AIR QUALITY EXPERT GROUP

Particulate Matter in the United Kingdom

Prepared for:
Department for Environment, Food and Rural Affairs;
Scottish Executive; Welsh Assembly Government; and
Department of the Environment in Northern Ireland

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Terms of reference

The Air Quality Expert Group was set-up in 2001 to provide independent scientific advice on air quality, in particular the air pollutants contained in the Air Quality Strategy (AQS) for England, Scotland, Wales and Northern Ireland and those covered by the EU Directive on ambient air quality assessment and management (the Air Quality Framework Directive). AQEG replaces the Airborne Particles Expert Group, who published their report on Source apportionment of airborne particulate matter in the UK in January 1999.

AQEG reports to the Secretary of State for Environment, Food and Rural Affairs, Scottish Ministers, the National Assembly for Wales and the Department of the Environment in Northern Ireland (the Government and Devolved Administrations). AQEG is an advisory non-departmental public body in England, Wales and Northern Ireland. In terms of the Scotland Act 1998, the Group is a jointly established body.

AQEG's main functions are:

- to give advice to ministers on levels, sources and characteristics of air pollutants in the UK;
- to assess the extent of exceedences of Air Quality Strategy objectives and proposed objectives, EU limit values and proposed or possible objectives and limit values, where monitoring data are not available;
- to analyse trends in pollutant concentrations;
- to assess current and future ambient concentrations of air pollutants in the UK; and
- to suggest potential priority areas for future research aimed at providing a better understanding of the issues that need to be addressed in setting air quality objectives.

The Group will not give approval for products or equipment.

Further information on AQEG can be found on the Group's website at:
http://www.defra.gov.uk/environment/airquality/aqeg/index.htm

Information on these pages includes the dates, agendas, and minutes of meetings as they become available, a list of the members, the Register of Interests and draft and final reports as they become available.

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Executive summary

When commissioning this report, Defra and the Devolved Administrations asked the Air Quality Expert Group (AQEG) a number of questions that are shown, in abbreviated form, in Figure 1. AQEG's answers to these questions, together with a rationale for each answer, are provided in Chapter 9. This executive summary provides a less detailed overview of the report and does not attempt to answer the questions explicitly.

Figure 1. Questions on particulate matter set by Defra and the Devolved Administrations for AQEG.

- Are the current assessment methods fit for purpose? How could they be improved?
- Are there sources missing from UK other European emissions inventories?
- Is the UK likely to achieve, with current abatement measures and technologies, the European Union limit values and the Air Quality Strategy objectives for PM_{10}? If not, why not? What levels of PM_{10} are likely to be achieved by current measures and policies?
- Will the UK be able to meet the range of targets for $PM_{2.5}$ as proposed in the draft CAFE Position Paper on Particulate Matter?
- What are the practical maximum feasible reductions of PM_{10} and $PM_{2.5}$ concentrations at hotspots and urban background, for example central London locations.
- Where and what are the main source contributors to current and future concentrations of PM_{10} and $PM_{2.5}$? What are the contributions of different sources to forecast exceedences of the EU limit values and UK objectives?
- What are the potential sources of abatement and types of measures to reduce particle concentrations at hotspots, at urban background, central London and across the whole country? What role can local/national/EU-wide measures play in meeting targets? Are there alternatives to emissions reduction?
- A number of recent studies have highlighted the health effects of certain components of particulate matter. Where further abatement techniques are known, how might they specifically affect reduction of the different particulate matter metrics and chemical components?
- What have we learned from the measured data on ultrafines, including source apportionment? Are the observed trends real? What fraction of ultrafine particles volatilise?
- How does the UK source apportionment for PM_{10}, $PM_{2.5}$ and other metrics compare with other modelling in Europe? Is road traffic more important than current models show? How is the coarser fraction accounted for?

- Can we explain the trends in measured PM_{10}, sulphur and black smoke since 1992?
- What are the differences between strategies that address hotspots of exceedence and those that aim to reduce population exposure? Should policy evaluation consider impacts on population exposure as well as concentrations at specific locations?

Properties of particulate matter

Atmospheric particulate matter consists of solid or liquid matter in sizes that range from a few nanometres (nm) in diameter to around 100 micrometres (100 µm). Its chemical composition includes sulphates, nitrates, ammonium, sodium chloride, elemental and organic carbon and a range of minerals. It contains both primary components, emitted directly into the atmosphere, and secondary components formed in the atmosphere by chemical reactions. The metric generally employed for particles in the UK is PM_{10}, which, to a good approximation, describes the mass of particles with a size of less than 10 µm diameter; similarly $PM_{2.5}$ describes the mass of particles with a size of less than 2.5 µm diameter. An older, but still useful, metric measures the blackness of particulate matter and is termed black smoke.

The balance of evidence currently available suggests that it is combustion-derived components of which PM_{10} that are primarily responsible for the harmful effects on human health. These components are comprised predominantly of fine (less than 2.5 µm) and ultrafine (less than 100 nm) carbon-containing particles and may be enriched with trace metals or specific organic compounds. There is generally less evidence to connect secondary inorganic particulate matter and coarse particles with adverse health effects. However the latter, in particular, cannot be ruled out since certain sources of these particles may be enriched with components of putative high risk (for example, soluble trace metals). The coarse fraction also contains biological material such as pollen and may be proportionally enriched with endotoxin, both of which factors can lead to adverse health effects.

Objectives and limit values for concentrations of particulate matter

The Air Quality Strategy objectives for particulate matter are based on the health effects, which can result from both short-term and long-term exposure, and are linked mainly to respiratory and cardiovascular effects. The European Union limit value for PM_{10} that came into force on 1 January 2005 is 50 µg m^{-3} per 24-h period, with up to 35 exceedences per year allowed. The annual limit value that also came into force on 1 January 2005 is 40 µg m^{-3}. The European Union has proposed an indicative limit value that should be met by 1 January 2010. It is also 50 µg m^{-3} per 24-h period, but the number of allowed exceedences is reduced to seven per year and the annual limit value is halved to 20 µg m^{-3}.The Air Quality Strategy adopted the 2005 values as objectives and the 2010 values as provisional objectives but modified them to make the annual objective more stringent (18 µg m^{-3}) in Scotland and less stringent (23 µg m^{-3}) in London. The 2010 limit values will be reviewed by the EC in the light of further experience and information and currently have no legal standing.

Sources of particulate matter and trends in emissions

Particulate matter derives from both human-made and natural sources. Road transport gives rise to primary particles from engine emissions, from tyre and brake wear and from other non-exhaust traffic emissions. Other primary sources include stationary combustion processes (industrial, commercial and domestic), quarrying, construction and non-road mobile sources; natural sources include sea spray and Saharan dust. Secondary particulate matter is formed from emissions of ammonia (NH_3), sulphur dioxide (SO_2) and nitrogen oxides as well as emissions of organic compounds both from combustion sources and vegetation.

