UNIVERSITY OF GREENWICH

School of Social Sciences and Law

LOCAL AIR QUALITY MANAGEMENT AND CLIMATE CHANGE: TOOLS FOR JOINED UP POLICY

by

Matthew P Ireland

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Abstract

All tiers of UK Government are required to include a range of environmental objectives in developing land use and transportation plans. The current trend towards regionalisation of governance brings with it uncertainty and debate. Part of this debate must include determination of the optimum scale for implementing land use and transportation plans to incorporate environmental objectives. This thesis is the result of developing tools to assist in this debate, using the environmental objectives relating to local air quality management and global climate change.

A major outcome of this research is the development of robust techniques and tools for the estimation of atmospheric emissions and subsequent air quality impact from land use, including industry, and transportation activities. An extensive number of evaluation techniques have been used to verify the robustness of these tools. These tools have been applied to a geographically diverse part of the UK representing a sub-regional level of governance and have been demonstrated to be applicable at the regional scale.

The tools have been used to provide a spatially detailed and source specific profile of emissions of key pollutants over a period from 1990 to 2020 allowing for the consideration of devolving the climate change objectives for reducing greenhouse gas emissions from the national to local scale. There is a balance between source size and the extent of geographical area (scale of governance) to ensure meaningful emission reduction measures can be implemented equitably. The tools have also been used to test the impact on local air quality of reducing emissions through assessment of a number of land use and transportation scenarios. The results indicate meeting climate change objectives will deliver more in terms of local air quality management than vice versa. There is expected to be a future limit to local air quality management in terms of reducing emissions and a need to reduce population exposure through land use planning. The issue of source scale may be overcome by adopting off setting policies.

The results of this research suggest the inclusion of local air quality management and climate change objectives in land use and transportation planning could be achieved at the regional or sub-regional scale, rather than local (district) scale of governance.

Acknowledgements

In the course of this research there have been many assistants both willing and unaware.

Most willing have been Liz Shier at Kent County Council, to whom I am eternally grateful for the tireless help in developing the geographical information system and keeping me alert to the progress of Southampton FC, and Professor Bernard Fisher as my academic supervisor. Bernard's ability to ask questions that do not appear immediately relevant but later reveal themselves as central to the subject in debate is a skill I can only aspire to. Although I have nurtured an ambition to attain a Doctorate in Philosophy since 1990 the key proponent of this work is Libby Street, formally of Kent County Council. I am indebted to Libby for telling me off in the first place for not having this qualification! I would like to thank my employers Mott MacDonald and the Kent and Medway Air Quality Partnership for supporting this research, the latter with European funding.

Those less aware of their assistance include the many friends and colleagues in the air quality field, particularly those associated with the National Society for Clean Air and Environmental Protection (NSCA). My understanding of local air quality management has advanced largely as a result of participating in the Air Quality Committee of the NSCA. I am more than grateful for the various opportunities the Society has provided for presenting my thoughts on incorporating climate change policies within local air quality management.

Finally, I would like to thank my family and friends for their support over the years, especially my mother and sister for proof reading, my two sons for diversion, my wife for her patience, and the quiz team for some essential rest and relaxation.

Dedicated to

Marcus B Adams

1965 - 2002

ante hoc finitus

Preface

The idea for this study evolved from the author's involvement with Local Air Quality Management. The author acknowledges sponsorship by the European Council funding, through Kent County Council under the INTERREG II Programme, and Mott MacDonald. A key outcome of the research has been to fully develop an Air Quality Management System, a stand alone strategic tool, which includes (coarse scale) air quality assessment of planning applications, a comprehensive (and fine scale) atmospheric emissions inventory and a dispersion model system integrated within the land use and transportation planning functions. It is expected the system could be developed by any regional authority. In the course of this research, the emissions inventory and model system has been applied to assess planning applications in Kent and Medway, develop the Local Transport Plan to address local air quality and climate change policies, and respond to government consultation on proposals for a major airport development in north Kent.

Research supervision was provided by Professor B. E. A. Fisher of the School of Earth and Environmental Sciences, and later the Environment Agency.

The author of this thesis has utilised a number of novel techniques to develop an atmospheric emissions inventory linked to land use and transportation activity data. The emissions inventory includes two pollutants associated with local air quality and one associated with global warming, is disaggregated by source category, both spatially to a fine resolution and temporally over a period of 30 years. The author has also developed a novel and efficient dispersion model system to provide estimates of local air quality. Both the emissions inventory and dispersion model system have been fully evaluated in the course of this research. These tools have been used to assess a series of land use and transportation policy options to demonstrate how they can be optimised to address both local air quality management and climate change.

The author has contributed to developing guidance on the application of local air quality management policy in the UK and is responsible for co-ordinating the principle UK forum for dispersion model users. He has presented and published the arguments developed during this research and its findings.

Publications by the author during PhD research

Journal Publications:

Ireland M P, Shier E, Smith M and Fisher B E A (2003) *Evaluation of Tools and Techniques for Rapid Estimation of Air Quality at the Local to Regional Scale*, in preparation for submission to Atmospheric Environment

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List of Symbols and Acronyms

ADMS	Atmospheric Dispersion Model System
AERMOD	American Meteorological Society / United States Environmental Protection Agency Regulatory Model
APEG	Airborne Particles Expert Group
AQAP	Air Quality Action Plan
AQMA	Air Quality Management Area
AQS	Air Quality Strategy for England, Scotland, Wales and Northern Ireland
ATDL	Atmospheric Turbulence Diffusion Laboratory model
CCP	Climate Change Programme
CERC	Cambridge Environmental Research Consultants
CNG	Compressed Natural Gas
^{o}C	degrees Celcius
DEFRA	Department for Environment, Food and Rural Affairs
DETR	Department for Environment, Transport and the Regions
DoE	Department of Environment
EMEP	Co-operative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe
EPAQS	Expert Panel on Air Quality Standards
ETSU	Energy Technology Support Unit
EU	European Union
GHG	Greenhouse Gases
GIS	Geographical Information System
HDV	Heavy Duty Vehicle
ITE	Institute of Terrestrial Ecology
KCC	Kent County Council
LAQM	Local Air Quality Management
LRC	London Research Centre
LPG	Liquefied Petroleum Gas
LTP	Local Transport Plan
$\mu g/m^3$	micrograms per cubic metre
mg/m^3	milligrammes per cubic metre
mbar	millibar
NAEI	National Atmospheric Emissions Inventory
NETCEN	National Environmental Technology Centre
NO_2	Nitrogen dioxide
NO_x	Nitrogen oxides
NSCA	National Society for Clean Air and Environmental Protection
OECD	Organisation for Economically Developed Countries
OPDM	Office of the Deputy Prime Minister
PC	Personal Computer
PM_{10}	Particles with a mean diameter of ten microns or less
$PM_{2.5}$	Particles with a mean diameter of two and half microns or less
PPG	Planning Policy Guidance Note
ppb	parts per billion
ppm	parts per million
R&A	Review and Assessment, with Stages I to IV

RCEP	Royal Commission on Environmental Pollution
TTR	Transport & Travel Research Ltd
TRICS	Trip Rate Information Computer System
UK	United Kingdom
UN	United Nations
UNEP	United Nations Environment Programme
WHO	World Health Organisation

1 INTRODUCTION

This Chapter provides an overview of environmental policy relevant to climate change and local air quality management highlighting the commonality in these themes. This overview is supported with more detailed description in the Appendix. Assessment criteria for climate change, in terms of reducing greenhouse gases, are identified. The roles of government tiers are compared, in the context of devolution from national to regional scales.

1.1 SUSTAINABLE DEVELOPMENT

1.1.1 The Earth Charter

The United Nations Conference on Environment and Development, held at Rio de Janeiro in June 1992, can be identified as the common origin for local air quality and climate change policies in the UK. The object of this Conference, often referred to as the Rio Summit, was to negotiate an Earth Charter marking out the principles for economic and environmental behaviour of peoples and nations. These negotiations produced (UN, 1992):

- the **Rio Declaration on Environment and Development:** a statement of 27 principles upon which the nations have agreed to base their actions in dealing with environment and development issues

- **Agenda 21:** a 40 chapter action blueprint on specific issues relating to sustainable development; requiring a consistent approach to both planning and environmental policy

- a **Statement on Forest Principles:** for the management, conservation and sustainable development of forests

- the **Framework Convention on Climate Change:** to stabilise greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system

- the **Biodiversity Convention:** for the conservation of biological diversity, the sustainable use of its components and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources.

The delegates of the Rio Summit agreed to reconvene after five years to consider progress on the above (Grubb *et al*, 1999). This meeting was held at Kyoto and is discussed further in Section 1.4.

1.1.2 A Better Quality of Life

During the Rio Summit, the UK Government signed up to the principles of sustainable development[1] and published its commitment soon after:

> *"The United Kingdom is determined to make sustainable development the touchstone of its policies. We recognise that this means a change of attitudes throughout the nation. That change cannot be achieved overnight, but gives no ground for defeatism – it should act as a spur for action."*
>
> (DoE, 1994a).

One of the recommendations for enacting Agenda 21 was for each country to produce a national sustainable development strategy. One of the first national strategies was produced for the UK in 1994, from which the above quotation is taken. This Strategy included a series of sustainable indicators including air quality. With respect to climate change, a target was agreed at the Rio Summit to stabilise greenhouse gas emissions to 1990 levels by the year 2000. The UK was one of only a few OECD countries to achieve this (DETR, 2000a; see also Table A.1. of the Appendix)[2].

Following consultation in 1998, a revised sustainable development strategy for the UK was published; *A Better Quality for Life* (DETR, 1999a). Two key changes from the original policy are the inclusion of the social dimension of sustainable development and the devolution of administrations in Scotland, Wales and Northern Ireland. The current extent of devolution was not considered likely to affect the management of local air quality as the Air Quality Strategy, along with associated legislation and guidance, described in Section A.2, was expected to be adopted wholesale by the devolved administrations (DETR, 2000b, Chapter 1) particularly given the merging of UK Air Quality Objectives with EU Air Quality Limits[3]. The effect of devolution raises the

1 *Sustainable development* is commonly defined as '*development which meets the need of the present without compromising the ability of future generations to meet their own needs*', after the Brundtland Commission (WCED, 1987). Although Cullingworth and Nadin (1998, p166) provide some useful alternatives, the subject of defining sustainable development remains an ongoing debate. The UK Round Table (2000) provides a practical definition, suggesting 'development' implies *change, progress, potential for improvement.* 'Sustainability' adds to that concept the idea of *durability.* This implies change must not only be economically viable but also environmentally and socially successful. In considering the UK planning system in terms of creating development plans and providing development control, the UK Round Table considers the former provides the framework for ensuring environmental and social success (i.e. public interest) with the latter ensuring the private interests of the developer do not prevail unreasonably over the public interest.

2 This target was also achieved in Kent and Medway; see Section 6.2.

3 DETR (2001b) has adopted of the EC Air Quality Framework Directive and First Daughter Directive

question of subdividing the Greenhouse Gas (GHG) emissions reduction target to individual countries and even further, to regions or local authorities. The potential for further devolution, to the regional level, is discussed further in Section 1.4.2. The implications of local to regional scale air quality management are considered beneficial to reducing GHG emissions (see Section 1.3). However, the potential for devolving GHG emissions reduction targets to the sub national level may prove problematic. As demonstrated in Chapter 6, the contribution from individual industrial processes to total GHG emissions becomes more dominant the smaller the study area, suggesting any measures adopted by a local authority, for example, would be completely overshadowed by emissions from a nearby power station. In contrast, a further example presented in Chapter 6 demonstrates the impact of switching freight from road to rail would have in improving local air quality and reducing greenhouse gas emissions.

DETR (1999a) states the UK Strategy for sustainable development is based on meeting four broad objectives:

- social progress which recognises the need of everyone
- effective protection of the environment
- prudent use of natural resources
- maintenance of high and stable levels of economic growth and employment.

The effective protection of the environment is defined in the Strategy as *acting to limit global environmental threats, such as* ***climate change****; to protect human health and safety from hazards such as poor* ***air quality*** *and toxic chemicals; and to protect things which people need or value, such as wildlife, landscapes and historic buildings*. This definition is useful in providing a high profile link between climate change and air quality.

Following initial work by the DoE (1996a) a set of some 150 sustainable indicators, including a subset of 14 headline indicators, have been developed to monitor the progress in achieving sustainable development. The indicators directly relevant to climate change and air quality are:

- emissions of greenhouse gases (headline indicator)
- carbon dioxide emissions by end user
- days when air pollution is moderate or high (headline indicator)

via the proposed Air Quality Limit Values (England) Regulations 2001 (H M Government, 2000a, 2000l). Importantly, the obligation for local authorities remains to *work towards* achieving the UK Air Quality Objectives (see Section 3.2) with the statutory duty for complying with the EU Limits residing with the Secretary of State. Evidently, similar proposed Regulations will be required for Wales, Scotland and Northern Ireland.

- concentrations and emissions of selected air pollutants
- sulphur dioxide and nitrogen oxide emissions.

These indicators could provide a means for monitoring the effectiveness of local air quality management and the climate change programme in the UK. However, guidance from the NSCA (Williamson, 2000b, 2001) on air quality action planning suggests both direct and indirect indicators should be used to monitor the effectiveness of AQAPs. As illustrated in Figure A.1 in the Appendix, year on year variations in air quality due to meteorological effects can mask expected improvements. The use of indicators such as traffic flow, for example, can assist in the process of consulting with non-air quality professionals and the general public as changes in traffic flow are more readily observed than changes in air quality. Note that the UK has been committed to reducing national emissions of pollutants since 1988. This is described further in Section A.1 and summarised in Table A.1 of the Appendix.

1.1.3 Local Planning and Pollution Control

In detailing the results of a 1995 survey of 36 Unitary Development Plans, Bruff and Wood (2000) consider the local government planning function to be significant in implementing sustainable development, citing the DETR's *Modernising Planning* agenda and the substantiation of this strategy as a policy goal for both local and regional planning bodies. With reference to the 1998 shift in government policy to greater inclusion of social progress, Bruff and Wood (2000) also refer to new initiatives such as *community planning* (to promote an over-arching framework of economic, social and environmental well-being based on full community participation) and *best value* (a new duty placed on councils to deliver services to clear standards based on cost and quality, and reflecting sustainable development principles). The Office of the Deputy Prime Minister (2002b) has provided guidance on this matter, requiring the integration of planning requirements within the community strategy or the best value performance plan where practicable and, more specifically, calls for the local air quality strategy to be subsumed under the Local Transport Plan if traffic is the main derivative of air pollution.

In title at least the planning guidance most relevant to this study is PPG23 : *Planning and Pollution Control* published by the Department of the Environment in 1997. The latest consultation draft of this guidance note, issued by the Office of the Deputy Prime

Minister (2002a) states the purpose and scope of the guidance specifically includes *ways in which the planning system can contribute in the longer term to improving air and water quality, and to reducing emissions of greenhouse gases.* In summary, the planning authority is required to consider both air quality and climate change (in terms of greenhouse gas emissions and impact on land use) as material for consideration. The high level link between air quality and climate change at national policy level has filtered through to being a significant consideration for local and regional planners.

PPG23 does not consider the option to off set emissions from one source by reducing emissions from another. This is the basic principle to emissions trading (see footnote 5, page 10) and has been successfully applied by the Environment Agency (1999) to UK electricity generators. In this case, operators of UK power stations are limited to a company ceiling for NO_x and SO_x emissions rather than individual ceilings for each station. Operators have the flexibility to decide which power station to operate. In all circumstances, local air quality criteria must be adhered to. Although the Environment Agency has stated that offsetting is not applicable under pollution control regulations (H M Government, 2000b) as they apply within the fence of the licensed site only (Powlesland, 2002) a precedent does appear to be in place. The use of planning gain, as Section 106 agreements under the Town and Country Planning Act 1990 (H M Government, 1990) is a similar concept to offsetting although, with respect to air quality management, there appears to be no examples of this mechanism being used in this manner except to require the installation and operation of air quality monitoring stations (Bull and Ireland, 1994).

1.2 UK AIR QUALITY MANAGEMENT

1.2.1 Concepts

Current central government policy on air quality management, as depicted in the AQS (DETR, 2000b) may be considered in terms of:

- reducing pollution emissions at source on a national scale
- setting Air Quality Objectives
- local management of pollution 'hot spots' – locations where achieving Air Quality Objectives is not expected.

The development of air quality management policy in the UK is described in Section A.2 of the Appendix. As discussed below, the first of these policy areas were developed as a

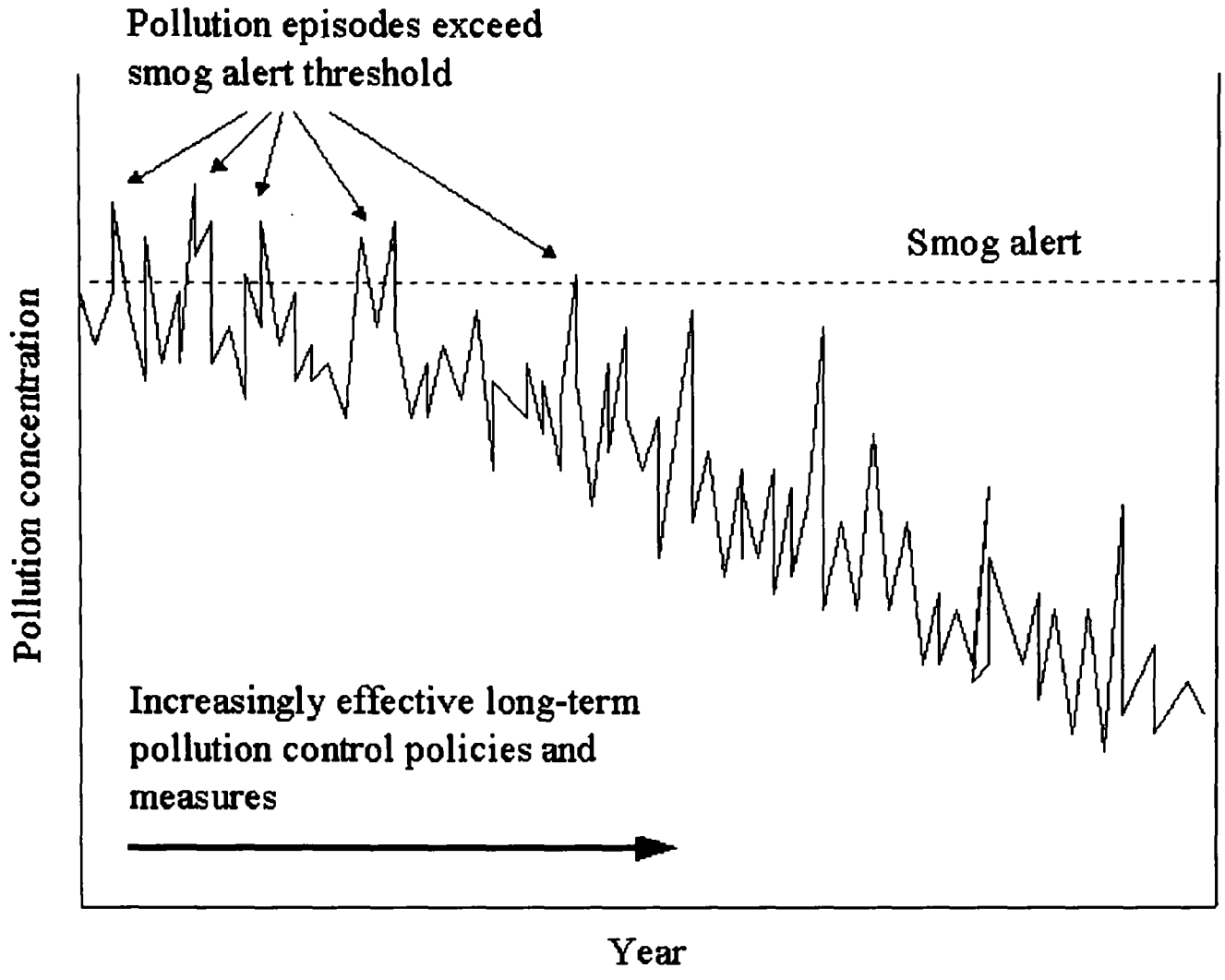
reaction to European Union Directives. In contrast, the setting of air quality objectives (quality standards to be achieved by some future date) and provision of local management is a concept developed in the UK and mirrored at the European level.

This concept of managing short term air pollution events by reducing long term average emissions is not new. Elsom (1996) for example, describes how this approach was used to tackle the London smogs by long term management of domestic coal burning and is used today to manage urban smog more characterised by complex photochemistry than sulphur dioxide and smoke. This is illustrated in Figure 1.1.

In the context of this research, this form of management is favoured as long term average concentrations of air pollution are considerably easier and quicker to model than short term episodes. Moreover, by managing local air quality using long term measures we are also providing a similar framework for managing the reduction of GHG emissions. The need for a rapid assessment tool should not be underestimated, as the development of an AQAP, involving the consideration of many management measures in varying degrees, also needs to be responsive to changes arising from obtaining committee approval for its implementation (see section 1.3). As described by Ireland (1998) Ireland and Fisher (1999) and Fisher *et al* (1999) and discussed further in Chapter 3, this method of assessment has required establishing relationships between long term means and short term episodes.

1.2.2 Local Air Quality Management

As prescribed in sections 82 to 84 of Part IV of the Environment Act 1995 (HM Government, 1995) local authorities have a key role in ensuring the AQS succeeds and have regard to guidance issued by central government. References to GHG emissions are included in this guidance, specifically with reference to transport (DETR, 2000g; paragraph 1.12) and in the context of Air Quality Strategies contributing to Local Agenda 21 initiatives (DETR, 2000f; paragraph 4.02).



Reducing baseline emissions over several years will eventually reduce the frequency of occasions when smog alerts are exceeded, although variability in weather conditions from year to year may obscure this trend for a while.

Figure 1.1 : The Long Term Reduction in Emissions Reduces the Frequency and Magnitude of Short Term Pollution Episodes

(Source: Elsom, 1996)

The overall process of local air quality management has been described and much reviewed by others from the basic elements (Elsom and Crabbe, 1996 Elsom, 1999; the regional implications raised by this paper are considered in section 1.5; and Elsom *et al,* 2000) to a focused overview of the statutory duties for UK local authorities by Murley (2002) with a comprehensive description of the air quality management process, as practiced in the UK, from EC Directives to Review and Assessment (R&A) by local authorities provided by Beattie *et al* (2001). In comparing the development and implementation of the UK AQS with the EC Framework Directive on Air Quality Management, Elsom (1999) considers the two strategic frameworks to be similar, with

the UK government proposing to implement the EC Directive with minimal legislative changes (DETR, 2001a, b). Elsom suggests the experiences of UK local authorities may prove useful to other member states, highlighting the need for guidance and support from central governments to be available in time.

Much of the review to date has focused on the requirements for the first round of R&A designation of Air Quality Management Areas (AQMAs) and subsequent implementation of Air Quality Action Plans (AQAPs). Beattie *et al* (2001) propose an ideal model for future air quality management practice in local authorities with the implementation of a long term (10 year) AQAP irrespective of whether an AQMA is designated. This proposal is in the same timescale as regional planning. The bottom half of Figure 2.2 illustrates the cyclic nature of air quality management. The process continues well beyond 2005 and, similar to land use and transportation planning, needs to become embodied within the ongoing workload of local authorities. Considering each element in turn (monitoring, emission inventories and modelling) a business management viewpoint can be adopted that the more effort required to assess a management scenario, in formulating an AQAP, the less likely it is to be adopted:

- too much detail in the earlier stages of AQM may slow the assessment of alternate AQAPs and therefore reduce the potential to assess comprehensively a variety of options for AQM

- too many resources and too much time spent on the review and assessment stages of air quality management could reduce the time and resources available for the development of effective AQAPs

- too much time and resources spent on the earlier stages may actually mean that the information and results collected/generated in the earlier stages are not available for inclusion in the development of AQAPs.

Whereas initial detailed R&A is required, the formulation of AQAPs requires a more streamlined approach to ensure inclusion of all relevant factors implicit in implementing an AQAP. This streamlining of the R&A process is possible because of the cyclical nature of local air quality management but only in conjunction with established ambient monitoring regimes. Techniques for achieving this are considered in Chapter 3.

1.3 CLIMATE CHANGE PROGRAMME

1.3.1 Policy Origins

The UK has been developing environmental policy addressing climate change since 1990 with the introduction of the non-fossil fuel obligation (NFFO) programme to encourage so called renewable forms of power generation, including wind and landfill gas turbines, energy crops and biofuels. The Climate Change Programme (CCP) was initiated formally in 1995 and, following consultation of a draft in 2000, was adopted later that same year (DETR, 2000a).

In contrast to the USA, the UK government has accepted the scientific view that climate change is occurring (see Section A.3 and Table A.2 in the Appendix); the introduction to the CCP states that some climate change is inevitable and anthropogenic contributions to greenhouse gases need to be curbed. The CCP considers measures to live with climate change (*reactive* policies) and measures to reduce GHG emissions and hence, minimise further change (*proactive* policies). This research focuses wholly on the *proactive* policies, some of which are described in Section A.4 of the Appendix.

1.3.2 Reducing Greenhouse Gas Emissions

Prior to the publication of the CCP, the Royal Commission on Environmental Pollution (2000) proposed a future energy policy for the UK with a focus on the climate change implications of not moving away from a carbon based economy. The Royal Commission concluded that UK climate change research to date suggests further global reductions of 60-70% below 1990 levels will be required within the next 50-100 years. This target is referred to in the CCP which includes a domestic goal for reducing CO_2 emissions by 20% below 1990 levels by 2010. The estimate of global reductions in GHGs required is considerably greater than the 5.2% reduction from developed nations agreed at Kyoto, suggesting that, even accounting for climate model uncertainty, much more needs to be done. Translating this target to the UK (a developed nation) suggests a domestic target in the order of 90% (DETR, 2000a; p9). Assuming no radical change in policy, this implies measures well beyond those included within the CCP will be required in the long term. Based on personal discussions with fellow practitioners, it is

generally recognised this will require a significant change in lifestyle for the UK population.

The CCP is based on implementing a series of sectoral measures aggregated to meet the UK Kyoto commitment. Based on existing policies, including increases in road fuel levy to 1999, climate change levy and meeting a 10% renewables target for power generation (introduced via the Utilities Act 2000, HM Government, 2000c) the UK is expected to cut GHG emissions by just over 13% and meet its Kyoto commitment of 12.5%[4]. The CCP acknowledges this achievement is mainly as a result of fuel switching in the energy sector during the 1990s[5].

As illustrated in Figure 1.2, UK GHG emissions are expected to increase from 2000 onwards, most significantly in the transport sector. Working from a 1990 = 100% baseline, the marked reduction in emissions by 2000 (14.6%) is primarily due to improvements in the energy and business sectors. The 33% reduction in GHG emissions from the energy sector is largely due to switching from oil and coal fired generation to more efficient and less carbon intensive gas fired plant. A similar (27%) reduction in emissions from the business sector is attributed to fuel switching to gas and the replacement of old equipment with new, more efficient plant. One significant reduction (3% of the UK GHG total) has been achieved by the single installation of emissions abatement technology to reduce N_2O emissions from a chemical works. The potential for one-off measures for reducing emissions from the energy and business sectors is now diminished with any further improvements expected from a diverse range of measures. Indeed, the forecast trends for the period 2000 to 2020 is a slight increase in emissions from these sectors. There has been no significant change in emissions from the domestic, agricultural, forestry or public sectors since 1990 and, as an aggregate, no significant change is forecast from these sectors up to 2020. In contrast, emissions from the transport sector have increased by 7% since 1990. This trend is expected to rise with

4 This target is actually the agreed target for the UK as part of the EU 'bubble' of 8%. The allocation of the EU bubble amongst member states, agreed in June 1998 (Grubb *et al*, 1999; pp122-124) is expected to be ratified in 2002; the tenth anniversary of the UN Framework Convention on Climate Change (DETR, 2000a; pp 32-33).

5 In proposing the European Climate Change Programme, the European Commission considers the UK switch in fuel technology is a unique event, similar to German Unification and unlikely to occur again. On a positive note, the EU has been successful in negotiating significant improvements in future road vehicle fuel efficiency from the motor manufacturing industry, is developing an EU wide renewable energy policy and consulting

an overall 35% increase in emissions by 2020. In summary, savings in the energy and business sectors since 1990 are expected to enable the UK to meet the Kyoto Protocol. However, the reductions achieved to date will be eroded, principally due to an increasing rise in emissions from the transport sector and only marginal changes in other sectors.

The CCP recognises further measures are required in the long term, not least because the domestic target of a 20% cut in CO_2 emissions will not be achieved on these projections. The CCP includes proposals for further measures. Many of these measures are unquantified or, at best, include a wide range of uncertainty. However, the CCP considers a 'realistic' view that these measures, if successful, are expected to cut GHG emissions to 21.5% below 1990 levels. Emissions of CO_2 are expected to fall from 13% to 17.5% below 1990 levels. The CCP optimistically concludes this shortfall in meeting the domestic CO_2 target will be met by as yet unquantified measures.

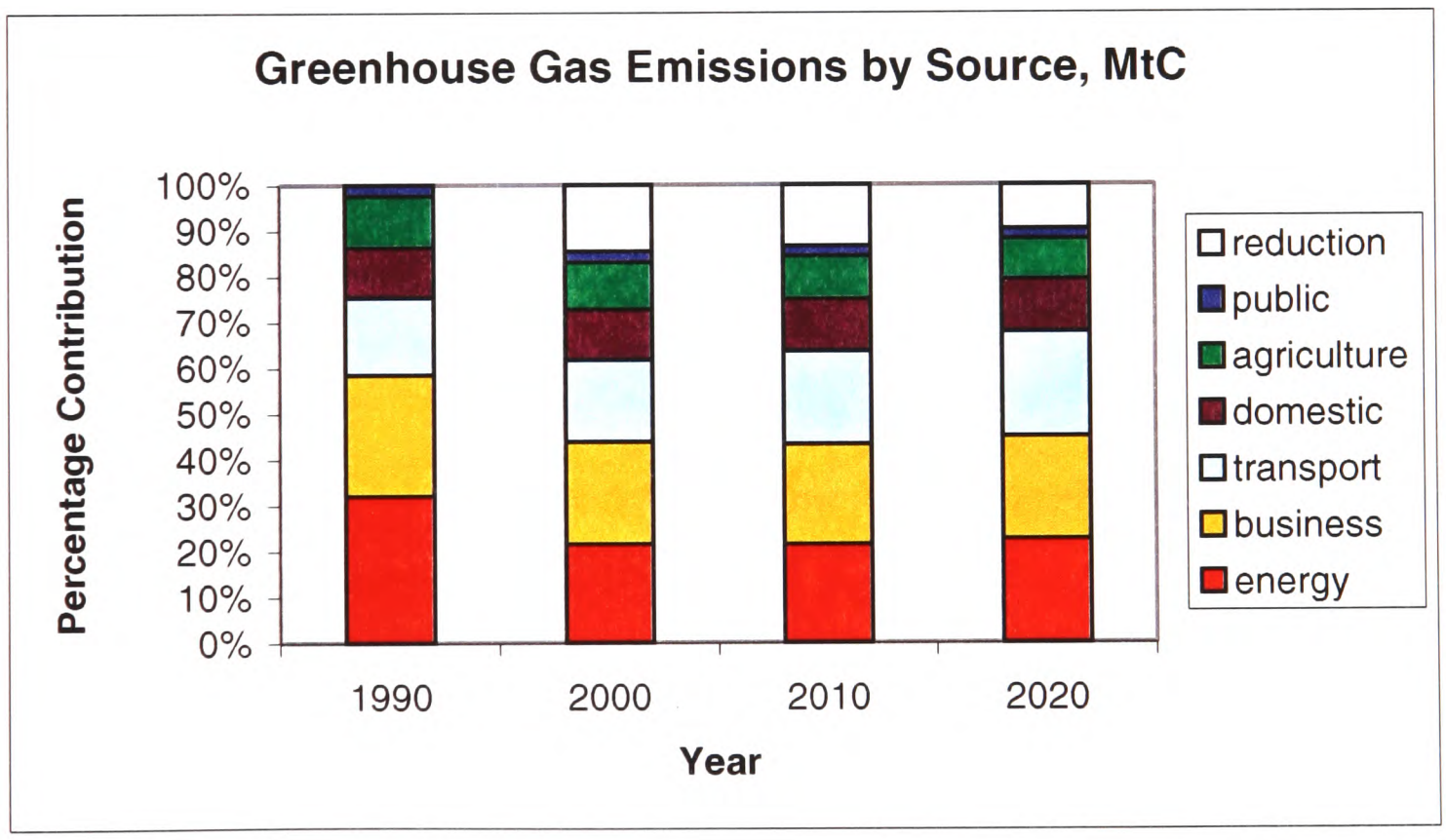


Figure 1.2 : Sectoral Contribution to UK Greenhouse Gas Emissions

(Source: DETR, 2000a)

on emissions trading of greenhouse gases (CEC, 2000a,b).

Although the CCP is based on a review of what can be achieved by each sector, it fal short of setting targets for each one. Emissions in the transport sector, particularly, roa transport, have risen since 1990 and are expected to continue rising beyond 2020 (se Figure 1.3). By 2010, national GHG emissions from road transport are expected to b 50% greater than in 1990. Even with the additional measures included in the draft CC emissions are expected to increase by 24% over the same period. More significantl and noted for its absence in the CCP, is the risk that savings made by the powe generators may be traded internationally, thus reducing the contribution to the UK total.

The CCP emphasises the UK target for reducing GHG emissions and goes on to confirr the agreement of the devolved administrations to work in partnership with the Ul government to meeting the both the Kyoto and domestic targets. Evidently, the adoption c these targets by the devolved administrations could occur in the future in a simila arrangement brokered at European level. The CCP also refers to the DETR wish for th implementation of Regional Sustainable Development Plans. One of the statutory purpose of these plans is to contribute to sustainable development, including climate change (an local air quality).

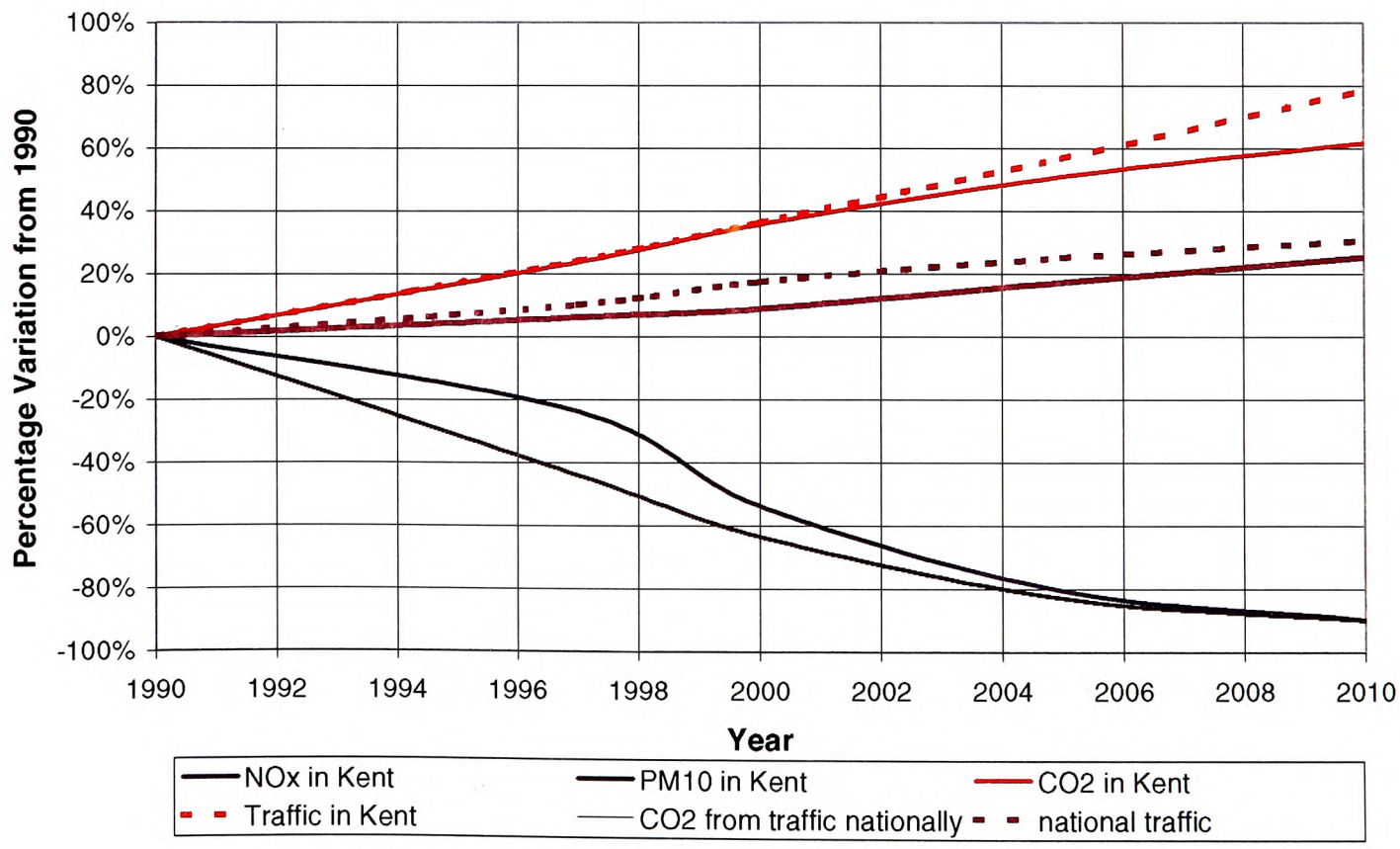


Figure 1.3 : Road Traffic Emissions in Kent

(Source: Kent County Council, 2000a)

1.4 DEVOLUTION AND ENVIRONMENTAL PLANNING

1.4.1 Tiers of Government and Responsibilities

The UK is administered through the enactment of laws passed through the Houses of Commons and Lords. With respect to environmental protection, these laws are usually translated into strategies or programme which are developed and implemented at a national scale, as prepared by the departments of central government, and cascade down progressively to a local scale (ie. to the district council); this is illustrated in Figure A.2 in the Appendix. The main tiers of government are described in Sections A.5 to A.10 of the Appendix, highlighting their current roles in implementing local air quality management and the CCP. The effect of devolution, in terms of devolving administration from UK Government to England, Scotland, Wales and Northern Ireland is not considered to significantly affect the conclusions drawn from this research as the relevant central government policies have been devolved as part of this process. However, the further devolution of governance to the regions is considered important.

1.4.2 Devolution to the Regions

In a review of regional policy and planning in Europe, Balchin and Sýhora (1999) present evidence of a trend towards devolution to the Regions[6] of Europe supported with reference to the underlying EU aim of creating economic cohesion in the EU. This economic policy is considered partially effective; the north/south divide across the EU is narrowing although full convergence is offset by regional differences. Supporting this policy are EU initiatives to create trans-European transport and telecommunications networks and to open up the gas and electricity supply industries with the aim of optimising opportunities for economic growth in all regions. In comparing the planning administrations of EU member states, Italy and Spain are presented as examples of fully devolved states with planning and legislative powers maintained at the regional level. The UK is considered a *devolving unitary state* similar in many respects with France, the Netherlands and Portugal.

In evaluating the impact of European Union membership on the planning activities of nine local planning authorities in England, Scotland and Wales, Bishop *et al* (2000)

provide a useful summary of the European Dimension to UK planning. This is summarised in Figure 1.4; the influence of EU on planning has been, so far, at the local rather than the national level. As noted also by Balchin and Sýhora, any direct EU influence on UK planning has been minimal to date despite the Maarstricht Treaty on European Union including explicit references to 'town and country planning' and 'land use'. However, many EU measures currently affect UK planning indirectly including, for example, environmental Directives and policies on transport, agriculture and regional development, in addition to direct funding (e.g. INTERREG[7]). Such funding is now directed via the newly formed Regional Development Agencies (RDAs). The RDAs themselves are considered to signify the third wave of regionalisation in the UK in the post war period and are far more significant because it is a UK wide programme of constitutional change (Morgan, 1999). Research by Allmendinger and Tewdwr-Jones (2000) highlight the uncertainty over both the future spatial dimension of planning processes and the scale links between the new regional level and existing national and local levels of governance.

Balchin and Sýhora (1999) consider the planning systems of EU member states will come under the influence of EU legislation[8]. The expectation is that a more structured 'integrated system' of planning (i.e. 'spatial planning') will be required at regional level directly linked to regional funding by the EU. This may lead to Regional Assemblies in the UK eventually attaining legislative powers[9]. Bishop *et al* (2000) consider the production of plans or strategies[10], other than statutory development plans, is influencing the statutory planning process through the provision of an additional context for

6 An example of a Region is the area covered by the Government Office for the South East of England.

7 INTERREG is an EU funding programme (provided via Annex 10 of the European Regional Development Fund) concerning transboundary regional development, of strategies and transnational links. For example, Kent County Council has received some £20.6 million funding through INTERREG I and II. One of the transnational links being developed between Kent County Council and Nord Pays de Calais concerns the monitoring, assessment and management of air quality (Bishop *et al*, 2000, Kent County Council, 1999).

8 Examples cited: the inclusion of town and country planning within the Maastricht Treaty, the proposed Directives *Assessment of the Effects of Certain Plans and Programmes on the Environment,* which will require all statutory land use plans to be subject to strategic environmental assessment, and *European Spatial Development Perspective*, aimed at supporting more balanced territorial development across Europe.

9 The use of spatial planning tools, such as a *Sustainable Development Strategy* is advocated by the UK Round Table on Sustainable Development (2000) who highlight the lack of integration between land use plans and public investment policies.

10 Referred to collectively as 'spatial planning' as opposed to UK 'planning' which refers quite specifically to the process of land use planning as pursued through the town and country planning system, ie. restricted to controlling development and the use of land, affected at a local level though the preparation of development plans and the granting or withholding of planning permission.

development plans, the joint development of policies and the establishment of new policy networks. In theory, this will assist in the rescaling of UK governance. However, Gibbs and Jones (2001) present evidence of continued tension between local and regional scale government, citing Local Agenda 21 as an example. The tools developed in this research will not resolve this issue but provide the means of testing land use and transportation policies to address air quality management and climate change objectives at various scales.

a period of *'splendid isolation'*	**a period of *'gradual reappraisal'***	**a period of *'broader based involvement in Europe'***	**a period of *'spatial planning'***
1945	1973	1986 (1991)	1998
Town & Country Pl. Act 1947 established system of land use planning based on preparation of development plans, providing a system of development control characterised by a high degree of discretion at both officer and political level. Contrasts with systems developed in most other European countries, which are based on legally binding zoning.	UK joins European Economic Community Gradual reappraisal and increased awareness of the transnational nature of economic and environmental policy issues.	Single European Act, 1986. A growing awareness amongst planners of how the European Commission operated and increased willingness to engage with Europe.	Originated in 1991 with the publication of *Europe 2000* but effected in the UK from 1997 with the change of government and 1998 with the ministerial statement *Modernising Planning*: (*"there needs to be a significant European dimension to our* [UK] *planning system"*)
'Europe' has little impact on planning in the UK	Impact of 'Europe' on the planning system is minimal but gradually increasing	Impact of 'Europe' on national planning is resisted but increases at local level	European policy vacuum at national and regional levels recognised.

Figure 1.4 : The Increasing European Dimension to UK Planning

(*after*: Bishop, Tewdwr-Jones and Wilkinson, 2000)

1.5 CONVERGING LOCAL AIR QUALITY MANAGEMENT WITH NATIONAL EMISSION REDUCTIONS

As described in Sections A.5 to A.10 of the Appendix, the statutory responsibility for meeting the GHG emissions reduction target currently lies with central government. At the other end of the scale, statutory responsibility for managing local air quality lies with the district councils. Devolvement of the UK target for reducing GHG emissions to the regional or sub-regional level is technically possible via the development plan route in combination with AQAPs and LTPs without the need for change in policy or guidance. Equally, the responsibility for local air quality management could be passed on through County Council and Regional Conference levels. Current policies for air quality management and the reduction of greenhouse gas emissions are both in line with the principles of Agenda 21 in this respect. Proposals for amending regional planning guidance include reference to undertaking environmental and sustainable appraisals of regional development plans with specific reference to the GHG emissions reduction target and the AQS (ODPM, 2000, 2002a).

Responsibilities for reducing GHG emissions could be considered to lie at all tiers of government considered. This partly contrasts with air quality management for which statutory responsibility lies with central government and the lowest tiers (ie. district council, metropolitan district council or unitary authority) only. This is illustrated in Figure 1.5 below. The optimum tier of government for combining GHG emission reduction targets with local air quality management will be a function of being sufficiently large scale for emissions from different sectors to be comparable and being sufficiently local for the implementation of air quality management measures. This suggests immediately that setting GHG emissions targets at the district level is not appropriate and implementing air quality management strictly in terms of the implied compliance dates included in the Air Quality Objectives at a regional level is not possible. One means of overcoming this is for pollution control and land use planning policies to adopt offsetting. This issue is considered in Chapter 6 with a series of land use and transportation scenarios.

The above argument may, in practice, be largely academic as policy makers focus more on ozone and fine particulates. For example, Cannibal and Lemon (2000) consider a

regional approach is essential for reducing tropospheric ozone formation. Similarly, the AQS (DETR, 2002b) clearly demonstrates the need for a regional scale approach to reducing particulate concentrations. In describing the evolution of air quality management policy and practice in the UK, Beattie *et al* (2001) consider the UK experience may act as a model for implementation of the EC Air Quality Framework Directive. Moreover, the results of survey work cited by Beattie *et al* (2001, 2000, 1999) suggest almost all urban authorities have formed regional groupings to consider management of local air quality. This suggests the current model of air quality management, with most focus on local 'hot spots' and compliance with air quality standards within a few years, will be replaced by a more regional framework with longer time horizons. This framework could, therefore, include greenhouse gas emission reduction targets as well as local air quality objectives.

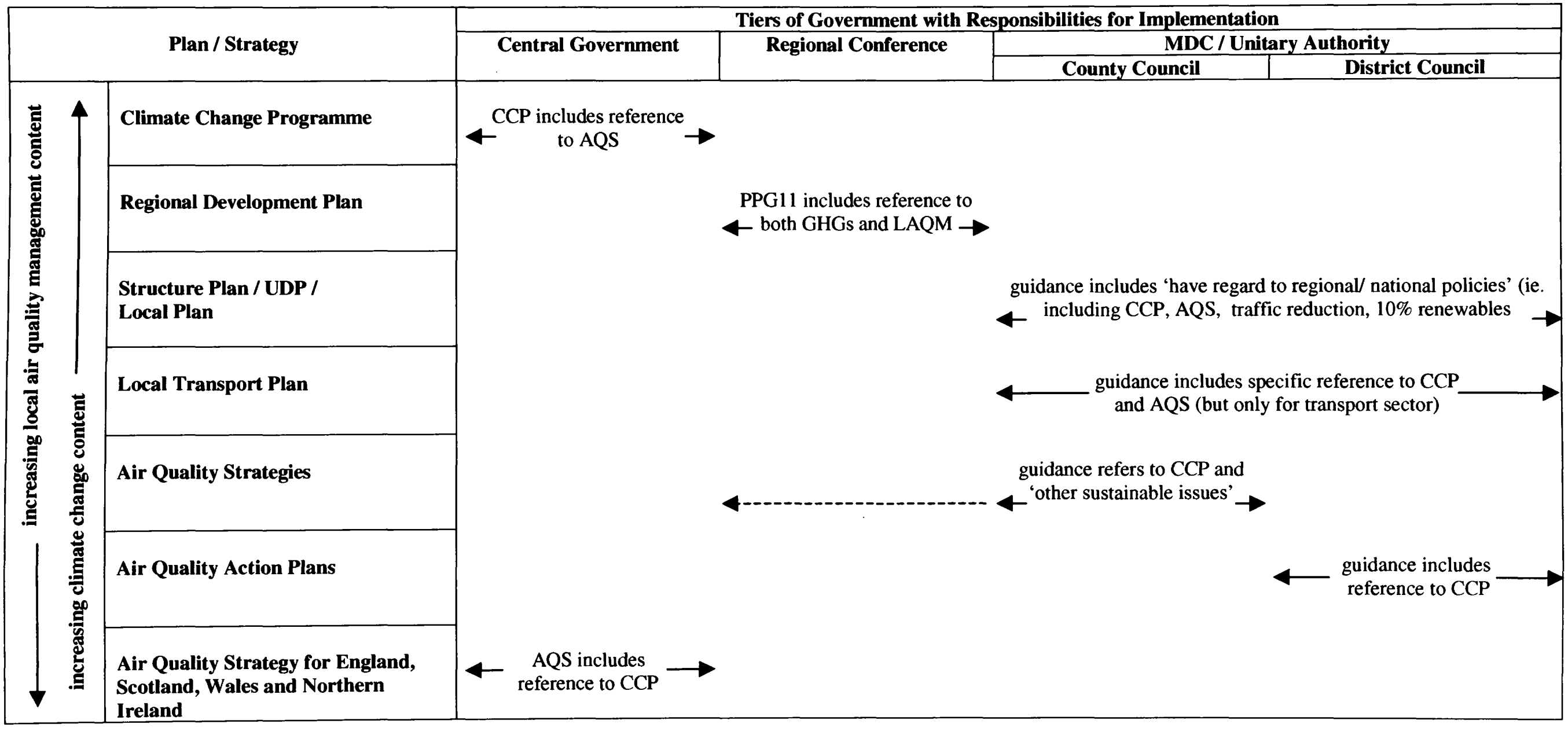


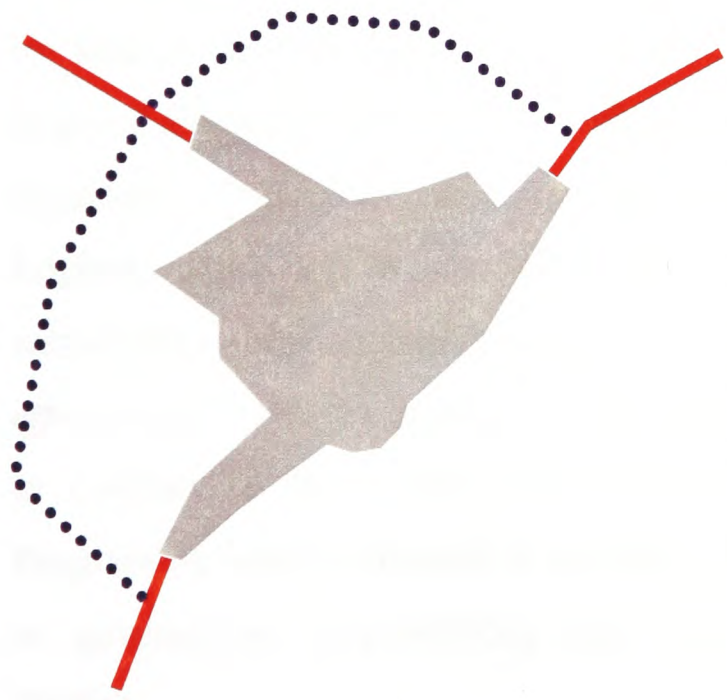
Figure 1.5 : Converging Local Air Quality Management with National Emissions Reductions of Greenhouse Gases
(Source : this work)

2 HYPOTHESIS AND METHODOLOGY

This Chapter provides some background to the research and, through discussion of a typical example of land use and transportation policy issues, sets out the hypothesis.

2.1 ORIGINS

This research originated from work undertaken in developing the Kent Air Quality Management System (KCC, 1995) at a time when local air quality management was becoming a statutory duty (HM Government, 1995) but prior to the development of a coherent national policy towards climate change, albeit now recognised as a key issue since the run up to the Kyoto Summit (DETR, 2000a). Discussion with local authority planning officers in 1996 regarding a proposed bypass highlighted the potential policy conflict between improving local air quality and reducing greenhouse gas emissions. This is illustrated in Figure 2.1. Table 2.1 summarises the potential for further conflict based on further discussion with the same officers.



A proposed town centre bypass (illustrated by the dashed blue line) would improve local air quality by reducing traffic volumes in the town centre (grey shading) but, by increasing the distance vehicles travel, would increase greenhouse gas emissions. Note: (1) the bypass alone would not ensure reductions in traffic volumes; complementary measures such as traffic calming or downgrading of roads is also required; and (2) the bypass provides new potential for development zoning from the existing urban fringe to the bypass itself. See main text for discussion of neighbouring towns.

FIGURE 2.1 : The Policy Conflict Between Improving Local Air Quality and Reducing Greenhouse Gas Emissions

(Source : this study)

Table 2.1

Air Quality Management Measures and Emission Reductions

Management Measure	Pollutant Emission Reductions							
	Local							Global
	Carbon monoxide	Benzene	1,3-Butadiene	Sulphur dioxide	Lead	Nitrogen Dioxide	PM_{10}	Carbon dioxide
Improved vehicle emission control	✓	✓	✓	-	-	✓	✓	✗
Promoting cleaner fuels	-	✓	✓	-	-	-	✓	✗
Road traffic reduction	✓	✓	✓	-	-	✓	✓	✓
Industry/domestic fuel switching (oil to gas)	✓	-	-	✓	-	✓	✓	✓
Domestic energy efficiency	✓	-	-	-	-	✓	✓	✓

Key:
✓ reduction in emissions
✗ increase in emissions
- not applicable or no significant change in emissions

(Source : this study)

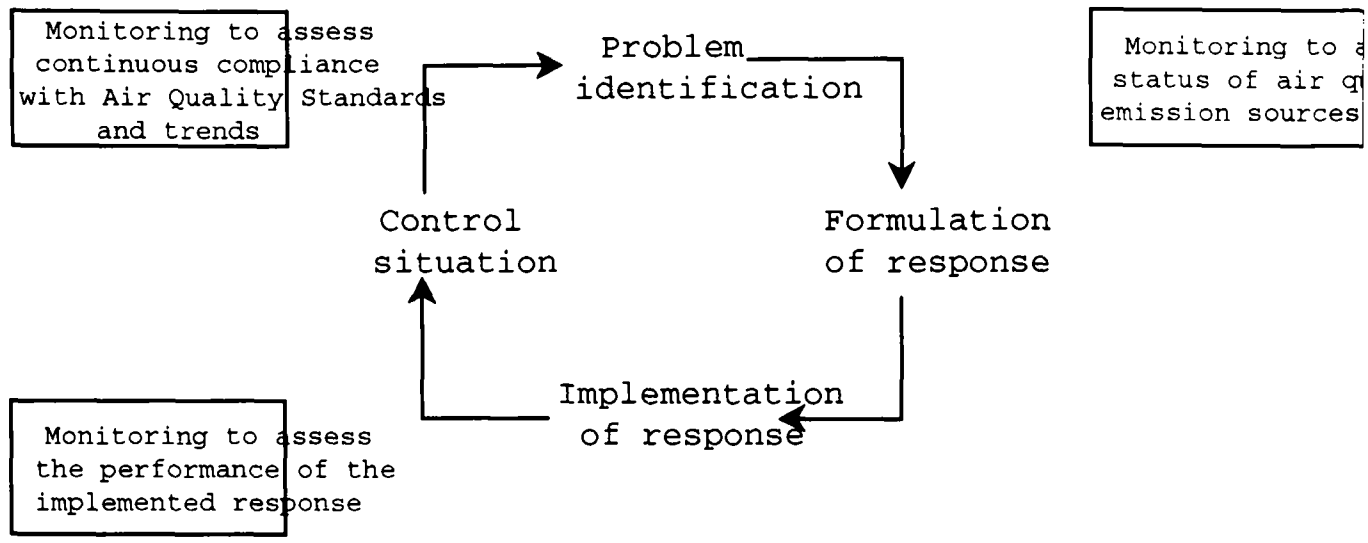
As described in Chapter 1 of this thesis, the common origin of local air quality management and climate change policy in the UK is the Rio Summit in 1992, the former being implemented by local authorities and the latter by national government. The need for local air quality management arises from the recognition that national measures to improve air quality are only cost effective to a point, leaving 'hotspots' that require local measures. One of the stated guiding principles of the UK Air Quality Strategy for England, Scotland, Wales and Northern Ireland (AQS) (DETR, 2000b) is that of subsidiarity. Subsidiarity is the principle that states *action should be taken at the lowest effective level of governance*. Local air quality management can be seen as 'bottom up' in contrast to the current 'top down' approach advocated in the Climate Change Programme which, although it includes a number of initiatives implemented at all levels of government, responsibility has not devolved from central government (DETR, 2000a).

Guidance for implementing local air quality management duties, as prescribed by Part IV of the Environment Act 1995, has evolved during the course of this research.

Membership of the Air Quality Committee of the National Society for Clean Air and Environmental Protection (NSCA) has provided early insight in to the development of the Air Quality Strategy for England, Wales, Scotland and Northern Ireland, Air Quality Regulations and associated guidance published by the Department for Environment, Transport and the Regions (DETR, 1997a-e, 1998a-d, 2000c-l, 2001a) and its successor the Department for the Environment, Food and Rural Affairs (DEFRA, 2002a-b). As the guidance evolved, the need to follow best business management practice, as described by Krajewski (1996) became clear. Making the process cyclical allows for continual improvement and the inclusion of new pollutants, such as polycyclic aromatic hydrocarbons. This is illustrated in Figure 2.2 and made implicit in the latest draft guidance issued by DEFRA (2002a).

The town illustrated in Figure 2.1 cannot be considered in isolation. Introducing traffic restriction measures in the town, for example, might lead to business moving to an adjacent town with less restriction. This is well recognised in local transport planning, which tends to be implemented at the county rather than district scale (Beattie *et al*, 2001). As road traffic is responsible for the majority of pollution 'hot spots' (Woodfield, 2001a-b) one conclusion from this is that local air quality management will not be implemented at the local (District) level beyond monitoring compliance and management performance; the three boxes in the lower part of Figure 2.2 may be considered equivalent to the Review and Assessment undertaken by the local authority. This conclusion is supported by the draft local air quality management policy guidance published for consultation by DEFRA (2002a) which states:

> *The Local Government White Paper , "Strong Local Leadership – Quality Public Services" sets out proposals to reform council services and enhance local democracy and community leadership in England only. Paragraph 4.30 of the White Paper, which was published in December 2001, states that "We (i.e. Her Majesty's Government) will no longer require the production of a separate air quality management action plan where an air quality problem arises because of transport pollution. Instead, councils will be free to address this through their local transport plan (LTP)".*
>
> (DEFRA, 2000a, paragraph 3.30)



The idealised WHO/UNEP approach developed from a study of the air quality management and assessment capabilities of 20 European cities (above) is compared to the cyclical process of UK local air quality management below, as described by current draft guidance. Each stage of Monitoring, indicated above, is synonymous with the rounds of Review and Assessment shown below. Similarly, the Implementation of Response is the Air Quality Action Plan, etcetera. Local authorities in the UK are currently on the right hand side of the cycle; formulating its response by consulting in the preparation of Air Quality Action Plans.

FIGURE 2.2 :The Cyclical Process of Local Air Quality Management

Source : top : *after* WHO/UNEP (1997), bottom : this study

This represents an opportunity for the timetable for compliance embodied within the Air Quality Objectives[11] to slip. More so, as '*Section 84(2)(b) of the 1995 Act makes clear that authorities are required to act "in pursuit of the achievement of air quality...objectives in the designated area". They are not under a legal obligation to achieve the objectives.*' (DEFRA, 2000a, paragraph 3.26).

As argued in Chapter 1 of this thesis, widening the geographical area (or level of governance) and extending the schedule of implementation for an AQAP could enable the existing local air quality management cycle to be co-ordinated with, or even incorporate, other policies, such as reducing greenhouse gas emissions, noise or any one or more of the 30 initiatives identified by the NSCA (Williamson, 2000a-b). However, in reviewing the progress made by highways authorities to incorporate air quality management within local transport plans, Beattie *et al* (2001) noted the diversity in progress, particularly given the different cycles for local transport plans and local air quality management, and the need for more explicit guidance. This appears to be similar to experience in the United States. For example, Chang (2000) cites Atlanta, Georgia, where successive air quality improvement programs with a five year planning horizon are not in step with the 20 year planning horizon, typical of industrial development and regional transport where measures to improve air quality will occur. This example of tension in planning is reflected by tension in UK governance as regionalisation progresses (see Section 1.4). Resolving these issues will be assisted by use of systematic or quantitative tools to assess the implications of including environmental objectives within land use and transportation policies. There is limited evidence in the literature of quantitative attempts to include more than one environmental objective within local transport planning. Two recent examples are Yang and Chen (2000) who describe a systems based approach to assessing local and regional air quality impacts of a number of transport management scenarios, and Spadara and Rabl (2001) who evaluated the damage costs of automotive air pollution from two case studies: one local (across Paris) and one regional (from Paris to Lyon). The second of these studies indicated damage costs for carbon dioxide are less than for nitrogen dioxide with particles having the greatest cost impact in urban areas.

11 Typically, air quality standards have to be achieved by 2003 – 2010, see Table 3.2.

2.2 THE STUDY AREA

The area included in this study is the county of Kent and the Unitary Authority of Medway within the south east region of England (see Figure 2.3). The study area is diverse with significant historic and projected change. The presence of major transport corridors, industry, small and larger conurbations, and contrasting rural parts suggests the area is expected to illustrate most of the issues arising from air quality management.

The area is a peninsula covering some 388,878 hectares and is surrounded by coastal waters on three sides. The southern shore of the Thames Estuary forms the northern coastline, into which flows the River Darent and the Medway. The east coast includes the Stour Estuary and the south east coast includes Romney Marsh and parts of the River Rother which flows into the sea at Rye (in East Sussex). The major landforms from north to south are: the Thames Basin; the North Downs; the Weald; and Romney Marsh (KCC, 1993, 1995). The study area is considered to represent a significant economic region to test potential policy impacts between LAQM and climate change.

In 1995 the combined population of Kent and Medway was 1.5 million of which 72% live in urban areas of over 10, 000 (KCC, 2000a). The major urban areas with a population over 100, 000 are: the Medway Towns (Rochester/Strood, Chataham, Gillingham); Thameside (Dartford, Gravesend/Northfleet); Maidstone and the Medway Gap; and the Thanet Towns.

The area is dissected by a number of nationally important transport corridors including the M25, M20/M26 and M2/A2, linking to the Continent. Industry, including a number of power stations, paper mills and cement works, are generally located in the north of the study area (KCC, 1995).

The continued long term strategy for the area is to stimulate economic activity and employment, capitalising on existing relationships with mainland Europe, with priority being afforded to the east and north of Kent (KCC, 1993).

Although the methodologies described in this chapter have been used to estimate emissions and dispersion at the *county* level their extension to the *regional* scale is

considered valid. By identifying suitable activity data commonly available to local authorities and using a GIS platform commonly used by local authorities, the methodologies developed for this study are considered transferable elsewhere. This is the subject of further research to extend the study area to include other authorities within the south east region of England and local authorities in Northern France and Belgium as part of the EU funded Interreg III Programme (KCC, 2002).

