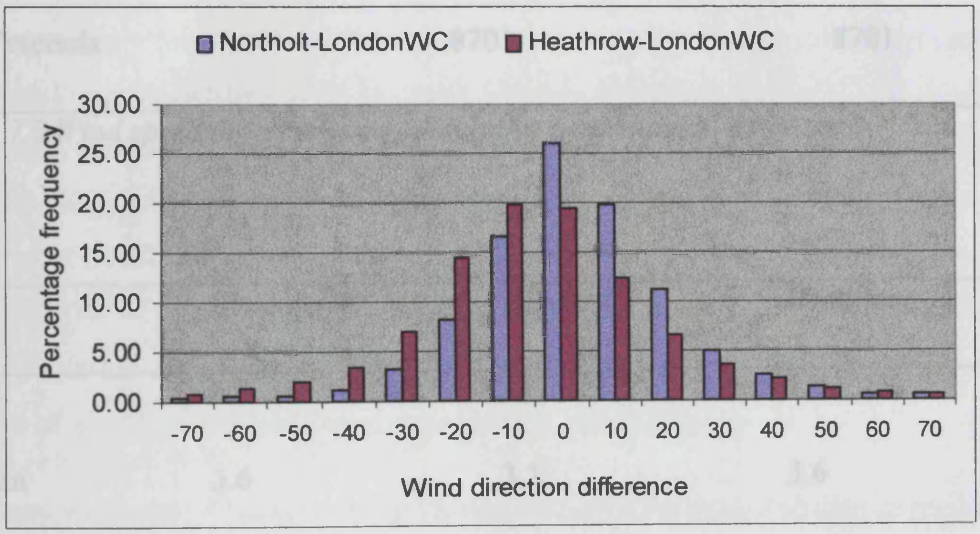


Figure 7.3 Differences in wind direction between synoptic sites - 1999.

Wind speed

Wind speed is again fairly consistent between the three sites, only varying by up to +/- $1ms^{-1}$ on 36% or 42% of the time for Northolt and Heathrow respectively. Table 7.3 show that wind speed very rarely varies by more than $5ms^{-1}$ for the London sites.

The wind speed statistics (Table 7.4) do not show any marked slowing of the wind due to the increased surface roughness of the urban environment.

Wind speed difference	Northolt – London WC	Heathrow – London WC
up to +/- $1ms^{-1}$	36.3	41.8
up to +/- 2 ms^{-1}	75.6	82.6
up to +/- 3 ms^{-1}	94.3	96.5
up to +/- 5 ms^{-1}	99.8	99.9
up to +/- 7 ms^{-1}	100	100
No of records	8701	8701

Table 7.3 Wind speed differences as cumulative percentages –1999.

	London WC	Northolt	Heathrow
Mean	4.0	3.6	3.5
Median	3.6	3.1	3.6
Maximum	13.4	16.5	13.9

Table 7.4 Wind speed statistics (ms^{-1}) – 1999.

7.2.2 Background pollution data

There are several urban background sites in Greater London at which monitoring is either carried out by NETCEN or by the relevant local authorities. However there are only three that are relatively near Barnes and that carry out monitoring for NO_x, NO_2 or NO. These are NETCEN sites at Teddington (9km southwest of Barnes) and the West London site at Kensington (5 km to the east). There is additionally a site run by the London Borough of Richmond near Castlenau in Barnes (wetland site), but data from this site were only intermittent. Background data for a seven-day period for all these sites are shown in Figure 7.4. Data for a two-day period during December 1999, from three of these sites along with monitored and predicted NOx for a site on Castlenau – Holy Trinity Church are presented in Figure 7.5. This shows how on these particular days the monitored pollutant was very similar to the 'background' value recorded at Kensington. Barnes and Teddington background values were also very similar. The wind direction on these two days was predominately from the north-northeast; i.e. blowing pollutants from more built up part of London north of the river. The modelled data were produced using local Barnes meteorology.

The prevailing wind direction is from the southwest so it is likely that background pollution levels in Barnes would be closer to those found at Teddington. Bearing mind that background levels of NO_x can be quite variable depending on wind direction a set of background data were created using data from both Teddington and West London as a function of wind direction. For wind direction 136° to 269° Teddington data were used and from 270° to 135°, West London. This is a procedure recommended by CERC for use when modelling with ADMS is a large urban area (McHugh, 2000). Ideally more sources of would be available to represent more compass points.

To assess which set of background performs best with the modelled data in predicting pollution levels at Holy Trinity Church, correlation coefficients and FAC2 analysis was carried out (Figure 7.6 and Table 7.5).

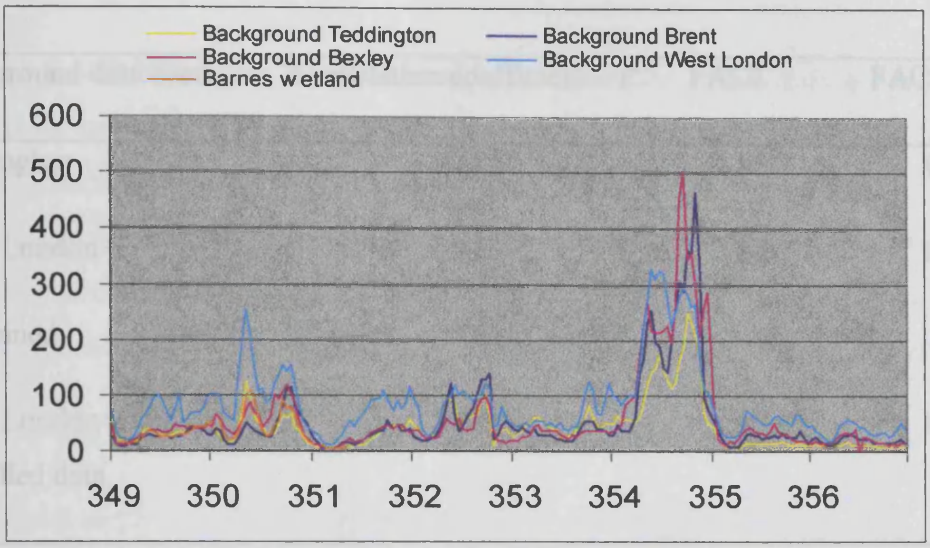


Figure 7.4 Background NO_x levels (ppb) at London background sites - Julian day 349-356, 1999.

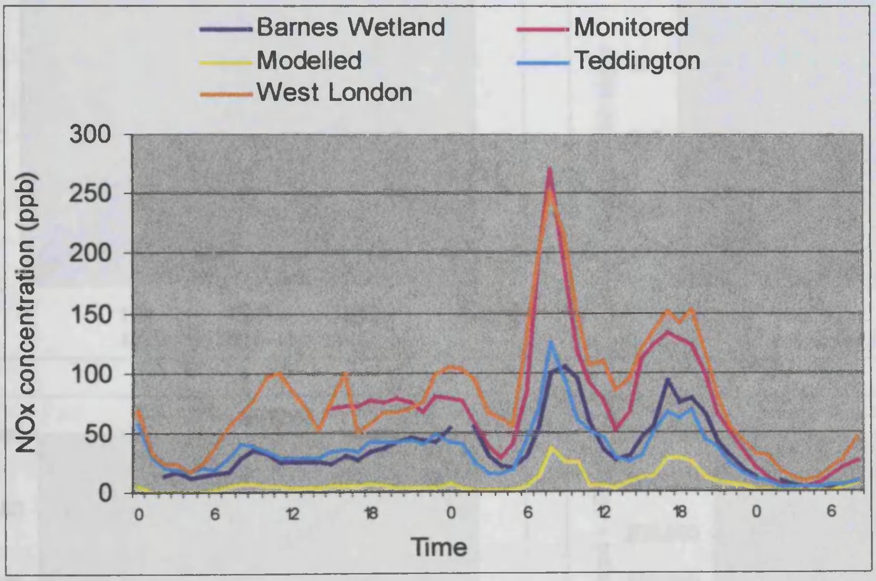


Figure 7.5 Modelled, monitored roadside (Holy Trinity) and monitored background NO_x over a two day period December 1999.

Background data used	Correlation coefficient – r^2	FAC2	FAC>2
Teddington	0.86	85.7	0.8
West London	0.89	80.2	19.8
Combined	0.81	86	1.0
West London without modelled data	0.90	87.8	12.2

Table 7.5 Model performance statistics using different background data.

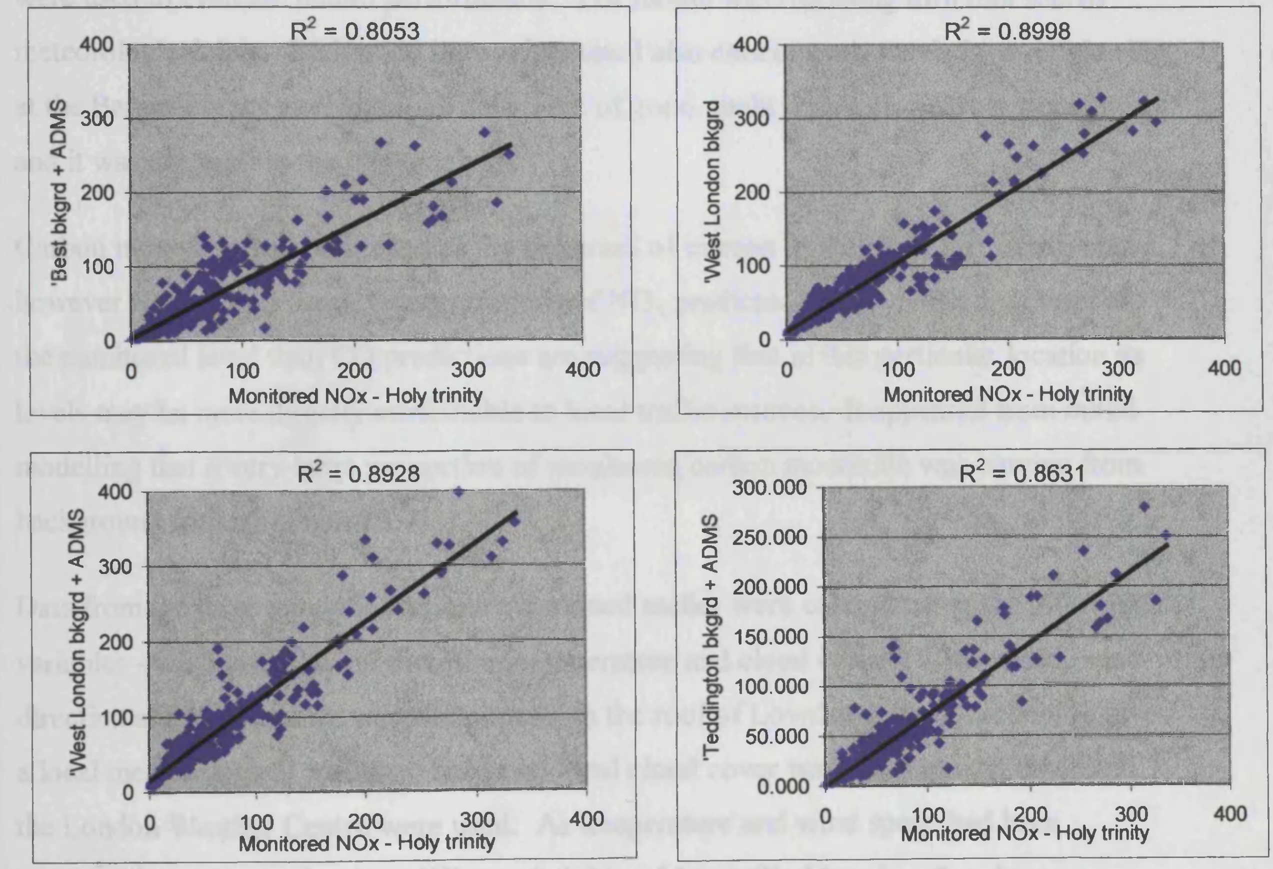


Figure 7.6 Relationships between monitored NO_x at Holy Trinity Church and model output plus background data December 1999.

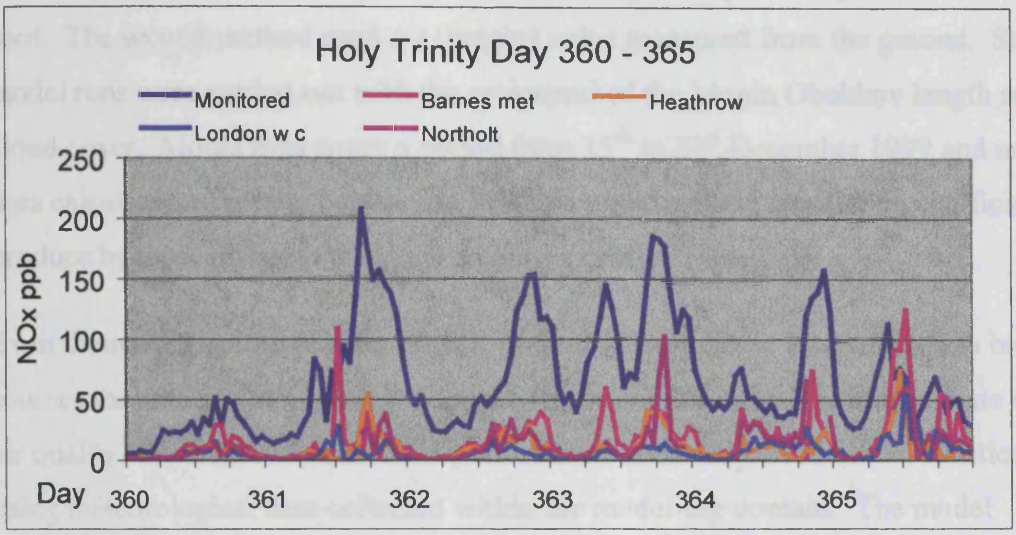
Although the West London data, either with or without the ADMS output produce the best correlation with the monitored data, there is a tendency to overpredict NO_x levels in Barnes. This is shown by the percentage of data points in which the predicted value is more than double the monitored – FAC>2. The combined background data perform marginally better than the Teddington data. This was used in the last modelling exercise in which the performance of the model was assessed in relation to the meteorological data input set.

7.2.3 Model performance in relation to meteorological datasets

Air quality data from Horiba Mobile Pollution Laboratory located at Holy Trinity Church were used to evaluate model performance. The model was run using different sets of meteorological data. Richmond borough council also concurrently carried out monitoring at the Barnes library site, although data were of good quality, pollution levels were lower and it was not used in the final analysis.

Carbon monoxide had been used as the pollutant of interest in the other part of this study, however NO_x is used here. Concentrations of NO_x predicted by the model are closer to the monitored level than CO predictions are suggesting that in this particular location its levels may be more directly attributable to local traffic sources. It appeared from initial modelling that a very large proportion of monitored carbon monoxide was coming from background sources (Figure 7.7).

Data from the three synoptic stations mentioned earlier were entered using the following variables - wind speed, wind direction, temperature and cloud cover. Wind speed, wind direction and temperature were monitored on the roof of Lowther Primary School to give a local meteorological measure, but as no local cloud cover was available the data from the London Weather Centre were used. As temperature and wind speed had been monitored at the school at two different heights a Monin Obukhov length value was determined. Two different methods were used. Firstly using a z value calculated from roof height on the assumption that surrounding building roofs are acting as a 'canopy'.



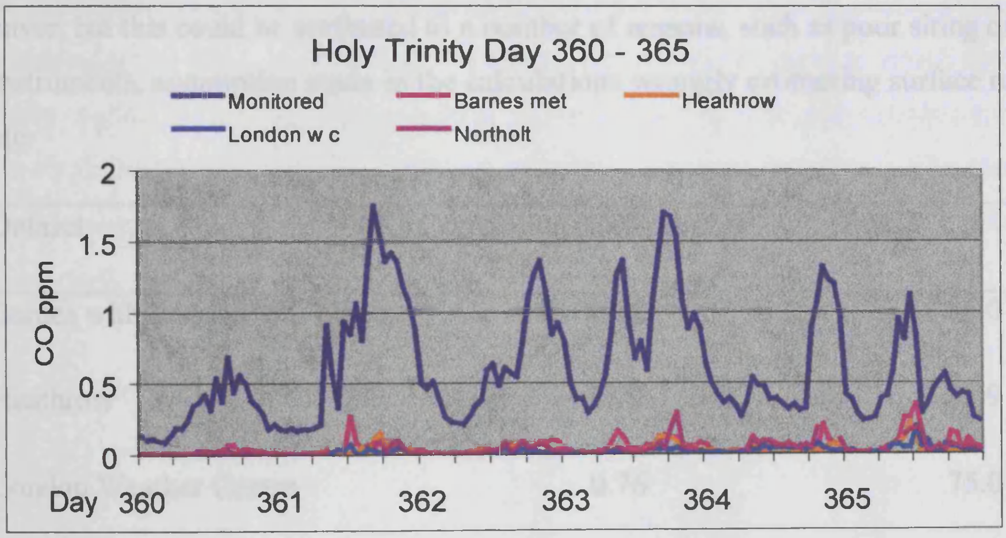


Figure 7.7 Modelled NO_x and CO using different meteorological datasets.

This may in fact be the best method for calculating a temperature profile from a flat black roof. The second method used a z (height) value measured from the ground. Some model runs were carried out with the reciprocal of the Monin Obukhov length replacing cloud cover. Model runs cover a period from 15th to 31st December 1999 and used traffic data calculated on a daily basis. The FAC2 statistic and the correlation coefficient produce by these different data sets are given below (Table 7.6).

Even though a large proportion of NO_x present in Barnes can be attributed to background sources the addition of a modelled local component does give a better estimate of local air quality. Both these measures of performance show improved model function when using meteorological data collected within the modelling domain. The model performance is not improved by inputting a Monin-Obukhov value instead of cloud cover, but this could be attributed to a number of reasons, such as poor siting of instruments, assumption made in the calculations wrongly estimating surface roughness etc.

Dataset	Correlation coefficient R^2	FAC2
Barnes with London WC cloud	0.80	86.0
Heathrow	0.72	72.9
London Weather Centre	0.76	75.0
Northolt	0.78	79.6
Barnes with LMO – method 1	0.68	77.4
Barnes with LMO – method 2	0.68	78.2
Only background data	0.73	54.8

Table 7.6 Performance statistics using different meteorological data.