There have been substantial reductions in emissions in recent decades. Primary UK PM_{10} emissions fell from 570 kt in 1970 to 200 kt in 2000. A further fall of 28% is expected between 2000 and 2010 and predicted reductions in UK emissions of secondary precursors over this period range from 52% (SO_2) to 10% (NH_3). It is clear, however, that reductions will level off and total UK PM_{10} emissions are expected to change little between 2010 and 2020 with current measures. The Department for Transport now expects greater market penetration of diesel cars in the UK than it had previously forecast and had been assumed in the earlier versions of the National Atmospheric Emissions Inventory used in the air quality models that are discussed in this report. This will lead to slightly higher PM_{10} emissions than were originally predicted.

Monitoring of particulate matter

UK monitoring networks primarily use the tapered element oscillating microbalance (TEOM) analyser. This provides real-time data with a short time resolution that is essential for the delivery of up-to-date public information. The TEOM uses a heated inlet, which leads to losses of semi-volatile compounds. The European reference method uses an unheated, filter-based gravimetric method. This method provides 24-h concentrations some days after the measurement. The differences in sampling lead to some inconsistency in results. Currently, as an interim measure, TEOM values are scaled by a factor of 1.3 to account for the losses of semi-volatile components. This is not ideal, as the real factor is highly variable from day to day and place to place. It would therefore be advantageous to identify a continuous method of quantifying the semi-volatile components.

The Clean Air for Europe (CAFE) programme is currently considering the introduction of targets for $PM_{2.5}$. Only a small number of sites (15) in the UK have co-located monitoring of PM_{10} and $PM_{2.5}$. Although current evidence in this report indicates a strong correlation between daily PM_{10} and $PM_{2.5}$ concentrations, AQEG views this number as inadequate and recommends an increase in the number of sites measuring both.

Current measured concentrations and composition of particulate matter

The report collates measurements from 240 monitoring sites, including kerbside, roadside, urban background/centre, industrial and rural/remote locations. Data from these sites and from other measurements have been used to assess sources, distributions and concentrations of particulate matter in the UK. Annual mean PM_{10} concentrations are highest at roadside sites and lowest at rural sites, demonstrating the importance of road traffic as a source. There is evidence that the contribution of heavy duty vehicles is much greater than that of light duty vehicles. The concentration gradient between roadside, urban background and rural concentrations is much less pronounced than is found for nitrogen oxides and indicates a greater regional background contribution to PM_{10}. This regional background contribution is a substantial fraction of the total even in London. There is a considerable variability, from month to month, in the number of exceedences of 24-h average concentrations of 50 $\mu g\ m^{-3}$. This demonstrates episodicity in concentrations derived from a range of sources that contribute to PM_{10}. Episodes of high secondary particulate matter also result from air masses arriving in the UK from European sources during anticyclones. Other episodic sources include sea salt, Saharan dust and biomass burning. Locally elevated particulate matter concentrations can result from construction activities, local roads, industries and domestic premises burning solid fuel.

Observed and predicted trends in concentrations of particulate matter

Monitoring sites that have long enough records to establish trends show downward trends in PM_{10} concentrations, but the steep decline observed over the period 1992–1999 has given way to a flattening out or even a slight increase over the period 2000–2003. It is unclear to what extent this change in behaviour arises from year-to-year variations in the weather. Black smoke records provide an 80-year record in London and show a decrease of a factor of about 50 over this period, largely as a result of the phase-out of coal burning. Measurements in London, though, show evidence of a slower decline in black smoke concentration over the last 10 years, related to emissions from diesel road traffic. Declines have also been seen in rural PM_{10} measurements that can be related, in part, to reduced emissions of SO_2 and hence of production of secondary particulate sulphate. The overall reductions show, however, that other sources – primary or secondary – must also contribute to rural concentrations.

Attainability of the Air Quality Strategy's objectives

Models incorporating assessments of future source strengths are used to predict future concentrations of particulate matter and likely exceedences of limit values and objectives in 2005 and 2010 as well as to develop mitigation policies. The models routinely used for national and local policy support in the UK include empirical components, based on monitoring data, and dispersion calculations. The diverse sources of particulate matter make predictions difficult and problems are encountered with: (i) the coarse fraction, which has a large range of sources, including road dust; (ii) the background concentrations and their dependence on both primary and secondary sources; and (iii) the increment in concentrations occurring at the road side. Calculations show the annual mean limit value set for 2005 being met nearly everywhere, but with some exceedences of the limit of 35 days with 24-hour averages above 50 $\mu g\ m^{-3}$, especially in London. However, substantial exceedences both of the more stringent indicative annual mean limit values and of the smaller number of days above 50 $\mu g\ m^{-3}$ suggested for 2010 are likely throughout the UK. It is clear, given the substantial background particulate matter concentrations and the extent of the exceedences, that the additional reductions required by 2010 to meet the Stage 2 indicative limit values cannot be met by control of primary emissions alone.

Local air quality management is able to identify local hotspots that are not necessarily identified through national studies. Air Quality Management Areas (AQMAs) have been established by 63 local authorities where exceedences of the 2005 limit values for PM_{10} are likely. The majority of these are for traffic sources, but there are also AQMAs for industrial, commercial and domestic sources of PM_{10}. The Action Plans being developed will help to ensure that concentrations are driven down in these areas, but will probably make only a marginal contribution to the wider reduction in particulate matter concentrations because of the substantial background contribution. The exception to this might be in London and other major conurbations, where many local authorities are working together to develop larger scale plans, for example, the low emission zone initiative being developed in London.

Enhancement of policy assessment and improvement of air quality

The modelling of particulate matter concentrations is inherently more complex than for other common pollutants because of the need to combine the contributions from different sources, for example, long-range transport of secondary particulate matter, primary contributions from urban sources and very local contributions from individual roads. Models perform reasonably well for current years, but the complexity of particulate matter and the manner in which source contributions may change adds to the uncertainty in predicting future concentrations. Further work is required to improve and refine the models and to check their accuracy with respect to the different components and their sources. It is still not possible to relate all the observed PM_{10} mass to specific sources or to be certain about the relative contributions of different types of source, some of which remain obscure. Targeted imporvements in the monitoring network, including enhanced monitoring of chemical components of the particulate mass, would help to clarify these questions and uncertainties. These improvements should include the deployment of monitors for particulate sulphate, nitrate, elemental and organic carbon and iron. It is also recommended that further rural PM_{10} and $PM_{2.5}$ monitoring is undertaken to assess the background levels of particulate matter and hence determine the urban increment and to aid the understanding of particulate matter episodes. Co-location of particulate sulphate monitors with existing rural ozone, particulate nitrate, PM_{10} and $PM_{2.5}$ measurements would contribute substantially to our understanding of the link between elevated regional particulate matter concentrations and the concentrations of individual components.