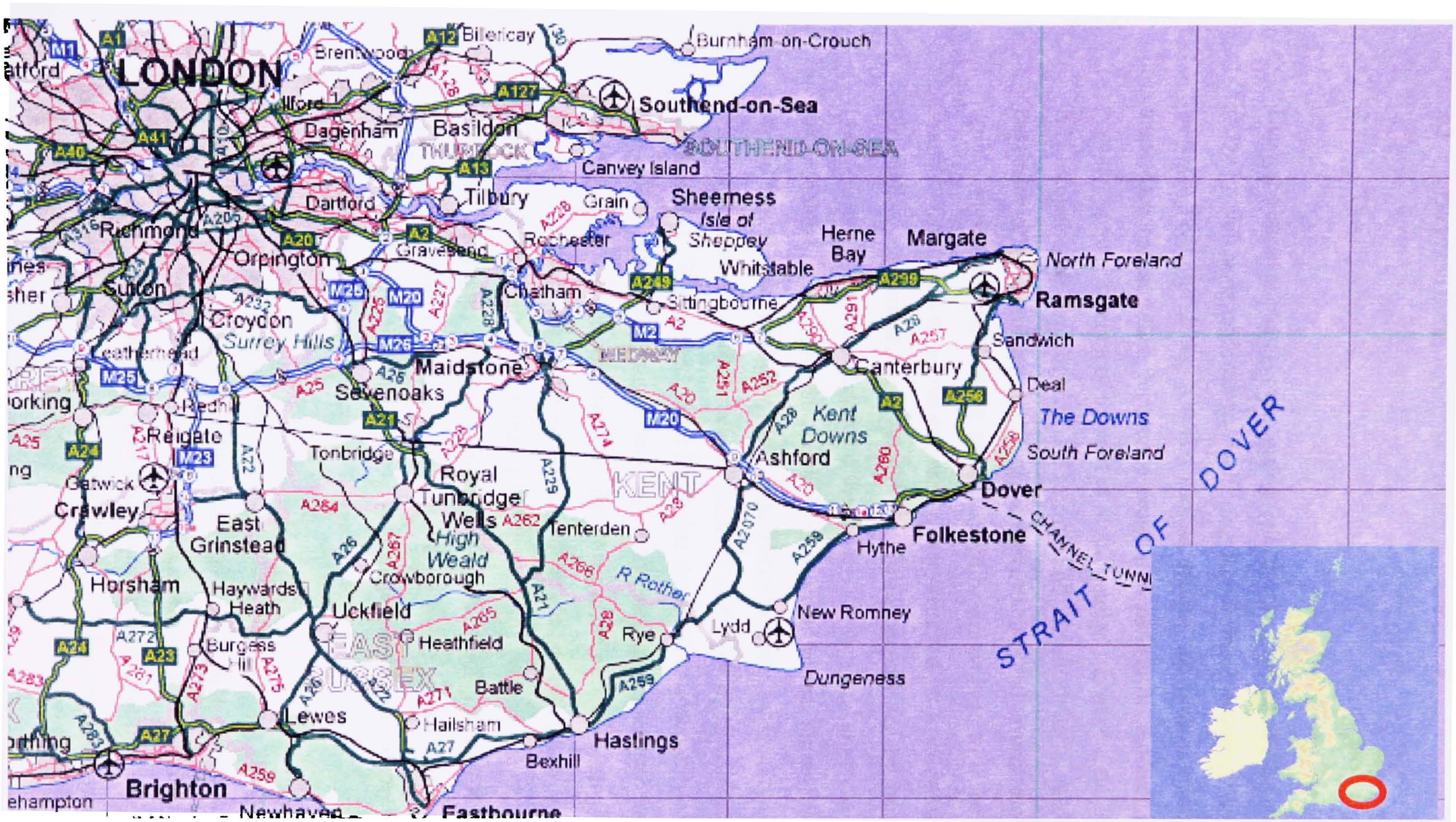


FIGURE 2.3 : The Geography of Kent and Medway

(Reproduced from Ordnance Survey map data by permission of Ordnance Survey. ©Crown copyright)

2.3 HYPOTHESIS AND METHODOLOGY

2.3.1 The Hypothesis

Using the business management analogy, the development of an Air Quality Action Plan (AQAP) can be considered as *evaluating management options*. If one considers a manager having to decide between various options, the time or resources required to study each option will influence the decision to include it. Another perspective is to consider options in terms of the cost of 'extra quality'. In most areas of manufacturing, it is recognised that quality can add to costs and that supplying a 'quality' in excess of the customer's requirements adds unnecessarily to costs and diverts resources away from other stages in the process. A parallel could be drawn here with the concept of Best Value[12]. One might argue the degree of quality required for a review and assessment of air quality is to simply comply with the guidance provided by central government (usually stated as the best defence if the results of review and assessment are questioned under judicial review). A less pragmatic view would be that the review and assessment should be sufficiently robust to determine whether certain Air Quality Objectives will or will not be achieved. Supplementary guidance developed by the Air Quality Committee of the NSCA (Williamson, 2001) highlights the inherent uncertainty of the review and assessment methodology suggesting this second view is less realistic for the purposes of land use and transportation planning. Returning to the business analogy and the cyclical process, one might argue the tools required should be sufficient to monitor or predict *changes* in air quality rather than provide definitive values. Furthermore, given the additional uncertainty attached to implementing Air Quality Action Plans, attempting to reduce to an absolute minimum any uncertainty in the review and assessment process may be erroneous if this delays actual implementation, simply representing an unnecessary burden on resources.

At the time of completing this research, local authorities are starting the process of developing Air Quality Action Plans or Strategies. In common with many of the initiatives referred to earlier, there is a need for public ownership of the plan or strategy in question through consultation and participation in pursuit of the principles of Local Agenda 21 (Williamson, 2001, DEFRA, 2002a). This implies the solutions to local air

12 See DLTR Best Value website: *www.local-regions.odpm.gov.uk/bestvalue/bvindex.htm*

quality are unlikely to be discrete actions but more a conglomeration of many initiatives and actions with varying degrees of effect dependant on community uptake. The level of inherent uncertainty in such community measures is well beyond that considered for the purposes of review and assessment, reinforcing the view that substantiating *change* is more realistic than providing precise *definitions* of air quality values. Murthy *et al* (1990) provides useful guidance here when discussing model validation; '*a model is adequate if it meets the requirements of the modeller – if it captures the salient features of the problem, and is capable of yielding a meaningful solution to the original problem*'. In summary, optimising the review and assessment methodology will allow resources to be better focused on evaluation and implementation of local air quality management options, i.e. tackling the problem rather than defining it to the n^{th} degree. In scientific terms this may be interpreted as recognising a degree of uncertainty and accepting there is an optimum at which attempts to reduce uncertainty further are not practical.

Linking the conclusions that local air quality management will not be implemented wholly at the District level, that short term deadlines inherent in the Air Quality Objectives are not critical and that the tools required should be sufficient to monitor or predict change leads to the hypothesis: ***sufficiently robust tools can be developed to test land use and transportation policies at a scale to improve local air quality whilst incorporating policies to reduce greenhouse gas emissions***. This hypothesis directly responds to the requirements of the proposed revision to PPG23: *Planning and Pollution Control* which states as one of its objectives to identify *ways in which the planning system can contribute in the longer term to improving air and water quality, and to reducing emissions of greenhouse gases* (ODPM, 2002a) and supports the high level UK policy links between local air quality and climate change described in Chapter 2 of this thesis.

2.3.2 GIS as a Platform

The decision was made to test the hypothesis using a Geographical Information System (GIS) platform to assist integration of the tools within day to day land use and transportation planning activities. GIS is now firmly established as a working tool within local government although its use in this field is likely to result in a shift in local

air quality management from environmental health to planning departments [13]. GIS has an important advantage in its ability to spatially check input data. As a planning control tool, the GIS used by a local authority needs to be accurate and hence, the base plan of roads, buildings or protected areas, for example, provides an accurate reference for checking the emissions inventory is spatially correct as well as providing the means for relating the results to land use and transportation planning. A review of work in compiling emission inventories for local and regional scale air quality modelling reveals a widespread use of GIS. For example: in Greater Manchester by Lindley *et al* (1996, 1998, 2000); Catalonia by Delgado *et al* (2000); Sacramento by Dai and Roche (2000); Lisbon by Borrego *et al* (2000); optimisation of an air quality monitoring network within an industrial area within the Basilicata Region of Southern Italy by Bonfiglio *et al* (1998); and Istanbul by Bozyazi *et al* (2000). Most of these examples used ArcView as the GIS platform. Both Dai and Roche (2000) and Lindley *et al* (1996, 1998, 2000) used proxy data to spatially disaggregate national, regional or county level emissions data to a finer resolution. Bonfiglio *et al* (1998) highlight the integration of land use data to both derive emissions and subsequent impact assessment; in this instance the effect of industrial emissions on areas of vintage cultivation. A further example is described by Briggs *et al* (2000) who used spatial regression to map traffic related air pollution in four urban areas.

Fotheringham and Wegener (2000) illustrate the wide application of GIS in spatial environmental modelling but argue GIS is not good for atmospheric modelling being poor at handling continuous phenomena and expressing simultaneous changes in multi-dimensional fields; atmospheric models requiring numerically intensive solutions to complex multi-variate systems of equations, and providing little support for dynamic modelling (feedback loops). The authors argue atmospheric models are '*well established in an aspatial, parameterised environment and unlikely to be reformulated in*

13 In providing an update to an original 1991 survey of all local authorities in Great Britain, Masser and Campbell (1996) noted 34.2% of all local authorities in 1991 reported to not having plans to introduce GIS whereas over 90% of county councils had GIS by 1993. They also note the increased provision of data by Ordnance Survey; suggesting greater uptake by local government using nationally consistent information although Shepherd (1996) reviewed the national availability of data including roads, rail, utilities and population, calling for a national database of infrastructure and settlement information. The planning and development departments of local government were most cited as controlling the GIS with one-third taking the lead in multi-departmental facilities. The GIS used in Kent is ArcView (ArcInfo) developed by Environmental Systems Research Institute Incorporated (1996). According to the survey previously cited, this is by far the most commonly used GIS in local government; in 1993 some 22% used this system. The next most popular system (MapInfo) was used by 7%.

an explicitly spatial context'. Methods for coupling GIS and models were considered by Fotheringham and Wegener, all of which are included in this study:

- ***isolated applications***; the use of a separate dispersion model to estimate ground level concentrations arising from Part A Processes (see Section 4.3.1)
- ***loose coupling***; the estimation of traffic emissions from major roads (see Section 4.3.3)
- ***tight coupling***; the application of the Matrix Model to estimate ground level concentrations arising from area sources (see Section 5.3)
- ***full integration***; the conversion of NO_x to NO_2 (although not strictly a spatially explicit function - see Section 5.6).

Moreover, this research includes a detailed and comprehensive review of inventory and model verification and validation studies to demonstrate the robustness of the methodology; see Sections 4.3 and 5.7.

2.3.3 Developing the Tools

The hypothesis has been tested by first developing the tools and then using them to assess a number of land use and transportation planning scenarios in the context of meeting local air quality and climate change policy objectives. The tools required can be considered in terms of assessment criteria, emissions inventories and dispersion models.

The review of UK policies relating to climate change and air quality management included in Chapter 1 identifies four targets for reducing greenhouse gas emissions[14], in terms of a future compliance date and a historical baseline, and two milestone years for the assessment of local air quality. With reference to the stated research hypothesis, the following years were selected for study:

- 1990: the base year for greenhouse gas emissions
- 1997: a scenario year for which ambient monitoring data are available for model validation

14 For the purposes of this study, no reference is made to the uncertainty of climate change. Paoli and Bass (1997) highlight the preference for policy makers to ignore uncertainty in issues such as climate change. Hence, for this study, reference is made to objectives for reducing greenhouse gas emissions only.

- 2000: a scenario year for which ambient monitoring data are available for model validation and a milestone year for greenhouse gas emissions
- 2005: a milestone year for local air quality
- 2010: a milestone year for local air quality and greenhouse gas emissions
- 2020: a reasonable horizon for regional land use planning, and the furthest point forward for which emissions data are available
- 2050: a long term planning horizon for reducing greenhouse gas emissions.

With the intention of ensuring the tools developed would be practicable, informal discussions were held with council officers and practitioners to consider how such tools would be used. This consultation confirmed the following conclusions:

- **emissions data** should be based on standard emissions factors and activities, and include a 1990 baseline and forecasts up to 2020. Use of recognised emissions factors would ensure consistency and enable comparisons with national data. The baseline year was chosen for greenhouse gas emissions. Forecast years included: current years, for comparison with observed ambient air quality; 2005 and 2010, for evaluation with air quality objectives; and 2020 being a reasonable horizon for regional scale planning with projections to 2050 desirable
- **activity data** should be based on information already available and maintained by the council. Primarily, this would ensure the workload of the council was not increased, ensure forecast data were available and facilitate dialogue with other policy makers (as, by definition, the data being collated and maintained by the council are for other purposes; Street and Ireland, 1997)[15]
- **dispersion modelling** should be based on recognised and defensible techniques. The results should be evaluated with reference to observed ambient levels. The modelling should enable rapid evaluation of scenarios to be assessed
- the tools should be **defensible** with implicit **accuracy** sufficient for long term land use and transportation planning on a regional scale.

These conclusions are in line with the principles of DETR (1997b-e, 1998a-d, 2000f-l, 2001a) and draft DEFRA (2002a-b) guidance on local air quality management and draft government guidance on planning and pollution control (OPDM, 2002a). For example, *'emissions data should be based on standard emissions factors and activities' and 'dispersion modelling should be based on recognised and defensible techniques'* is

15 The issue of resource and time constraints is highlighted by Lindley *et al* (1998, 2000) who also propose the use of *efficient and repeatable methods*. In testing the application of two dispersion models, Courthold and Whitwell (1998) highlight their use of model input data representative of what may be available. Secondary benefits to this approach include the ability to use census data

wholly consistent with local air quality management guidance. Specifying tools which *'are defensible with implicit accuracy sufficient for long term land use and transportation planning on a regional scale'* is considered in line with planning and pollution control guidance.

2.3.4 Testing the Tools

Having developed sufficiently robust tools, Chapter 6 demonstrates how they could be used to test land use and transportation policies. The spatial scale chosen is the same adopted by Kent County Council and Medway Council who are jointly implementing the Structure (land use) and Local Transport Plans, with time horizons of 10-15 and 5-10 years, respectively. A number of land use and transportation policy scenarios were assessed demonstrating the need for an integrated approach. Using the tools to analyse these policy scenarios leads to some useful conclusions highlighting the potential for measures to reduce greenhouse gas emissions to improve local air quality, the need to use land use planning to limit personal exposure and off setting as a mechanism for accommodating new development.

disaggregated by ward to assess population exposure (Jensen *et al*, 2001 and DEFRA, 2001).

3 KEY CRITERIA FOR ASSESSING UK AIR QUALITY

This Chapter provides the results of a detailed analysis of air quality criteria and identifies the key criteria for the assessment of local air quality.

3.1 AIR QUALITY MANAGEMENT TOOLS

As stated in Section 1.1 local authorities have a statutory duty to manage local air quality although guidance is limited in consideration of wider sustainability issues within this process. Nevertheless, there are various tools available in the UK for the development of local air quality action plans or strategies. These tools may be considered in two categories:

(a) Tools enabling air quality to be assessed:

- air quality objectives
- ambient monitoring
- emissions inventories
- air quality models.

(b) Tools enabling air quality to be improved:

- emission limits for individual sources
- total emission limit for an area
- air quality management plans
- management groups
- public information
- public liaison.

All of the tools enabling air quality to be assessed are implemented at the national level. The Air Quality Strategy (AQS) (DETR, 2000b) includes a number of air quality objectives. In its recent submission to the European Commission with respect to the Air Quality Daughter Directive, DETR (2001b) made use of national data from air quality monitoring networks and emission inventories, and surrogate spatial modelling techniques developed by Stedman (1998a-b, 1999a-b) Stedman *et al* (1998, 1999a-d, 2001a-c) and Stedman and Handley (2001). In meeting their statutory duties under Part IV of the Environment Act 1995 (HM Government, 1995) all district councils or equivalent in England, Scotland and Wales will have used these tools to a greater or lesser extent with those proceeding to a Stage III R&A requiring detailed application of monitoring, emission inventory and modelling tools (DETR, 2000l, 2001a and DEFRA,

2002b). With the exception of some metropolitan borough wide studies in, for example, the West Midlands (Birmingham City Council, 2000) and regional studies undertaken by the Environment Agency (2000a) there has been limited application of these tools at levels of governance between national and district council.

All of the tools enabling air quality to be improved are also implemented at the national level, as described in Chapter Two. With the exception of *total emission limit for an area*, all of these tools are being considered in developing AQAPs at the district council level or equivalent (DETR, 1997c, 2000f, DEFRA, 2002a, Williamson, 2000b, 2001). An aspect of this research, in coupling emissions reduction targets for greenhouse gases with local air quality objectives, provides some original investigation of the implementation of a total emission limit for an area at the local to regional scale.

This chapter provides a critical review of the relevant *ambient air quality objectives* and analysis of *ambient monitoring* data to identify the key criteria for assessment of air quality. Much of this work was undertaken during the beginnings of the first round of review and assessment of air quality by local authorities. The conclusions are supported by the results of local authority work in practice.

The development of an *emission inventory* and *air quality models* for application at a scale between national and local governance is described in Chapter Fours and Five.

3.2 AIR QUALITY OBJECTIVES

The setting of air quality criteria is the keystone to air quality management enabling the success or failure of any strategy or management plan to be measured. The AQS includes objectives for eight key pollutants, of which seven are included in the Air Quality Regulations (HM Government, 200a, 20010) [16]. The UK air quality objectives are actually comprised of air quality standards and objectives:

- an air quality **standard** is the level below which human health, flora and fauna is not harmed or amenity affected. For each pollutant detailed studies are required to assess at what concentration and over what time period these harmful effects will occur. The evidence available is not always consistent which can make agreement difficult on what level an air quality standard is set. This is

16 Due to be replaced by the Air Quality (Amendment) Regulations (DEFRA, 2002a).

compounded by the fact different pollutants may react in combination and be synergistic in effect.

- an air quality **objective** is the policy target for achieving the air quality standard, in terms of proportion of the year and by what date.

The ideal air quality objective should be based on the best scientific knowledge of the effects of the pollutant at a given level and also the political and economic reality of achieving that level within a given period.

The DETR (2001c) is consulting on the setting of Air Quality Bandings for the provision of information to the public. For the purposes of this research, the discussion is limited to air quality objectives only.

Many countries have established air quality standards based upon best scientific evidence but often including an element of watering down due to political pressure. An example of this was the relaxing of the US ozone standard in 1979 from 80 ppb to 120 ppb due to political concerns over the economic impact of compliance (Elsom, 1996 page 43). Such standards are often expressed in simple terms of concentration over a given time period (eg. 50 $\mu g/m^3$ as an annual mean) and it is not possible to separate out the political element in setting the standard.

In the UK most concern is addressed on the effects of air quality on human health[17]. The Department of Environment (superseded by DETR then DEFRA) set up an independent panel of UK scientific experts in air pollution and health effects referred to as the Expert Panel on Air Quality Standards (EPAQS). This Expert Panel is required to review available scientific evidence and recommend air quality standards for each of the key pollutants identified in the AQS. EPAQS (1994a-d, 1995a-b, 1996, 1998, 1999) has reported on all eight pollutants to date in addition to polycyclic aromatic hydrocarbons. The work of this Expert Panel is independent and to a large degree replicates the work of the World Health Organisation (WHO, 1984 and 1996) which made recommendations of acceptable levels for some 28 pollutants. The acceptance of WHO air quality guidelines within Europe (see comparison of WHO and EU limit values in Table 3.1) as well as by funding agencies such as the World Bank (1997) demonstrates a convergence of opinion suggesting the future work of the Expert Panel may be limited. However, the existence

17 Although there are UK Air Quality Standards for flora and fauna, this study is limited to health

of such an Expert Panel can be useful. In its revision of the 1987 guidelines, WHO was not able to set a definitive standard for PM_{10}; there being no safe limit, but produced a relationship between concentration and number of deaths. Such an approach is not practicable for the assessment and management of air quality. With some qualification, the Expert Panel was able to set an air quality standard for this pollutant enabling meaningful interpretation of monitoring data and the setting of targets and objectives.

Each UK objective is based on an air quality standard derived from a review of the health impacts of each pollutant, and a specific objective; a future date by which the air quality standard will be achieved for a proportion of the year. This is determined through consideration of the economic and political effort required to reduce current levels to below the standard. For some pollutants, such as carbon monoxide and 1,3 butadiene, the original NAQS published by the Department of Environment (1997a) suggests the standards for 2005 will be met by around the year 2000 based on current legislation and policies already in place. The revised AQS published by DETR (2000b) proposed bringing forward the date for compliance to 2003 for these pollutants. For other pollutants, such as nitrogen dioxide, the standard is unlikely to be achieved by the target year of 2005 unless additional measures are put in to place. This approach of setting *objectives* rather than *standards alone* was also adopted by the European Commission in setting limits for pollutants under the first Air Quality Daughter Directive (99/30/EC).

Air quality standards and limits are not necessarily applicable at all locations. The approach adopted in the UK is to limit public exposure to poor air quality. The standards and limits only apply where the public may be reasonably be expected to be present over the applicable averaging period. For example, air quality standards and limits based on the annual mean or averaging periods of 24-hours apply in the vicinity of housing, schools or hospitals whereas short term (1-hour or 15 minute mean) air quality standards or limits apply at all locations (DETR, 1997a, 1998d, 2000b, DEFRA, 2002a-b). The issue of personal exposure and the applicability of air quality standards and limits is discussed further in Section 6.5.

Local authorities have a duty to consider the UK air quality objectives in undertaking their local air quality management duties. The air quality objectives are not mandatory

related criteria.

(in contrast to the Air Quality Regulations 1989) although local authorities do need to demonstrate best intent in meeting them to comply with the Environment Act 1995. This includes, for example, industrial operators demonstrating *Best Available Techniques Not Entailing Excessive Cost* (BATNEEC). Setting air quality objectives nationally removes the need for local authorities to research and adopt their own criteria although these may have to be defended in court. Moreover, as air quality criteria are derived from epidemiological studies they should, theoretically, be consistent at least within the UK. Nonetheless, local authorities may refer to other guidelines such as those published by WHO or the EU.

The UK air quality objectives are considerably more stringent than the air quality standards implemented by the Air Quality Regulations (HM Government, 1989) in force when the NAQS was published and include for a further four pollutants. Whereas the now superseded UK air quality standards were based on EU Directives (80/779/EC, 82/884/EC and 85/203/EC for sulphur dioxide and particulates, lead and nitrogen dioxide, respectively) the objectives included in the original NAQS represented a departure from current EU thinking. The UK objectives were based on recommendations of health standards made by the EPAQS whereas the EU limit values are generally based on revised guidelines published by WHO (1996). The UK Government has ratified the EU Air Quality Framework and is actively negotiating on the various Daughter Directives setting EU limits for some thirteen pollutants. The revised AQS and Air Quality Regulations include additional objectives numerically similar to the published and proposed EU limits (see Table 3.1). Importantly, the statutory duty for complying with EU limits remains with central government and not local authorities. This was reiterated by the DETR (2001a) 'Stage IV' guidance. The introduction of further EU Daughter Directives is expected to extend the number of UK air quality objectives to include pollutants such as polycyclic aromatic hydrocarbons, cadmium, mercury and $PM_{2.5}$ (AQS, paragraph 137).

The air quality objectives are compared to the WHO guidelines and EU limits in Table 3.1. Ambient concentrations for gaseous pollutants are expressed in *parts per billion (ppb)* or *parts per million (ppm)*. The conversion to *micrograms per cubic metre ($\mu g/m^3$)* or *milligrammes per cubic metre (mg/m^3)* is assumed at reference conditions of 20°C and 1013 mb.

Table 3.1

Comparison Of WHO, EU and UK Health Based Air Quality Criteria

Pollutant	Averaging Period	WHO (1996)	EU Limit	1997 UK Objective (by 2005)	2000 UK Objective	Comments
Benzene	running annual mean	no safe limit designated (carcinogenic)	-	5 ppb (16.25 µg/m^3)	5 ppb by 2003	The EU has proposed a 1.5 ppb limit for benzene to be achieved by 2010. This is similar to the provisional UK target of 1ppb for 2010. DETR (2000b paragraph 153) expects the 2003 objective to be achieved but acknowledges further measures will be required to meet the 2010 objective at roadside locations.
1,3–butadiene	running annual mean	-	-	1 ppb (2.25 µg/m^3)	1 ppb by 2003	Neither the WHO nor EU have published or proposed a guideline or limit value for 1,3 butadiene. DETR (2000b para 160) expects the UK air quality objective will be achieved by 2003; no further objective is expected.

Table 3.1

Comparison Of WHO, EU and UK Health Based Air Quality Criteria

Pollutant	Averaging Period	WHO (1996)	EU Limit	1997 UK Objective (by 2005)	2000 UK Objective	Comments
Carbon monoxide	1-hour	25 ppm 30 mg/m^3		-	-	The UK objective, as a maximum rolling average, is more stringent than the WHO guideline. DETR (2000b paras 169-170) expects this objective to be achieved by 2003. The proposed EU limit of 8.6 ppm to be achieved by 2005 is slightly more stringent than the UK air quality objective.
	8-hour	8.6 ppm 10 mg/m^3		10 ppm (11.6 mg/m^3)	10 ppm by 2003	
Lead	annual	0.5 - 1.0 µg/m^3	0.5 µg/m^3 by 2005	0.5 µg/m^3	0.5 µg/m^3 by 2004 0.25 µg/m^3 by 2008	With the exception of a small number of specific industrial processes, DETR (2000b paras 180-182) expects the objective to be achieved although more stringent industrial emission limits may be imposed on a case by case basis.

Table 3.1

Comparison Of WHO, EU and UK Health Based Air Quality Criteria

Pollutant	Averaging Period	WHO (1996)	EU Limit	1997 UK Objective (by 2005)	2000 UK Objective	Comments
Nitrogen dioxide	1-hour	105 ppb (200 µg/m^3)	[a]105 ppb by 2010. 99.8th %ile	150 ppb (287 µg/m^3) Maximum	[a]105 ppb by 2010 99.8th %ile	The EU has adopted the WHO guideline. The 1997 UK 1-hour objective is less stringent than the EU limit but with a tighter deadline. The 2000 UK objective is in line with the EU limit but is phrased as provisional in the Regulations. DETR (2000b, para 195) recognises meeting the annual mean objective is expected to result in the 1-hour objective being achieved (see Section 3.3.3).
	annual	21-26 ppb (40-50 µg/m^3)	21 ppb by 2010	21 ppb	21 ppb by 2005	The annual mean criteria are the same for the UK, EU and WHO, although the UK objective is described as provisional in the Regulations.

Table 3.1

Comparison Of WHO, EU and UK Health Based Air Quality Criteria

Pollutant	Averaging Period	WHO (1996)	EU Limit	1997 UK Objective (by 2005)	2000 UK Objective	Comments
Fine Particles (PM_{10})	24-hour	no safe limit designated	[b]50 µg/m^3 by 2005, 90th %ile [b]50 µg/m^3 by 2010, 98th %ile	50 µg/m^3 99th %ile	[b]50 µg/m^3, by 2005, 90th %ile	WHO has not published a safe limit for PM_{10}. The EPAQS recommendation is based on a reasonable level of risk (an increase of one in 1000 deaths during a pollution episode) and recognition of the need to set a Standard for monitoring purposes. EU limit for 2005; now adopted as the UK 2000 objective. Differences in monitoring methods (DETR. 2000b, paras 247-248) suggest the UK objective is less stringent than the EU limit.
	annual	no safe limit designated	40 µg/m^3 by 2005 20 µg/m^3 by 2010	-	40 µg/m^3 by 2005	The EU has proposed Stage ll limits to be achieved by 2010. The number of allowable exceedances of the 24-hour limit will be reduced to seven times per year. The annual limit will be reduced to 20 µg/m^3; see Figure 3.1.

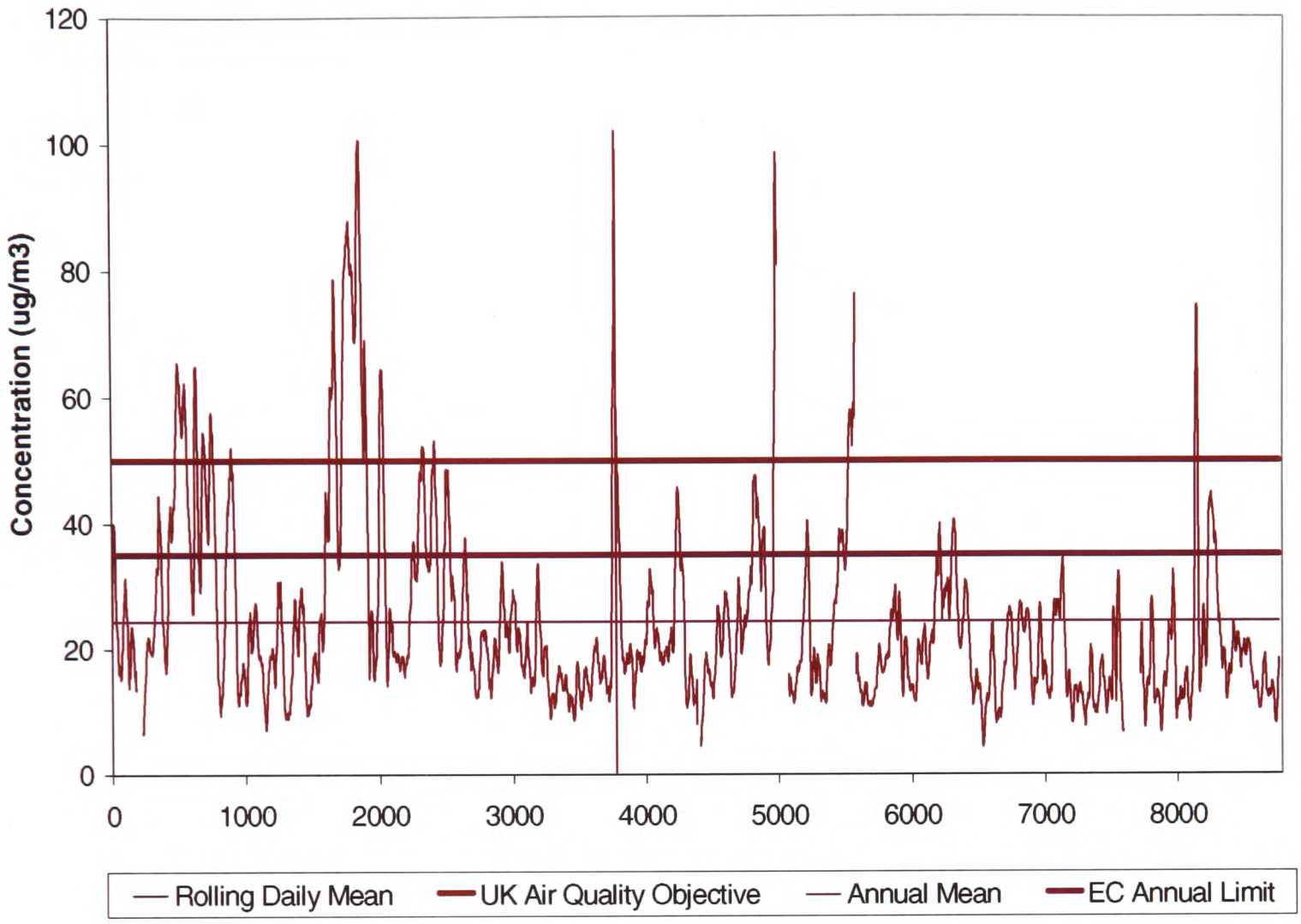
Table 3.1

Comparison Of WHO, EU and UK Health Based Air Quality Criteria

Pollutant	Averaging Period	WHO (1996)	EU Limit	1997 UK Objective (by 2005)	2000 UK Objective	Comments
Sulphur dioxide	10 minute	188 ppb (500 µg/m^3)	-	-	-	The criteria for sulphur dioxide published by all three bodies are similar and, between them, provide a range of standards for different averaging periods from 10 minutes to one year.
	15 minute	-	-	-	100 ppb, 99.9th %ile by 2005	
	1-hour	132 ppb (350 µg/m^3)	[c]132 ppb, by 2005	100 ppb (286 µg/m^3) 99.9th %ile	[c]132 ppb, by 2005	
	24-hour	47 ppb (125 µg/m^3)	[d]47 ppb, by 2005	-	[d]47 ppb, by 2005	
	Annual	50 µg/m^3	-	-	-	

Notes to EU Limits :

(a) Not to be exceeded more than 18 times per calendar year
(b) Not to be exceeded more than 35 times per calendar year, using a high volume sampler. The TEOM equivalent value is 35 µg/m^3
(c) Not to be exceeded more than 24 times per calendar year
(d) Not to be exceeded more than 3 times per calendar year



Work by the Airborne Particles Expert Group (1999) suggests local management of PM_{10} to meet the annual mean limit will not necessarily result in achieving the 24-hour UK objective or EU limit because of secondary contribution from external sources. Such sources are evidently outside the control of local authorities; the annual mean limit may prove to be a more practical objective for local air quality management. This is illustrated above for Bexley during 1996. The annual mean concentration was 24 $\mu g/m^3$; well below the EU limit of 40 $\mu g/m^3$ for 2005 and close to the proposed EU limit of 20 $\mu g/m^3$ for 2010. The UK air quality standard was regularly breached with the 99th percentile being 80 $\mu g/m^3$ for that year.

Figure 3.1 : Observed Particle Levels : Bexley 1996

(source: compiled from data in the National Air Quality Archive, *www.air-quality.co.uk/*)

3.3 AIR QUALITY MONITORING

The measurement of air quality in the UK is based on a small number of relatively standardised techniques. As described by DETR (1998a, 2000i) and DEFRA (2002b) guidance, these techniques range from the relatively simple and cheap diffusion tube to the more sophisticated and considerably more expensive continuous (or automatic) analysers. Diffusion tubes can be used to measure fortnightly or monthly mean concentrations of nitrogen dioxide, sulphur dioxide and benzene. Continuous analysers are able to measure short term mean concentrations (collated in aggregates of 15 minute means) of all the priority pollutants except lead. The monitoring of lead is undertaken using the M-type sampler. This consists of a filter attached to a sampler head (specially designed to minimise air flow across the filter) through which air is drawn via a pump. Samples of lead are taken as monthly means.

3.3.1 National Monitoring Networks

DEFRA (2002b) funds a number of nationwide air quality monitoring networks throughout the UK. These are now organised into the Advanced Urban and Rural Network (AURN) with 105 monitoring stations, and six sampler-based (non-automatic) programmes including over 1200 nitrogen dioxide diffusion tube sites (Clark *et al*, 1999). The networks include monitoring stations measuring one or more pollutants from a comprehensive range including:

automatic

- carbon monoxide
- benzene
- 1,3 butadiene
- particles (PM_{10} and $PM_{2.5}$)
- nitrogen oxides (NO_x; NO_2 and NO)
- sulphur dioxide
- ozone

non-automatic

- smoke and sulphur dioxide (as total acid gases)
- nitrogen dioxide
- lead
- acid deposition
- toxic organic micro pollutants (TOMPS).

All the priority pollutants are included within the national networks. However, these networks were not originally developed with local air quality management in mind although subsequent changes have occurred, including the addition of a number of

roadside sites. The objectives of the national monitoring programme are stated as follows:

- to understand air quality problems so that cost effective policies and solutions can be developed
- to assess how far standards and targets are being achieved
- to provide public information on current and forecast air quality
- to assist the assessment of personal exposure to air pollutants
- to support local authorities in their review and assessment of air quality
- to check compliance with existing European Community Directives.

Ireland (1998) identified some 19 roadside monitoring sites within the AURN suggesting they would provide useful data for model validation if correlated with road traffic data. This work has been progressed by Laxen *et al* (2001, 2002) with Laxen and Wilson (2002) publishing a new approach to deriving NO_2 from NO_x for air quality assessments of roads.

3.3.2 Local Monitoring Networks

In addition to the AURN and non-automatic networks, a number of local authorities co-ordinate their own monitoring activities. Bull and Ireland (1994) observed the growing trend by local authorities to use Section 106 agreements attached to planning permissions to reinforce local monitoring regimes. Examples of locally co-ordinated monitoring include:

- The London Air Quality Monitoring Network
- The Welsh Air Quality Forum
- The Kent and Medway Air Quality Monitoring Network.

The London Air Quality Network is considered to be the first large scale co-ordination of local authorities to construct and maintain a number of automatic monitoring stations. The project was initiated in 1992 by the South East Institute for Public Health. The London network currently includes a total of over 100 pollutant monitors stationed at 67 locations around London. Air quality data for London, updated daily, are available *on line* at www.erg.kcl.ac.uk/london/asp/home.asp (Beavers *et al*, 1999).

The Welsh Air Quality Forum was initiated in 1994 by the All Wales Chief Environmental Health Officers Panel. The first report of the Welsh Forum (1996) includes the results of 387 monitoring locations in Wales maintained by 31 of the 37 local authorities in the Principality.

The Kent and Medway Air Quality Monitoring Network was launched in late 1997 following recommendations made in the First Report of the Kent and Medway Air Quality Partnership (KCC, 1995). The network currently includes twenty automatic monitoring stations and over 180 non-automatic sampling sites. The network is maintained by SEIPH under a contract managed by the Association of Chief Environmental Health Officers in Kent and Medway (Barratt, 2000).

3.3.3 Results of Pollutant Specific Monitoring

Benzene

The AURN includes 12 automatic stations for benzene. The majority of local authorities do not regularly undertake routine monitoring of this pollutant and, if they do so, the monitoring is limited to benzene diffusion tubes. Data from automatic stations are available as 1-hour averages whereas the diffusion tubes provide monthly averages. The latter can be aggregated to provide a fixed annual mean but cannot provide a running maximum annual mean; the statistical descriptor of the air quality objective, which is slightly more stringent.

Using monitoring data from the national archive, Figure 3.2 provides a regression analysis of fixed versus running maximum annual means[18]. These results can be compared to analysis published by Stedman and Dore (1999a) who also report additional unpublished data:

$$\textbf{[running maximum annual mean]}_{ppb} = a * \textbf{[fixed mean]}_{ppb} + c \qquad \ldots.(1)$$

where:

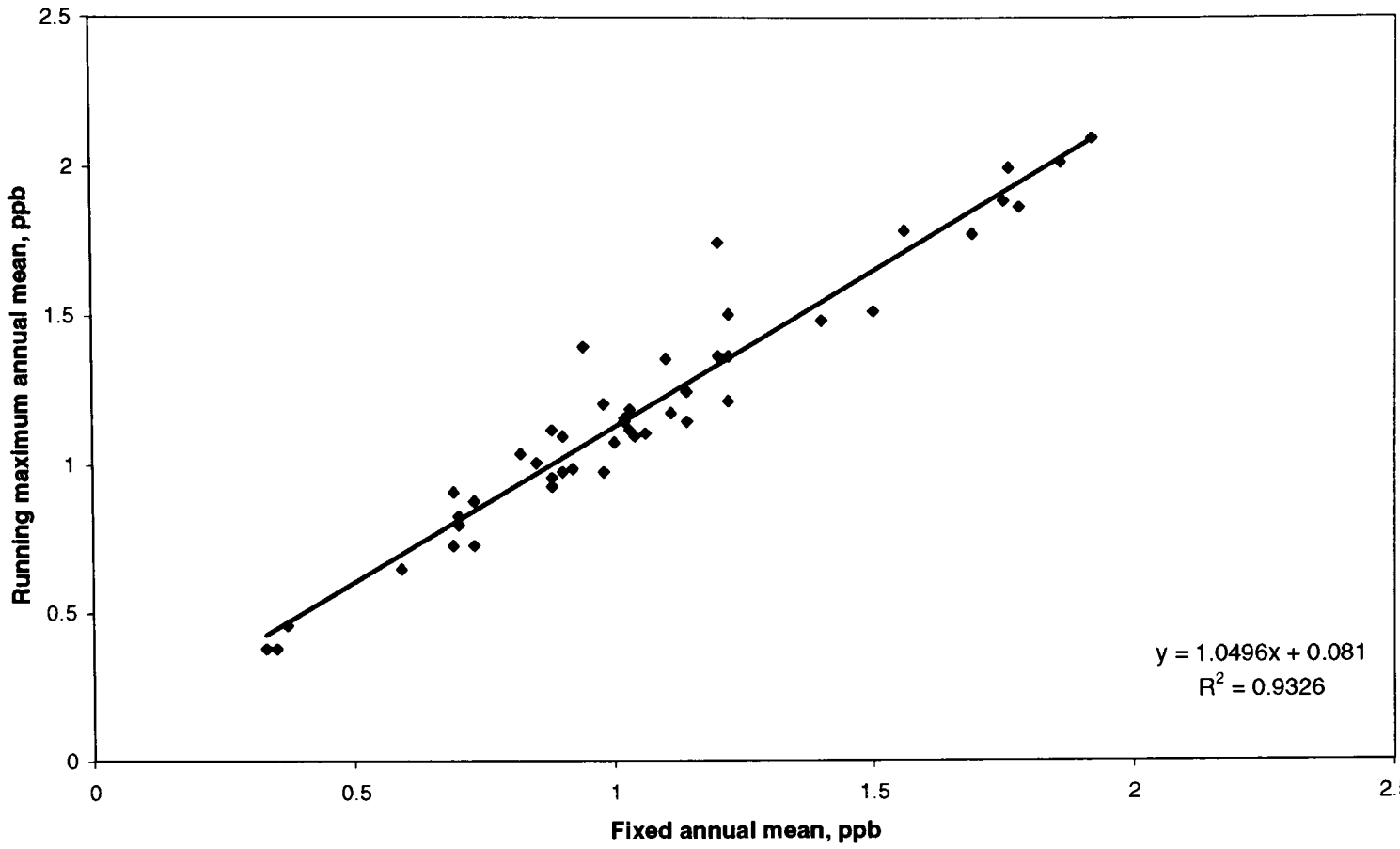
a	c	R^2	Reference
1.0496	0.081	0.9326	this work
1.102	0.004	0.960	Stedman and Dore (1999a)
1.1015	0.0198	0.9328	NETCEN (unpublished; cited by Stedman and Dore)

(95% confidence)

Forcing the regression to intercept = 0, yields a simple factor of 1.12 with little loss in the degree of correlation (R^2 = 0.9282, 95% confidence). DETR (1998a, 2000i) and DEFRA (2002b) suggest the use of benzene diffusion tubes is an acceptable method for R&A within a tolerance of +/- 30% of the air quality standard. On the presumption benzene levels are in decline, using this degree of tolerance suggests the air quality standard for 2005 will be achieved if the results of diffusion tube monitoring indicate current levels are below 3.5 ppb. Stedman and Dore (1999a) report monitoring data from 28 roadside locations across the UK of which 11 were above 3.5 ppb. National modelling of roadside benzene concentrations by Stedman (1999a,b) and Stedman and Dore reported in the AQS (pp A112-A115) indicate the air quality standard will be achieved by 2002. This work also considered the provisional UK standard of 1 ppb to be achieved by 2010 concluding this would also be achieved with the exception of some 25 road links in London and the M8 in Glasgow (50 links reported in the AQS).

The above work suggests the additional measures required to manage benzene levels in the UK are limited to a small number of roads and some industrial premises. Diffusion tubes offer a suitable means of monitoring the expected continued decline in benzene levels and can be used to assess air quality with reference to both the UK air quality objective and proposed EU limit.

18 In this and subsequent analyses presented in this Chapter a simple linear regression has been used in accordance with usual practice (see references in main text as appropriate).which assumes the *x* values are 'correct' by minimising the sum of the squared deviations in the *y* direction. Work by Ayers (2001) would suggest a more correct approach would be Reduced Major Axis (RMA) regression which minimises the product of the *x* and *y* deviations from the fitted line, ie. assumes neither *x* nor *y* values



The AQS reports fixed and running maximum annual mean concentrations of benzene observed over the period 1994 to 1998 at 13 sites including one influenced by industrial sources. The analysis presented above excludes the industrial site and includes 47 pairs of data.

Figure 3.2 : Relationship Between Fixed and Running Maximum Annual Mean Benzene Levels

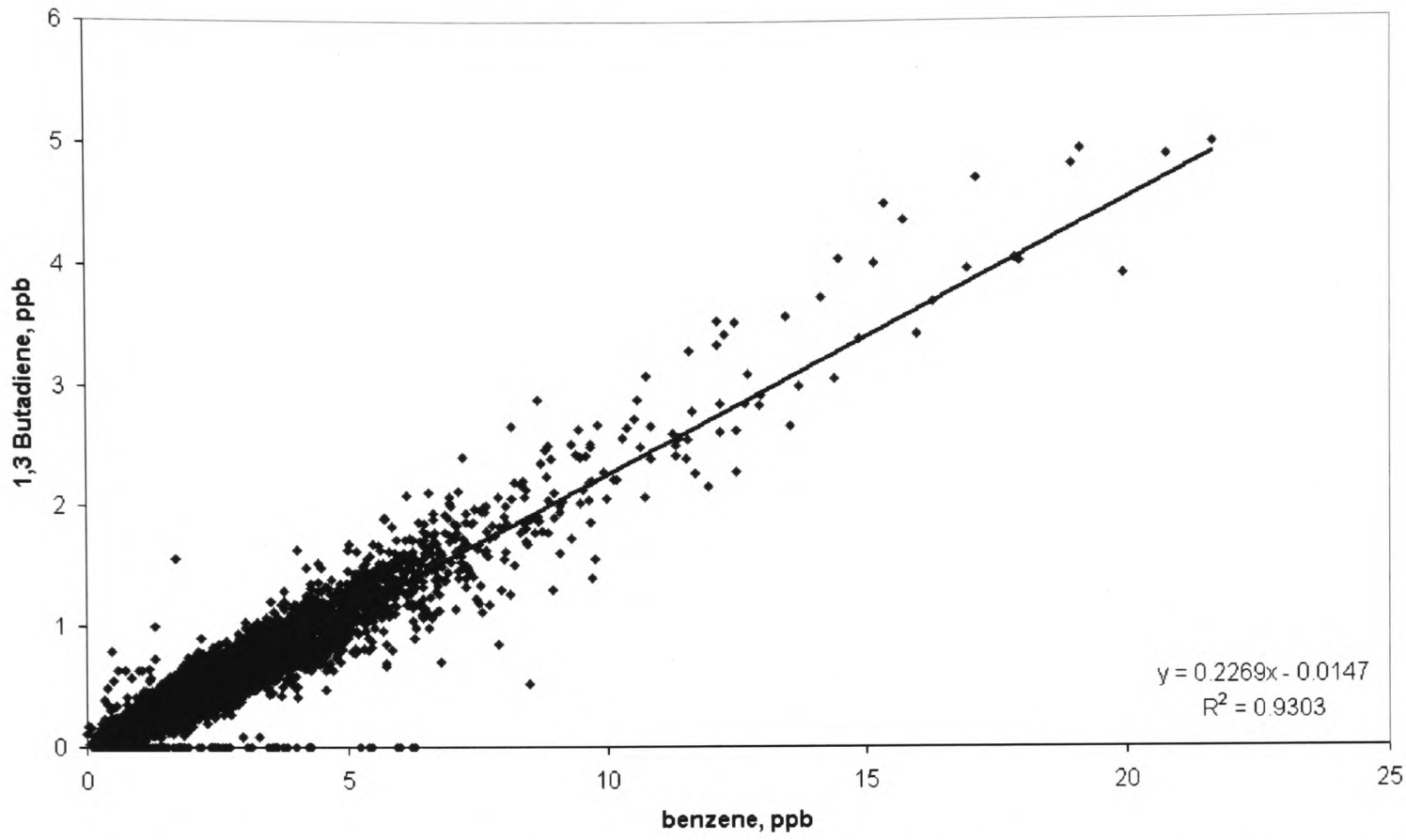
(source: compiled from data in the AQS, Table A4)

1,3 Butadiene

This pollutant can only be monitored using automatic equipment. Observations in the UK are currently undertaken at the same automated network sites that benzene is observed. However, as the main source of both these pollutants is road vehicles, the results of benzene monitoring could be used as a proxy in urban areas. Concurrent monitoring data from University College London (UCL) reveals a very high correlation between ambient concentrations of these two pollutants. This is illustrated in Figure 3.3 with the following regression equation:

are 'correct'.

$$[\text{1,3 butadiene}]_{ppb} = 0.2269 * [\text{benzene}]_{ppb} - 0.0147 \quad (R^2 = 0.9303) \qquad \ldots(2)$$



The above regression was constructed using hourly observations of made at the urban background site at UCL. Four years of observations yielded 31,300 pairs of data, excluding one outlier. The liner regression has a very high degree of correlation (R^2 = 0.9303).

Figure 3.3 : Observed Benzene and 1,3-butadiene levels : UCL, London : 1997-2000

(source: compiled from data in the National Air Quality Archive, *www.air-quality.co.uk/*)

Using this regression, the air quality objective of 1 ppb for 1,3 butadiene is likely to be met in urban areas if the running annual mean benzene concentration is below 5 ppb.

This conclusion, based on work by Ireland (1998) is supported by further analysis of data reported in the AQS. The results of monitoring benzene and 1,3 butadiene at twelve sites across the UK (1994-1998) are summarised in Figure 3.4. The derived regression from these summary data is similar to equation (2) above:

$$[\text{1,3 butadiene}]_{ppb} = 0.1991 * [\text{benzene}]_{ppb} + 0.0058 \quad (R^2 = 0.8908) \qquad \ldots(3)$$

Shifting the origin of both these regressions to (x=0, y=0) yields, for equations (2) and (3) respectively:

Shifting the origin of both these regressions to (x=0, y=0) yields, for equations (2) and (3) respectively:

$$\textbf{[1,3 butadiene]}_{ppb} = 0.2208 * \textbf{[benzene]}_{ppb} \quad (R^2 = 0.9289) \qquad \ldots(4)$$

$$\textbf{[1,3 butadiene]}_{ppb} = 0.2034 * \textbf{[benzene]}_{ppb} \quad (R^2 = 0.8903) \qquad \ldots(5)$$

The regressions are very similar, confirming the dominant contribution of road traffic to ambient concentrations of both pollutants and the lesser effect of local and inter-year meteorology.

From these regression, we can derive 1,3 butadiene concentrations will be approximately 20% of ambient benzene concentrations. Assuming levels of 1,3 butadiene are in decline and the dominant source continues to be road traffic, the air quality objective for this pollutant will be achieved if observed levels of benzene are below 5 ppb; or below 3.5 ppb if measured using diffusion tube. From data cited above, this appears already to be the case for most roadside locations. National modelling by Stedman and Dore (1999b) suggests the standard will be achieved at all locations by 2004.

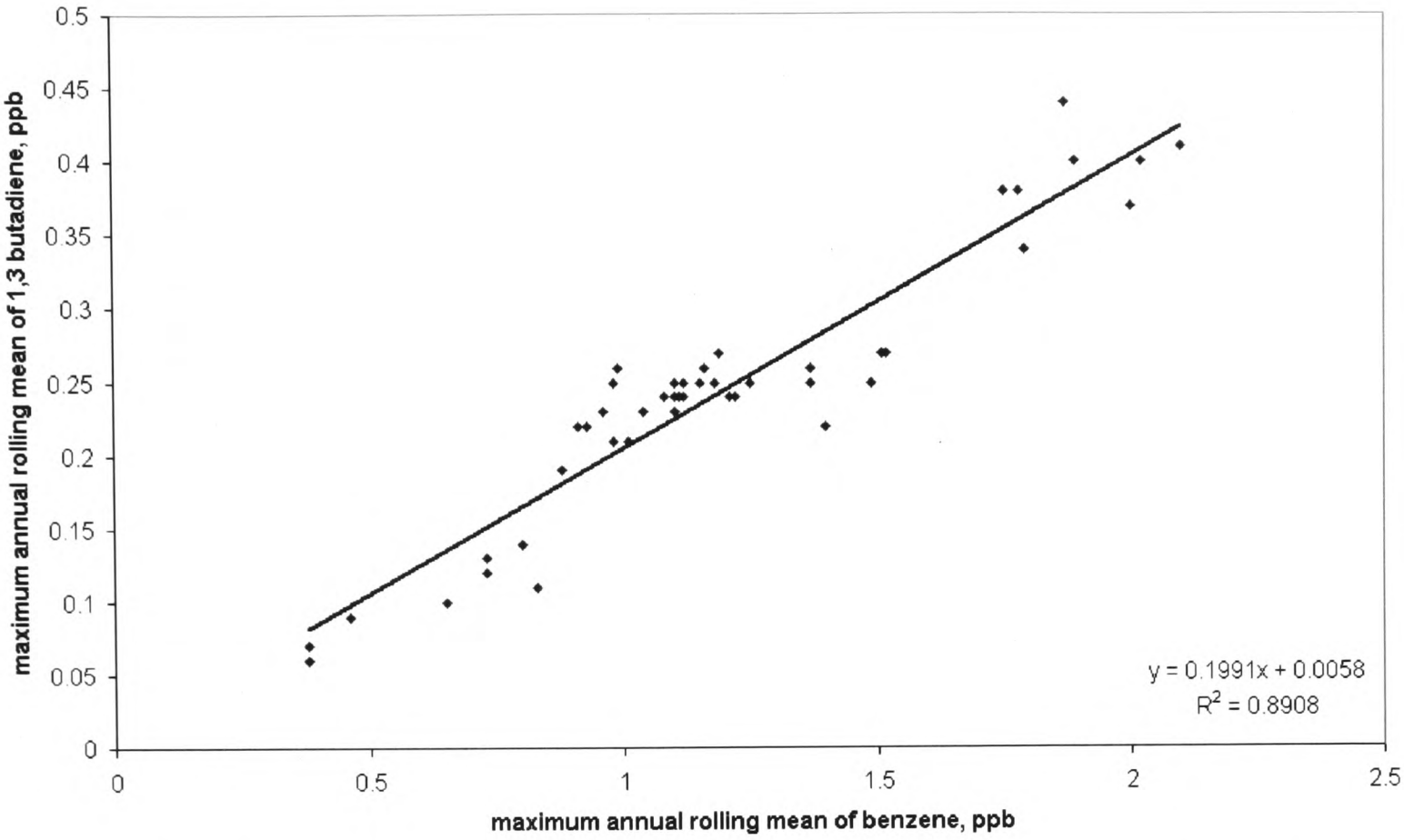
This work suggests no additional measures are required to manage 1,3 butadiene levels in the UK. Monitoring benzene as a proxy using diffusion tubes offers a suitable method for demonstrating the expected continued decline in 1,3 butadiene levels and can be used to assess air quality with reference to the UK air quality objective.

Carbon Monoxide

The only technique currently recognised by DETR (1998a, 2000i) and DEFRA (2002b) for monitoring carbon monoxide is automatic infra-red absorption. Less precise techniques are available based on electrochemical cells but are yet to be formally recognised.

The most significant source of carbon monoxide in urban areas is road traffic, suggesting annual mean observations of benzene could be used as a proxy. However, there is no strong correlation between the annual mean and the maximum 8-hour mean concentrations of carbon monoxide[19]. This is demonstrated in Figure 3.5.

19 The UK air quality objective is expressed as the *running* maximum 8-hour mean although the difference is negligible in the context of this discussion.



The above regression was constructed using annual summaries of the maximum running annual mean concentrations of benzene and 1,3 butadiene observed at eleven sites across the UK. A total of 45 pairs of data were compiled. Note: the DETR operate two additional sites: Middlesborough – an industrial site and Marylebone Road – data not yet available; both were excluded from this analysis.

Figure 3.4 : Observed Benzene and 1,3-butadiene levels : 1994 -1998

(source: compiled from data in the AQS, Tables A4 and A9)

Analysis of aggregated observations of carbon monoxide at urban locations by Archer *et al* (1998) and DETR (2000b) suggests a decline in the urban maximum 8-hour mean of 0.9 ppm per annum since 1990. This trend cannot be applied directly to observed kerbside levels owing to local effects such as street canyons. However, if a 'no change in policy' scenario (ie. future reductions in road traffic emissions will occur at least at the same rate as in the past[20]) is assumed, extrapolation of this trend to 2005 suggests the air quality objective will be achieved at locations where the current (2000) observed maximum 8-hour concentration of carbon monoxide is less than 13 ppm.

20 This assumption is based on a continual roll over of the national vehicle fleet, with new cars fitted with three way catalytic converters replacing the older vehicles without, and continual improvement in the performance of three way catalytic converters.

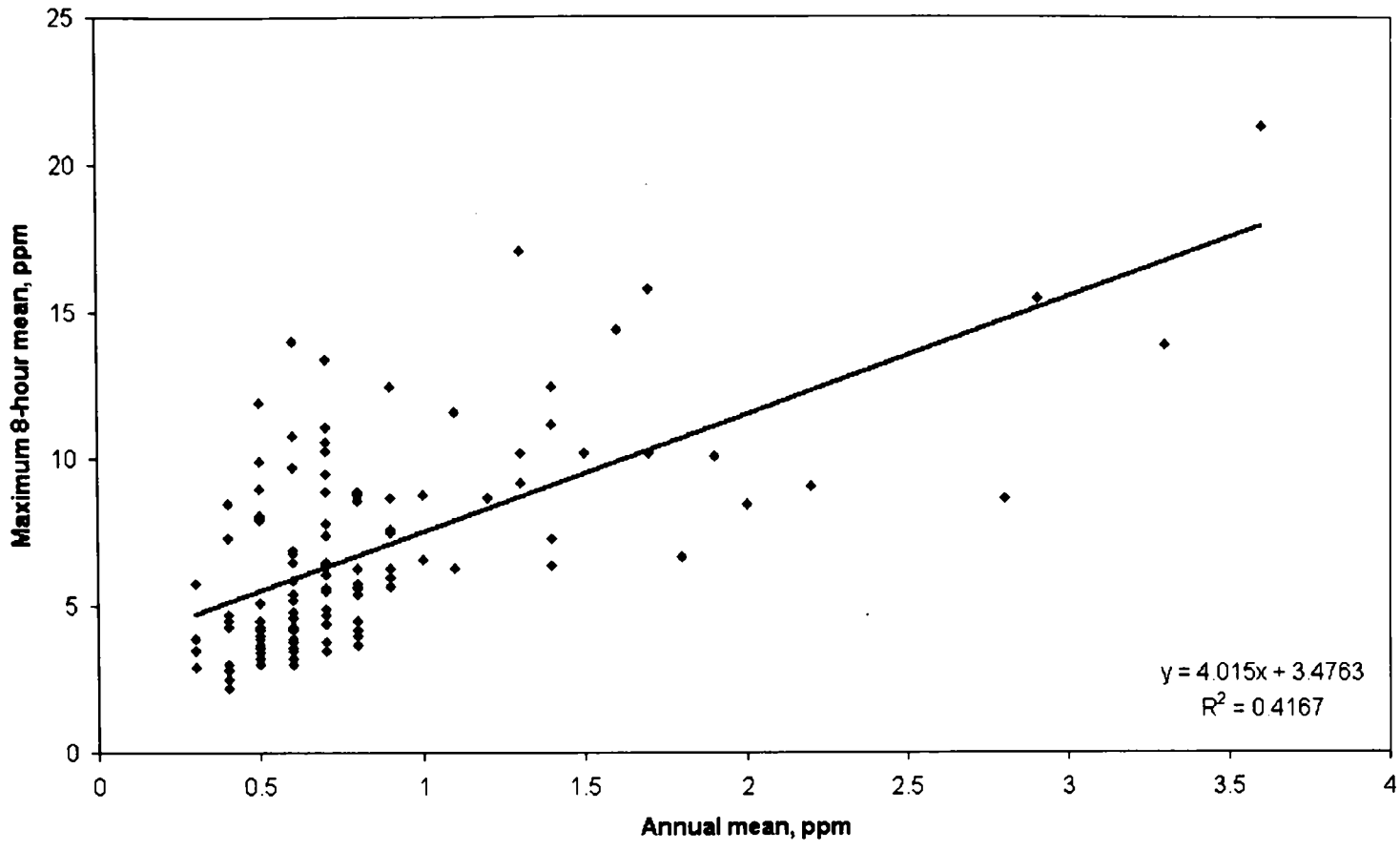
The only kerbside monitoring site with several years of data for ambient concentrations of carbon monoxide is the heavily congested Cromwell Road (AADTF = 60, 000). The maximum rolling 8-hour mean concentrations observed since 1990 are as follows:

Year	1990	1991	1992	1993	1994	1995	1996
ppm	15.5	13.9	8.7	9.1	10.1	6.7	6.4

The limited data set for Cromwell Road precludes drawing any conclusions of long term trends. The results do confirm the air quality standard (of 10 ppm) has not been exceeded since 1994 at this busy location and are well below 13 ppm. This suggests the air quality objective will not be breached. This is supported by data included in the AQS (Table A14) for seventeen roadside / kerbside monitoring stations during 1998; the maximum running 8-hour concentration of carbon monoxide being in the range 3.7 to 11 ppm with all but three sites below 7 ppm.

The results of modelling of roadside carbon monoxide levels for the 7508 most heavily trafficked links across the UK by Stedman *et al* (1999b) are summarised in the AQS; with only one link expected to breach the air quality standard and EU limit value by 2004 assuming extreme meteorology. In more typical years, the air quality standard and limit value would be achieved by this year.

The above work suggests no additional measures are required to manage carbon monoxide levels in the UK. Monitoring techniques are currently limited although, given the margin between observed values and the air quality standard, less precise techniques could be adopted to demonstrate the expected continued decline in carbon monoxide levels.



Ratified data from 35 monitoring stations in 1998 including historic data where available yielded some 124 pairs of data. The degree of correlation between the maximum 8-hour mean and the annual mean is low ($R^2 < 0.42$, 95% confidence level) and is influenced by the tail end data at higher concentrations; at annual mean concentrations <2 ppm (119 data points) $R^2 < 0.33$, 95% confidence level.

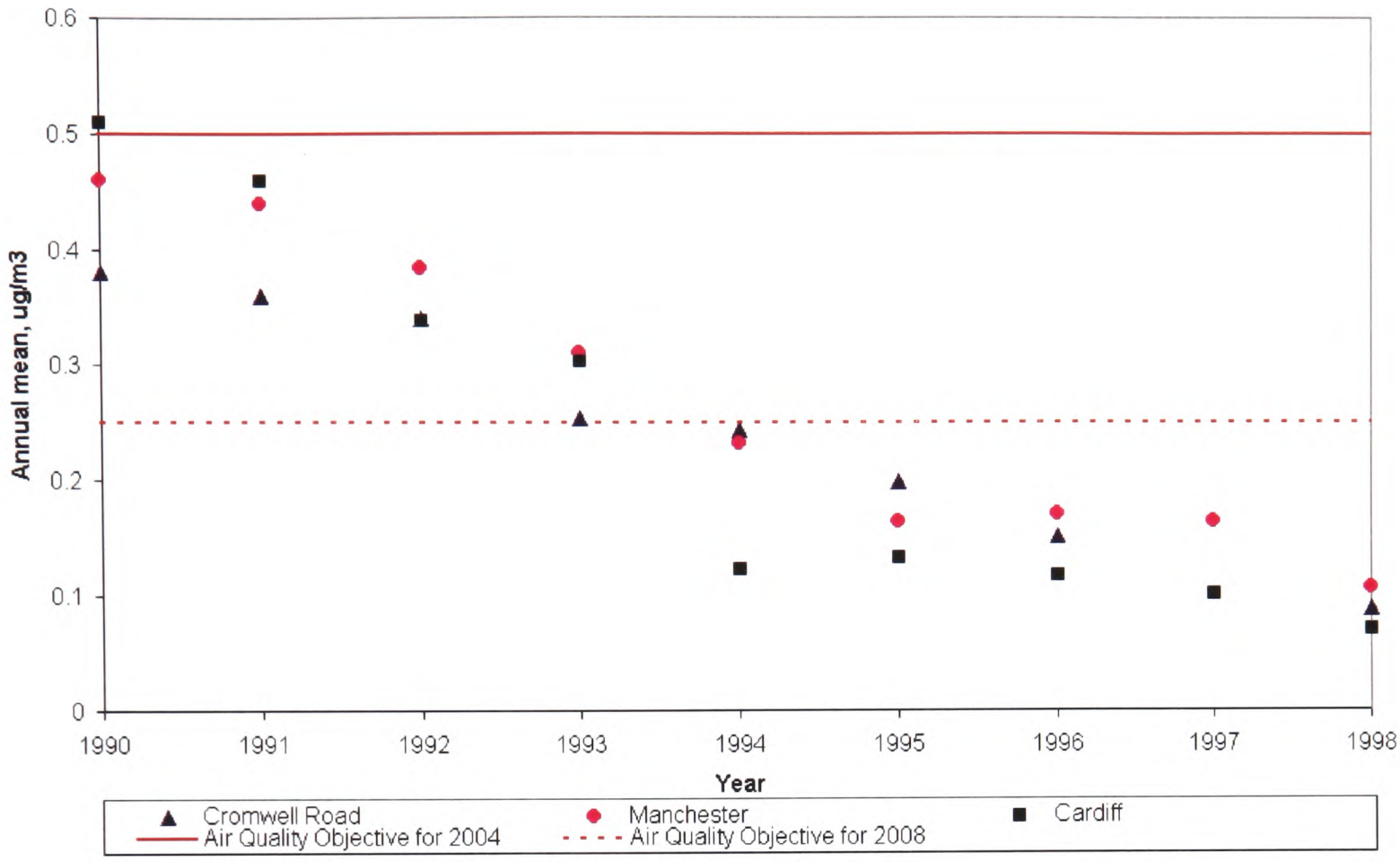
Figure 3.5 : Observed Carbon Monoxide Levels : AUN : 1988 to 1996

(source: compiled from data in the National Air Quality Archive, *www.air-quality.co.uk/*)

Lead

An analysis of aggregated data reported in the AQS (Table A18) indicates lead levels at all urban and rural sites monitored have been below 0.25 $\mu g/m^3$ (the air quality standard for 2008) since 1990. Observed levels of lead are higher at kerbside locations and near to specific metallurgic industries.

Figure 3.6 illustrates the consistent decline in kerbside concentrations of lead corresponding with the reduced consumption of leaded petrol at three UK sites. The air quality standard for 2008 has already been achieved at these sites since 1994 suggesting the assessment of lead need not include road traffic. Local air quality management for lead is limited to specific industrial processes already regulated by the Environment Agency or the local authority.