7 3 Conclusions

This modelling exercise was far from ideal for assessing the performance of an urban air quality model for a number of reasons. Only a very limited emission source database was used, consisting of only a few roads with very low traffic numbers. Many assumptions were made about the physical surroundings of the model domain and although relatively good quality traffic data were obtained, any emission from this traffic was swamped by background pollutants from the larger urban area. However it was valuable in that it highlighted some of the problems that might be encountered in trying to model changes in pollutant levels associated with small scale traffic management schemes within a large urban area. Pollution levels in Barnes were very similar to background levels recorded elsewhere, but the addition of the modelled component did generally improve the quality of the prediction and the exercise again showed the value of using locally obtained meteorological data. Even though the synoptic sites did not show as much variation as those studied in Chapter Four, in a large conurbation with a highly variable surface form the requirement for locally sourced data may be even more important. The use of background data from different sources as a function of wind direction was also shown to work well.

CHAPTER EIGHT

CLOUD COVER

8.1 Introduction

To run ADMS the meteorological data input set must contain wind speed and wind direction as well as the reciprocal of the Monin Obukhov length ($1/L_{MO}$) or surface sensible heat flux ($F_{\theta 0}$). If neither $1/L_{MO}$ nor $F_{\theta 0}$ is available then the Julian day, the hour of the day and cloud cover must be entered in order that the meteorological pre-processor can calculate $F_{\theta 0}$. Once the surface sensible heat flux has been calculated the model can then go on to calculate friction velocity and the reciprocal of the Monin Obukhov length. Julian day and hour are initially used to calculate the solar elevation (s). For values of s equal to or less than 0, i.e. at night, $F_{\theta 0}$ is a function of density, specific heat capacity, friction velocity and θ_*, where $\theta_* = 0.09(1 - 0.5(C_l/8)^2)$ and C_l is the cloud cover in oktas. During daytime when s is more than 0, $F_{\theta 0}$ is derived by using s and C_l to calculate initially incoming solar radiation (K^+) and then net radiation (Q^+).

$$K^+ = (990s - 30)\,(1 - 0.75(C_l/8)^{3.4})$$

$$Q^+ = ((1 - r)K^+ + 5.41 \times 10^{-13}T_0^{\,6} - 5.67 \times 10^{-8}T_0^{\,4} + 60(C_l/8))/1.12$$

where r is surface albedo, assumed to be 0.23 as an average of non snow-covered surfaces and T_0 is the near surface temperature. $F_{\theta 0}$ is then a function of net radiation, the modified Priestly-Taylor parameter, the specific latent heat of vaporization of water, specific heat capacity and the saturated specific humidity at the stated temperature (Thomson, 1992). The Priestly-Taylor parameter is a function of surface moisture and is used in the partitioning of energy between latent and sensible surface heat fluxes.

Obvious sources of error in using these equations come from deploying a default value for albedo and the modified Priestly-Taylor parameter, and by not taking into account the fact that incoming radiation is not only influenced by the amount of cloud cover, but by cloud base height and cloud density. The scheme used for estimating $F_{\theta 0}$ has been developed empirically and is based on results obtained over mid-latitude vegetated surfaces without snow and the model developers recognise that this can lead to uncertainty. However albedo is an optional input parameters so adjustment can be made for large urban or heavily forested surfaces. The same applies to the Priestly-Taylor parameter. The moist grassland value of 1.0 is used as default, but this may not be appropriate for 'dry' urban surfaces and alternative values could be input. A value of 0.45 is suggested for dry grassland and 0 for dry bare earth. With dry surfaces less moisture is available for evapotranspiration, as a result the latent heat flux will be reduced and the surface sensible heat flux relatively larger. Both albedo and the Priestly Taylor parameter may show seasonal variation which should be taken into account.

Although the effects of urban environments on boundary layer conditions are well documented (Oke, 1987), with the exception of very large urban areas, errors in albedo or the Priestly-Taylor parameter are unlikely to have a significant effect of model output (see Chapter Five). Errors resulting from the use of cloud cover in the calculation of surface sensible heat flux operate over a much larger spatial scale and are not dependent on changing surface characteristics. As such they are potentially more serious. Most synoptic sites that record cloud cover also record cloud type and height. Although this additional information is available, the ADMS model only uses cloud cover amount.

BOXURB, a box model for forecasting nitrogen dioxide in urban areas using synoptic observations or numerical forecasts of wind and cloud, does take into account cloud height when calculating sensible heat flux. The cloud cover value is effectively reduced if cloud is high (Middleton, 1998). Gardner and Dorling (1999) used readily available local meteorological data to train a Multilayer perceptron (MLP) neural network model to predict hourly NO_x and NO_2 in Central London. The meteorological parameters included low cloud amount, and the base altitude of the lowest cloud. The former, along with time of day, was considered to indicate likely total radiation levels and the latter along with wind speed and visibility to indicate boundary layer depth. Temperature and vapour pressure were also used. Although boundary layer depth and stability indices are important in determining pollutant concentrations, they are not readily available. The MLP model developers considered it important to use a small number of parameters and only those that were observed at most UK synoptic sites.

It should be noted also that cloud cover observations require observers (although some instrumentation is available for cloud base heights) and therefore cloud cover is not available from automatic synoptic weather stations. Net radiation would be more useful than cloud cover but is not routinely recorded at synoptic stations. Global radiation, defined as incoming direct and diffuse shortwave (0.3μm to 3μm) radiation is recorded at a few sites. Automatic observing systems do record a variety of different meteorological parameters that can be used for simple modelling, but none that allow determination of boundary layer height or turbulence characteristics.

The meteorological pre-processing for 'new generation' dispersion models is necessarily more complex than that required for models using Pasquil-Gifford stability curves as a measure of atmospheric turbulence and has been the subject of a research programme within the EU; COST-Action 710. The algorithms for calculating turbulence parameters from variables commonly recorded by automatic meteorological networks were developed during the 1980's, but still little is know about how well they perform or how dependent their performance is on atmospheric stability. Stübi *et al.* (1997) carried out some of the sensitivity analysis of the COST 710 programme. Only the part referring to calculation of sensible heat flux will be reviewed here as it has been already discussed in the context of ADMS. Hourly data was used to calculate heat flux by five different methods; one based on temperature profiles, cloud cover and solar elevation, two based on global radiation measurements and one using net radiation. With the net radiation value taken as reference the two global radiation methods performed best, whilst the temperature profile method tended to underestimate heat flux. The cloud cover method showed poor correlation, but without tendency to over- or under-predict. The sensible heat flux values were then used to calculate boundary layer height. Again using the net radiation vale as reference, the cloud cover method predicted boundary layer height to with +/-250m, where as the global radiation methods were accurate to within +/-150m.

Whilst radiation measurements are only available from 12 sites across the UK, if a more localised value is required, there is no alternative but to estimate incoming radiation from other more commonly observed meteorological parameters; hours of sunshine or cloud cover. Models have been developed using both these variables. Gul *et al.* (1998) compared the performance of two models, one that used a hourly sunshine fraction and temperature (Meteorological Radiation Model MRM), and one that used cloud cover (Cloud-cover Radiation Model CRM), using three years of hourly data from two sites in Switzerland. MRM generally provided better correlation with measured global irradiance, but neither model performed well in overcast conditions. In the case of CRM, the authors considered this to be due to lack of data on cloud type. They cite work by other researchers who claim that sunshine explains 70-85% of the insolation variance whereas cloud cover often accounts for less than 50% and seldom more than 70% (Bennett, 1969). This might be expected intuitively, as for anything less than completely over cast skies, incoming radiation will vary greatly depending on whether the sun is obscured. Gul *et al.* (1998) conclude that sunshine and temperature give a better estimate of irradiance.

Using hours of sunshine has the advantage that monitoring can be automated and can be measured continuously, however cloud cover as it requires no instrumentation is easy to record and is available from many sites, but always as spot readings. It is subject to human error. Surface based observations of cloud cover tend to be about 10% greater than satellite estimates. Ground based observations refer to an area of about 250km^2. (Barry and Chorley, 1992)

How cloud cover effects incoming radiation has been an area of active research for some time. Most of the early attempts at parameterising the relationships between cloudiness and radiation were carried out in the 1940's in the United States. At the time there were only 21 stations recording insolation and although it was already acknowledged that duration of bright sunshine was a better indicator, there were similarly few stations recording this. Attempts were made to quantify the relationship between insolation and the cloud cover that was more widely recorded.

One of the limitations of solely using cloud cover is that it takes no account of cloud density, but Haurwitz (1945) noted that even when cloud data was categorised by cloud cover and cloud density there was still a large scatter of radiation values within each group. Cloud density is not uniform, even with obscured skies, and as with cloud cover its assessment is subject to human error. It is also subject to systematic error. As estimates depend on the amount of light passing through the cloud, the same actual densities may be given different values depending on the height of the sun. With partly obscured skies radiation values will obviously vary greatly depending on whether or not the sun itself is obscured.

For low cloud covers, when the sun is high in the sky, density is largely irrelevant as the sun is rarely obscured, but once the elevation decreases, higher cloud density can result in higher insolation values due to reflection from the sides of dense clouds. As cloud cover increases, increasing density has a greater effect on reducing insolation. With complete cloud cover, depending on cloud density, insolation can be reduced by anything from 20% to 90 % of the clear sky value (Haurwitz, 1945). Both these points have implications for the uses cloud cover in modelling. Firstly that density can have the opposite effect to that expected when the sun is low in the sky. These are precisely the times when ADMS is most sensitive to cloud cover, but it seems from the sensitivity study (Chapter Five) that the model takes this into account. Secondly in the UK high cloud covers predominate, so it is important to consider cloud density.

Haurwitz (1948) later used the same data to compare insolation with cloud type. Only hours when the sky was completely obscured were analysed and only seven cloud types occurred frequently enough (two high, two middle and three low cloud types) for analysis. Haurwitz devised a series of formulae for estimating insolation as a function of air mass and these seven cloud types. The ratio of insolation with a clear sky to insolation with each individual cloud type remains fairly constant regardless of solar elevation. Although cloud density can vary for each type of cloud, generally the high clouds cut out about 20% of insolation, middle cloud cut out 50 to 60% and low clouds about 65 to 80%. This work is perhaps more useful as cloud type and cloud height are recorded at UK synoptic stations whereas cloud density is not.

Lumb (1963) continued this work in the UK using data several years worth of data obtained from weather ships. The empirical formulae that had been used up until then to calculate average daily short wave radiation (Q) was clearly subject to at least 50 % error as it did not take into account cloud type and could not be applied to hourly averages.

$$Q = Q_0 (1-0.71\ C)$$

Where Q_0 is the amount of radiation received with a cloudless sky and C is the fraction of the sky covered by cloud. Lumb devised a series of 9 categories that took into account cloud amount and cloud type and using data from a weather ship derived a series of formulae that could be applied to calculate incoming radiation for each category.

$$Q = 135s\ (a + b\ s)$$

Where s is the mean of the sine of solar elevation for each hour and, a and b are constants determined by the 9 cloud categories. When these formulae where used to estimate daily radiation at another weather ship where actual measurements were taken, the estimates varied from the measured values by rarely more than 10 %.

Kasten and Czeplak (1980) based their study on 10 years worth of hourly data. They also found that the ratio of global radiation under cloudy skies to radiation under clear skies could increase at low cloud covers (1 oktas) and decrease slowly to 6 oktas, but drop rapidly to a value of 0.25 with complete cloud cover. The relationship was found to be constant regardless of season and solar elevation. Diffuse radiation behaves rather differently in that the ratio of radiation under cloudy skies to radiation under clear skies increases up to about 6 oktas cloud cover before falling rapidly to 0.8 at 8 oktas. Reflection from the sides of clouds has a stronger influence here.

Kasten and Czeplak (1980) also looked at the effect of cloud type on global radiation. They divided cloud type into 5 groups, Cirrus, Altus, Cumulus, Stratus and Nimbostratus and only used data when there was complete cloud cover - 8 oktas. Again regardless of cloud type, the ratio of global radiation under cloudy skies to radiation under clear skies is fairly constant irrespective of season or solar elevation. The ratios have the following values 0.61, 0.27, 0.25, 0.18, and 0.16 respectively for the following groups Cirrus, Altus, Cumulus, Stratus and Nimbostratus. Diffuse radiation shows a marked sensitivity to cloud type and solar elevation. Skies overcast with cirrus clouds can bring about a relative two-fold increase in diffuse radiation as the sun rises to 60° elevation. Long-wave radiation levels under overcast skies are more dependent on temperature than cloud type or solar elevation, although both have an indirect effect. These and the findings of Haurwitz (1945 and 1948) suggest that either cloud type or density should be considered and not just cloud amount, and that net radiation would be even more useful.

Bearing in mind that ADMS is particularly sensitive to cloud cover changes either side of 7 oktas and that 7 oktas is the most frequently occurring cloud cover, the actual relationship between land based cloud cover and radiation in the UK is considered important. This was examined at 4 of the 12 sites in the UK that record both parameters on an hourly basis. These are the two nearest sites to Northampton; Bracknell, which is 90km to the south and Hemsby, a coastal site 170km to the northeast. Eskdalemiur in southeast Scotland and Aviemore in northeast Scotland were also studied. The possibility of systematic differences between these sites due to local topography and geographical location was recognised. Three years worth of data were used and although data could have been standardised by solar elevation it was decided to just look at three different hours 0900Z, 1200Z and 1500Z for two months of the year. Where datasets are complete this provides 90 data points in each category. This is not large enough to determine relationships between radiation values and particular cloud types or cloud heights but does indicate the broad qualitative nature of the relationship.

8.2 Results - Cloud cover and Radiation

Hourly synoptic and global radiation data from the four sites mentioned above were downloaded from the BADC database. Data from these sites confirmed the findings previously reported that the most frequently occurring cloud cover is 7 oktas. The average and median global radiation value for each cloud cover value within each category was calculated. Data for Bracknell is presented in Figure 8.1 and Table 8.1 and 8.2. All four sites followed the same broadly similar polynomial relationship of decreasing radiation with increasing cloud cover. This is particularly distinct in summer when radiation levels are higher. There is always a sharp decline in radiation from 6 to 8 oktas, but less distinction is shown between 0 and 5 oktas. In winter the trend is less clear and radiation levels are often higher at mid cloud levels. This does appear to be reflected in how ADMS deals with the cloud cover data during the middle of the day.

The most notable feature of the data is the spread of radiation values, found at all times and all sites for cloud cover 7 oktas. This could simply be a feature of natural variability associated with the increased number of data points, but it does nevertheless highlight the problem of using solely cloud cover as a proxy for radiation.

One might expect a narrow range of data at 8 oktas. The sun is always obscured, so the radiation value is solely a function of cloud type and density and elevation. Even though the radiation data is integrated over the whole hour, for cloud covers between 3 and 7, it is possible that the radiation is still to some extent a function of the amount of time that the sun is actually obscured. Though one would expect lower radiation values at 7 than 6 oktas, this is not always the case. There are frequently individual hours for which the 7 oktas radiation value is a high as that obtained under skies with only 1 or 2 oktas cloud cover. Modelling these hours would inevitably have increased the risk of error.

In winter, it is the change between 5 and 6 oktas that ADMS-Urban model is most sensitive to. The same risks applies here. Although there are fewer data points, maximum radiation values associated with 6 oktas are often equal or even higher than values for lower cloud covers.

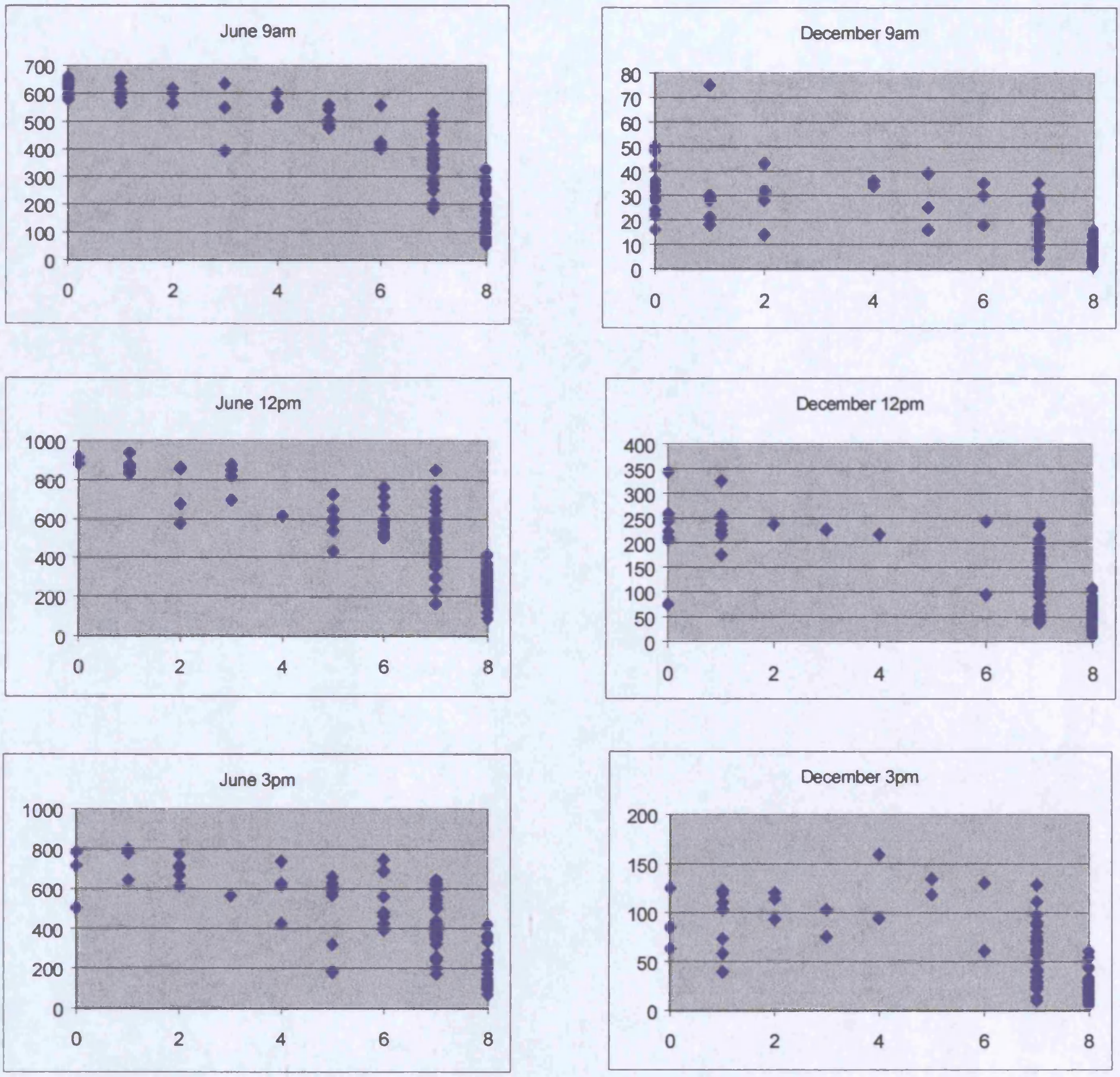


Figure 8.1 Cloud data for Bracknell, cloud cover (in oktas) against global radiation in wm^{-2}.