It is clear that while road traffic emissions are a major source of particulate matter near to roads, the regional background contribution, both rural and urban, is still dominant and must form a central component of mitigation strategies. AQEG recommends that consideration be given to additional forms of regulation to reduce mean population exposure, complementing concentration-based limit values, which tend to focus attention on local hotspots. The regional contribution and the links between the concentrations of different pollutants demonstrate the need for a more holistic approach to urban air quality management and its coupling to the control of acid rain, eutrophication and ground-level ozone.

Chapter 1

Introduction

1. The Air Quality Expert Group (AQEG) was established to help the Government and Devolved Administrations tackle air pollution by examining sources and concentrations of pollutants in the UK. It was set up following publication of the *Air Quality Strategy for England, Scotland, Wales and Northern Ireland* with the following functions:

- to advise the Government on levels, sources and characteristics of air pollutants in the United Kingdom (UK);
- to assess the extent of exceedences of existing and proposed Air Quality Strategy objectives and European Union (EU) limit values, where monitoring data are not available;
- to analyse trends in pollutant concentrations;
- to assess current and future ambient concentrations of air pollutants in the UK; and
- to suggest potential priority areas for research aimed at providing a better understanding of the issues that need to be addressed in setting air quality objectives.

2. The aim of this report, the second by the group, is to address these issues with respect to particulate matter (PM). The amenity effects of dust are not addressed in full here, but a study of the full database of visibility measurements at meteorological stations in the UK has been conducted previously (Doyle and Dorling, 2002; Doyle, 2004). Care has to be taken to account for natural visibility variability, but there is evidence at some sites of trends that can be ascribed to reductions in local and regional pollution. However, formulating air quality objectives for visibility, or predicting future trends, as part of an air quality management system would not be possible at the present time and hence has not formed part of this report.

1.1 Particulate matter

3. Atmospheric PM consists of a wide range of materials arising from a variety of sources. These may be solid or liquid particles that range in size and have complex chemical compositions. PM in the UK is measured using a range of metrics. The most frequently used is PM_{10}: this measure describes the mass of particles that pass a sampler entry with a 50% efficiency at 10 micrometres (µm). Thus as a good approximation, PM_{10} describes the mass of particles in the atmosphere with a diameter less than 10 µm. This is also the fraction of particles that is most likely to be deposited in the lung. The less commonly measured determinand $PM_{2.5}$ is defined similarly, but applies to particles with a diameter less than 2.5 µm. For many years atmospheric PM in the UK was measured in terms of its blackness, using a measure known as black smoke. Black smoke data still have value and this report also addresses particle data measured in this way.

1.2 Health effects of particulate matter

4. The Air Quality Strategy objectives[1] for PM are primarily based on health effects. The Department of Health's Committee on the Medical Effects of Air Pollutants (COMEAP) estimated that in Great Britain in 1996 PM_{10} pollution was associated with around 8,100 deaths brought forward and 10,500 hospital admissions that were either brought forward or would not otherwise have happened (COMEAP, 1998).

This was due to short-term (acute) exposure. It is likely that the health impacts of long-term (chronic) exposure on the population were even greater. COMEAP have estimated that over the lifetime of the current population of England and Wales, the health gain that would result from a reduction of 1 $\mu g\ m^{-3}$ of $PM_{2.5}$ could lead to a gain of 0.2 – 0.5 million life years. Estimates derived using the same COMEAP coefficients, but more recent exposure data, are lower.

5. PM causes inflammation of the airways that may worsen existing lung disease and enhance the sensitivity to allergens of people with hay fever and asthma. It may also alter the ability of blood to clot and the circulation of red blood cells and platelets, a mechanism that could explain the adverse influence of inhaled particles on cardiovascular morbidity (illness) and mortality (death).

6. There have been a limited number of health studies on a range of PM parameters, including $PM_{2.5}$, $PM_{1.0}$, $PM_{0.1}$, particle numbers and surface area and there is much debate about the effects of the various size fractions. This report covers all the particle parameters for which UK measurements are available.

7. PM_{10} standards are generally based on available epidemiological studies relating to the relevant geographical area. For example, the PM_{10} standard recommended by the Expert Panel on Air Quality Standards (EPAQS) was derived from studies done in London and Birmingham and was later confirmed by a larger study done in the West Midlands once further PM_{10} and $PM_{2.5}$ monitoring data were available.

8. The health effects of PM will be addressed in brief in Chapter 3.

1.3 Emissions

9. PM may be either primary or secondary. Primary particles such as carbon particles from combustion, salt from the sea and mineral particles derived from stone abrasion are released directly into the air, whereas secondary particles are formed in the atmosphere by chemical reactions that lead to the formation of low volatility compounds that condense into particles.

10. The main sources of primary PM are road transport (combustion by-products, brake and tyre wear and re-entrainment of dust from road surfaces); stationary combustion (mainly domestic coal burning); and industrial processes (production of metals, cement, lime, coke and chemicals, bulk handling of dusty materials, construction, mining and quarrying). In 2001, road transport accounted for about

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[1] Objectives are policy targets, which outline what the Government intends should be achieved in the light of air quality standards.

16% (combustion only) of total UK emissions of PM_{10}, with further major contributions from the electricity supply industry (10%) and the industrial (13%) and domestic (18%) sectors. In London, road transport accounted for 68% of PM_{10} emissions. It is worth noting that this 68% value does not include the long-range PM component originating outside the London area or coarse PM_{10} sources because these are difficult to quantify. UK PM_{10} emissions have declined substantially over the last 30 years, with an overall reduction of 67% between 1970 and 2001. The decline is mainly due to reductions in coal use. Domestic emissions have fallen by 85% between 1970 and 2001 and road transport emissions have declined by 12.5% over the same timescale.

11. Emissions are likely to decline further over the coming years as new technology and emissions standards continue to be introduced. Estimates of current and future emissions are provided via emissions inventories, which allocate emissions to categories – for example, road transport and power generation – and also provide spatial maps. The National Atmospheric Emissions Inventory (NAEI) covers the whole of the UK, mapping emissions on a 1 km x 1 km spatial grid. There are, in addition, several local inventories, for example, the London Atmospheric Emissions Inventory (LAEI).