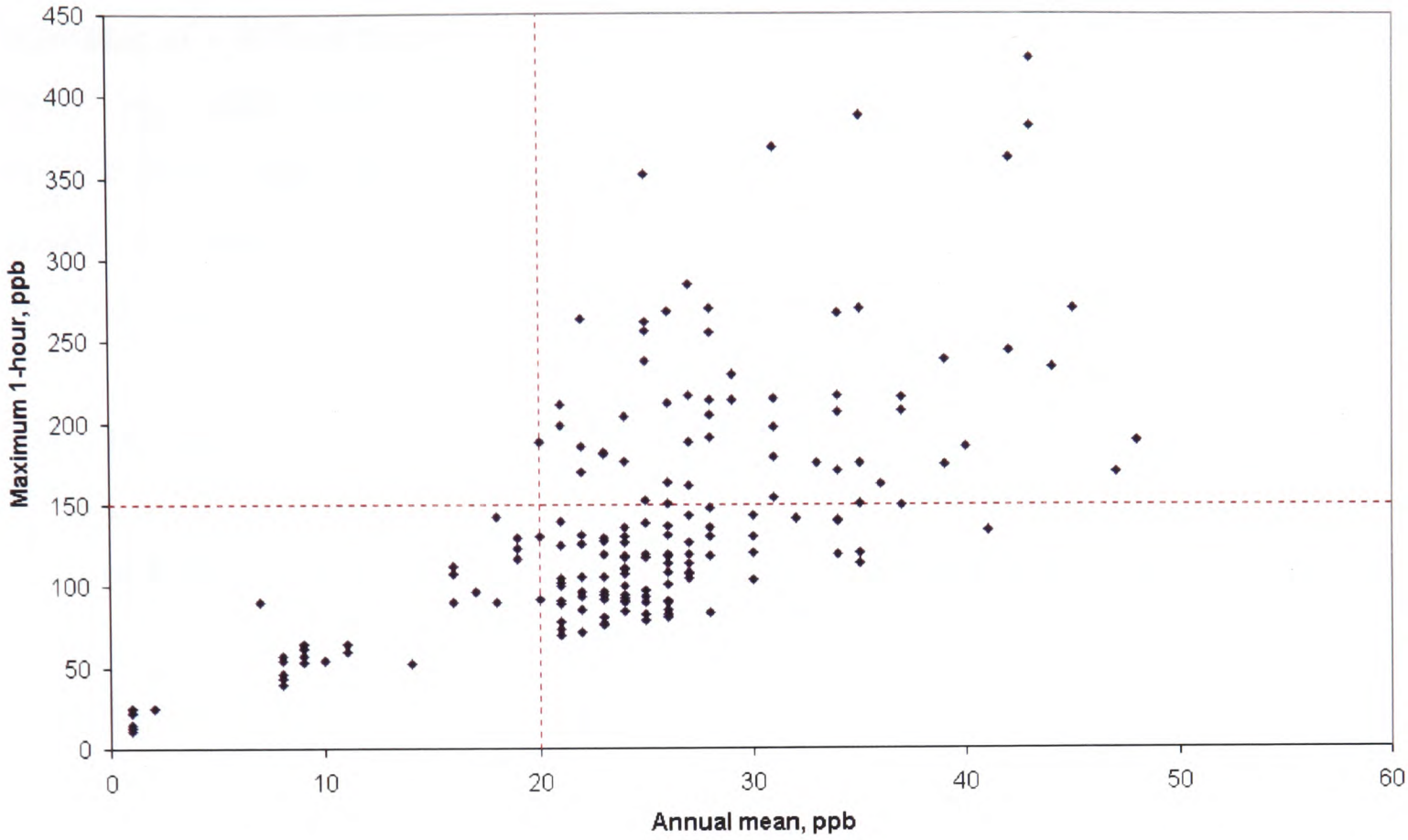
Kerbside monitoring data from central London, Cardiff and Manchester (1990 to 1998) all demonstrate a similar decline in levels of lead at locations dominated by vehicular emissions.

Figure 3.6 : Observed Lead Levels : 1990 to 1998

(source: compiled from data in the AQS, Table A18, DETR, 2000b)

Nitrogen Dioxide

Compliance with the 1-hour air quality standard can only be determined using automatic equipment. However, the most commonly used method is the diffusion tube, providing a fortnightly or monthly mean value, that can only be used to assess nitrogen dioxide levels with respect to the annual mean standard of 21 ppb (40 $\mu g/m^3$). Figure 3.7 is a scatterplot of the number of exceedances of the maximum 1-hour standard (as per the *1997 Air Quality Regulations*) against the annual mean observed from all automatic monitoring stations in the UK since 1979 (Ireland, 1998). As expected, there is no direct correlation between the averaging times. However, for the years and locations with an annual mean value of less than 20 ppb, the maximum 1-hour objective was not exceeded.



Ratified data from 47 monitoring stations in 1996 including historic data where available yielded some 180 pairs of data. The red dashed lines indicate the air quality standards relevant when the work was undertaken. There is little correlation between the two sets of data ($R^2 < 0.43$, 95% confidence level for a straight line regression) but for all annual mean values less than 20 ppb the corresponding maximum 1-hour value was within the air quality standard.

Figure 3.7 : Relationship Between the Annual Mean and Maximum 1-hour Concentration of Nitrogen Dioxide : 1987 to 1996

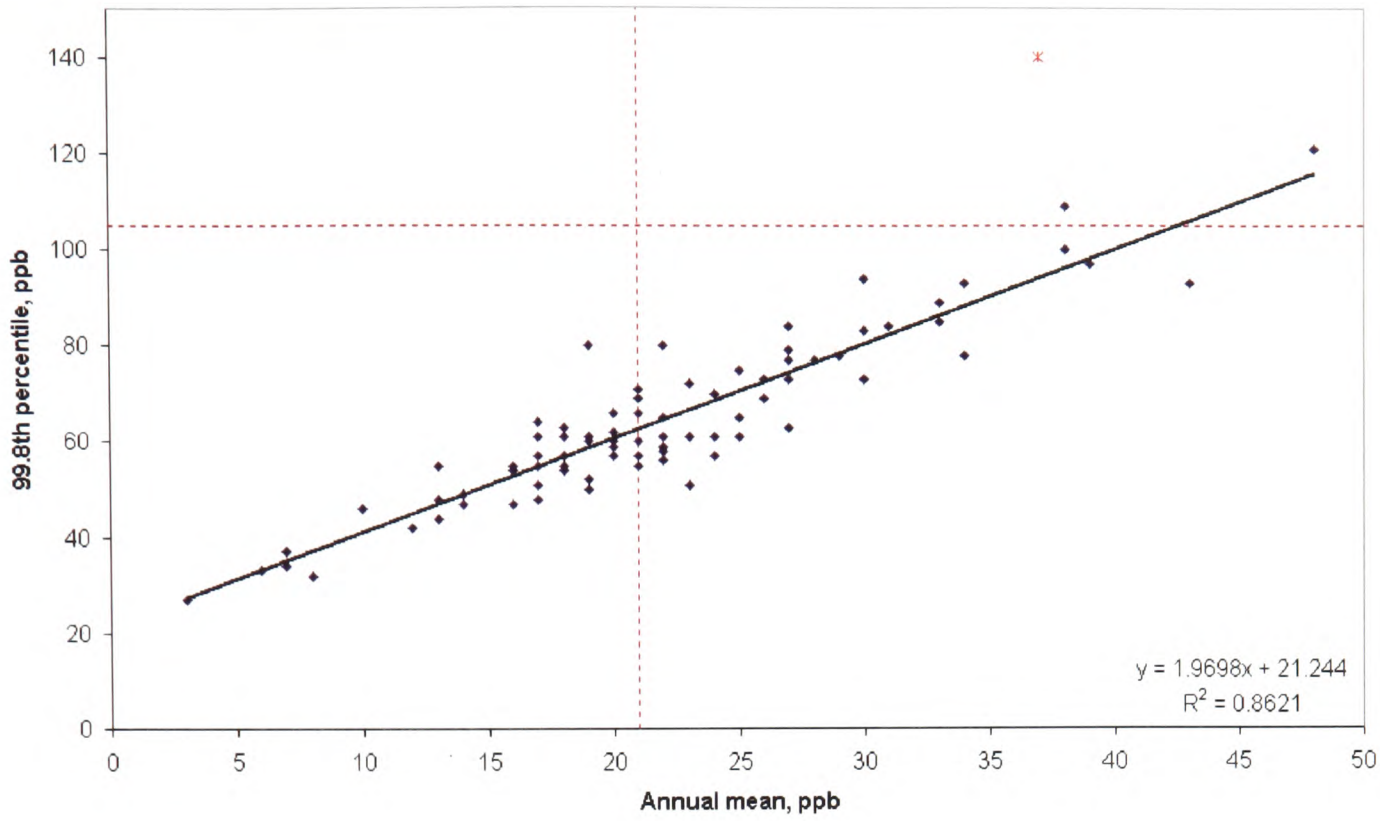
(source: Ireland, 1998)

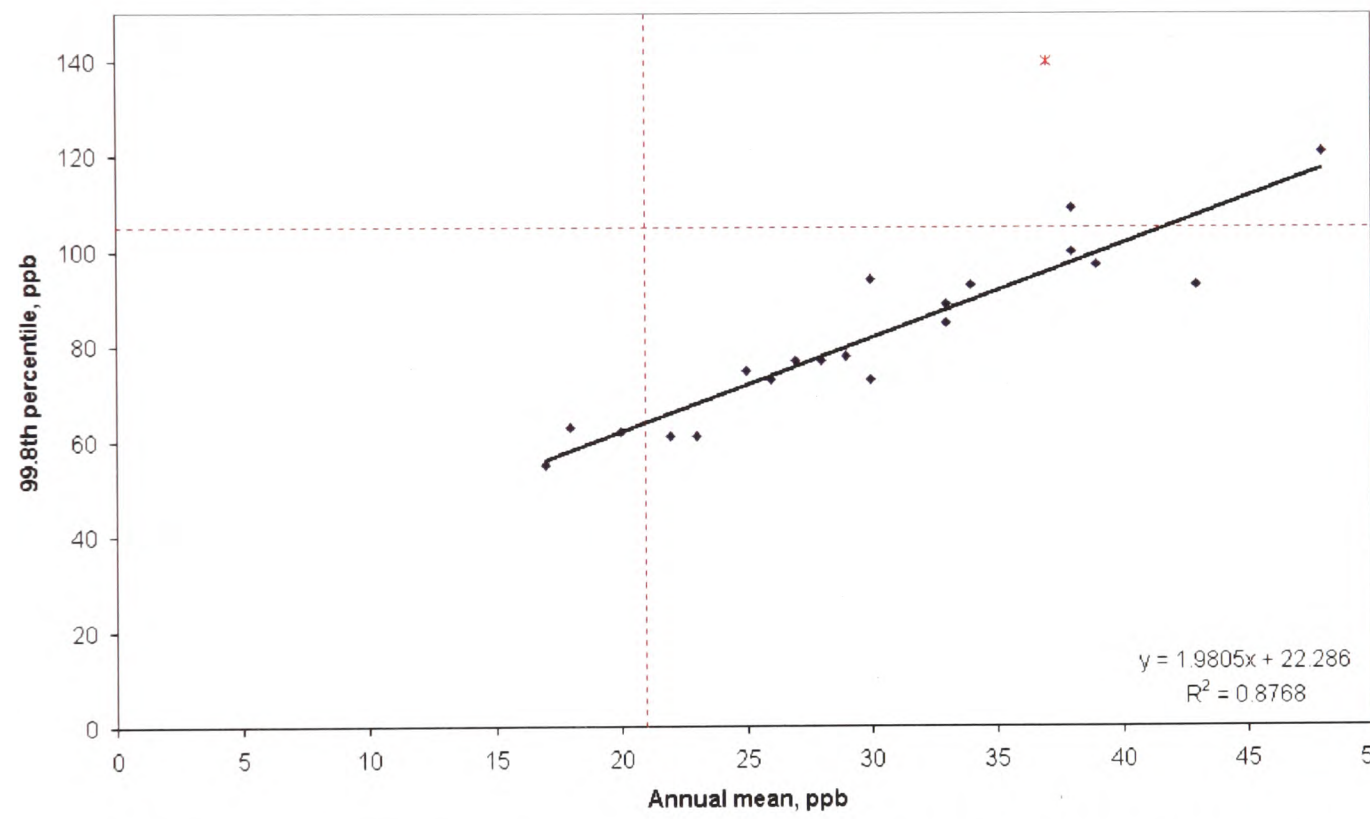
Since the revision of the air quality objective (150 ppb as the 99.8th percentile of 1-hour values) this relationship has been revisited using 1998 data from 85 sites within the AURN (Figure 3.8). This analysis should be treated as preliminary as variations in the 99.8th percentile of hourly means are expected with different meteorological years; this year also being characterised by good dispersion and hence, generally good air quality. However, the straight line regression yields a high level of confidence with only one outlier suggesting the analysis is robust. The 99.8th percentile of hourly values is expected to be within the air quality standard when the annual mean is less than approximately 35 ppb.

Monitoring guidance published by DETR (1998a, 2000i) and DEFRA (2002b) suggests the use of nitrogen dioxide diffusion tubes is an acceptable method for R&A within a tolerance of ± 30% of the air quality standard[21]. This suggests breaches of the short term (99.8th percentile of hourly values) objective are unlikely if observed annual mean levels are less than 24 ppb. Interpretation of diffusion tube monitoring data in this way may be useful to identify locations where more detailed assessment, including automatic monitoring, is required.

Only 2% of the data cited in Figure 3.8 were above the short term standard whereas 53% of sites were above the annual mean standard, suggesting the latter will be much harder to achieve. Analysis of aggregated data from urban centre and urban background monitoring sites by Archer *et al* (1998) reported in the AQS (page A143) suggests the annual mean concentration has been declining by between 0.3 to 1.5 ppb per annum since 1987 with the typical decline being approximately 0.7 ppb. Extrapolating the annual mean trend, assuming a 'no change in policy' scenario as before, suggests the annual mean objective will be achieved in urban centre and urban background locations if the observed concentration in 1998 was less than 16.5 ppb. From the data cited in Figure 3.8, only 21% of the 85 sites recorded annual mean concentrations less than 16.5 ppb in 1998. Archer et al (1998) also report trend analysis for the maximum 1-hour values although only the 98th percentile of hourly values are reported as statistically robust in the AQS. Given the argument presented around Figure 3.8; that the short term objective appears to have already been met at most (98%) locations, further consideration of this trend analysis is not made in this research.

21 In reporting data from the UK NO_2 network, Loader *et al* (2002) suggest annual averages of NO_2 calculated from six month of data are likely to be within ± 10% of the annual average at urban and suburban sites and within ± 20% at roadside sites. Earlier work by Heal *et al* (1999) which suggested the diffusion tube method generated results 22-24% greater than observations using chemiluminesence appears to have been addressed by improved laboratory QA/QC; Bush *et al* (2001) reports an uncertainty of ± 10-18% for annual means.





The top graph was constructed using ratified data from 85 monitoring stations in 1998. The red dashed lines indicate the current (2000) air quality standards. There is a strong degree of correlation between the two sets of data: R^2 =0.8220, 95% confidence level for a straight line regression increasing to 0.8621 excluding one outlier (shown in red). Although limited to one year, the analysis suggests the short term objective will be achieved at locations where the annual mean is less than approximately 35 ppb.

The bottom graph was constructed using only roadside data from the same data set (21 sites). The regression and degree of correlation are very similar. Note that almost all the roadside sites breach the annual mean air quality standard.

Figure 3.8 : Relationship Between Annual Mean and 99.8th Percentile of 1-hour Mean Nitrogen Dioxide : 1998

(source: compiled from data in the AQS, Table A22).

Modelling by Stedman *et al* (1999c, 2001a-b) of urban locations across the UK is cited in the AQS, taking into account measures to further reduce emissions of nitrogen oxides in the future. The results suggest annual mean concentrations will be less than 18 ppb at 23 of the 31 sites modelled by 2005 and at 28 of the sites by 2009. Only one location (London Bloomsbury) is not expected to achieve the air quality standard by 2009. The implication here is that urban concentrations of nitrogen dioxide will continue to decline with most locations meeting the objectives. The exception to this is roadside locations. The modelling by Stedman *et al* included roadside locations, also cited in the AQS, and a number of vehicular emission reduction scenarios which, if implemented, would reduce the number of locations not complying with the air quality standard by 2010 to a handful of road links. It is implicit from the AQS that national measures to reduce emissions from the vehicle fleet will not be sufficient to achieve the objectives for nitrogen dioxide and local measures are required.

Although mean winter/summer ratios of NO_2 concentrations are generally in the order of 1.2 to 1.6 at background locations (Atkins and Lee, 1995, Hargreaves *et al*, 2000, Loader *et al*, 2002) short term peak concentrations of nitrogen dioxide are associated with summer photochemistry (the increased availability of ozone increasing the rate of oxidation of nitrogen monoxide; $NO + O_3 \rightarrow NO_2$) and winter smogs, where the trimolecular reaction is thought to be one of the dominant processes ($NO + \frac{1}{2}O_2 \rightarrow NO_2$). In explaining why national modelling of nitrogen dioxide was limited to the annual mean, Stedman *et al* (1999c) cite earlier work by Bower *et al* (1994) who considered the rate of the trimolecular reaction to be dependant on the square of the nitrogen monoxide concentration. Hence, any reductions in emissions of nitrogen oxides will have a marked effect in reducing winter episodes of nitrogen dioxide. Stedman *et al* (1999c) support this by citing unpublished work by Derwent who modelled ozone trajectories across London. Derwent estimated the annual mean would fall to 70% of current values but winter peaks would fall to less than 50% of current values, taking into account the expected emissions reductions of nitrogen oxides. The AQS refers to other modelling studies by Derwent supporting the case that the annual mean objective is likely to be more stringent than the short term one. In one of these studies, Dixon *et al* (2000, 2001) estimate, from a study to fit polynomials to binned observations of nitrogen oxides and nitrogen dioxide, a reduction of 32-43% in the maximum observed concentration of nitrogen oxides (1991-1997) will be required to

meet the 1997 maximum 1-hour objective. Assuming the concentration of nitrogen oxides is directly related to emissions, this reduction compares with various emissions reduction scenarios cited in the AQS of 56% by 2005, 67% by 2009 (both from a 1996 baseline) and further reductions of 7-13% (from the same base year). The studies cited in the AQS consistently predict breaches of the annual mean air quality standard in both 2005 and 2010.

From the above analysis, using diffusion tubes will provide suitable data for evaluation of ambient air quality with reference to both the short term and long term air quality objectives. Monitoring data suggests the short term standard is achieved if the annual mean concentration of nitrogen dioxide observed using diffusion tubes is less than 35 ppb. Analysis of AURN data reveals the long term objective will be much harder to achieve; this is supported by national and urban scale modelling.

Particles

The Airborne Particles Expert Group, APEG (1999) identified three sources of particles:

Total PM_{10} = (primary + secondary + coarse) particles ...(6)

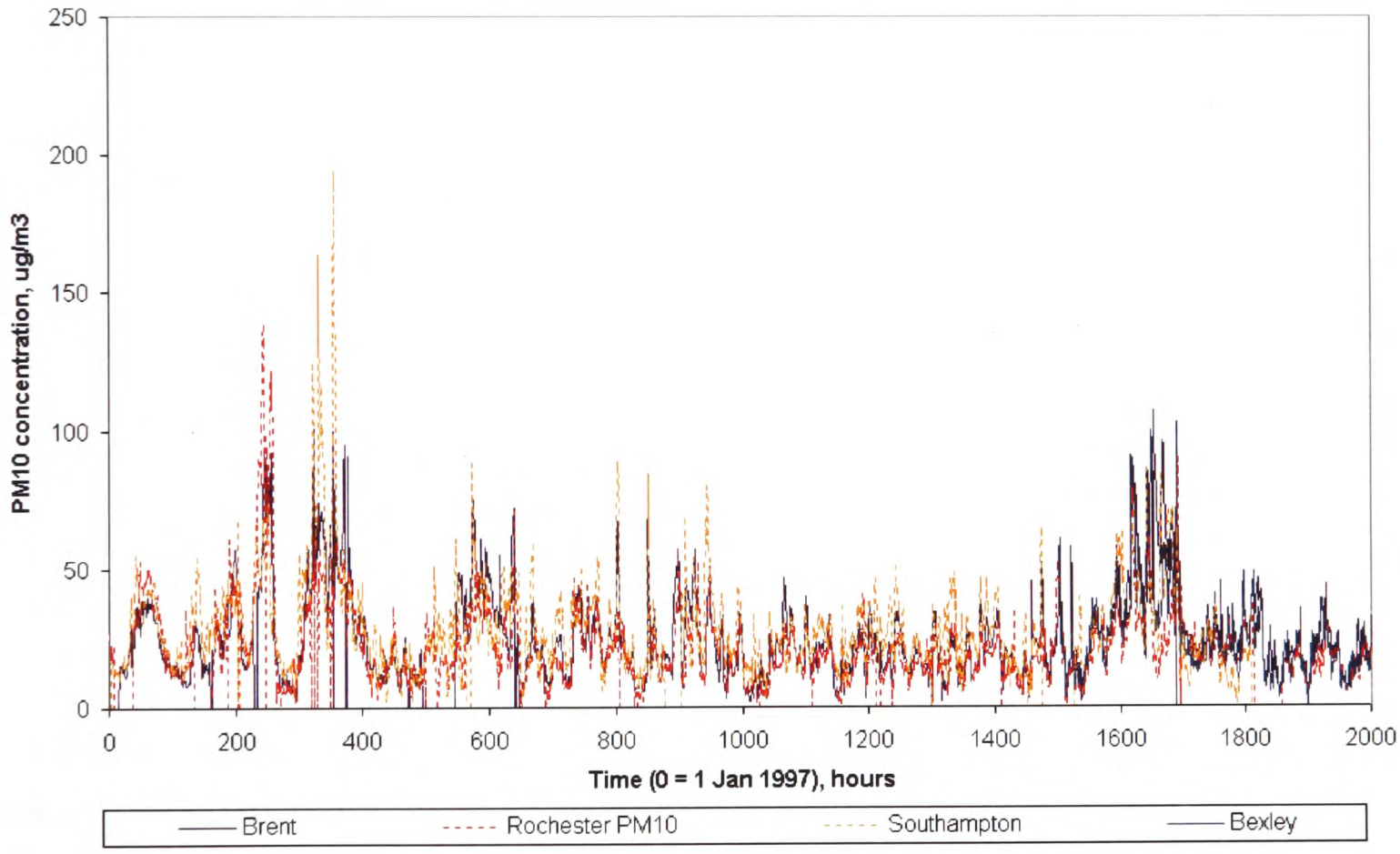
Primary particles generated in mainland Europe are also observed in the UK. Estimates suggest the annual mean contribution from sources outside the UK ranges from 1.9 $\mu g/m^3$ in London to 1.0 $\mu g/m^3$ in Edinburgh. APEG states this contribution is likely to be much greater during episodes. Secondary particles are formed within the atmosphere by chemical reaction or condensation of gases. Major contributors are sulphur dioxide and nitrogen oxides forming, through oxidation, sulphate and nitrate salts. A certain amount of particulate matter forms naturally from sources such as evaporation of water from sea spray, wind-borne pollen and dust. These sources form a background level of coarse particles.

APEG reports annual mean concentrations of PM_{10} in urban areas in the range 10 to 40 $\mu g/m^3$, made up typically as follows:

3 $\mu g/m^3$	:	Primary
10 $\mu g/m^3$	:	Secondary
6 - 10 $\mu g/m^3$	:	Coarse

In comparison, rural background levels are reported to be typically up to 10 $\mu g/m^3$

Seasonal variations in PM_{10} are observed with the additional effects of meteorology also being important. Breaches of the air quality standard for PM_{10}, expressed as a rolling maximum 24-hour mean, are observed in summer months - which may be wholly due to sources from the Continent and photochemical processes - and during winter months due to temperature inversions trapping emissions at ground level. However, evidence reported by King *et al* (2000) suggests most urban PM_{10} does not originate from local sources and the occurrence of pollution episodes tends to occur across whole regions of the country. This is supported by example in Figure 3.9; observations made at four monitoring sites in the south east of England demonstrate a number of episodes occurring concurrently.



From an analysis of hourly sequential data of PM_{10} concentrations observed at a number of sites across the south east during 1997-1998, the extract above is for the first 2000 hours of 1997 (up to 24 March 1997). All sites exhibit a similar pattern, due to variations in the meteorological influence on the local dispersion of emissions. Observations on the south coast include a high frequency of elevated concentrations, either due to sea salt or continental emissions. The latter appears to be significant during the episode (350-400 hours) with all sites exhibiting elevated concentrations.

Figure 3.9 : An Example of a Regional PM_{10} Episode Across South East England

(source: compiled from data in the Air Quality Archive, *www.air-quality.co.uk/*)

The monitoring of PM_{10} to observe such events is only possible using automatic techniques providing aggregates of less than daily (ie. hourly) mean concentrations (Ruuskanen *et al*, 2001). In the UK this is achieved using the *tapered oscillating electronic microbalance* (TEOM) instrument. This instrument is not consistent with the *high volume sampler* prescribed in the EU Daughter Directive, albeit the latter only provides daily mean values. Work by Moorcroft *et al* (1999) sponsored by the DETR suggested applying a factor of 1.3 to TEOM results for comparison with the high volume sampler whilst recognising local variations will apply[22]. This factor was used in the AQS (Table A32) to report ambient levels of PM_{10} in the UK monitored using the TEOM with reference to the EU limit values. The data reported in the AQS are included in Figure 3.10.

From these data we can derive a relationship between the annual mean and the 90th percentile of daily means (the EU Stage I limit value):

[90th percentile of daily means] = 1.5518 * [annual mean] + 0.8461 ...(7)

(all values in $\mu g/m^3$, gravimetric, $R^2 = 0.8656$, 95% confidence)

Although there is no correlation between the annual mean and the higher percentile statistics, the results do indicate the EU Stage I short term limit value will be achieved if annual mean concentrations are less than 32 $\mu g/m^3$. Analysis presented in the AQS concludes the EU Stage I and Stage II short term limit values will be achieved if annual mean concentrations are less than 28 $\mu g/m^3$ and 17 $\mu g/m^3$, respectively.

Modelling by Stedman *et al* (1998, 1999d, 2001a, c) of future PM_{10} concentrations has been undertaken for a number of urban sites across the UK. Predictions were made using 1995 and 1996 meteorology (see Chapter 4) and secondary contributions for both a 'business as usual' scenario and a 'primary and secondary emissions reduction' scenario. The results, cited in the AQS (Tables A34 and A35) lead to the following conclusions:

22 The reason for the inconsistency is the preheat chamber in the TEOM, raising the air inlet temperature to 50°C, with subsequent loss of volatiles. As the concentration of volatiles varies in the air, there is no direct relationship between TEOM and gravimetric measurements. However, subsequent unpublished work by Moorcroft based on lowering the air inlet temperature demonstrates a consistent relationship between the two instruments (based on own discussions with equipment suppliers).

- the 1997 UK air quality standard will be widely exceeded in 2005 and 2010 for both scenarios, albeit to a lesser extent in 2010

- the EU Stage I limit values (90^{th} percentile of daily means and annual mean) will be achieved in 2005 and 2010 at all locations, with few exceptions only for the short term limit value

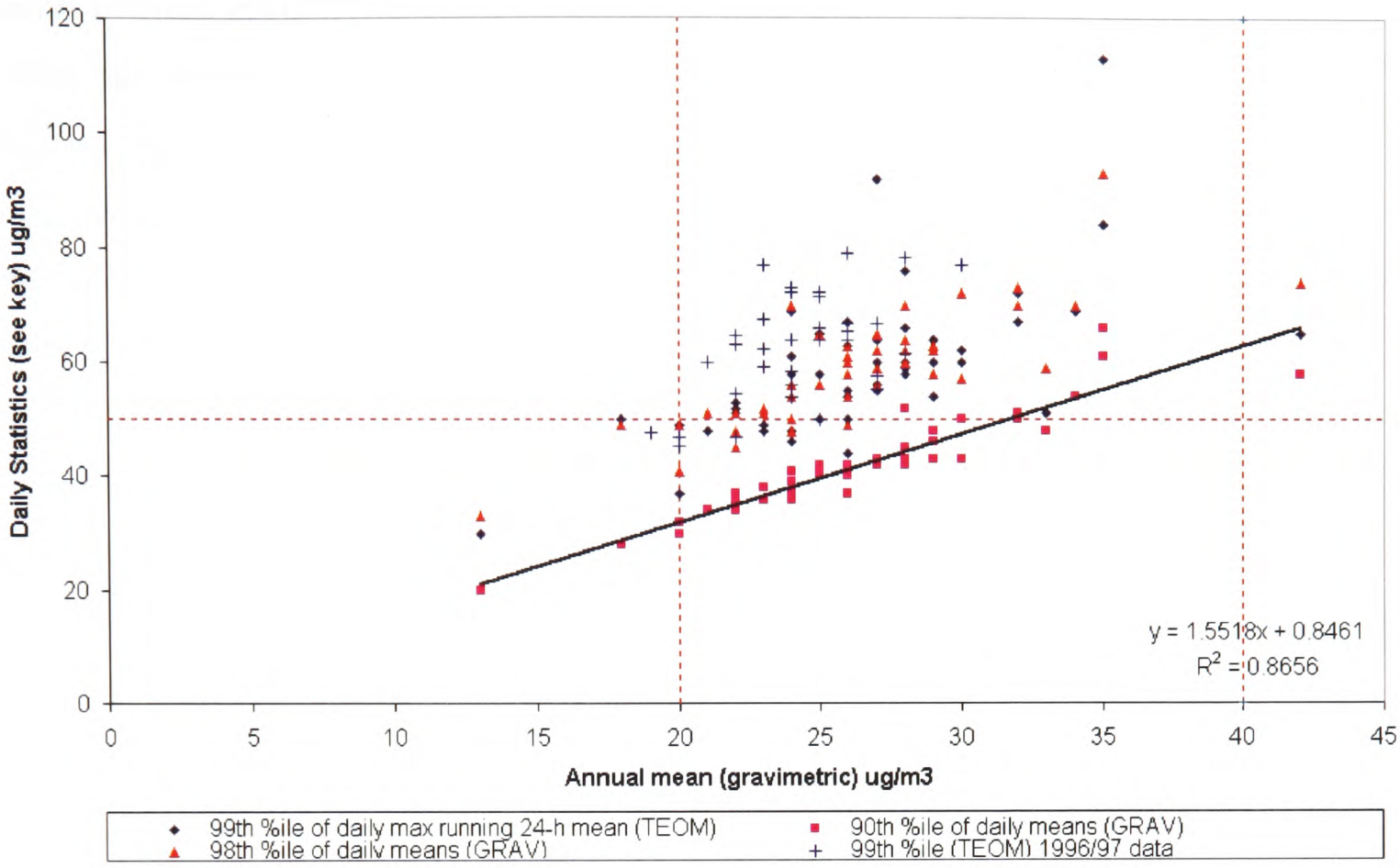
- The EU Stage II limit value (98^{th} percentile of daily means and annual mean) will be widely exceeded in 2005 and 2010 for both scenarios.

Stedman *et al* (1999d, 2001c) also conclude the EU Stage II annual limit is more stringent than the EU Stage II short term limit. This is consistent with the analysis presented in Figure 3.10. This will require significant reduction in both local emissions of primary PM_{10} and regional / national / pan-European emissions of primary and secondary PM_{10}. Only continued monitoring and analysis of short term (aggregates of one hour) variations in ambient levels of PM_{10} will enable source apportionment to be made. For the purposes of modelling and subsequent local air quality management this is important as it enables the background contribution - the proportion of PM_{10} that cannot be directly managed - to be determined.

Sulphur Dioxide

The air quality criteria for sulphur dioxide are measured as either fifteen minute, hourly or daily means suggesting only the use of continuous monitoring equipment will enable local authorities to demonstrate compliance. The monitoring equipment is relatively expensive so the number of monitoring sites across the UK is limited. This in turn limits the information available regarding the frequency and location of any breaches.

In urban areas where coal is still burnt, EPAQS (1995a) suggests the 15 minute air quality standard (100 ppb for the 99.9^{th} percentile of 15 minute means) is unlikely to be exceeded if either the maximum daily mean concentration of sulphur dioxide is below 28 ppb or if the 98% percentile of daily means is below 19 ppb. Ireland (1998) summarised monitoring data from the smoke and sulphur dioxide monitoring network for 1995 and 1996, concluding that most urban locations are likely to experience short term exceedances on the basis of analysis by EPAQS. A review of the exceedances reported from national monitoring sites confirms this observation; see Figure 3.11.



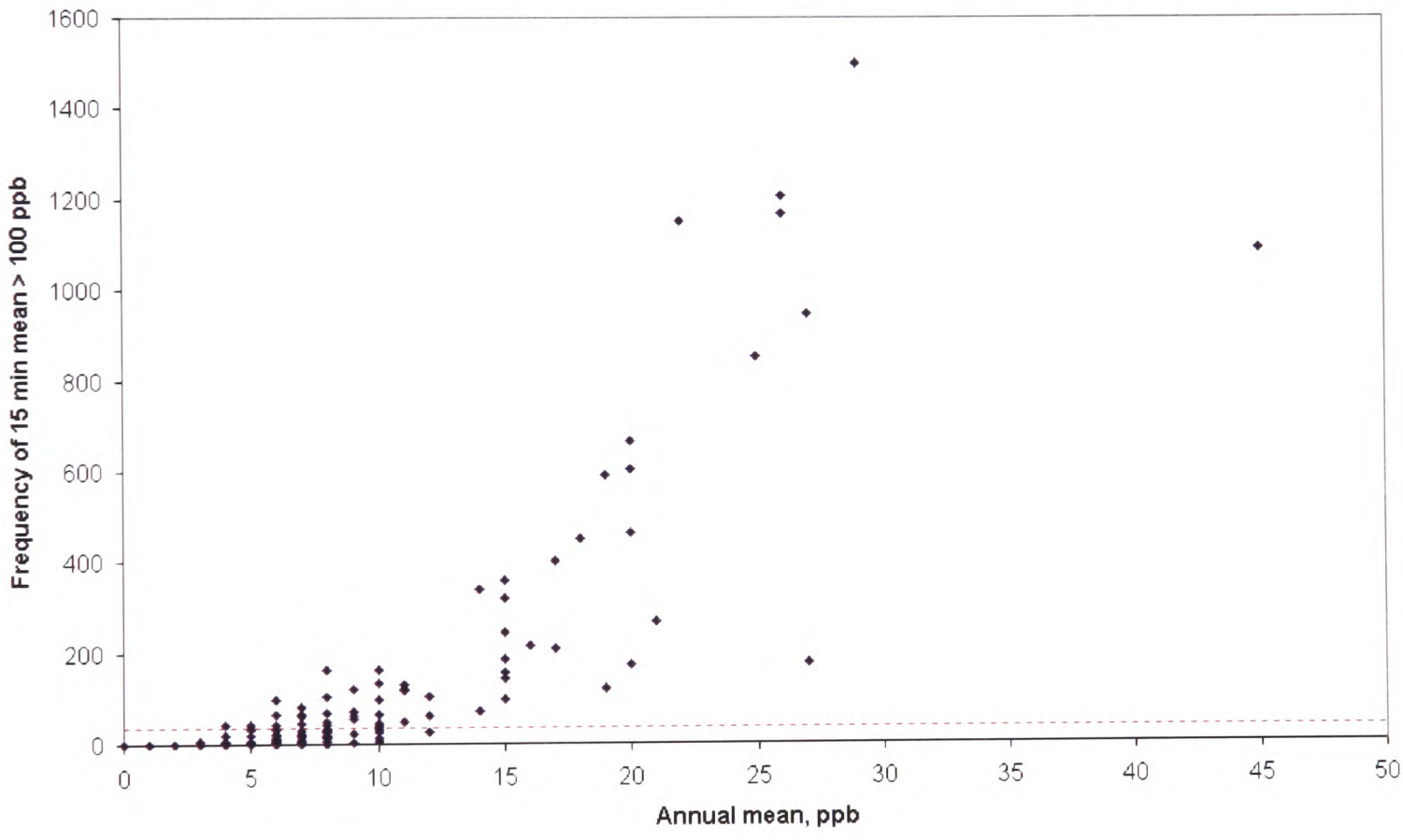
The data for 1998 (solid symbols) include 49 sites across the UK. The data for 1996-97 are based on 18 sites (total of 36 pairs of data). All data are derived from TEOM data using the 1.3 factor where applicable (see main text). For the 1998 data, the EU Stage I Limit (90th percentile of daily means) is generally achieved and shows a high degree of correlation with the annual mean (R^2 = 0.8656, 95% confidence level). The degree of correlation is much diminished with higher percentiles. The UK air quality objective (99th percentile of daily maximum running means, TEOM) appears to be similar with the EU Stage II Limit (98th percentile of daily means, gravimetric). Data for 1996 and 1997 are included to illustrate 1998 was a 'good' year for air quality with sufficient dispersion and low secondary contribution from the continent. Note, however, the majority of sites breached the air quality standards and limit values, with the exception of the Stage I EU limit values (90th percentile of daily means and annual mean).

Figure 3.10 : Relationship Between Annual Mean and Daily Statistical Meters of PM_{10} : 1998; 1996 and 1997

(source: compiled from data in the AQS, Table A32 and Stedman, 1998b).

The AQS reports monitoring data for 1998 across the UK. The results are compared to the three air quality criteria in Figure 3.12. The criteria are achieved at nearly all the sites with the 15 minute standard being breached at more sites than either the 1-hour or daily standards.

Although annual mean levels of sulphur dioxide in urban areas are higher than in rural areas (Abbott *et al*, 1999) the reduction in domestic coal use since the Clean Air Act 1956 has more or less eliminated the occurrence of extended periods of high sulphur dioxide and smoke concentrations in urban areas. Moreover, industrial use of coal has also decreased with many operators using boilers fired on gas; the pattern of sulphur dioxide today is characterised by low annual mean concentrations in urban areas with short term peak concentrations due to the grounding of plumes from large industrial sources burning coal or oil. In studies undertaken for the UK power generators, Laxen (1996) suggested the UK 15 minute objective for sulphur dioxide was unachievable in the vicinity of coal and oil fired plant. Subsequent studies by Fisher and Acres (2000) suggest the introduction of more stringent emission limits by the Environment Agency will ensure the objective is met at these locations.

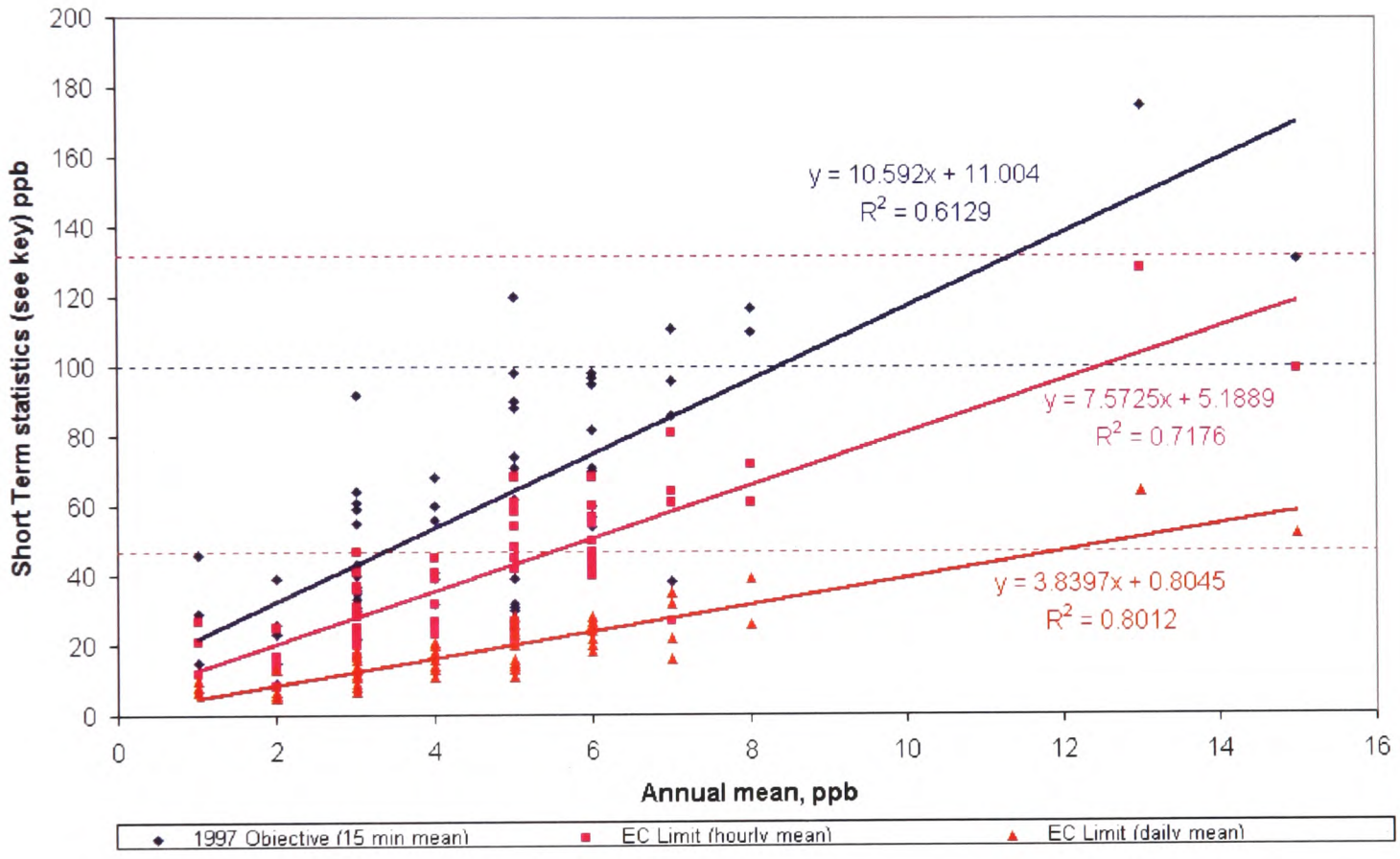


Data from 40 sites across the UK in 1996 with historical data to 1988 yielded 148 pairs of data. The above frequency analysis illustrates the number of 15 minute periods in one year the 15 minute air quality standard of 100 ppb was breached. The air quality objective allows 35 breaches of this value (shown by the red line); 65 of the 148 observations failed to meet the air quality objective, albeit prior to 2005.

Figure 3.11 : Observed Sulphur Dioxide : AUN : 1979 to 1996

(source: Ireland, 1998)

The analysis for sulphur dioxide suggests episodes are associated with either single point sources dominating short term concentrations or poor dispersion conditions in urban coal burning areas. The Environment Agency (1999) has stated the major point source contributors (power stations) will be sufficiently regulated to significantly reduce the potential for breaching the 15 minute objective. Other point sources are generally regulated either by the Environment Agency or the local authority. More diffuse urban sources can be reduced through the specification of fuel supplies (via designation of smoke control zones, for example).



Data from 62 sites across the UK in 1998 are presented with reference to the 1997 UK air quality objective and EU limits. The 15 minute standard was breached at 6 sites, whereas the EU (hourly) limit was not breached and the EU (daily) limit only breached twice. Comparison of the number of breaches of the 15 minute standard (see Figure 3.11) confirms 1998 was a 'good' year for air quality.

All three statistical meters demonstrate some correlation with the annual mean, although previous analysis by Ireland (1998) with several years of data suggests there is only a very poor correlation between the annual mean and the 99.9th percentile of 15 minute means. Assuming an uncertainty of 30%, the above results suggest both EU limits will be achieved if the annual mean concentration is less than 8 ppb.

Figure 3.12 : Comparison of Short Term Criteria for Sulphur Dioxide

(source: data collated from the AQS, Table A39)

(source: data collated from the AQS, Table A39)

3.4 CONCLUSIONS

A number of air quality management tools are available to assess and improve air quality. Of the former, air quality objectives provide the benchmark to determine if air quality management is succeeding. Given the fundamental role of air quality objectives, it is useful to acknowledge the convergence of scientific and government opinion in setting them. The UK air quality objectives and EU limits are, to a large extent, equivalent.

Monitoring the success of air quality management requires direct measurement to provide infallible evidence of change in air pollution levels. Given the influence of meteorology, this evidence can only be accumulated by observing trends over several years. How we determine what monitoring data are infallible is the subject of continued debate not considered here. Given limited resources, there is a balance required between high quality data, available from only a small number of sites, and lesser quality data available at many locations. Based on discussions with environmental health officers, anecdotal evidence suggests the public will place greater trust in monitoring rather than modelling data, with no regard to the actual data quality.

The UK has extensive national monitoring networks and many local authorities operate automatic monitoring sites to national operating standards generating high quality data. Analysis of these data provides an assessment of current air quality with reference to the relevant air quality standards and, by projection of historical trends (assuming a 'business as usual' scenario) an assessment can be made of whether the relevant air quality objective will be achieved. Further analysis of these data provides for the development of surrogates and definitions of tolerance when using simpler monitoring techniques.

The results, summarised in Table 3.3, are based on work undertaken principally in 1997-1998 and are supported by work in practice; the subsequent designation of AQMAs following detailed (Stage III) review and assessment by local authorities.

The analysis described in this Chapter leads to the conclusion that not all of the fifteen UK air quality objectives included in the Air Quality Regulations 2000 require detailed consideration at all levels of governance.

At a national level, the UK Government is obliged, in accordance with 99/30/EC to monitor and report air quality with reference to EU limits. This is being achieved by minor modifications to the national monitoring networks and using national scale modelling techniques, where predictions are below a proportion of the EU limit. All of the UK air quality objectives derived from EU limits therefore have to be formally monitored and assessed to prescribed levels of detail by Central Government. Subsequent measures to ensure the EU limits are met are also the responsibility of Central Government.

At a local level, local authorities are obliged to assess the likelihood of achieving the UK air quality objectives, based on local circumstances and national trends in emissions, and exhibit best endeavours to improve local air quality. The inference from DEFRA guidance is that monitoring beyond the first round of R&A is only required for pollutants not expected to meet the air quality objectives. Of the 113 local authorities which had designated or announced intention to designate AQMAs by May 2001, Woodfield (2001a-b) reported:

- 40% had/would designate for nitrogen dioxide (annual mean objective) only
- 52% had/would designate for nitrogen dioxide (annual mean objective) and PM_{10} (24 hour objective) only
- 4% had/would designate for nitrogen dioxide (annual mean objective) and sulphur dioxide (15 minute objective) only

Woodfield (2001a-b) provided further analysis based on source type. Of the 113 local authorities, the proportion of AQMAs due to source types were:

- 70% traffic only
- 20% traffic and industrial
- 5% domestic and traffic
- 5% industry only.

There is no formal obligation for meeting air quality objectives at the regional scale. Planning at the regional level requires working to longer time scales. The approach to air quality management must be one of ensuring long term reductions in air pollution levels, as depicted in Figure 2.1. The planning role of regional governance and the regulatory role of the Environment Agency and local authorities needs to be clearly differentiated when assessing the contribution of industry to short term peak concentrations of sulphur dioxide and lead. However, the effect of meteorology giving rise to short term peak concentrations can be addressed at the regional level by ensuring the long term mean is sufficiently low to prevent such episodes occurring. There is strong evidence to support this in the case of nitrogen dioxide but less so for PM_{10} where the contribution from continental Europe can be significant during short term episodes.

In conclusion, this review of air quality objectives and ambient monitoring data suggests the key criteria for local and regional government in assessing and managing UK air quality are:

- the annual mean concentration of nitrogen dioxide, not to exceed 40 $\mu g/m^3$ (21 ppb) by 2005
- the annual mean concentrations of PM_{10}, not to exceed 28 $\mu g/m^3$ by 2005 or 17 $\mu g/m^3$ by 2010.

Both these criteria are applicable in the vicinity of housing, schools or hospitals.

The additional tools enabling air quality to be assessed (emissions inventory and dispersion model) have therefore been developed at the approximate regional scale based on annual mean emissions of nitrogen oxides and PM_{10}. The development of an emissions inventory and dispersion model is described in Chapters 4 and 5 respectively.

Table 3.2

Key Criteria for Assessing UK Air Quality

Pollutant	Air Quality Objective	Current Air Quality	Expected Compliance	Contribution to AQMAs	Monitoring Technique
Benzene	5 ppb, running annual mean, by 2003, 1.6 ppb by 2010	Roadside levels < 3 ppb	Yes, excepting specific industrial sources.	None designated.	Diffusion tube < 3.5 ppb
1,3 butadiene	1 ppb, running annual mean, by 2003	Roadside levels < 1 ppb	Yes	None designated.	Assume to be 20% of observed benzene levels
Carbon monoxide	10 ppm, running 8-hour mean, by 2003, 8.6 ppm by 2010	Roadside levels < 11 ppm	Yes	None designated.	Automatic analyser or electro-chemical device
Lead	0.5 $\mu g/m^3$, annual mean, by 2004, 0.25 $\mu g/m^3$, by 2008	Roadside levels <0.2 $\mu g/m^3$	Yes, excepting specific industrial sources.	Not known.	M-type sampler
Nitrogen dioxide	105 ppb, 1-hour mean not to be exceeded >18 times p.a, by 2005 (UK) or 2010 (EU)	All sites < 105 ppb, with few exceptions	Yes	100% (annual mean objective only)	Short term objective; diffusion tube > 22 ppb, annual mean
	21 ppb, annual mean, by 2005 (UK) or 2010 (EU)	Most urban and roadside levels > 21 ppb	No		Long term objective; diffusion tube > 15 ppb

Table 3.2

Key Criteria for Assessing UK Air Quality

Pollutant	Air Quality Objective	Current Air Quality	Expected Compliance	Contribution to AQMAs	Monitoring Technique
Particles	50 µg/m^3, 24-hour mean not to be exceeded >35 times p.a, by 2005 (Stage I) and >7 times p.a by 2010 (Stage II)	Ambient levels < Stage I limit	Stage I objectives already achieved, with few exceptions.	52%	Automatic analyser; Stage I and II: short term objectives met if corresponding annual mean objectives are achieved, subject to contributions from continental Europe
	40 µg/m^3, annual mean, by 2005 (Stage I) or 20 µg/m^3, by 2010 (Stage I))	Ambient levels < Stage I limit	Stage II objectives not expected to be achieved.		
Sulphur dioxide	132 ppb, 1-hour mean not to be exceeded >24 times p.a, by 2005	Ambient levels below 1-hour and 24-hour mean standards in 1998; breaches expected in other years.	Yes, subject to industry specific source control	4%	1-hour and 24-hour objectives; diffusion tube < 8 ppb, annual mean
	47 ppb, 24-hour mean not to be exceeded >3 times p.a, by 2005				
	100 ppb, 15 minute mean not to be exceeded >35 times p.a, by 2005	15-minute standard regularly breached			15 minute objective; automatic analyser

4 LOCAL TO REGIONAL SCALE EMISSIONS

This Chapter provides a review of available methodologies for the development of emission inventories. The use of a 'bottom up' approach is advocated to provide a direct link between atmospheric emissions and land use and transportation planning data. The development of an inventory of local to regional scale emissions is described for key pollutants from 1990 to 2020. A comprehensive evaluation of the emissions inventory tool is provided to demonstrate its robustness.

4.1 EMISSIONS INVENTORIES

The compilation of atmospheric emissions data is undertaken by a number of organisations around the world for a variety of reasons, including air quality management and monitoring progress in reducing greenhouse gas emissions. There are two common approaches: estimation of emissions from the 'top down' using fuel use statistics, for example, to provide aggregated totals classed by sector and estimation of emissions from the 'bottom up' using spatially defined activity data (Hutchinson and Clewley, 1996, DETR, 2000j). In practice, regional scale studies have required a combination of the two (Borrego *et al*, 2000, Sturm *et al*, 1999 and Lindley *et al*, 1996, 1998). On the largest scale is the inventory maintained under the United Nations Convention on Climatic Change which collates emission data expressed on a national basis provided by some 36 countries globally (United Nations, 1999). Guidelines for compiling national emissions of GHGs have been produced by the Inter-Governmental Panel for Climate Change (Bolin and Sundararaman, 1996). The next level down, with respect to the UK, is the CORINAIR inventory based on 50 km by 50 km grid squares across Europe and first completed in 1990 for the year 1985. The preparation of this inventory included the compilation of an Emission Factor Handbook with a number of default factors that may be used in the absence of more specific data (European Environment Agency, 1996). There are several examples of using the CORINAIR inventory as a reference for disaggregation of emissions to local land use, including that of Cosmi *et al* (1999) for the Basilicata region of Italy. For a number of years the DETR has been maintaining a UK National Atmospheric Emissions Inventory (NAEI: Goodwin *et al,* 1997, 2001). With the exception of carbon dioxide, this is based on 10 km grid squares and includes information for seven pollutants including nitrogen oxides and PM_{10}. Carbon dioxide emissions are reported nationally although DETR (2000a) has indicated regional disaggregation is being considered. Baldasano *et al*

(1999) provide an example of how greenhouse gas emission data can be compiled for a region (Barcelona) and reported as CO_2 per capita. In describing the compilation of district level inventories of greenhouse gases in India, Garg *et al* (2001) consider such disaggregation useful, both in terms of developing sound mitigation strategies at manageable smaller scales and improving national estimates, although no specific consideration of uncertainties is reported.

Experience from maintaining the NAEI enabled the DETR (1999e) to publish company guidelines for estimating GHG emissions using many of the techniques and default emission factors included in the IPCC and CORINAIR work. These guidelines form the basis of the CBI methodology developed by its Emissions Trading Group for companies to trade GHG emissions within arrangements to be introduced as part of implementing the Climate Change Levy (CBI Emissions Trading Group, 2000). These trading arrangements could include offsetting schemes with local government (such as the support of public transport by industrial developers).

The reported accuracy of the annual mean NAEI based on 10 km grid squares is shown in Table 4.1. The accuracy for 1 km grid squares is not yet reported but is likely to be less precise. These estimates of accuracy are useful in providing a means of determining the overall confidence in detailed modelling studies for the purposes of local authority review and assessment of air quality and for regional scale studies.

Table 4.1
Accuracy of National Estimates of Atmospheric Emissions in the UK

POLLUTANT	ACCURACY (+/-)
Methane	30 to 40%
Sulphur dioxide	10%
Nitrogen oxides	30%
Non-methane VOC_5	50%
Carbon monoxide	40%
Black smoke	20 to 25%
Note: Stated accuracy for carbon dioxide is 5% for the UK as a whole.	

(Source : Goodwin *et al*, 1997, 2001)

The first comprehensive urban emissions inventory was prepared by the London Research Centre (LRC; now part of the Greater London Authority) in 1995 for the West Midlands as part of the DETR Air and Environment Quality Research Programme (Hutchinson and Clewley, 1996). The authors of this inventory consider it suitable to be

used as a template for other UK local authorities should they wish to prepare their own emission inventories although no reference to validation or uncertainty analysis is made.

A series of emission inventories has now been published by the LRC based on the same methodology for the following urban areas (Buckingham *et al.* 1997a-c; 1998; Hutchinson and Clewley, 1996):

- Bristol
- London
- Greater Manchester
- Merseyside
- Southampton and Portsmouth
- Swansea and Port Talbot
- West Midlands.

In developing the West Midlands Emissions Inventory, the LRC set out its methodology for identifying suitable emission factors starting with those developed in the UK and, should those not be available, factors developed in Europe or elsewhere. This methodology has been adopted in the course of this research. Based on first hand experience in compiling the urban inventories, Sadler (1998) provides an overview of emission sources, their significance and ease of estimation, in terms of the quality of emission factors and availability of activity data. This work suggests emission sources can be ranked in terms of data quality; see Table 4.15.

In considering the application of the detailed urban emission inventories for the purposes of Stage III R&As, Vawda (1998) highlighted the following shortfalls:

- traffic flows based on traffic models with a basic vehicles mix and extrapolated by limited traffic counts
- greater resolution required in variations in traffic speeds along links
- better information required of vehicle types / mixes
- no future predictions made for 2005
- PM_{10}, benzene and 1,3 butadiene estimates generally poorer than other pollutants
- emissions expressed as annual mean
- no temporal / seasonal variations in vehicle emissions assumed.

These shortfalls suggested significant improvements will be required to use the urban inventories for Stage III R&As. Considerable investment was made in preparing those urban inventories (for example, in excess of £50,000 for the West Midlands Inventory) which is unlikely to be sustained in estimating temporal variations in emissions in the future. A significant step change in resource requirements is required to temporally disaggregate emissions data. The usual approach for traffic emissions is to apply national or local diurnal and seasonal traffic profiles, as described by Lindley *et al* (1998) although more complex statistical profiling techniques, as described by Niemeier *et al* (1999) are available. Similarly, Carslaw *et al* (2000) describe the use of operational data to profile annual mean emissions data from large combustion plant in the Thames Estuary. There is a lack of DETR guidance on estimating temporal variations in emissions. In this study the preparation of an emissions inventory of nitrogen oxides and particulates has been limited to estimating annual mean emissions. This is not considered to be significant for, as demonstrated in the preceding chapter, most concern for nitrogen dioxide and PM_{10} lies with achieving air quality objectives based on the annual mean or derivative. Furthermore, emissions of GHGs are expressed as annual means, enabling common activity data to be used thus simplifying the inventory compilation.

Although the urban inventories include carbon dioxide, Vawda's review (1998) was limited to Stage III R&As. Notwithstanding the points cited above, the urban inventories are, therefore, useful as a starting point for compiling GHG emissions. These are usually expressed as annual means and hence, issues of temporal resolution are not applicable. There are differences in spatial resolution and the methods used for compiling local air quality related pollutants and GHG emissions. The NAEI for GHGs refers to the IPCC guidance and the use of energy or fuel consumption data for estimating emissions from a country as a whole. Similarly, the DETR and CBI Emissions Trading Group guidance refers to aggregate fuel consumption data for company vehicle fleets in preference to mileage rates. The use of such methods supports the observation made by Reynolds and Broderick (2000) that emission inventories are obsolete by virtue of the time taken to compile and verify the data. Lindley *et al* (2000) describe the use of national and regional trends in activity or proxy data to extrapolate to future and past years. This is in line with the urban inventories and guidance for compiling emission inventories (DETR, 2000j) which provide methods for estimating

emissions using spatially derived activity data, such as traffic flows on specific road links; recognising the limited availability of fuel sales data.

The use of activity data also has the advantage of enabling analysis of various source control measures (development of AQAPs) beyond gross fuel substitution. By developing techniques for estimating GHG emissions based on common activity data, the effects of air quality management can also be considered in terms of GHG emissions in both temporal and spatial scales, and in terms of broader local environmental issues such as noise.

A further advantage of using activity derived emissions data is the ability to link with data compiled and forecast by government for planning purposes. National projections of energy and fuel demand are based on gross trends in the economy, population, land use and transport, taking little account of regional variation. These projections are then amended to meet national objectives with the amendments often passed on to more local government, industry or business as targets; almost every measure within the Climate Change Programme is an example of this. For local government to influence these trends, it needs to incorporate these targets within the planning process. By linking emissions to activity data rather than gross trends, local government is able to assess the impact of adopting various measures or policies in preparing and implementing its various plans. However, a further important point made by Lindley *et al* (2000) is the inevitable differences in methodologies for compiling inventories including updates, such as the UK National Atmospheric Emissions Inventory (Gilham *et al*, 1994, Goodwin *et al*, 1999, 2000) This presents two potential problem in comparing inventories. For inventories of the same geographical area Lindley and Longhurst (1998) recommend careful documentation and comparison of the methodologies used. For inventories updated, Gilham *et al* (1994; see also Sturm *et al*, 1999) rework the estimates for previous years using the updated methodology. How this impacts on policies based on emission reduction targets (eg. CO_2) is unclear at this time.

Activity data such as traffic flows, population density and land use classification, have therefore been used as the common reference for preparing a regional (annual mean) emissions inventory of nitrogen oxides, particulates and carbon dioxide.

The review of available literature has been restricted to the following main sources for compiling a regional emissions inventory, although significant reference to published work has been made, particularly with regard to verification (Section 4.4):

- the **DETR databank of Atmospheric Emissions** *(www.naei.og.uk/)* was prepared by LRC in collaboration with RSK Environment. It is considered to be the definitive reference source for actual emissions data and emission factors expected from industrial and other activities in the UK. This database makes significant use of work published by NETCEN: Salway *et al* (1997) and Goodwin *et al* (1997, 2001), and Lloyd's Register (1995). The database was designed to provide data and technical information to local authorities undertaking a local emissions inventory. The database does not include CO_2 emission factors, except those derived from the NAEI or published by Lloyd's Register, and is therefore limited for the purposes of building a regional emissions inventory.

- the DETR (1999) **Design Manual for Roads and Bridges**. In developing guidance for Reviews and Assessments (see chapter 2) the DETR commissioned the Transport Research Laboratory to provide an updated version of the DMRB as a screening method for estimating roadside air quality for a Stage II R&A. The DMRB also includes speed and year related emission factors for a range of individual and aggregated vehicle types for use in a Stage III R&A, and factors for estimating carbon dioxide emissions.

- The West Midlands (and other) **Urban Emissions Inventories** considered best practice. The West Midlands Inventory covers a total area in excess of 899 square kilometres and a population of 2.5 million; approximately double that of Kent.

- The **London Energy Study** prepared under the European Urban and Regional Energy Management Programme by LRC (Chell and Hutchinson, 1993). The methodologies used to estimate energy use in the four sectors of domestic, commercial, industrial and transport (road, rail, air) can be extended to estimate atmospheric pollutant emissions. The London Energy Study also considered measures to reduce energy consumption. Such measures have an obvious role to play in air quality management and reducing GHGs.

- The **National Atmospheric Emissions Inventory for the UK** is compiled by NETCEN. Spatially disaggregated data for 1995 are currently from the DETR website for nitrogen oxides, particles and other air quality related pollutants. Forecasts of future disaggregated emissions are not available although forecasts of future ambient concentrations are (Stedman *et al*, 1998, 1999a-b, 1999d, 2001a-c).

4.2 METHODOLOGY FOR ESTIMATING REGIONAL EMISSIONS

The references identified above provide a variety of emission factors which, in conjunction with activity data, can be used to estimate emissions of air quality related pollutants, such as nitrogen oxides and particulates, and GHGs including carbon dioxide. The regional atmospheric emissions inventory compiled for this study includes three categories of sources:

- **Industrial sources :** Part A, Part B and other
- **Transport sources :** road, rail, shipping and airports
- **Population / land use related sources** : domestic, commercial land use categories and agriculture.

Industrial sources are represented in the regional emissions inventory as either point sources (Part A Processes) 250 m by 250 m area sources (Part B and other). Transport and other sources are also represented as 250 m by 250 m area sources.

Emissions from significant industrial sources outside the region also need to be considered. Part A Process emissions from the Thames Corridor (London and Essex) were also included within the modelling study although not within the emissions compilation. The import of nitrogen oxides and PM_{10} from other sources outside the region has been incorporated within the modelling procedures with reference to rural background monitoring data and reported contributions to secondary particulates; see Section 5.4.

4.2.1 Part A and Part B Processes

The NAEI includes industry as individual Part A Processes and aggregated emissions from other ('small') industry per 1 km^2. National emissions for all industry are estimated from Energy Statistics (fuel consumption). From this total (for each pollutant) the emissions from Part A Processes is removed. The category of small industry therefore includes Part B Processes and smaller industrial sources. The national emissions are distributed based on the Science Policy Research Unit (SPRU) study of boiler insurance policies (boiler size and fuel usage by 10 km grid squares for 1992). Land cover data maintained by the Institute of Terrestrial Ecology (ITE) was used to

apportion emissions to 1 km grid squares with urban land use. Emissions are estimated using fuel emission factors for different levels.

LRC used the Chemical Release Inventory for Part A Processes, as validated by the Environment Agency. Part B Processes have been recorded based on data held by the local authorities. LRC noted the high degree of participation from these authorities in supplying these data. Information on large boiler plant was also collected by LRC, although how this was achieved is not detailed. LRC make the comment that the availability of emissions data for Part B Processes is scant and a more formalised system, based on Part A Processes, needs to be established. Sadler (1998) considers Part A processes to be the most significant source of local air pollution (rated 3, on a scale of 0-3) along with road traffic, with Part B processes marginally less significant. Sadler also notes a wide inconsistency in the quality of emission factors (1-3, on the same scale) and availability of activity data (1-3, same scale) for Part A Processes. The quality of emission factors and availability of data for Part B processes are considered generally to be lower, 2 and 1, respectively.