Cloud cover	Frequency	Ave. global radiation wm^{-2}	Minimum value	Maximum value
0	1	897	-	-
1	6	868	-	-
2	4	740	-	-
3	4	806	-	-
4	1	612	-	-
5	6	586	-	-
6	9	600	-	-
7	32	473	164	846
8	26	234	89	412

Table 8.1 Cloud data for Bracknell June, 1200Z, 1996-1999.

Cloud cover	Frequency	Ave. global radiation wm^{-2}	Minimum value	Maximum value
0	9	228	-	-
1	9	241	-	-
2	1	236	-	-
3	2	225	-	-
4	1	217	-	-
5	0	-	-	-
6	2	169	-	-
7	31	125	38	238
8	34	54	12	108

Figure 8.2 Cloud data for Bracknell December, 1200Z, 1996-1999.

The 1200Z June data for all sites was analysed further. Hours with 7 oktas cloud cover were selected to see if there was any relationship between the parameters used to characterise the cloud cover and global radiation. Within these datasets cumulus clouds predominate at the lowest cloud level, but they occur over the whole range of global radiation values. Where mid-levels of cloud are present these are dominated by altocumulus clouds and they generally cover a greater proportion of the sky than the low clouds.

These are small datasets (about 50 records for each site) and it is not possible to say which factors control radiation levels. At individual sites only the following show a significant association with global radiation at the 0.01 confidence level (Spearmann Rank); at Bracknell, cloud amount at the lowest level; Eskadalemiur, cloud type and cloud base at height at the second level. At Hemsby second level cloud base height is significant at the 0.05 confidence level. When the whole dataset of all four sites is combined the following parameters show a significant correlation with global radiation at the 0.01 probability level; cloud base height of the lowest cloud level, and cloud type and base height for the second layer.

The situation is broadly similar at 0090Z and 1500Z, although the pattern of cloud type is slightly different and when the sun is slightly lower in the sky, the third cloud level seems to play a greater part in determining radiation levels.

8.3 Conclusions

As the sensitivity of ADMS is limited to only certain times of day and certain cloud cover changes, error in cloud cover data has little effect overall, although it can be substantial for individual hours. From a regulatory point of view, this is likely to be most significant in summer when maximum model sensitivity coincides with maximum vehicular emissions.

From the relationships demonstrated here, it would appear that using cloud cover alone may not be a very reliable way of estimating the amount of radiation reaching the earth's surface. Although only a small dataset was examined, the results do suggest that, particularly at 7 oktas cloud cover, some additional information is required to reliably give an estimate of net radiation. However, a larger dataset would need to be analysed before it would be possible suggest ways of incorporating cloud type and cloud height data as part of the model input.

The error in model output that may result due a poor estimate of net radiation is harder to quantify. In a modelling exercise, not reported here, deliberate error was introduced. Twelve separate days in July and August 1999 were selected during which 7 oktas cloud cover predominated. ADMS was run using real meteorological data and again with cloud cover increased by 1 okta. The effect on predicted CO and the way ADMS determines boundary layer characteristics was examined. Generally increasing cloud cover leads to an increase model output, but reductions can occur under certain circumstances. It is possible for output to be reduced even during the hours when the model is most sensitive to increased cloud cover. Even at times of day and at cloud cover values at which the sensitivity study showed cloud cover to have little effect there can be quite large increases or decreases in predicted CO.

How ADMS uses the input data to calculate the boundary layer parameters may give some insight into why increasing cloud cover does not always produce consistent results. Friction velocity is increased during the day and decreased during the night by increasing cloud cover. Time of day seem to have little effect, but largest change, both positive and negative, occurs when wind speeds are low and at cloud cover 7. Increasing cloud cover only causes friction velocity to increase when the atmosphere is stable and boundary layer heights are less than 300m. Predictably the effect on heat flux of changing the cloud cover is much more pronounced. Surface heat flux is substantially reduced as increasing cloud cover decreases the amount of incoming solar radiation and not surprisingly the largest effects are seen at midday. At night heat flux is increased as cloud cover increases. When convective forces have the dominant influence, wind speed makes little difference, whereas during the night the increased mixing caused by higher wind speeds means that the effect cloud cover has on heat flux is dependent on wind speed.

These changes in friction velocity and heat flux in turn influence how the model determines the Monin Obukhov length and the boundary layer height. Although the sensitivity of surface heat flux to changing cloud cover is principally determined by time of day during daylight hours and by wind speed at night and that of friction velocity is determined by wind speed, the sensitivity of 1/LMO does not follow the same patterns. A change in cloud cover only alters the Monin Obukhov length significantly when wind speed are low, when the original cloud cover value is 7 and at certain timesof day; 0700Z, 0800Z, 1600Z and 1700Z. Monin Obukhov length is the dominant influence on how ADMS output responds to changing cloud cover.

Although one might expect boundary layer height to have an influence on dispersion of pollutants and hence the model output, unlike 1/LMO, there does not appear to be any relationship between the change in output caused by increasing cloud cover and the boundary layer height. Boundary layer height is largely unaffected by changing cloud cover at night. There is a general tendency for boundary layer height to be reduced, particularly in late afternoon, when it would normally be reaching a maximum value. However if wind speeds are high, even though the increased cloud cover would normally suppress convection, the boundary layer height can increase.

Boundary layer height does show more sensitivity than 1/LMO to changes in cloud cover over the complete range of cloud cover values. Reductions in height can occur even with relatively clear skies but this lowering of the boundary layer height is not translated into an increase in model output. During the day, the degree of change in boundary layer height seems less dependent on one particular variable than 1/LMO. It may be this that causes the unpredictable nature of the response to changing cloud cover.

The combination of using cloud cover data that are not directly related to specific global radiation values and the complex nature of the model response to cloud cover as an input suggest that there is great potential for error. If neither the reciprocal of the Monin Obukhov length nor the surface sensible heat flux are available, then at least steps should be taken increase the coverage of either net or incoming solar radiation monitoring across the country. This would provide a more direct and accurate path to the calculation of surface sensible heat flux.

CHAPTER NINE

THE NAME MODEL

9.1 Introduction

The NAME lagrangian dispersion model, developed by the UK Meteorological Office (Ryall *et al,* 1995), was used in this study to provide a background component of NO_x in Northampton. This is similar to the study proposed by Chatterton *et al.* (2000) to investigate the sources and dispersion of background pollutants in Norwich. Seika *et al.* (1998) also describe the use of an Ambient Background Model and its application for urban air quality management in London.

The inclusion of this particular exercise is important within the context of the whole study for two reasons. One is that it addresses the problem of providing a background value for urban air quality modelling and the other is because NAME uses as input, meteorological data from a Numerical Weather Prediction (NWP) model. This same meteorological data was also used to run ADMS.

Providing background data for urban dispersion modelling is problematic. The background should be from sources not explicitly modelled or if sources from the whole area are included in the inventory and modelled for, then background should be from a rural site. The developers of the ADMS model suggest two possible approaches; either to model the entire urban area (main roads and industrial sources), industrial sources outside the area and include the remaining emissions as areas sources, in which case background data for a nearby rural site should be included <u>or</u> to model part of the urban area in detail with large industrial sources from outside and in this case to add a background derived from the surrounding urban area as a function of wind direction (McHugh, 2000).

The problem with the first option is that there are relatively few rural monitoring sites, particularly for some pollutants. Of the 112 automatic monitoring sites in the UK that provide hourly data, there are only 19 that are classified as rural or remote. All of these monitor ozone, but oxides of nitrogen and sulphur dioxide are only monitored at seven, PM 10 at three and hydrocarbons at only one. None monitor for carbon monoxide. Rural monitoring sites are not evenly spread across the country; for example, there are none in the Midlands. For some pollutants the rural value will have little bearing on the urban value, but where long-range transport is an important factor knowledge of concentrations at distant sites is vital. Sulphur dioxide and nitrogen dioxide, even though they are used to estimate secondary particulate concentrations, are still only monitored at a few rural sites.

If area sources are used in the modelling and local emission inventories are not available, it is possible to obtain emission estimates from NETCEN on a 1km x 1km grid basis (URL.1). Estimates for seven pollutants are available from Part A industrial processes and also as a general figure that includes small industry, transport, domestic and 'other' sources. Data are currently available based on1998 estimates. However this is only available as an annual average in tonnes/year/km^2. These data do not show seasonal or daily emission patterns, which is a problem previously recognised in an ADMS-Urban evaluation study (Owen *et al.* 1999).

Modelling the second option is not necessarily any easier, although less demanding on computer time. Most modelling domains will not have monitored data from an urban background site. There are only a few NETCEN urban background monitoring sites across the UK and although many local authorities may be carrying out their own monitoring, they are unlikely to have background data when limited resources will be put into monitoring pollution 'hot spots'. There are 37 NETCEN sites classified as suburban or urban background and 9 of these are in London. Only 3 other cities have multiple sites where it would be possible to use different data depending on wind direction. Urban background concentration is likely to be more dependent on wind direction than rural values.

The nearest urban background NETCEN site to Northampton is Leamington Spa, a smaller town and situated approximately 48km to the east. In this exercise hourly NO_x

values from Leamington were added to ADMS output, as a substitute for background levels in Northampton and to provide a comparison with NAME.

Background concentration estimates are also available for 6 pollutants from NETCEN on a 1km grid square basis for the whole country. These are available as annual average for 1996 or as projected concentrations for the year 2004. Again they do not show seasonal or diurnal variation and may be interpolated from very limited data.

An obvious adaptation that could be used with either set of NETCEN data (emission or background) is to apply a model to the data that takes into account meteorological conditions and seasonal/diurnal variation.

The NAME (Nuclear Accident Response Model) model was developed to simulate the transport and dispersion of airborne radioactive pollutants over medium and long ranges. It is currently being adapted by the UK Meteorological Office to replace BOXURB in the routine forecasting of background NO_x concentrations. NAME is able to provide hourly NO_x data with a 15km grid square resolution for the whole UK using the NETCEN emission data. Traffic derived emissions are modified with weighting factors determined by traffic patterns on Cromwell Road, London. The meteorological data used to run the model comes from the UK Met Office NWP model and is available on an $11km^2$ resolution in analysis or forecast mode. (Manning, 1999: Ryall *et al.,* 1995)

NAME NO_x output was provided by the Meteorological Office for this study, as a traffic component and an industrial component for 1999, for a grid cell centred on Northampton. The ADMS-Urban model was run for a seven-week summer and a five-week winter period in 1999 for two sites in Northampton where monitored NOx and traffic data were available. NAME was used in conjunction with ADMS to provide background data. The ability of NAME to provide accurate background data has been reported elsewhere (Manning, 1999). In comparing NAME predictions with monitored data from ten UK cites, Manning found that it generally under-predicted, though correlation coefficients (r) varied from 0.44 to 0.69, the fraction of model output with a factor of two was very variable; from 26% for Belfast to 65% for London.

Details of the sites used in the Northampton modelling exercise are given in Appendix Two. Both sites are situated near busy arterial roads and although hourly average traffic flows are similar for both sites, at one (Kingsthorpe) there are more local traffic movements so morning and evening rush hour peaks are less pronounced. The Cliftonville site is nearer the town centre, but much less built up. At Cliftonville, the inlet to the Horiba monitor is placed on the roof of a two storey building (council offices) some distance back from the road and as such could effectively be called an urban background site.

The model runs for Kingsthorpe were carried out using meteorological data from Wittering and using data obtained from the mast on the Horiba mobile air pollution laboratory. Cliftonville model runs were carried out using only Wittering data. ADMS was also run with the same meteorology (NWP) that was used to run the NAME model.

In order to assess how well NAME provided a background component the ADMS model was run with two different scenarios; using roads in the immediate vicinity of the monitoring site and using a database containing 489 road sources from the whole of Northampton and including a section of the M1. In the latter case only the industrial component of NAME was added to the results. When output from part of the urban area modelled in ADMS and NAME traffic and industrial components from a 15km grid cell are combined, the risk of 'counting' some roads twice is acknowledged. The data present here suggests that this has not been a problem, but logically there must be a limit on the fraction of the urban area on which this approach can be taken.

Performance of the models was assessed on paired hourly predicted and observed values using the following statistical tests; bias, fractional bias (FB), normalised mean square error (NMSE), an index of agreement, correlation coefficient (r^2) and the percentage of data points with a factor of two (FAC2). Details of these tests are given in Appendix Nine.

9.2 Results

9.2.1 Kingsthorpe - Summer

Although both models under-estimate the NO_x levels, the NAME model in particular appears to be good at following the general trends shown by the pollutant. Where ADMS uses local meteorology, this highlights how with low wind speed the model can, on occasions, greatly over-estimate pollution levels (see Figure 9.1 and Tables 9.1 and 9.2).

Sources modelled in ADMS	Met. data used in ADMS	Background data used	Bias	FB	NMSE	Index of Agreement
Local roads	Wittering	NAME (Total)	-9.81	-0.28	0.62	0.73
	Horiba	NAME (Total)	2.03	0.05	1.29	0.64
	NWP	NAME (Total)	-18.48	-0.59	0.97	0.55
	Wittering	Leamington Spa	-6.71	-0.18	0.48	0.78
	Horiba	Leamington Spa	5.05	0.12	1.19	0.65
All roads	Wittering	NAME (Ind.)	-12.53	-0.37	0.82	0.69
	Horiba	NAME (Ind.)	12.28	0.26	1.66	0.58
	NWP	NAME (Ind.)	-23.82	-0.84	1.62	0.52
	Wittering	Leamington Spa	-1.10	-0.29	0.50	0.77
	Horiba	Leamington Spa	23.67	0.45	1.64	0.43

Table9.1 Performance measures Kingsthorpe - July and August 1999.

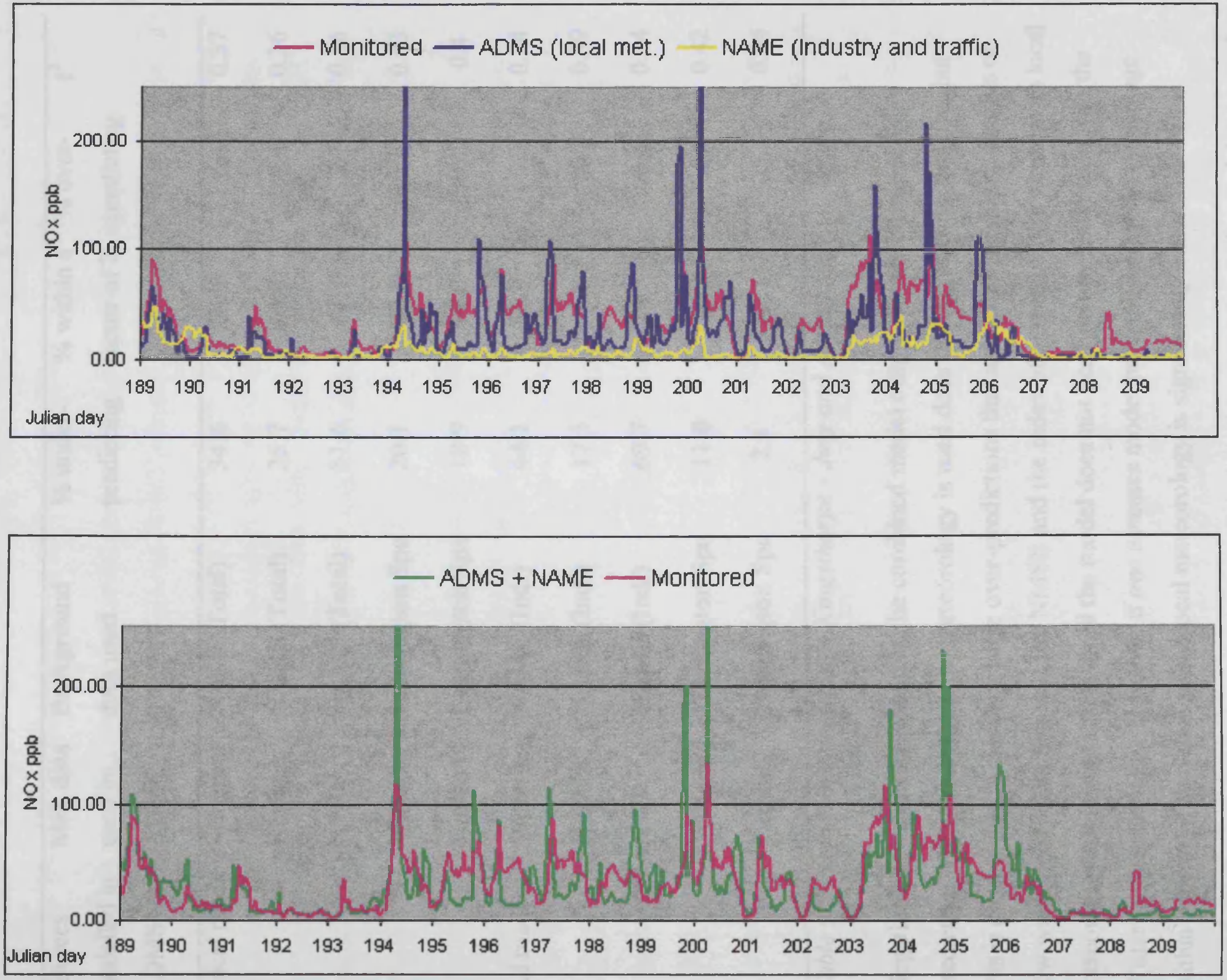


Figure 9.1 Monitored and modelled NO_x (ADMS and NAME), Kingsthorpe - summer 1999.