12. Secondary particles are less easy to ascribe to their original sources. They are comprised mainly of ammonium sulphate and nitrate. These originate from the oxidation of sulphur and nitrogen oxides in the atmosphere to acids, which are then neutralised by atmospheric ammonia derived mainly from agricultural sources. The chemical processes involved in their formation are relatively slow and their persistence in the atmosphere is prolonged. Thus, while road traffic may be the main source of the original oxides of nitrogen and coal and oil burning the main source of sulphur oxides, the secondary particles are distributed more evenly throughout the air with fewer differences between urban and rural areas. They can also travel large distances, resulting in the transport of particles across national boundaries.

1.4 Measurements of particulate matter

13. The UK has an extensive national network of 64 automatic monitoring sites and an additional seven gravimetric analysers that measure PM. These comprise the Automatic Urban and Rural Network (AURN) and some affiliated sites from the London Air Quality Network (LAQN). The AURN sites include both those operated for the Department for Environment, Food and Rural Affairs (Defra) and the Devolved Administrations and those owned and operated by other organisations, mainly local authorities. These sites use a Tapered Element Oscillating Microbalance (TEOM), sequential gravimetric analyser (Partisol) or ß-attenuation monitor (one site only) to measure PM_{10}. The automated instruments (TEOM and ß-attenuation monitor) were chosen mainly for their reliability in the field and their ability to provide near real-time data for public information. In addition, several regional networks are operated, for example by the Highways Agency and power generators. Collective measurements, some extending over 10 years, provide an invaluable resource with which to assess the temporal and spatial variations in PM_{10}, including exceedences of the air quality objectives. These data have been used extensively in this report.

14. In addition, national monitoring of a number of other metrics and components, including $PM_{2.5}$, particulate nitrate, particulate sulphate, elemental and organic carbon, polycyclic aromatic hydrocarbons (PAHs), black smoke, heavy metals and particle numbers, also takes place.

1.5 Modelling

15. Although the monitoring sites have a wide spatial coverage, they can provide only a representation of pollutant concentrations at a relatively small number of points throughout the UK. When gradients of concentration are relatively small, as occurs in rural areas, measurements can be interpolated to give the spatial variation in concentration, which is broadly consistent with regional models. At the urban and local scales as well as close to significant sources, measurements give a limited indication of the spatial variation in concentration, and models are required to determine the complete picture. In addition, measured data cannot be used in a simple way to project future concentrations and to assess future exceedences because of the complex nature of the likely future emissions of primary PM and of the processes involved in the formation of secondary PM. As a result, modelling is essential for assessing the links between estimates of emissions and of atmospheric concentrations, to provide maps of particle concentration and to estimate future concentrations and exceedences.

16. A robust fundamental approach – in which meteorology and chemistry are modelled on a range of scales, from metres to many kilometres, using mathematical formulations – is not yet available for assessment purposes. A semi-empirical approach, utilising statistical analysis of large amounts of monitoring data – coupled in some circumstances with dispersion modelling – has been widely used in the UK for policy development. It forms a central component of the analysis and projections presented in this report and is coupled with surface concentration modelling to generate maps of PM_{10} concentrations across the UK. Urban scale and local models that can take account of local features such as street canyons and smaller averaging times are also used to produce maps for urban areas; these maps are then used to test the assumptions implicit in the national mapping approach and for air quality review and assessment. In this report numerical models for regional scales up to the European scale have been used mainly to assess and project the contribution of secondary particulates.

1.6 Policy background

17. PM emissions and concentrations in the UK are subject to a number of national and international agreements and initiatives. These are outlined below and are described in more detail in individual chapters where appropriate.

1.6.1 International legislation and agreements

18. There are four EU directives that relate directly to emissions and concentrations of PM:

i. The First Air Quality Daughter Directive (1999/30/EC) sets limit values for 24-h and annual average PM_{10} that must be achieved throughout the European Community (EC) by 1 January 2005. The Directive also contains Stage II indicative limit values for both 24-h and annual average PM_{10} to be achieved by 1 January 2010 (Table 1.1). These Stage II limit values are only indicative and will be reviewed by the EC in light of further information on health and environmental effects, technical feasibility and experience gained in the application of Stage I limit values by Member States. They have no legal standing. Monitoring of $PM_{2.5}$ is also included in this directive, although no limit values have been set.

ii. The Auto-Oil Programme introduced tighter European vehicle emission and fuel quality standards by regulating particulates, carbon monoxide, hydrocarbons and nitrogen oxides. In 1998, three Auto-Oil proposals for the introduction of further standards from 2000 were agreed. The key elements to this agreement – now consolidated into two directives – are:

- a stringent series of emission standards (known as the Euro III standard) to apply to all new cars and light vans sold from January 2001 and to all new heavy duty vehicles sold from October 2001;
- more stringent standards (Euro IV standards) to apply from 1 January 2006 for new cars and light vans and from October 2006 to new heavy-duty vehicles;
- tighter quality specifications to apply to all new petrol and diesel fuel sold from 1 January 2000 and 1 January 2005; and
- a ban on the general marketing of leaded petrol from 1 January 2000.

iii. For industrial plants, two additional directives provided (both directly and indirectly) controls on emissions of both particles and the secondary particle precursors sulphur dioxide and nitrogen oxides: the Large Combustion Plant Directive (2001/80/EC) established controls on emissions from large combustion plants, which include power stations, oil refineries and large energy producers within industry, particularly the metals sector.

iv. The Integrated Pollution Prevention and Control (IPPC) Directive (96/61/EC) requires site-specific permits, that take account of the characteristics of each installation, its location and the state of the local environment. A wide range of installations and environmental impacts are taken into account by the regulator in considering a permit application. Conditions must be included that address any transboundary pollution caused by the installation. The Directive requires individual industrial plants to take any measures necessary to comply with any relevant EC legislation.

19. The National Emission Ceilings Directive (2001/81/EC) does not regulate PM directly but affects particle precursors such as ammonia (NH_3), sulphur dioxide (SO_2) and nitrogen oxides (NO_X). It establishes for each Member State a national limit for emissions of certain atmospheric pollutants including NO_X, sulphur (as SO_2), NH_3 and non-methane volatile organic compounds (NMVOCs). The National Emission Ceilings Directive incorporates (and in some cases goes beyond) ceilings established previously under the UNECE Convention on Long-Range Transboundary Air Pollution Gothenburg Protocol. Under the Directive, the UK is committed to reducing its national annual emissions of NO_X to below 1167 kt, SO_2 to below 585 kt, NH_3 to below 297 kt and NMVOCs to below 1200 kt, all by 2010.

20. The EC's ongoing Clean Air for Europe (CAFE) programme, culminating in the publication of a Thematic Strategy on air quality in summer 2005, will lead to a number of revisions to existing Directives, including the first, second and third Daughter Directives as well as the National Emission Ceilings Directive. Revisions to the first Daughter Directive may include the introduction of controls on $PM_{2.5}$. The CAFE Technical Working Group on Particulate Matter has suggested a range of ambient $PM_{2.5}$ concentrations for consideration in CAFE's integrated assessment of possible future PM targets. This range encompasses $PM_{2.5}$ concentrations that are equivalent to or more stringent than the comparable Stage II indicative annual average limit value for PM_{10}. This is the range of possible limit values discussed in Chapter 9.