Industrial emissions data in Kent and Medway have been compiled for Part A and Part B Processes. The emissions data for Part A Processes for 1997-2000 were derived from a combination of interrogation of the Chemical Release Inventory, reference to a study of the Thames Estuary by Carslaw *et al* (2000) and personnel communication with the Environment Agency. Extrapolation for other years (1990 to 2020) was based on judgement and discussions with the Environment Agency. These data provide direct input to the dispersion model as elevated point sources (see Section 5.2). Similar to the LRC findings, the quality of data for Part B sources in Kent and Medway is relatively poor. Data on the location and type (with reference to the relevant Process Guidance Note) of each Part B Process were compiled. Emission estimates were then made based on the type of process. For each combustion process, the fuel type and thermal rating was defined and subsequently the emission rates estimated using standard thermodynamic combustion calculations. The general assumption was made that combustion processes switched from oil to gas between 1990 and 1997, the exception to this being waste oil burners. Non-combustion emissions were derived from a combination of local knowledge and judgement to derive activity data, such as plant capacity and operating hours, and emission factors included in the DETR Atmospheric

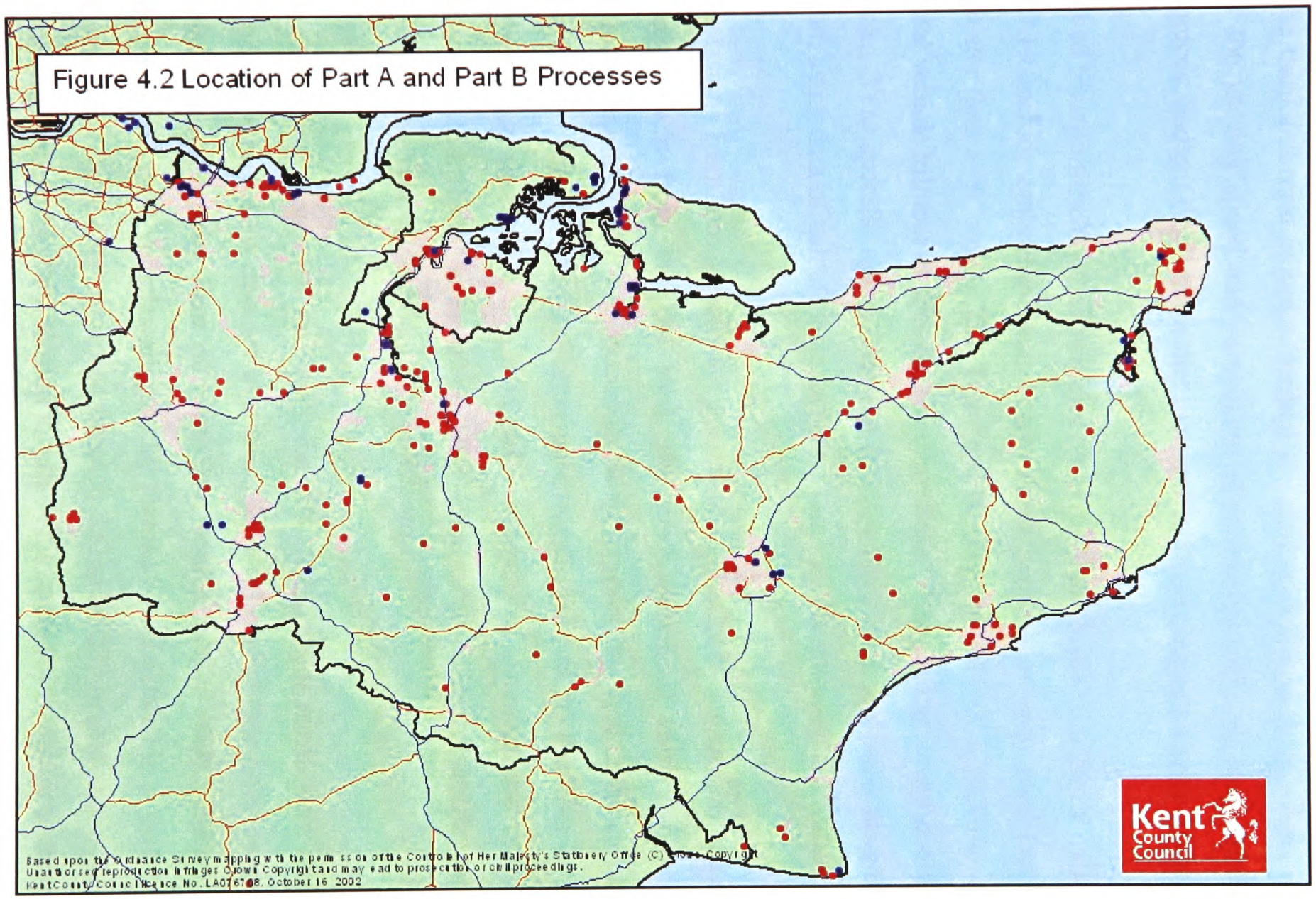
Emissions Databank. These sources were modelled as area sources (see Section 4.2.2). Although not ideal, this approach is considered to provide a worst case estimate of ground level contributions from Part B Processes; by implicitly assuming significant local building wake effects (see Section 5.3.4).

The study area includes 162 sources associated with Part A Processes and 421 Part B Processes. The location of each Part A and Part B source is included in Figure 4.1 illustrating the general industrialisation towards the north east corner of the study area. This Figure also locates Part A Processes outside the study area which would be expected to have an influence. These have been included implicitly as part of the background contribution to NO_x and PM_{10}. This approach is supported by the results of Carslaw *et al* (2000) who suggest the annual mean NO_2 contribution from Part A Processes along the whole of the Thames Estuary in 1998, from Slough to the Isle of Grain, was between 0.6-1.4 $\mu g/m^3$ in East London, 0.8 $\mu g/m^3$ in Thurrock, along the northern shore of the East Thames Estuary, and 1 $\mu g/m^3$ in Rochester. All of these contributions represent 3.5 % or less of the Air Quality Objective Value. One of the key findings of the work by Carslaw *et al* (2000) was the dominance of Grain and Kingsnorth Power Stations on air quality in the east Thames Estuary; both of which are included in the Kent and Medway emissions inventory.

A review of other published inventories (cited in this chapter) suggests industrial emissions are assumed to be static over time. The inventory developed for this research accounts for fuel switching (from oil to gas) economic growth and planned closures, upgrades or new installations.

4.2.2 Other Industry

In addition to Part A and Part B Processes, point sources in the West Midlands are identified as including central heating plants serving large groups of buildings such as hospitals, boiler plants supplying process heat to industry. Basic data in terms of land use are available but do not include details of plant specification or fuel consumption. The technique adopted in this study is described in Section 4.3.12.



(Part A Processes in blue, Part B Processes in red)

4.2.3 Major Roads

Delgado *et al* (2000) describes three principal methods to estimates traffic emissions. The simplest is to take account only of the total kilometres travelled by the whole vehicle fleet. The second is to assign speed-dependant emission factors for each trip. The third and most complex method is called modal modelling, where account is taken of the varying speed and subsequent emissions during each stage of the trip. Of all sources, Sadler (1998) notes that road traffic has the highest degree of data quality.

Road transport includes all vehicles on major roads and minor roads. Major roads in the NAEI include emissions from Trunk, Primary and Motorway road links (i.e. any road greater than a 'B' road). The Transport Division of the DETR provides estimates of the total vehicle kilometres travelled in these links. The average speed is estimated for each link. Year specific speed emission profiles have been compiled for each vehicle type and, using data of vehicle type on each road link, the emission calculated.

Emissions from minor roads are estimated nationally by subtracting the emissions from major roads from the UK total, the latter calculated from national total vehicle kilometres travelled. The remainder is apportioned by land cover (urban or rural) and population density. In rural areas an average speed of 56 km/hour is assumed and in urban areas (defined in the land cover database as concrete and roofing) an average speed of 28 km/hour is assumed. The same speed emission profiles are then used for each vehicle type to calculate total emissions.

In compiling a regional inventory for this study, major roads were defined simply as those for which flow data are available. In practice, the threshold was roads having an annual average daily flow of more than approximately 4000 vehicles; similar to the threshold of 5000 vehicles adopted by Delgado *et al* (2000) in a study of emissions in Catalonia. These data are stored for each 250 m by 250 m grid square in the region with each link assumed to be 250 m in length. This approach is similar to that of Valkonen *et al* (1996) who represented roads as 1 km grid squares for estimating urban emissions in Espoo, Finland but represents a potential calculation error as the road link may be shorter or longer. Lindley *et al* (1998, 2000) used the GIS to map links as lines and hence compute actual link lengths within each grid square. Although this overcomes

this potential error, the authors acknowledge the start and end of each link was defined by the co-ordinates of the relevant road junction identified by the Greater Manchester Traffic Model with the link assumed to be a straight line between these points implying an underestimation of total link length and hence, emissions. In either case, the degree of error is not expected to be significant over the study area. To estimate speed related vehicle emission factors for NO_x, PM_{10} and CO_2 on these roads, the Design Manual for Roads and Bridges (DMRB) (DETR, 1999, revised by DEFRA, 2002) method was used by:

1. using the standard emission factors for a 1996 fleet car travelling at 100 km/h
2. correcting for speeds between 5 and 140 km/h for cars and HDVs
3. correcting for years 2000, 2005, 2010 and 2020.

This approach is consistent with DETR (2000j,l) guidance for R&A but differs from that used to compile the NAEI.

The 1999 version of DMRB does not include emission factors for years before 1996 and the current (2002) revision does not include CO_2 or emission factors beyond 2010. Emissions for 1990 were back projected from the tables of year correction factors (for cars and HDVs) based on the following suppositions:

- the significant change in legislation over this period was the introduction of three way catalytic converters which, although mandatory for all petrol cars registered after 1993, took place over several years before this
- the relatively low turnover of the national fleet (<10% per annum) means any changes due to the introduction of new vehicles with lower emissions takes place over several years; the resultant lag dampens significant changes in fleet emissions enabling extrapolation over periods of a few (<5) years
- CO_2 emissions from cars increased slightly between 1990 and 1995 with increased vehicle size, emissions from HDVs remained unchanged.

The year correction factors for cars and HDVs between 1996 and 2020, including back projection to 1990 based on discussions with the Transport Research Laboratory (DETR, 1999g, Hickman, 2000) are illustrated in Figures 4.2 and 4.3, including the revised emission factors for NO_x and PM_{10} (DEFRA, 2002b). The effect of the recent voluntary agreement by car manufacturers to reduce CO_2 emissions from new cars by at least 25%

from 2008 (DETR, 2000a, pp84-85) has been included assuming an annual fleet turnover of 10% with 20% of cars being diesel.

Between 1990 and 2010, vehicular emissions of NO_x and PM_{10} were expected to have reduced significantly by more than 75% although the revised emission factors suggest the reduction will not be so marked. This is due to a previous optimistic assumption of the degree new vehicles would comply with new emission limits, most markedly in the case of PM_{10}. In contrast, emissions of CO_2 rose slightly up to 1995 with the trend for larger cars (eg. four wheel drive utility vehicles). The effects of the voluntary agreement are evident up to approximately 2015. The increase in HDV CO_2 emissions from about 2010 is the result of introducing larger vehicle sizes. Although not reported, revised emission factors for CO_2 are not expected to be significantly different as the key factor is engine size and hence, fuel consumption, rather than the performance of catalytic converters or particulate traps.

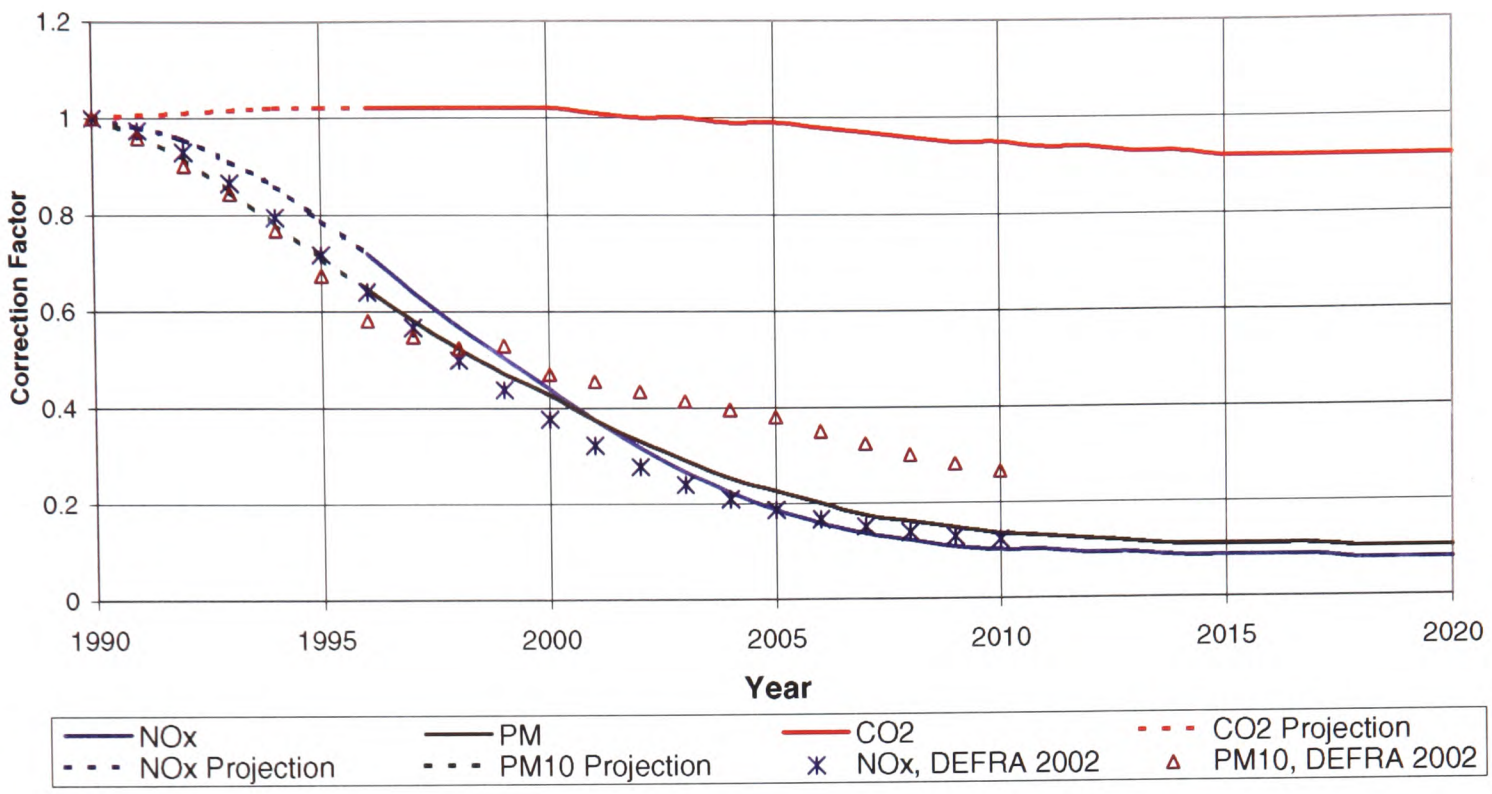


Figure 4.2 : Year Correction Factors for Light Duty Vehicular Emissions

(Source: DETR, 1999g and Hickman, 2000)

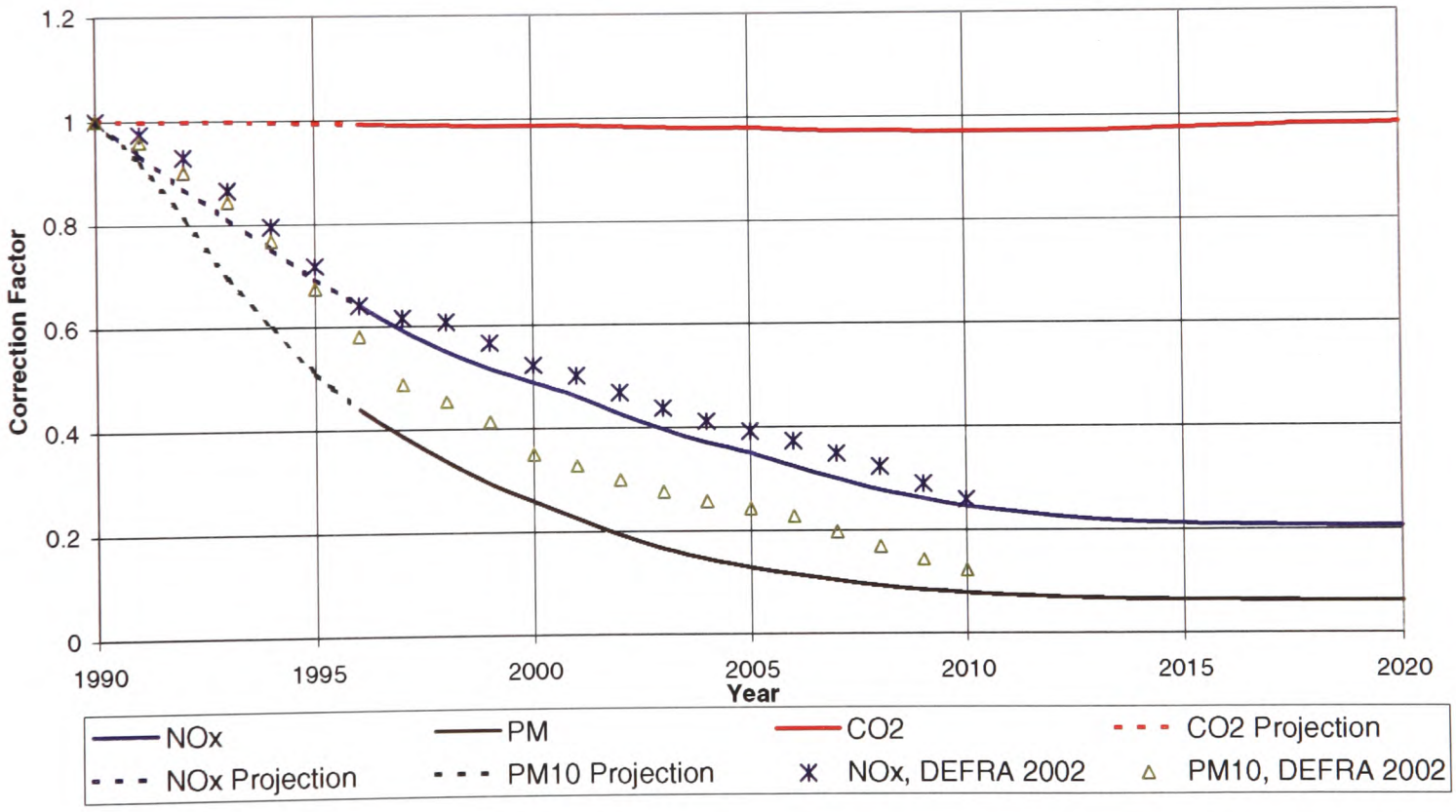


Figure 4.3 : Year Correction Factors for Heavy Duty Vehicular Emissions

(Source: DETR, 1999g, Hickman, 2000 and DEFRA, 2002)

Raw traffic data were available for this study for the years 1997, 2005 and 2010. Extrapolation for other years was made using the following factors (KCC, 2000b; Figure 6):

Year	**Index (1996 = 100)**
1990	92
1996	100
2000	108
2005	115
2010	120
2020	130 *

* Projections for 2020 not available from KCC but are assumed to follow the same trend.

The compilation of emissions estimates from major roads in this research is therefore unique in two aspects. Firstly, each link is represented as a 250 m by 250 m area source rather than a 1 km by 1 km area source or line source. Although this precludes the inventory being used directly for very local scale modelling, in defining the boundary of AQMAs for example, the loss of spatial resolution is not considered significant over the local to regional scale. This local resolution issue was overcome to a large extent by developing a method for modelling roadside air quality, as described in Section 5.5. Secondly, inventory includes estimate of carbon dioxide emissions back projected to 1990 from published data for 1996.

4.2.4 Domestic Traffic on Minor Roads

Emissions from minor roads, for which flow data are not available, have been estimated using the procedures described below. In contrast to the NAEI, this method is a 'bottom up' approach using observations of trip generation for different land use categories. This method requires interrogation of the Kent Landuse GIS for population densities and urban or rural classification. The results are expressed in tonnes per head of population per annum for each 250 m grid square:

(a)

trips per annum	=	8 trips per household per day (JMP, 1996)
		*365 / 2.2 people per household
	=	664

(b) minor traffic emissions (urban areas)	=	664 * car emission factor @ 28 km/h * 5 (assumed average distance to nearest urban road with traffic data) * 1/1,000,000
(c) minor traffic emissions (rural areas)	=	664 * car emission factor @ 56 km/h * 10 (assumed average distance to nearest rural road with traffic data) * 1/1,000,000

Notes: (1) A cold start is assumed to last the first 2 km of the journey, as per the West Midlands Inventory and DEFRA (2002b)
(2) People per household based on total population of Kent and Medway divided by total number of tenures (585341, 1991 census).

This method is similar to that used by LRC and assumes cold start emissions, in grammes per cold start trip are numerically equivalent to speed related emissions in grammes per kilometre. Lindley *et al* (1999) estimated the total number of cold starts in the UK to be 72.5 % of all trips in urban areas based on work by ETSU published in 1994 and assigned cold start emission factors, also published by ETSU, accordingly. CORINAIR includes a method for estimating the proportion of cold start emissions for urban trips (eg. Delgado *et al*, 2000, Reynolds and Broderick, 2000 and Mensink *et al*, 2000; the latter adopting an amended set of emissions factors derived from the COPERT II programme). Minor emissions from HDVs are not included. The emission factors, summarised in Table 4.2, have been applied to a population database disaggregated by 250 m squares for the base case years between 1990 and 2020. Population data were derived from a combination of census and forecast data for each district. The emissions for the total trip are therefore included within each 250 m square. Although this results in some spatial inaccuracy, the contribution from minor road emissions to local air quality is not considered significant. Over a typical study area of more than a few kilometres square, the total emissions will be representative. By accounting for local shifts in population over time, the method developed for this research is considered an improvement from other published techniques.

Table 4.2
Minor Urban and Rural Traffic Emission Factors
(Tonnes per head of population per annum)

Minor Road	Year	NO_x	PM_{10}	CO_2
Urban	1990	0.00468	0.000232	0.7175
	1997	0.00333	0.000156	0.7322
	2000	0.00253	0.000118	0.7322
	2005	0.00156	0.000079	0.7102
	2010	0.00106	0.000054	0.6809
	2020	0.00090	0.000046	0.6590
Rural	1990	0.01049	0.000339	0.5250
	1997	0.00772	0.000227	0.5357
	2000	0.00557	0.000176	0.5357
	2005	0.00311	0.000121	0.5197
	2010	0.00205	0.000081	0.4982
	2020	0.00181	0.000064	0.4822
Notes:	NO_x and PM_{10} emissions calculated using revised DEFRA (2002) emission factors. CO_2 emissions calculated using DETR (1999) factors.			

(Source : this study)

4.2.5 Public Car Parks

A method for estimating emissions from stationary traffic was developed using the DMRB during the Heathrow Terminal Five Public Inquiry (Hackman, *personal communication*). This method assumes vehicles in parking areas are travelling at an average of 5 km/h. Time related emissions from parking areas can be estimated per vehicle assuming each vehicle takes an average of ten minutes to park (including entrance and exit):

[DMRB emission factor @5 km/h] : g/km * 5 / 6 = g/vehicle ...(8)

Using this method requires activity data in the form of car park patronage data which was expected to be available in terms of revenue. Such data were not available during the course of this research and hence, car park emissions have not been explicitly included. The contribution from these sources is not expected to be significant across urban areas as a whole.

4.2.6 Generated Trips by Developed Land Use

The Trip Rate Information Computer System (TRICS) database maintained by JMP (1998) has been reviewed to create a table of road vehicle trips generated by land use classification. These are provided in Table 4.3. Vehicular emissions were estimated assuming a speed of 28 km/h, with cold starts and with each trip being 500 m before a main road is reached.

In this study, the land use classifications in the maintained databases do not correlate fully with those detailed in the table above. Categories B1A, B1C, B8 and SG included in the land use databases have been assigned to B1B. Similarly, C3 has been assigned to C1.

Neither historical nor forecast data from the land use databases are available, although the Structure Plan (eg. KCC, 1993) includes targets for economic growth by sector. These are derived from regional economic forecasts, in terms of gross employment, which themselves are based on national estimates. The employment forecast for the southeast region published in 2000, is included as Figure 4.4. This forecast represents the 'average' for the region and is used as a basis for planning economic development on regional and subregional levels. Economic growth targets included in Structure or Local Plans are therefore based on a local government's aspirations for meeting or achieving more than the regional forecast within different sectors. KCC, for example, aspires to achieving more than the regional forecast as a result of developments such as Ebbsfleet and Kent International Airport (KCC, 1993; Policies ED1, ED2, ED3). At the time of this study, more local forecasts or targets of economic development were not available. The regional economic forecast was therefore used by coupling with current land use data for the current year to generate base case years between 1990 and 2020.

This method is limited in spatial detail but, as more local forecasts are developed, has the potential to be resolved to at least the district council (Local Plan) level. For the purposes of this study, the contribution of land use class related emissions to local air quality is expected to be minimal and is not considered significant. DETR (1998b, 2000j) and draft DEFRA (2002b) guidance does not address this source although its contribution to greenhouse gases will be significant (compare Figures 6.1 - 6.3). The

method is considered valid for estimating GHG emissions aggregated across the study area. In contrast to the 'static' inventories cited in this chapter, this methodology allows for a direct link between land use and emissions to be established on both temporal and spatial scales.

Table 4.3
Traffic Generation Factors By Land Use Class

Land Use Class		Variable	Traffic Generation Factor	
Code	Use Class		Trips per Day per 100m^2	% HGVs
A1	Retail	Gross Floor Area	150	2
A2	Business (office)	Gross Floor Area	10	0
A3	Catering	Gross Floor Area	98	2
B1 B1B B1C	Business (non- office)	Gross Floor Area	10	20
B2	General Industry	Gross Floor Area	15	50
	Warehousing	Gross Floor Area	5	98
C1	Hotels	No. of bedrooms	6 per bedroom	0.1
C2	Institutions	Gross Floor Area	11	1
D1	Non Residential Institutions	Gross Floor Area	11	1
D2	Sports and recreation Halls	Gross Floor Area	0.4	1

(Source : this study)

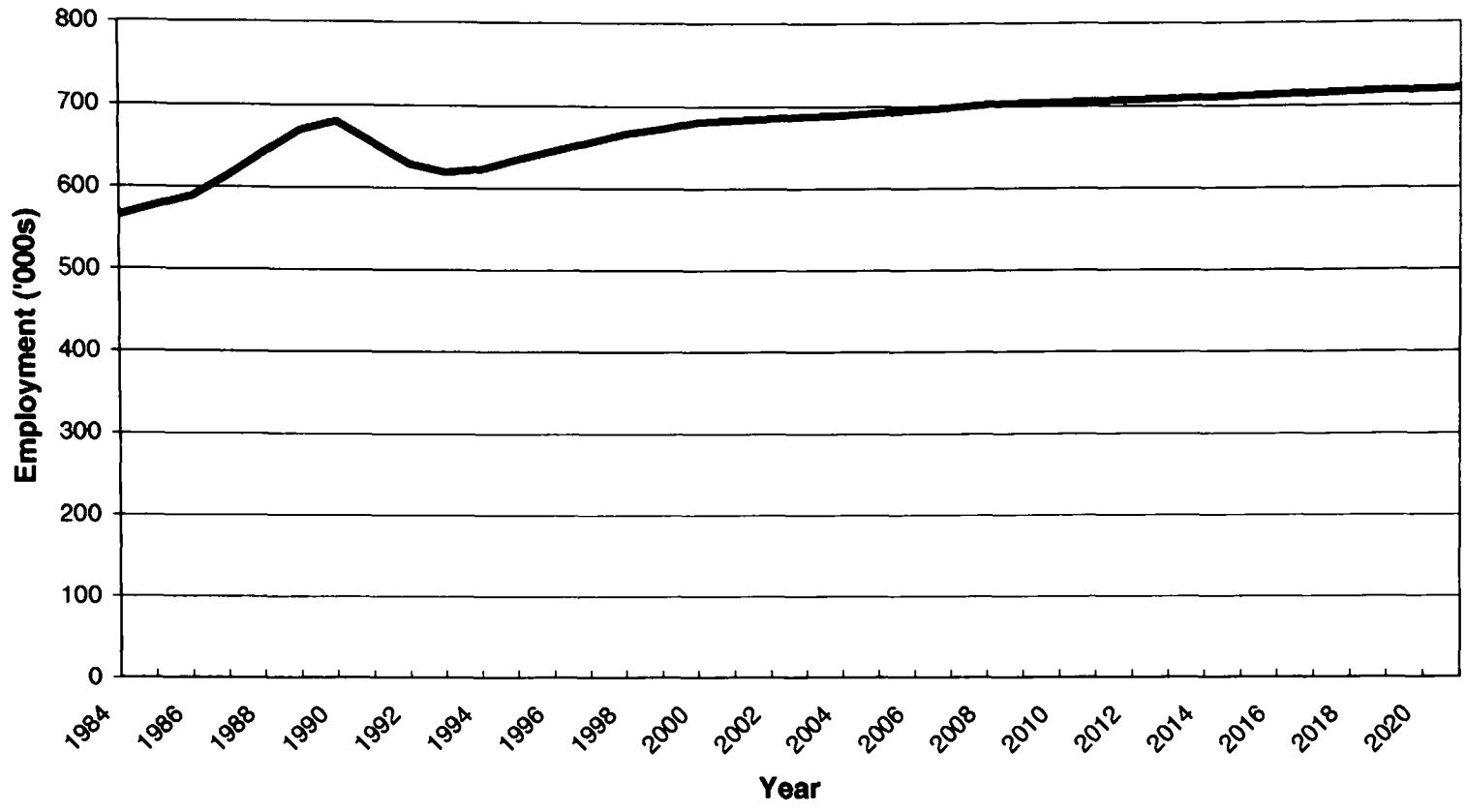


Figure 4.4 : South East Employment Forecasts: 1984 – 2021

(Source: BSL, 2000)

4.2.7 Road Dust

In compiling the urban inventories, LRC considered two factors for road surface dust emissions. The first, published by the USEPA and recently criticised by Venkatram (2000) led to emission estimates well in excess of that expected. A second emission factor (of 0.1 g/km) was used based on studies undertaken in London in 1983. LRC concluded that neither of these factors was sufficiently robust to include an estimate of road dust emissions. This view is supported by Sturm *et al* (1999) and earlier work by the Quality of Urban Air Review Group (QUARG, 1996). Reynolds and Broderick (2000) adopt a different view; using the USEPA factor to estimate emissions in Dublin, to be less uncertain than not accounting for this source at all. In responding to criticism by Venkatram (2000) who demonstrated by statistical analysis of the original field data that this emission factor may under or over estimate actual emissions by a factor of 14, the originator of the emission factor (Nicholson, 2001) considered vehicular exhaust emissions to be far more dominant in urban areas although work by the Airborne Particles Expert Group (1999) suggests that re-suspended dusts may represent 50% of the total roadside increment of PM_{10}. However, measurements of road traffic emissions

made in a Swiss tunnel by Wengartner *et al* (1997) suggest the contribution of tyre wear to traffic emissions is approximately 2% of total suspended particulates with the size range being between 20 and 40 μm mean diameter. Sadler (1998) does not refer to this source.

The regional emissions inventory compiled for this study does not include road surface dust emissions directly. In assessing local air quality, this source is accounted for in the use of ambient monitoring data to estimate ambient PM_{10} concentrations. This approach is consistent with that of Stedman *et al* (2001a). Emissions of NO_x and CO_2 are not associated with road surface dust.

4.2.8 Rail

The NAEI derives railway emissions from the total UK emissions from this sector distributed by the length of railway links. A similar method is used in the urban inventories with Sadler (1998) suggesting a high degree of confidence in estimating emissions from this source.

With the exception of one small line, all trains in the study area are electrified. Emissions from this source have not been included in this study.

4.2.9 Shipping

Emissions from shipping are included for different vessel types based on results from the *The Marine Exhaust Emissions Research Programme* initiated in 1989 in response to discussions at the International Maritime Organisation regarding the reduction of air pollutants from shipping. The research was undertaken by Lloyd's Register (1995) and has been further developed by Cooper and Andreasson (1999) who report emissions from individual ships in Göteborg, Sweden. Sadler (1998) considers shipping one of the easier sources to estimate.

The Lloyd's Register research team suggest emissions fell significantly in squares adjacent to the Dover Straits where shipping lanes diverge with observed reductions in shipping densities. Estimates by Lloyds Register are reported as being higher than those used by EMEP for acid deposition estimates.

The NAEI data for 1995 includes national NO_x and SO_2 emissions for shipping proportioned by port arrivals and assigned to the general 10 x 10 km grid square including the port. These have been aggregated as emissions located within 250 m grid squares along shipping lanes and port entrances.

For the purposes of this study, the NAEI shipping NO_x and SO_2 emissions data for 1996 have been factored for each year using the annual passenger volumes provided in Table 4.4 below as a proxy. PM_{10} and CO_2 emissions are derived from the SO_2 data based on ratios for shipping fuel published by Salway *et al* (1997). This method assumes emissions of these three pollutants are directly related to fuel consumption; the NAEI includes emission factors for black smoke but not PM_{10}. The latter were estimated using the ratio of these two pollutants reported for non-road transport aggregated for the UK in the same report.

Table 4.4
Passenger Volumes in Regional Ports

Year	Dover	Ramsgate	Folkestone	Sheerness
1990	*17000000*	*3300000*	*800000*	*900000*
1995	17905061	3392192	727820	74287
1996	18978105	2794639	917883	21000
1997	21322433	1713853	866988	-
2000	*19547192*	-	*719767*	-
2005	*20151052*	-	*586115*	-
2010	20754913	-	452464	-
2020	*21962634*	-	*185161*	-

Notes: Numbers in italics based on a linear regression from reported data for 1995 –1997 (data supplied by KCC Highways and Transportation)

(Source : this study)

4.2.10 Airports

The NAEI includes emission estimates for nine national airports and emission factors for large and smaller airports with a strong correlation between emissions and aircraft movements, albeit based on a small sample size; see Figure 4.5.

This relationship cannot be extrapolated from large aircraft movements (associated with national airports) to smaller aircraft movements present at regional airports. However, this observed correlation provides some confidence in relating emissions to aircraft movements although Sadler (1998) does not assign a high degree of confidence in estimating emissions from this source category.

There are three small airports in Kent and Medway: at Rochester; Lydd and Manston. Annual emission from these airports have been estimated using the NO_x, SO_2 and CO_2 emission factors for small airports published by Salway *et al* (1997, Table A17; the same method for estimating PM_{10} emissions from shipping was used). Aircraft movement data were collated by contacting each airport manager and used as a proxy for extrapolating annual mean emissions for each year 1990-2020 in a similar method to that adopted for estimating shipping emissions. The results are summarised in Table 4.5. Note that the proposed airport at Cliffe is not included. This proposed development, discussed further in Section 6.4, provides an example of the need to include future development within the inventory.

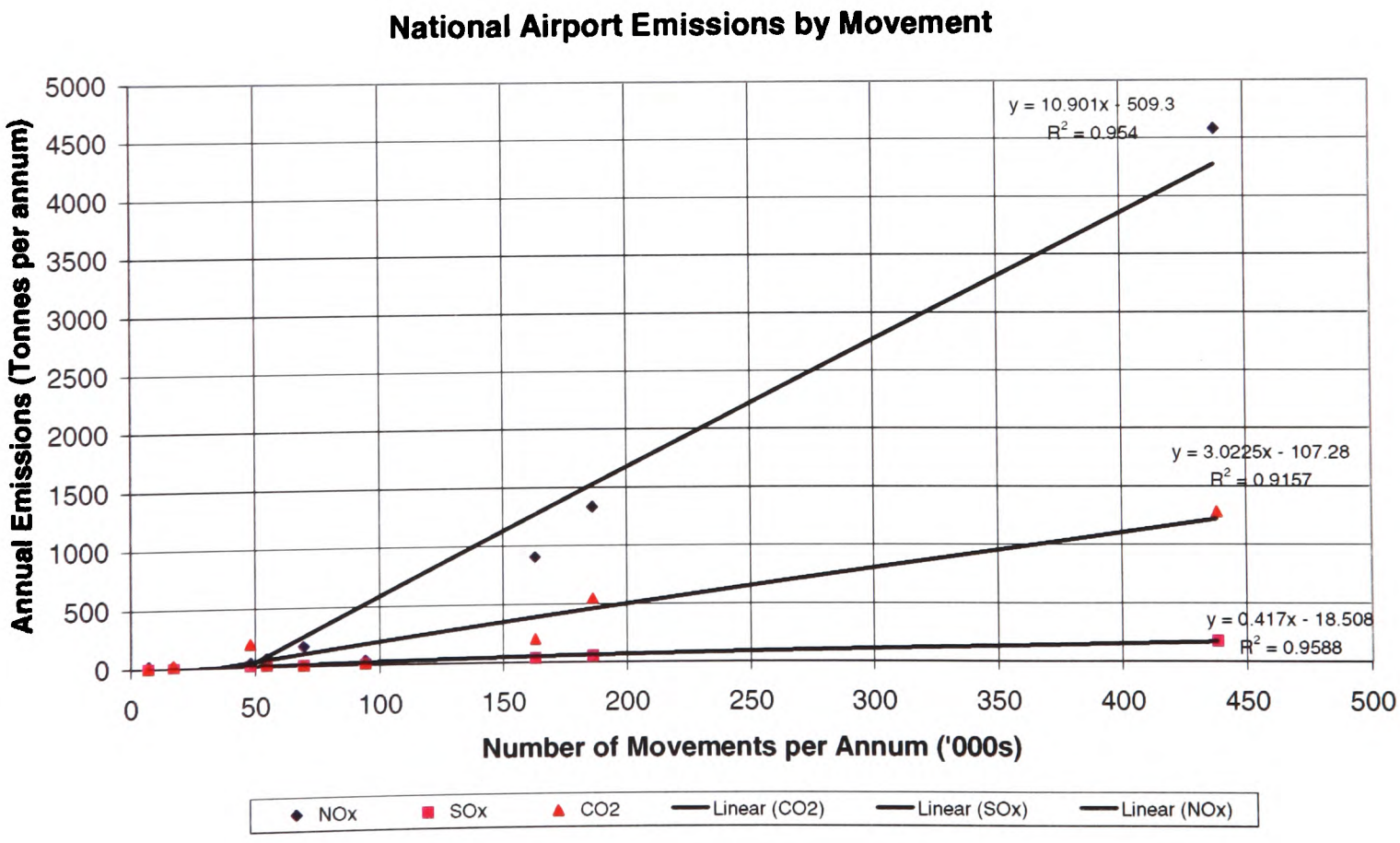


Figure 4.5: UK National Airport Emissions by Movement

(Source: Salway *et al* (1997)

Table 4.5
Airport Emissions for Small Airports (Tonnes per annum)

Airport	Year	No. of aircraft movements per annum	NO_x	PM_{10}	CO_2
(kg/ movement)			7.88	0.117	401
Rochester	1990	40230	317	4.71	16132
	1997	31852	251	3.73	12773
	2000	25868	204	3.03	10373
	2005	24139	190	2.83	9680
	2010	20000	158	2.34	8020
	2020	20000	158	2.34	8020
Lydd	1990	26000	205	3.05	10426
	1997	30000	236	3.52	12030
	2000	40000	315	4.69	16040
	2005	45492	358	5.33	18242
	2010	50984	402	5.97	20445
	2020	56476	445	6.62	22647
Manston	1990	20000	158	2.34	8020
	1997	20000	158	2.34	8020
	2000	20000	158	2.34	8020
	2005	20000	158	2.34	8020
	2010	20000	158	2.34	8020
	2020	20000	158	2.34	8020

Notes: (a) aircraft movement data obtained from the airport managers
(b) future years extrapolated from historical data
(c) Medway Local Plan indicates Rochester is to close in 2004 and be replaced by housing. For this study, the airport is assumed to continue with 20000 movements per annum.
(d) An application for the expansion of Kent International Airport (Manston) is currently being prepared. For this study, the airport is assumed to continue with 20000 movements per annum.

(Source : this study)

4.2.11 Domestic Sector

National estimates of domestic emissions are derived from national fuel use data distributed by population density. The NAEI assumes there is no gas supply available in areas with a density of less than 200 people per square kilometre. Coal consumption in urban areas (defined by concrete and roofing land cover) is adjusted by smoke control zones to account for the use of non-sulphur solid fuels. For example, if in a district 10%

of the total area is uncontrolled then 90% of total solid fuel consumption in each grid square in that District is assumed to be non-sulphur solid fuel.

In compiling the urban inventories, the LRC used fuel sales data supplied by British Gas, the coal industry and Petroleum Industries Association (Buckingham *et al*, 1997a, b, c; 1998). Gas and coal sales were available on the basis of postcode, national grid reference and local authority area. Allowances were made due to the loss of British Gas market share since the opening up of competition in the gas supply sector. The authors of the LRC reports make the comment that future availability of gas sales data may be limited due to its commercial nature. No information of how building emissions are estimated is included in the reports of urban inventories. Sadler (1998) notes the high quality of emissions data for gas consumption and reasonable availability of activity data.

Chell and Hutchinson (1993) report domestic energy consumption by end use in London for 1989 proportioned as follows:

Space Heating	56.6 %
Water Heating	25.3 %
Lights and Appliances	10.8 %
Cooking	7.3 %.

These end uses reflect the different types of energy consumed. In 1991 80% of total domestic energy use in Greater London was gas and 17% was electricity. Future demand for electricity is likely to increase with increased availability of electrical appliances such as dishwashers. This may be counterbalanced by an overall increase in the number of centrally heated homes predominantly fuelled by gas.

In terms of local emissions, the domestic use of gas (rather than electricity) needs to be included in an emissions inventory. The above figures indicate the majority of properties in urban areas use gas for space and water heating which may make up 81.9% of the total energy demand. In rural areas, where mains gas supplies may not be available, the use of gas is likely to be much lower although approximately 85% of domestic dwellings use gas nationally.

When estimating emissions from the domestic sector, the age and type of housing needs to be considered. The high proportion of energy used for space heating means that the majority of emissions due to domestic use of gas will occur during the winter months. This may exacerbate winter pollution episodes formed during low level inversions but does suggest there is considerable potential for reducing these emissions through the promotion of home insulation and other energy conservation measures. Information on the annual average consumption of gas by property type up to 1995 has been made available to the Sussex Air Quality Forum by British Gas (Consulting Environmental Scientists, 1997) and is summarised in Table 4.6.

Table 4.6
Annual Mean Gas Consumption By Domestic Property Type (m^3)

Domestic Property Type	Pre 1919	'19 – '44	'45 – '70	'71 – '95
Detached	3583.13	2939.286	2449.405	2281.446
Semi-detached	2468.067	1856.882	1637.602	1511.633
Terrace	2094.824	1688.923	1548.957	1506.967
Flat	1101.066	886.4513	807.1373	741.8198

(Source: CES, 1997)

The annual mean emissions of NO_x, PM_{10} and CO_2 have been estimated from these values based on DETR factors for gaseous fuels. Approximately 80 % of the domestic sector in Kent and Medway is provided by mains gas with the remainder, predominantly in rural areas, supplied with either bottled gas, oil, coal or wood (Seeboard, 2001, *personal communication*). Emissions from the domestic sector are therefore under estimated using this method but not significantly.

The emissions from different property types are summarised in Table 4.7.

Table 4.7
Annual Mean Emissions By Domestic Property Type
(Tonnes per 1000 people per Annum)

Property Type	Pre 1919	'19 – '44	'45 – '70	'71 – '95
(a) NO_x				
Detached	1.80434	1.48012	1.23343	1.14886
Semi-detached	1.24283	0.93506	0.82464	0.76121
Terrace	1.05488	0.85049	0.78	0.75886
Flat	0.55446	0.44639	0.40645	0.37355
(c) PM_{10}				
Detached	1.7866×10^{-7}	1.4649×10^{-7}	1.2207×10^{-7}	1.1370×10^{-7}
Semi-detached	1.2300×10^{-7}	9.2542×10^{-8}	8.1614×10^{-8}	7.5336×10^{-8}
Terrace	1.0440×10^{-7}	8.4171×10^{-8}	7.7196×10^{-8}	7.5103×10^{-8}
Flat	5.4874×10^{-8}	4.4178×10^{-8}	4.0226×10^{-8}	3.6970×10^{-8}
(d) CO_2				
Detached	2181.20	1789.26	1491.05	1388.81
Semi-detached	1502.41	1130.36	996.87	920.19
Terrace	1275.20	1028.12	942.91	917.35
Flat	670.26	539.62	491.34	451.58

Notes: Assumes all properties use domestic gas, 1 m^3 of natural gas = 10.3 kWh and domestic boiler efficiency of 65%

(Source : this study)

In compiling the regional emissions inventory for this study, the population census data for surveys undertaken in 1921, 1951, 1971 and 1991 were analysed to determine the age and type of housing within each district (KCC, 2000b). For example, the increase in housing stock between the 1921 and 1951 surveys is assumed to be new housing built between 1919 and 1944. Only the 1991 census includes sufficient information to differentiate between house types and was assumed for each year. These data were then combined with population forecasts for 1996 2001, 2006, 2011 and 2021 to extrapolate for future years assuming uniform population growths relative for rural (4%) and urban (1.5%) areas observed in each district in recent years and all new houses are built to the 1995 specification.

This method does not take account of future savings in domestic emissions or of any improvements in building insulation. Although not specifically quoted, analysis of the CCP suggests carbon dioxide emissions from the domestic sector by 2010 will be in the order of 10 - 11% below 1990 levels (DETR, 2000a, pp 48 and 114-116; see also McEvoy *et al* (1999). Interpretation of these estimates requires care as the DETR numbers include carbon dioxide emissions associated with electricity generation.

As a conservative approach, domestic emissions estimated using the method detailed above were adjusted as follows:

- 1990: as estimated using domestic emission factors and housing stock data
- 2000: as estimated using domestic emission factors and housing stock data
- 2005: as estimated for year 2000, extrapolated using population forecasts for 2005 assuming new build with 1995 specification. Total emissions (ie. all housing) reduced by 5%.
- 2010: as estimated for year 2000, extrapolated using population forecasts for 2010 assuming new build with 1995 specification. Total emissions (ie. all housing) reduced by 10%
- 2020: as estimated for year 2000, extrapolated using population forecasts for 2020 assuming new build with 1995 specification. Total emissions (ie. all housing) reduced by 15%.

The emissions for each district within the study area were aggregated in proportion to the age and number of house types to provide an emission rate per person for each of the scenario years (1990 to 2020). The results, summarised in Table 4.8 below, are then multiplied by the population value for each 250 m square.

Table 4.8
Annual Mean Emissions Aggregated by Domestic Property Age and Type (Tonnes per person per annum)

		NO_x	PM_{10}	CO_2
Kent and Medway	1990	0.0006333	6.267E-11	0.7655
	1997	0.0006265	6.200E-11	0.7573
	2000	0.0006257	6.193E-11	0.7564
	2005	0.0005999	5.937E-11	0.7251
	2010	0.0005663	5.605E-11	0.6846
	2020	0.0005317	5.262E-11	0.6427

Notes: (1) All new build post 1995 assumed to have same specification as for 1995.
(2) Emissions for 2005, 2010 and 2020 are reduced by 5%, 10% and 15% respectively.

(Source : this study)

In comparison with published inventories, this approach for quantifying domestic emissions is more detailed, in terms of building type, spatial density and change in strength and distribution of emissions over time. DETR (1998b, 2000j) and draft DEFRA (2002b) guidance does not address this source although its contribution to greenhouse gases will be significant (compare Figures 6.1 - 6.3). The method is considered valid for estimating GHG emissions aggregated across the study area and, given the important role local authorities have in local housing provision, provides a simple tool for land use planning.

4.2.12 Emissions by Developed Land Use

Commercial energy consumption was also estimated for London by Chell and Hutchinson (1993). In this context, the commercial sector does not include industrial premises. Chell and Hutchinson (1993) estimated total energy used by the commercial sector in 1991 was split equally between gas and electricity.

Chell and Hutchinson (1993) provide tables of energy performance, in terms of Gigajoules of energy consumed per square metre, for both public and commercial buildings. The public buildings include libraries, museums, colleges and schools (both with and without swimming pools). The energy performances are classified as either good, fair or poor. The commercial buildings include naturally ventilated and air conditioned offices. These are classified as either typical or good practice. For public buildings, excluding swimming pools, the 'fair' ratings range between 0.65 and 1.55 $GJ/m^2/annum$ with the majority in the 0.7 - 1.3 $GJ/m^2/annum$ range. Using the upper range for 'fair' ratings, the average energy rating for public buildings may be estimated as 1.1 $GJ/m^2/annum$. In commercial buildings of 'typical' perfomance the energy consumption ranges between 0.89 and 2.28 $GJ/m^2/annum$ with prestige air conditioned offices being the most energy intensive. The energy consumption for a standard air conditioned office is estimated to be 1.53 $GJ/m^2/annum$.

Industrial Energy Consumption for small industrial units used in the London Energy Study was determined using the results of a study undertaken in 1981 which estimated this to be 2.7 $GJ/m^2/annum$ excluding electricity and heavy industries. Chell and Hutchinson (1993) then extrapolated this figure to estimate an overall industrial energy

consumption rate based on the proportion of light industrial and total industrial floor space. This is then further extrapolated using total industrial employment figures to estimate the energy consumption per employee in the industrial sector. Employment figures for London were then used to determine the spatial distribution of industrial energy consumption. LRC note that due to the large number of head offices in London, this degree of extrapolation produced some anomalies.

Although energy consumption by industrial processes is also considered in the London Energy Study, for the purposes of this review the emissions to atmosphere of such processes can be determined elsewhere.

With reference to the DETR emission factors for different fuels, this information on commercial and industrial emissions has been used to compile estimates of NO_x, PM_{10} and CO_2 emissions by land use categories in Table 4.9. The same landuse databases described in section 3.5.8 were then used for this study. The draft CCP suggests CO_2 emissions from the commercial sector by 2010 will be in the order of 5% below 1990 levels (DETR, 2000a, pp 48 and 114-116). Adopting a similar conservative approach as for domestic emissions, the estimation of emissions for 2005, 2010 and 2020 were reduced by 2.5%, 5% and 7.5% below 2000 levels, respectively.

This method is limited in spatial detail but, as more local forecasts are developed, has the potential to be resolved to at least the district council (Local Plan) level. For the purposes of this study, the contribution of land use class related emissions to local air quality is expected to be minimal and is not considered significant. DETR (1998b, 2000j) and draft DEFRA (2002b) guidance does not include for this source although its contribution to greenhouse gases will be significant (compare Figures 6.1 - 6.3). The method is considered valid for estimating GHG emissions aggregated across the study area. In contrast to the 'static' inventories cited in this chapter, this methodology allows for a direct link between land use and emissions to be established on both temporal and spatial scales.

Table 4.9
Commercial Building Emissions By Land Use Classification

Land Use Class		Energy Consumption (GJ/m^2)	[7] Annual Mean Emissions ($kTonnes/m^2/annum$)		
Code	Use Class		NO_x	PM_{10}	CO_2
A1	Retail	[1] 0.79 to 1.12	4.568×10^{-08}	4.52×10^{-15}	5.52×10^{-05}
A2	Business (office)	[2] 1.53	6.989×10^{-08}	6.917×10^{-15}	8.448×10^{-05}
A3	Catering	[3] 0.83 to 1.13	4.568×10^{-08}	4.52×10^{-15}	5.52×10^{-05}
B1 B1B	Business / Light Industry	[2] 1.53 to [4] 2.7	1.0×10^{-08}	1.0×10^{-14}	11.5×10^{-05}
B2	General Industry	[4] 2.7	1.233×10^{-08}	1.22×10^{-14}	0.000149
C1	Hotels	[3] 0.83 to 1.13	4.568×10^{-08}	4.52×10^{-15}	5.52×10^{-05}
C2	Institutions	[3] 0.83 to1.13	4.568×10^{-08}	4.52×10^{-15}	5.52×10^{-05}
D1	Non Residential Institutions	[5] 1.1	5.025×10^{-08}	4.97×10^{-15}	6.07×10^{-05}
D2	Sports and recreation Halls	[6] 0.72 to 1.22	4.568×10^{-08}	4.52×10^{-15}	5.52×10^{-05}

Notes :
1. Assumed to be equivalent to the space heating requirement of a museum or art gallery (ie. a large open area accessed by the public)
2. Based on typical office energy consumption for a standard, air conditioned office.
3. Based on the energy consumption for small industrial units but applied to all industry, as per the LRC Energy Study. This does not include process emissions.
4. Assumed to be equivalent to the space heating requirement of a residential polytechnic.
5. Based on the upper range of "fair" ratings for public buildings
6. Without a swimming pool.
7. Energy consumption by burning of natural gas is assumed to be 50% (see text). Volume of natural gas to generate 1 GJ = 26.8 m^3 = 276 kWh (3.1)

(Source : this study)

4.2.13 Construction Dust

The LRC urban inventories (Buckingham *et al*, 1997a, b, c; 1998) estimate construction dust emissions using the NAEI national total of 4000 tonnes per annum and apportioning by population in the West Midlands. As coarse particles may represent up to 20 % of urban levels of PM_{10} (Turnbull and Harrison, 2000) rising to 50 % in the summer months (Harrison *et al*, 1997, 1999) this appears to be a rather gross assumption that

requires further study. Based on the work of Sadler (1998) a very poor data quality rating is suggested for this source category.

The duration of construction projects is typically less than one or two years and tends to be concentrated within small, well defined areas. Given the broad spatial and temporal scale of the regional inventory, construction emissions were not directly estimated in compiling the regional emissions inventory for this study. The 'other' source category for PM_{10} included in the NAEI was assumed to be split equally between construction dust and agriculture. No differentiation was made in the spatial distribution of construction emissions across the study area.

4.2.14 Agriculture

The NAEI includes emissions associated with agricultural activities based on the ITE land cover dataset. These are now aggregated within the 'other' source category for PM_{10}. Assuming a 50% split between construction and agriculture for this source category, emissions were evenly distributed across the rural areas. A data quality rating similar to construction can be applied to this source category.

4.3 VERIFICATION OF EMISSIONS

Lindley and Longhurst (1998) describe some of the potential errors and uncertainty in emissions inventories, including selection of emission factors, assumptions made in activity data and any changes in activity data following compilation. In quantifying some of these errors for a regional emissions inventory in North West England, the authors make comparisons with other estimation methods (for vehicular NO_x) and national emissions estimates (NAEI). This work was extended by Lindley *et al* (1999) in making a spatial comparison of the local inventory and the NAEI extract for the study area. Pulles and Builtjes (1998) consider the *validation* of an emissions inventory in terms of checking procedural quality and *verification* as checking scientific quality; the former being considered more important for the policy maker. The DETR (2000j) guidance on estimating emissions for the purposes of review and assessment highlights the uncertainty in deriving such estimates and lists some six key areas for consideration including some of those highlighted above. In essence, the suggested approach is to

accept a degree of uncertainty and provide a *reasonable estimate*. With the exception of one statistic on traffic counts provided by the DETR and national estimates of uncertainty for the NAEI (Table 4.1) there appears to be little in the literature providing any indication of the expected accuracy of local emissions factors or estimates. This may explain the emphasis placed by Pulles and Builtjes (1998) on validation. The DETR guidance appears to be in line with LRC 'best practice' ie. to critically evaluate each estimate and provide suitable justification for its use, identifying any shortfalls if known. A good example of this is the evaluation of road dust in the urban inventories (Section 4.2.7).

In describing the validation of an emissions inventory for Antwerp, Mensink (2000) cites the need for:

- an analysis of the *completeness* of an emissions inventory; whether all emission activities have been included
- an analysis of its *uncertainty*; the degree of accuracy (the measure of truth of an estimate) and precision (repeatability)
- *validation*; comparison with an independent approach.

Although Mensink refers to validation with reference to both field data and all three processes listed above, Zannetti (1990) more strictly defines this as *'the theoretical ability [of the model] with error-free inputs'*. In this context, Zannetti uses the term *evaluation* when comparing to field data and *verification* being the successful validation and/or evaluation of the model, i.e. the overall process. In discussing uncertainties of modelling emissions from road transport in West Germany, Kühlwein and Friedrich (2000) define *uncertainty* as a general expression of unknown possible deviations of true emissions from calculated emissions data and *error* as quantifiable uncertainties.

Mensink describes two methods for validation [*sic*] the first being based on a combination of comparing modelled traffic flows and modelled vehicular emissions with observed traffic flow and emissions. No account was taken of the uncertainty in observations; a limitation in modelling highlighted by both Zannetti (1990, p319) and Murthy *et al* (1990) who go on to describe three approaches to verification:

- Approach 1; based on a critical examination of basic assumptions; if these are invalid, the model is inadequate.

- Approach 2; based on model behaviour, comparing modelled with observed data. Ideally, the data used for verification should be independent of the data used for model development (calibration).

- Approach 3; a combination of 1 and 2.

The second approach described by Mensink (2000) was an unsuccessful attempt to compare the pollution flux (estimated using the emissions inventory and a simple box model) with in-flight observations across the study area.

Palmgren *et al* (1999) derived actual car fleet emissions for a street in central Copenhagen by using a line source model to back calculate vehicular contributions to observed kerbside concentrations. Consideration was given to uncertainties in both dispersion modelling and ambient monitoring data. Although a useful complement to emissions data derived from national statistics, this type of analysis is site specific. A similar conclusion was made by Mulholland and Seinfeld (1995) who used inverse modelling techniques to define the range of acceptable tolerance in carbon monoxide emissions across the South Coast Air Basin of California.

Funk *et al* (2001) compare the ratios of observed concentrations with the ratios of estimated emissions of non-methane hydrocarbons (NMHC) / NO_x and carbon monoxide / NO_x. This is referred to as a 'top down' approach most suitable for identifying areas of an inventory that warrant improvement. The main objective of this work on the US/Mexican border was to verify estimates of NMHC on the basis that estimates of NO_x and carbon monoxide were considered reliable.

Sturm *et al* (1999) and Lindley *et al* (1996, 2000) undertook trend analyses of emissions and air quality over time, comparing the % change in annual mean levels of pollutants with estimated annual mean emissions between two years. Although useful, Sturm *et al* recognise this technique is limited due to the effects of meteorology; a period of five years would be required to derive a statistically robust trend (Archer and Sykes, 1998).

El-Fadel *et al* (2001) employed two techniques to consider the uncertainty of emission factors used to compile an inventory of greenhouse gases for Lebanon. The first was a reference approach using national fuel supply and specification data coupled with emission factors used by the IPCC, Austria, the UK and the USA. Estimated CO_2

emissions were within 7% of each other implying a degree of confidence in the emission factors. The second approach used emission factors derived at the national level for different industrial sectors and applied to Lebanon. The example presented was of the cement industry where variations in estimated CO_2 emissions were in the order of –60% to + 12% implying a lower degree of confidence.

In reviewing the performance of local authorities undertaking review and assessment, Woodfield (2001a) reports some use of the Monte Carlo method, as described by Hammersley and Carter (1964) and Rubenstein (1981), to account for the range of uncertainty in emissions and modelling data. The usual method requires ascribing margins of tolerance to each parameter and randomly selecting a value for each within this range. The function (total emissions) is then calculated. This is usually repeated some 10000 - 20000 times and the confidence regions identified. Although not utilised in local air quality management to date, Oppenshaw *et al* (1991) have described a means of using this technique to account for spatial uncertainty. In providing a sensitivity analysis on carbon dioxide emissions, the DTI (2001) rejected a Monte Carlo analysis at the time on the basis of the '*complex structure of the energy demand model*'. A simplified model was used, based on using energy demand, price, GDP and temperature as independent variables. In fitting this grossly simplified model to historical data a forecast interval of +/- 6% at two standard deviations (ie. the approximate 95% confidence limit) was indicated. This was used to define upper and lower probable limits (uncertainty). This was supported by a study of variations in individual key assumptions to the full energy demand model. Notwithstanding the above, this method of defining uncertainty has been used in this study, as described in Section 4.3.3.

Lindley *et al* (2000) tabulate four main techniques with subsequent discrete elements for verification of an emissions inventory. Most of these elements have been incorporated in this study; as summarised in Table 4.10. One of the key principles was to use published emission factors have been used along with activity data obtained from auditable sources. For example, commercial land use data held in Kent and Medway are based on information used to collect business rates and hence, open to public scrutiny. No analysis was made of activity data; a potential source of error highlighted by Reynolds and Broderick (2000) who suggest traffic volumes may be underestimated by 10-20% leading to an underestimation of emissions of 50% (partly due to increases in

Table 4.10
Techniques for Verification of an Emissions Inventory

Verification Technique	Elements	This Study
Documentation of data and Procedures	Details of the selection of procedures, assumptions and factors	Full details provided in Section 4.3; a critical evaluation of each estimate was made in accordance with LRC 'best practice' (Hutchinson and Clewley, 1996, DETR, 2000j) and *Approach 1*; Murthy *et al* (1990) An *independent review* was undertaken by a third party for part qualification as registered auditor with the Institution of Environmental Management and Assessment (Critten, 2001).
	Data quality objectives	The emissions inventory was subject to quality assurance procedures (ISO 9001) as part of Mott MacDonald project management requirements.
Quality checks in relation to the application of data	Basic applications as a verification technique	The emissions data were checked in terms of providing, in combination with dispersion modelling, estimates of annual mean NO_2 and PM_{10} concentrations on a local to regional scale. The inventory provides a reasonable disaggregation of emissions by sector over the study area. The spatial resolution of the inventory does not allow for very local scale (ie. tens of metres) assessment.
Comparison of alternative estimates	Consideration of alternative methods and data: Comparisons of emissions densities and factors	Reference to national emission factors where applicable. *Completeness check* for NO_x and PM_{10} with reference to the urban inventories. *Consistency check* for NO_x and PM_{10} by comparing emissions with underlying geography (eg. increased emissions with urban density) and with local NAEI extract and urban inventories. Comparison of CO_2 emissions with national inventory.
Uncertainty estimates	Spatial and temporal issues as well as overall magnitude of emissions	Spatial comparison of NO_x and PM_{10} emissions with local NAEI extract; Temporal evaluation by comparing annual trends with observed data / national estimates: Lindley and Longhurst (1996); Lindley *et al* 1999, 2000; Sadler (1998); and Niemeier, Lin & Utts (1999).
	Sensitivity studies	*Data quality ratings* provided for NO_x and PM_{10} as per Sadler (1998) Table 1. Parameter sensitivity considered in terms of results being highly sensitive to one parameter, implying the estimate is less valid: Buckland and Middleton (1999). Specific *checks on systematic error and bias* not undertaken. Monte Carlo analysis undertaken to identify bounds of confidence.
Ground truth verification	Survey analyses	*Random sampling and remote measurement* not undertaken.
	Monitoring analyses	Comparison of observed trends in ambient levels of NO_x and PM_{10}, and trends in emissions: Sturm *et al* (1999); Lindley *et al* (1996). Evaluation of the emissions inventory and dispersion model combined, with reference to observed ambient concentrations; in accordance with common practice (eg. Renolds and Broderick, 2000) and *Approach 2*; Murthy *et al* (1990); see Section 4.11.
	Source sampling	Comparison of observed NO_x / PM_{10} ratios and emissions: Funk *et al* (2001).
	Measurement of operating parameters	*Receptor modelling* not undertaken.

(Source : *after* Lindley *et al*, 2000)

traffic speed implicit with lower traffic volumes) and by Rodier and Johnston (2001) who raise the whole question of uncertainties in the underlying socio-economic assumptions in travel demand models used to derive traffic volumes.

The use of arbitrary factors, such as 10% applied to an inventory of greenhouse gas emissions in Barcelona (Baldasano, 1998) was not adopted. Given the simplistic emission equations used (rarely more than two or three parameters) any assessment of parameter sensitivity would be limited and largely meaningless, accepting that Part A Process data are locally dominant.

4.3.1 Comparison With Other Inventories

Completeness and consistency was considered by comparing the method with those of the NAEI and urban inventories. The results of estimating NO_x, PM_{10} and CO_2 emissions from all sectors in Kent and Medway are summarised and compared to estimates derived from national and urban studies in Table 4.11. The study area and population for each study are included where available for comparison. The emissions are all reported in kilotonnes per annum with NO_x emissions generally being a factor of 6-12 greater than PM_{10} emissions and CO_2 emissions typically a factor of 200-300 greater than NO_x emissions.

The urban inventories appear generally consistent; dividing total NO_x emissions by area suggests emissions are between 0.44 and 0.59 kTonnes/km^2. Middlesborough is an exception; being dominated by large industry. Emission densities in Kent are much lower (eg. 0.026 kTonnes/km^2 for NO_x) reflecting the large proportion of rural area.

Subtracting Part A emissions from the inventory for this study allows direct comparison with NAEI data. The results, summarised in Table 4.12, indicate NO_x estimates within 10% and PM_{10} estimates within 15% of equivalent NAEI data suggesting a high degree of confidence in the data, particularly given the very different methodologies for compiling these inventories.

Table 4.11
Summary Comparison of Emission Inventories

Inventory	Area (km^2)	Population (millions)	NO_x (kTonnes)	PM_{10} (kTonnes)	CO_2 (kTonnes)
This Study (1997)		1.56	101.4	7.9	8864
This Study (2000)		1.6	94.8	7.2	9190
NAEI (Kent and Medway) (1998)	3908	1.575	[6]40.145	[6]2.847	nr
NAEI (UK) (1998)	[2]244880	[2]5659	1835	184	546000
Bristol (1996)	181	0.402	10.152	1.162	1995
Glasgow (1996)	239	[3]0.6	11.894	1.023	2721
Gr. Manchester (1995)	1552	2.577	75.434	11.188	21486
London (1995)	2466	[4]7.8	147.581	9.844	34243
Merseyside (1996)	1009	1.409	44.404	14.332	12973
Middlesborough (1996)	261	0.145	32.314	2.226	20477
Southampton & Portsmouth (1996)	465	0.406	23.012	2.147	4007
Swansea & Port Talbot (1996)	358	0.368	15.759	2.312	2463
West Midlands (year not reported)	899	[5]2.62	46.519	4.021	28050
West Yorkshire (1996)	655	2.113	24.79	2.003	9704

nr not reported
[1] population data from National Statistics (2001) unless otherwise stated
[2] from National Statistics (2001)
[3] data from Glasgow City Council website
[4] data from Buckingham C, Sadler L and Shah S (1998)
[5] data from Hutchinson and Clewley (1996)
[6] excludes Part A emissions
other data from NAEI (DETR, 1998), LRC urban inventories and this work.