Sources modelled in ADMS	Met. data used in ADMS	Background data used	% under-predicting	% within a factor of 2	% over-predicting	r^2
Local roads	Wittering	NAME Total)	34.6	58.9	6.4	0.37
	Horiba	NAME (Total)	25.7	64.9	9.4	0.36
	NWP	NAME (Total)	51.6	45.7	2.7	0.36
	Wittering	Leamington Spa	20.1	72.8	7.1	0.43
	Horiba	Leamington Spa	10.9	79.0	10.2	0.4
All roads	Wittering	NAME (Ind.)	44.1	50.6	5.2	0.34
	Horiba	NAME (Ind.)	17.5	66.2	16.3	0.39
	NWP	NAME (Ind.)	69.7	29.5	0.8	0.34
	Wittering	Leamington Spa	11.8	78.2	10.0	0.42
	Horiba	Leamington Spa	2.1	72.7	25.2	0.44

Table 9.2 Performance measures Kingsthorpe - July and August 1999.

Negative bias values indicate that the combined model output is generally under-predicting. Only when the local meteorology is used does the bias become positive, but this is entirely due to the very large over-predictions that are occurring during periods of low wind speed. The fact that the NMSE and the index of agreement are worse with local meteorology suggested that overall the model does not perform any better than with the Wittering meteorology. However if one assesses model performance by the percentage within a factor of two then using local meteorology is significantly better.

Using the NWP meteorology, by all measures of performance shows the model to severely under-predict, but the r^2 value varies little and this suggests that the data still allow ADMS to pick up the general trends in NO_x. If one looks at correlation coefficients on just the ADMS output using local road sources only and NWP, local and Wittering meteorology, the values are 0.34, 0.32 and 0.31 respectively. The NAME output on its own only correlates with the monitored data with an r^2 value of 0.16.

There is little to choose in terms of model performance between the two modelling scenarios; local road sources and the total NAME output or using roads from the whole of Northampton and only adding the industrial component of NAME. Modelling using the 489 road sources with the local meteorology actually produces reasonably good results with an r^2 value of 0.38 and 66% of the data with a factor of 2, but high FB and NMSE values suggest there is a tendency to over-predict and a large amount of variability in the results.

Generally, if one considers the percentage of results within a factor of 2, using local meteorology it is seems best to model all the road sources, but if one is using data from a more distant synoptic site or the NWP data, a better performance is obtained by modelling only the local roads and adding the total NAME component as background. The errors produced by using in-exact meteorology are amplified, if ADMS is used to model the whole urban area.

The addition of a monitored background value (Leamington Spa) even it is not from the modelling domain produces better results than using a modelled background. However the addition of a monitored urban background value is probably not appropriate if one models the whole urban area, addition of a rural value would then be more appropriate.

9.2.2 Kingsthorpe - Winter

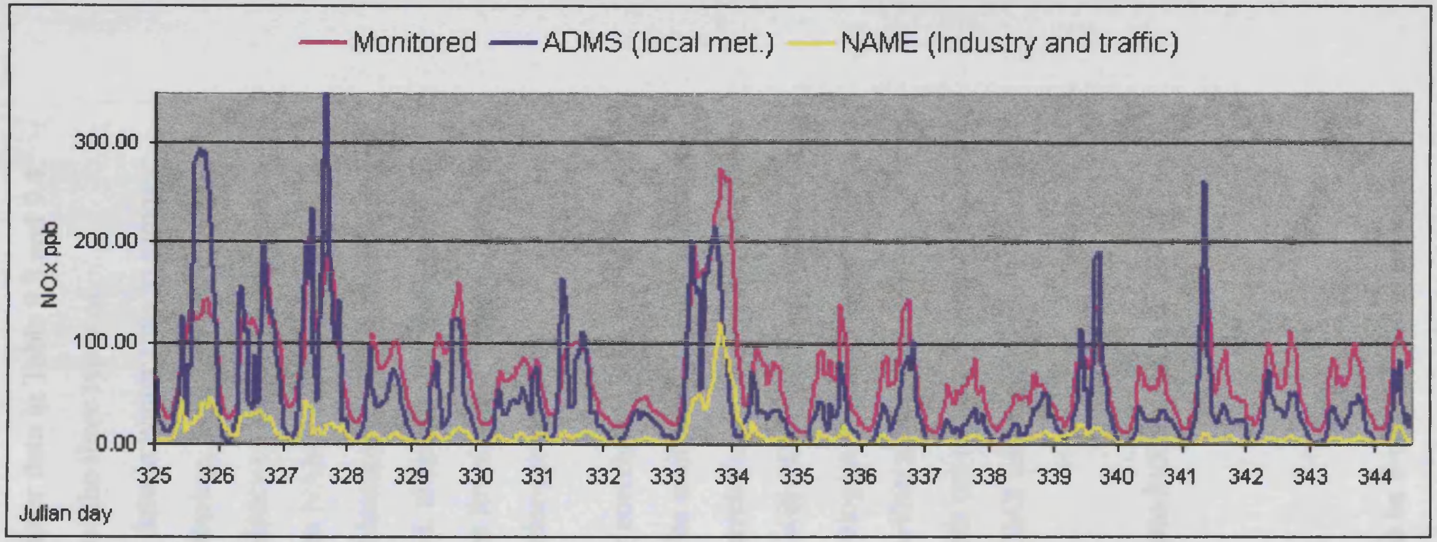
The models are both able to replicate the trends shown by the monitored data and during the latter part of the period the combined output is able to give a very good prediction. NO_x levels are generally fairly low and do not change much from day to day during the first week of December. The prevailing weather conditions suggest neutral stability. The wind was constantly from the southwest and speeds were generally quite high. Daytime temperatures were variable from day to day. The NWP model and ADMS using local meteorology both calculate the reciprocal of the Monin Obukhov length to be small and positive. NWP estimates the daytime boundary layer height to be about 1000m.

Sources modelled in ADMS	Met.data used in ADMS	Background data used	Bias	FB	NMSE	Index of Agreement
Local roads	Wittering	NAME (Total)	-26.91	-0.48	0.48	0.78
	Horiba	NAME (Total)	-1.34	-0.02	0.64	0.79
	NWP	NAME (Total)	-37.48	-0.74	1.11	0.69
	Wittering	Leamington Spa	-11.67	-0.18	0.20	0.89
	Horiba	Leamington Spa	14.06	0.18	0.69	0.77
All roads	Wittering	NAME (Ind.)	-27.92	-0.50	0.61	0.72
	Horiba	NAME (Ind.)	4.84	0.07	0.90	0.73
	NWP	NAME (Ind.)	-45.49	-0.97	1.91	0.57

Table 9.3 Performance measures Kingsthorpe - November and December 1999.

Sources modelled in ADMS	Met. data used in ADMS	Background data used	% under-predicting	% within a factor of 2	% over-predicting	r^2
Local roads	Wittering	NAME (Total)	44.5	55.2	0.3	0.64
	Horiba	NAME (Total)	27.6	67.1	5.4	0.62
	NWP	NAME (Total)	75.9	23.9	0.2	0.52
	Wittering	Leamington Spa	14.5	84.6	0.9	0.71
	Horiba	Leamington Spa	6.0	86.7	7.3	0.67
All roads	Wittering	NAME (Ind.)	47.5	52.2	0.3	0.51
	Horiba	NAME (Ind.)	26.0	66.7	7.3	0.56
	NWP	NAME (Ind.)	90.2	9.7	0.1	0.45

Table 9.4 Performance measures Kingsthorpe - November and December 1999.



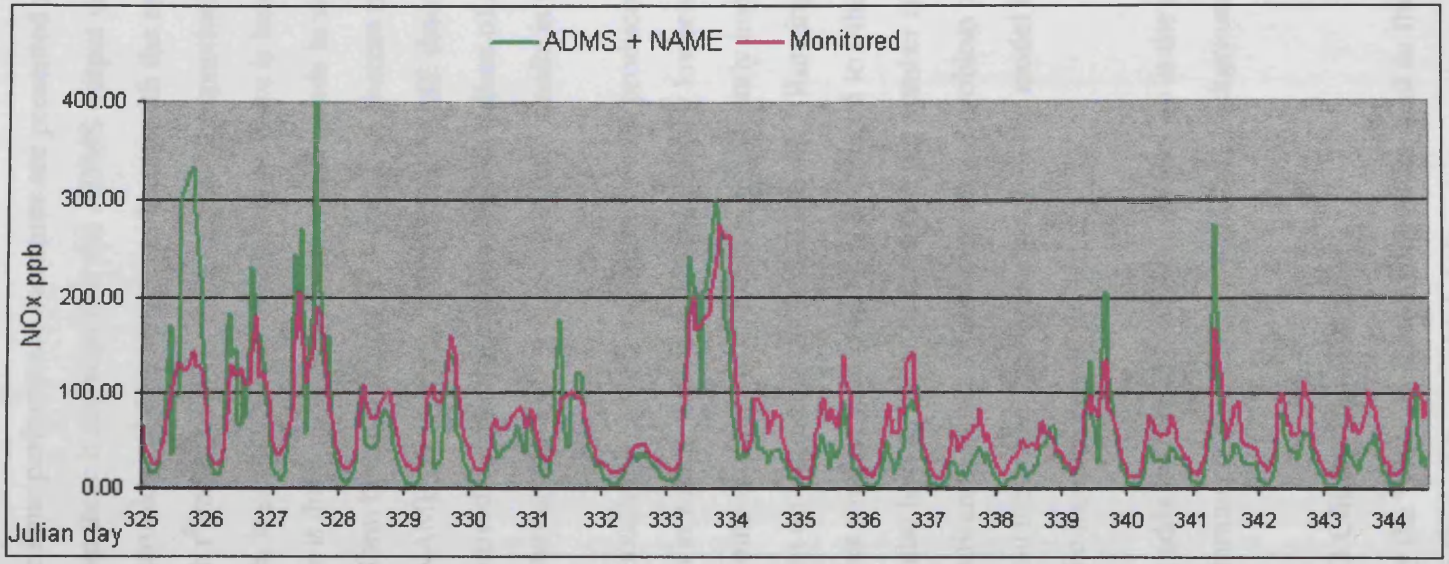


Figure 9.2 Monitored and modelled NO_x (ADMS and NAME), Kingsthorpe - winter 1999.

The same performance measures are presented for the winter data in Table 9.3 and 9.4. From these it can seen that the ADMS output using any of the three types of meteorological data correlates better with the monitored data in winter than in summer; with r^2 values of 0.48, 0.56, and 0.44 respectively for Wittering, local and NWP. Local data, in particular, perform better as there is less of a tendency to severely over-predict. This is due to generally higher wind speeds in winter. The NAME model also seems to perform better in winter ($r^2 = 0.42$). A feature noted by Manning (1999) in his assessment of NAME performance. When the NAME data are added, although this also results in improved correlation and gives good indices of agreement for all three datasets, using the Wittering and NWP meteorology still results in model under-prediction.

Although the NAME model on its own produces a better correlation with the monitored data in winter, it does produce relatively lower values. The main outcome of combining the output of the two models is significantly improved correlation in winter, but it has less effect on increasing the FAC2 statistic. Running ADMS with NWP meteorology produces higher average predicted NO_x compared to other meteorologies, but as observed values are also higher, for both scenarios the number of results within a factor of two is significantly lower. This suggests the problem may lie with the meteorological data used to run the NAME model rather than the model itself. ADMS seems to be more sensitive to the meteorology in winter.

The addition of real background data, as in the summer, produces better overall performance than adding a modelled background.

9.2.3 Cliftonville - Summer

Only the Wittering meteorology was used so there tend to be fewer occasions on which ADMS predicts very high NO_x concentrations. Both models, but especially NAME, are reasonably good at picking up the trends. There are occasions when ADMS completely fails to predict some of the higher monitored NO_x concentrations. However the combined output appears to give satisfactory results (see Table 9.5. and 9.6).

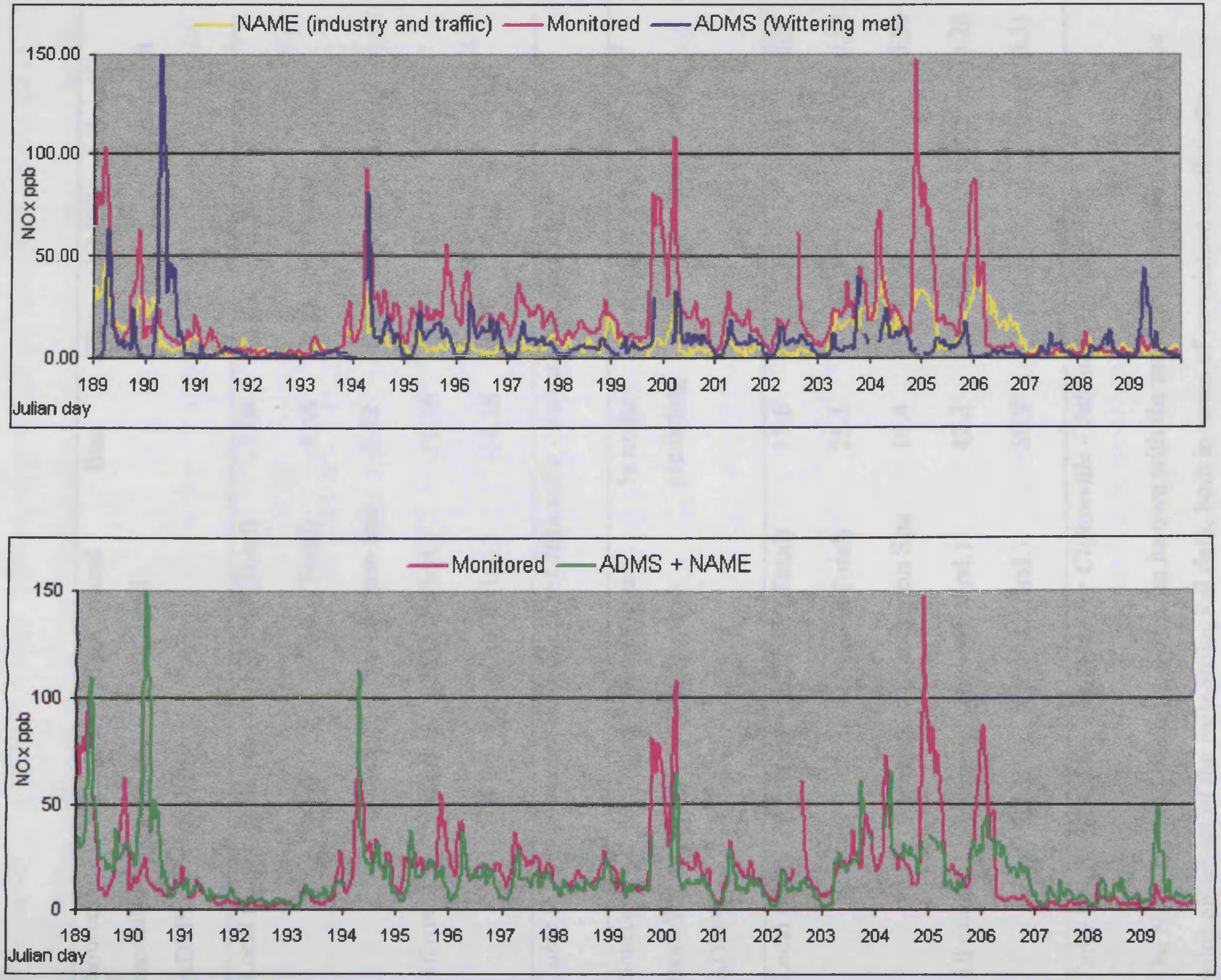


Figure 9.3 Monitored and modelled NO_x (ADMS and NAME), Cliftonville - summer 1999.

Sources modelled in ADMS	Met. data used in ADMS	Background data used	Bias	FB	NMSE	Index of Agreement
Local roads	Wittering	NAME (Total)	-2.08	-0.09	0.79	0.71
	NWP	NAME (Total)	-5.66	-0.25	0.86	0.68
	Wittering	Leamington Spa	0.82	0.03	0.69	0.71
All roads	Wittering	NAME (Ind.)	-12.36	-0.65	1.67	0.52
	NWP	NAME (Ind.)	-14.28	-0.80	2.16	0.46

Table 9.5 Performance measures for Cliftonville - July and August 1999.

Sources modelled in ADMS	Met. data used in ADMS	Background data used	% under-predicting	% within a factor of 2	% over-predicting	r^2
Local roads	Wittering	NAME (Total)	15.6	67.3	17.2	0.31
	NWP	NAME (Total)	25.1	58.7	16.2	0.32
	Wittering	Leamington Spa	10.4	65.5	24.1	0.30
All roads	Wittering	NAME (Ind.)	42.3	51.5	6.2	0.28
	NWP	NAME (Ind.)	50.9	44.1	5.0	0.31

Table.9.6 Performance measures for Cliftonville - July and August 1999.

The NAME model performs better on its own with the monitored data than ADMS does using either set of the meteorological data, both in terms of correlation coefficients (NAME $r^2 = 0.39$, ADMS (Wittering met.) $r^2 = 0.10$) and FAC2 (NAME = 39.5%, ADMS = 32.8%). The nature of the Cliftonville site means that less of the monitored NO_x can be attributed to road sources in the immediate vicinity, so these results are not entirely unexpected. Because NAME performs so well the combined output from both models produces reasonably good results. Similarly even though the individual performance of ADMS using NWP data is poor, combined results are reasonable and much better than at Kingsthorpe.

Both sets of meteorological data produce a lower average predicted value when modelling all road sources than nearby roads only. This may be because the .upl file for 489 road sources contains traffic flows from the SATURN model on the A45 and the file for local roads uses data from actual traffic counts, the latter are nearly 50 % higher. However, although the A45 is a busy road, it is to the east of the monitoring site and winds are rarely from this direction. Adding only the industrial component of NAME is not sufficient to bring the predicted values anywhere near the monitored levels. Figure 9.3 shows model output for local roads only and the total NAME component. The combined model outputs under-predict, have a high NMSE and low indices of agreement.

For the Cliftonville site using the Leamington Spa data as background does slightly improve model performance. Although bias, fractional bias and NMSE are improved and the index of agreement is identical, the FAC2 statistic is marginally worse. This latter point is due to a slightly increased tendency to over-predict and may be due to the fact that Leamington data are inappropriate for use at Cliftonville, a site that could itself be described as 'urban background'.

9.2.4 Cliftonville - Winter

Apart from three days during November when high NO_x concentrations were recorded, there was less day to day variability during this period than there was during the summer. Both models found it easier to replicate the monitored pattern. Cliftonville seems to be less subject to seasonal variation in performance than Kingsthorpe. However monitored values are slightly higher and both models are more likely to under-predict (see Figure 9.4 and Table 9.7 and 9.8).