1.6.2 National framework

21. Part IV of the Environment Act 1995 required the Government to produce a National Air Quality Strategy containing standards and objectives and also measures for achieving the objectives (Table 1.1). The first edition of the Strategy, published in 1997, fulfilled that requirement. It contained objectives for annual mean and 24-h exposure to PM_{10}. The Strategy was reviewed in 1999 and the Air Quality Strategy for England, Scotland, Wales and Northern Ireland was published in January 2000. This contained revisions to the PM_{10} objectives. The PM_{10} objectives were reviewed again in September 2001 and revised in an Addendum to the Strategy published in February 2003. The Act also laid the foundations for the system of Local Air Quality Management (LAQM). The Air Quality Strategy is undergoing a further review during 2004/5, this time focusing on a more measures-based approach.

22. Local Air Quality Management forms an important part of the Government's Strategy for meeting both the UK air quality objectives and EU limit values. The Strategy recognises that national measures, for example, controls on emissions from new motor vehicles, will not always be the most appropriate way to deal with localised hotspots. LAQM requires all local authorities to carry out regular reviews and assessments of air quality in their area, with a view to identifying all those locations at which one or more of the objectives is unlikely to be met by the relevant date, taking account of relevant exposure (Defra, 2003). Such locations have to be formally designated as Air Quality Management Areas (AQMAs) and an Action Plan prepared for moving towards meeting the objectives. Now in its sixth year, the LAQM process has identified the need for 132 of 407 local authorities in England, Scotland and Wales to declare AQMAs. (Northern Ireland is working to a different timetable.) Of these, 92% are for nitrogen dioxide (NO_2), 45% for PM_{10} (mostly in combination with NO_2) and 4% for SO_2 exceedences. The PM_{10} declarations are mainly for road traffic, although in some cases exceedences are due to combined contributory emissions from industry. These local studies of PM_{10} have generated information that supplements the findings of the national studies, and both are drawn upon in this report. For instance, although the national studies were inevitably focused on London and other major conurbations, as well as on the most heavily trafficked roads, the LAQM studies have shown that exceedences can also occur in smaller towns, where narrow streets coincide with congested traffic.

Table 1.1 EU limit values and Air Quality Strategy (AQS) objectives for PM_{10}.

Legislation	24-hourly values	Annual values ($\mu g\ m^{-3}$)	Achieved by
EU First Daughter Directive (99/30/EC) Stage I limit value	50 $\mu g\ m^{-3}$ with up to 35 exceedences per year	40	1 January 2005
EU First Daughter Directive (99/30/EC) Stage II indicative limit value	50 $\mu g\ m^{-3}$ with up to 7 exceedences per year	20	1 January 2010
AQS (2000) statutory objectives	50 $\mu g\ m^{-3}$ with up to 35 exceedences per year	40	31 December 2004
AQS Addendum (2003) – all parts of UK except London and Scotland provisional objectives	50 $\mu g\ m^{-3}$ with up to 7 exceedences per year	20	31 December 2010
AQS Addendum (2003) – Scotland provisional objectives	50 $\mu g\ m^{-3}$ with up to 7 exceedences per year	18	31 December 2010
AQS Addendum (2003) – London provisional objectives	50 $\mu g\ m^{-3}$ with up to 10 exceedences per year	23	31 December 2010

23. The Greater London Authority Act received Royal Assent in November 1999. It provides for the Mayor of London to prepare and publish a Mayor's Air Quality Strategy (MAQS) for Greater London. London local authorities must take the MAQS into account when exercising their functions under Part IV of the Environment Act 1995.

24. The MAQS, published in September 2002, sets out how the Mayor intends to implement the National Strategy in Greater London and achieve the objectives in the relevant regulations. It also contains:

- information about current and likely future air quality in Greater London;
- the measures that are to be taken by the Greater London Authority, Transport for London and the London Development Agency to implement the Strategy;
- information about measures the Mayor will encourage other people and organisations to take; and
- any other proposals and policies the Mayor considers appropriate.

25. Part I of the Environmental Protection Act 1990 has been the main mechanism for minimising air pollution from industrial sources for a number of years. This Act established two pollution control regimes: the Local Air Pollution Control (LAPC) regime and the Integrated Pollution Control (IPC) regime, which not only takes account of releases to air, but also to land and water.

26. The IPC and LAPC are being succeeded by the new pollution prevention and control (PPC) regime, implementing the IPPC Directive 96/61/EC. There are separate regulations in Scotland and it will be for the new Executive to determine the arrangements for Northern Ireland. Although the terminology used differs slightly, the principles underpinning both IPC and PPC are similar. Regulators are required to ensure that pollution from industry is minimised through the use of the best available techniques (BAT), which take into account the balance between the costs and environmental benefits. Both systems also require regulators to take account of the characteristics of each installation, its geographical location and local environmental conditions. In England and Wales, PPC is regulated by the Environment Agency and local authorities, in Scotland, by the Scottish Environment Protection Agency (SEPA). In Northern Ireland, PPC is the responsibility of the Industrial Pollution and Radiochemical Inspectorate of the Environment and Heritage Service and district councils.

1.7 Structure of the report

27. This report aims, for the specific case of PM, to fulfil the tasks placed on AQEG and outlined in paragraph 1. It draws on several previous reports including *Airborne Particulate Matter in the UK* (QUARG, 1996), *Source Apportionment of Airborne Particulate Matter in the UK* (APEG, 1999), *Health Aspects of Air Pollution with Particulate Matter, Ozone and Nitrogen Dioxide* (WHO, 2003) and the *Second Position Paper on Particulate Matter* (CAFE, 2004).

28. Chapter 2 provides essential background material, including the physical characteristics and chemical composition of airborne PM, mass closure issues, receptor modelling and source apportionment. Chapter 3 briefly discusses the health effects of particles. Chapters 4 and 5 review the main issues relating to the construction of emissions inventories and the measurement and monitoring of PM. Where possible and appropriate, the Group has included a discussion of fundamental as well as practical issues, although much less detail is provided than for the AQEG report *Nitrogen Dioxide in the United Kingdom* (AQEG, 2004) because of the availability of previous reports on PM.

29. Chapters 6 and 7 provide a detailed analysis of data and an examination of historic trends in PM_{10}. Chapter 6 considers the data and asks what they imply about the sources, properties and spatial distributions of PM. Chapter 7 considers what are the main past and present trends in PM in the UK.