(Source : this study)

Table 4.12
Comparison With NAEI Data for Kent (excluding Part A Emissions)

Inventory	NO_x (ktonnes)	PM_{10} (ktonnes)
This Study (1997)	44.1	3.260
This Study (2000)	37.3	2.719
NAEI (Kent and Medway) (1998)	40.145	2.847

(Source : this study)

These summary data have been subdivided by source category in Tables 4.13 to 4.15. with some source categories aggregated to allow comparison between inventories. All the inventories show some consistency in identifying road transport and industry as the dominant sources of NO_x and a generally similar pattern for PM_{10}. Industrial NO_x

emissions in Kent are estimated to be in the order of 59 – 63% similar to Swansea and Port Talbot, reflecting the significant industrial element to land use in Kent. The proportion of PM_{10} emissions from industry in Kent (64-68%) are also comparable to Swansea and Port Talbot. Road traffic in Kent contributes approximately 34-38% of NO_x and 32-35% of PM_{10}. As may be expected, these proportions are somewhat lower than urban areas but are broadly comparable to UK data. Although both industry and traffic are significant contributors of CO_2, all the inventories reflect the importance of emissions from the residential and commercial sectors.

The inventory of Kent and Medway therefore appears to be consistent when compared to other UK national and urban studies both in terms of total and sector contributions. The most direct comparison, with equivalent NAEI data, suggests a high degree of consistency.

Table 4.13
Total NO_x Emissions by Sector (%)

	Part As	Part Bs	Other Industry	Major Roads	Minor Roads	Residential	Commercial	Shipping	Airports	Agriculture / Construction
This Study 1997	56.5	2.4	-	26.9	10.8	0.7	0.1	1.9	0.6	-
This Study 2000	60.7	2.6	-	23.9	9.7	0.8	0.1	1.5	0.7	--
NAEI (Kent and Medway 1998)		5.4		61.2		4.6	28.7			
NAEI (UK 1997)	37			48		4	-	4	1	-
Bristol	18			60.8		7.4		-	0.7	0
Glasgow	3.2			75.6		10.4		-	2.6	0
Gr. Manchester	24.3			63.1		8.6		-	1.2	0
London	5.3			75.1		11		-	2.7	0
Merseyside	43.7			41.5		8.5		-	0.08	0
Middlesborough	74.8			17.2		2.2		-	0	0
Southampton and Portsmouth	33.2			47.2		5		-	0.05	0
Swansea and Port Talbot	63.6			27.9		3.7		-	0	0
West Midlands	3.7			84.7		6.4		-	3.2	0
West Yorkshire	9.3			72.8		11.3		-	0	0

Notes:
NAEI (UK) data for 1997 from DETR (2000b); Part A data not available
NAEI (Kent and Medway) for 1998; Part A data not available
Urban Inventory Data: for Merseyside, Bristol and Southampton/Portsmouth; Buckingham *et al* (1997); for West Yorkshire, Glasgow, Swansea / Port Talbot and Middlesborough; Buckingham *et al* (1998); for West Midlands; Hutchinson and Clewley (1996); for Greater Manchester; Buckingham *et al* (1997); for London; Buckingham *et al* (1997)

(Source : this study)

Table 4.14
Total PM_{10} Emissions by Sector (%)

	Part As	Part Bs	Other Industry	Major Roads	Minor Roads	Residential	Commercial	Shipping	Airports	Agriculture / Construction
This Study 1997	58.8	5.3	-	30.5	4.9	<0.01	0	0.5	0.1	0.01
This Study 2000	62.4	5.7	-	26.7	4.7	<0.01	0	0.4	0.1	0.01
NAEI (Kent and Medway 1998)	0.32 (excl Part As)			1.2		0.7	0.7			
NAEI (UK 1997)	54			23		15	-	1	0	-
Bristol	52.7			35.5		1.6		-	-	-
Glasgow	11.8			73.2		4.5		-	-	-
Gr. Manchester	24.9			31.3		43.5		-	-	-
London	9.9			77.3		2.4		-	0	-
Merseyside	7.7			90.1		0.7		-	-	-
Middlesborough	73.4			19.1		0.1		-	-	-
Southampton and Portsmouth	58.4			23.9		11.7		-	-	-
Swansea and Port Talbot	80.1			12.9		5.4		-	-	-
West Midlands	27.7			55.9		8.5		-	-	-
West Yorkshire	28.5			63.7		1.5		-	-	-

Notes: NAEI (UK) data for 1997 from DETR (2000b); Part A data not available
NAEI (Kent and Medway) for 1998
Urban Inventory Data: for Merseyside, Bristol and Southampton/Portsmouth; Buckingham *et al* (1997); for West Yorkshire, Glasgow, Swansea / Port Talbot and Middlesborough; Buckingham *et al* (1998); for West Midlands; Hutchinson and Clewley (1996); for Greater Manchester; Buckingham *et al* (1997); for London; Buckingham *et al* (1997)

(Source : this study)

Table 4.15
Total CO_2 Emissions by Sector (%)

	Part As	Part Bs	Other Industry	Major Roads	Minor Roads	Residential	Commercial	Shipping	Airports	Agriculture / Construction
This Study 1997	37.3	6.8	-	25.3	14.3	11.7	3.9	0.4	0.4	-
This Study 2000	36.2	6.5	-	26.5	15.1	11.3	3.8	0.3	0.4	-
NAEI (UK 1997?	-	-	-	-	-	-	-	-	-	-
Bristol	22.3			25.6		46.5			1.1	-
Glasgow	14.4			29.4		51.6		-	-	
Gr. Manchester	17.3			21.9		59.4		-	0.9	-
London	64			28.7		54.6		-	3.7	-
Merseyside	63			18		17.8		-	0.1	-
Middlesborough	4.2			92.9		2.5		-	-	-
Southampton and Portsmouth	37.2			24.2		32.2		-	1.7	-
Swansea and Port Talbot	54.7			16.7		26.6		-	-	-
West Midlands	12			42.8		40.4		-	0.2	-
West Yorkshire	47.7			16.7		34.7		-		

Notes: NAEI (UK) data for 1997 from DETR (2000b)
Urban Inventory Data: for Merseyside, Bristol and Southampton/Portsmouth; Buckingham *et al* (1997); for West Yorkshire, Glasgow, Swansea / Port Talbot and Middlesborough; Buckingham *et al* (1998); for West Midlands; Hutchinson and Clewley (1996); for Greater Manchester; Buckingham *et al* (1997); for London; Buckingham *et al* (1997)

(Source : this study)

4.3.2 Spatial Analysis

The annual mean emissions of NO_x, PM_{10} and CO_2 estimated for 2000 (excluding Part A emissions) are presented in Figures 4.6, 4.7 and 4.8 respectively. The spatial correlation between main roads, urban areas and emissions can be discerned by comparing these Figures with Figure 2.3. NO_x emissions are generally less than two tonnes per $250m^2$ in rural areas and between two and three tonnes per $250m^2$ in urban areas. The road network is clearly highlighted. Maximum emissions per $250m^2$ are associated with the local airports, suggesting further assessment is required although this source represents less than 1% of total emissions for Kent and Medway (see Table 4.13). The pattern of PM_{10} and CO_2 emissions (Figures 4.7 and 4.8, respectively) is similar to that for NO_x although the urban areas are more distinguished for CO_2, reflecting the relative contribution made by building related emissions as a proportion of total CO_2 for Kent and Medway. Building related emissions of NO_x for Kent and Medway are estimated to be 0.8% compared to 15.6% of total CO_2 emissions.

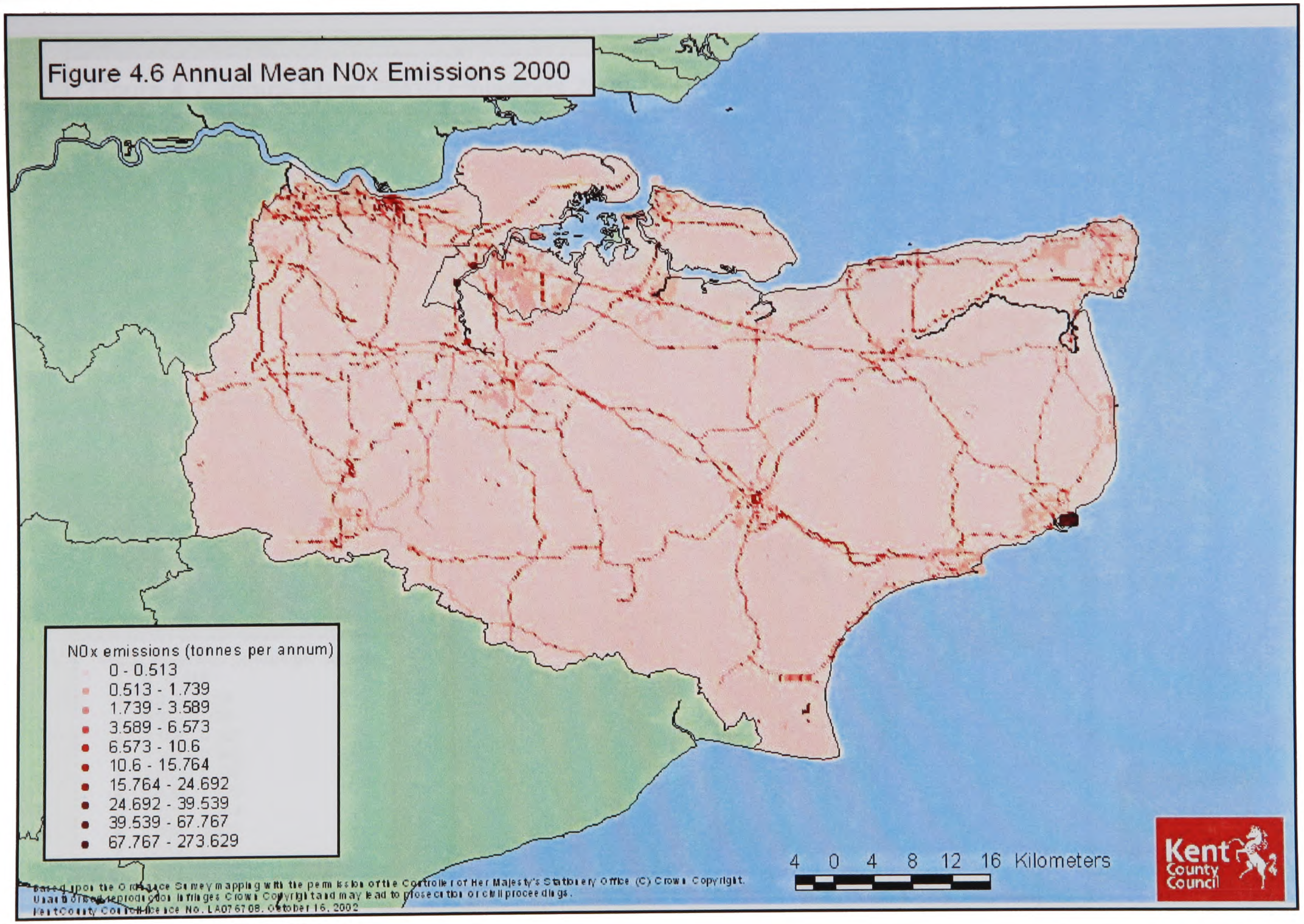


Figure 4.6 Annual Mean N0x Emissions 2000
NOx emissions (tonnes per annum)
0 - 0.513
0.513 - 1.739
1.739 - 3.589
3.589 - 6.573
6.573 - 10.6
10.6 - 15.764
15.764 - 24.692
24.692 - 39.539
39.539 - 67.767
67.767 - 273.629
4 0 4 8 12 16 Kilometers
Kent County Council
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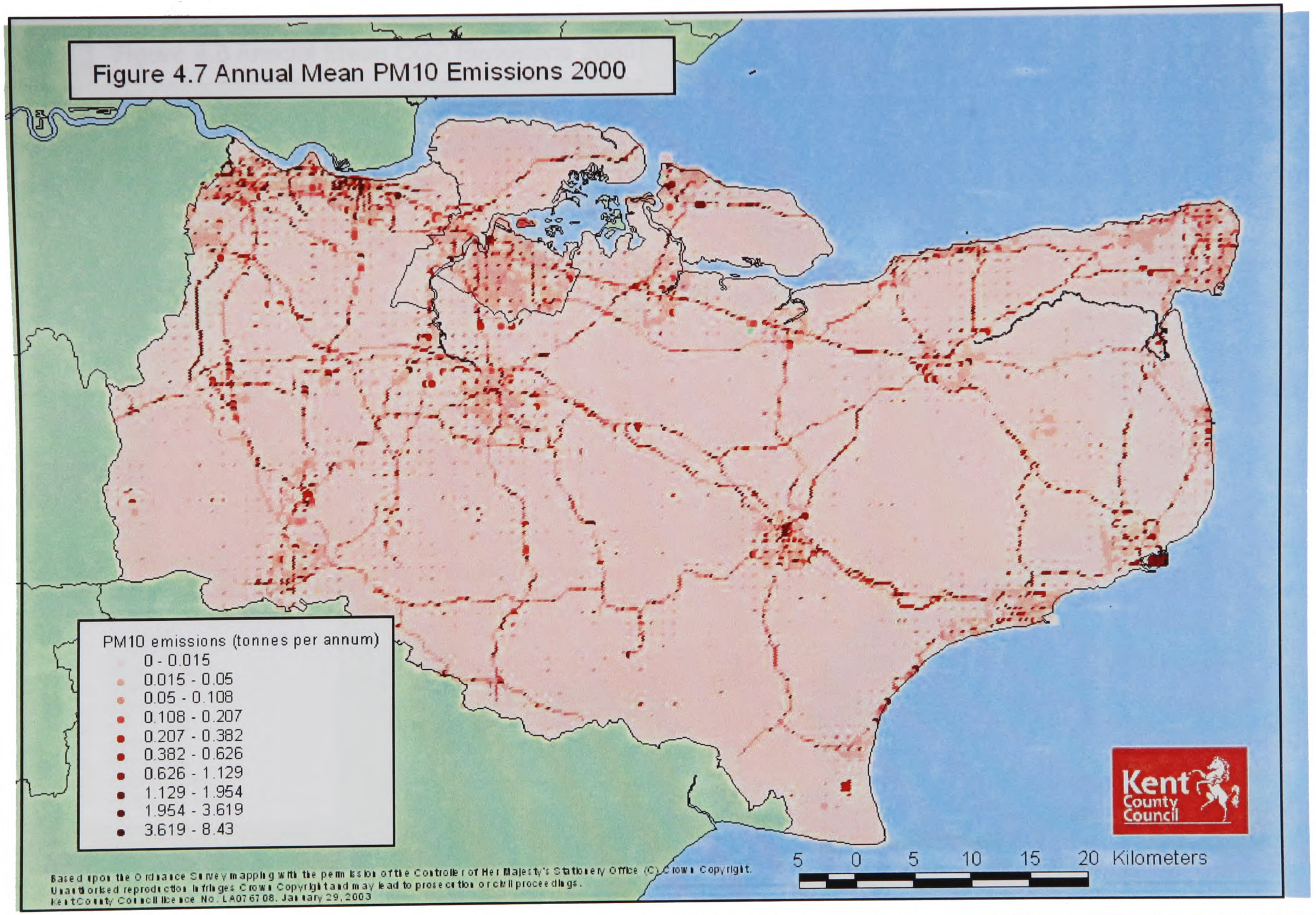
Figure 4.7 Annual Mean PM10 Emissions 2000



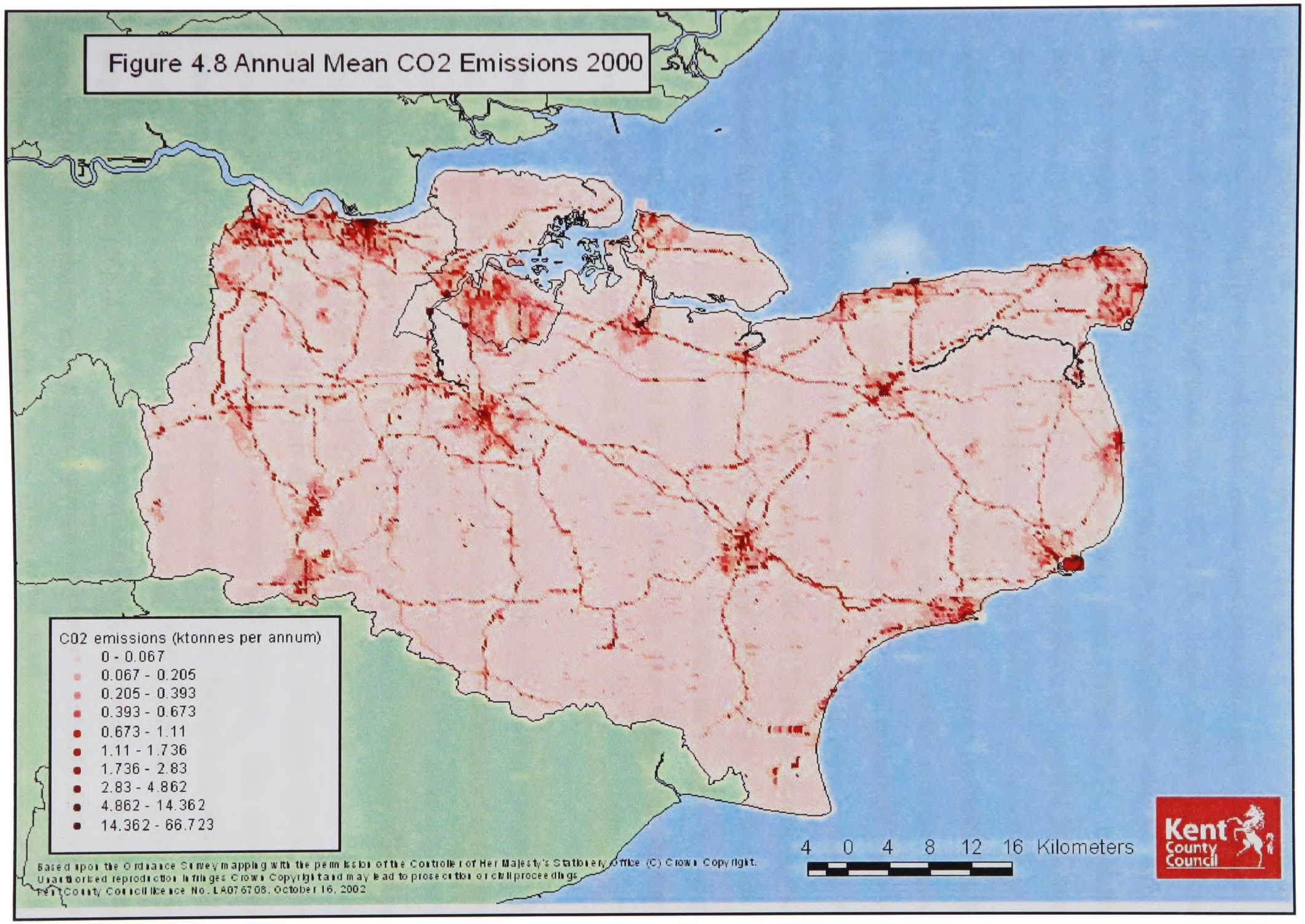


Figure 4.8 Annual Mean CO2 Emissions 2000
C02 emissions (ktonnes per annum)
0 - 0.067
0.067 - 0.205
0.205 - 0.393
0.393 - 0.673
0.673 - 1.11
1.11 - 1.736
1.736 - 2.83
2.83 - 4.862
4.862 - 14.362
14.362 - 66.723
4 0 4 8 12 16 Kilometers
Kent County Council
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4.3.3 Uncertainty Analysis of Emissions

Following the Tier 1 method described by the IPCC (2000) in its Good Practice Guide (see also Salway *et al*, 2001) and the work of Sadler (1998), estimates of tolerance (range of uncertainty) for each emissions source have been compiled in conjunction with other published data and judgement; see Table 4.16. These estimates of tolerance are applied to current emissions (1997-2000) assuming a minimum value of zero, and a further margin applied for extrapolation to other years (Table 4.17). Evidently, the tolerances provided could be debated but, with the absence of specific published data, are considered sufficient for demonstrating some consideration of uncertainty. These tolerances have been used as the basis for a Monte Carlo analysis (described by Salway *et al* (2001)) to determine the overall degree of uncertainty of the emissions inventory. For each year, the total emission for each source category was calculated and a random value within the relevant tolerance generated. Following this method, some 10, 000 random values were generated for each source and the 97.5^{th} percentile confidence limit calculated to provide a range of uncertainty, the range within which 97.5% of the values lie (i.e. the mean ± twice the standard deviation). The estimated ranges of uncertainty, expressed as twice the standard deviation divided by the mean, for the emissions inventory are summarised in Table 4.17.

The results of this uncertainty analysis are comparable with national estimates for NO_x and PM_{10} (see Table 4.1; Goodwin *et al*, 1997) suggesting the inventory is sufficient for local to regional scale studies. Although the uncertainty of this inventory for CO_2 is comparable to NO_x and PM_{10}, it is considerably larger than the 5% reported for national emissions of CO_2 (Goodwin *et al*, 1997). This may be expected, as the national estimate is based on the 'top down' approach of estimating CO_2 emissions directly from fuel consumption which, being taxed, is relatively well defined. The methodology is sensitive to the assumed margins of tolerance applied (ie. Tables 4.15 and 4.16) which, in the case of this work, and that of Goodwin *et al* (1997) and Salway *et al* (2001) is largely based on judgement. Experience in compiling this inventory suggests reducing the degree of uncertainty to less than reported nationally is unlikely without significant additional resources to compile more accurate emission factors and activity data. Whether such effort is warranted is considered in Section 5.7 in validating the dispersion of these emissions across the study area.

Table 4.16
Assumed Tolerances in Emissions Data Estimates
(for Monte Carlo Analysis)

Source	Quality Rating (0-3)	Tolerance (+/- %)	Comment
Part A Processes	1-3	35	Data reported and verified by the Environment Agency. Sturm *et al* (1999) suggests ±35% based on measurement data
Part B Processes	1-2	100	Data generally unreported so based on typical emissions by process type. Sturm *et al* (1999) suggests ±65% based on heating capacity. Additional tolerance added for uncertainty in activity data.
Other Industry	0-1	100	30% tolerance estimated for NAEI; Sturm *et al* (1999) suggests ±65% based on heating capacity. Additional tolerance included for spatial error
Major Roads	3	50	DETR (1999g) does not provide estimates of uncertainty. Vogel *et al* (2000) suggest a difference of ±20% between estimated and measured vehicular emissions; Bull and Zimmann (1997) suggest a typical factor of four between emissions models; Mensink *et al* (2000) report a range of ±80% between measured hot NO_x emissions; Kuhlwein and Friedrich (2000) suggest a statistical error for hot NO_x emissions of ±35% (at 68.3% confidence limits); El-Fadel and Hashisho (2000) suggest ±65%, John *et al* (1999) consider greater than ±10 to 20%, Reynolds and Broderick (2000) and Colvile et al (2001) suggest a global factor of two. ±50% selected as a compromise between reported values.
Minor Roads	2-3	75	See discussion for Major Roads; additional tolerance added for spatial and activity data error.
Shipping	3	50	30% tolerance estimated for NAEI, additional tolerance included for spatial error and comments by Sadler (1998)
Airports	1-2	50	30% tolerance estimated for NAEI, additional tolerance included for spatial error and comments by Sadler (1998)
Residential Buildings	2-3	50	30% tolerance estimated for NAEI, additional tolerance included for spatial and activity data error and comments by Sadler (1998)
Commercial Buildings	1-2	50	30% tolerance estimated for NAEI, additional tolerance included for spatial and activity data error and comments by Sadler (1998)
Agriculture	0-2	50	30% tolerance estimated for NAEI, additional tolerance included for spatial error and comments by Sadler (1998)
Notes:	Quality rating based on aggregate of emissions factor quality and availability of activity data		

Source : this study *after* Sadler (1998)

Table 4.17
Applied Tolerances in Annual Mean Emissions by Year

Year	Tolerance (%)
1990	±10 %
1997	±0 %
2000	±0 %
2005	±5 %
2010	±10 %
2020	±20 %

(Source : this study)

Table 4.18
Range of Uncertainty in Annual Mean Emissions

Year	NO_x	PM_{10}	CO_2
1990	± 50%	± 53%	± 43%
1997	± 39%	± 43%	± 45%
2000	± 38%	± 41%	± 46%
2005	± 40%	± 43%	± 49%
2010	± 43%	± 44%	± 52%
2020	± 51%	± 53%	± 57%
Note:	Range of uncertainty expressed as the total for all source categories. See text for methodology.		

(Source : this study)

This method does not provide a measure of spatial error. The discussion associated with Figures 4.6 to 4.8 provides a qualitative assessment of spatial consistency, concluding that the pattern of emissions follows the underlying land use pattern. The difference in spatial resolution between this inventory and the equivalent data from the NAEI precludes a spatial comparison as undertaken by Lindley *et al* (2000).

A final means of verification was applied by comparing the ratios of observed concentrations and estimated emissions (1997-2000; see Section 5.7.3) at grid squares with concurrent data (after Funk *et al*, 2001). The results, summarised in Table 4.19, are inconclusive. The ratio of observed NO_x to PM_{10} concentrations are in the order of 2 to 5 whereas the ratio of NO_x to PM_{10} emissions range from 2 to 40. This is most likely due to the large background component of PM_{10} which typically represents 80-90% of background observations and 40-50% of kerbside observations.

Table 4.19
Comparison of NO_x : PM_{10} Ratios of Observed Concentrations and Estimated Emissions

Monitoring Site (1997-2000)	Class	Annual Mean NO_x : PM_{10} Ratios	
		Observed	Emissions
Folkeston Suburban	b	1.8	38
Luton Background	b	3.3	4.2
Maidstone Rural (Detling)	b	1.5	45
Sevenoaks Background (Greatness)	b	2.5	35
Stoke Rural	b	1.8	38
Gravesham Ind Background (Northfleet)	i	1.1	6.9
Chatham Roadside (A2)	r	4.8	52
Dartford Roadside (St Clements)	r	2.0	2.9
Gravesham Roadside (A2)	r	4.6	2.1
Maidstone Roadside (Fairmeadow)	r	5.7	48

(Source : this study, *after* Funk *et al*, 2001)

4.4 FROM COUNTY TO REGIONAL SCALE

Expanding the emissions inventory from the county to regional scale can be considered in terms of data availability as the GIS is sufficiently powerful to handle very large datasets. All Part A Process data are available from the Environment Agency in a consistent format. All emission factors are published with the majority being those used for the purposes of local authority R&A. The availability of activity data in each local authority within several authorities South East England was determined by telephone survey. The results are presented in Table 4.20 clearly demonstrating the potential for extending the scale of the tools described in this Chapter for estimating atmospheric emissions of CO_2, NO_x and PM_{10}, and subsequent dispersion to estimate ground level concentrations of NO_x, NO_2 and PM_{10}.

Table 4.20

Methodology for a Regional Scale Emissions Inventory

Source Category	Regional Scale Inventory	Availability of Activity Data						
		Kent	Medway	Essex	Brighton	West Sussex	East Sussex	East Anglia
Part A Processes	Location and source data available from the Environment Agency	✓	✓	✓	✓	✓	✓	✓
Part B Processes	Location and relevant Process Guidance Note available from local authority records. Source data and fuel derived emission factors based on Process Guidance Notes	✓	✓	✓	✓	✓	✓	✓
Smaller Industry	Residual from NAEI less Part B emissions	✓	✓	✓	✓	✓	✓	✓
Major Roads	DMRB requires AADTF data (number of vehicles, %HDVs and speed). May require standard conversion from peak hour 24-hour flow data.	✓	✓	✓	✓	✓	✓	✓
Minor Roads	Derived from urban and rural population densities disaggregated in GIS to 250 m grid squares from ward population data	✓	✓	✓	✓	✓	✓	✓
Traffic Generation by Landuse	Emission factors derived from DMRB and TRICS database. Activity data from GIS database of commercial land use.	✓	a	b	✓	c	c	b
Shipping	Lloyd's Register and NAEI	✓	✓	✓	✓	✓	✓	✓
Airports	NAEI data and / or emission factors derived from aircraft movements (provide by each airport)	✓	✓	✓	✓	✓	✓	✓
Trains	DETR Emission factors. Activity data from Railtrack?	✓	✓	✓	✓	✓	✓	✓
Domestic Emissions	Sussex Domestic Gas Consumption Data and fuel derived emission factors. Activity data from population data and age of housing (disaggregated for each District based on Census data)	✓	✓	✓	✓	✓	✓	✓
Land Use Emissions	LRC London Energy Study for typical energy consumption rates and fuel derived emission factors. Activity data from GIS database of commercial land use.	✓	a	b	✓	c	c	b
Agricultural Activities	NAEI data and GIS disaggregation of urban and rural land use	✓	✓	✓	✓	✓	✓	✓
Note :	a data available, but not for this study b requires formatting to GIS c data included in Sussex Air Quality Emissions Inventory							

(Source : this study)

4.5 CONCLUSIONS

The compilation of an emissions inventory from the 'bottom up', to allow direct linkage with land use, transportation and economic forecast data, has been undertaken for Kent and Medway. Through the use of nationally available and published emission factors the resultant inventory of annual mean emissions of NO_x, PM_{10} and CO_2 is consistent with other UK national and urban emission inventories. This approach to developing the inventory is, in itself, unique and provides for much more detail both spatially and temporally. Whilst remaining consistent with DETR (1998a, 2000j) and draft DEFRA (2002b) local air quality management guidance, this study demonstrates how it could be extended to include greenhouse gases such as CO_2. This requires consideration of land uses not associated with local air quality management, all of which are subject to planning control.

To complete this study, a comprehensive review of emission inventory evaluation techniques was undertaken and used. Whilst accepting that no one evaluation technique is correct the results suggest a high degree of confidence in the inventory data enabling its use for development of appropriate land use and transportation policies across the study area. Although the use of tolerances consistent with the limited amount of literature available yields uncertainty estimates of emissions similar to NAEI data there appears to be over reliance on 'expert judgement' on assigning such tolerances with no clear guidance. For example, the draft guidance provided by DEFRA (2002b) for evaluating uncertainty in monitoring data runs to many pages with comparable guidance on emissions inventories being significantly shorter.

This study has therefore proven that an emissions inventory can be developed from the bottom up using 'real' data used by land use and transportation planners. This approach is unique and, although this has required some novel application of detail, the comprehensive evaluation exercise demonstrates its robustness for assessing the air quality and climate change implications of land use and transportation policies at the local to regional scale. Moreover, a review of the activity data available within neighbouring local authorities suggests the techniques described in this chapter could be used to extend the inventory at least across the whole of south east England, if not the whole country.

5 LOCAL TO REGIONAL SCALE DISPERSION

This Chapter describes the development of a number of novel techniques to describe the dispersion of atmospheric pollutants across the study area. Central to these techniques is the matrix model enabling rapid assessments of local air quality to be made. A comprehensive evaluation of the dispersion modelling tools are provided to demonstrate their robustness.

5.1 INTRODUCTION

In collaboration with the Department of Environment, the Royal Meteorological Society (1995) has issued guidelines on atmospheric dispersion modelling identifying ten points for consideration. Three of these are considered particularly relevant to this research:

- **justification of choice of model procedure**; this research demonstrates *fitness for purpose* for using the atmospheric emissions inventory and the modelling techniques described in this and the preceding chapter for estimating annual mean emissions and subsequent ground level concentrations

- **sensitivity analysis**; has been undertaken to establish the *robustness* of the atmospheric emissions inventory and modelling techniques in combination with *validation* of the techniques used

- **uncertainty and variability**; of the inventory and modelling techniques are defined through a combination of *verification* and *evaluation* of techniques and results.

This chapter describes the use and development of a number of dispersion models and their application across the study area of Kent and Medway using the emissions inventory described in Chapter 4. Although the development and use of atmospheric dispersion models is well documented (eg. Hanna *et al,* 1982, Zannetti, 1990, Namdeo and Colls, 1996, Middleton, 1997, DETR (1998c, 2000k, DEFRA, 2002b and Hall *et al,* 2000a) deciding which model to use requires an appreciation of the scales and uncertainties in the emissions inventory and concentration field. One of the key aims of this research is to develop tools enabling rapid evaluation (see Section 2.3.3) implying a compromise between accuracy and run time.

The first part of this chapter (Section 5.2) provides a review of the application of dispersion models and justification for the development of a new model procedure. Sections 5.3 to 5.7 describe in detail the approaches adopted for modelling:

- elevated sources
- ground level sources
- background contributions
- roadside air quality
- conversion of NO_x to NO_2.

These sections include the results of sensitivity analyses to demonstrate the robustness of the techniques developed during the course of this research. The final part of this chapter (Section 5.8) uses the results of a comprehensive review of model evaluation studies to identify and apply a range of model evaluation methods to determine model uncertainty and further demonstrate the robustness of the model procedure.

The dispersion models have been used in conjunction with the emission inventories described in Chapter 4 to evaluate a number of land use and transportation scenarios. An assessment of atmospheric emissions and air quality for the 'business as usual' and management scenarios is presented in Chapter 6.

5.2 APPLICATION OF DISPERSION MODELS

5.2.1 Short Range Models

Examples of using short range[23] models for regional scale studies include:

- Athens, Greece by Moussiopoulos *et al* (1997)
- Birmingham, UK by Veal and Appleby (1997)
- Espoo, Finland by Valkomen *et al* (1996)
- Helsinki, Finland by Karppinen *et al* (2000a-b)
- Istanbul, Turkey by Şen (1998)
- London, UK by Owen *et al* (1999, 2000) and Carslaw *et al* (2001)
- Wuppertal, Germany by Brűcher *et al* (2000).

23 Short range models are defined as those able to describe the dispersion of a pollutant released within the lower atmosphere up to a distance of approximately 50 km downwind.

The key inputs for all short range dispersion models are:

- source characteristics (point, line, area or volume)
- emissions data
- meteorological data
- topographical data
- wake effects (including buildings and street canyons)
- location of receptors.

Models also require a definition of averaging periods. This is normally based on aggregates of 1-hour; the common period for logging meteorological data. In rare circumstances, emissions data may be available in aggregates of 1-hour but are more usually available as long term mean or short term peak (maximum) values. In a study of air quality in the lower Trent Valley, UK, Futter (2000) used two well known models (ADMS-3 and AERMOD) for estimating ground level concentrations of SO_2 associated with coal fired power stations emissions. The study utilised historical hourly emissions data. The model results were compared to ambient monitoring data for the same period. Futter (2000) concluded that predictions of both models were comparable up to about the 99th percentile of hourly means with AERMOD estimating large over predictions and ADMS-3 moderate under predictions at higher percentiles. Both models were considered broadly representative for long term statistics but less successful for individual hours. Owen *et al* (1999) highlights the inherent assumption of steady state models, including ADMS-3 and AERMOD, that pollutants emitted during an hour will arrive at the receptor within the same hour. This will lead to differences in hour by hour comparison of modelled and monitored data over a regional scale domain. This is demonstrated by Veal and Appleby (1997) using modelling and monitoring data for Birmingham, UK, who report modelled and monitored peak hour concentrations of similar magnitude but occurring at different times. In describing the development of a highly complex model of Graz, Austria, Almbauer *et al* (2000) reach a similar conclusion emphasising the model's ability to estimate annual mean concentrations.

Modelling guidance published by the DETR (1998c, 2000k) refers to six screening models, six intermediate models and thirteen advanced models. To achieve the objectives set out in Section 4.1, this research requires a model similar in performance to advanced models. The Users' Manual for one advanced model, ADMS-3 (CERC, 1999) includes an estimated run time of one hour for a single source, no complex terrain, 441 receptors and one year of hourly sequential meteorological data. These data suggest the

user would wait approximately one week to run ADMS-3 for all 162 Part A sources (Section 4.3.1) and for only a limited grid of receptors within Kent and Medway. Evidently computer processing speeds are increasing along with improved model optimisation by the developers but personal communication with a number of local authority users of advanced models for urban area studies suggest each model run can take in the order of days (usually one or two weeks) to run. An extreme example of using significant computing resources is the model of Helsinki described by Karppinnen *et al* (2000a-b) that requires 24 hours to run on a super-computer Cray C94. This represents a significant time limiting step within the overall process of developing an AQAP suggesting a practical limit to the number of management options that can be considered (see also the example of assessing the impact of a proposed London Airport in North Kent described in Chapter 6). As argued in Chapter 1 of this thesis, the longer an assessment of a management scenario takes the less likely it is to be considered. Long model run times limit the ability of the air quality manager to consider all available options in developing an AQAP. This explains one of the requirements expressed by local authority officers for a rapid assessment tool (Section 1.3.3). Moreover, long model run times also reduce the ability for model sensitivity studies and subsequent robustness of the assessment. This is considered further in Section 5.7 (model evaluation). The conclusion from this - that a robust model with practicable run times for the study area is limited to estimating annual mean concentrations only - is supported by the conclusions reached in preceding Chapters.

5.2.2 Model Procedures

An overview of the model procedures developed during this research is provided in this subsection. The structure of most urban models (in this context, defined as those including point, line, area and volume sources) is to use a point source model separate from an area, volume and / or line source model. The point source model used to estimate dispersion from elevated sources (i.e. Part A Processes) was the proprietary model AERMOD (see Section 5.3 for definition) developed for the United States Environmental Protection Agency (USEPA). The sensitivity of this model to input data, particularly topographical and meteorological data, is considered in Section 5.3. The dispersal of ground level (line and area source) emissions has been modelled using a technique developed specifically for this research – the *Matrix Model.* Similar

approaches have been adopted in the development of national maps of UK air quality, e.g. Abbott and Vincent (1999). Stedman *et al* (2001a,b) used the approach described by Abbott and Vincent (1999) and also modelled the contribution made by large industrial sources depicted as 1 km^2 area sources with a release height of 50 m. The sensitivity of the *Matrix Model* to input parameters is considered in detail. The *Matrix Model* provides estimates of non-roadside concentrations. A roadside enhancement factor has been developed for this model as part of this research. Empirically derived NO_x and PM_{10} relationships are included to describe background contributions and atmospheric chemistry to provide total estimates of annual mean concentrations of NO_2 and PM_{10}.

5.2.3 The Matrix Model

The simplest form of describing dispersion from ground level sources within an urban area is to assume emissions (Q_a in g/s) are constant over a distance (Δx in m) running approximately from one edge of the urban area to another. The pollutant is assumed uniformly mixed in a layer of depth (z in m) between the ground and the mixing height, and the wind speed (u in m/s) is constant. If conditions are steady state (no change in concentration or mixing height with time) the solution is:

$$C = \frac{\Delta x}{z_i} \cdot \frac{Q_a}{u} \qquad \ldots(7)$$

This is the simple *Box Model* and has been widely used. Lyons and Moy (1998) for example, describe using the Box Model to illustrate the strong relationship between vehicle kilometres travelled, urbanised area and air quality for thirty major cities around the world. Ballasio (1997) uses a formation of the box model to describe the build up of traffic pollution within a tunnel. Hanna *et al* (1982) describe how this model was developed further into the *Atmospheric Turbulence and Diffusion Laboratory (ATDL) Model*; a box model with the height of the top lid proportional to the vertical dispersion parameter σ_z (Δx) rather than the mixing height; the former being significantly less for grid distances less than 10 km. The ATDL model was used to develop the first county wide dispersion model for Kent and Medway in conjunction with the Industrial Source Complex model for point sources (KCC, 1995).

For a study area as diverse as Kent and Medway, including both urban and rural elements, the use of a simple box model is inappropriate as emissions are not uniform. The ATDL model, in which Kent and Medway was separated in to 1 km grid square area sources, is more realistic. However, at the 250 m grid square scale, the model run time was estimated to be in the order of days. Further problems were encountered in attempting to incorporate the model within the GIS platform.

After applying the ATDL model to several urban areas, Hanna *et al* (1982) noted the calculated concentration $C_{x,y}$ was usually proportional to the emission $Q_{x,y}$ in the grid square in which the receptor was located. This observation led to the development, in the course of this research, of the *Matrix Model* described below where the contribution of an area source to each receptor is spatially proportional. This model assumes each area source – or 'box' – is similar in dimensions and emission characteristics. For a unit emission, therefore, the *pattern* of dispersion and subsequent concentrations in surrounding receptors is dependant on meteorology alone. This pattern – or *matrix* – can be applied to each emission source and the results summed for each receptor; hence removing the dispersion element of the calculation. The basic matrix is formed using a dispersion model to derive factors between a unit area emission and contributions to concentrations in a grid of receptors, centred on the area source. The matrix is then applied in GIS using a script similar for interpolation of receptor values, cutting out the numerically intensive calculations for each source. This approach reduces the number of calculation steps required to estimate the concentration in each receptor and, with over 60, 000 receptors in the study area, significantly reduces the processing time to less than one minute.

The *Matrix Model* can be considered similar to the *transfer-matrix* derived by Seika *et al* (1998) using the Ambient Background Model. The *transfer-matrix* describes the relationship between emissions sources and annual mean concentration at a given receptor for a given meteorological data set, emissions pattern and emission height. Seika *et al* (1998) then propose using the *transfer-matrix* for air quality management purposes; evaluating the effect of various emission reduction scenarios. Stedman *et al* (2001a) recently adopted a similar approach in modelling national projections of PM_{10} and NO_x emissions in support of the DETR (2001c) consultation on proposed air quality objectives.

Extension of the *Matrix Model* to derive a three dimensional description of regional scale dispersion and hence, include regional chemistry, is not recommended. Sportisse (2001) provides a theoretical comparison between Box and Eulerian Models, the latter commonly used to evaluate long range dispersion at regional and continental scales (e.g. Stedman and Williams (1992) Sofiev (2000) and Malcolm *et al* (2000)) concluding that the implicit assumption of uniform mixing within each box is not sufficient to represent the non-linear atmospheric chemistry and deposition.

In accordance with best practice (Royal Meteorological Society, 1995) this Chapter includes the results of a sensitivity analysis of the *Matrix Model* to various inputs. In describing the development of a new technique for modelling traffic air pollution in tunnels, Bellasio (1997) considers this is a fundamental part of the model validation; to identify the critical inputs and compare the sensitivity of nature to these same inputs. In comparing ADMS predictions of power station SO_2 emissions in the lower Trent Valley with observed data, Futter (2000) used sensitivity analysis to conclude that small variations in meteorological inputs can lead to significant step changes in modelled concentrations, principally as a result of the transition from stable to unstable conditions but also due to the effect of increasing boundary layer depth. Lanzani and Tamponi (1995) undertook sensitivity studies of a microscale Lagrangian particle model for an urban street canyon, describing the sensitivity of the model to receptor location within the canyon and wind speed. Evidently, the model described in this research is not applicable to specific situations such as tunnels or street canyons. However, a review of published work for these two scenarios suggests the data compiled for this study would allow such investigations to be undertaken. For example, Vardoulakis *et al* (2000) used the AEOLIUS model to predict carbon monoxide and benzene concentrations within a Parisian street canyon. This model, developed by Buckland (1998; see also Buckland and Middleton, 1999, Rafferty, 1997) requires hourly traffic flows and a background concentration value, the latter could be derived using the *Matrix Model.*

5.3 DISPERSION OF ELEVATED EMISSIONS

The dispersal of pollutants released from elevated point sources is best described using one of the new generation dispersion models; referred to by DETR (2000k) as advanced models. Those commonly used in the UK include ADMS and AERMOD. ADMS (UK

Atmospheric Dispersion Modelling System) described by Hunt *et al* (1991) and CERC (1999) was originally developed in the UK with part funding by the Environment Agency. This model represents a significant departure from the traditional Gaussian based approach to describing the dispersion of pollutants during convective (unstable) atmospheric conditions. During stable conditions the model reverts to the Gaussian approach. AERMOD, described by the United States Environmental Protection Agency (2001) was developed by the US American Meteorological Society and Environmental Protection Agency Regulatory Model Improvement Committee. In a number of respects AERMOD is similar to ADMS-3 as both incorporate the latest understanding of the atmospheric boundary layer and are certainly an improvement compared to Gaussian models such as R-91 or ISC (DoE, 1996). A model inter-comparison by Hall *et al* (2000a,b) included running ADMS-3 and AERMOD with the same pre-processed meteorological data; both models generated similar results for a scenario with no complex terrain or building effects. Other reported model inter-comparison studies (e.g. Futter (2000) Veal and Appleby (1997) and McHugh *et al* (1999)) are consistent in demonstrating similar performance by the two models to estimate annual mean concentrations. Ireland *et al* (2002) consider the similarities between ADMS-3 and AERMOD sufficient to advocate the use of both models in the assessment of point sources, recognising the overall consistency in model performance as a means of increasing the robustness of the assessment and providing an indication of model uncertainty. Ireland *et al* (2002) further consider any differences in the results generated by the user should lead to a more intensive investigation in to why such differences were occurring including sensitivity analysis of input data.

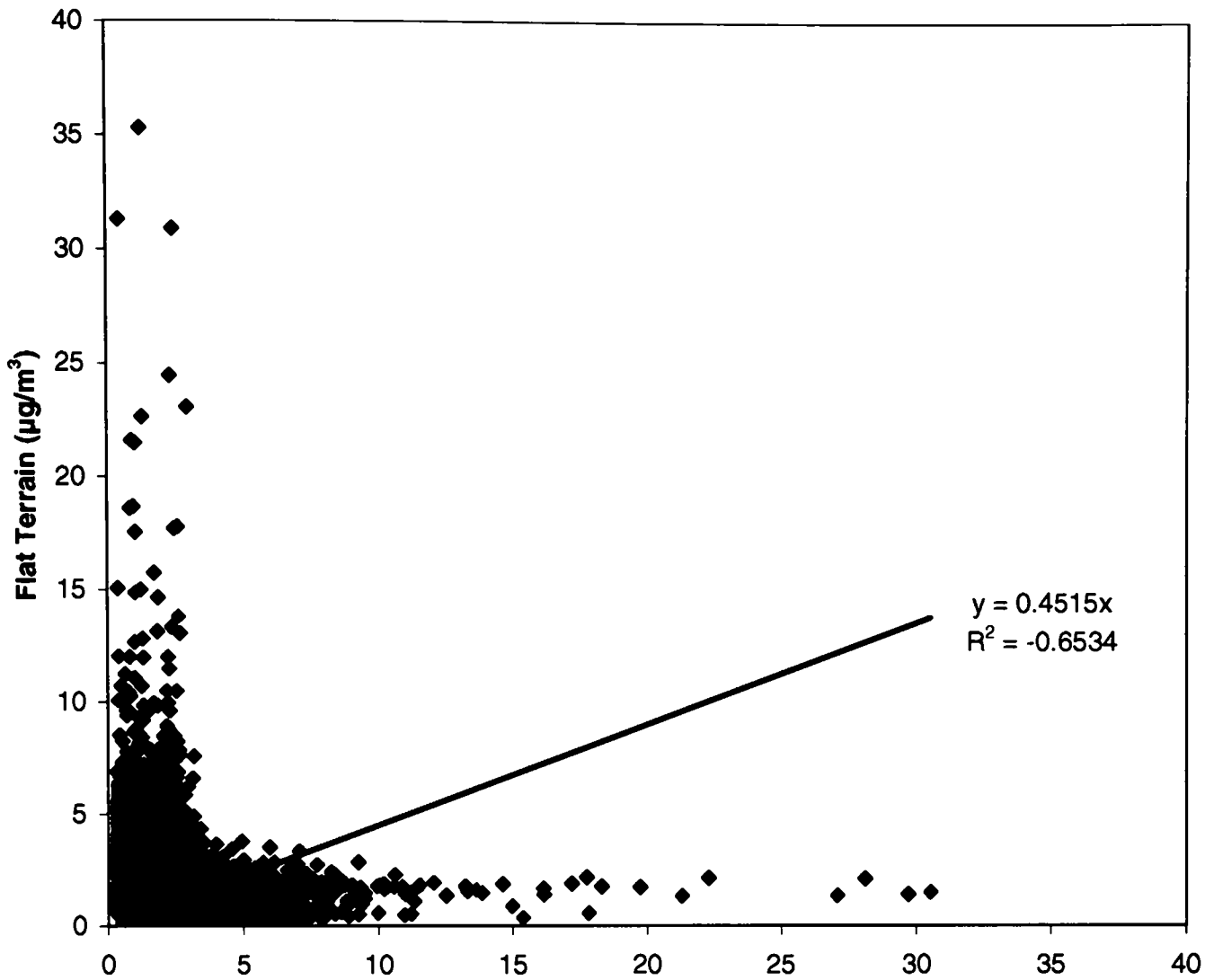
5.3.1 Receptor Grid and Topographical Data

Receptors were defined for a regular cartesian grid aligned with the 250 m by 250 m stock matrix adopted for ground level emissions (Section 4.1.2). The DEFRA *model helpline* recommends using a receptor spacing of 10 m for defining the extent of traffic source related AQMAs (*personal communication*). This is clearly impractical over the study area and is far in excess of the resolution of emissions data developed in the course of this research or available to local authorities for the purposes of LAQM. The scale selected for this study represents an optimum compromise between spatial resolution

and model run time, and allows for easy results analysis including population exposure, for example, described in Section 5.8.7.

The topography of the study area has been published previously (KCC, 1995). Topographical data are available digitally from the Ordnance Survey for the UK in National Terrain Format (NTF) files. Unfortunately, the cost of these data precluded their use in this research. Using a 1 km by 1 km Cartesian grid of receptors, spot heights were read off Ordnance Survey 1:50,000 sheets and the data processed using AERMAP (the terrain pre-processor for AERMOD). Using 1990 NO_x emissions data for Part A Processes, AERMOD was run with both flat and elevated terrain for the 1 km by 1 km Cartesian grid of receptors across Kent and Medway. In the case of modelling with flat terrain, the source elevations were all set to zero. The results, expressed as annual mean NO_x concentrations, are presented in Figures 5.1 and 5.2.

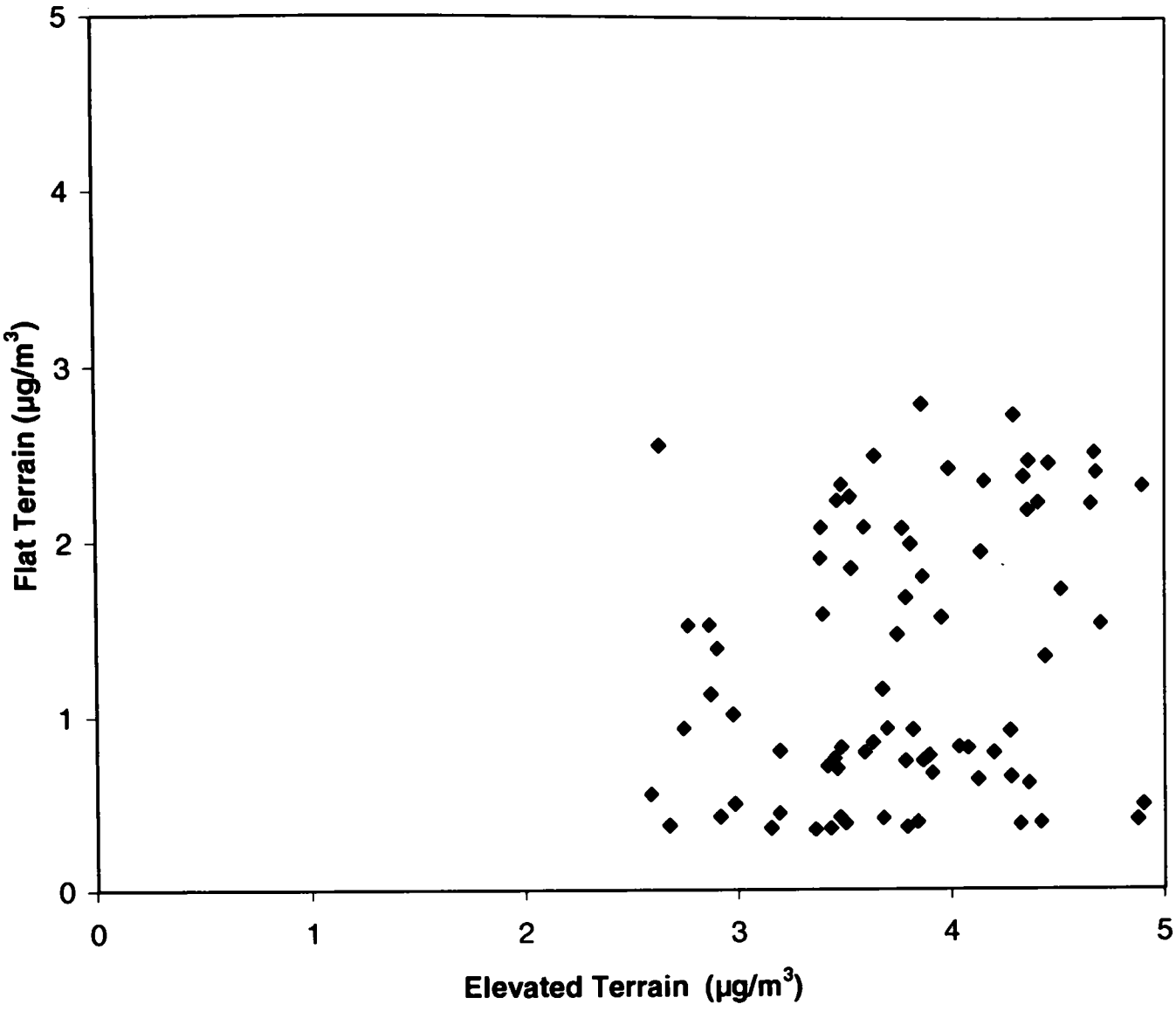
AERMOD was used to model the effects of topography across the whole of the study area (Figure 5.1) providing only a weak conclusion that the effect of elevated and complex terrain is significant. The majority of the study area is relatively flat and hence, the effects of terrain are limited. Restricting the model domain to the North Kent / Medway area (Figure 5.2) characterised by the North Downs and complex terrain associated with the Medway Valley (see Section 4.1.3 and Figure 4.1) clearly demonstrates the effect of elevated and complex terrain is significant in the dispersal of elevated emissions. Adopting the precautionary principle requires the inclusion of elevated terrain when modelling elevated emissions. The dispersal of ground level emissions is not significantly affected by elevated terrain.



Scatterplot of annual mean NO_x concentrations modelled using AERMOD and Gatwick (1996) meteorological data (9043 data pairs). The model was first run with elevated sources and a Cartesian grid of elevated receptors at 1 km intervals (x axis). The results are compared to running the model with the base elevation of all sources set to zero and a flat grid of receptors (y axis).

FIGURE 5.1. : EFFECTS OF TOPOGRAPHY ON A UNIFORM 1 KM GRID FOR KENT AND MEDWAY

(Source : this study)



Scatterplot of annual mean NO_x concentrations modelled using AERMOD and Gatwick (1996) meteorological data within the North Kent/Medway Valley area only – an area of elevated and complex terrain (80 data pairs). The model was first run with elevated sources and a Cartesian grid of elevated receptors at 1 km intervals (x axis). The results are compared to running the model with the base elevation of all sources set to zero and a flat grid of receptors (y axis). The results confirm running the model with an elevated terrain produces higher estimates of ground level concentrations (see Figure 5.1).

FIGURE 5.2. : EFFECTS OF TOPOGRAPHY ON A 1 KM GRID WITHIN AN AREA OF ELEVATED AND COMPLEX TERRAIN

(Source : this study)

5.3.2 Surface Roughness

Surface roughness affects local wind speed and vertical mixing by inducing turbulence. A high surface roughness will result in lower near surface wind, for example. The surface roughness is a function of land use and will therefore vary across the study area. For this study a constant surface roughness length was assumed across the whole study area. Evidently this is not the case but is a restriction within dispersion models such as ADMS or AERMOD. Bottema (1997) describes some progress in modelling urban roughness, by varying surface roughness length dependant on source and receptor locations, which could potentially be incorporated within dispersion models. Owen *et al* (1999) adopted a surface roughness of 2 m representing 'cities' and 'open woodland' to model London. Rotach (1999) describes the roughness sublayer, which may extend 2-5 times the local building height and hence, is considered important in urban modelling of low level sources. Rotach (1999) considers this to partly explain the underprediction by urban models and demonstrates, using data for Copenhagen, improved representation of the roughness sublayer can improve model performance. Buckland and Middleton (1999) describe the Europe-wide use of the Operational Street Pollution Model, developed by Berkowicz *et al* (1996) and also consider the type of building and urban layout to be important; see also Hall *et al*, 1996, 1999. These observations imply that models developed in Europe may not be wholly applicable elsewhere, including extreme locations such as Hong Kong (Xia and Leung, 2001).

The effect of individual buildings on local dispersion is well documented (e.g. Hall *et al*, 1996, 1999) but was not included implicitly in this study. Practical constraints precluded the compilation of dimensions for individual buildings across the study area and, given the normal change in urban landscape expected over a thirty year period, such an exercise was considered unlikely to improve model accuracy. Recent wind tunnel studies by Mavroidis and Griffiths (2001) of urban dispersion concluded the effect of individual buildings was not as great as the effect of turbulence created due to general urban building array. This suggests the use of a singular descriptor, such as surface roughness, may be sufficient for the purposes of this research. For this study, a surface roughness of 0.5 m was adopted, representing 'parkland' and 'open suburbia'. The sensitivity of the model to surface roughness cannot be treated rigorously although sensitivity to meteorological data gives insight into potential inaccuracies (see next

section). The effect of surface roughness is not expected to be large as this study is not considering major urban areas such as London.

5.3.3 Meteorological Data

In describing the development of the Kent Air Quality Management System, KCC (1995) identified four meteorological stations providing observations suitable for dispersion modelling studies in Kent:

- **Gatwick**; representative of inland areas of Kent
- **Manston** and **Shoeburyness**; representative of coastal zones to the north and east
- **Herstmonceux**; representative of elevated areas (the High Weald).

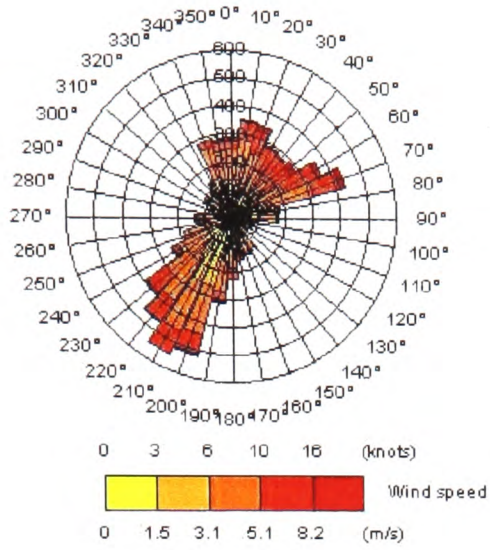
The original work by KCC (1995) was based on using long term (10 year) statistically aggregated data. For the purposes of this work, the decision to use hourly sequential data was made should studies of short-term episodes be required in the future. Studies of nitrogen dioxide and PM_{10} episodes and modelling reported in the AQS (DETR, 2000b) consistently use either 1995 or 1996 meteorological data. 1996 is considered a 'typical' year in terms of the effect meteorology has on air quality. 1995 is considered a 'worst case' year with higher prevalence of easterly winds increasing the influence of secondary pollutants originating from the continent, giving rise to more frequent episodes of high ozone (increasing the formation of NO_2) and secondary PM_{10}.

The DETR (2000k) and the Environment Agency (2002) recommends three and five years of meteorological data respectively. However, a limited budget dictated purchase of hourly sequential data, suitable as input for both AERMOD and ADMS 3, for Gatwick (1996) to provide a reasonable representation of inland areas and Manston (1995, 1996) to provide an indication of coastal effects (see Figure 5.3). Although the meteorology associated with the High Weald is not studied, the low population of this area and low density of emissions suggests the results are not critical to this omission.

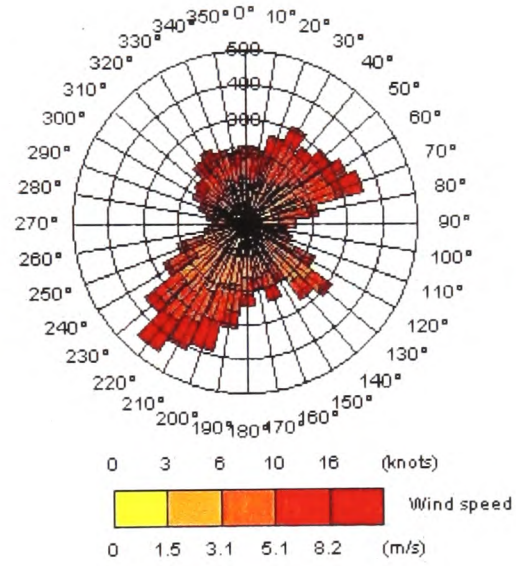
By using three meteorological data sets, this work aims to provide some indication of the model sensitivity to these data in a systematic manner. The raw data are themselves subject to uncertainty. For example, Seibert *et al* (2000) compare seven operational methods for determining the mixing height; recognised by Futter (2000) as a critical input. Hall *et al* (1999) compared data sets based on the same observations but provided

by UK and US suppliers, concluding that the source of data (either UK or US) was not important but should be acknowledged with the key recommendation that several years of data should be used to assess year on year variability and increase the number of combinations of meteorological data modelled. No assessment of meteorological uncertainty is currently made of data supplied for dispersion modelling. However, as the use of meteorological modelling to provide short-range dispersion model input data increases, some definition of uncertainty is expected (Seaman, 2000).

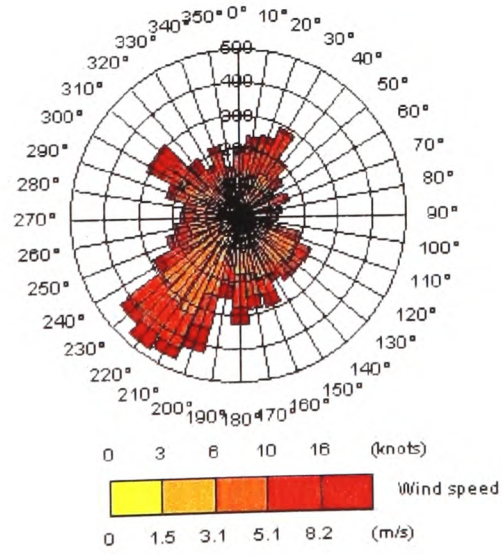
To test the sensitivity of the model to meteorological data, 1990 NO_x emissions data for Part A Processes were used, running AERMOD with an elevated terrain for the 1 km by 1 km Cartesian grid of receptors across Kent and Medway. The results, expressed as annual mean NO_x contributions (concentrations) from modelled sources, are presented in Figure 5.4. The effect of background contributions and atmospheric chemistry were excluded from this analysis. The results suggest, for the three meteorological datasets used, Gatwick 1996 data generates the most conservative results across the study area. In contrast, comparison of the absolute maximum values (31.59 $\mu g/m^3$, 25.88 $\mu g/m^3$ and 30.53 $\mu g/m^3$ for Manston 1995, Manston 1996 and Gatwick 1996, respectively) suggests Manston 1995 data should be used. However, given that the aim of this study is to assess air quality across the study area as a whole, the conclusions drawn from Figure 5.4 are considered less sensitive to individual source contributions, more robust and more applicable.



(a) Gatwick (1996) data exhibiting the typical UK pattern of predominantly south westerly winds with a secondary predominance of north easterlies.



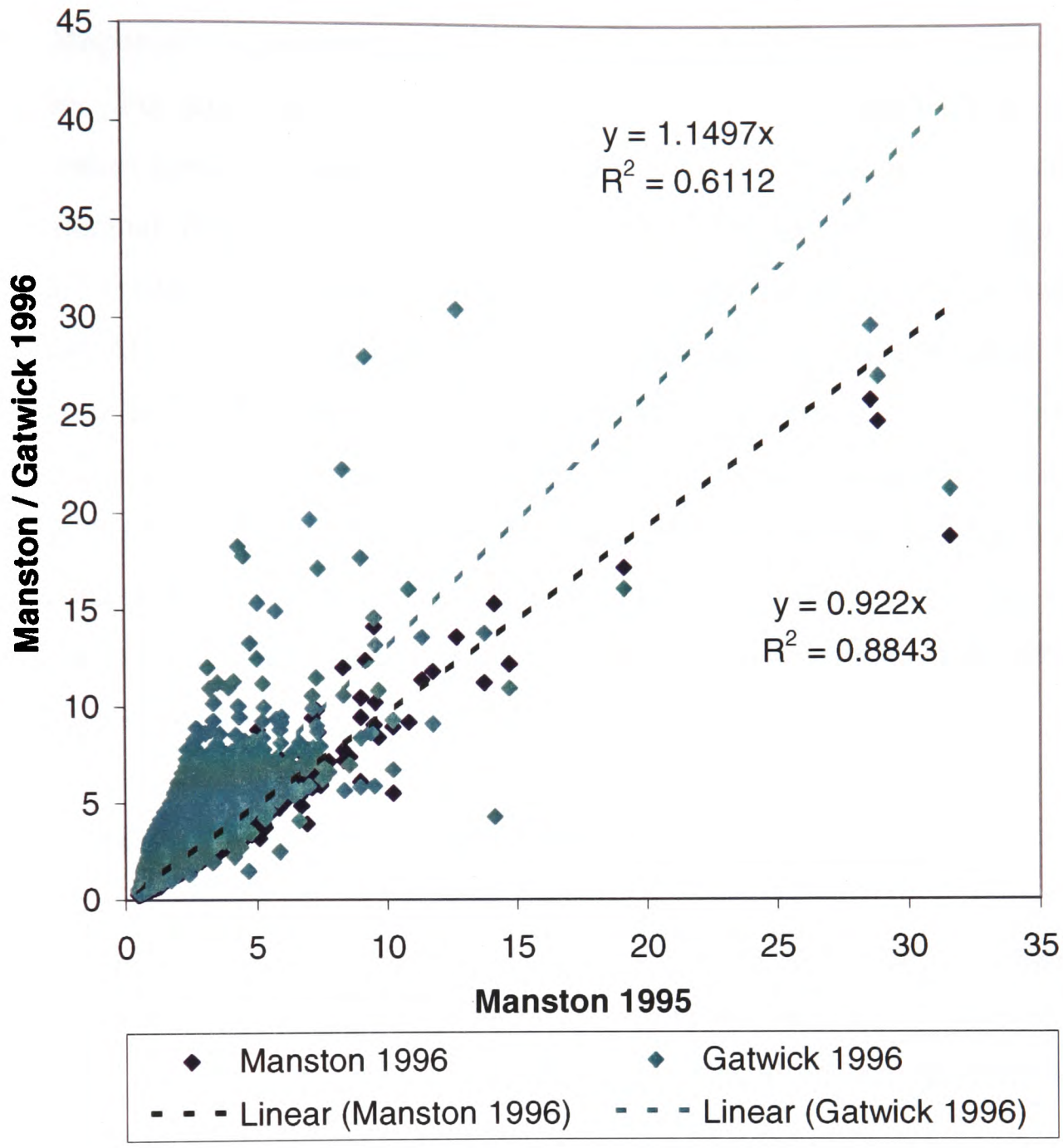
(b) Manston (1996) data illustrating the influence of coastal winds (stronger winds from the north east and the presence of winds from the south east) compared to inland observations made at Gatwick.