Although the index of agreement and correlation coefficients are unaffected the bias, fraction bias and NMSE statistics are far worse in winter regardless of the meteorological data used. It is only when one considers the FAC2 values that it is obvious that the Wittering data are performing slightly better. This was also seen with the Kingsthorpe data. Because both models are relatively poorer at predicting NO_x at Cliftonville during winter using Leamington data as background produces the best results. It seems likely that general meteorological conditions in winter result in even less of the NO_x present at Cliftonville being attributable to local sources than in summer. The addition of what is effectively a 'town centre' background brings modelled NO_x closer to the real values.

Sources modelled in ADMS	Met. data used in ADMS	Background data used	Bias	FB	NMSE	Index of Agreement
Local roads	Wittering	NAME (Total)	-12.55	-0.45	1.37	0.70
	NWP	NAME (Total)	-17.44	-0.66	1.83	0.65
	Wittering	Leamington Spa	2.55	0.09	0.6	0.84
All roads	Wittering	NAME (Ind.)	-15.0	-0.56	1.95	0.43
	NWP	NAME (Ind.)	-22.09	-0.93	3.31	0.37

Table 9.7 Performance measures for Cliftonville - November and December 1999.

Sources modelled in ADMS	Met. data used in ADMS	Background data used	% under-predicting	% within a factor of 2	% over-predicting	r^2
Local roads	Wittering	NAME (Total)	28.7	66.1	5.2	0.44
	NWP	NAME (Total)	50.1	45.6	4.3	0.62
	Wittering	Leamington Spa	2.8	77.8	19.4	0.53
All roads	Wittering	NAME (Ind.)	36.2	58.4	5.3	0.31
	NWP	NAME (Ind.)	67.4	31.3	1.3	0.53

Table 9.8 Performance measures for Cliftonville - November and December 1999.

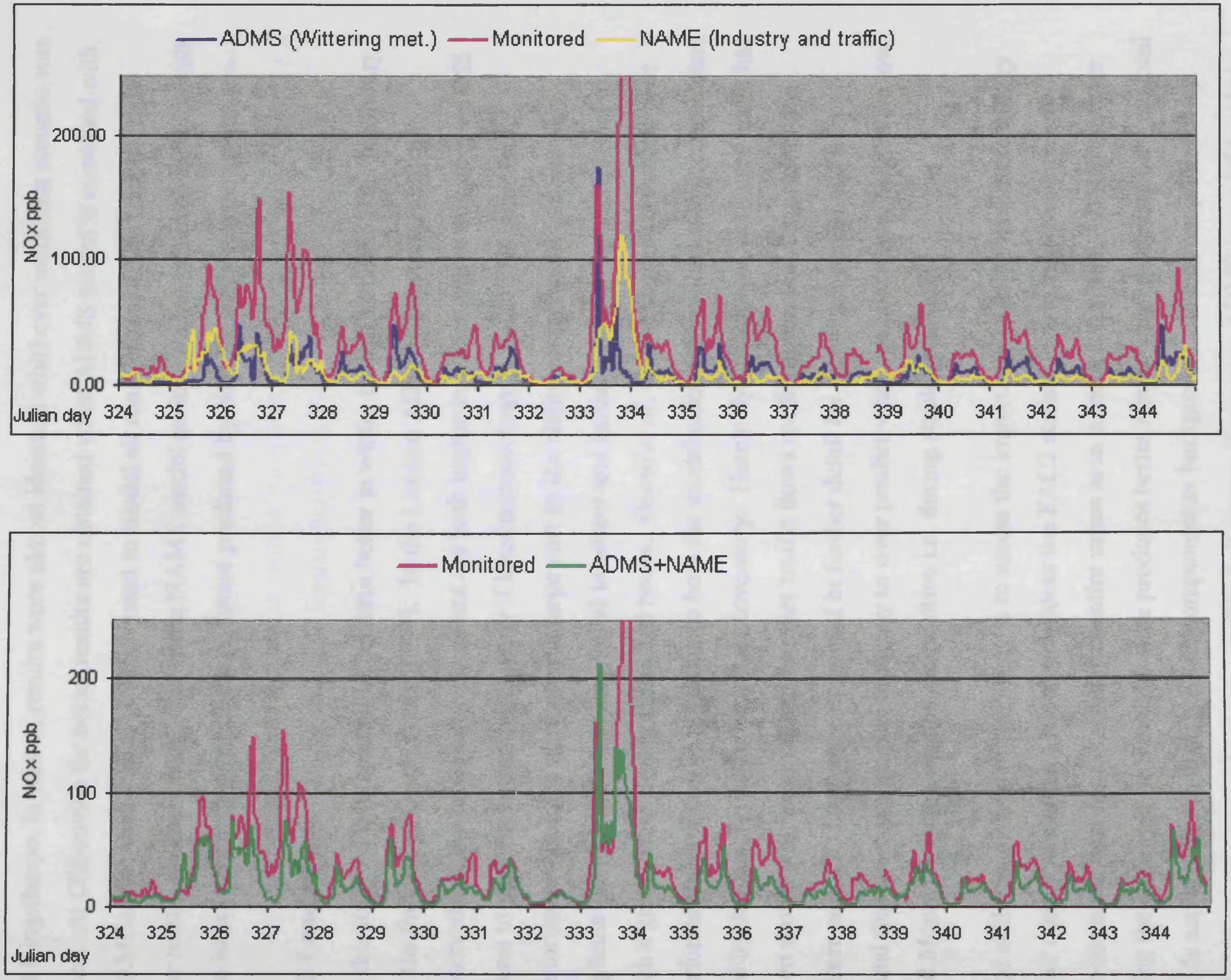


Figure 9.4 Monitored and modelled NO_x (ADMS and NAME), Cliftonville - winter 1999.

At Kingsthorpe, in winter, results were almost identical whichever modelling scenario was used. At Cliftonville far better results are obtained when ADMS output is combined with NAME rather than when ADMS is used to model all road. There doesn't appear to be any reason for this except that as the NAME model on its own out-performs ADMS, with the larger the component of the pollutant predicted by NAME, the better the results are.

9.3 Conclusions

In this study ADMS seemed to perform better in winter than in summer. This is contrary to the findings of Owen *et al.* (1999). In the London ADMS evaluation study, the observed and the modelled NO_x values are both higher in winter, but in this study ADMS seems to perform better, particularly if local meteorology is used. This improved performance could be attributed in large part to the ability of the model to accurately replicate the higher NO_x values found in winter and because it does not over predict to such a great extent during individual hours. Owen *et al.* (1999) suggested that the poor performance in winter may be due to how the model calculates various stability parameters and by deficiencies in the emissions inventory. Here only line sources were used and it has been shown with local monitoring that traffic flows are fairly constant for summer and winter periods. Data were examined in further detail for the winter period and it was found that there is a greater tendency to over predict when boundary layer heights are low and Monin Obukhov lengths are positive i.e. during stable conditions.

One of the main aims of this study is to assess the importance of using local meteorology in air quality modelling. If one considers the FAC2 statistic, using local meteorology produces the best results when the entire urban area is modelled within ADMS, however using the Wittering or the NWP data produces better results when modelling only the local roads and adding the total NAME component as background. The errors produced by using in-exact meteorology are amplified, it seems, if one uses ADMS to model to whole urban area. At Cliftonville, where no local data are available, it seems to make very little difference if the Wittering or the NWP data is used.

Using a model such as ADMS in a predictive capacity, if short term hourly forecasts were required, would by necessity involve the use of a meteorological model. It would appear, from the examples used here, that better performance is achieved by using ADMS to model the local area and NAME to provide a background value that includes other traffic and industrial sources from the immediate surroundings. Although background data from Leamington generally produces better results, such data would not always be available and NAME does provide a more than adequate substitute.

CHAPTER TEN

DISCUSSION AND CONCLUSIONS

10.1 Discussion

With models becoming increasingly used in the management of local air quality there is a danger that policy decisions will be made based on their predictions without a true understanding of the risks involved. Within the context of air quality management, Beychok (1998) notes that many model users are unaware of the assumptions and constraints that are inherent in them and how the propagation of small errors can lead to large variations in the model prediction. He refers specifically to some of the problems inherent in Gaussian plume modelling, particularly the assumption that wind speed, wind direction, and turbulence are constant between source and receptor. There is a mistaken idea that precision equates with accuracy. Errors in input data can be translated into larger errors in model output and this is often not fully appreciated.

It is frequently accepted in modelling exercises that no local meteorological data will be available and it is even suggested in some circumstances that the use of routine collected data is preferable. For example for regulatory modelling in Denmark, the Danish Environmental Protection Agency recommends the use of a specific set of data from Karstrup airport (Olesen, 1995a). It is recognised however that while this is administratively easier it is at the expense of accuracy. Even in the USA, the EPA when regulating emission sources, place no requirement for site-specific data and prefer the use of routinely collected data. This is seen as a way of avoiding disputes that could arise over questions of how representative the measurements are, the height and response of the sensors and the manner of electronically processing data (Turner 1997). This situation could to some be resolved by setting in place sufficient quality control.

In the UK it is also widely accepted that site specific data are not available. However the Royal Meteorological Society in their guidance notes on the use of models do now suggest a more pro-active approach is necessary especially where long term studies are likely and where dispersion will be affected by local terrain (Royal Meteorological Society, 1995). Compared to some other parts of Europe the UK is particularly poorly served by urban synoptic weather stations.

An aim of the EU COST 715 action - Meteorology applied to Urban Air Pollution Problems, was to advise modellers on the most appropriate meteorological data for use in urban dispersion modelling (URL4). The COST 715 programme made a number of recommendations; one of which was that all methods of data preparation for modelling would benefit from more field observations. They also recommended that local meteorological measurements be taken, co-located with air pollution monitors principally in order that the meteorological data could be used to interpret pollutant behaviour. The reason that more urban monitoring is not carried out was seen as a problem with the siting of instruments and their different requirements rather than cost.

The lack of urban meteorological monitoring often means that modelling studies have to rely on data gathered at a remote site, the problems that this can create are recognised. Ziomass *et al.* (1995) used regional meteorological variables in a modelling study carried out in Athens as no local data was available, but they implied that the variables would then represent conditions over the whole Athens area and the same methodology could be used. They acknowledged that by using regional rather than local data would result in some loss of accuracy. Similarly Nambeo and Colls (1996) used data from Nottinghamshire (40 km away) to model in Leicester and recognised that even this distance would decrease the accuracy of the prediction.

This study has attempted to demonstrate the variation that exists in the meteorological variables used to run ADMS-Urban over three spatial scales. Within the urban environment, with a 35 km range and between three synoptic stations sited between 50 and 70 kilometres from Northampton. Three synoptic sites within Greater London were also compared. No attempt was made to characterise these differences in meteorology in terms of topography or the urban surface. Differences obviously do exist between sites but for the most part they are within the measurement resolution. For example wind direction is recorded with an instrument resolution of +/- 10° and on nearly 50% of occasions neighbouring synoptic stations either record identical wind direction or a 10-degree difference. Similarly cloud cover has a resolution of +/- 1 okta, but cloud cover is identical or varies by 1 okta approximately 60% of the time. Very large differences between sites for any parameter rarely occur, but it is systematic error or variation at certain times of day that can cause significant error in model output.

The sensitivity study showed ADMS to be more sensitive at certain times of day and that this was also dependent on time of year. Although diurnal trends can be detected in certain meteorological parameters, for example wind speeds tend to be lower in late afternoon, the differences between synoptic sites showed no such trends. One finding that does have significance for model outcome is that wind direction and cloud cover show less consistency when wind speeds were low. When pollution levels are likely to be highest as a result of low wind speed there is also the potential for greatest error in the meteorological data, if it is obtained from a distant site.

There has only been one other study recently reported in the UK that investigated model output in relation to the source of meteorological data. AEOLIUS, a street canyon model, was used with different sets of input data. The performance of the model was found to be largely dependent on traffic flow and wind direction. In the AEOLIUS study the street level wind that was used to calculate carbon monoxide concentrations was derived from roof level wind by several different methods so no direct comparison can be made with this study. However Manning *et al.* (2000) did find that roof top wind speed was considerably reduced compared to Manchester airport wind speed. Comparison of wind direction showed broad agreement. These finding are in accordance with those found in this study.

The Royal Meteorological Society (1995) suggest that model sensitivity should form part of any assessment, as part of a process that aids the communication of results and is especially useful where there is a choice of input variables. It is also a means of verifying certain claims about how well the model performs (a means of auditing the model output). Moussiopoulos et al (1999) suggest that sensitivity tests should be 'performed for as many situations as possible in order to derive a minimum set of accuracy, completeness and transparency requirements to the input data for the policy application in question'. A sensitivity study was used here to determine the significance of possible variation in meteorological parameters and to determine the course of further investigations. It did highlight some areas where regulatory model users would need to pay particular attention to the quality of the model output.

In summer ADMS seems to be most sensitive to cloud cover at the times of day when traffic related emissions are highest. Under certain conditions it appears that the model could completely fail to predict the peak in pollution levels that are associated with the morning rush hour. The change between 7 and 8 oktas cloud cover has a significant effect on model output in summer. In winter this change is seen at midday and occurs between 6 and 7 oktas. As 7 oktas is the most frequently occurring cloud amount in the UK, slight variation, real or otherwise, between monitoring sites can result in very different model output.

ADMS uses the cloud cover data to estimate incoming solar radiation and through a series of equations to derive a measure of atmospheric stability, this is then used to determine the dispersion capabilities of the lower atmosphere. Comparison between cloud cover and global radiation recorded at four sites in the UK suggest that cloud cover alone may be a very poor estimate of incoming radiation. Seven oktas cloud cover can be associated with a wide range of radiation values if cloud base height and cloud type are not taken into account. A modelling exercise (not reported here) in which deliberate error was introduced in the cloud cover data showed little effect overall, however this did mask individual hours when altering cloud cover by one okta could produce extreme results. Sensitivity to cloud cover is critical at certain times and certain cloud amounts. From a regulatory point of view cloud cover is most significant in summer when maximum model sensitivity coincides with maximum traffic emission. A more reliable estimate of incoming radiation to remove this potential source of error may be preferable to one that relies on visual interpretation and as such is subject to human error. Pyranometers are relatively cheap and easy to install and would give a far more accurate and local measure of global or diffuse radiation. ADMS-Urban does allow for use of surface heat flux if these data are available, however they are not readily available to routine model users. The use of satellite remote sensing has also been suggested by Mensink and de Ridder (2000). Although satellite data may have good spatial resolution (in some cases down to 1km) and provide daily coverage it still may not have sufficient temporal resolution to model short term averages.

Gaussian models, such as ADMS, are most frequently used for regulatory purposes, as they only require modest computer capacity. However they do show certain systematic errors that the more demanding Lagrangian models do not. They tend to over-estimate in low wind conditions and with parallel winds. In parallel or nearly parallel wind directions they are most sensitive to the assumption of steady state homogenous wind flow (Oettl *et al.*, 2001). With parallel wind the background contribution from the road will also be substantially increased. The developers of the ADMS model acknowledge that modelling traffic emissions in low wind conditions could lead to over-prediction and that no facility for varying meteorological parameters over a large urban area exists in the model program (Carruthers *et al.,* 2000). Although it may have been useful to study the direct effect of general synoptic weather or even local weather on pollution levels, this was not carried out. The effect of using different sets of meteorological data at varying distances from the modelling domain on model output was investigated.

Using various performance measures the model does generally seem to perform best when using local meteorological data. However data collected elsewhere from within the urban area does not necessarily perform any better than data from the nearest synoptic site

The modelling exercise in London also showed that the closest match with monitored air quality data was found with a local background value and a modelled component using local meteorological data. It would be unfair to claim that this was a measure of model performance when the model output only forms a small part of the predicted pollutant concentration, but it did appear that ADMS was able to better follow hourly trends in concentration with local meteorology.

Introducing error to real meteorological data showed that even small changes in the data can produce very large changes in model output, especially if cloud cover and wind speed work together to dampen the dispersion process. The changes were particularly large at certain times of day. Although some uncertainty is inevitable in meteorological data, it would seem wise to strive for the most appropriate data set possible in order to minimise the risk of error. It was a recommendation of the COST 715 action that national weather services should work with model users to advise on the setting up of local meteorological monitoring facilities (URL4), this would clearly seem beneficial.

It is perhaps worth putting the possible errors in meteorological data into context. The potential errors relating to model formulation and the due to the stochastic nature of the dispersion process have been discussed earlier (Chapter 2). Model users need to be aware of the error or uncertainty attached to all model inputs. Weil *et al.* (1992) report that several studies had found model performance to be site dependent and suggested that model evaluation studies should include a variety of sites and dispersion climatologies. Hanna (1993) similarly reports how the same model can overpredict by 40% at one site and underpredict by the same amount at another. Grimmond and Oke (1999) have commented on the difficulties of estimating surface roughness with the urban environment. Hall *et al.*, (1999) have demonstrated difference in model output using meteorological data from the same site, but supplied by different agencies.

The uncertainty in emission data has been noted as being highly significant. Moussiopoulos *et al.* (1999) found that although uncertainty in meteorological data played a critical role in certain situations, minimising emission uncertainties is far more important for increasing confidence in model output when used for air quality assessments. They review studies that had investigated uncertainty in meteorological data, initial and boundary concentrations of pollutants and emissions data, and generally found the later two to possess a higher degree of uncertainty. Gram (1996) had previously stressed the importance of accurately defining the traffic characteristics and how important this was for modelling emissions. The sensitivity to daily variation in driving speed, road gradient and traffic composition, in terms of light and heavy vehicles was highlighted. Even sophisticated models such as ADMS only allow one speed to be entered per stretch of road, they do not take into account road gradient and assume a constant proportion of heavy to light vehicles throughout the day (CERC, 1999). Kuhlwein and Friedrich (1999) have also commented on emission uncertainty in the context of the temporal resolution of the emission data. Pielke and Uliasz (1998) referring to the use of meteorological models for providing input to dispersion models, question the point of using advanced meteorological models when there is still such huge uncertainty in emission data.

However in this study the variation in traffic flows was found to be very small and where traffic emissions dominate the emissions inventory, as in Northampton and many other Western European towns and cities, the uncertainty in emissions data may be more important in determining the background than rather roadside concentrations. In this situation the influence of meteorological inputs becomes more crucial.