30. Chapter 8 provides some essential background information on the modelling of PM and considers the data from current models, including current and predicted future PM concentrations.

31. The Discussion (Chapter 9) summarises the answers to the questions set for AQEG by Defra and the Devolved Administrations.

- Are the current assessment methods (emissions inventories, measurements and modelling) fit for purpose? How could they be improved?
- Are there sources missing from (a) the UK emissions inventories and (b) other European inventories?
- Is the UK likely to achieve, with current abatement measures and technologies, (a) the Stage I and indicative Stage II annual and daily mean PM_{10} limit values in the First Air Quality Directive and (b) the Air Quality Strategy objectives for PM_{10}? If not, why not? What levels of PM_{10} are likely to be achieved by current measures and policies?
- Will the UK be able to meet the range of targets for $PM_{2.5}$ as proposed in the draft CAFE Position Paper on Particulate Matter? If not, why not? What levels of $PM_{2.5}$ are likely to be achieved by current measures and policies?
- What are the practical maximum feasible reductions of PM_{10} and $PM_{2.5}$ concentrations at (a) hotspots and (b) urban background, for example, central London locations?
- Where and what are the main source contributors to current and future concentrations of PM_{10} and $PM_{2.5}$? What are the contributions of different sources to forecast exceedences of the EU limit values and UK objectives?
- What are the potential sources of abatement and types of measures to reduce particle concentrations at (a) hotspots, such as near busy roads, (b) at urban background, central London and (c) across the whole country? What role can local/national/EU-wide measures play in meeting targets? These measures should be defined as technical (for example, vehicle standards); non-technical (for example, traffic management systems); and international (for example, controlling European/hemispheric emissions). Are there alternatives to emissions reduction?
- A number of recent studies (including the WHO report) have highlighted the health effects of certain components of particulate matter (that is, fines, ultrafines, particle number, metals and elemental/organic carbon). Where further abatement techniques are known, how might they specifically affect reduction of the different PM metrics (for example, PM_{10} , $PM_{2.5}$, PM_1 and particle numbers) and chemical components?
- Ultrafines – What have we learned from the measured data, including source apportionment? Are the observed trends real? What fraction of ultrafine particles volatilise?
- Source apportionment – How does the UK source apportionment for PM_{10} , $PM_{2.5}$ and other metrics compare with other modelling in Europe? Is road traffic more important than current models show? How is the coarser fraction between 2.5 and 10 μm accounted for?

- Can we explain the trends in measured PM_{10} , sulphur and black smoke since 1992?
- What are the differences between strategies that address hotspots of exceedence and those that aim to reduce population exposure? Should policy evaluation consider impacts on population exposure, as well as concentrations at specific locations?

32. Chapter 10 draws conclusions and makes recommendations.

Chapter 2

What is particulate matter?

Key points

- Airborne PM includes a wide range of particle sizes and many different chemical constituents. It contains both primary components, which are emitted directly into the atmosphere, and secondary components, which are formed within the atmosphere as a result of chemical reactions.
- Conceptually, airborne particles are usually considered as belonging to three size modes. The smallest is the nucleation mode, which includes particles newly formed through source condensation processes or through atmospheric chemical reactions. These particles grow – through coagulation and vapour condensation – into the accumulation mode, where they may have a long atmospheric lifetime. Particles with a diameter greater than ~1 μm, are referred to as the coarse mode; they are typically generated through mechanical processes such as quarrying or sea spray formation by breaking waves.
- Airborne particles can range in size from a few nanometres (nm) to around 100 μm in diameter. By far the greatest number of particles falls into the ultrafine size range, which is less than 100 nm, whereas the larger particles – although contributing rather little to particle number – contribute the major proportion of particle mass.
- Some constituents of airborne particles, most notably ammonium nitrate (NH_4NO_3) and PAHs are termed semi-volatile and are able to partition between particles and the vapour phase.
- Major chemical components of airborne particles include sulphates, nitrates, ammonium, sodium chloride (NaCl), elemental and organic carbon, mineral particles and coarse, iron-rich particles generated by vehicles.
- Particles in the atmosphere contain chemically bound water that is not removed completely under the drying conditions used in the standard European weighing procedures.

2.1 Introduction

33. Airborne PM presents a far greater complexity than most other common air pollutants. Not only is it a mixture of different chemical substances, but individual particles also span a wide range of sizes. Both chemical composition and size can provide valuable insights into the sources of airborne particles, and these parameters also determine the atmospheric behaviour and fate of particles as well as influencing human health effects. Because these issues have been examined in detail in reports by the Quality of Urban Air Review Group (QUARG, 1996) and the Airborne Particles Expert Group (APEG, 1999), they will be discussed only briefly here.

34. The most important distinction is between primary and secondary atmospheric particles. Primary particles are those emitted directly from a source and therefore include particles that arise directly from combustion sources, such as road vehicles and power stations, as well as those generated by mechanical processes, for example, quarrying and agricultural harvesting. The land and the sea are both major sources of primary particles, through entrainment of soils by the wind and the generation of marine aerosol particles by the bursting of air bubbles entrained in breaking waves. Secondary particles are not emitted directly from sources, either natural or anthropogenic. Rather, they are formed in the atmosphere as a result of chemical reactions that lead to the formation of substances of low volatility, which consequently condense into the solid or liquid phase, thereby becoming PM. Such particles generally result from atmospheric oxidation processes and the substances oxidised may be either natural or anthropogenic in origin.

Box 2.1 Definitions of PM

The following definitions of particle size categories are widely accepted.

- Nanoparticles: particles smaller than 50 nm in diameter.
- Ultrafine particles: particles smaller than 100 nm in diameter.
- Fine particles: particles in the fine mode (see Figure 2.1) or those in the $PM_{2.5}$ fraction. The two may differ.
- Coarse particles: fraction of the measured particle mass concentration determined from PM_{10} minus $PM_{2.5}$.
- $PM_{2.5}$: airborne particulate matter passing a sampling inlet with a 50% efficiency cut-off at 2.5 μm aerodynamic diameter and which transmits particles of below this size.
- $PM_{2.5–10}$: known as PM_{coarse}.
- PM_{10}: airborne particulate matter passing a sampling inlet with a 50% efficiency cut-off at 10 μm aerodynamic diameter and which transmits particles of below this size.