(c) Manston (1995) data also exhibits predominance of winds from the south west but with a secondary predominance of winds from the south east rather than the north east.

FIGURE 5.3 : REPRESENTATIVE WEATHER STATIONS

(wind roses constructed from data supplied by Trinity Consultants, Inc)



Scatterplot of annual mean NO_x concentrations modelled using AERMOD and three meteorological data sets (Gatwick, 1996 and Manston, 1995, 1996) (9045 data pairs for both plots). Comparing the Manston 1995 and 1996 results (in blue) suggests the latter are slightly more conservative over the study area, by approximately 8%. The Gatwick 1996 results (in green) are more conservative than Manston 1995 results by approximately 15%. Running the model with Gatwick 1996 data appears to generate the most conservative results for the meteorological data selected.

FIGURE 5.4 : ELEVATED SOURCE MODEL SENSITIVITY TO METEOROLOGY

(Source : this study)

5.3.4 Building Dimensions

The initial dispersal of a pollutant can be subject to considerable influence from building wake effects. For an elevated point source, experience suggests this tends to increase maximum (short term) concentrations and shift them closer to the source. The effect on long term (annual mean) concentrations is much less significant. Building wake effects are normally confined to situations with low stack heights, relative to local buildings. For example, AERMOD assumes there are no building wake effects if the stack is more than 2.5 times the height of the stack, whereas ADMS-3 assumes a factor of three. The assessment and subsequent management of this type of complex situation is considered outside the local air quality management regime but falls in to the remit of industrial pollution control when more focus is given to potential short term impacts. The scope of this study is limited to annual means so individual building data for each elevated point source were not obtained.

5.3.5 Compiling Point Source Model Results

Given the long run time for the point source model (>5 hours) a *QBasic* routine was written to generate an (*x, y, z, c*) ASCII text file of receptors based on a 250 m by 250 m Cartesian grid, with each receptor at the centre of the corresponding 1 km by 1 km grid square assumed to have the same height and annual mean concentration.

This method precludes the use of ADMS-3 which requires inclusion of the complete terrain field to calculate the wind field. The version of ADMS-3 available is limited to 1200 receptors. For regional scale studies the density of the receptor grid would be too coarse although a restricted access version of this model (ADMS-Urban) can handle larger receptor grids.

5.4 DISPERSION OF GROUND LEVEL EMISSIONS

5.4.1 Developing the Matrix Model

The structure of the matrix and dispersion pattern used as the basis for the model developed for this study is described in Figures 5.5 and 5.6. The dimensions of the

matrix were resolved using area sources of 250 m by 250 m as a compromise between resolution and size of data files. Although greater resolution is possible as computers become more powerful, there is considered to be a practical constraint in the availability of land use data at resolutions greater than the scale adopted in this study.

The concentration at any given receptor will be the sum of contributions made from 49 area sources:

$$\mathbf{[receptor]}_{x,y} = \sum \mathbf{(emission)}_{i,j} \mathbf{\times (dispersion\ factor)}_{-i/250,\,-j/250} \quad \textbf{...(8)}$$

for: $i = -750$ to $+750$ and $j = -750$ to $+750$.

where: (x, y) is the location of the receptor
(i, j) is the location of the emission source
$(-i/250, -j/250)$ is the relative location of the dispersion factor for emission source $(i/250, j/250)$

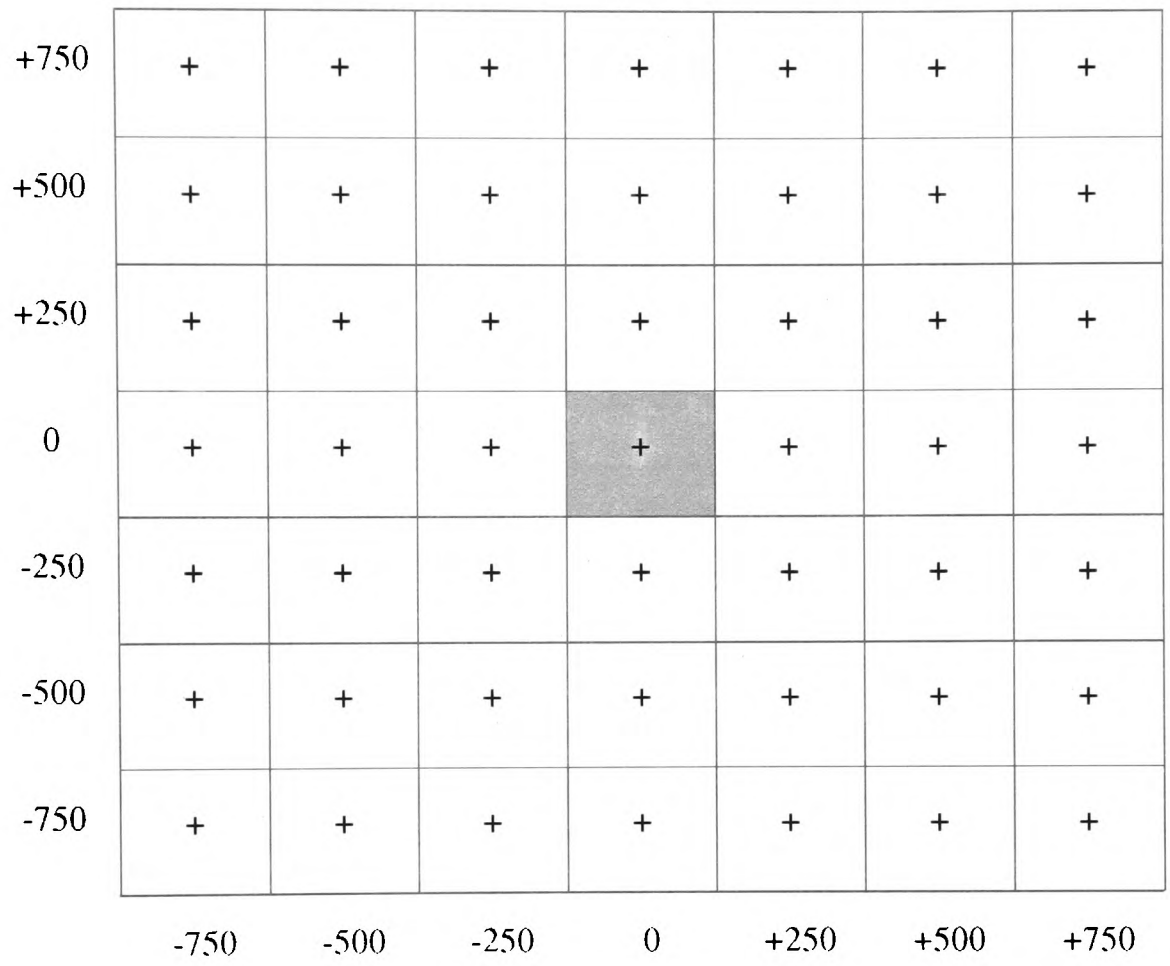
(all distances in metres)

The matrix pattern was applied to the emissions inventory in ArcView using the *ConFiT* subroutine originally developed by Behrens (2001) to provide a method for interpolating between points on a regular grid.

Stedman *et al* (2001a) provide coefficients for estimating the background NO_x and PM_{10} concentrations ($\mu g/m^3$) less the rural contribution, from 1 km^2 area sources (tonnes per annum) using an empirical approach. For large urban areas, the coefficients are in the ranges 3.66 to 4.53 and 3.6 to 6.01 for NO_x and PM_{10}, respectively. For elsewhere in the UK, the coefficients are in the ranges 7.18 to 8.61 and 7.63 to 11.95. Allowing for the difference in area dimensions, the coefficients reported by Stedman *et al* (2001a) are consistent with the dispersion matrix developed in this study. However, Stedman *et al* (2001a) do not provide an explanation for the range of coefficient values. The approach adopted in this study is considered more systematic being based on a source-receptor relationship.

Analysis of area emissions data for 1998 maintained in the National Air Quality Information Archive (*http://www.airquality.co.uk/archive/index.php*) suggests typical urban non-roadside NO_x emissions were in the order of 10 - 20 tonnes per km^2 in Central London (Camden) and up to 3 tonnes per km^2 in provincial towns (Brighton). Applying

the central dispersion factor provided in Figure 5.7 suggests an estimated annual mean urban background NO_x concentration in the order of 48 - 96 $\mu g/m^3$ for Central London and up to 15 $\mu g/m^3$ for smaller urban areas; within the range of observations reported in the AQS (DETR, 2000b).



The structure of the matrix model is shown above. The central square, shaded grey, represents the 250 m by 250 m area emission source. The receptors are centred within each grid square (receptor location denoted by '+'). The dispersion model (AERMOD or ADMS) is run with a single area emission source centred at (0,0) with a flat Cartesian receptor grid of origin (-750, -750) and spacings of 250 m.

FIGURE 5.5 : Basic Structure of the Matrix Model

(Source : this study)

+750	0.026	0.040	0.072	0.150	0.218	0.161	0.079
+500	0.039	0.054	0.109	0.302	0.382	0.163	0.070
+250	0.061	0.099	0.202	1.003	0.572	0.136	0.059
0	0.094	0.189	0.612	4.757	0.391	0.097	0.047
-250	0.109	0.179	0.355	0.684	0.209	0.072	0.038
-500	0.070	0.095	0.168	0.209	0.139	0.054	0.033
-750	0.045	0.069	0.098	0.103	0.092	0.048	0.025
	-750	-500	-250	0	+250	+500	+750

The results of running AERMOD with Gatwick 1996 data, surface roughness = 0.5 m, are presented above to illustrate the dispersion pattern. The results in each square are annual mean concentrations (in $\mu g/m^3$) of a pollutant released from the central area source. The area source represents a unit release of one tonne per annum per 250 m by 250 m area source. In the model this is equivalent to 5.074×10^{-7} $g/m^2/s$. This dispersion pattern is for a release at 3 m above ground level with an initial vertical dimension of 5 m.

The highest concentration is calculated at the central receptor. As expected, with the predominance of south westerly winds observed in the UK, the higher concentrations are towards the north east. The concentrations for the receptors at the edge of the matrix are less than 5% of the central receptor value; increasing the matrix size beyond the (7, 7) array size would not significantly affect the model results.

FIGURE 5.6: The Dispersion Pattern

(Source : this study)

5.4.2 Matrix Model Sensitivity Analysis

The general dispersion pattern for the *Matrix Model* was developed using both ADMS and AERMOD in a series of sensitivity analyses described below. Both AERMOD and ADMS include algorithms for modelling area and volume sources although only the latter calculates concentrations within a volume source. The inputs for modelling an area source within ADMS and AERMOD are compared in Table 5.1.

Table 5.1
Area Source Inputs

AERMOD	ADMS	Comments
x and y co-ordinates of S-W corner (m) x and y length (m) angle from North (°)	x and y co-ordinates of each corner (m)	Both models similar in defining the dimensions and location of the area source.
base elevation (m)	base elevation (m)	In both cases the source was defined as 250 m by 250 m at 0 m base elevation.
release height (m)	release height (m)	Both models similar in defining the initial release height. In both cases the initial release height was defined between 1 – 5 m.
initial vertical dimension (m)	not available	AERMOD includes an option to define the initial vertical dimension to the area source plume. USEPA (2001) states "*this parameter may be important when the area source algorithm is used to model mechanically generated emission sources, such as mobile sources. In these cases, the emissions may be turbulently mixed near the source by the process that is generating the emissions, and therefore occupy some initial depth*". This option is not available with ADMS. The initial vertical dimension in AERMOD was defined between 1 – 15 m with the minimum equal to the release height and the maximum representing a well mixed urban centre street canyon. Stedman *et al* (2001a) suggest 10 m for traffic and 30 m for non-traffic sources.
emission rate ($g/m^2/s$)	emission rate ($g/m^2/s$)	Both models similar in defining the emission rate. In both cases the emission rate was defined as 5.074×10^{-7} $g/m^2/s$ (equivalent to 1 tonne per annum per 250 m by 250 m grid square)

(Source : this study)

Both models were run to assess sensitivity to release height and initial vertical dimension using hourly sequential meteorological data for Gatwick (1996). Further analysis was made of the models' sensitivity to different meteorological data (Manston, 1995 and 1996) and surface roughness lengths. ADMS was also used to derive a dispersion pattern based on a volume source with the height defined by the initial vertical dimension. As the dispersal of ground level emissions is localised, with the highest concentration within the area source itself, the effects of terrain on the dispersion of these sources is ignored. The results are presented in Tables 5.2 to 5.5 and discussed below.

The two models predict higher concentrations with area sources closest to ground level, both in terms of release height and initial vertical dimension (Table 5.2). The spread of results using ADMS is wider than for AERMOD. Both models provide consistent results with an area source of release height 3 m and an initial vertical dimension of between 5 to 10 m. These values correspond well with reality.

Although variations with different meteorological years are observed (Table 5.3) the two models generally agree with the ADMS results within the range of AERMOD results. For the three meteorological data sets used, the Gatwick (1996) data generates the highest concentrations.

Grosch and Lee (1999) report the results of sensitivity studies using AERMOD to model surface and elevated sources with a range of surface characteristics which influence local meteorology (albedo, bowen ratio and surface roughness length). The results indicate the model is sensitive to all three parameters in short term modelling but sensitive only to the surface roughness length when modelling annual means. ADMS and AERMOD were run with different surface roughness lengths to assess the sensitivity of the Matrix Model to this parameter (Table 5.4). For AERMOD, this required rerunning the AERMET pre-processor to generate the relevant input parameters. The AERMET pre-processor allows the albedo, daytime Bowen ratio and surface roughness length to be varied (see USEPA (1998) for definitions). In this work, only the surface roughness length was varied as this has the greatest effect. Both models generated comparable results for surface roughness lengths between 0.2 m and 0.5 m, ie. the ADMS values were in the range of AERMOD values.

Modelling a volume source with ADMS did not generate consistent results; increasing the initial vertical dimension is expected to reduce concentrations. The model does not follow this expected pattern, albeit the values generated were within the range of values generated by both ADMS and AERMOD for an equivalent area source.

In conclusion, the results presented in Tables 5.2 and 5.4 suggest both models appear to generate reasonably consistent results when modelling an area source with a release height of 3 m, an initial vertical dimension in the range 5-10 m and a surface roughness length of 0.5 m. These conditions are considered to most represent the situation being modelled. Use of Gatwick (1996) meteorological data represents the worst case for the data selected (see Table 5.3) although it is recognised analysis with further years would be ideal. The results presented in Table 5.5 suggest modelling the source as a volume does not appear to provide robust data.

Table 5.2
Sensitivity of ADMS and AERMOD to Area Source Model Inputs

Release Height (m)	Initial vertical Dimension (m)	ADMS ($\mu g/m^3$)	AERMOD ($\mu g/m^3$)
1	1	10.688	8.254
1	5		5.466
1	10		3.775
1	15		2.866
2	2	6.930	6.713
2	5		5.299
2	10		3.714
2	15		2.829
3	3	4.637	5.427
3	5		4.757
3	10		3.368
3	15		2.557
4	4	3.157	4.374
4	5		4.156
4	10		3.028
4	15		2.304
5	5	2.195	3.557
5	10		2.724
5	15		2.091

Notes: Gatwick (1996) meteorological data used with surface roughness of 0.5 m
Model results presented for central receptor, ie. at the centre of area source; see Figure 5.7.

(Source : this study)

Table 5.3
Sensitivity of Area Source Models to Meteorological Data

Meteorological Data	ADMS ($\mu g/m^3$)	AERMOD ($\mu g/m^3$)
Gatwick (1996)	4.637	3.368 - 4.757
Manston (1995)	2.715	1.991 - 2.701
Manston (1996)	2.985	2.163 - 2.974

Notes: Model results presented for central receptor, ie. at the centre of area source; see Figure 5.7
Surface roughness = 0.5 m for both models.
Release height = 3 m for both models.
The range of AERMOD results corresponds to initial vertical dimensions of 5 to10 m.

(Source : this study)

Table 5.4
Sensitivity of Area Source Models to Surface Roughness

Surface Roughness (m)	ADMS ($\mu g/m^3$)	AERMOD ($\mu g/m^3$)
0.1 ('root crops')	3.307	3.335 - 5.221
0.2 ('agriculture – minimum')	4.230	3.409 – 5.158
0.3 ('agriculture – maximum')	4.659	3.426 - 5.049
0.5 ('parkland, open suburbia')	4.637	3.368 - 4.757
1.0 ('cities, woodlands')	3.931	2.888 – 3.876
1.5 ('large urban areas')	3.502	2.453 - 3.219

Notes: Gatwick (1996) meteorological data used with surface roughness of 0.5 m
Model results presented for central receptor, ie. at the centre of the area source; see Figure 5.7.
The range of AERMOD results corresponds to initial vertical dimensions of 5 to10 m.

(Source : this study)

Table 5.5
Comparison of Modelling Area and Volume Sources

Release Height (m)	Initial vertical Dimension (m)	Volume Source ADMS ($\mu g/m^3$)	Area Source	
			ADMS ($\mu g/m^3$)	AERMOD ($\mu g/m^3$)
3	3	4.799	4.637	5.427
3	5	5.140		4.757
3	10	5.435		3.368
3	15	4.5017		2.557

Notes: Gatwick (1996) meteorological data used with surface roughness of 0.5 m
Model results presented for central receptor, ie. at the centre of source; see Figure 5.7.

(Source : this study)

5.4.3 Building Dimensions and Street Canyons

The complexity of building wake effects on ground level sources has been reviewed by Kukadia *et al* (1999) in considering future ventilation design in urban buildings. Wind tunnel experiments appear to present the best means describing the highly fluctuating dispersion around buildings. There are two areas of interest: street canyons and closely packed building arrays. The former is the subject of model development elsewhere (see, for example, Buckland and Middleton (1999), and Scaperdas and Colvile (1999) who consider the effect of buildings and street canyons in terms of how representative monitoring data are). However, such cases are site specific and of more interest in detailed traffic management schemes; they are not considered further in this work. Similarly, computational fluid dynamic (CFD) techniques are often used to assess the dispersion around building arrays; with many shortfalls noted, and empirical models are considered useful in predicting dispersal patterns in a broad sense although Croxford and Penn (1998) describe using an array of monitors to provide kerbside resolution. Again, these techniques are site specific. In their review, Kukadia *et al* (1999) cite work in Australia where it was concluded '*urban form does matter*'; allowing urban sprawl was good for dispersal (and hence, air quality) but led to increased commuting (CO_2 emissions). Evidently, the tools developed in this study would not be able to assess this potential interaction.

Accepting that building wake effects cannot be described in detail, the approach taken in this study was conventional, in using a surface roughness value to describe the effect of land use on turbulence.

5.5 BACKGROUND CONTRIBUTIONS

5.5.1 Background NO_x

The emissions inventory and model compiled for this study is limited to within the study area of Kent and Medway. The contribution from sources outside the study area, such as Part A Processes along the Thames Estuary, urban agglomerations and Continental Europe have been implicitly included in assuming a background contribution of 20 $\mu g/m^3$ across the whole study area based on the observed annual means at Lullington

Heath and Stoke. Further discussion of the contribution from Part A Processes is included in Section 4.3.1.

5.5.2 Background PM_{10}

Total PM_{10} concentrations can be considered in terms of three fractions:

- **primary PM_{10}** : as derived from the emissions inventory and models
- **secondary PM_{10}** : comprising principally of sulphate and nitrates
- **other PM_{10}** : arising from sea salt spray, wind erosion of soils, and other sources.

Modelling studies with the West Midlands inventory reported by Sadler (1998) suggests as little as 20% of ambient PM_{10} is estimated with an emissions inventory; with the majority being secondary or from outside the area covered. There has been extensive research in to the source apportionment of PM_{10} in recent years, not least the summary provided by the Airborne Particles Expert Group (1999) from which the above categories are derived and have been used to project PM_{10} concentrations in future years (see Table 5.6 below). The Airborne Particles Expert Group (1999) also estimate primary traffic emissions make up approximately 29% of total observed PM_{10} with 45% coming from secondary sources. Kukkonen *et al* (2001) expand on this, and the work of Harrison *et al* (1997) in developing a model on the assumption that local traffic is responsible for a substantial fraction of urban PM_{10}. A similar method was developed by Stedman *et al* (1998; 2001; see also Stedman, 1998) for national mapping and forecasting of PM_{10} across the UK in reviewing the Air Quality Strategy.

Malcolm *et al* (2000) provide an update to modelling the long-range transport of secondary PM_{10} to the UK; work which contributed to the review of the UK Air Quality Objectives (DEFRA, 2001). The work by Malcolm *et al* (2000) highlights the influence of Continental Europe on ambient sulphate levels in Kent and Medway; 40% compared to an average of 25% contribution observed across other regions in the UK. The Airborne Particles Expert Group (1999) estimates the contribution of sulphate and nitrates to urban levels of PM_{10} to be 6 $\mu g/m^3$ and 4 $\mu g/m^3$ respectively although Malcolm *et al* (2000) comment that the paucity of nitrate data limits the analysis of receptor modelling for this fraction of secondary PM_{10} with Stedman *et al* (2001c) further commenting that the use of nitrate levels as an indicator of secondary PM_{10} is becoming more important with improved sulphur emission controls.

The proportion of PM_{10} defined as 'other' is reported to be typically 20-50% with no expected decline in future years. Stedman *et al* (2001c) undertook a regression analysis to estimate the relative contributions of PM_{10} using NO_x as an indicator of primary combustion with the residual C representing the 'other' fraction of PM_{10}:

$$\textit{measured}\ PM_{10} = A.[\textit{measured}\ NO_x] + B.[\textit{measured}\ \text{sulphate}] + C \qquad ...(9)$$

Using this method, Stedman *et al* (2001c) estimate the annual mean contribution of 'other' PM_{10} observed at London Bloomsbury (1996-1998) was in the range 8-11 $\mu g/m^3$ which, although not implicit from the text, includes a contribution from long range transported primary PM_{10} over Europe, estimated by ApSimon *et al* (2001) to be in the order of 2 $\mu g/m^3$. Given the increased traffic intensity of Central London and the expected greater level of construction activity, the 'other' fraction of PM_{10} in Kent and Medway is assumed to be 5 $\mu g/m^3$.

Table 5.6
Secondary and 'Other' Contributions to PM_{10} ($\mu g/m^3$)

Year	Secondary (sulphate + nitrates)	Background contribution (including 'other')
1990	10	15
1997	10	15
2000	9	14
2005	8	13
2010	7	12
2020	7	12

Note: The results of Europe wide measures to reduce the formation of secondary PM_{10} implies levels will fall by 2010 to between 50 and 70% of 1996 levels. The contribution from 'other' sources is not expected to decline: APEG (1999); Stedman (2001a).

(Source : this study)

5.6 ROADSIDE AIR QUALITY

The method described in Sections 5.2 to 5.4 provides estimates of non-roadside concentrations of NO_x and PM_{10} based on the dispersion modelling of primary emissions and accounting for background contributions. Woodfield (2001a-b) concludes from a review of the results of local authority R&A that most concern lies with exposure to air pollution at the roadside. The use of a line source model, such as the CALINE series or ADMS-Roads, requires emissions data to be maintained as discrete lines rather than

areas; Lindley *et al* (2000) and Briggs *et al* (2000) both describe how the ArcView GIS can maintain data in such a form. However, generating meaningful results using a line source model requires a dense grid of receptors, typically at 5 m, 10 m, 25 m, 50 m, and 100 m from the road centreline and evenly strung at 25-50 m intervals along the road. The generation of such an irregular receptor grid will tend to skew the data and not enable population exposure studies, as described in Section 6.5, to be undertaken.

As part of the work to review the Air Quality Strategy, Stedman *et al* (1999a-b, and later 2001b) describe an empirical model for estimating roadside concentrations of NO_2 where the roadside concentration is estimated as the sum of background concentration and a 'roadside enhancement'. The background concentration is provided by national mapping of air pollution across the UK, also undertaken by Stedman *et al* (1998). The roadside enhancement was estimated from the observed relationship between vehicular emissions (from the NAEI) and kerbside monitoring data along ten roads. In this instance, 'kerbside' included monitoring sites within 5-10 m of the kerbside. Stedman *et al* (1999a-b, 2001c) defined the roadside enhancement as the difference between the observed kerbside level and relevant background level provided by national mapping.

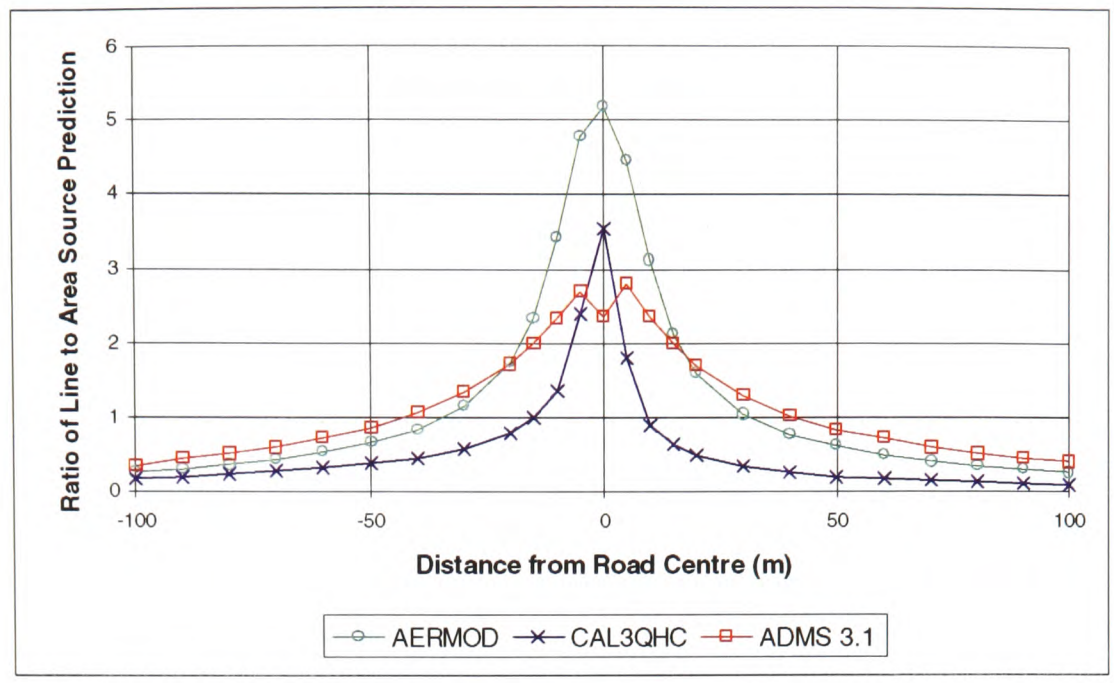
A similar approach was used in this study by using first estimating the non-kerbside concentrations of NO_x and PM_{10}, and then applying an enhancement factor. However, in this study, a systematic approach was developed to derive the enhancement factor, from a comparison of modelling a unit emission from equivalent area and line sources, applied to the vehicular emissions within the grid square being modelled. By using local vehicular emissions data, this technique is expected to have some advantage over the work of Stedman *et al* (1999a-b, 2001c) who recognised the limitations of using NAEI data where the class of road only dictates the assumed speed and hence, vehicular emissions. A further advantage of this approach is that the enhancement factor could be derived for a range of distances from the kerbside. Although this approach has been evaluated with reference to monitoring data within Kent and Medway only (see Section 5.8) it could be applied elsewhere in the UK.

The models ADMS-3, AERMOD and Cal3QHC were used to derive the ratios between a unit emission (1 tonne per annum) from an area source of 250 m by 250 m and an

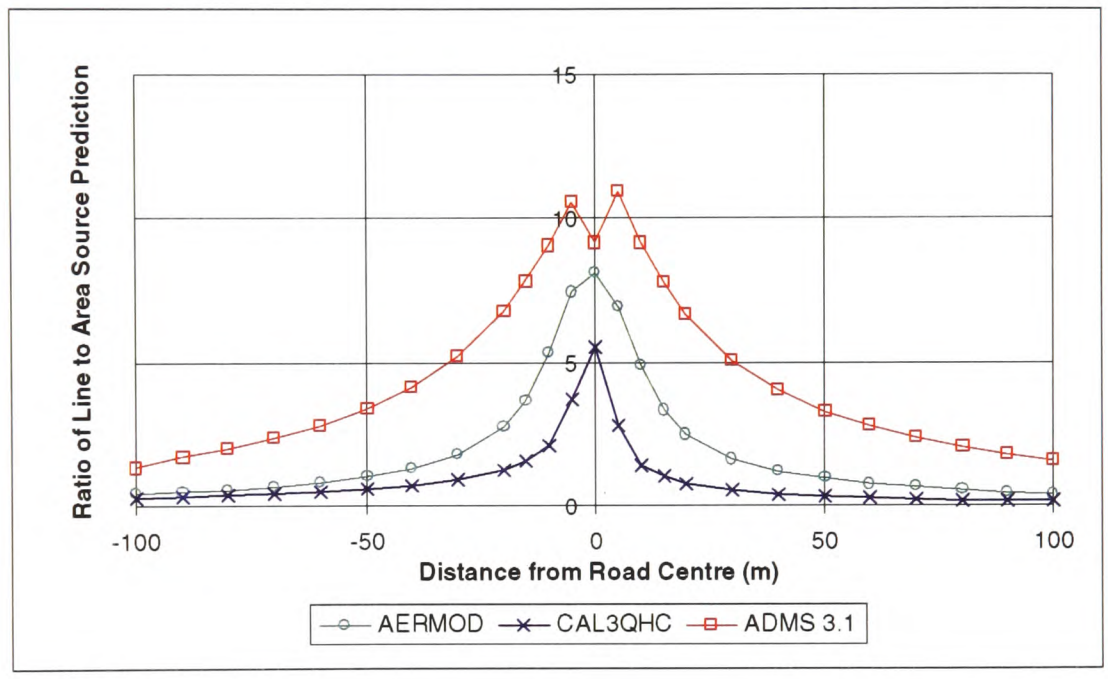
equivalent line source 250 m in length. A series of model runs were undertaken using hourly sequential meteorological data from Gatwick (1996) and Manston (1995, 1996).

The models were first run with area source release heights of 1 m, 3 m and 5 m (Figure 5.7). The model results are very sensitive to source release height. With a release height of 1 m the ratio of area to line source is between 2.5 and 5, with a release height of 5 m this ratio increases to between 9 and 23. For the release height of 3 m, established in Section 5.3 as being most representative of ground level sources, the ratio of line to area sources is in the range 4 to 11 at 10 m from the road centreline. Variations in line source orientation (north/south, east/west, north-east/south-west and north-west/south-east; Figure 5.8) extends the area to line source ratio range to 6.5 – 13 at 10 m from the road centreline. In comparison, the ratio of area to line sources is not as sensitive to the width of the line source (Figure 5.9). Running the models with an area source release height of 3 m and line source widths of 8 – 12 m generates a ratio between 0.8 and 1.25. In conclusion, assuming an area source release height of 3 m, the ratio at 10 m from the road centreline lies in the range 4 – 13. In the process of evaluating the model (see Section 5.7) a roadside enhancement factor of 8.5 was adopted.

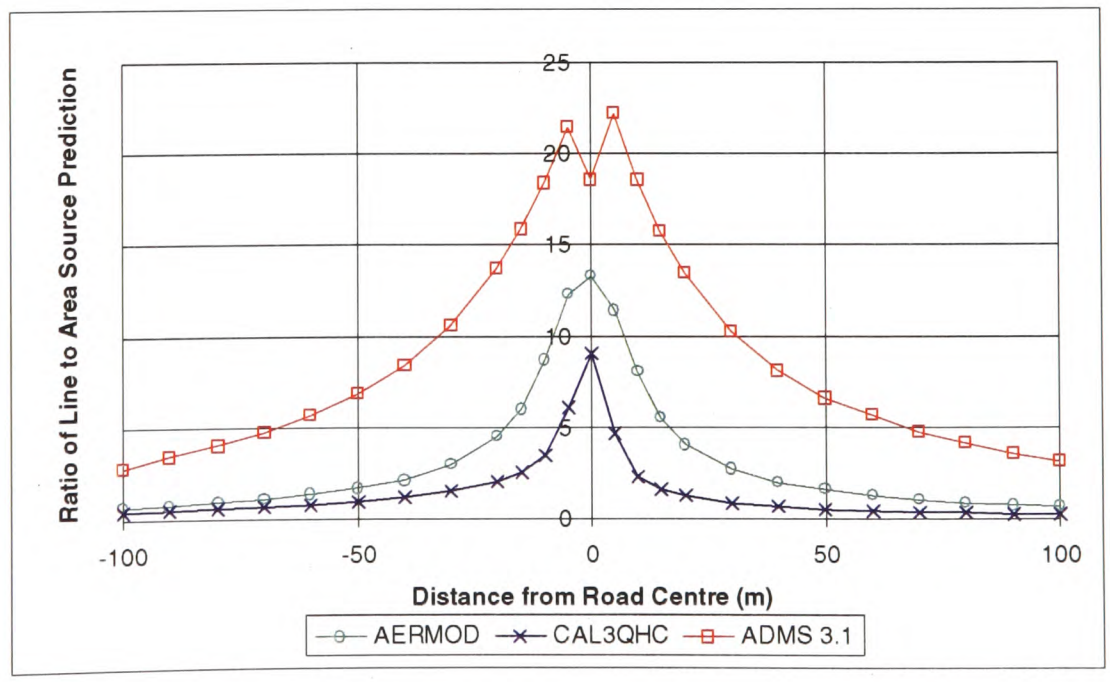
The use of this technique has the disadvantage of providing roadside concentrations at a fixed point from the road centreline; Stedman *et al* (1999a-b, 2001c) suggest their method is applicable at approximately 6 m, this study uses 10 m. However, the method developed in this study could be used to derive enhancement factors for a range of distances if required although analysis by Laxen *et al* (2001, 2002) suggests elevated NO_x and NO_2 concentrations are limited to within a few (<20) metres of the road centreline. This technique is not considered valid for very localised studies. Berkowicz *et al* (1996) highlight the very site specific nature of air quality monitoring stations and the difficulty in determining how representative they are of a wider area. This implies the roadside NO_x to NO_2 contribution and the model evaluation will be subject to uncertainty. Berkowicz *et al* (1996) consider street configuration and local meteorology to be the key influences in very localised variations. This is the subject of research recently commenced in Central London (Colvile, 2002, *personal communication*). The emissions inventory and models developed by this research are not considered applicable at the very localised scale.



(a) Area Source Release Height of 1 m



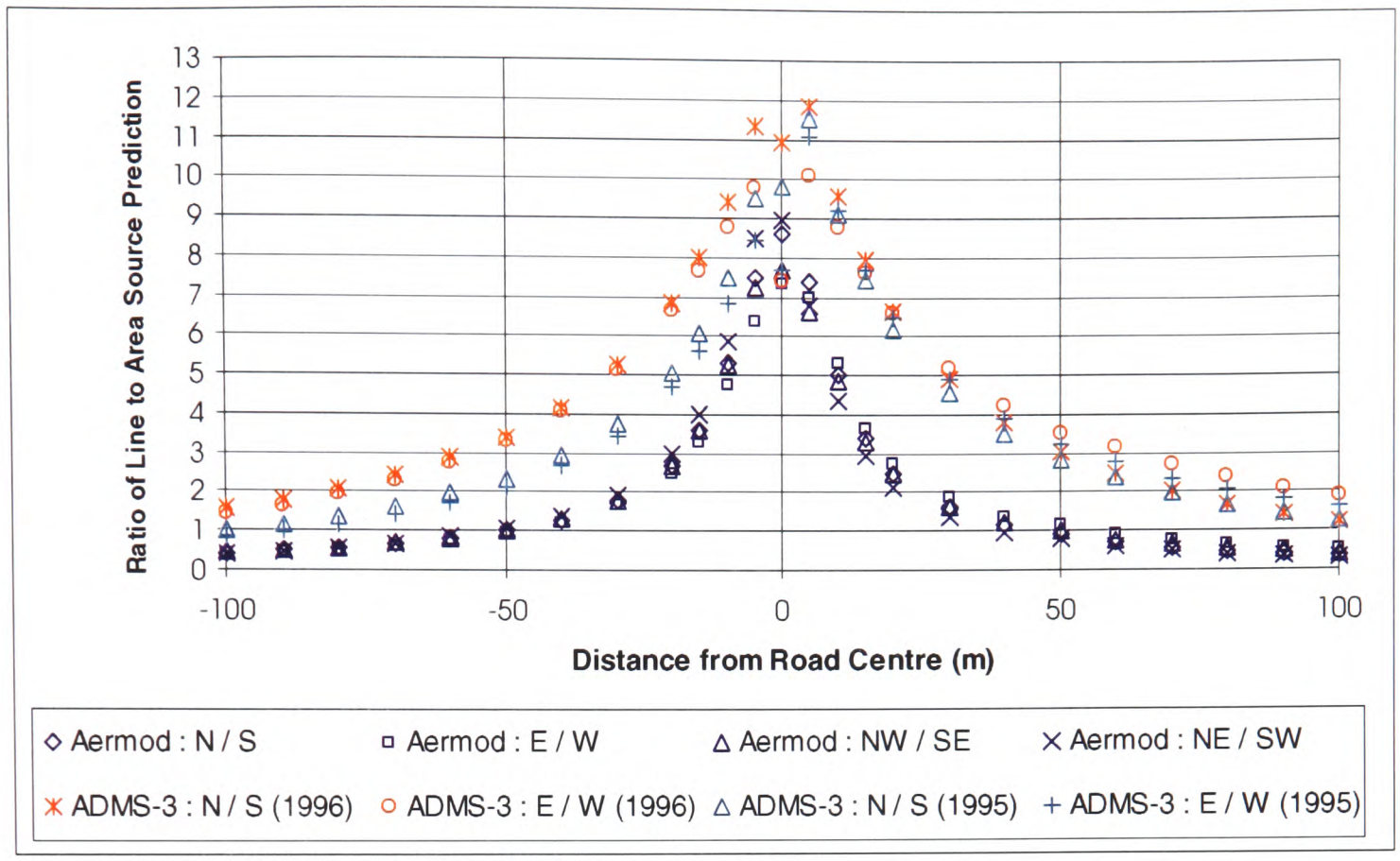
(b) Area Source Release Height of 3 m



(c) Area Source Release Height of 5 m

FIGURE 5.7 : Ratios of Modelled Area and Line Sources by Release Height

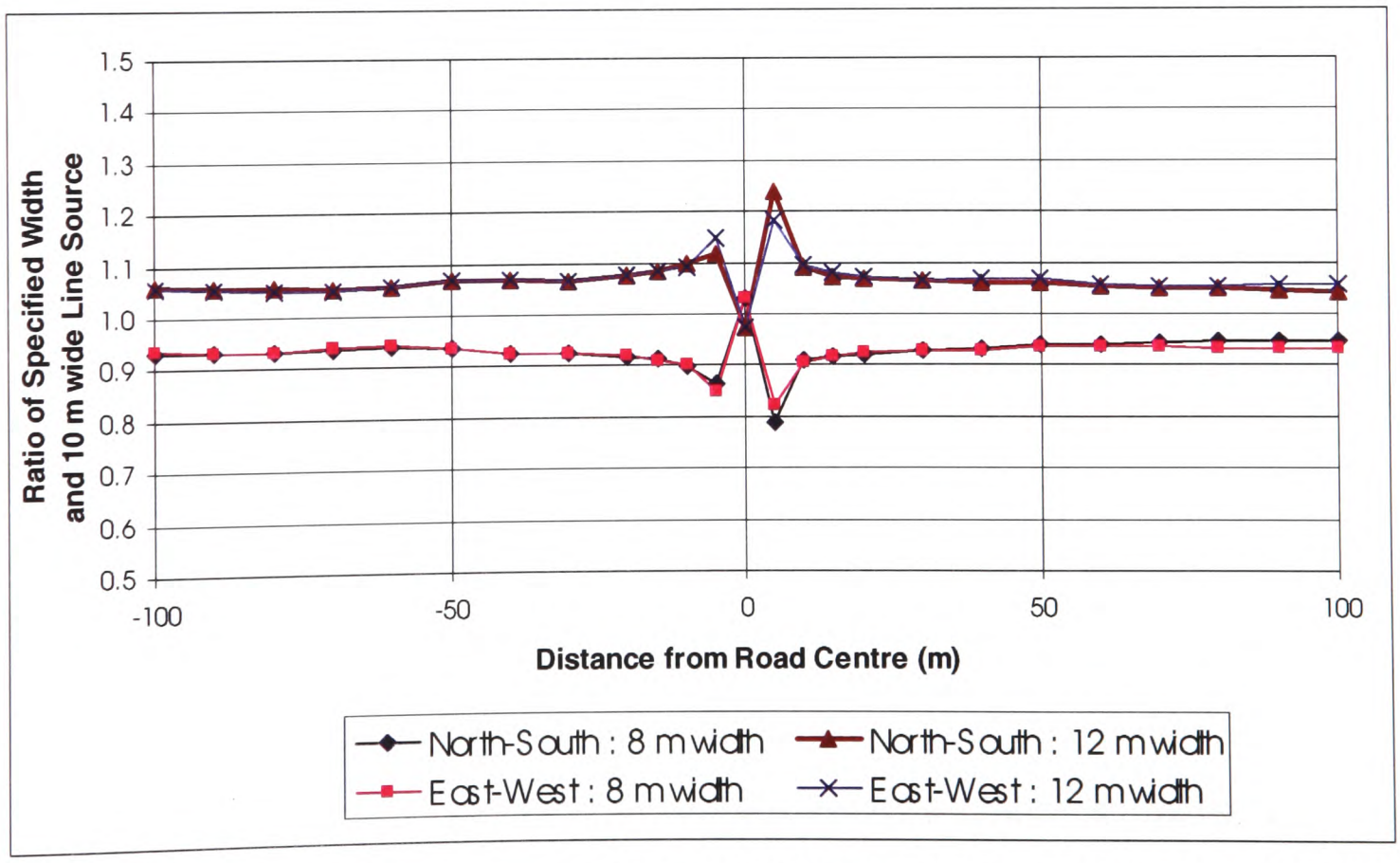
(Source : this study)



Results presented were generated using Aermod and ADMS-3 with release height of 3 m and line source width of 10 m for Aermod.

FIGURE 5.8 : Sensitivity of Line Source Model to Source Width (1)

(Source : this study)



Results presented were generated using Cal3QHC. Similar analysis using AERMOD reveals the model is not sensitive to this parameter (the ratios all being within 4%). ADMS-3 does not allow specification of line source width.

FIGURE 5.9 : Sensitivity of Line Source Model to Source Width (2)

(Source : this study)

5.7 CONVERSION OF NITROGEN OXIDES TO NITROGEN DIOXIDE

The relationship between NO_x, O_3 and hydrocarbons in urban and rural environments has been the subject of much research over the last decade and before. Sillman (1999) provides a convenient review of conclusions drawn from photochemical models and field measurements with a focus on the uncertainty of predicting the response of O_3 formation to future reductions in NO_x and volatile organic compounds (VOC) emissions. Sillman (1999) highlights the difference between (United States) policy makers protecting urban and rural environments; the former generally being (relatively) NO_x-saturated and hence, O_3 generation is more sensitive to VOC reductions. Sillman (1999) states '*in general, NO_x emissions within an urban area determine the total amount of O_3 formed after the air moves downwind and chemistry has run to completion; ...a polluted air mass is most likely to have VOC-sensitive chemistry when it is close to its emissions sources*'. The term *close* is relative; in densely populated parts of Northern Europe, rural O_3 also tends to be more VOC-sensitive (Sillman, 1999; see also Bower *et al*, 1994).

Hewitt (2001) cites empirical studies of the nitrogen chemistry of power station plumes, including those of Janssen *et al* (1980)[24], suggesting a removal rate of NO_x in the order of up to 30% per hour in sunny conditions via oxidation (to NO_2 in the presence of O_3 and hydroxyl, hydroperoxy and alkylperoxy radicals) and dry deposition (via HNO_3); see also Imhoff *et al* (2001). Hewitt also reports model calculations suggesting a removal rate in the order of 80% in three hours; comparable to empirical studies. The effect of urban (more polluted) air is highlighted; driving the oxidation reactions with increased VOCs present.

Extending earlier work by Derwent and Middleton (1996; see also Derwent *et al*, 1995) who derived an empirical relationship between NO_x and NO_2 using observations from Exhibition Road in Central London, Dixon *et al* (2001) developed a series of similarly derived relationships for a number of urban background locations across the UK. By assuming an O_3-saturated atmosphere, these were then used to demonstrate that an approximate 50% reduction in NO_x emissions (relative to the period 1991-1997) will be

[24] The empirical work of Janssen *et al* (1988) is commonly used in power station plume modelling but is also used by Karppinen *et al* (2000) in estimating the conversion of NO_x to NO_2 in a model of Helsinki.

required to meet the short term UK Air Quality Objective. Dixon *et al* (2001) exclude the heavily trafficked kerbside data from this work as the atmosphere is NO_x-saturated; further build up of NO_2 being the result of direct emissions. Owen *et al* (2000) compared using the Derwent-Middleton relationship with using the chemistry module within ADMS-Urban to model London, concluding that the empirical relationship performed well and should not be ignored. However, short term fluctuations are far more complex involving, for example, nitrous acid (HONO) emitted directly or converted from NO_2 on surfaces such as road tunnels (Kurtenbach *et al*, 2001). Modelling of short term fluctuations in NO_2 is expected therefore to remain uncertain.

For reasons presented in Chapter 3, the key NO_2 criterion of concern is the annual mean objective. Stedman *et al* (1998, 2001) summarise the atmospheric chemistry stating that ambient levels of NO_2 are directly related to NO_x, assuming O_3 concentrations remain relatively unchanged. Stedman *et al* (1998, 2001b) cite Bower *et al* (1994) who suggest the trimolecular reaction of NO_2 with O_2 may be one of the dominant reactions during rare winter smog episode days. Stedman *et al* (1998, 2001b) conclude that, as the rate of reaction depends on the square of the NO concentration, further reductions in NO_x emissions will reduce the importance of this pathway and hence, the annual mean objective is of prime concern. This theory is borne out by the analysis of monitoring data by Ireland (1998) presented in Section 3.3.3. Karppinen (2000a-b) states the NO_2/NO_x ratio is approximately 10% in traffic emissions with most of the NO oxidised within a few minutes. The NO_2/NO_x ratio for stationary sources is less and the NO oxidation period in the order of 10-20 minutes. The annual mean NO_2/NO_x ratio at locations remote from NO_x sources is in the order of 90%, and approximately 50% in the vicinity of busy roads. Stedman *et al* (2001c) report a rural annual mean NO_2/NO_x ratio of 0.833 for the UK, based on observations made at 32 sites during 1990 to 1998.

Application of detailed atmospheric chemistry models, principally developed to estimate O_3 generation, is not considered efficient favouring a more empirical approach. In proposing a number of techniques for rapid assessment of local air quality, Ireland (1998) derived an empirical NO_x/NO_2 relationship based on annual mean observations from all monitoring stations in the Advanced Urban Network (now the Enhanced Urban and Rural Network) for all years 1987-1997. Stedman *et al* (1998) provide a similar relationship based on 1996 data. Fisher *et al* (1999) compared the empirical relationship

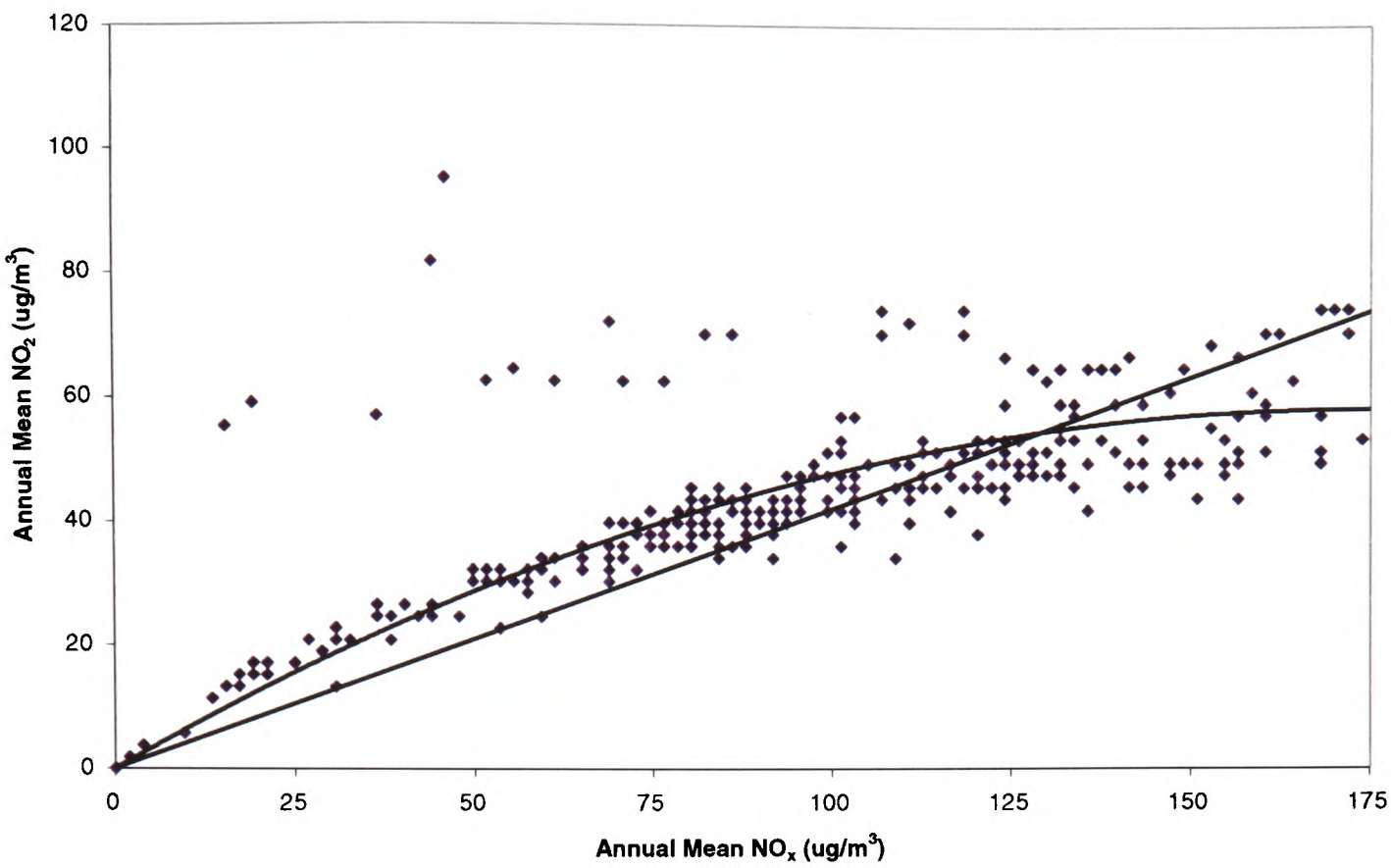
detailed in Volume 11 of the Design Manual for Roads and Bridges (DEFRA, 2002c, *updated*) with those compiled by ENTEC and Ireland (1998) concluding the latter appeared to be the most pessimistic. Stedman *et al* (1998) note that any such empirical relationship will not remain unchanged as future NO_x emissions reduce and the balance of atmospheric chemistry changes, as noted by Sillman (1999) Dixon *et al* (2000) Carslaw *et al* (2000) and DETR (2000b). This may explain the different NO_x/NO_2 relationship observed by Ireland (1998) who used several years of data (see Figure 5.10). Carslaw *et al* (2000) considered this approach for London but concluded that, with only 13 monitoring sites in the study area, a statistically robust relationship could not be derived. Although not implicit in the discussions of Carslaw *et al* (2000) this suggests a further issue in the variation of O_3 across the UK and the effect on NO_x/NO_2 ratios observed at different geographical locations as demonstrated by Dixon *et al* (2001). An example of this is the consistently higher NO_x/NO_2 ratios observed in central and inner London compared to outer London.

From analysis of monitoring data, Stedman *et al* (2001c) provide a relationship between roadside increments of NO_x and NO_2:

$$[NO_2]_{\text{roadside increment}} = 0.1615 \,.\, [NO_x]_{\text{roadside increment}} \qquad ...(10)$$

(all concentrations in $\mu g/m^3$, $r^2 = 0.81$ based on 23 data points).

In this research, the model was evaluated assuming NO_2:NO_x ratios of 0.75 for background locations and 0.15 for kerbside locations, respectively. These ratios are similar to those observed at rural and kerbside locations in Kent and Medway, and by Stedman *et al* (2001c) although the final ratios selected were based on fitting the modelled data with observations (see Figures 5.12 and 5.13, and associated text). For the purposes of model evaluation, intermediate locations were modelled assuming the same NO_2:NO_x ratio for background sites and half the roadside enhancement factor (see Section 5.5). This 'smoothing' approach is similar to that proposed by Laxen and Wilson (2002) although in this case these ratios appear less conservative compared to empirical data (Figure 5.10) highlighting the need for model evaluation with reference to local monitoring data.



A total or 338 paired (annual mean NO_x and NO_2) observations from kerbside, roadside, urban centre, urban background and rural sites across the UK were used to construct the above scatter plot.

The equation for the linear regression through zero is:
$[NO_2] = 0.4238 \,.\, [NO_x]$ $(r^2 = 0.6052)$

The equation for the curved regression is:
$[NO_2] = - 0.002 \,.\, [NO_x]^2 + 0.6786 \,.\, [NO_x]$, for $[NO_x] < 175$ µg/m^3 $(r^2 = 0.741)$

Using the quadratic regression, the NO_2/NO_x ratio at background locations (assuming $[NO_x] = 20$ $\mu g/m^3$) would be in the order of 0.6. Similarly, the NO_2/NO_x ratio at kerbside locations would be in the order of 0.3, assuming $[NO_x] = 175$ $\mu g/m^3$.

Figure 5.10 : Observed Annual Mean Nitrogen Oxides : AUN : 1979 to 1996

(Source : this study, compiled from data reported in the National Air Quality Information Archive)

5.8 MODEL EVALUATION

5.8.1 Approach

Model evaluation is the quantification of the performance of the model in real cases with real data (Zannetti, 1990). This allows identification of two sources of error (DETR, 2000k):

- ***systematic error*** (bias) which can be quantified and used to calibrate the model
- ***random error*** (uncertainty) may also occur even after the bias has been accounted for, expressed as a range in which the model answer will lie.

DETR (2000k) suggests the systematic error should be corrected and defines the random error as the square root of the mean sum of the squared differences between the predicted and measured concentrations:

$$RMS = \sqrt{\frac{\sum (C_o - C_p)^2}{n}} \qquad ...(11)$$

where : C_o = *observed value*
C_p = *predicted value*
N = *number of values*

Having corrected for bias, DETR (2000k) suggests, as a rule of thumb, '*if the prediction of an annual mean concentration lies within ± 50% of the measurement, a user would not consider that the model has behaved badly*'.

Between them Zannetti (1990) and Hanna *et al* (1999) cite eight methods for model evaluation:

- bias evaluation
- error analysis
- variance of the differences
- time correlation
- space correlation
- peak analysis
- distribution functions
- regression analysis.

The evaluation of bias, analysis of error and variance on the differences can be performed using a number of statistical tests. There are two model evaluation methodologies currently used based on a combination of statistical measures, including some of those described above; the Model Validation Toolkit widely distributed by Olesen (1997, 1999) and the emerging American Society for Testing and Materials (ASTM) Standard Practice (Irwin, 1997). McHugh *et al* (1999) compared these evaluation methods as applied to three models and three field data sets. Although the work was based around short term (1-hour) values, the conclusion was made that no single statistic should be used to signify confidence in model performance. A review of model evaluation studies (Table 5.11, presented at the end of this Chapter) suggests the following should be included as best practice:

- tabulation of observed and predicted data
- summary statistics of the number of data pairs, minimum, maximum, mean and standard deviation of observed and predicted data
- graphical presentation of data as scatter plots and regression analysis with 1:1 best fit line and factor of two boundaries
- reporting of the *bias*, *fractional bias* and *normalised mean deviation* as measures of systematic error (bias)
- reporting of the *index of agreement*, *normalised mean square error*, *Pearson's correlation coefficient*, *fraction of data within a factor of two* and *Root Mean Square Difference in Concentrations* as measures of random error (uncertainty).

As all emissions and dispersion estimates are as annual means, an assessment of time correlation is not appropriate for this study. The correlation of observed versus modelled results in space has been assessed qualitatively along with an analysis of the magnitude and location of peak concentrations in Section 5.7.8

The *Bias* is calculated as follows:

$$Bias = C_o - C_p \qquad \ldots(12)$$

where : C_o = *observed value*
C_p = *predicted value*

This parameter is reported in the units of the study in question. Comparison with published data is therefore meaningless.

The *Fractional Bias (FB)* is a measure of the agreement of the mean concentrations. FB is non-linear bounded by ± 2 where a positive value indicates the model is over predicting and a negative value indicates an under prediction:

$$FB = \frac{\overline{C}_p - \overline{C}_o}{0.5.(\overline{C}_p + \overline{C}_o)} \qquad \ldots(13)$$

where : $\dot{C}_o$ = *mean of observed values*
$\dot{C}_p$ = *mean of predicted values*
FB = 1.99 = no agreement
FB = 1.98 = model predicts within a factor of 200
FB = 1.81 = model predicts within a factor of 20
FB = 1.00 = model predicts within a factor of 3
FB = 0.67 = model predicts within a factor of 2

Karppinen *et al* (2000a-b) Kukkonen *et al* (2000, 2001a-c) and Kousa *et al* (2001) use *FB* to evaluate model performance by comparing hourly observed and modelled NO_2 and PM_{10} concentrations at four sites in Helsinki, NO_x and NO_2 at a separate site near to a major road in southern Finland (at three heights; the lowest height of 3.5 m is reported here) and NO_x and NO_2 at a street canyon site also in Helsinki. Results are reported in the range –0.092 to +0.118 for NO_2 and –0.04 to +0.09 (PM_{10}) for the four sites in Helsinki, +0.061 for NO_x and –0.192 for NO_2 at the major road and +0.045 (NO_x) and +0.22 (NO_2) in the street canyon. Kousa *et al* (2001, 2002) extends the Helsinki study, reporting results for seven sites for two years; *FB* in the range –0.289 to +0.255 (NO_2). Kukkonen *et al* (2001b) suggest an absolute value of FB less than approximately 0.1 could be caused by the inaccuracies of the measurement system for concentration and meteorological data[25]. An absolute value of FB greater than 0.1 - 0.2, together with a relatively high correlation coefficient (see below for definition) is considered to indicate a systematic error. In comparing two sets of statistical tests for model performance using field data, McHugh *et al* (1999) reports results for three models (ISCST3, AERMOD and ADMS3) compared with hourly mean observations from three field data sets. McHugh *et al* (1999) report *FB* in the range –0.442 to +0.855.

[25] This appears consistent with Carslaw *et al* (2000) who conclude, from national and London wide monitoring data, the uncertainty associated with annual mean NO_2 observations is around 10%, based on automatic measurements.

The *Normalised Mean Deviation (NMD)* is used as a measure of uncertainty:

$$NMD = \frac{(C_o - C_p)}{C_p} \qquad ...(14)$$

where : C_o = *observed value*
C_p = *predicted value*
NMD =0 = perfect agreement

Häggkvist (1997) compared the results of a model developed for Stockholm with observed concentrations at three monitoring sites, reporting *NMD* in the range –0.32 to +0.58.

The *Index of Agreement* (IA) is a measure of the correlation of the predicted and observed concentrations, in this case, providing an indication of the spatial accuracy:

$$IA = 1 - \frac{\overline{(C_p - C_o)^2}}{\overline{\left(|C_p - \dot{C}_o| + |C_o - \dot{C}_o|\right)^2}} \qquad ...(15)$$

where : C_o = *observed value*
C_p = *predicted value*
$\dot{C}_o$ = *mean of observed values*
$\dot{C}_p$ = *mean of predicted values*
IA = 1 = perfect agreement

Kukkonen *et al* (2000, 2001a, 2001c) report *IA* in the ranges +0.65 to +0.82 (NO_2) and +0.86 to +0.96 (PM_{10}) for the Helsinki sites and 0.849 (NO_x) and 0.735 (NO_2) at the major roadside site. Kousa *et al* (2001) report *IA* in the range +0.65 to +0.82 (NO_2). Kukkonen *et al* (2000) suggest 0.4 represents a totally random predicted time series having the same range of variables as the observed series.

The *Normalised Mean Square Error* (NMSE) is a measure of the correlation of the predicted and observed concentrations, in this case, providing an indication of the spatial accuracy:

$$NMSE = \frac{\overline{(C_p - C_o)^2}}{(\overline{C_p})^2} \qquad ...(16)$$

where : C_o = *observed value*
C_p = *predicted value*
NMSE =1 = perfect agreement

Kukkonen *et al* (2000) report *NMSE* in the range 0.26 to 0.45 (NO_2) for Helsinki and 0.215 and 0.209 (NO_x, NO_2 respectively) at the major roadside site. Kousa *et al* (2001) report *NMSE* in the range 0.21 to 1.19 (NO_2). McHugh *et al* (1999) report *NMSE* in the range 0.57 to 3.83. Häggkvist (1997) reported *NMSE* in the range 0.34 to 1.10.

Correlation Coefficient (R^2) is a measure of the correlation of the predicted and observed concentrations, in this case, providing an indication of the spatial accuracy:

$$R^2 = \left[\frac{(C_o - \overline{C_o})(C_p - \overline{C_p})}{\sigma_o \sigma_p} \right] \qquad \ldots(17)$$

where : C_o = *observed value*
C_p = *predicted value*
$\dot{C}_o$ = *mean of observed values*
$\dot{C}_p$ = *mean of predicted values*
σ_o = *standard deviation of observed values*
σ_p = *standard deviation of predicted values*
R^2 = *1 = perfect agreement*

Kukkonen *et al* (2000, 2001a, 2001b) report R^2 in the ranges 0.50 to 0.65 (NO_2) and 0.79 to 0.95 (PM_{10}) for the Helsinki sites, 0.794 (NO_x) and 0.80 (NO_2) at the major roadside site, and 0.77 (NO_x) and 0.83 (NO_2) in the street canyon. Kousa *et al* (2001) report r^2 in the range 0.39 to 0.68 (NO_2). McHugh *et al* (1999) report R^2 in the range 0.162 to 0.749. Häggkvist (1997) reports r^2 in the range 0.37 to 0.64. In a comparison of CALINE4, AAQuiRE and ADMS-Urban, Courthold and Whitewall (1998) report a R^2 in the range 0.678 to 0.896 for annual mean NO_2 (10 observations).

The fraction of data for which $0.5 \leq (P/O) \leq 2$ (*Fa2*) is a measure of data scatter:

$$Fa2 = \frac{1}{2} \leq \frac{C_p}{C_o} \leq 2 \qquad \ldots(18)$$

where : C_o = *observed value*
C_p = *predicted value*

Kukkonen *et al* (2001) report *Fa2* of 0.937 (NO_x) and 0.667 (NO_2) at the major roadside site in southern Finland. McHugh *et al* (1999) report *Fa2* in the range 0.256 to 0.669. Hanna *et al* (1999) compared ISCST3, AERMOD and ADMS3 using six field datasets, reporting *Fa2* in the range 0.064 to 0.8. This statistic is more relevant to hourly data.

The emissions inventory has been subject to Monte Carlo analysis. Bergin and Milford (2000) describe further statistical techniques for evaluating the performance of a photochemical model. The basic case is to generate model results using Monte Carlo analysis and compare with observed data. Bergin and Milford (2000) describe the use of Bayesian Monte Carlo analysis to refine uncertainty estimates by using a continuous likelihood function to weight the results of individual Monte Carlo simulations. Although such techniques may prove useful with large data sets with thousands of observations, the limited number of observations in this study (less than one thousand) suggests such advanced statistical techniques are not appropriate.