10.2 Conclusions

This study has sought to show how dependent the successful modelling of traffic derived urban air pollution is on the quality of the meteorological data used in the modelling process. The significance of this issue is based on many different features. Models are being increasing used in air quality management programmes and as Russell states (1988) the cost implications of making a wrong decision based on model predictions are immense. There is also an increasing trend, with improved computing capabilities, towards the use of long term sequential data (URL2) and therefore any error in the data due to location that might in other circumstances be smoothed out by the use of statistical data will be more manifest. In addition, it is not always appropriate to use statistical data as there are many situations in which sequential data are required, for example to gains insights into the development of pollution episodes or in traffic management schemes.

Some understanding of the inaccuracies in dispersion modelling has led to the adoption of worst case or conservative assumptions in the selection of input parameters such that predictions tend to over-estimate. This results in a cautious approach to regulatory planning and licensing (Vawda, 1999). However as this study demonstrates, even a cautious approach without a full understanding of how the model performs, could give an erroneous outcome. Regulatory model users in particular are concerned with exeedences of the strategy levels set by government, these are most likely to occur with occasional peak events which could well be missed by a model using unrepresentative or statistical data.

This study has shown by means of a sensitivity study which parameters and over what range these have greatest effect on modelling outcome and there are a number of key points.

It has been shown that at certain times of day the model is extremely sensitive to small changes in wind speed and more importantly cloud cover, which as mentioned earlier is one of the parameters for which it is most difficult to obtain reliable information. In summer this sensitivity is shown precisely at the times of days when pollution levels are likely to be highest.

This is important as it gives the greatest potential for error, as errors in meteorological input tend to be greatest when emissions are also highest.

In the case of wind speed and direction, it is the variation in the range of 2-3ms^{-1} wind speed or 20 to 30 degrees wind direction that occur with sufficient frequency to cause problems in modelling. The fact that greatest variation between meteorological sites also occurs at low winds speeds, i.e. the situation in which pollution is likely to be highest, that again gives greatest potential for error. However this variation does not appear to show a trend with regard to time of day, or year, otherwise the problem would be further exacerbated.

This suggests that the Royal Meteorological Society (1995) recommendation that calibration between distant and local could be applied is unworkable, as it is the small individual changes in meteorological parameters that have critical importance on model output.

ADMS is most sensitive to cloud changes either side of 7 oktas, the most frequently occurring cloud cover in the UK. The work has also shown the wide range of global radiation levels that can occur at this cloud cover level, undoubtedly due to variation in cloud base height and cloud type. This analysis was not extended further, but it does suggest that cloud cover may be an unreliable input and that including some additional information such as cloud type or height may well refine the pre-processing calculations.

The above conclusions were reinforced by the performance of the model in Barnes. Even though the Barnes data set may not have been a very good measure of model performance due to the high background component. The success of the modelling was clearly improved with the use of local data, despite the three synoptic sites in or around London being more consistent than the three rural sites that surround Northampton.

This sensitivity of the model to small variations in meteorological inputs on certain occasions is particularly crucial as peak pollution episodes are more likely at those times.

This had not been properly identified in earlier research and this study provides model users with important additional understanding of how the air pollution model is likely to behave.

It is obviously important that urban model users are aware of the potential problems that can exist and might consider that if investing in a model, it is then worth ensuring good quality data to run it and should consider providing their own meteorological station. Due to the highly variable nature of the urban environment, the siting of this within the urban area is not straightforward and would need to the subject of a much consideration, as this is an area still needing further research.

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Web sites

URL1 http://www.aeat.co.uk/netcen/airqual

URL2 http://www.dmu.dk/AtmosphericEnvironment/cost_710.htm

URL3 http://www.mi.uni-hamburg.de/techn...ologie/cost/cost_615/finalrep.html

URL4 http://www.dmu.dk/AtmosphericEnvironment/cost715.htm

URL5 http:// www.badc.rl.ac.uk/data/surface/ukmo_guide.html

URL6 http://www.hammersmithbridge.co.uk

URL7 http://www.its.leeds.ac.uk/software/saturn/

URL8 http://www.learian.co.uk/products/streetbox.html

Appendix One

COST Programmes

COST (European Cooperation in the Field of Scientific and Technical Research) is a framework for international research and development cooperation established between member countries of the European Union and other European counties. COST actions cover a number of scientific disciplines and aim to provide a forum for the coordination of research and dissemination of research findings. Each action is made up of a number of working groups concentrating on specialised areas and is set up for a period of five years. The following COST actions are concerned with air pollution, these include:

- COST 615: Database, Monitoring and Modelling of Urban Air Pollution. Working groups concentrating on databases, monitoring and modelling. (URL 3)
- COST 616: Mobile Sources of Air Pollution in Urban Areas.
- COST 617: Stationary Sources of Air Pollution in Urban Areas.
- COST 618: Institution Building and Information Policy in the Field of Urban Air Pollution.

These four Cost actions make up the CITAIR (Science and research for better air in European cities) programme and were started in May 1993.

- COST 710: Harmonisation in the Pre-Processing of Meteorological Data for Dispersion Models. This action started in 1994 and consisted of four working groups: Surface energy balance, mixing layer depth, vertical profiles of wind, temperature and turbulence, and complex terrain. (URL 2)
- COST 715: Meteorology Applied to Urban Air Pollution Problems. This action started in 1998 and consists of the following working groups: Urban wind field, surface energy budget and mixing height in urban areas, meteorology during peak pollution episodes and input data for urban air pollution models. (URL 4)

Appendix Two

Study areas

Northampton

Northampton is a rapidly expanding town in central England with a population of 197,000 and covering an area of approximately 80km^2. It has mostly light industry, but proximity to the M1 motorway produces a heavy reliance on distribution and warehousing. The river Nene that runs from west to east bisects the town. The commercial centre of the town is within the river basin (altitude 60m). The majority of the residential areas are found to the north and to the south on ground that rises a further 50 to 60 metres.

Two areas within the town were used in modelling exercises. One centred on the Borough Council offices at Cliftonville. These are located on low lying land to the east of the town centre (1km from centre) on a busy route into the town and less than a kilometre from a bypass and link road to the M1 motorway. The surrounding area is open, consisting of parkland, school playing fields and a golf course. The second area was Kingsthorpe, an area covering approximately 10km^2 on the northern edge of the town, but bisected by a major commuting route into the town. This area contains a local shopping centre and several schools; it consequently has much local traffic and pedestrian activity. This area has been previously subject to a major study assessing the impact of traffic management intervention on air pollution and human exposure (NAPS). The daily traffic patterns in these two areas reflect their difference in nature (see below)

A third location approximately 1Km west of the town centre was also used to take meteorological and air quality reading, but due to lack of traffic data no modelling work was carried out.

Hour starting (BST)	Kingsthorpe		Bedford Road		Cliftonville Road	
	Ave flow	TVEF	Ave flow	TVEF	Ave flow	TVEF
00:00	310	0.15	196	0.20	88	0.17
01:00	178	0.09	116	0.12	52	0.10
02:00	143	0.07	82	0.08	40	0.08
03:00	99	0.05	50	0.05	38	0.07
04:00	215	0.11	73	0.07	62	0.12
05:00	552	0.27	248	0.25	159	0.31
06:00	1124	0.56	521	0.53	327	0.63
07:00	2780	1.38	1503	1.52	995	1.93
08:00	3579	1.78	2021	2.04	1740	3.38
09:00	2921	1.45	1737	1.76	1273	2.47
10:00	2644	1.31	1677	1.70	989	1.92
11:00	2699	1.34	1807	1.83	1020	1.98
12:00	2879	1.43	1986	2.01	1193	2.32
13:00	2988	1.48	2020	2.04	1373	2.67
14:00	2956	1.47	1931	1.95	1228	2.38
15:00	3302	1.64	1888	1.91	1255	2.44
16:00	3581	1.78	2091	2.11	1454	2.82
17:00	3801	1.89	2238	2.26	1506	2.93
18:00	3244	1.61	1536	1.55	1002	1.94
19:00	2697	1.34	1182	1.20	723	1.40
20:00	2115	1.05	880	0.89	560	1.09
21:00	1598	0.79	636	0.64	403	0.78
22:00	1182	0.59	518	0.52	269	0.52
23:00	756	0.38	380	0.38	164	0.32
Total flow	48343		27316		17913	

Results of traffic surveys carried out in Northampton during July 1999

The average flows are hourly two-way weekday flows. The time varying emission factors (TVEF) are calculated for modelling purposes based on the average hourly flow for the entire week.

Barnes

Barnes is part of the London Borough of Richmond. It is confined by a bend of the river Thames and carries through its centre a road (Castlenau) linking Roehampton in the south with one of the bridges crossing the river - Hammersmith Bridge. This is not one of the major thoroughfares across the river as it carries only single carriageway traffic and has a weight restriction of 7.5 tonnes imposed. The bridge was closed in February 1997 to allow for essential maintenance work on the 110 year old structure. Before closure the average two-way 24hr traffic flow was 30,700 vehicles. The neighbouring Chiswick and Putney bridges both have over 50,000 vehicles per day (URL 6). The other major road (Lonsdale Road) within Barnes closely follows the line of the river and connects Mortlake with Hammersmith.

To the east of Castlenau, the area enclosed by the river is largely taken up by, now disused, Thames Water plc reservoirs and playing fields. The area enclosed by Lonsdale road and Castlenau is entirely residential. The major influence on local traffic patterns is St Paul's school, a large private school situated on Lonsdale road. Whilst the bridge was closed there was very little traffic activity on the south side of the river, however only half a kilometre to the north on the other side of the river is the Hammersmith flyover, a multi-lane highway. All bridge traffic except buses was diverted over Putney Bridge and Chiswick Bridge. (URL6)

Appendix Three

Model set up

Entry of data to ADMS is carried out through a series of screens, each of these is now described in turn.

Set up - The name of the site and project name are entered along with options of running chemical reactions, dry deposition and wet deposition. Specific site data can also be defined. A surface roughness value can be entered to define the characteristic roughness length of the study area. CERC recommend 0.5 as being appropriate for parkland and open suburbia and a value 1.0 for cities and woodland. In the absence of any real data, the value of 0.5 was selected as being more suitable for Northampton. Surface roughness of 1.0 was thought inappropriate for a town such as Northampton where buildings are rarely over four storeys high. Surface roughness is particularly difficult to estimate in urban areas and is highly variable, as it depend on both building height and density. It is a recognised deficiency of ADMS that only one value can be entered for the whole modelling domain (Carruthers *et al.,* 2000; Owens *et al.,* 1999). The latitude of the domain is entered; the default value is 52°. This is correct for Northampton where the actual latitude is 52°15′N. Latitude for Barnes is nearer 51°30′N and ADMS was adjusted accordingly.

The final option is the ability to set a minimum value of Monin Obukhov length, if this is left to the meteorological pre-processor to calculate. Under stable conditions a value would typically be between 2 and 20m, in convective condition its value would be negative and less than 20m. In neutral conditions the value would more than 20 and either positive or negative. In urban areas the urban heat island effect and the increased surface roughness has the effect of preventing the atmosphere from ever becoming very stable. The larger the urban area the greater the effect and the larger the minimum value below which the Monin Obukhov length cannot fall. To take this into account ADMS recommends appropriate minimum values depending on the type of urban environment, 10m for small towns with population less than 50,000 and 30m for cites and large town or a mixed urban/industrial environment. For large conurbations with population over 1 million a value of 100m is recommended. Thirty was selected for both Northampton and Barnes. Although Barnes is obviously part of a very large conurbation, 30m was still used, as it is upwind of the conurbation when one considers the prevailing wind direction. There are many large areas of open space such as Kew Gardens, Bushy Park, Hampton Court Park, Richmond Park, Wimbledon Common near Barnes. These and the river Thames, all would have a cooling effect on the local environment (Avissar, 1996). These minimum values seem appropriate for these types of urban terrain.

Source - The source screen allows entry of information pertaining to emission sources within the study area. There are three source types available; road, industrial and grid. There were no significant industrial sources in either of the areas studied. A Grid source could be used to model a number of emission sources that are closely located at one site such as number of chimneys within an industrial complex or domestic heating from individual areas within an larger urban area. It also allows for the possibility of grouping together suburban roads as a single area source. In Kingsthorpe and in Barnes all road that were likely to significantly contribute to pollutant levels were modelled separately. Road sources can be defined in ArcView and then transferred to ADMS or by creating a road.xls file in the Access Emission Inventory. The Cliftonville and Barnes modelling scenarios were created by the former method, the Kingsthorpe modelling scenario had been created previously for the NAPS study so were already in emissions database format. For road sources there is also the possibility of defining road elevation, road width (width of carriage way or if canyon height is entered, this is building to building width). No roads were elevated in this study. The width was left as the default setting (10m) unless roads were known to be multi-laned or dual carriageway. Canyon height was not considered applicable except in Barnes.
ADMS contains a database of vehicle emission factors and it is possible to select these for the year of interest. Version 1.51 of ADMS-Urban uses the Design Manual for Roads and Bridges (DMRB) vehicle emission data set issued in 1994 (Department of Transport, 1993). With this set it was only possible to model for years ending in a 5 or an 0. Version 1.53 gives the option of using the 1994 values or a more recent set of emission factors issued in 1999. These allow modelling for any year. Some of the earlier work in this

project used version 1.51 with the year set at 2000 for 1999. Later work used version 1.53 with emissions set at 1999 or 2000 as appropriate. Predicted concentrations will be different depending on the data set used. However as no direct comparisons were made, this is unlikely to have much effect except it may contribute to some error when comparing monitored with modelled results. It is possible to enter user-defined values, but ADMS defaults were always used. It was considered important to use the model in a way that was appropriate for general use by a regulatory authority.

Each road source is also assigned an average hourly vehicle flow and average speed for a light duty and a heavy-duty component of the vehicle fleet. The vehicle flow will be adjusted accordingly throughout the 24-hr hours and for weekdays/ Saturdays/Sundays by means of time varying emission factors. The model does not permit the entry of factors to allow for vehicle speed changes throughout the day. This is considered a deficiency in the modelling as clearly emissions do vary with vehicle speed (DoT, 1993) and heavily trafficked roads are most likely to be subject to a reduction in speed during busy times of day. For this project traffic flow data came from a variety of sources.

Time varying emissions are calculated by:

TVEF = average flow for particular hour/average hourly flow for whole week.

Meteorology - Meteorological data can be entered *'from file'* or by the *'enter by hand'* option and can be hourly sequential or statistical data. There are a number of options whereby certain parameters may be entered directly to ADMS, if accurate values are known, otherwise they are calculated by the meteorological pre-processor. The minimum variables required are wind speed and wind angle with either surface heat flux (Wm^{-2}) or year, Julian day, time, cloud cover (Oktas). Additional values that may be entered by hand include boundary layer height, surface temperature and lateral spread - the standard deviation of the mean wind direction. If data is entered by file a greater range of input parameters is allowed, in addition to the parameters listed above, it is possible to enter the reciprocal of the Monin Obukhov length. A complete list of possible variables is given below. Different output values may be obtained depending on the combination of input data is used. CERC recommend that boundary layer height should always be entered if a better estimate can be provided than that which would be calculated by the model

however this data would not be readily available to most model users. It is may be determined by radiosonde or Lidar, or can be calculated from wind speed and temperature profiles. It is especially difficult to obtain values for urban areas. If boundary layer height is not available, but the reciprocal of the Monin Obukhov length or surface sensible heat flux or cloud cover are, the reciprocal of the Monin Obukhov will always be used in preference. It is always useful to add temperature to produce a better estimate of boundary layer height.
It is also possible to enter the height at which wind speed was determined (as the model recalculates this to the standard value of 10m), the wind sector size (10° for sequential data, but generally 30° for statistical data) and whether or not the meteorological data is representative of the modelling site. If it is unrepresentative, as is likely to be the case when using data supplied from a remote meteorological station for urban modelling, it is possible to enter a precipitation factor to allow for any difference in snow/rainfall and to give the surface roughness of the meteorological site. The surface roughness of these sites is not readily available and an estimate has to be made based on the knowledge that the recommended siting for a meteorological station is in short grass and well away from trees and other large obstructions (URL5). Precipitation was not used in this study.

Chemistry - ADMS has the ability to resolve the main chemical reactions between nitric oxide, nitrogen dioxide, ozone and volatile organic compounds. Three different chemistry options are available, but they can only be run with hourly sequential data and when cloud cover is specified. The three options are: -

- Derwent/Middleton Correlation which uses a simple function to estimate the concentration of NO_2 from a given concentration of NO_x.

$[NO_2] = 2.166 - [NO_x]\,(1.236 - 3.348A + 1.933A^2 - 0.326A^3)$

Where $A = \log_{10}([NO_x])$

This is only valid over the range 9ppb to 1141.5ppb. It only applies to urban areas and there is some question as to how transferable it is to areas other than where the original work was carried out. (McHugh, 2000)

- GRS (Generic Reaction Set) which uses a set of eight chemical reactions to model the interactions of NO, NO_2, VOCs and O_3 in the atmosphere. Background data are required for NO_x, NO_2 and O_3.
- GRS + Box Model is an extension of GRS to be used when modelling very large urban areas.

Grids - The grid screen is used to define the area over which the model calculates pollutant concentration and the location of individual receptor points. If the gridded output is selected there are three options that define how the gridding is carried out. For modelling road sources the 'intelligent' option is recommended, whereby grid size varies with distance from source and grids are smallest where pollutant gradients are steepest. Three co-ordinates define point source, for gridding it is also possible to specify a height at which the concentration is to be determined. The pollutant concentration at any specific receptor point is different if the model is run for specified point only or for point and gridded output. The latter gives a more accurate prediction (pers comm Anne Danskin, CERC), but substantially increases model run times. In this study the model was run for point receptors only and this is an acknowledged additional source of error.