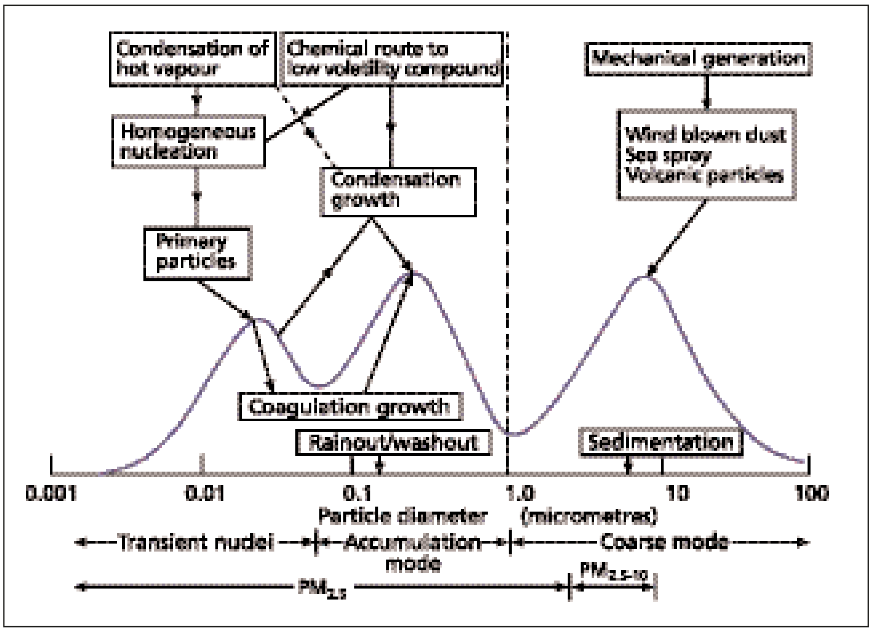
35. The process by which secondary particles are formed is termed nucleation; this term refers to the processes by which molecules of low volatility condense to form solid or liquid matter. There are two distinct types of nucleation process (Hinds, 1999). The first is heterogeneous nucleation. Most secondary particle formation in the atmosphere occurs by this process, newly formed substances condense onto existing particles causing the growth of those particles. The condensation processes are most effective on small and medium sized particles, causing these to grow larger. The second process is called homogeneous nucleation. Some newly formed molecules have extremely low vapour pressure and, in the absence of an abundance of pre-existing particles, will condense with one another to form wholly new particles. The best known process of homogeneous nucleation occurs when sulphuric acid (H_2SO_4) is formed from the atmospheric oxidation of SO_2. This H_2SO_4 can nucleate with water vapour (binary nucleation) or, more readily, with water vapour and ammonia (ternary nucleation) to form droplets of sulphate solution. Measurements made in relatively clean air situations, both over coastlines and the mainland, demonstrate that new particles are frequently formed by condensation of the atmospheric oxidation products of both halocarbons generated from exposed seaweed and hydrocarbons released from forests. In neither case, however, is it wholly clear whether particle formation results from homogeneous nucleation of the oxidation product itself or whether condensation occurs on a primary nucleus formed from H_2SO_4 nucleation.

36. From an air pollution perspective, the secondary components that are of the greatest importance are sulphate arising from the oxidation of SO_2, nitrate from the oxidation of NOx, ammonium from the acid neutralisation of NH_3 gas and secondary organic compounds formed from the atmospheric oxidation of VOCs, largely hydrocarbons.

2.2 Key physical characteristics of airborne particles

37. The behaviour of particles in the atmosphere and within the human respiratory system is determined largely, but not wholly, by their physical properties, which have a strong dependence upon particle size. Figure 2.1 shows, in schematic form, the typical size distribution of airborne particles. Sizes range over several orders of magnitude. The smallest, freshly nucleated, particles are only 1–2 nm in diameter and contain only tens of molecules. At the other extreme, particles may be up to ~100 µm in diameter, which is comparable to a human hair. Particles as large as this rapidly settle out of the air and are of minor health significance because, although they can be inhaled, they do not generally penetrate beyond the nose and mouth. Consequently, air pollution research generally focuses on particles ≤10 µm in diameter, although it is important to recognise that larger particles are also present and contribute mass to 'total suspended' particles (TSP).

Figure 2.1 Schematic diagram of the size distribution of airborne particles.



38. Figure 2.1 divides particles conceptually into three subgroups. These arise in different ways and have quite distinct atmospheric behaviour; they are 'therefore' referred to as separate modes of the distribution.

2.2.1 Nucleation mode

39. The smallest group of particles, with diametres of ≤50 nm, are usually referred to as the nucleation mode; however, some describe the smallest particles, of 10 nm and below, as the nucleation mode and the particles falling between10–50 nm as the Aitken mode. Such particles will have been newly formed either by homogeneous nucleation in the atmosphere (see on previous page) or by nucleation processes that occur within the emissions from high temperature sources and lead to the emission of primary nucleation mode particles. Particles within the nucleation category soon become part of the Aitken category by condensational growth. Such particles have a relatively transient existence in the atmosphere since they readily transform into larger particles and also deposit quite efficiently to surfaces because of their highly diffusive nature, which results from their low individual mass and high tendency to Brownian motion.

2.2.2 Accumulation mode

40. Growth of nucleation mode particles, primarily by vapour condensation but also as a result of coagulation processes, leads to formation of the accumulation mode of particles that are typically between 50 nm and 1 µm in size. Such particles are too large to be subject to rapid Brownian motion and too small to settle from the air rapidly under gravity. Their further growth is inhibited because they do not coagulate as rapidly as fine and coarse particles do and there are diffusion barriers to their growth by condensation. Particles in the nucleation mode can, therefore, have a long atmospheric lifetime, typically 7–30 days, although they are subject to removal by incorporation into rain and this can significantly shorten their atmospheric lifetime.

2.2.3 Coarse particle mode

41. Particles greater than ~1 µm in diameter are typically generated mechanically rather than through the nucleation and condensation processes. Thus, dusts from industrial operations such as quarrying mainly fall into the coarse particle mode as do wind-blown soil and also sea spray. Within this size range, gravitational settling velocities become appreciable and therefore atmospheric lifetimes – although significant – are much shorter than for the accumulation mode particles.

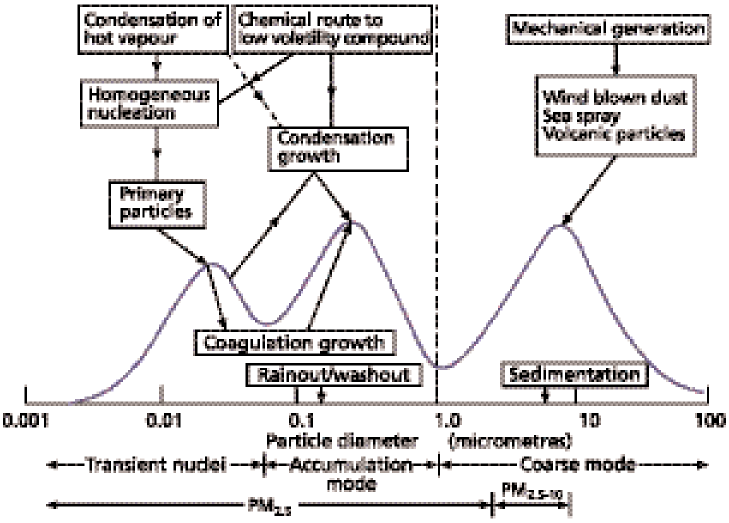
42. It is worth noting at this point that the minimum in particle abundance between the accumulation and coarse particle modes is at ~1 µm and not at 2.5 µm, the size fraction typically used to distinguish coarse and fine fractions during routine air sampling. Thus, typically, a significant proportion of mechanically generated dusts can be sampled within $PM_{2.5}$. One example is Saharan dust, which after long-range transport is substantially within the $PM_{2.5}$ fraction.