5.8.2 Evaluation Methodology

The observed annual mean concentrations of NO_2, NO_x and PM_{10} for the years 1997 to 2000 were compiled from automatic monitoring stations (NO_2, NO_x and PM_{10}) and diffusion tubes (NO_2 only) within the Kent Air Quality Monitoring Network (King's College London, 2001, South East Institute for Public Health, 1998, 1999, 2000). Following rejection of unrepresentative monitoring data (see discussion below) a total of 157 sites across Kent and Medway (39 background, 34 intermediate and 84 kerbside) were used to evaluate the model for NO_2. A total of 15 sites across Kent and Medway were used to evaluate the model for NO_x (8 background, 1 intermediate and 6 kerbside) and 10 sites for PM_{10} (5 background, 1 intermediate and 4 kerbside). The results are presented In Figures 5.12 to 5.14 as scatterplots (observed versus modelled) with evaluation statistics summarised in Table 5.8.

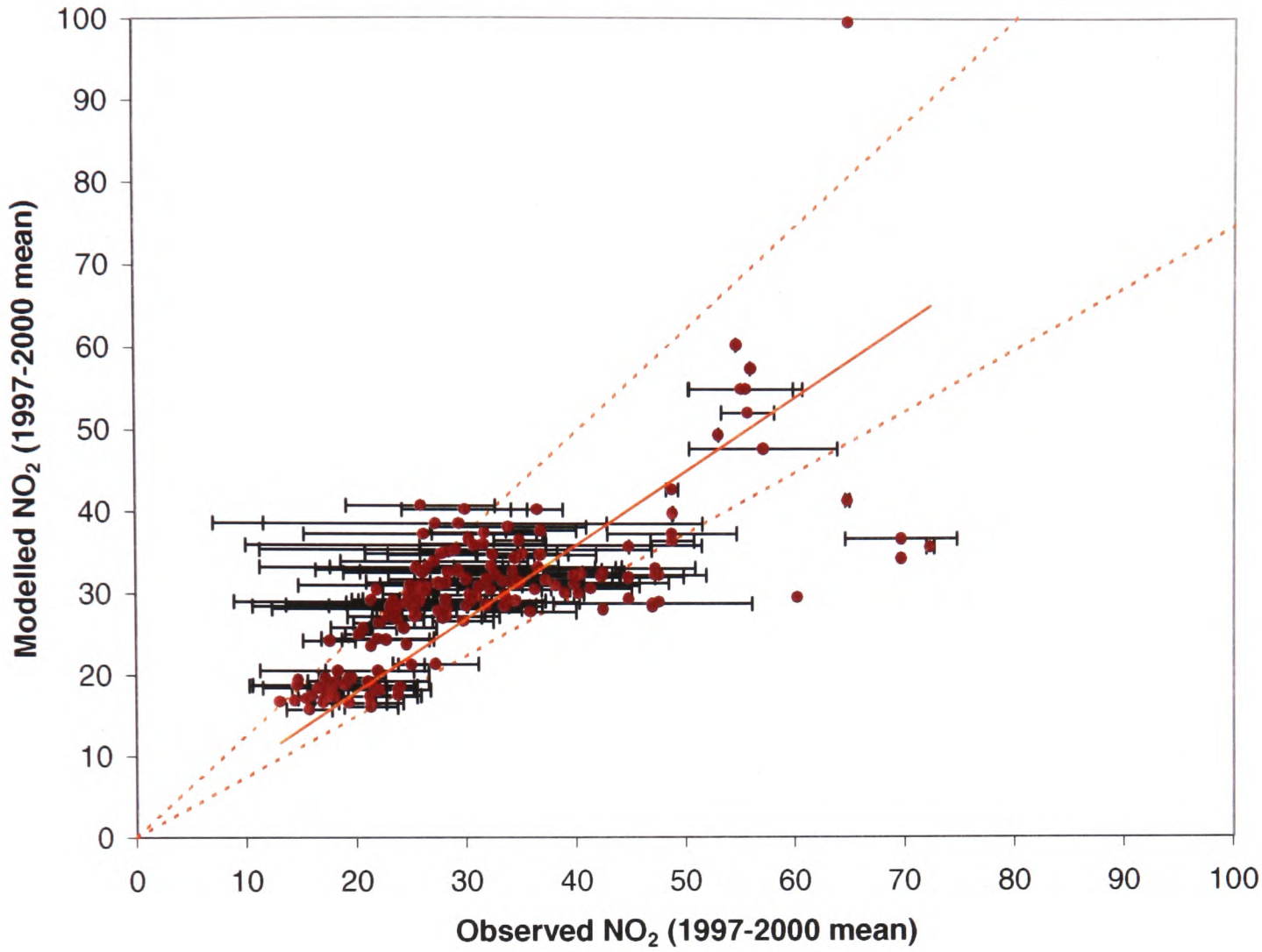
The model was calibrated by comparing observed and modelled data for NO_2 with adjustments made to the roadside enhancement factor (see Section 5.5) and the NO_x : NO_2 ratio for background, intermediate and kerbside locations (see Section 5.6). No further calibration was carried out for NO_x or PM_{10}.

Further evaluation was undertaken by comparing the results with national mapping statistics published by Stedman *et al* (2001a) in terms of correlation (r^2) and reported population weighted background NO_2 and PM_{10} concentrations for countries within the UK (Tables 5.9 and 5.10). Finally, maps of annual mean concentrations of NO_2 and PM_{10} for 2005 were generated and locations where the model indicates where the

relevant Air Quality Standard may be breached were compared to designated Air Quality Management Areas.

5.8.3 Inter Year Variability

Significant inter-year variability was observed in monitoring data with subsequent variation in comparing monitoring and modelling data. This is illustrated in Figure 5.11, a correlation plot of modelled versus observed annual mean concentrations of NO_2, with error bars included to represent ±1 standard deviation from the four year observed mean. The range of actual minimum and maximum values (annual mean for a single year) were -57% and +214% of the four year mean. For the purposes of model evaluation, four year means were derived for both monitoring and modelling data (1997 to 2000) to smooth this variability.



Regression Plot of observed and modelled annual mean NO_2 concentrations 1997 – 2000 with error bars representing ± 1 standard deviation from the four year observed mean (all concentrations in $\mu g/m^3$). Points without error bars represent sites with insufficient data to calculate a standard deviation.

FIGURE 5.11. : Illustration of Inter-Year Variability

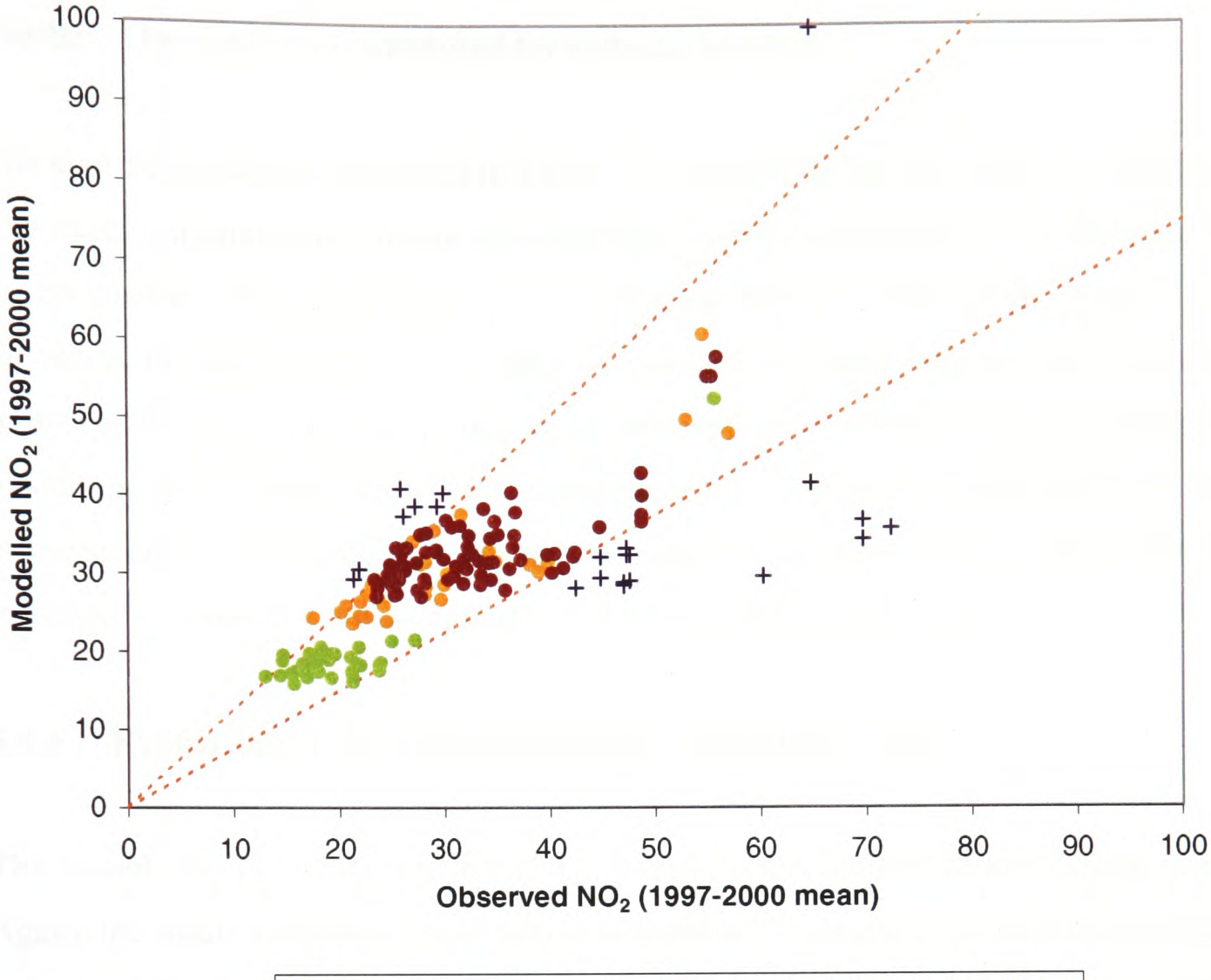
(Source : this study)

5.8.4 Evaluation of the Kent and Medway Model for NO_2

Figure 5.12 includes the linear regression of modelled versus observed annual mean concentrations of NO_2 in Kent and Medway The model appears to represent near unity ±30%; well within the range of inter-year variability in monitoring data and within the range accepted by DEFRA (2000k) for models used for R&A. The results are also disaggregated by class, with intermediate and kerbside sites representing locations with higher observed and modelled concentrations with significant overlap. Some 20 outliers were identified for kerbside locations. Four of the sites included only one year of monitoring data and hence, may be subject to uncertainty due to inter-year variation. Inspection of the emissions inventory suggested some local roads may not be included in

the traffic models and surveys undertaken by KCC Highways and Transportation leading to model results being lower than observed values.

The statistical analysis presented in Table 5.7 suggests the model performs well for NO_2. The mean, minimum, maximum and standard deviation of modelled and observed values are all similar. The measures of *bias* all confirm the analysis presented in Figure 5.12; that the model is slightly conservative. The absolute value of *fractional bias* is less than 0.1. With reference to Figure 5.11, this result corroborates the suggestion by Kukkonen *et al* (2001b) this could be caused by the inaccuracies of the measurement system for concentration and meteorological data. The measures of uncertainty are all well within the results of published model valuation studies. The *normalised mean deviation* of - .01 is within the range reported by Häggkvist (1997). Similarly, the *Index of Agreement* of 0.93 and *Normalised Mean Square Error* of 0.2 are slightly better than reported by Kukkonen *et al* (2000, 2001a, 2001c) Kousa *et al* (2001) McHugh *et al* (1999) and Häggkvist (1997). The *Correlation Coefficient* (R^2) of 0.7012 is better than reported by Kukkonen *et al* (2000, 2001a, 2001b) Kousa *et al* (2001) McHugh *et al* (1999) Häggkvist (1997) and in the range reported by Courthold and Whitewall (1998).



Regression plot of observed and modelled annual mean NO_2 concentrations (1997 – 2000 means, all concentrations in $\mu g/m^3$) with the dashed red lines representing ±25%.. A total of 157 data pairs (excluding outliers) yields a correlation (R^2) of 0.7012. From the linear regression, the model under predicts by 1.5% (y = 0.9848x) within an overall range of uncertainty of ±30%.. See main text for discussion on outliers.

FIGURE 5.12. : Kent and Medway Model Evaluation for NO_2

(Source : this study)

5.8.5 Evaluation of the Kent and Medway Model for NO_x

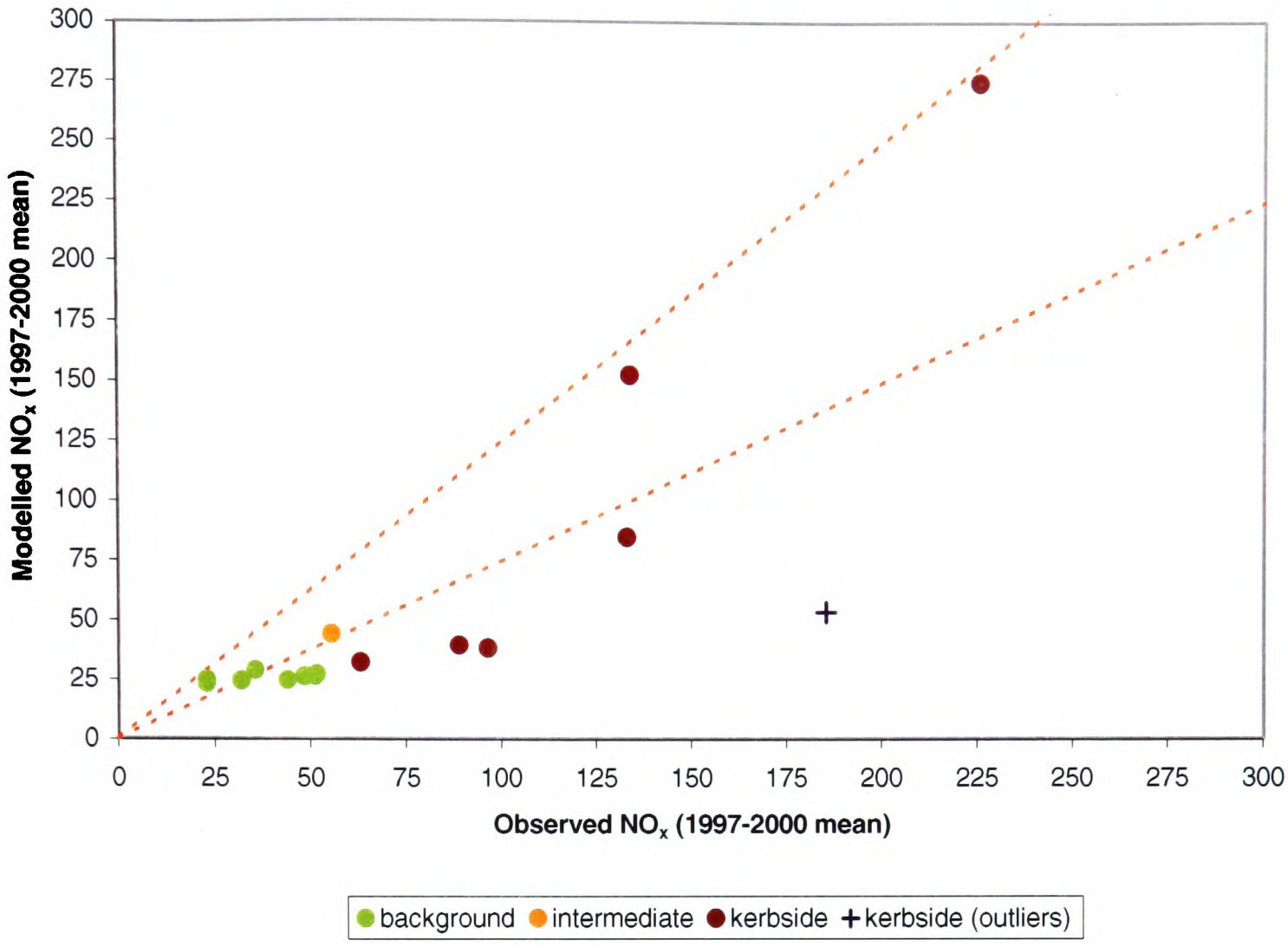
Figure 5.13 includes the linear regression of modelled versus observed annual mean concentrations of NO_x. The model appears to represent near unity ±20%; well within the range of inter-year variability in monitoring data and within the range accepted by DEFRA (2000k) for models used for R&A. However, the distribution of results, with a single high value, suggests the range of error is expected to be greater for lower NO_x concentrations. The results are disaggregated by class, with intermediate and kerbside

sites representing locations with higher observed and modelled concentrations with no overlap. One outlier was identified for kerbside locations.

The statistical analysis presented in Table 5.7 suggests the model performs well for NO_x. The mean, minimum, maximum and standard deviation of modelled and observed values are all similar. The measures of *bias* supports the analysis presented in Figure 5.14; that the model is slightly conservative although inspection of data suggests this interpretation is skewed by the single high value. Over the main range of observed concentrations, the model appears to underpredict NO_x concentrations. Although further statistical analysis is considered limited, due to the number and distribution of data pairs, the results presented in Table 5.7 are comparable with other published studies.

5.8.6 Evaluation of the Kent and Medway Model for PM_{10}

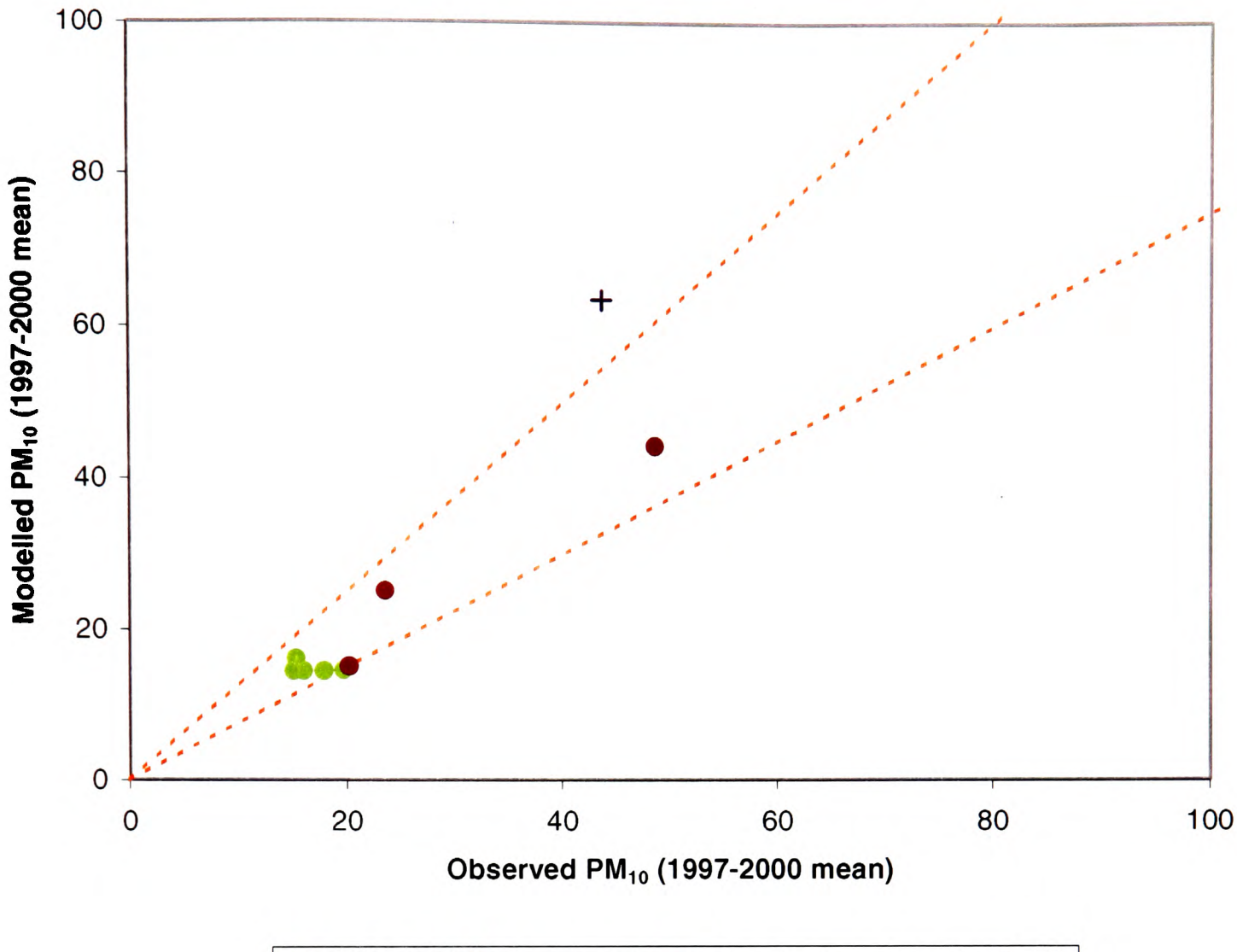
The model also performs well for PM_{10}, based on the limited observed data available. Again, the mean, minimum, maximum and standard deviation of modelled and observed values are all similar. The model appears to slightly underpredict (bias = 2.59) although the spread of data (see Figure 5.15) precludes any firm conclusions. There is one outlier. As for NO_2 and NO_x, the statistical results presented in Table 5.7 are comparable with other published studies.



Regression plot of observed and modelled annual mean NO_x concentrations (1997 – 2000 means) with the dashed red lines representing ±25%.. A total of 15 data pairs yields a correlation (R^2) of 0.8005. From the linear regression, the model represents near unity (y = 0.918) within an overall range of uncertainty of ±20%.. See main text for discussion on outliers.

FIGURE 5.13. : Kent and Medway Model Evaluation for NO_x

(Source : this study)



Regression plot of observed and modelled annual mean PM_{10} concentrations (1997 – 2000 means) with the dashed red lines representing ±25%. A total of 10 data pairs yields a correlation (R^2) of 0.9401. From the linear regression, the model represents near unity (y = 0.9244x) within an overall range of uncertainty of ±6%..

FIGURE 5.14. : Kent and Medway Model Evaluation for PM_{10}

(Source : this study)

Table 5.7
Statistical Analysis of Observed (O) and Predicted (P)
Annual Mean Concentrations in Kent and Medway

Statistic	NO_2	NO_x	PM_{10}
Summary statistics			
Number of data pairs	157	15	10
Mean	28.6(O) 28.9(P)	73.7(O) 58.2(P)	23.8(O) 21.2(P)
Minimum	13.0(O) 15.8(P)	22.9(O) 23.3(P)	15.2(O) 14.5(P)
Maximum	57.1(O) 60.5(P)	226(O) 275(P)	48.8(O) 50.6(P)
Standard deviation	9.53(O) 8.33(P)	55.0(O) 69.0(P)	12.1(O) 13.2(P)
Measures of bias			
Bias (O-P)	-0.25	15.46	2.59
FB	0.01	-0.23	-0.11
Measures of uncertainty			
NMD ((O-P)/P)	-0.01	0.54	0.17
NRMSE (RMSE/O)	0.16	0.53	0.17
R^2	0.70	0.80	0.94
IA	0.93	0.93	0.98
Fa2	1.00	0.87	0.90
RMS	0.46	0.32	0.35

Notes:
O = observed, P = Predicted, all in $\mu g/m^3$
NMD = Normalised mean deviation
FB = Fractional bias
NRMSE = Normalised root mean square error
R^2 = Correlation coefficient
IA = Index of agreement
Fa2 = Fraction within a factor of two
RMS = Root mean square difference in concentrations

(Source : this study)

5.8.7 Comparison with National Mapping

The Kent and Medway model results for NO_2, NO_x and PM_{10} are compared to the correlations (r^2) reported by Stedman *et al* (2001a) in evaluating the results of modelling and observed data across the UK (see Table 5.8). Although the R^2 values are comparable for all three pollutants it is important to note that the work of Stedman *et al* (2001a) is empirical whereas this study based on a process model which relates sources to concentrations at receptor points. Both approaches may be considered as equally robust although the source-receptor approach provides for application in other situations.

The population weighted background NO_2 concentrations estimated for Kent and Medway (Table 5.9) are within the range of national mapping results published by Stedman *et al* (2001b) providing further evidence the model is performing sufficiently.

However, the population weighted background PM_{10} concentrations are lower for Kent and Medway than comparable national mapping data. The reason for this is not clear although most likely related to the choice of background contribution (see Section 5.5.2).

Table 5.8
Model Evaluation With National Mapping Method

	This Study (1997, 2000)		National Mapping (1996-1999)	
	R^2	Number of sites	R^2	Number of sites
NO_2	0.70	157	0.71 – 0.84	33 – 55
NO_x	0.80	15	0.68 - 0.83	33 - 55
PM_{10}	0.94	10	0.36 – 1.00	4 – 40
Notes:	National Mapping statistics from Stedman *et al* (2001a)			

(Source : this study)

Table 5.9
Population Weighted Background NO_2 and PM_{10} Concentrations

	NO_2		PM_{10}	
	1996 - 1999	2010	1996 – 1999	2010
Scotland	18.813 – 23.069	11.618 – 13.191	18.379 – 21.940	15.108 – 16.312
Wales	20.169 – 24.389	12.438 – 13.819	19.588 – 25.243	16.309 - 18.330
Northern Ireland	16.528 – 20.430	10.935 – 12.156	20.623 – 25.721	16.526 – 17.862
Inner London	47.386 – 54.697	34.213 – 37.837	27.272 – 37.808	20.497 – 24.068
Outer London	40.703 – 46.791	28.944 – 31.533	25.462 – 34.673	19.579 – 22.847
Rest of England	29.837 – 34.943	20.491 – 22.080	21.965 – 28.559	17.701 – 19.990
UK	29.565 – 34.542	20.341 – 21.981	21.984 – 28.587	17.641 – 19.913
Kent and Medway	30.13	20.62	18.01	13.43
Notes:	National Mapping statistics from Stedman *et al* (2001b) data for Kent and Medway reported for 2000 and 2010			

(Source : this study)

5.8.8 Spatial Evaluation

The model results for NO_2 and PM_{10} in 2005 are presented in Figures 5.16 and 5.17 respectively. In both cases, the concentrations increase with urban intensity, traffic levels and proximity to industry; the highest concentrations are generally in the North Kent / Medway Valley area and along the principal transport corridors. Secondary areas of higher concentrations are at airports and seaports (Lydd and Dover, reflecting the approximations made in emissions (see Section 4.4.2 and Figures 4.6 and 4.7).

The annual mean NO_2 concentrations predicted for 2005 (Figure 5.15) breach the UK Air Quality Objective of 40 $\mu g/m^3$ at the side of most of the major roads in Kent and Medway with concentrated areas of the objective being breached in Dartford, Gravesham and Ashford. When compared to the results of R&A by local authorities, in terms of designated AQMAs, the model appears to be very conservative. Using this model alone would have resulted in significantly more stretches of road being designated AQMAs, including ten of the 11 AQMAs currently designated by local authorities. The principal difference between the results is the consideration of public exposure at the local level. The model is therefore not as accurate as local authority R&A but does provide the land use and transportation planner with a clear indication of where certain land uses, such as housing, should be restricted.

The annual mean PM_{10} concentrations predicted for 2005 (Figure 5.16) breach the short term UK Air Quality Objective (equivalent to an annual mean of 28 $\mu g/m^3$; see Section 3.4) at the side of most motorways in Kent and Medway. Eight of the 11 AQMAs designated by local authorities for this pollutant are predicted by the model. Although, similar to NO_2, the principal difference between the results is the consideration of public exposure at the local level, some areas where the objective would be breached were missed. Overall, the results for PM_{10} support those for NO_2, providing further evidence for implementing land use and transportation policy to reduce exposure to air pollution by restricting housing near to main roads.

5.9 CONCLUSIONS

A source-receptor modelling approach has been developed and evaluated by use of sensitivity analyses, calibration and evaluation using data from the Kent and Medway Air Quality Monitoring Network. The sensitivity analyses demonstrate the model is robust, but provides an indication of the model uncertainty, which is particularly affected by meteorology. The model has been calibrated using two parameters: a roadside enhancement factor to derive roadside concentrations from area source model results; and NO_x to NO_2 conversion factors for background and roadside locations. Evaluation of the model was undertaken using a range of statistical tests confirming the model performs well compared to other published work. With reference to DEFRA (2000k) guidance, the model is considered sufficient for the purposes of R&A.

The modelling approach is very efficient, with each model run (industrial and all other sources) compilation and GIS mapping of results taking less than two working days. The majority of this time is taken in running the industrial source model. Running the *Matrix Model*, specifically developed in the course of this research, for all other sources takes less than five minutes. The modelling approach therefore provides land use and transportation planners with a rapid tool for evaluating future scenarios within a practicable timescale covering a range of practical alternative options.

Comparison of the model results with the review and assessment of air quality undertaken by local authorities further demonstrates the good performance of the model but also reveals some shortcomings. At the local scale, the emissions inventory requires refinement. Emissions from seaports and airports appear to be overestimated. Emissions of some urban roads appear to be underestimated where the results do not correlate with the designation of Air Quality Management Areas. Overall, the model appears to provide a conservative result when compared to the designation of Air Quality Management Areas. This is principally due to the model not accounting for exposure at the local scale, particularly with respect to housing adjacent to roads. For the land use and transportation planner, the model is sufficient in this respect but, if required for local scale air quality management, more detailed data on the location of housing adjacent to main roads is required.

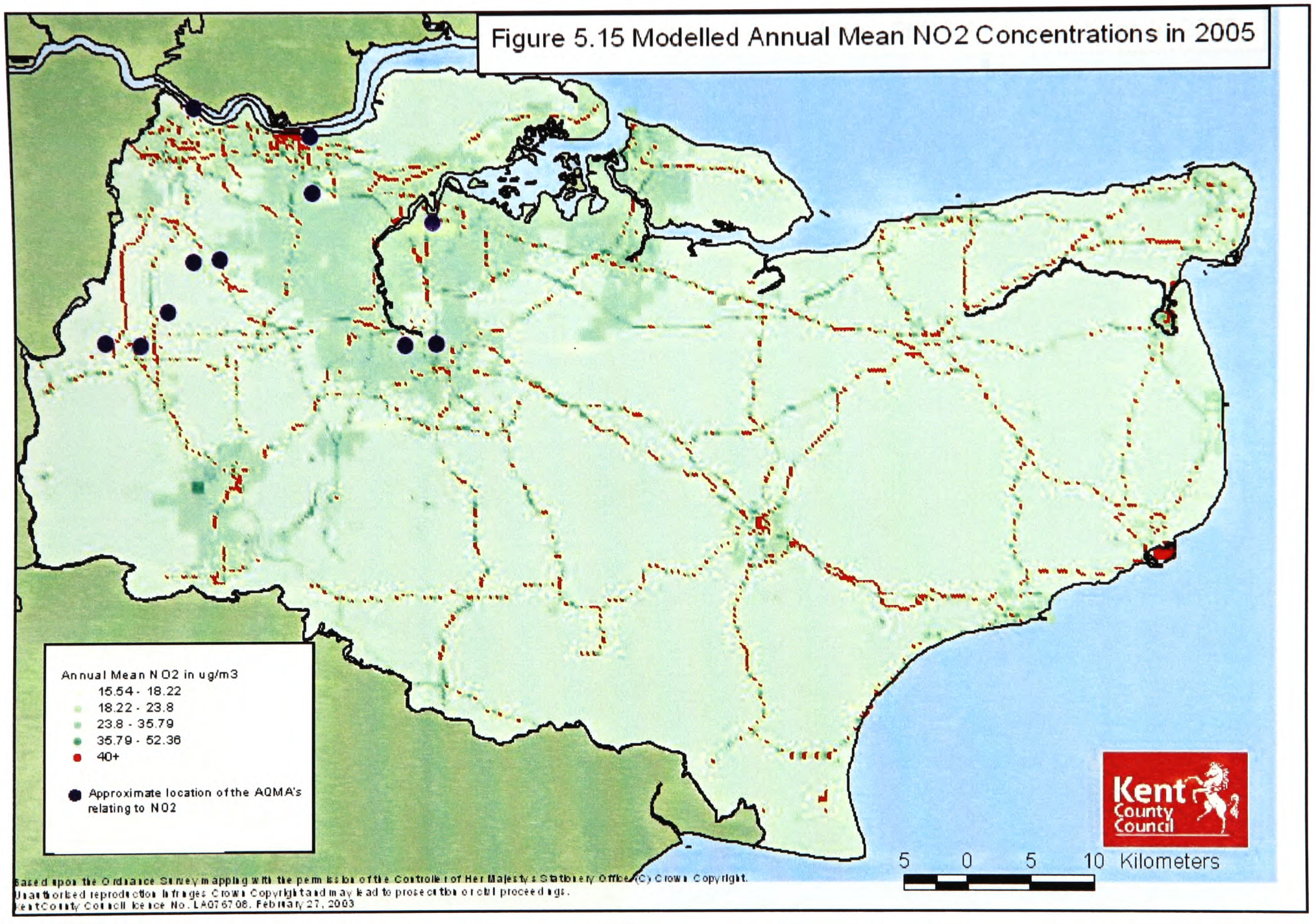


Figure 5.15 Modelled Annual Mean NO2 Concentrations in 2005

Figure 5.16 Modelled Annual Mean PM10 Concentrations in 2005

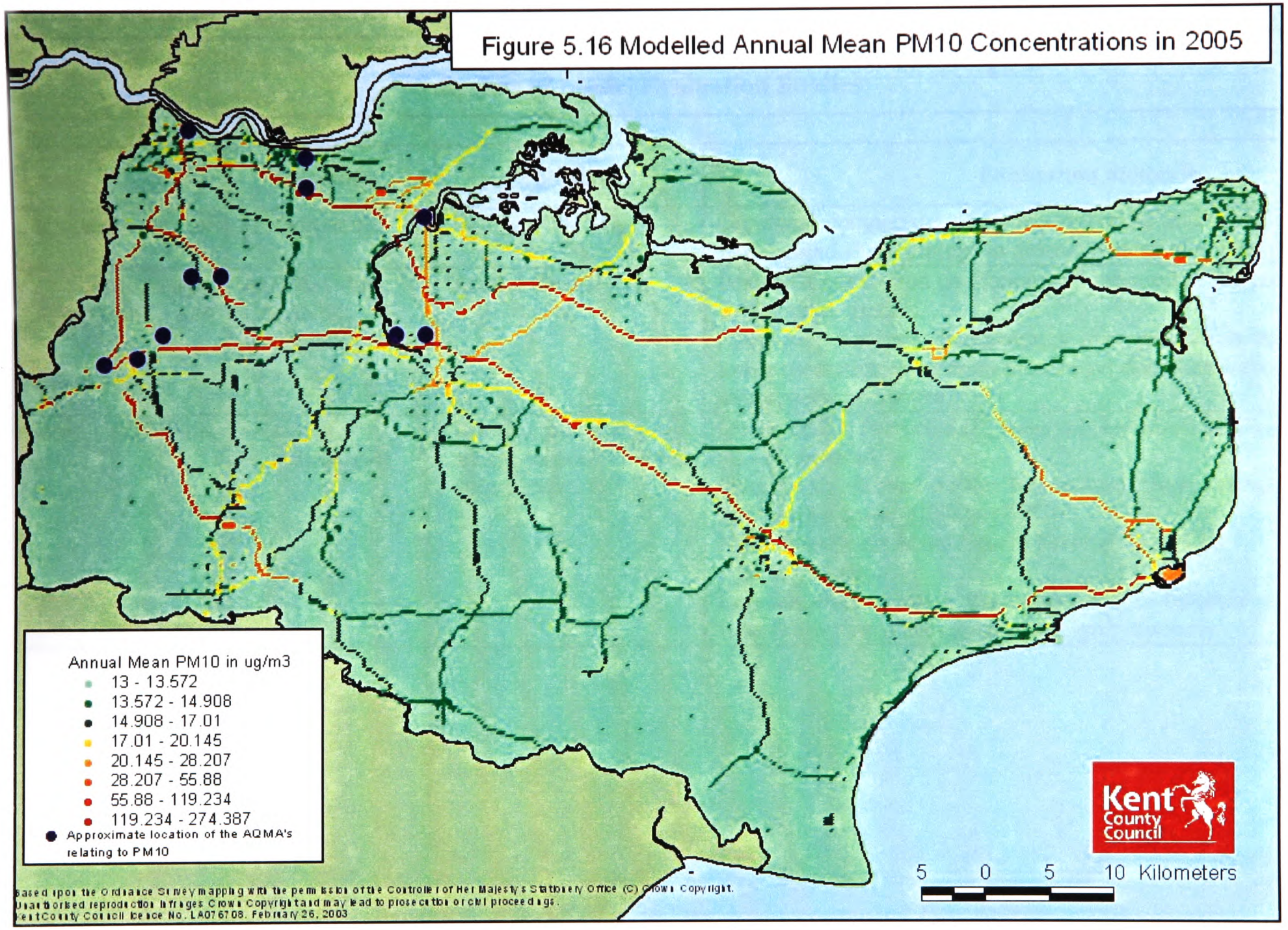


Table 5.10
Review of Model Evaluation Studies

Reference	Model(s)	Pollutant	Evaluation methods
Graz, Austria; Almbauer *et al* (2000)	Study specific	Hourly, NO, NO_2	Graphical comparison of hourly observed and modelled NO and NO_2. Highlighted shortcomings in diurnal pattern of emissions inventory
M4/M5 motorway interchange, Bristol, UK; Courthold and Whitwell (1998)	ADMS-Urban, CALINE4, AAQuiRE	Annual mean, NO_2	Pearson's correlation coefficient (r^2) reported in the range 0.678 to 0.896 for the three models. Regression analysis demonstrates over prediction near motorway and under prediction away from motorway; due to underlying assumptions in atmospheric chemistry.
Stockholm, Sweden; Häggkvist (1997)	Gaussian, Eulerian (grid)	1-week summer and winter means, 15-week winter mean, NO_x	Observed (O) Predicted (P) standard deviations (O,P) and Bias (O - P) reported. Uncertainty defined as the Normalised Mean Deviation (NMD = (O-P)/P) in the range –32% to +58%. Normalised Root Mean Square Error; 0.34 to 1.10 r^2; 0.37 to 0.64. Less uncertainty when the closest model receptor value was selected from those within the radius of one grid distance.

Table 5.10
Review of Model Evaluation Studies

Reference	Model(s)	Pollutant	Evaluation methods
Hanna SR, Egan BA, Purdum J and Waglar J (1999) Evaluation of the ADMS, AERMOD and ISC3 dispersion models with the OPTEX, Duke Forest, Indianopolis and Lovett field data sets, Sixth International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Applications, Rouen, Frnace, 11-14 October 1999	ISCST3, AERMOD, ADMS3		Simple tabular presentation of the maximum observed and modelled, geometric mean, geometric variance. Fa2; 0.064 to 0.8.
Ozone simulation over Southern Taiwan; Kuang and Chen (1998)	Urban Airshed Model		Correlation (not specified) = 0.62.
Comparison of Modelling system with urban measurements, Helsinki, Finland; Karppinen *et al* (2000)	Study specific		Mean (O, P) maximum (O, P) and standard deviation (O, P) reported. Bar chart of O versus P hourly, daily and annual mean NO_x and NO_2 concentrations, with reference to relevant Air Quality Standards. Index of Agreement (IA); –0.75 to +0.79 r^2, 0.5 to 0.65 Normalised Mean Square Error (NMSE); 0.26 to 0.45 Fractional Bias (FB); -0.092 to +0.118

Table 5.10
Review of Model Evaluation Studies

Reference	Model(s)	Pollutant	Evaluation methods
Statistical and Diagnostic evaluation of Helsinki Model, Finland; Kousa *et al* (2001)	Study specific		Mean (O, P) maximum (O, P) and standard deviation (O, P) reported. Time series plots of monthly mean O and P Box and whisker plots of O versus P for different wind speed classes, wind direction and Pasquill stability classes IA; +0.65 to +0.82 r^2; 0.39 to 0.68 NMSE; 0.21 to 1.19 FB; -0.289 to +0.255
Semi-empirical model of urban PM_{10} in Helsinki, Finland; Kukkonen *et al* (2001)	Study specific	Annual mean for each hour of the day, PM_{10}, at four monitoring stations, one year of data	FB; reported in the range –4 to +9% IA; reported in the range 0.85 to 0.94 Squared correlation coefficient (r^2) = 36 to 90%
Evaluation of CAR-FMI Model near a major road, Finland; Kukkonen *et al* (2001)	CAR-FMI	Hourly NO_x, NO_2 at three heights; only 3.5 m results reported here	No. of data points (N=125), Observed (O) Predicted (P) mean (O, P) maximum (O, P) and standard deviations (O, P) reported. Graphical presentation of scatter plot with 1:1, 0.5:1 and 2:1 boundaries IA = 0.849 (NO_x) and 0.735 (NO_2) r^2 = 0.794 (NO_x) and 0.800 (NO_2) NMSE = 0.215 (NO_x) and 0.209 (NO_2) FB = +0.061 (NO_x) and -0.192 (NO_2) Factor of Two (Fa2 = $0.5 \leq (P/O) \leq 2$) = 0.937 (NO_x) and 0.667 (NO_2)
Evaluation of Street Canyon Model, Helsinki, Finland; Kukkonen *et al* (2001)	OSPM	Hourly NO_x, NO_2	No. of data points (N=1125), Observed (O) Predicted (P) reported. Graphical presentation of scatter plot with 1:1, 0.5:1 and 2:1 boundaries r^2 = 0.77 (NO_x) and 0.83 (NO_2) FB = +0.045 (NO_x) and +0.22 (NO_2); the latter considered a systematic over prediction

Table 5.10
Review of Model Evaluation Studies

Reference	Model(s)	Pollutant	Evaluation methods
First validation trials of a microscale lagrangian particle model for street canyons, Italy; Lanzani and Tamponi (1995)	Study specific	Not detailed, hourly	Results presented from two street canyon studies; Cologne and Lyon. NMSE = 0.057 (Cologne) and 0.062 (Lyon) Bias = 0.186 (Cologne) and 0.254 (Lyon) FB = 0.170 (Cologne) and 0.225 (Lyon)
New formulation of σ_y and σ_z Mangia *et al* (1998)	Study specific, Hybrid Plume Dispersion Model (HPDM) OML	SF_6 tracer, maximum hourly	Validation of model by performance comparison with two existing models using the Copenhagen and Kincaid field data sets. Graphical presentation of scatter plots with 1:1, 0.5:1 and 2:1 boundaries r^2; 0.92 – 0.36 (Mangia) 0.44 – 0.87 (HPDM) 0.15 – 0.82 (OML) NMSE; 0.10 – 0.74 (Mangia) 0.61 – 0.75 (HPDM) 1.12 – 1.24 (OML) FB; 0.092 – 0.16 (Mangia) 0.19 – 0.55 (HPDM) 0.14 – 0.76 (OML) Fa2; 66 – 92% (Mangia) 56 – 65% (HPDM) 22 – 55% (OML)
Comparison of Model Evaluation Methodologies, McHugh *et al (1999)*	ISCST3, AERMOD, ADMS3	Uses four field datasets (Parairie grass, Kincaid, Indianapolis)	Uses the Model Validation Kit (Olesen, 1999) applied to both the Maximum Arcwise Concentration (MAC) observed and the Near Centreline (NC) observed. For all models and datasets presented: BIAS; -146.40 to +33.22 (nb: different units of concentration) NMSE; 0.57 to 3.83 r^2; 0.162 to 0.749 Fa2; 0.256 to 0.759 Fb; -0.442 to +0.855

Table 5.10
Review of Model Evaluation Studies

Reference	Model(s)	Pollutant	Evaluation methods
Hybrid plume model for local scale dispersion; Nikmo *et al* (1999)	Study specific and one cited model (O'Brien, 1970)	SF_6 tracer, maximum hourly	Validation of model by performance comparison with an existing model using the Kincaid field data set. Graphical presentation of scatter plots with 1:1, 0.5:1 and 2:1 boundaries No. of data points (N=253), mean (O, P) reported. r^2; 0.136 (Nikmo *et al*) 0.156 (O'Brien) NMSE; 1.56 (Nikmo *et al*) 1.49 (O'Brien) FB; -0.085 (Nikmo *et al*) –0.078 (O'Brien)
New generation model for a large urban area; Owen *et al* (1999)	ADMS-Urban	Annual mean and hourly percentiles of SO_2 and NO_x; for four urban background monitoring sites. Annual mean NO_x reported here	Graphical analysis of peak concentrations with wind direction and Monin-Obukhov length Summary statistics of summer and winter (three month mean) concentrations of NO_x for two diurnal emission patterns (LRC and SEIPH): Mean (O, P) and standard deviation (O, P) reported Bias; -6 to +35 (LRC) +0.25 to +29 (SEIPH) FB; -0.18 to +0.75 (LRC) +0.01 to +0.77 (SEIPH) NMSE; 0.55 to 1.91 (LRC) 1.02 to 3.01 (SEIPH) r^2; 0.16 to 0.55 (LRC) 0.15 to 0.33 (SEIPH) Fa2; 24 to 67% (LRC) 17 to 57% (SEIPH)
Development of a spatial weighting method for comparing modelled and observed data; where monitoring sites are not co-located with model receptor grid, Istanbul, Turkey; Şen (1998)	Study specific	Monthly mean SO_2 and Suspended Particulate Matter (SPM) for 16 monitoring sites.	Results presented for one month for all 16 sites. % error defined as (O-P)/O (reported to be <11% with most being <5%)

Table 5.10
Review of Model Evaluation Studies

Reference	Model(s)	Pollutant	Evaluation methods
Application of Ambient Background Model in London, UK; Seika *et al* (1998)	Ambient Background Model (ABM)	Annual mean NO_x, PM_{10}, C_6H_6, CO, one monitoring station, one year of data	Results presented as histogram with separate modelled components (background, secondary and local contributions). NO_x, PM_{10}, and C_6H_6, within ±10% of observed value CO within ±25% of observed value
Model for low wind speeds, India; Sharon *et al* (1996)	Study specific	SF_6 tracer,3-minute and 30-minute samples	Evaluation with field experiment on New Delhi sports ground. Tabular presentation of results comparing observed and predicted 3-min and 30-min concentrations. Fractions within factors of 2, 4 and 6 calculated
Model evaluation of ISC using emissions inventory and monitoring data compiled for Jamshedpur, India; Sivacoumar *et al* (2001)	ISC	Daily mean NO_x; 15 day campaign with 19 monitoring sites	No. of data points (N=285), Observed (O) Predicted (P) mean (O, P) maximum (O, P) and standard deviations (O, P) reported. Bar chart comparison of maximum O and P at each monitoring site Linear regression; intercept and slope, COR = 0.71 Systematic Mean Square Error (MSE) = 13.30 Unsystematic MSE = 7.27 Total RMSE = 15.16 IA = 0.68
Benchmark Study for London, UK; Stanger (1997)	Indic AirViro	Annual mean and hourly percentiles of SO_2 and NO_x; for four urban background monitoring sites.	Tabular comparison of observed and predicted three month mean winter and summer concentrations; no statistical analysis. For NO_x: 72 – 132 (O) 779 - 571 (P); Bloomsbury 69 –122 (O) 257 – 335 (P); Bridge Place 32 – 74 (O) 71 – 83 (P); Bexley 43 – 80 (O) 112 – 463 (P); Greenwich NO_2 results are better, but also show marked model over prediction.

Table 5.10
Review of Model Evaluation Studies

Reference	Model(s)	Pollutant	Evaluation methods
Benchmark Study for the West Midlands, UK, including street canyon, specific hour pollution episodes and winter/summer season scenarios; Veal and Appleby (1997)	Indic AirViro, ADMS-Urban	NO_x, SO_2 and PM_{10}, hourly and seasonal means	Graphical presentation of cumulative frequencies for hourly data. Tabular summaries of mean and standard deviations (O, P) mean error, mean absolute error, RMSE and r^2 for street canyon and episode studies (hourly data). Comparison of O and P only for seasonal scenarios.
Development of two field data sets (with (w) and without (w/o) a fence) to evaluate a dense gas local scale dispersion (CFD) model, Greece; Venetsanos *et al* (1998)	Study specific	Propane, 2-3 minute means	Results presented graphically to show concentration (period mean, ±1 stdev) as function of distance downwind. Tabular summary of: FB; -0.03 (w) +0.14 (w/o) NMSE; 0.10 (w) 0.19 (w/o) MRB; +0.18 (w) +0.02 (w/o) MRSE; 0.74 (w) 0.55 (w/o) Geometric mean bias (MG); +0.51 (w) +0.50 (w/o) Geometric mean variance (LN (VG)); 6.76 (w) 9.6 (w/o) Fa2; 70% (w) 80% (w/o) Fa5; 83.3% (w) 90% (w/o) Fa10; 86.7% (w) 90% (w/o)
Evaluation of a weighting factor within a complex terrain model (AERMOD) by comparison with CTDMPlus using field data from four complex sites, US; Venkatram *et al* (2001)	AERMOD, CTDMPlus	SF_6 tracer, 1-hour, 3-hour, 24-hour means	Quantile-Quantile plots presented; results reported to be *generally within a factor of two* for Tracy Power Plant, Nevada. Other results reveal model under prediction at lower end of concentration scale and over prediction at higher end of concentration scale; the former probably due to under estimated background contribution.

Note: Formulae for statistical expressions provided in main text.

(Source : this study)

6 TREND ANALYSIS AND FUTURE SCENARIOS

This Chapter provides the results of testing the tools developed in Chapters 3 to 5 (assessment criteria, emission inventory and dispersion models) to assess a number of land use and transportation planning scenarios. The results are used to evaluate the impact of these policy scenarios in improving local air quality whilst incorporating policies to reduce greenhouse gas emissions.

6.1 INTRODUCTION

A number of land use and transportation planning scenarios have been developed and assessed using the tools described in chapters 3 to 5. The first scenario (A) provides the baseline, or 'business as usual' scenario with an analysis of trends in emissions from 1990 to 2020 and subsequent change in local air quality. With reference to national emission reduction targets described in Chapter 2, four alternative future scenarios are then described.

- Scenario B : Isle of Grain and Kingsnorth Power Stations
- Scenario C : Reducing Heavy Goods Vehicles on Main Roads
- Scenario D : Low Emission Vehicles
- Scenario E : Meeting the RCEP (2000) Target for 2050

The impacts of these alternative emissions reduction scenarios are then assessed, in terms of local air quality, to test the policy interface between local air quality management and the national climate change programme. The results provide suggestions for optimum policies to meet both emission reduction targets and air quality objectives. The use of the emissions inventory and models to assess alternative scenarios compared to 'business as usual' is similar to that reported by Stedman *et al* (2001a,b) who developed a number of NO_x and PM_{10} reduction scenarios to determine future reductions in population exposure and assess potential compliance in 2010 with the Stage II Air Quality Limit for PM_{10} although any consequent reduction or increase in CO_2 emissions were not reported.

In addition to the above scenarios based on alternative emission profiles with little or no change to land use, a final scenario (F) was derived to assess the impact of a new development. This scenario was based on recent proposals by UK Government for a new London Airport on the north Kent Marshes (DEFRA, 2002).

6.2 SCENARIO A : BUSINESS AS USUAL

6.2.1 Emission Trends

Trends in emissions of NO_x, PM_{10} and CO_2 by sector for the years 1990 to 2020 are illustrated in Figures 6.1 to 6.3. Emissions of NO_x (Figure 6.1) are projected to decline from 124.7 kilotonnes in 1990 to 74.16 kilotonnes in 2010 with only a very small further reduction by 2020. Although emission from industry decline slightly, the most significant reduction is in road transport; from 52.8 kilotonnes in 1990 to 15.4 kilotonnes in 2020, a fall of 71% despite high growth in road traffic. Across all sectors, NO_x emissions in 2010 are expected to be 59% of emissions in 1990 and comparable to the emissions reduction target for reducing emissions between 1980 and 2010 (Table 2.1). The trend in PM_{10} emissions follows a similar pattern although significant reductions from industry, associated principally with fuel switching from coal and oil to gas, are observed between 1990 and 1997. Again, the most significant reduction is observed in road transport contributing to total emission reductions from 11.6 kilotonnes in 1990 to 5.3 kilotonnes in 2020; a reduction of 54%. Stedman *et al* (2001a) provide similar projections for the UK as a whole. The UK total urban road traffic NO_x and PM_{10} emissions in 2020, for example, are projected to be 19% of 1990 emissions, with UK total non-road NO_x and PM_{10} emissions 58% and 34% of 1990 emissions, respectively.

Emissions of CO_2 in Kent and Medway are predicted to fall from 9.3 megatonnes in 1990 to 9.0 megatonnes in 2010 and rise thereafter. The fall between 1990 and 2010 is 2.5%; well below the UK Government target of 20% and the national commitment to the Kyoto Protocol of 12.5% over the same period. Despite reductions in industrial, residential and commercial contributions, the forecast increase in road traffic is the cause of this trend in CO_2 emissions. Road transport emissions are projected to rise by almost 39%, from 3.1 megatonnes in 1990 to 4.3 megatonnes in 2010.

The validity of these projections of CO_2 can be checked by comparing with national forecasts illustrated in Figure 2.4 (DETR, 2000a). National projections also suggest a reduction in emissions from the industrial sector and increases from the transport sector albeit, not as large (approximately 30% over the same period). However, the geographical location of Kent, with strategic transport corridors from continental Europe

to the UK, is reflected in the greater proportion of total emissions contributed by this sector (40–42%; see Table 4.14) compared to approximately 23% nationally.

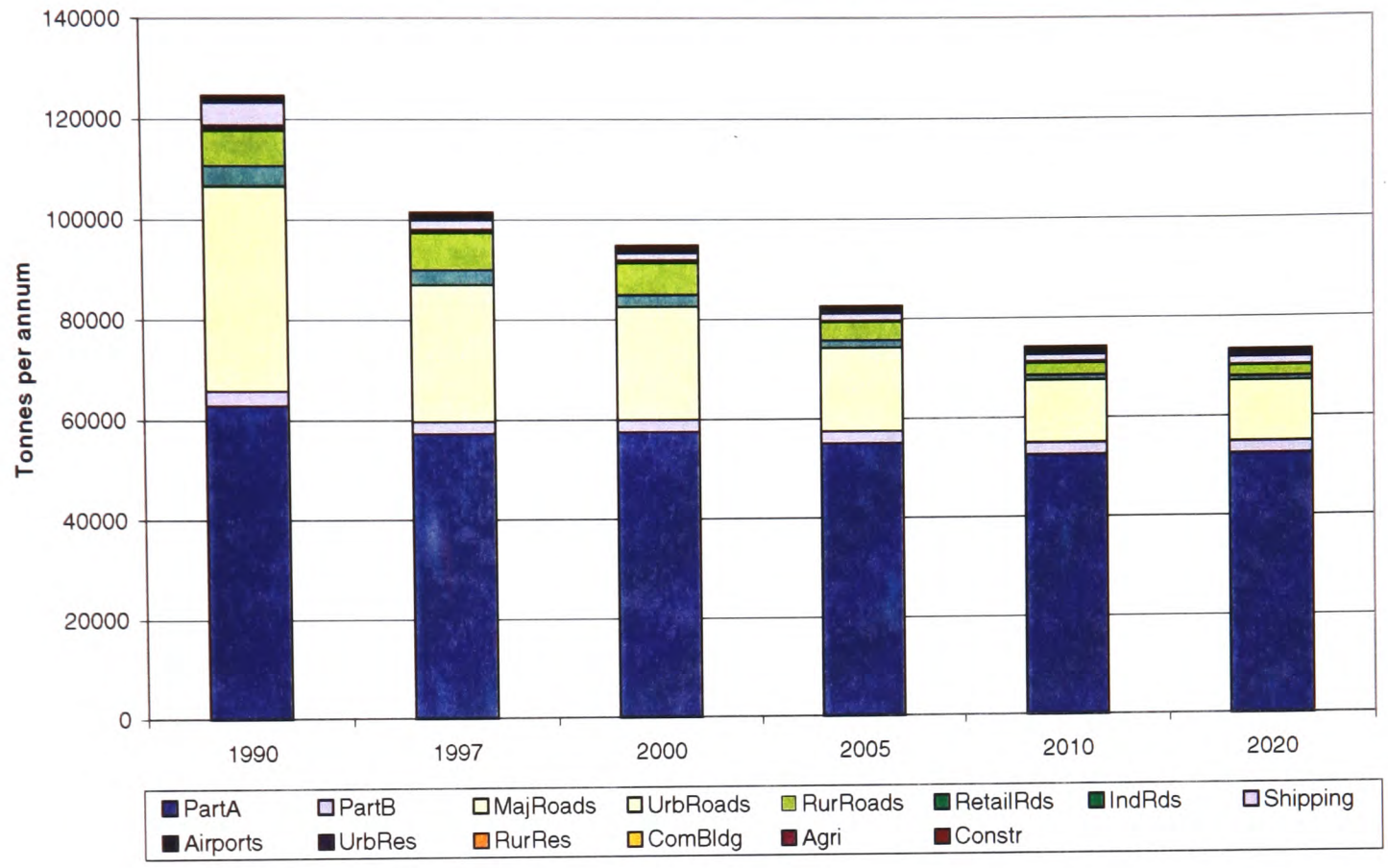


Figure 6.1 : NOx Emissions : 1990 to 2020

(Source: this study)

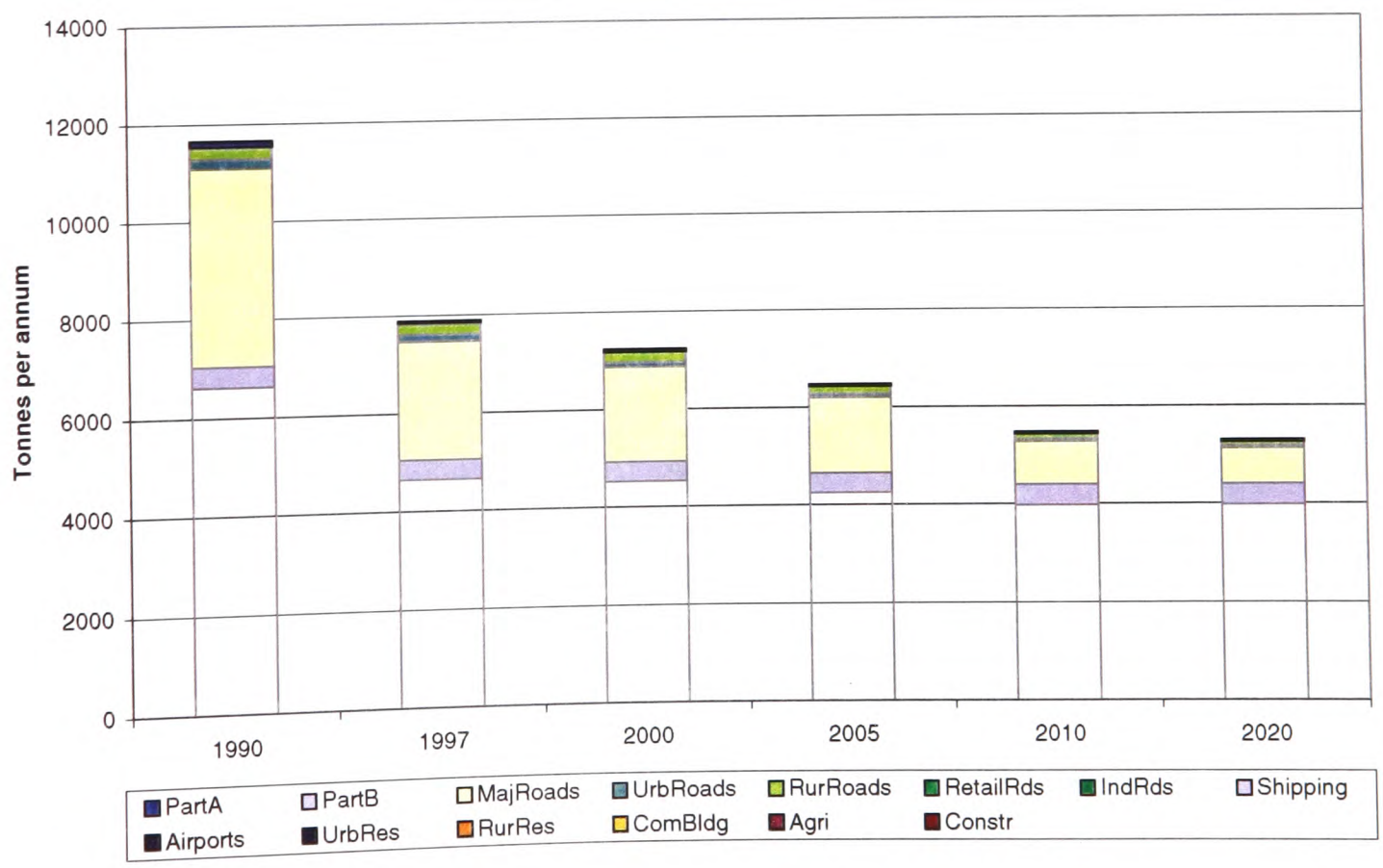


Figure 6.2 : PM_{10} Emissions : 1990 to 2020

(Source: this study)

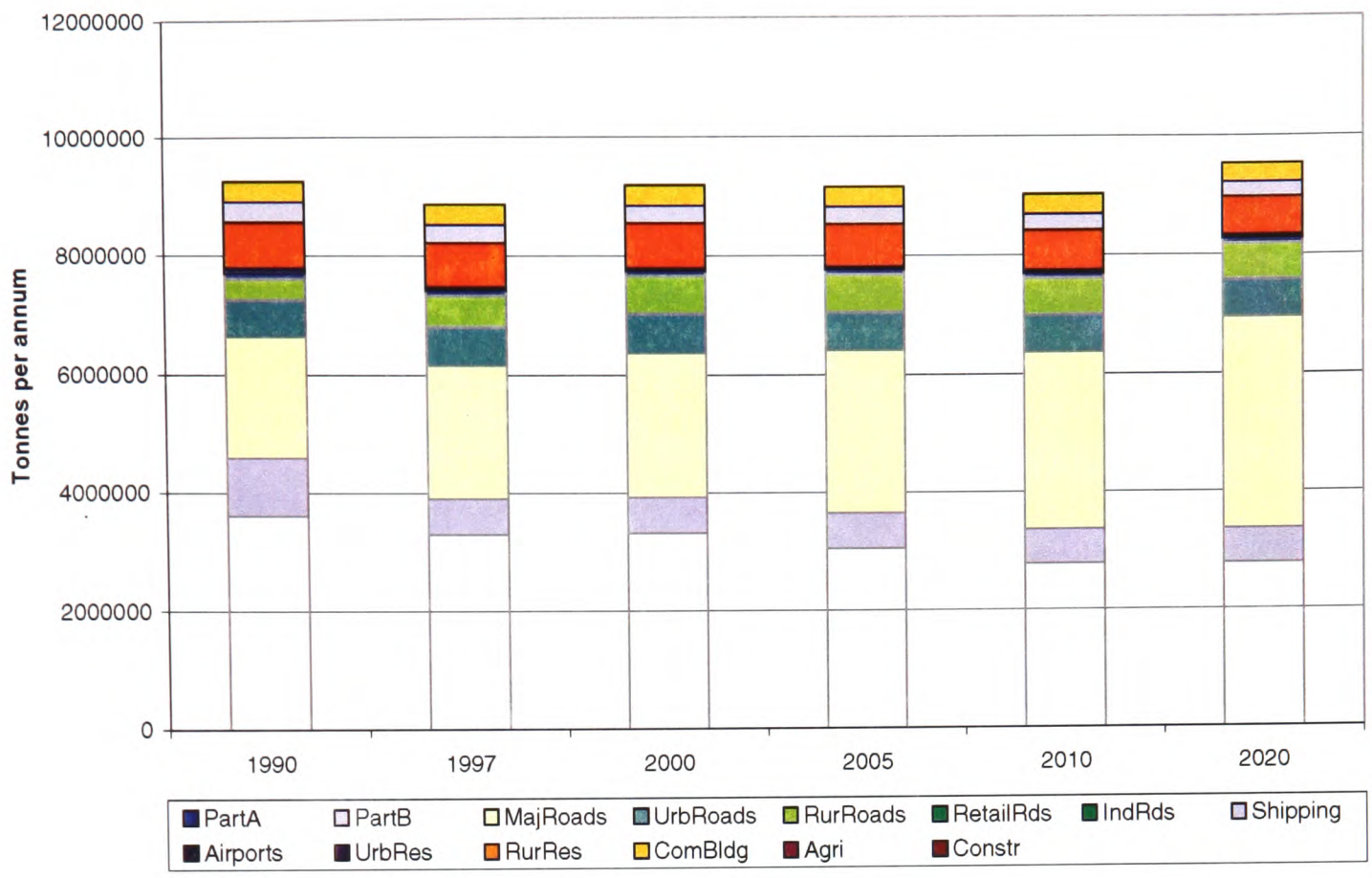


Figure 6.3 : CO_2 Emissions : 1990 to 2020

(Source: this study)

6.2.2 Air Quality Trends

Modelling emissions of NO_x and PM_{10} from 1990 to 2020 allows for assessment of improvements in air quality, depicted in terms of annual mean NO_2 and PM_{10}. This is considered particularly useful as the monitoring data is typically limited to 1996 onwards. A spatial analysis of the results is presented in Section 5.7.8. The trends in air quality, in terms of total number of 250 m by 250 m grid squares breaching the relevant air quality criteria at roadside locations, are presented in Table 6.1. The results illustrate the improvement in local air quality observed since 1990 and forecast future improvements to 2010. These improvements are primarily associated with vehicular emissions control technology (see Figures 4.3 and 4.4). Both NO_x and PM_{10} reductions achieved by individual vehicular emissions control, are expected to be off set by further traffic growth. This is illustrated in Figures 6.1 to 6.3 above and by the results in Table 6.1. This in contrast to the UK as a whole, where roadside air quality is predicted to improve (Stedman *et al*, 2001a-b) and reflects the higher than average traffic growth forecast for Kent and Medway (see Figure 2.3). Further improvements in vehicular PM_{10} emissions control being introduced within the vehicle fleet will result in reducing emissions and some further improvement in ambient concentrations of this pollutant

between 2010 and 2020. All of these squares include a main road where traffic emissions are greatest. Whether local people reside adjacent to these main roads at these locations is not included for in this study. Further work would be required to determine the degree of exposure within each of the grid squares where a breach of air quality criteria is modelled.