Output - The output screen is used to select which pollutants are to be included in the model run, the required averaging time and the units to be used. If a number of emission sources are included in the model set up they can be grouped together into a number of sub-groups. This screen allows for a selection to be made for each model run. The modelling scenario file for Kingsthorpe contained all the roads used in the NAPS study, although initially roads in the immediate vicinity of the monitoring point were selected and modelled for separately, there seemed little advantage in doing this and all roads were latter included in model runs

ADMS Meteorological input variables

ADMS notation	Standard notation	Definition
U	U	Wind speed
UGSTAR	Ug	Geostrophic wind speed normalised by the friction velocity
PHI	Φ	Wind direction (direction wind is coming from in degrees)
DELTAPHI	$\Delta\Phi$	Geostrophic wind minus surface wind direction (degrees)
FTHETA0	$F\theta_0$	Surface heat flux
RECIPLMO	$1/L_{MO}$	Reciprocal of the Monin-Obukhov length
H	h	Boundary layer depth
NU	N_u	Bouyancy frequency above boundary layer
DELTATHETA	$\Delta\theta$	Temperature jump across the boundary layer top
T0C	$T^c{}_0$	Near surface temperature (°C)
P	P	Precipitation rate (mm/hour)
CL	Cl	Cloud amount (oktas)
R	r	Surface albedo
ALPHA	α	Modified Priestly-Taylor parameter
TDAY	t_{day}	Julian day number
THOUR	t_{hour}	Local time (hours)
FR	fr	Frequency of occasions when these conditions occur (arbitrary units, e.g. percentage of occasions or number of hours per year)
SIGMATHETA	$\sigma\theta$	Standard deviation of wind direction (degrees)

SI units are used except where stated. (CERC, 1999)

Appendix Four

Meteorological Data

Meteorological data used to assess the local variation in cloud cover, wind speed and direction and to provide input data into the dispersion model came from a variety of sources. These are described below.

British Atmospheric Data Centre (BADC)

The British Atmospheric Data Centre is the Natural Environment Research Council's data centre for Atmospheric sciences and holds long term data sets of atmospheric data produced by NERC funded projects and data produced by third parties such as the UK Meteorological office. Three types of data from the UKMO surface data set were accessed using Telnet.

Synoptic data – There are currently 267 synoptic stations in the UK at which observations are taken simultaneously at these hours; 0000Z, 0600Z, 1200Z,and 1800Z. However, many of them also make 24-hour observations. Only some stations are staffed 24 hours a day and are able to make cloud cover observations. Data from several of the UK synoptic stations that make hourly surface synoptic weather observations were used in this study. Only data for wind speed, wind direction, cloud cover and temperature were required for model input. Definition of this data and the units used is given below. There are differences in the way both wind speed and directions are recorded depending on the type of instrumentation used. The averaging period may vary if either the wind direction varies by more than 30 degrees or the speed varies by more than 10 knots in the proceeding 10 minutes.

A list of the sites used is this study is also given below.

Radiation data – There are 40 meteorological stations that recorded hourly solar radiation during 1999. They record both the global and diffuse components of radiation. Only a few of these also record cloud cover. Details are given below

Parameters	Description
Hour of observation	Hour of observation (GMT)
Wind direction	The mean direction is for a period of 10 minutes ending at the hour of observation. Recorded to the nearest 10 degrees.
Wind speed	Speeds are adjusted to a standard height of 10 metres. The mean speed is for a period of 10 minutes ending at the hour of observation. Recorded as Knots to the nearest whole unit.
Total cloud amount	The proportion of the sky covered by cloud is recorded as 'oktas' or eighths. A value of 9 is sometime used to indicate and obscured sky i.e. in the case of dense fog or mist.
Cloud type	Determined by WMO codes for low, medium and high cloud types
Cloud base height	Height above ground of the lowest cloud base, recorded in decametres.
Temperature	The measurement of air temperature is by means of a dry-bulb thermometer housed in either a white painted louvered screen or a Gill screen at a height of 1.25 metres above short grass. Temperatures are reported to 0.1 degree Celsius.
Global solar irradiation amount	'Global' radiation is defined as that obtained from both the sun and the surrounding environment. Recorded as Wm^{-2}.
Diffuse solar irradiation amount	'Diffuse' radiation is defined as that obtained when a shade ring is used to block out the sun's direct radiation. Recorded as Wm^{-2}.

Weather observations recorded for synoptic and radiation data and as defined by BADC (URL5).

Site Name	County	Grid ref.	Lat.	Long.	Elevation	Station code
Aviemore	Invernes s-shire	NH896143	57.206	-3.827	228	WMO 03063 DCNN 0585
Benson (Samos)	Oxon.	SU625917	51.620	-1.097	57	WMO 03658
Bracknell (Samos - Beaufort Park)	Berks.	SU846664	51.389	-0.783	74	WMO 03763 DCNN5592
Brize Norton	Oxon.	SP292067	51.757	-1.576	81	WMO 03649
Coleshill	Warks.	SP211869	52.479	-1.689	96	WMO 03535
Elmdon	West Midlands	SP167841	52.454	-1.754	98	WMO 03534
Eskdalemuir	Dumfries -shire	NT235026	55.311	-3.205	242	WMO 03162 DCNN 6679
Heathrow	Greater London	TQ077767	51.478	-0.448	25	WMO 03772
Hemsby	Norfolk	TG493162	52.685	1.689	14	WMO 03496 DCNN 3095
London WC (Samos)	Greater London	TQ302800	51.503	-0.123	42	WMO 03779
Northolt	Greater London	TQ099846	51.549	-0.414	34	WMO 03672
Norwich WC	Norfolk	TG233082	52.625	1.299	35	WMO 03492
Wittering	Cambs.	TF043026	52.610	-0.459	73	WMO 03462

Meteorological stations from which data was used in this study (URL 5).

(Elevation above sea level in metres)

Climatological data - There is a large number of stations spread across the UK (541 in 1999) that make climatological readings daily at 0900Z. There were accessed for cloud cover data. Details of those within a 70km radius of Northampton are given below.

Site name	County	Grid Ref.	Lat.	Long.	Distance from N'pton (km)
Moulton Park	Northamptonshire	SP765644	52.272	-0.878	0
Rockingham	Northamptonshire	SP865918	52.516	-0.725	27
Monk's Wood	Cambridgeshire	TL201796	52.400	-0.272	48
Cambridge NIAB	Cambridgeshire	TL435606	52.224	0.101	67
Cambridge Botanic Gardens	Cambridgeshire	TL456572	52.193	0.130	70
Mepal*	Cambridgeshire				70
Marholm	Cambridgeshire	TF145020	52.603	-0.309	
Woburn	Bedfordshire	SP964360	52.013	-0.595	35
Grendon Underwood	Buckinghamshire	SP677215	51.887	-1.016	45
Rugby	Warwickshire	SP507749	52.369	-1.255	29
Wellesbourne	Warwickshire	SP271565	52.205	-1.603	49
Stratford upon Avon	Warwickshire	SP164549	52.191	-1.760	61
Newtown Linford	Leicestershire	SK530095	52.680	-1.215	52

Sites that record cloud cover at 9.00am (URL 5).
* No data available for Mepal

Roadside weather stations

Northamptonshire County Council and The Highways Agency both have a number of roadside weather stations spread across the county which are maintained and data managed by consultants – Vaisala. These are operated for part of the year as part of the road de-icing programme. Data were made available for this project from winter 1997/98. Although the siting of these station does not comply with the strict requirements used in the Meteorological. Office synoptic network, the data is nonetheless useful for analysing variation in meteorological data over a smaller spatial scale than is available from the synoptic stations.

Direct comparisons cannot be made with synoptic data as readings are of a different format and are taken at a height of six metres.

The location of these is shown in Figure 3.3 and further details of these sites are given below.

Site name	Road	Distance from Northampton	Direction from Northampton
Farthinghoe	A422	28Km	SW
Naseby	A14	20Km	NNW
Rothwell	A6	22Km	NNE
Upper Benefield	A427	35Km	NE
Hill Top Harrowden	A509	17Km	NE
Bythorn	A14	35Km	ENE
Collingtree	A508	5Km	S

Roadside weather stations.

UCN data

University College Northampton has an automatic weather station sited within the college grounds at Park Campus, Boughton Green Road, Northampton (referred to in the text as 'Moulton Park'). This consists of a 6m mast with wind vane, cup anemometer, temperature and relative humidity sensor. This was used to provide wind speed, wind direction and temperature data for a fixed site within Northampton and was used to make comparisons with data recorded from elsewhere within the urban area and for some model runs.

The college is also designated as a climatological station within the Meteorological Office's data collection system (see above). Readings of Daily Weather Observations are taken at 0900Z. Cloud cover is one of the data entry requirements of ADMS-Urban and this provided the only truly local reading.

The location is shown in Figure 3.1.

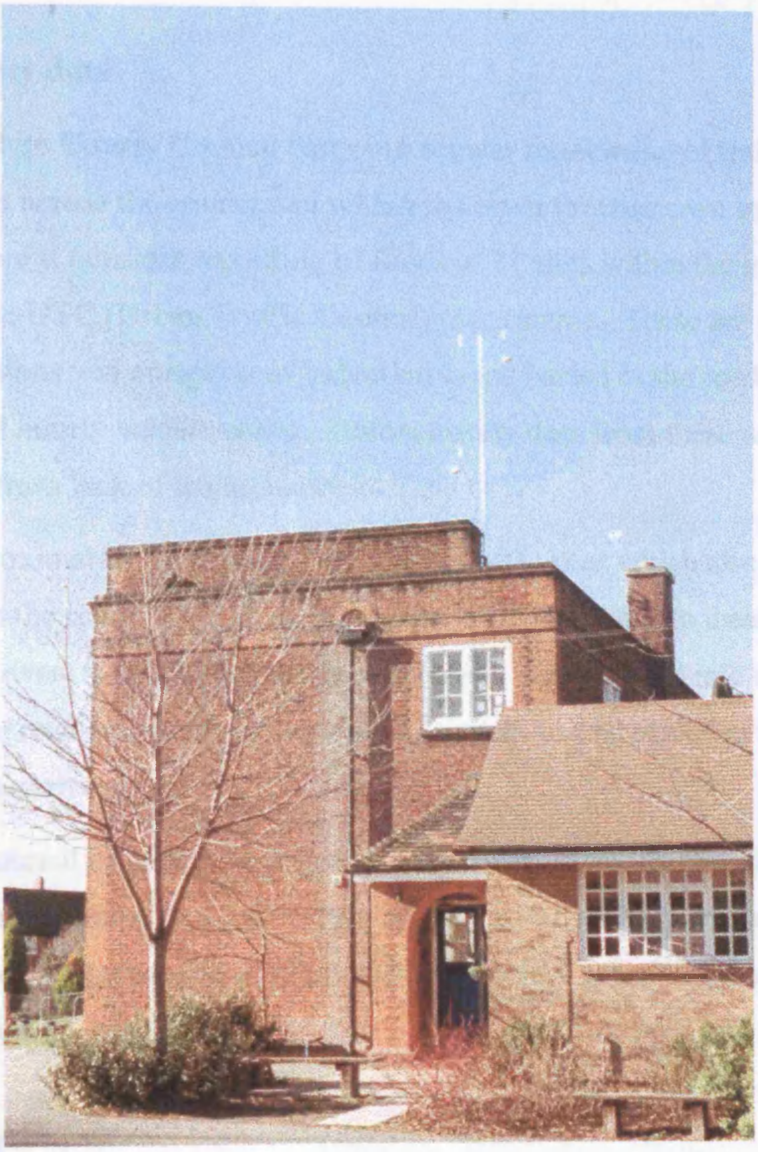
Special survey data

Concurrent with the pollution monitoring that was carried out for the purposes of this project; meteorological recordings were also made. The Horiba mobile pollution

laboratory has a meteorological mast which records wind speed, direction and temperature as 15-minute averages. These readings are made at a height of 6m. Equipment mounted on a Clark mast includes a Lee-Integer combined Humidity and Temperature probe (DCH24T), A Met-Check Wind speed sensor (MET 5204) of the cup anemometer type and a Met-Check wind vane (MET5212) (Horiba, 1995).
In order to obtain temperature and wind speed profiles readings were required at two different heights, this was not possible with the Horiba mast so additional data collection was carried out. A sectional aluminium mast mounted with a wind vane, two anemometers and two temperature probes was erected for a two month period on the roof of the council offices at Cliftonville, Northampton and on the roof of Lowther primary school, Barnes for 4 months (see overleaf). The instrumentation comprised of Vector Instrument anemometers (3cup inline type A100R) and potentiometer windvanes (W200P), also temperature probes supplied by Grant Instruments. Squirrel Data Loggers from Grant Instruments were used for data recording purposes.

NWP data

The NAME model is a Lagrangian particle model developed by the UK Met. Office and designed to forecast nitrogen oxide concentrations up to six days in advance. NAME has a resolution of 15km x 15km. Predictions, for use within this study, for a grid centred on Northampton were supplied by the UK Meteorological. Office. The meteorological input for NAME is provided by the Met. Office Numerical Weather Prediction (NWP) suite of models. NWP predicts or interpolates data for a number of meteorological parameters on a 12km horizontal grid at meso-scale resolution or a 60km grid for regional resolution. Data, with the latter resolution, was supplied for four months during 1999. The following parameters were used to run ADMS; wind speed and direction, surface temperature, surface heat flux, boundary layer height and reciprocal of Monin Obukhov length. NWP can predict cloud cover, but this was not available for the time period of interest.



Meteorological mast on the roof of Lowther Primary School

Appendix Five

Traffic Data

Local Authority data

Northamptonshire County Council carry out regular monitoring of traffic flows at a number of sites across the county and within the town for their own traffic management purposes. There is constant recording of flows at 11 sites within the town centre that form part of the UTC (Urban Traffic Control) programme. These are generally sited near major intersections and comprise of induction loops buried in the road. Data are recorded as a directional hourly vehicle count. Unfortunately data from these are often unreliable as they suffer from lack of maintenance.

There are approximately 30 County Council run ATC sites which also have induction loops buried in the road. The actual counters are only attached to these for a two-week period in June every year. These are situated on all major routes into the town and at selected sites spread across the urban area. Counters can be placed at these sites at any time for special surveys.

The County Council as part of their sustainable transport policy also carry out manual traffic counts at a number of sites for just one day a year. These are carried out in June or September. Counts are carried out for twelve hours and vehicle type is classified into seven categories.

Special surveys

Northampton - Counts were carried out by Northamptonshire County Council at three sites specifically for this project either using the ATC sites or by placing pneumatic tubes across the road. For a two week period in July 1999 counts were carried out on Cliftonville Road (A4501), Bedford Road (A428) and Harborough Road (A508). Counts were also carried out on the Harborough Road for a two-week period in October 1999.

Harborough Road forms the busiest section of road through the Kingsthorpe study area. Traffic counts were carried out only 50m from the modelling receptor point. Data from this site was used to provide the time varying emission factors that were applied to the whole area.

Cliftonville Road and Bedford Road are adjacent to the Northampton Borough Council offices. The hourly flows on Cliftonville Road were used to calculate the time varying emission factors.

Barnes – Richmond Borough Council carried out traffic counts using pneumatic tubes on Castlenau and Lonsdale Road. The location of these is show in Figure 3.2. Counts were carried out for a three-week period at each site, but unfortunately there were not concurrent and only overlapped by 9 days. The Castlenau tube broke soon after Hammersmith Bridge reopened and was not replaced. These counts also gave an estimate of vehicle speed.

Modelling for Barnes covered a transitional period before and after the re-opening of Hammersmith Bridge. Time varying emission factors were calculated using Castlenau for the period up until the bridge opened on 21st December 1999 and data from here and Lonsdale was used to calculate an average hourly flow.

SATURN traffic model

SATURN is combined traffic simulation and assignment model developed in the 1970s by Leeds University, Institute of Transport studies for the analysis of traffic management schemes. It is widely used in the evaluation of road schemes and for research projects linked to all aspects of road use. It can operate on many levels from the simulation of flow at individual junctions to, in assignment mode, the analysis of networks with up to 6000 links. (URL 7)

Output from the SATURN traffic assignment model had previously been obtained from Northamptonshire County Council and used in a study on the effects of traffic management interventions on air pollution and human exposure in Northampton. The model was calibrated on 15/10/1997 using data from detailed surveys on traffic flows and turning movements. SATURN produces a vehicle flow for each road section which is representative of the average hourly weekday flow. It also produces an estimate of vehicle speed for each road section. The traffic counts carried out in 1999 gave similar values to the SATURN output from 1997, so no further calibration was considered necessary.

Saturn data was used in both Kingsthorpe and Cliftonville to provide average hourly flows on minor roads.

Appendix Six

Air Quality Monitoring Equipment

Horiba Mobile Air Pollution Laboratory

The Horiba mobile contains monitors for gaseous and particulate pollutants housed in a purpose built self-contained trailer unit. The common inlet for gaseous pollutants is at a height of approximately three metres. The air is drawn in through Teflon tubing and split in a glass manifold to the various analysers. Details of each analyser are given below:

CO monitor – APMA-360; using non-dispersive infrared absorpiometry with a lower limit of detection of 0.05ppm and operating range up to 100ppm. Air is drawn in at a rate of 1.5litres per minute.

NOx monitor – APNA-360; using chemiluminescence to determine concentrations of NO and NO_x and by difference, NO_2. The lower limit of detection for NOx is 0.5ppb. Flow rate is set at 0.8l/min.

The unit also houses a non-dispersive ultraviolet absorptiometer (APOA-360) to monitor for ambient ozone, a sulphur dioxide monitor (APSA_360) using ultraviolet luminescence methods as its measuring principle and a Rupprecht and Patashnick TEOM Series 1400a PM10 monitor (Horiba, 1995). Although the monitor collected data for these pollutants, the data were not used for the purposes of assessing the performance of ADMS.

The monitors are set to auto-zero using ambient air that has been drawn through a Purafil and a charcoal filter every 73 hours. They also calibrate with the same frequency using span gases supplied by Messer Ltd and carrying a certificate of analysis. Additional manual calibration was also carried out whenever the filters were changed. This was carried out on a fortnightly basis. Horiba personnel serviced the unit on a six-monthly basis.

Similar fixed site Horiba analysers are installed at Northampton Borough Council offices on Bedford Road, Northampton. The inlet is placed on the roof of the building a height of approximately six metres.

Learian Streetbox

The Learian Streetbox uses a chemical sensing technique to detect ambient concentrations of carbon monoxide and nitrogen dioxide. The CO sensor has an operating range of 0 to 500ppm with a resolution of 0.1ppm and the NO_2 sensor has a range of 0 to 20ppm and a resolution of 0.02ppm. Streetboxes are small battery operated units that can be mounted on any suitable structure such as a lamppost. The only consideration regarding their siting is that they should be out of reach to reduce the likelihood of theft or vandalism and they should not face due south.

(Learian Ltd, 1997; URL8)

The performance of the Streetbox monitors was compared with the Horiba. One of the monitors was placed on the roof of the Horiba and although correlation was good (r^2 = 0.94), it gave relatively higher CO readings. The following linear relationship between the two types of monitor was found to exist;

Horiba CO = (Streetbox CO x 0.785) + 0.107.

When the two Streetbox monitors had been placed at the same location, one metre apart, for a ten-day period the CO values were again in close agreement. However, one monitor gave consistently higher readings and the correlation was poorer (r^2 = 0.81). The relationship between the two monitors was found to be;

Streetbox a = (Streetbox b x 0.82) + 0.083.

Appendix Seven

Background data

NETCEN

The NETCEN automated air quality network operated on behalf of the DETR by AEA Technology consists of currently 112 sites, of which 93 are classified as urban. These produce hourly pollution levels for an array of determinands, though not every site records the same range. Sixty-five sites record carbon monoxide concentrations and 80 sites record nitrogen oxides. Within the urban classification there are several sub-classifications such as roadside or urban centre. However there does not appear to be any constancy in how these are assigned. Thirty-eight sites in England, Scotland and Wales are classified as urban background or suburban.

It could be suggested that urban background levels for determinands such as carbon monoxide that are principally traffic derived will not vary much from one urban centre to another of similar size and within a particular region of the UK. Although urban background sites are not evenly distributed, data from a NETCEN site will give a general indication of likely levels elsewhere. Data from the Leamington site (1999) were downloaded from the NETCEN website (URL 1) to produce hourly averages for CO. These were used in the NAME exercise. Data from two London urban background sites were used in the Barnes study. In this case NO_x levels for each hour from 15th December to 31st December was used. Details of these sites are given below.

Learian Streetbox

A Streetbox was placed within the grounds of University College Northampton, Park Campus for two weeks during March 2000 and a further two weeks during May 2000. Although only 50m distant from access roads within the college, it was 300m from a road carrying an average of 1000 vehicles per hour and as such could well be described as an urban background site. Data were used from the initial two-week period to calculate a representative hourly average carbon monoxide concentration for each hour of the day. The component of carbon monoxide derived from local sources was removed by running ADMS with this as a receptor site and subtracting the hourly average obtained from the model output from the monitored value. These data were then used in Northampton modelling exercises.

Site Name	Site type	Brief Site description
Leamington Spa	Urban background	The monitoring station is to the rear of a town centre building. Apart from an access road , the nearest road is 50m away and is free flowing. The inlet manifold is 4m high.
London Teddington	Urban background	The nearest road is 500m from the monitoring station. The inlet manifold is 15m above ground level. There is parkland to the south and west.
West London	Urban background	The monitoring station is sited within a council depot. The nearest road, 50m away, is a relatively busy single carriageway. The inlet manifold is 30m above ground level. The surrounding area is built up.

NETCEN background sites. (URL 1).

Appendix Eight

Monitoring programmes

Northampton

Westbridge - The Horiba monitor was on site from 7th December 1998 to 12th April 1999. Data recording did not start until 7th January 1999 when the monitor was moved to a roadside position. Data from the UTC site proved to be erratic and the road layout meant that a special traffic survey was impractical. Meteorological data were collected from the mast on the Horiba. There were used in conjunction with data from Meteorological Office sites and roadside weather stations to analyse spatial variation in wind speed, wind direction and temperature.

The Mounts - The Horiba monitor was on site from 12th April to 24th June. No traffic data were available from the nearby UTC site, as a result no modelling was carried out for this site.

Kingsthorpe -The Horiba monitor was on site from 24th June to 10th December collecting air quality and meteorological data. Meteorological data were also recorded on the roof at Cliftonville from 13th July to 9th August. Traffic data were record using the induction loop on the A508 from 1st July to 15th July, on Bedford Road using pneumatic tubes from 20th July to 30th July and on Cliftonville Road using the induction loop from 5th July to 29th July. A further period of traffic sampling was carried out on the A508 from 18th October to 12th November.

The Horiba was again at Kingsthorpe from 18th February to 11th May 2000. During this period some background carbon monoxide monitoring was carried out at the college campus. Learian Streetbox CO monitors were placed on lampposts in Kingsthorpe from 14th April to 11th May.

Modelling and meteorological analysis were carried out using data from this site.

Barnes

A Horiba monitor equipped with a TEOM and NO_x analyser was installed at Lowther Primary School and the meteorological mast erected on the roof in November 22nd 1999. These were left in place for the duration of the monitoring programme i.e. until 17th February 2000. Learian Streetboxes were placed on lampposts on Castlenau from 19th November until 21st December. The Horiba monitored equipped with a TEOM and gaseous analysers was moved to Holy Trinity Church on Castlenau on 15th December and left in place until 17th February. The bridge re-opened 21st December. Streetboxes units were returned to Barnes on 25th January and left in place until 17th February.

Traffic counts were carried out on Castlenau from 30th November until 22nd December. The counter was damaged two days after the bridge re-opened and was not replace by the local authority. Counts on Lonsdale Road covered the period from 14th December until 3rd January. Unfortunately this monitoring ceased before St. Paul's school re-opened after the Christmas holidays. As a result no traffic counts were available that were representative of the term-time situation with the bridge open.

Richmond Borough Council carried out air quality monitoring at the library on Castlenau and the wetland site situated near the southern end of Castlenau.

Although a Horiba monitor was placed at Lowther Primary School and Streetbox monitors were placed on lampposts on Castlenau, only data from the Horiba monitor placed outside Holy Trinity Church, Castlenau was used in the work carried out for the study. Similarly data from the monitor at Barnes library is not reported here, however data from the wetland site is shown in Figure 7.4 and 7.5.

Appendix Nine

Model Performance

Although dispersion models had been in use for some time it was not until the 1970's and early 1980's that there was considered to be any need for formal evaluation procedures. There needed to be a way of establishing model reliability in performing specific tasks. Many different procedures for assessing performance have been proposed and some of these are reviewed here.

Model evaluation

One of the first attempts at standardising procedures was made at a workshop convened by the American Meteorological Society (AMS) in September 1980. At this recommendations were made for model performance evaluation measures and methods, and performance standards were proposed (Fox, 1981). Previous to this model evaluation had merely consisted of graphical plots of predicted against measured concentration values and the calculation of simple parameters such as correlation coefficients, however it became clear that this was no longer adequate (Zanetti, 1990).

How evaluation should be carried out is still a subject of much debate. Beck *et al.* (1997) suggested that the existence of many definitions of the term validation, may well be a symptom precisely of how intractable the problem of model validation is. One outcome of the 1980 AMS workshop was to distinguish between the concepts of model verification, evaluation and validation. They defined evaluation as the process of examining and appraising the performance by comparing the model's concentration estimates to measured air quality data and validation as the establishment of a conclusion by detailed and copious evidence that leads to a formal recognition (Fox, 1981).

Zanetti (1990) suggested that model validity should be considered as the 'theoretical' ability of the model to *perform* with error-free inputs and that a model only can be validated against a theory or against another model, but not against measurements. He considered model evaluation to be the quantification of the performance of the model in real cases with real data. Successful validation and/or evaluation of the model leads to model verification (Zanetti, 1990). Olesen (1995b) however defines validation as the comparison of model predictions with experimental (*or observed*) data. Beck *et al.* (1997) themselves suggested that validation is the determination of the correctness of a model with respect to user's needs and requirements. In other words, a judgement about the validity of a model to perform its design tasks reliably, that is, with minimum risk of an undesirable outcome.

However, the overall term for model validation/evaluation is defined, many of those actually carrying out validation procedures have agreed that there are broadly two type of performance measure involved. Operational performance refers to comparison with monitored data exclusively within an application context and will lead to the creation of statistical performance measures. Scientific or diagnostic performance measures are aimed at understanding model behaviour, confirming that the model is based on sound physical principles and that it gives good predictions for the right reasons. Weil *et al.* (1992) pointed out that although most model users are only concerned with operational performance, investigating the model physics is important if one is to have faith in performance when modelling new situations and beyond the range of existing data.

The AMS 1980 workshop recognised that a distinction had to be made between evaluating point source models and urban models, not only because the format of the experimental data is different, but in how performance is assessed (Fox, 1981). There are several datasets available for evaluating short-range point source models, notably those using data from the Kincaid experiment in the USA conducted in 1980-81 and experiments in Copenhagen, Denmark and Lillestrøm, Norway, conducted in 1978-78 and 1987 respectively. Following on from model evaluation work within the EC and the series of conferences entitled 'Harmonisation with Atmospheric Dispersion Modelling for Regulatory Purposes', a model validation kit has been developed with these datasets specifically in mind (Olesen, 1995b). However as yet, there are no such formal procedures for evaluating urban models. Oettl *et al.* (2001) in particular stressed the need for good quality datasets to evaluate ground level line source models.

Apart from the practical problems, there are many theoretical considerations to be taken into account. These fall broadly into two groups, firstly considering what exactly the observed pollution data and the model output data actually represent and how they should best be described, and secondly where the sources of error and uncertainty derive from and how this should be accounted for in the validation process.

The observed concentration of a pollutant at any particular monitoring point is largely governed by the random process of turbulence and as such should be considered a *random* variable itself with an ensemble mean concentration C and an associated distribution described by σ_c. The observations should best be described statistically by a probability distribution parametised by C and σ_c, but as dispersion models only predict C, this makes comparison difficult (Weil *et al.*, 1992). Some workers (McNair *et al.,* 1996; Zanetti, 1990) have considered the space and time limitations that even 'error-free' measurements have. They only represent a very small region around the measurement point, so comparison with grid averaged model output will not be appropriate.

The 1980 AMS workshop also considered whether models should be judged on the ability to predict the highest peak values or whether upper percentiles should be compared. They concluded that evaluations applied to estimates of mean performance would provide more information about overall performance than evaluations applied to extreme values, and that performance should be judged on the same averaging times as are required by air quality standards (Fox, 1981). After the 1980 workshop, it was felt that there was too much emphasis had been placed on operational performance and that there should be more testing of model physics. Different kinds of approach have subsequently been suggested. Irwin and Lee (1997) discuss the problem of comparing data where natural variability occurs. They claim that in these situations it is more robust to compare model evaluation data that has been stratified into regimes or ensembles, based primarily on the model physics and considering the range of applications the model is designed for. Weil *et al.* (1992) suggest that models developed and tested with intensive datasets of limited duration should also be evaluated with routine monitoring records to capture all combinations of meteorological conditions.

Model evaluation should, if carried out properly, determine the range of circumstances over which the model will perform adequately, should define the accuracy of the inputs required to implement the model and if possible should identify and quantify the reasons for differences between predictions and observed values. Beck *et al.* (1997) noted that successful validation does not necessarily guarantee accurate predictions and that the social consequences of accepting poor models are much more serious than the consequences of rejecting good models. From this standpoint they suggest that model validations should always by carried out to test the hypothesis that the model is invalid. It should be considered an integral part of the evaluation process that weaknesses in the model are identified and that this directs attentions to topics for further research (Mole *et al.*, 1993; Russell, 1988).

Measures of performance

There are many different types of performance measure in use. The 1980 AMS workshop identified three categories of data that they could be applied to

- Predicted and observed concentration values paired in space and time,
- Peak concentrations data either paired or unpaired,
- Cumulative frequency distributions of unpaired predicted and observed concentrations.

The first is considered the most stringent, as it tests the models' ability to perform well at all locations and under all conditions. Comparison of peak concentrations only tests how well the model performs under worst case conditions.

There are two types of performance measure that can be applied to these groups of data; measures of difference and measures of correlation. Depending on the type of problem under investigation different types of data and different measures will be appropriate (Fox, 1981). For example, when models are used to carry out 'worst case' type analysis to ensure compliance with regulatory standards, the case could be made for only studying performance for concentrations in the upper half of the percentile range. Error in predicting low concentration may be of little consequence and the measurement error associated with monitoring low concentrations will be significantly greater rendering comparisons virtually meaningless. It has been suggested that measurement error on concentrations of 2.0ppm carbon monoxide could be 25% or greater and over 50% on typical background levels of 1ppm (Noll *et al.,* 1978; Rao *et al.,* 1980).

There are often statistical problems associated with performance evaluation. Firstly many of the recommended statistical tests assume a normal distribution for the difference between observed and predicted concentrations; this is not always the case. In addition many tests are designed to determine whether or not two sets of data came from the same population. While in air pollution modelling it is already known that they do not (Fox, 1981; Weil *et al.,* 1992). Details of the statistical tests most commonly used in model evaluation studies are given below.

It is generally acknowledged that correlation coefficients can give misleading results. Fractional bias is suggested as an alternative measure, but applied to the top 25 percent of values rather than the actual peak value. Values for fractional bias vary between +2.0 and -2.0, extreme under-prediction to over-prediction respectively. As it is dimensionless it can be used to compare data with different concentration levels (Cox and Tikvart, 1990). The index of agreement is another statistical test that has been developed recently. This determines the degree to which the magnitude and sign of the observed value about the mean observation are related to the predicted deviation about the mean of the predicted values. Perfect agreement between observed and predicted values gives an index of agreement of 1.0, but experiments have shown that even if the set of predicted concentrations is completely randomised, the IA will be approximately 0.4 (Oettl *et al.,* 2001). The index of agreement has recently been used in two line source model evaluation studies by Sivacoumar and Thanasekaran (1999) and Karppinen *et al.* (2000a) and to compare a Gaussian and a Lagrangian model (Oettl *et al* 2001).

Model Sensitivity

Sensitivity analysis is just one component of a range of tests that can be used to evaluate model performance. Performance characteristics are generally based on either measures of difference (residuals) or measures of association (correlation) and so both require comparisons to be made with real field data. Sensitivity analysis studies the response of the model to varying levels of input and generally involves characterising the relationship between change in output and the incremental variation in input parameters, it only requires the use of realistic scenarios. It does not directly indicate the reliability of the model predictions, but the performance would be suspect if small changes in an input parameter lead to large changes in output (Beck *et al.,* 1997). It is a way of determining systematically the effect of uncertainty on model output (Hwang *et al.*, 1997).

Using sensitivity to evaluate how well the model performs over a variety of situations gives a measure of its transferability and an indication of the desired accuracy of input data. Bellasio (1997) carried out sensitivity analysis as part of the model evaluation, not only because it identifies the critical inputs but also allows the evaluator to determine if nature displays the same sensitivity to these inputs as the model.

Results of several different sensitivity studies have been reported. In early work by Noll *et al.* (1978) sensitivity analysis was carried out on three highway line source models. The sensitivity of model predictions to changes in input parameters was determined by calculation of normalised concentration (Cû/Q) vs normal distance to the highway edge for different wind angles and for stability classes B and E (C = pollution concentration, û = wind speed, Q = emission rate). Classes A and F were omitted because they represent extreme conditions. In the UK classes A, C and E are most frequently observed. Although the extremes occur less frequently, sensitivity for these stability classes would have given some measure of the limits of applicability of these models and it would have been particularly useful to assess the performance under the classes, which are likely to lead to the most severe pollution episodes. The results gave some interesting insights into how the models responded to changing stability class, wind angle and distance form the highway and highlighted a problem in one of the models of discontinuous predictions when the model changed from using a parallel wind calculation to Gaussian distribution (Noll, 1978). The analysis gave an indication of why poor results occurred in certain situations.

The American Meteorological Society workshop of 1980 considered that sensitivity analysis with respect to both emission and meteorological data should be the final part of a model evaluation study (Fox, 1981). However Bullock *et al.* (1998) thought that comparison of model output with observations should only be carried out after sensitivity analysis had determined which input parameters are most crucial and what required degree of accuracy was required.

Statistical tests

Bias

A measure of difference. In a perfect model bias would be zero.

$$\text{Bias} = \bar{C}_p - \bar{C}_o$$

Fractional Bias

A measure of difference. Fractional bias is a dimensionless number therefore it can be used for comparing data categories with different concentration levels. Values range between -2.0 and +2.0, representing extreme over-prediction to extreme under-prediction.

$$FB = (\bar{C}_p - \bar{C}_o) / (0.5 (\bar{C}_p + \bar{C}_o))$$

Normalised Mean Square Error

A measure of correlation. NMSE is also dimensionless and in a perfect model would equal zero.

$$NMSE = \overline{(C_p - C_o)^2} / (\bar{C}_p \times \bar{C}_o)$$

Correlation Coefficient

A measure of correlation. Pearson correlation measures the extent to which two variables are linearly related. $r = 1$ with perfect positive correlation and and -1 with perfect negative correlation.

$$r = 1/n\Sigma [(C_p - \bar{C}_p) \times (C_o - \bar{C}_o)] / \sigma_p \times \sigma_o$$

Index of Agreement

An alternative measure of correlation recommended for model evaluation (Wilmot, C 1981 cited Karppinen, 2000). The index ranges from zero to 1 for perfect agreement between predicted and observed values. If the index of agreement is carried out using randomised datasets with identical means, the I of A varies between 0.39 and 0.41. Values must be above this limit to show agreement.

$$I \text{ of } A = 1 - \overline{(C_p - C_o)^2} / \; 1/n\Sigma [(C_p - \bar{C}_o) + (C_o - \bar{C}_o)]^2$$

Fraction of data within a factor of two

Indicates the proportion of predicted values that are within a factor of +/- 2 of the observed values

$$\text{FAC2} = \textit{fraction of data for which } 0.5 \leq C_p / C_o \leq 2$$

(Cox and Tikvart, 1990; Hanna, 1993; Karppinen, 2000)

Spearmann rank correlation

The Spearmann rank correlation is a nonparametric measure of the relationship between two sets of ranked values. It is used when data is not normally distributed. r = 1 with perfect positive correlation and and -1 with perfect negative correlation.

$$r_s = 1 - ((6\Sigma d^2)/(n^3 - n))$$

Mann Whitney

The Mann Whitney U test is a test of whether there is a significant difference between two sample sets of data. It is a non-parametric test. It is carried out on values that have been ranked out of the total set of data.

$$U_x = n_x n_y + (n_x(n_x + 1)/2) - \Sigma r_x$$

$$U_y = n_x n_y + (n_y(n_y + 1)/2) - \Sigma r_y$$

Where n_x and n_y are numbers of data points in each sample. U_x and U_y are the statistic compared with critical values in statistical tables and used to accept or reject the null hypothesis.

(Ebdon, 1999)

Definitions

C_p = Predicted concentration
C_o = Observed concentration
n = Number of data points
σ = Standard deviation
d = Difference in ranking
r = rank
Σ = sum of...