Table 6.1
Modelled Trends in Air Quality

Year	Number of Grid Squares Breaching Air Quality Criteria (out of a total of 62638 in the study area)		
	NO_2 (40 µg/m^3)	PM_{10} (28 µg/m^3)	PM_{10} (17 µg/m^3)
1990	732	2763	12702
1997	416	1995	9172
2000	322	1643	7553
2005	213	1498	6887
2010	129	1234	5673
2020	139	1183	5496

Note: The key criteria for PM_{10} are the short term objectives for 2005 and 2010, equivalent to annual mean concentrations of 28 µg/m^3 and 17 µg/m^3, respectively (see Section 3.3.3 *Particles*).

(Source: this study)

6.3 ALTERNATIVE EMISSION SCENARIOS

The results of trend analysis suggests that emissions of CO_2 in Kent and Medway need to be reduced significantly beyond the business as usual scenario if the UK Government target of 20% and the national commitment to the Kyoto Protocol of 12.5% (1990 to 2010) is to be adopted locally. The results of modelling local air quality, presented in Chapter 5, indicate that road traffic is the major cause of designating AQMAs. The large background contribution to PM_{10} concentrations (Section 5.5.2) suggests meeting the EU Stage II objective for 2010 will require reductions in regional emissions.

6.3.1 Scenario B : Isle of Grain and Kingsnorth Power Stations

The study area includes the oil fired Isle of Grain and the dual fired (oil or coal) Kingsnorth power stations. In terms of Part A Processes emissions in Kent and Medway, these two power stations represent 37-45% of NO_x, less than 1% of PM_{10} and

39 – 50% of CO_2 alone. Given the age of these stations, both commissioned in the 1970s, one reasonable assumption is they are both retired between 2005 and 2010. In addition to reducing NO_x emissions, this would significantly reduce total CO_2 emissions, from all source categories in the study area, to 18% below 1990 levels in 2010 and 14% below 1990 levels by 2020.

A further assumption for the future operation of these two power stations is co-firing with bio-fuels such as wood chips (ENDS, 2002). Although NO_x and PM_{10} emissions are not expected to vary significantly, co-firing with bio-fuels up to 15% by weight would reduce station CO_2 emissions by approximately 7%. The impact, in terms of local air quality would be insignificant but this measure would reduce total CO_2 emissions in the study area in 2010 to 3.7% below 1990 levels.

6.3.2 Scenario C : Reducing Heavy Goods Vehicles on Main Roads

As illustrated in Figure 2.5, traffic growth in Kent and Medway is significantly above the national average. Recent disruptions to Channel Tunnel rail freight services has also led to increased movement of goods by road, above that forecast previously.

Assuming a slowing down in traffic growth more in line with national projections would realise a reduction in road traffic to 20% below that currently forecast for 2010. Compared to a 1990 baseline, this would slightly improve NO_x and PM_{10} emissions from a 41% to a 43% reduction and from a 53% to a 55% reduction, respectively. CO_2 emissions would be reduced significantly from 3.5% to 12% below 1990 levels by 2010.

A further scenario was considered assuming measures were introduced to encourage the movement of goods from road to rail, resulting in removing 25% of heavy goods vehicles from main roads in the study area by 2020. Rail lines in Kent and Medway are electrified via the National Grid with no local emissions (see Section 4.3.8).

Compared to the business as usual scenario for 2020, this measure would reduce NO_x and PM_{10} emissions from main roads by 16% and 4%, respectively. In terms of total emissions from all source categories, this is equivalent to 3% and 1% for NO_x and PM_{10}

respectively. CO_2 emissions from main roads would be reduced to 8.6% below the 2020 baseline.

The impact of these measures in terms of local air quality is illustrated in Table 6.2, following the same methodology as for Table 6.1. The results indicate a small benefit from reducing HDV traffic on main roads by 25% using the metric of the number of grid squares where air quality criteria are breached. Local air quality would remain similar to that modelled for 2010 2020 suggesting this management option would assist in stabilising local air quality at roadside locations.

Table 6.2
Local Air Quality Impact of Reducing Heavy Goods on Main Roads

Year 2020	Number of Grid Squares Breaching Air Quality Criteria (out of a total of 62638 in the study area)		
	NO_2 (40 µg/m^3)	PM_{10} (28 µg/m^3)	PM_{10} (17 µg/m^3)
Business as Usual	139	1183	5496
25% shift in goods from road to rail	132	1155	5220

(Source: this study)

6.3.3 Scenario D : Low Emission Vehicles

Following promotion by the NSCA (1998, 1999a,b) some nine local authorities are considering the introduction of Low Emission Zones to reduce local traffic pollution in urban areas (TTR, *personal communication*). This is expected to promote the introduction of cars fuelled by Liquefied Petroleum Gas (LPG) or Compressed Natural Gas (CNG) which would be allowed in such zones, in preference to older technology vehicles with higher emissions. As technology advances, the introduction of electric vehicles is expected. Life cycle analyses of different fuels (petrol, diesel, compressed natural gas, liquefied petroleum gas, low sulphur diesel, rape methyl ester and electricity) by Davies (1999) suggests the difference in CO_2 emissions are not significant (ie. national total will not vary). Davies (1999) continued her work undertaking a review of vehicle emission factors as part of a study into introducing low emitting vehicles in

Southwark, London. This work included determining whether tail pipe emissions where statistically different, given the range of data uncertainty. Davies (1999) concludes that CO_2, NO_x and PM_{10} emissions from cars using compressed natural gas are significantly lower than emissions from petrol or diesel cars. Similarly, NO_x and PM_{10} emissions from heavy goods vehicles are also significantly lower with compressed natural gas although CO_2 emissions are significantly higher. Tail pipe CO_2, NO_x and PM_{10} emissions are zero for all electric vehicles, although this technology is considered impracticable for heavy goods vehicles.

A low emissions vehicle scenario was developed for 2020 assuming hybrid cars (ie. using petrol/diesel fuels in rural areas/main roads, switching to electric for short journeys) and no change in heavy goods vehicles. For this scenario, vehicular emissions on urban roads are assumed to be zero and emissions from cars on major and rural roads reduced by 25%, to reflect the use of electric power for short trips.

Compared to the business as usual scenario for 2020, the promotion of low emission vehicles would reduce NO_x and PM_{10} emissions from main roads by 16% and 4%, respectively. In terms of total emissions from all source categories, this is equivalent to 3% and 1% for NO_x and PM_{10} respectively. CO_2 emissions from main roads would be reduced by 16% compared to the base case scenario for 2020. Assuming the increased electrical demand was met by renewable sources, CO_2 emissions for Kent and Medway would be reduced by less than 2% over the period 1990 to 2020.

The impact of these measures in terms of local air quality is illustrated in Table 6.3, following the same methodology as for Table 6.1. The model results do not indicate a significant improvement in peak concentrations of local NO_2. In contrast, peak concentrations of local PM_{10} concentrations would fall with the number of grid squares breaching the air quality criteria falling by 11-14%. These results suggest the introduction of low emission vehicles would be more effective than reducing the number of HDVs on main roads by 25%. However, further analysis of the source contributions made by light and heavy duty vehicles on main roads suggests most of this benefit is due to the reduction in LDVs on main roads rather than reductions in urban and rural traffic emissions. These results reflect the significant contribution made by traffic on main roads in Kent and Medway to air pollution, in contrast to major cities with extensive

areas of dense urban traffic. The conclusion would be different if the assessment considered population exposure to elevated air pollution. The introduction of Low Emission Zones would reduce air pollution levels in locations more densely populated s than along side main roads. However, forecasting land use to determine population exposure over a 20 year horizon will introduce a high level of uncertainty (see Section 5.9) which limits the extent of analysis.

Table 6.3
Local Air Quality Impact of Low Emission Vehicles

Year 2020	Number of Grid Squares Breaching Air Quality Criteria (out of a total of 62638 in the study area)		
	NO_2 (40 µg/m^3)	PM_{10} (28 µg/m^3)	PM_{10} (17 µg/m^3)
Business as Usual	139	1183	5496
Low Emission Zones	138	1023	4941

(Source: this study)

6.3.4 Scenario E : Meeting the RCEP Target for 2050

As described in Section 2.3.2, the Royal Commission on Environmental Pollution (2000) suggests global reductions of 60-70% below 1990 levels will be required within the next 50-100 years; translated as a domestic target in the order of 90% (DETR, 2000a; p9). CO_2 emissions from Kent and Medway could be reduced by:

- closing Kingsnorth and Grain Power Stations (replaced with renewable technologies) and, through efficiencies, reducing remaining Part A emissions by 20% and Part B emissions by 50%

- using a combination of traffic reduction measures, fuel switching and electric vehicles to reduce vehicular emissions on main roads to 50% of those in 2020. All vehicles on urban roads are required to be electric (ie. zero emissions). Further use of traffic reduction measures, fuel switching and electric vehicles to reduce vehicular emissions from rural roads to 25% of those in 2020, and from retail and industrial roads to 50% of those in 2020

- a freeze in shipping and airport growth, ie. emissions remain as for 2020

- further efficiency measures to reduce building emissions to 50% of those in 2020.

This scenario would reduce total CO_2 emissions in 2050 to 50% below 1990 levels; in line with the Royal Commission's recommendation. NO_x and PM_{10} emissions would be reduced by from all sectors to 34% and 26% of 1990 emissions, respectively.

The impact of these measures in terms of local air quality is illustrated in Table 6.4, following the same methodology as for Table 6.1. Compared to the management options described above, the package of measures to reduce CO_2 emissions by 50% would significantly improve local quality. The number of grid squares breaching the Air Quality Objective for NO_2 would fall from 139 to 53. These remaining grid squares are associated with airports where emissions are overestimated (see Section 5.9) and some roadside locations. A similar improvement is modelled for PM_{10}. Although this scenario focuses on meeting a climate change policy objective it represents the most effective option for improving local air quality.

Table 6.4
Local Air Quality Impact of Meeting RCEP Target

Year 2020	Number of Grid Squares Breaching Air Quality Criteria (out of a total of 62638 in the study area)		
	NO_2 (40 µg/m^3)	PM_{10} (28 µg/m^3)	PM_{10} (17 µg/m^3)
Business as Usual	139	1183	5496
Meeting RCEP Target	53	716	2069

(Source: this study)

6.4 SCENARIO F : A NEW AIRPORT AT CLIFFE

The final scenario considered in this study provides an example of using the tools to assess potential new development, as opposed to changes in the emissions profile of existing sources. The tools were used to assess the potential impact of the new development in a strategic rather than detailed manner, reflecting the nature of the development and availability of information.

6.4.1 South East Region Airport Study

In 2002 the UK Government published a Consultation Paper detailing proposals for the expansion of airport services in southeast England. These proposals include development of Stansted Airport and either a new runway at Heathrow or a new airport at Cliffe in the north Kent marshes (DEFRA, 2002d). The proposals are not detailed and provide information at a strategic level only. The tools developed in this study are considered appropriate for informing the planning authority of the potential impact of such a proposal within the constraints of a limited resource budget and available information.

6.4.2 Impact on Atmospheric Emissions

With limited time and resources available for the study, the assumption was made that the proposed airport would be equivalent, in terms of operations, to the present day Gatwick by 2010 and the present day Heathrow by 2020 (MM, 2002). The British Airports Authority (BAA) has published atmospheric emissions inventories for both these airports (British Airports Authority, 2000, 2002). The results of these inventories were overlain on to the 'business as usual' results for the Kent and Medway inventory presented in Chapter 4. The BAA emission inventories do not include off-site road traffic. For the purposes of this study, a transportation planner (Swanson, *personal communication*) was asked to provide an estimate of additional patronage of existing roads and the emissions for these roads increased pro rata. These estimates are summarised in Table 6.5. For the assessment year of 2020, the proposed airport at Cliffe would increase CO_2 emissions in Kent and Medway to 20% above 1990 levels. NO_x emissions would be 61% of 1990 levels, compared to the business as usual scenario of 58% for the same year. PM_{10} emissions in Kent and Medway would not change significantly as a result of the proposed airport.

Table 6.5
Cliffe Airport : Assumed Increase in Local Traffic

Road	% Split	Annual Average Vehicles Per Day	
		LDVs	HDVs
Airport Access Road	100	7000	700
A2 (north)	80	5600	560
M25 (west)	40	2800	280
M25 (east)	40	2800	280
M2 (south)	20	1400	140
M2 (south of Bluebell Hill)	5	350	35
A229 (M2 to M20)	15	1050	105
M20 (west)	10	700	70
M20 (east)	5	350	35

(Source: MM, 2002)

6.4.3 Impact on Local Air Quality

BAA (2000) has also commissioned air quality modelling studies for Gatwick. The results of these modelling studies have been overlain on to the results of running the Kent and Medway Model with 'business as usual' emissions and additional road traffic emissions, as per Table 6.5. For the purposes of evaluating the impact of the proposed airport, reference was made to the Air Quality Limit Value for NO_x (to protect of vegetation) and NO_2 (to protect human health). For the purposes of this discussion, only the results for NO_2 are presented. The dispersion modelling results are presented in Figure 6.4.

In general, the annual mean concentration of NO_2 at non-roadside locations is predicted to remain within the UK Air Quality Standard of 40 $\mu g/m^3$ across Kent and Medway. However, in addition to locations in the immediate vicinity of the airport, roadside concentrations may continue to breach the Standard, particularly along the A2/M2 corridor where Air Quality Management Areas have been designated by local authorities in accordance with Part IV of the Environment Act 1995. The additional traffic generated from the proposed airport would contribute further to poor air quality along the A2/M2 corridor and hence, would conflict with any such local measures to improve local air quality. The environs of the proposed airport is sparsely populated. At the regional scale the increase in population exposure to poor air quality will be affected

more by concentrations being elevated at the roadside rather than in the immediate vicinity of the airport where the population is very low.

6.5 CONCLUSIONS

Assuming a 'business as usual' scenario (A) emissions of NO_x and PM_{10} in Kent and Medway have declined since 1990 but are projected to stabilise between 2010 and 2020. The principal cause for this trend is improvements in vehicle exhaust emissions, with traffic growth expected to override these improvements between 2010 and 2020. The implication is that without further vehicle emissions control technology, further traffic growth beyond 2020 will result in emissions rising. The trend in CO_2 emissions is less clear with no significant change in emissions projected between 1990 and 2020. Any improvements made in other source categories will be offset by increased road traffic emissions. Further traffic growth beyond 2020 will inevitably result in CO_2 emissions continuing to increase. These trends in emissions are reflected in the modelled trends in local air quality, using number of grid squares breaching air quality criteria as a metric. For both NO_2 and PM_{10}, local air quality is expected to improve significantly by 2010 compared to 1990. Further improvements after 2010 are limited and, with no restraint on traffic growth, the number of local hotspots may well increase beyond the period 2010-2020. Further study is required to determine the degree of exposure within these grid squares. However, at the strategic level, the results are considered useful to land use planners in identifying locations where development should be discouraged if it is likely to increase exposure to poor air quality. The 'business as usual' scenario is therefore expected to deliver improvements in local air quality, albeit with the need to consider exposure in future land use provision, but will eventually lead to an upturn in greenhouse gas emissions and hence, will not assist in meeting climate change policy objectives.

The first policy scenario (B) considered highlights the major contribution that power stations make to CO_2 emissions within Kent and Medway. Closing two of the fossil fuelled power stations, or switching to a biofuel, would reduce CO_2 emissions significantly and off set any increases in emissions from road traffic. However, in this scenario there is no significant improvement in local air quality. This scenario makes a

useful contribution to meeting climate change policy objectives but does not assist in local air quality management.

The second and third policy scenarios (C and D) consider two traffic management options. In both cases there would be a significant reduction in CO_2 emissions. Reducing HDV traffic on main roads would have less of an impact on local air quality compared to the introduction of Low Emission Zones and LDVs with zero emissions. Both these scenarios make a useful contribution to meeting climate change policy objectives and assist in local air quality management.

The fourth policy scenario (E) considers a package of measures with the aim of meeting a more stringent climate change policy target. The subsequent impact on local air quality is significant although some local hot spots may remain. These results suggest that climate change policies are likely to have a greater impact on land use and transportation planning although some local measures may be required, in terms of land use provision to minimise population exposure, to meet air quality management objectives.

This Chapter includes a final scenario (F) assuming new development to demonstrate how the tools allow for development within scenarios based on changes in the profile of emissions from existing sources. This provides an example to demonstrate how the emissions inventory and model system can be used at the strategic level to consider the potential impact of major developments. The scenario was based on an actual proposal for a major new airport in response to increased demand for air travel in South East England. The proposed airport would significantly add to NO_x and CO_2 emissions in Kent and Medway, far outweighing any benefit from other scenarios. The impact on air quality, in terms of human health impacts, would not be significantly different to the 'business as usual' scenario.

Scenario F demonstrates that any land use or transportation policies developed to meet local air quality management and climate change objectives will be sensitive to the impact of new development. One means of overcoming this would be to allow the development but only with offsetting of emissions elsewhere. If, for example, the measures included in Scenario E were adopted to compensate the increase in greenhouse

gas emissions associated directly with the proposed airport air quality would be improved across the whole of Kent and Medway.

These conclusions are based on results that do not include very localised information on population exposure. This has the potential to affect any land use and transportation policies, particularly with reference to managing emissions from main roads. However, there is considerable uncertainty associated with estimates of future land use. A more positive approach would be to implement policies to limit exposure by discouraging housing development. Such an approach is supported by the model. Accepting this, the conclusions drawn from this study are strengthened; reducing CO_2 emissions to meet long term targets will reduce population weighted exposure to poor air quality more effectively than vice versa.

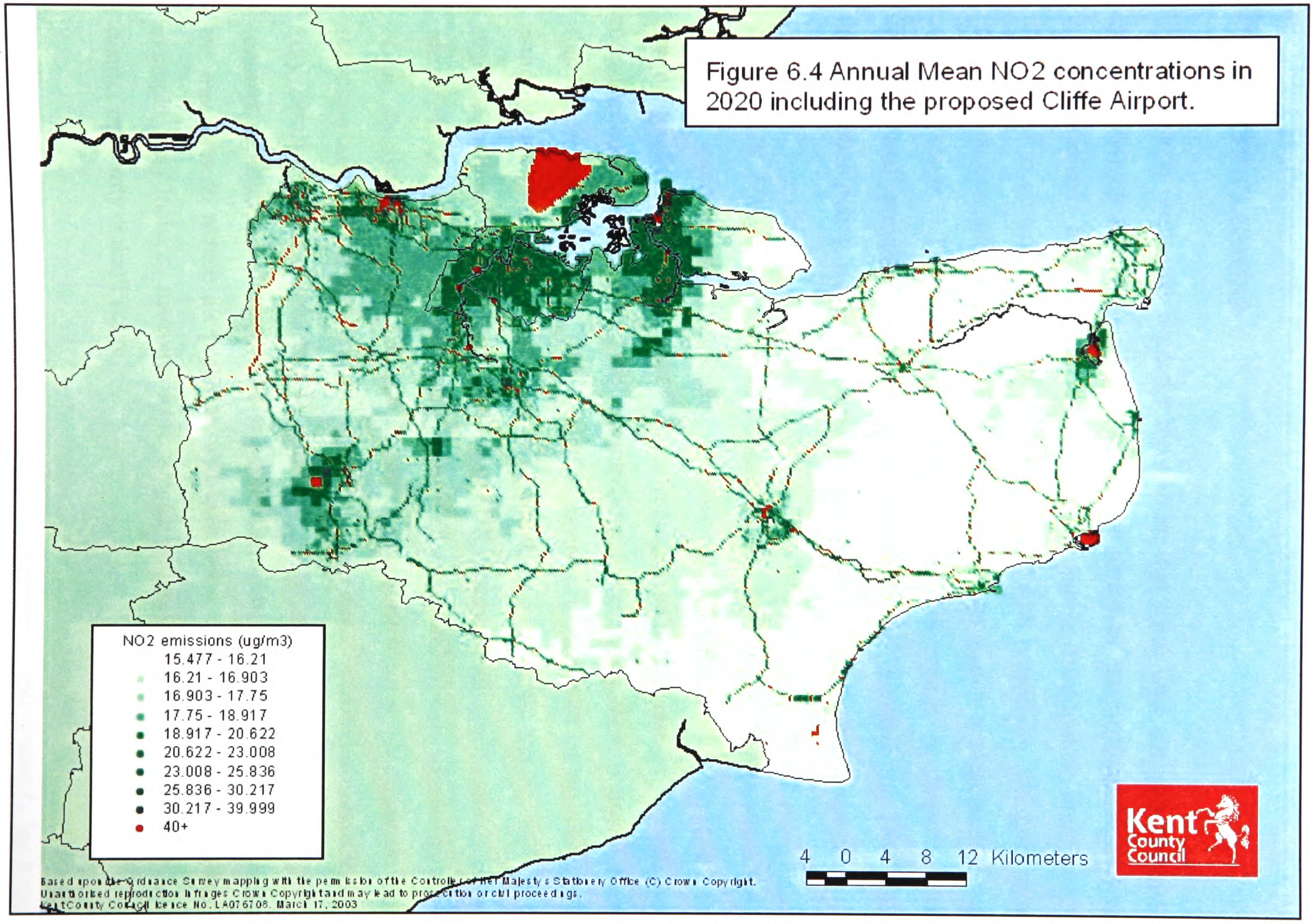


Figure 6.4 Annual Mean NO2 concentrations in 2020 including the proposed Cliffe Airport.

7 CONCLUSIONS AND FURTHER WORK

This Chapter draws together the results and discussion presented in preceding chapters and provides some conclusions and recommendations for further work.

7.1 CONCLUSIONS

This research originated from the recognition that local land use and transportation planners require practical tools for rapid assessment of the implications of strategic policy decisions. In the context of achieving sustainable development, these assessment tools need to include for a wide range of criteria with potentially divergent aims and scales. The examples used in this research are local air quality management and global climate change. The thesis has brought together diverging, or disparate strategic thinking and provides some thoughts on the future direction of land use and transportation policy, if it is to address air quality and climate change concerns.

Current UK policies on air quality management and climate change share the same origin of international commitment to sustainable development. However, despite frequent references in key policy documents and guidance, recognising the need to consider both policies in land use and transportation planning, there is little evidence of this happening in practice. The lack of assessment tools is part of the reason for this although the difference in the scale over which these policies can be applied (local, regional, national) is important. The current UK process of air quality management is to identify local 'hot spots', which may be as small as a road junction, where exposure may occur. In contrast, the UK Climate Change Programme is primarily based on emission reduction targets for greenhouse gases applied at the devolved administration level. The very detailed process described by guidance to identify local air quality 'hotspots' requires significant additional time and resources to be applied at the scale over which the measures to reduce air pollution will take place. The Local Transport Plan, for example, may includes measures to reduce air pollution at the 'hotspot' applied at the district or county council scale.

CO_2 is the principal greenhouse gas and, for the purposes of this research, has been used as the metric for assessing the impact on climate change of land use and transportation policies. Two targets for reducing CO_2 emissions have been identified.

A review of relevant air quality criteria reveals significant parity with UK Government, the European Union and the World Health Organisation. A detailed analysis of monitoring data leads to the conclusion that three of the twelve UK Air Quality Objectives are directly relevant to land use and transportation planners with other pollutants addressed by industrial pollution control.

The compilation of an emissions inventory from the 'bottom up', to allow direct linkage with land use, transportation and economic forecast data, has been undertaken for Kent and Medway. Through the use of nationally available emission factors the resultant inventory of annual mean emissions of NO_x, PM_{10} and CO_2 are consistent with other UK national and urban emission inventories. The use of a range of inventory verification tests suggests a high degree of confidence in the inventory data enabling its use for development of appropriate land use and transportation policies across the study area. A review of the activity data available within neighbouring local authorities suggests these techniques could be used to extend the inventory at least across the whole of south east England.

A modelling approach has been developed and evaluated by use of sensitivity analyses, calibration and evaluation using data from the Kent and Medway Air Quality Monitoring Network. The sensitivity analyses demonstrate the model is stable and provides an indication of the model uncertainty, which is particularly affected by meteorology. The model has been calibrated using two parameters: a roadside enhancement factor to derive roadside concentrations from area source model results; and NO_x to NO_2 conversion factors for background and roadside locations. Evaluation of the model was undertaken using a range of statistical tests confirming the model performs well compared to published work. With reference to DEFRA (2000k) guidance, the model is considered sufficient for the purposes of R&A.

The modelling approach provides land use and transportation planners with sufficiently robust tools for the evaluation of future scenarios within a practicable timescale, ensuring the number of options assessed is not unduly limited. A series of land use and transportation policy scenarios were devised to test these tools.

For the 'business as usual' scenario, emissions of NO_x and PM_{10} in Kent and Medway have declined since 1990 and are projected to stabilise between 2010 and 2020. The principal cause for this trend is improvements in vehicle exhaust emissions, with traffic growth expected to match these improvements between 2010 and 2020. The trend in CO_2 emissions is less clear with no significant change in emissions projected between 1990 and 2020; any improvements made in other source categories will be offset by increased road traffic emissions. The implication is that without further vehicle emissions control technology, further traffic growth beyond 2020 will result in emissions rising. This is reflected in the modelled trends in local air quality with significant improvements made by 2010 but limited improvement thereafter.

Focusing on single source categories is not expected to achieve both climate change policy or air quality management objectives. Closing two of the fossil fuelled power stations or switching to a biofuel for example, would reduce CO_2 emissions in Kent and Medway to almost 20% below 1990 levels alone. Unfortunately, this would not result in any significant improvement in local air quality, in terms of NO_2 and PM_{10}. The introduction of Low Emission Zones and LDVs with zero emissions would have more of an impact on local air quality than reducing HDV traffic on main roads with most benefit being seen in terms of PM_{10} concentrations. Both these traffic management scenarios would make a useful contribution towards meeting climate change policy objectives but would not be sufficient alone.

A package of measures with the aim of meeting a more stringent climate change policy target would have a significant impact in improving local air quality although some local hot spots may remain. These results suggest that climate change policies are likely to have the greater impact on land use and transportation planning although some local measures may still be required to meet air quality management objectives as well.

Using the tools to assess the implications of a new development, in this case a new airport, reveals how fragile land use and transportation policies to improve local air quality and reduce greenhouse gases are. Scale is important. However, by off setting greenhouse gas emissions directly associated with the new development by implementing policies across the study area, there is potential to realise significant air quality improvements.

This research has led to the development of suitable assessment tools and applied them to assess land use and transportation policies in the context of improving local air quality whilst reducing greenhouse gas emissions. This appears to be the first time quantified methods have been applied to assess the policy interface between air quality management and climate change in terms of land use and transportation planning. The need to develop tools in this way enabling a local policy link between air quality management and climate change to be made is now recognised by the UK Air Quality Community albeit no guidance is yet available. On behalf of NSCA, the author is preparing guidance on the inclusion of carbon dioxide emissions within atmospheric emissions inventory developed by local authorities for the purposes of LAQM.

The key conclusion from this work is that land use and transportation planning will be most affected by policies to reduce greenhouse gas emissions rather than improve local air quality management. Application of greenhouse gas emission reduction targets at the local (i.e. district) level is impractical as single sources become dominant. This research demonstrates the RCEP greenhouse gas emission reduction target could be applied at the sub regional scale (ie. county council or unitary authority) at a similar level that Local Transport Plans are implemented. Applying any emission reduction targets at this level would require careful consideration if including certain Part A Processes. If policy objectives for reducing greenhouse gas emissions are to be achieved then local air quality management may simply be an exercise to minimise exposure. An example of this may be to identify locations where certain land uses, such as housing, would not be appropriate. However, the impact of new development on such policies is important. The tools developed in this study have been used to demonstrate there is a limit to the extent of development that can be accommodated at the local to regional scale. One means of overcoming this is offsetting, where the increase in greenhouse gas emissions or reduction in local air quality due to a new development are compensated by measures implemented elsewhere. The example presented in this thesis suggests significant improvements in local air quality over a wide area could be achieved by offsetting greenhouse gas emissions associated with new development. The success of applying policy in this manner would require careful consideration of the source types involved, particularly large industrial processes. Importantly, the tools developed in this research would allow offsetting to be quantified and hence, applied in an equitable manner, enabling for its incorporation within land use and transportation planning.

7.2 FURTHER WORK

A number of areas of further work have been identified to progress this research:

- On-going maintenance and operation of the air quality monitoring network is vital to provide sufficient data to verify the emissions inventory and model results. Continuity is important as several years' data are required to account for inter year variability. However, some relocation of monitoring sites may prove beneficial to provide a more even distribution of site classes (rural, urban background and roadside) with roadside sites reflecting a range of traffic flows.

- The methodology for compiling the emissions inventory could be applied in other locations or for the whole of the UK to provide further verification. This would also provide an opportunity to further verify the dispersion models with reference to ambient air quality monitoring data from the AURN and the models developed by Stedman *et al* (2001a,b).

- A more detailed assessment of whether UK Air Quality Objectives will be achieved requires consideration of exposure. Local investigations are required to determine whether any sensitive receptors are present in grid squares where air quality is modelled as poor.

- The land use and transportation measures implicit in the assumptions made in estimating emissions for future years and the various scenarios could be tested to determine their political, social and economic acceptability.

- The tools developed in this research could be combined with other studies identifying land use constraints (e.g. flood plains) to assist in identifying potential locations for housing allocation.

- The emission inventory and dispersion model tools enable policy makers to consider offsetting to provide a quantifiable and transparent means of accommodating new development although refinement of emissions data for seaports, airports and some urban roads is required.

- Workshops could be held with land use and transportation planners to derive a whole series of scenarios to be tested using the tools developed in this research.

- The model could be developed to provide noise mapping for Kent and Medway. This would require derivation of empirical relationships between land use categories, as defined by the emissions inventory, and ambient noise levels.

In addition to these specific research issues, the products of this research is now available to evaluate more fully Structure Plan and Local Transport Plan policies. Moreover, there is no technical reason why these tools could not be used by the Government Office of the South East to consider such policies applied regionally.

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APPENDIX : ENVIRONMENTAL POLICY AND GOVERNMENT

A.1 REDUCING NATIONAL EMISSIONS

The UK Government has signed a number of agreements to reduce national emissions of sulphur dioxide, nitrogen oxides and volatile organic compounds. The national reduction targets for these pollutants are summarised in Table A.1 below. The GHG emissions reduction targets are included for comparison.

In the mid 1990s, the DoE (1996b) suggested future emissions of sulphur dioxide would be reduced as a result of:

- continued switching from industrial burning of coal or oil to gas
- completion of the installation of flue gas desulphurisation plant to coal-fired power stations
- continued improvement in emissions required by Local Authority Air Pollution Control and Integrated Pollution Control.

The UK Government admitted such measures may not, in themselves, be sufficient to meet the 2010 target of an 80% reduction. Reductions in the sulphur content of industrial and motor oil have occurred through the implementation of 93/12/EC (Sulphur Content of Fuels Directive) and more are expected with the implementation of 99/32/EC (same subject). The reluctance of the UK Power Industry to retrofit flue gas desulphurisation (FGD) units when industry specific sulphur reduction targets and the UK Air Quality Objective can be achieved without them suggests this target is even less likely to be achieved [i]. For example, retrofitting FGD to a coal fired unit can effectively reduce sulphur emissions from that unit by 90%; more than required to meet the annual emission limits set by the Environment Agency (MM, 2000). Operators could choose to use international coals, with perhaps 50% lower sulphur content than UK coals, and reduce unit load to achieve a similar result, as demonstrated by Fisher and Acres (2000). This second approach was recognised by the Environment Agency (1999) in a consultation paper which moved away from the original requirement for all operators to retrofit FGD to one where alternatives could be considered but measures would be put in place to encourage FGD.

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i Of the 13 coal fired power stations operating in the UK only two have been installed with FGD to date with a further two being installed in the near future. In compiling the latest energy statistics, the DTI (2000) assumed no further FGD units will be installed. With subsequent closure of unabated plant, SO_2 emissions are projected to reduce from 718 – 725 ktonnes in 2000 to 191 – 237 ktonnes in 2005

Table A.1
International Commitments for Reducing Atmospheric Emissions in the UK

Year	SO_2	NOx	Particulates	VOCs	GHGs
				-	-
1980	base year	2416 kT	-	-	-
1988	-	-	base year	-	-
1990	-	-	-	-	base year
1993	70% of 1980	-	-	64% of 1988	-
-	levels	-	-	levels	-
1994	-	Freeze to 1980	-	-	-
-	-	levels	-	-	-
2000	50% of 1980	70% of 1980	-	-	Stabilise at
-	levels	levels	-	-	1990 levels
-	-	-	-	-	-
2005	30% of 1980	-	-	-	-
-	levels	-	-	-	-
-	-	-	-	-	-
2007	-	-	-	43% of 1990	-
-	-	-	-	levels	-
-	-	-	-	-	-
2010	20% of 1980	1181 kT	-	1200 kT	87.5% of 1990
-	levels	-	-	-	levels
-	(625 kT)	-	-	-	-

(after: Murley(2000))

Although large industry represents by far the greatest source of sulphur dioxide (approximately 67% of the UK total in 1995 was from power generation alone: Salway *et al*, 1997) the impact of these sources, in terms of number of people affected may only be as significant as sources located in urban areas. Such sources include domestic burning of smokeless fuels, coal and oil, smaller industrial processes, waste oil burners and to a lesser extent, road traffic. Emissions from all of these disparate sources will be reduced as implementation of 99/32/EC takes place. The UK strategy for reducing emissions of sulphur dioxide principally addresses large industrial sources. As described by Laxen (1996) the effect of this central government policy on local air quality will be the continued reduction in background concentrations and reduction in the frequency of short term peak concentrations due to plume grounding.

In urban areas with no large industrial sources, the effect of the 1996 national sulphur strategy will be minimal. Although emissions of sulphur dioxide have reduced dramatically since the Clean Air Act 1956 levels may still be sufficiently high to lead to short term breaches of the relevant air quality criteria. Such breaches typically occur

and even further to 60 – 97 kTonnes by 2020.

during still winter days with a low level inversion layer trapping emissions near to ground level. The further reductions in the sulphur content of oil expected will reduce this potential. In an update of Government road fuel levy policy, Holman (2001) suggests current reductions in road traffic emissions of sulphur dioxide occurred as a side effect of measures to reduce particulate emissions and, with respect to road transport emissions, increasing the effectiveness of three way catalytic converters, by reducing the sulphur content of petrol and diesel. Further sulphur reductions are expected following agreement by European car manufacturers to reduce new light duty vehicle carbon dioxide emissions by approximately 25% over the period 1996 to 2008; lean burn engines emit more nitrogen oxides and hence will require advanced techniques for its removal, such as selective catalytic reduction, which is more sensitive to sulphur.

Through a number of EU Directives, the Acidification Strategy and the Convention on Long Range Transboundary Air Pollution, the UK Government is committed to a national reduction in emissions of nitrogen oxides from certain large industries such as power stations and other sectors such as road transport (DETR, 2000b; pp24-26). The EU has proposed a target of a 30% reduction by 2000 (using 1980 as a baseline). The impact of increased road traffic emissions suggests that without some means of reducing actual vehicle numbers this target will not be achieved. Observed levels of nitrogen dioxide across London increased between 1993 and 1995, despite the growing use of catalytic converters (see Figure A.1). There is a large scatter in individual results (as maximum and minimum monthly mean observations for all sites) and the year on year variations implying some degree of caution in assuming any trend over the whole period. Despite a slight reduction in nitrogen dioxide concentrations between 1994 and 1998, the observed trend from 1990 to 1999 suggests no discernible improvement in air quality.

Despite this example of a lack of apparent improvement in air quality, the process for reducing national emissions in the UK is well established and, to date, successful. The principles for reducing atmospheric emissions, including GHGs, are therefore not unknown in the UK and many of the policy and legislative mechanisms used for reducing 'local air quality' pollutants are available for reducing GHGs.

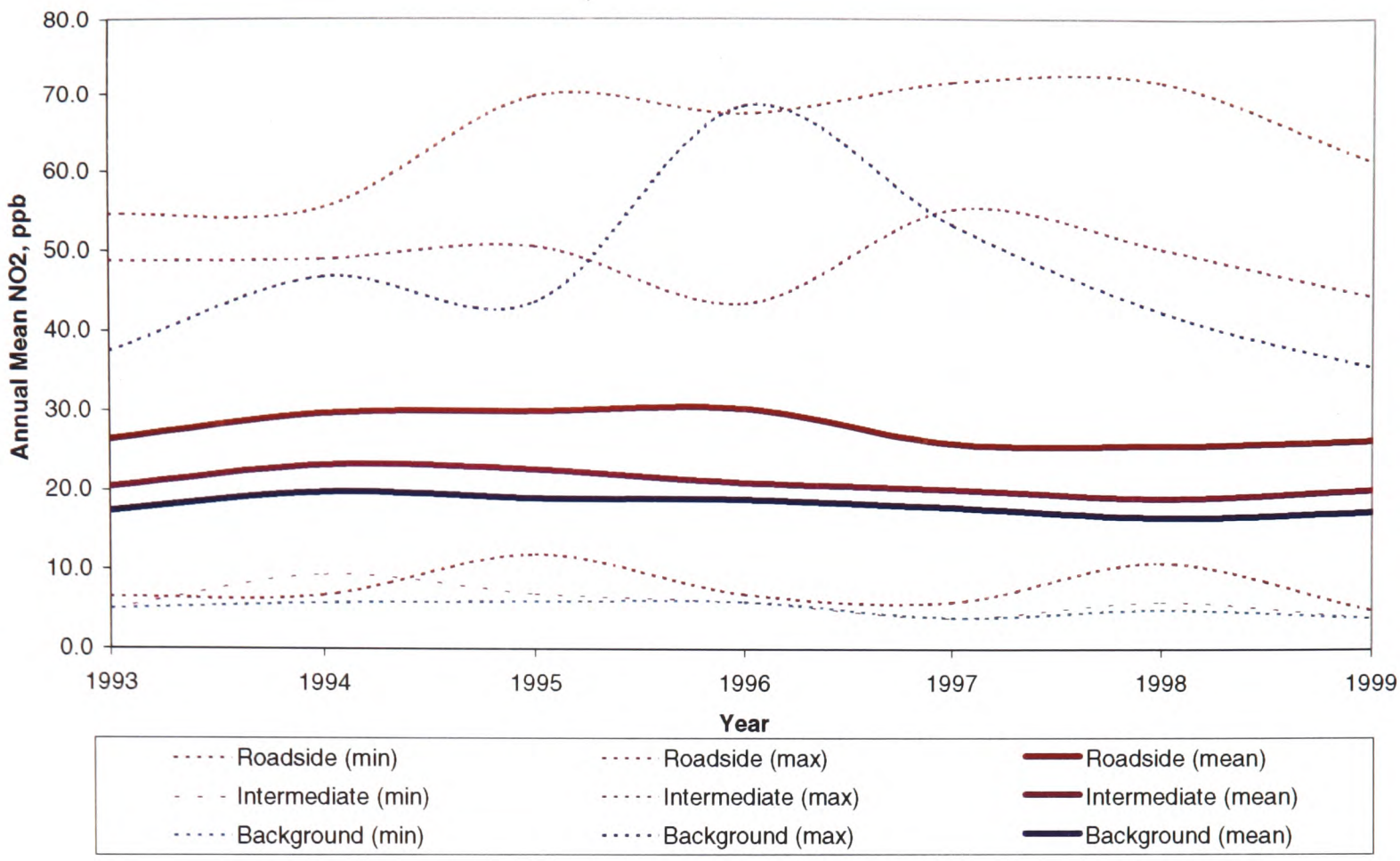

Figure A.1: Observed Trends in Annual Mean Concentrations of Nitrogen Dioxide in London 1990 to 2000

(Source: compiled from observation data from over 80 monitoring stations, National Air Quality Archive, DETR)

A.2 THE AIR QUALITY STRATEGY FOR ENGLAND, SCOTLAND, WALES AND NORTHERN IRELAND

In the mid 1990s, the DoE (1994b) set out proposals for introducing legislation:

- requiring periodic reviews of air quality by local authorities
- provide for the establishment of air quality management areas in those places where national air quality targets are not met
- placing powers and obligations on local authorities and other relevant bodies to prepare plans for remedying air quality problems in Air Quality Management Areas
- securing effective co-ordination of all activities which can influence air quality improvement in the most cost effective manner in those areas where it is most needed.

These proposals, and the implied technical processes required, were considered by Elsom and Crabbe (1996) in the context of air quality management procedures practised elsewhere. This review concluded the UK approach is not unique and builds upon best developed techniques. Elsom and Longhurst (1997) further developed the argument for co-ordinating the various bodies responsible for land use, transport and energy policies in addition to those responsible for the assessment of air quality. This aspect of co-ordination within local government is the subject of research elsewhere[ii] with an established monitoring programme based on regular questionnaires sent to local authorities co-ordinated by the University of the West of England (Woodfield, 2001, Woodfield *et al* 2001).

The appropriate legislation was formally enacted under Part IV of the Environment Act 1995 (HM Government, 1995) of which:

- **s80** places a duty upon Central Government to publish a National Air Quality Strategy (NAQS)

- **s81**[iii]

- **s82** places a duty upon each UK local authority to carry out a review of air quality from time to time. This review should identify whether air quality standards and objectives are being achieved. Moreover, this review should identify areas where air quality standards are not being achieved and where air quality objectives are not likely to be achieved

- **s83** requires by order that a local authority designate as an air quality management area (AQMA) any part of its area in which it appears that the standards or objectives are not, or are unlikely to be, achieved. Such an order can be revoked following a subsequent review which indicates otherwise

- **s84** refers to the duties for local authorities in designated areas. Within twelve months of declaring an AQMA, the local authority is required to have completed a report of the likely extent of non-achievement of air quality standards or objectives. In addition, the local authority is required to prepare an Air Quality Action Plan, including a schedule for implementing the measures within it. For two tier local authorities, this Air Quality Action Plan needs to be agreed with the relevant County Council, with reference to the Secretary of State where necessary.

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ii See, for example, Longhurst *et al* (1996) Crabbe *et al* (1999) Elsom *et al* (2000) Beattie and Longhurst (2000) and Ing *et al* (2001).

iii Although not directly relevant to this discussion, s81 of the 1995 Act places an onus on the Environment Agency to consult with local authorities when reviewing and authorising IPC licences (and PPC Permits). This effectively means if local authorities can identify a large industrial operator as the main cause for not meeting an air quality objective then the responsibility for meeting its duties under s84 can be passed on.

Central government fulfilled its current duties, relating to air quality, under Part IV of the 1995 Act in publishing the NAQS and associated guidance for local authorities (DoE, 1997a; DETR, 1997a-e and DETR, 1998 a-d). The NAQS and air quality objectives were reviewed by the DETR and updates published in January 2000 (DETR, 2000b; note the new title to reflect devolution; and 2000c). This update focused on the revision of air quality objectives required for subsequent alignment with the EU Air Quality Daughter Directives and the implications for meeting these objectives (a detailed discussion of air quality standards and objectives is included in Chapter Three). The AQS review led to a subsequent revision of the associated guidance (DETR, 2000d-l) including detailed guidance for the assessment of PM_{10} with reference to the EU Stage 1 Limit (Moorcroft *et al*, 1999) and further draft guidance related to the subsequent review and assessment following declaration of an AQMA, as required under s84 of the 1995 Act (DETR, 2001b).

Personal discussions with officers suggests most local authorities followed the guidance to the letter as a demonstrable form of defence should Air Quality Management Areas (s83) be declared. This anecdotal evidence appears to be supported by more formal questioning by Beattie *et al* (2001) who go on to observe that, in practice, the subsequent revision of guidance has caused some difficulties. The DETR did not publish further guidance within this round of review and assessment (see section 2.2.3) and was reliant on the NSCA (1999; 2000a-b, 2001) to produce supplementary guidance.

A.3 CLIMATE CHANGE POLICY

Under the Climate Change Convention, an ongoing programme of conferences was established to agree how the Convention would be achieved in practice, recognising the uncertainty in the science and the need for international co-operation. Emerging scientific evidence quickly established the need for actual reduction in GHG emissions below 1990 levels. This led to the Kyoto Summit, held in December 1997, and the Kyoto Protocol with legally binding targets to reduce GHG emissions from developed countries to 5.2% below 1990 levels by the end of the period 2008-2012. The European Community is committed to reducing GHG emissions to 8% below 1990 levels. In June 1998, as part of

European negotiations, the UK Government agreed to a legally binding target of a 12.5% reduction (Grubb *et al,* 1999).

The DETR is funding research in to the impacts of climate change as the UK contribution to ongoing discussions on interpreting Annex 2 of the UN Framework Convention on Climate Change. In the third of its annual reports, the Hadley Research Centre at the UK Met Office (1997, 1998, 1999) considered the climate impacts of three CO_2 emission scenarios :

- 'business as usual' (ie. no change in current upward trends in emissions)
- stabilisation of CO_2 emissions to 750 ppm (equivalent to approximately three times pre Industrial levels)
- stabilisation of CO_2 emissions to 550 ppm (equivalent to approximately twice pre Industrial levels).

The Met Office predictions of climate change described in the 1999 report were made using an improved climate model with a much higher resolution (1.25° latitude x 1.25° longitude) than before. Ocean currents, such as the Gulf Stream, were now better represented. This resulted in the 'flux corrections' between atmospheric and oceanic sub models used in previous versions of the climate model, which led to unrealistic future trends, being removed. The revised model also includes the cooling effect of sulphate aerosols, and the reduction of this effect expected with reduced sulphur emissions expected in the future (see Table A.1 above).

Only limited model validation has been reported to date although '*initial results indicate climate models can simulate reasonably well the climate change of the past 140 years, providing confidence in predictions of the future*' (Met Office 1998, p2). Unfortunately, no measure of uncertainty is given although the report states that comparisons of model simulations and observations indicate that human-made greenhouse gases have contributed substantially to global warming over the past 50 years. Moreover, the model is considered weak in terms of positive and negative feedback mechanisms. The only conclusion to drawn from this is that, unless there is a reduction in GHG emissions, significant changes in our climate will continue and are likely to accelerate, ie. the scale and magnitude of these impacts cannot be precisely predicted. The Inter-Governmental Panel on Climate Change (IPCC) 'business as usual' scenario for increases in

greenhouse gas emissions result in a further warming of about 3°C over the next 100 years. Increases in greenhouse gases result in a slowing down of the North Atlantic ocean circulation, but even with this, Europe still warms (see Table A.2 overleaf).

Despite stated reservations in predicting the scale and magnitude of impacts to any detail, this global scale research was used by Wade, Hossell and Hough (1999) as the basis for considering the expected climate change impacts to the southeast region of the UK, incorporating initial results from limited *regional* scale modelling by the Hadley Centre. In this context, the regional scale included the majority of England, extending west beyond Bristol and north beyond Leeds. The authors provide sufficient caution to these regional level predictions to suggest even finer resolution (ie. to actual Government Office regions) would not generate meaningful data. Climate impact modelling, therefore, has progressed to providing predictions of future impacts at national level, sufficient to enable assessments of each country making up the UK but not to any finer resolution. The key findings to this more localised scoping study for the future, which includes Kent, are:

- rising sea levels will leave many coastal and estuarine zones prone to flooding and coastal erosion

- greater seasonal variation in climate and a change in the pattern of extremes will occur. For example, wetter winters and hotter, drier summers leading to an imbalance in water resource and demand

- there is a potential for a shift in agricultural towards more 'temperate' crops

- increased migration and settlement of flora and fauna from temperate zones, including malaria carrying mosquitoes.

These findings, endorsed by organisations including the Government Office for the South East, relevant county and district councils, the Environment Agency, the Met Office, ADAS and Thames Water, lead to a conclusion that some climate change will occur with consequent changes in lifestyle inevitable.

Table A.2 : Climate Change and its Impacts

Impact	Emissions Scenario		
	Business as Usual	**CO_2 Stabilised at 750 ppm [1]**	**CO_2 Stabilised at 550 ppm [1]**
Mean Global Temperature [2]	+2°C by 2050s +3°C by 2080s	+2°C by 2100s +3°C by 2230s	+2°C by 2150s +2°C by 2230s
Global Sea Level	+40 cm by 2080s	delayed by 25 years	delayed by 40 years
Natural Ecosystems	By 2080s: substantial dieback of tropical forests and grasslands, especially in S America and southern Africa; considerable growth of forests in N America, northern Asia and China. CO_2 sequestration increases during 21 century. This sink is lost in 2070s due to forest dieback.	Dieback delayed by 100 years. CO_2 sequestration increases during 21 century; sink is lost in 2170s due to forest dieback.	Dieback substantially reduced, even by 2230s. CO_2 sequestration increases during 21 century; sink eventually lost due to forest dieback.
Availability of river water	By 2080s: substantial decreases in Australia, India, southern Africa, most of S America and Europe, and the Middle East. Increases in N America, Asia (esp. central Asia) and central eastern Africa.	Rate of change in water availability generally slowed by 100 years.	Rate of change in water availability generally delayed further.
Water resource stress [3]	By 2050s: 2.3-3.2 billion people experience an increase in water stress. By 2080s: 3-3.6 billion people with increased water stress.	2.2 billion people experience an increase in water stress. Relatively little change from 'business as usual' scenario.	1.7 billion people experience an increase in water stress. 0.9 billion people experience an increase in water stress.
Agriculture [4]	By 2080s: cereal yields increased in N America, China, Argentina and much of Europe, reduced in Africa, Middle East and particularly India.	Fewer regions experience negative impacts; Africa and India adversely affected. Production increased in parts of central Asia.	Fewer regions experience negative impacts; Africa and India still adversely affected Parts of S America worse off.
Coastal Effects [5]	By 2080s: number of people flooded increased from 13 million to 95 million; 60% in S Asia and 20% in S E Asia.	Number of people flooded increased from 13 million to 34 million; mainly in southern Asia and S E Asia.	Number of people flooded increased from 13 million to 19 million; mainly in southern Asia and S E Asia.
Human Health	An estimated 290 million additional people will be at risk of *falciparum* malaria [6] in 2080s; greatest increases in China and central Asia. Estimated reduction in winter related mortality exceeds summer heat related mortality in many temperate zone cities.	Estimated additional people at risk reduced to 255 million; greatest increases also in China and central Asia. Effect of reduced winter related mortality moderated.	Estimated additional people at risk reduced to 175 million; greatest increases also in China and central Asia. Effect of reduced winter related mortality moderated.

Notes:
[1] 750 ppm CO_2 equivalent to three times pre Industrial levels; 500 ppm equivalent to twice pre Industrial levels
[2] Land areas warming twice as fast as oceans. Large changes (+ve and -ve) in precipitation.
[3] Defined as those countries using more than 20% of national annual average run off.
[4] The food system may be expected to accommodate regional changes at a global level; Africa and India at risk.
[5] In all scenarios; compounded decline in coastal wetland areas; progressive impact on coastal low-lying areas.
[6] *falciparum* malaria is clinically more dangerous than the more widespread *vivax* malaria.

(Sources: Met Office, 1997, 1998, 1999, RCEP 2000 and DETR 2000a).

A.4 MEASURES IN THE CLIMATE CHANGE PROGRAMME

The road fuel duty escalator was introduced in the April 1993 Budget with the aim of increasing the cost of road fuel year on year at a rate of 3% above inflation to encourage fuel economy, either by choice of vehicle (smaller engines) choice of transport or driving habits. This escalator was later increased to 5% and, with the change in government in 1997, further raised to 6% coupled with a differentiation in road tax to encourage smaller engines. The road fuel duty escalator was abandoned in 1999 (HM Treasury, 2001) due to political pressure; providing a valuable message to government of how difficult the task will be to influence a significant change in the way we live our lives if climate change is to be addressed [iv].

The Climate Change Levy on the business end user was effected from 1 April 2001 following recommendations made by Lord Marshall. This policy measure is expected to save the equivalent of 1.5 MtC per annum by 2010. The Emissions Trading Scheme has developed via a business led initiative in response to this Levy (UKETG, *online*). In developing this scheme, a series of guidelines for companies to calculate GHG emissions was required. These guidelines are discussed further in Chapter 3 [v].

The 10% renewables target effectively extends the objectives of the Non Fossil Fuel Obligation (NFFO) programme. With the introduction of the New Electricity Trading Arrangements (NETA) (HM Government, 2000b) all electricity suppliers are required to meet targets for the proportion of electricity they purchase derived from renewable forms of generation. The target for 2010 is 10%. Electricity suppliers will be required to hold sufficient renewable tickets, obtained either by purchasing from a renewable generator or via market trading, to meet this target. Suppliers who do not hold sufficient tickets will be required to purchase from the government at a pre-determined minimum price. Revenues raised by the government will be used to support emerging renewable technologies. In theory, this system will enable government to increase the minimum price renewable generators can obtain for electricity and hence, encourage the introduction of newer technologies (DTI, 1999).

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iv A similar observation could be made regarding the future implementation of Air Quality Action Plans.

v The CCP (DETR, 2000a; p36) refers to the development of a methodology for preparing regional GHG inventories; not available during the course of this research.

Nuclear power stations currently generate 26% of UK electricity. The first stations are due for closure in 2005. The large capital cost of nuclear power prohibits private financing of new build and, given the public resistance to nuclear power and recent performance of the UK nuclear industry, there is little prospect of government funded projects. The 10% renewables target alone will not compensate for this loss in power generation. A balance in shifting power generation towards more efficient plant, increasing renewable energy and reducing demand will be required. The first two could be achieved through increased competition and changes to the tariff, as indicated above. The easiest means of reducing demand is to increase energy costs, such as introducing a carbon tax, for example. A review of implementing environmental taxes undertaken by the European Environment Agency (1996) highlights the need to carefully consider the potential impacts on competitiveness and low income groups. This is echoed in the CCP and recognised, for example, in the implementation of the Climate Change Levyvi: energy intensive industries will be eligible to some form of tax relief; the revenue will be 'recycled' back in the form of reduced national insurance contributions; and the domestic sector is exempt.

A.5 CENTRAL GOVERNMENT

UK legislation and national policy regarding air quality is developed within Central Government level often as a response to European Union Directives. The principal Departments with a role to play in air quality management are: the Department of the Environment, Food and rural Affairs (DEFRA); the Department of Trade and Industry (DTI); and, to a lesser extent, the Department of Health (DoH) as well as the Welsh Office, the Scottish Office and Northern Ireland Office.

Central Government responsibilities with reference to air quality management and climate change include:

- compliance with international commitments to reducing emissions of a number of pollutants including NO_x, SO_2, VOCs and GHGs
- compliance with a number of EC air quality and national emissions reduction related Directives
- compliance with EC Directives related to road vehicular emissions
- compliance with the Integrated Pollution Prevention and Control Directive

vi Emissions trading could, in theory, provide a means of financing carbon off setting projects such as public transport provision or AQAPs. This is being considered further by Kent County Council in the development of energy policies within the Structure Plan.

- maintaining a [National] Air Quality Strategy
- responsibility for the provision and maintenance of primary roads
- publishing a national plan including targets for road traffic reduction.

The impact of devolving administrations in Scotland, Wales and Northern Ireland is not considered to be significant with respect to national government (ie. non-local authority) responsibilities as both the AQS and CCP refer to agreements reached for the relevant policies, measures and responsibilities to be upheld by each devolved administration. The conclusions reached in this research, whilst based on an area of England, are considered to be equally applicable elsewhere.

The legal responsibilities of Central Government, with respect to local air quality management or reducing GHG emissions, are far reaching. The AQS stems from the EC Directive 96/62/EC and the principal target included in the CCP is expected to be ratified in 2002.

A.6 REGIONAL ASSEMBLIES

The UK has been devolved in to individual countries and the Greater London Assembly (GLA) (DETR (*on line*) and HM Government, 1999) and a number of directly elected Regional Assemblies formed to co-ordinate economic development, transport and planning by providing a regional framework for the preparation of local authority development plans using a 15 to 20 year horizon. Unlike the GLA, the Regional Assemblies do not have legislative powers at present. In a review of EU member state government structure and planning policies, Balchin and Sýhora (1999) describe the UK as a *devolving unitary state*. As considered further in section 2.6, the devolution process is expected to continue and may lead to Regional Assemblies in the UK eventually attaining legislative powers.

DETR proposals for amending regional planning guidance include reference to: undertaking environmental and sustainable appraisals of regional development plans with specific reference to the GHG emissions reduction target and the AQS (DETR, 1999b; paragraphs 2.30 and 13.2, respectively).

The adoption of *Regional Air Quality Strategies*, required by the GLA, promoted by the NSCA (2001) and considered further by Beattie *et al* (2001a,b; see section 2.6) suggests they may be a role for Regional Assemblies in air quality management.

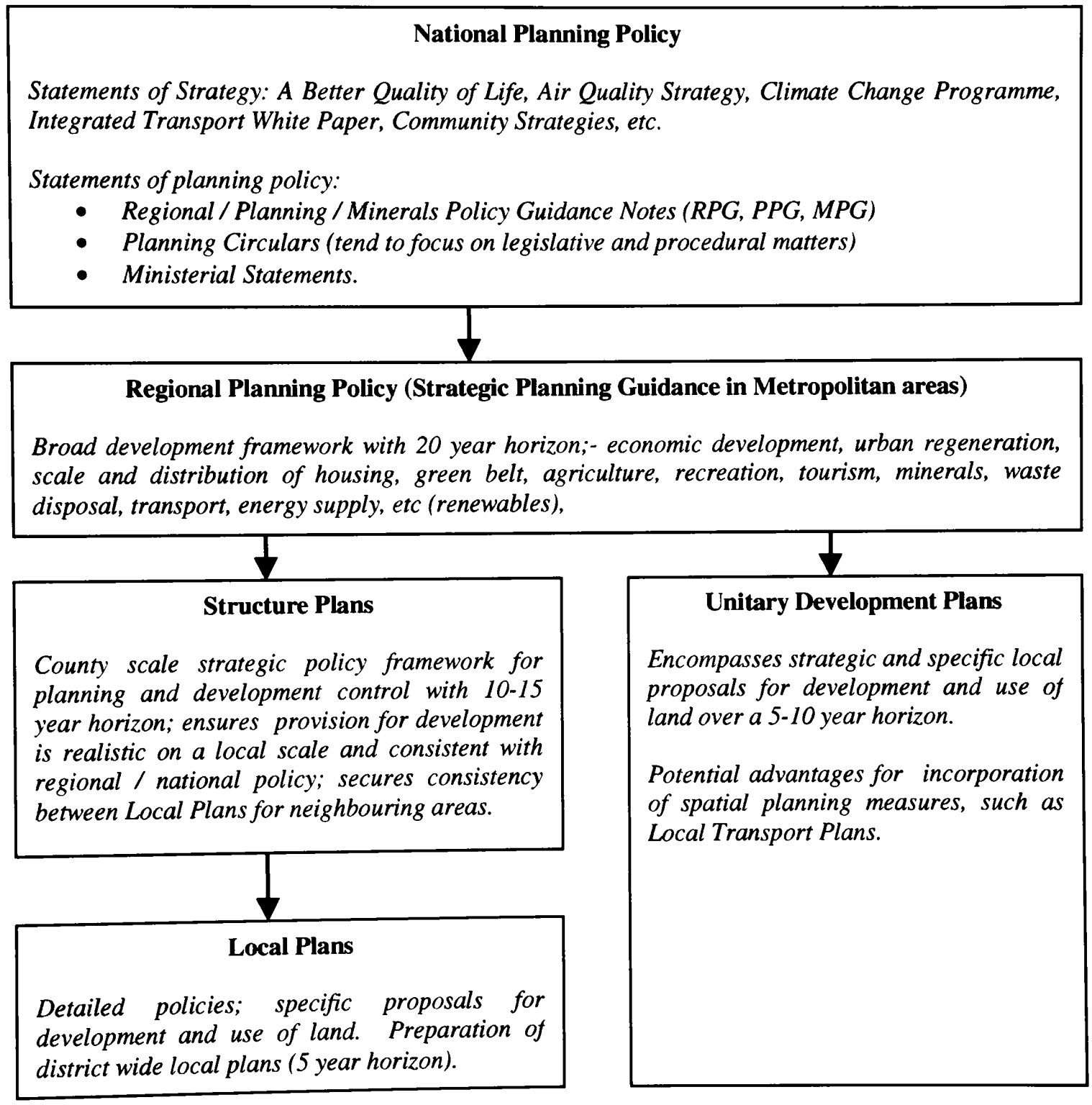

Figure A.2 : UK Planning – The Plan-Led System

(*after*, DETR (*on line*))

A.7 COUNTY COUNCILS

Following the reorganisation of local government completed in April 1998, there are now 34 county councils in England. Their primary role is to provide regional government. The functions of county councils relevant to local air quality management are as follows:

- maintenance of the Structure Plan
- maintenance of the Waste and Minerals Local Plans
- responsibility for the provision and maintenance of non-primary roads
- maintenance of a Transport Management Strategy (Local Transport Plan)
- statutory consultee for local authority air quality reviews and assessments, and air quality action plans
- statutory consultee for major planning applications and Integrated Pollution Control Licence applications
- provision of education and library services.

Although county councils have a statutory duty as consultees to local authority air quality reviews and assessments (principally in providing information) they have no specific requirements to undertake reviews and assessments themselves. Perhaps the most important role for county councils in air quality management is in the co-ordination of activities undertaken by individual district councils, such as the co-ordination of rural public transport to relieve congestion by out-of-town commuters and shoppers in urban areas. This co-ordinating role may either be achieved using the existing development plan system or local and regional transport plans.

County Councils have no legal obligations, beyond the provision of information, with respect to local air quality management or reducing GHG emissions although, if preparing LTPs, guidance does refer to improving local air quality and reducing GHGs.

A.8 DISTRICT COUNCILS

The 237 district councils in England generally represent the next tier of local government below county councils. The Environment Act 1995 specifically places the responsibility for undertaking air quality reviews upon local authorities at the district council level.

The Statutory duties of district councils with respect to air quality can be summarised as follows:

- Part B Process licensing authority [vii]
- maintenance of Part B Process Register
- enforcing statutory nuisance legislation
- enforcement of the Clean Air Act 1993, including regulation of smoke control zones
- planning authority for new development
- maintenance of Local Plans
- review and assessment of air quality
- declaration of Air Quality Management Areas, if required
- Implementation of Air Quality Action Plans.

In addition, district councils have discretionary duties to undertake air quality monitoring and maintain a register of Part A Processes in their area.

District councils are legally obliged to do all that is fair and reasonable to ensure the UK Air Quality Objectives are met within their area [viii] but are not directly obliged to reduce GHG emissions beyond complying with guidance for the preparation of LTPs. They are required however, under the Home Energy Conservation Act 1995, to report on how significant energy savings can be made in the domestic sector. This has the potential to contribute directly to reducing GHG emissions.

A.9 METROPOLITAN DISTRICT COUNCILS

There are a number of large urban conglomerates in the UK, including London, the West Midlands and Greater Manchester, which are locally governed by a total of 36 Metropolitan District Councils (MDC) and the 32 London Boroughs. These councils carry out the functions of both county council and district councils. Their roles and responsibilities with respect to air quality will be as described for district and county councils above. Cullingworth and Nadin (*ibid*) suggest MDCs (and Unitary Authorities) represent a streamlined approach to the provision of services to large numbers of people in relatively small areas. With respect to local air quality management, encompassing diverse

vii Part B Processes as per the Environmental Protection Act 1990; to be replaced by A2 installations under the Pollution Prevention and Control (England) Regulations 2000 (HM Government, 2000a)

viii But are not obliged to demonstrate compliance, if the action required is beyond their control.

responsibilities such as monitoring and modelling of air pollution, regulation of industrial emissions, transport and land use planning, this is evidently an advantage.

The legal obligations for MDCs, with respect to local air quality management and reducing GHG emissions are the same as for district and county councils.

A.10 UNITARY AUTHORITIES

The Local Government Act 1994 (and Local Government etc (Scotland) Act 1994) (HM Government, 1994) set way for the establishment of 34 unitary authorities. Such authorities, usually formed by the agglomeration of two or three neighbouring district authorities [ix] perform the same functions as both district and county councils. Their air quality roles and responsibilities are as described above for MDCs.

ix In Scotland, the 53 District and 9 Regional Councils have been replaced by 29 Unitary Councils with the 9 Island Councils remaining unchanged. In Wales, the 8 County and 37 District Councils have been replaced by 22 Unitary Councils.