43. Coagulation, the process by which particles collide and coalesce together, can be important in aerosol dynamics. This process, which is rapid only when particle number concentrations are high, leads to a general increase in the size of particles. It is, therefore, an important process in the transfer of nucleation mode particles into the accumulation mode. In the absence of fresh emission or new particle formation through nucleation, coagulation processes lead to a gradual reduction in the number of particles and an increase in the mean size.

2.2.4 Number, surface area and volume size distributions

44. While Figure 2.1 shows typical particle size distribution in a schematic form, in practice, the form of size distribution is highly dependent upon whether it is expressed in terms of particle number, surface area or volume/mass. In a typical sample of airborne particles, the vast majority, numerically, are very small in size: it is usual for 70–80% of the number count of particles to be in the ultrafine size range (<100 nm in diameter). However, such small particles have a relatively small surface area and a tiny mass compared to larger particles; therefore, if the distribution of surface area or mass according to particle size is plotted, it looks quite different from the distribution of particle number with particle size. This is exemplified by Figure 2.2, which is taken from measurements of particle number size distributions in suburban Birmingham; these data were then transformed into surface area and volume distributions, assuming spherical particle geometry.

Figure 2.2 Particle size distribution measured in Birmingham.



45. Provided there are no major changes in particle density with particle size, the size distribution of particle mass is very similar to that of volume. Table 2.1 shows the effect of dividing up a single 10 µm particle into particles of 1, 0.1 and 0.01 µm on relative number and relative surface area. Thus, it takes one billion 0.01 µm particles to equal the volume and mass of one 10 µm particle. That one billion 0.01 µm particles has a surface area one million times larger than the 10 µm particle of the same overall volume. Thus, in Figure 2.2, the number size distribution gives great emphasis to the nucleation mode, with the accumulation mode visible. The surface area distribution gives greatest emphasis to the accumulation mode with rather little surface area being associated with nucleation and coarse mode particles. On the other hand, the volume distribution in this case shows two approximately equal modes with a minimum that is close to 1 µm. The nucleation mode contributes insignificantly to particle volume in this example.

Table 2.1 Influence of particle size on particle number and surface area for a given particle mass, assuming spherical geometry.

Particle diameter (µm)	Relative number of particles	Relative surface area
10	1	10
1	10^3	1^2
0.1	10^6	1^4
0.01	10^9	1^6

2.2.5 Semi-volatile components

46. As indicated above, condensation of materials of low volatility is a significant growth mechanism for airborne particles. However, some of the condensable materials themselves have significant vapour pressures and will tend to equilibrate between the particle and vapour phases. An example of substances that undergo such equilibration are the PAHs, which exhibit a wide range of vapour pressures and, therefore, show a range of behaviours, with the two- and three-ringed compounds (for example, naphthalene and anthracene) existing predominantly in the vapour phase (see Table 2.2) and the five-ringed compounds such as benzo(a)pyrene being predominantly incorporated into airborne particles. For the four-ringed compounds such as pyrene, the partition is more equal. The extent of partition into airborne particles is dependent on the air temperature as well as the airborne particle loading. Yamasaki *et al*. (1982) proposed a temperature-dependent equilibrium constant. This is the ratio of the vapour phase (PAH) concentration (A) to the concentration in the particulate phase taken as the particle phase (PAH) concentration in air (F) divided by the concentration of total suspended particles (P):

$$\mathrm{K} = A/(F/P) = AP/F$$

This equilibrium ratio is strongly temperature dependent, and Foreman and Bidleman (1987) proposed the following relationship:

$$\log K = \log p^{\circ}_L + \log k'$$

Here, $\log p^0_L$ is the supercooled (conventionally referred to as sub-cooled) liquid phase vapour pressure of the compound and log k' is an intercept from a plot of log K versus $\log p^0_L$ for the class of compounds.

Table 2.2 Percentage of various PAHs in the vapour and particle phases determined from samples collected in Birmingham (Lim, 1999).

Compound	Number of rings	Vapour phase (%)	Particle phase (%)
Phenanthrene	3	98	2
Anthracene	3	97	3
Fluoranthene	4	95	5
Pyrene	4	93	7
Benzo(a)anthracene	4	12	88
Benzo(b)fluoranthene	5	4	96
Benzo(e)pyrene	5	2	98
Benzo(a)pyrene	5	<2	>98
Coronene	7	<2	>98

47. A more complex situation exists for particulate compounds formed by reversible chemical reactions. The best known example is that of NH_4NO_3, formed from the reversible reaction of gas phase NH_3 and nitric acid (HNO_3):

$$NH_3 + HNO_3 \leftrightarrow NH_4\ NO_3$$

Not only is the equilibrium ratio for the above process dependent upon temperature, it is also strongly affected by relative humidity above the deliquescence point of NH_4NO_3, since NH_4NO_3 then dissolves into an aqueous solution and its thermodynamic properties are changed. The existence of NH_4NO_3 in particles is strongly favoured by low temperature and high relative humidity; diurnal variations in the equilibrium between particulate NH_4NO_3 on the one hand and vapour phase NH_3 and HNO_3 on the other can be a result of normal daily cycles in humidity and temperature (Allen *et al.*, 1989). Sampling instruments that heat the incoming air tend to drive off semi-volatile constituents such as low molecular weight PAH, NH_4NO_3 and water.

2.3 Chemical composition of airborne particles

48. Airborne particles have a very diverse chemical composition that varies both in time and space. As indicated in Section 2.2, particles from specific sources can fall into characteristic size ranges and, therefore, particle composition can vary with particle size, reflecting this effect. For example, sulphate arising from the oxidation of SO_2 is typically present in fine particles, whereas silicon from the resuspension of soils and surface dusts is normally found in coarse particles. Due to the processes of condensation and coagulation, however, particles can become internally mixed. Such an internally mixed aerosol is a mix of particles containing components that have arisen from different sources. In contrast in externally mixed aerosols, particles have a composition that reflects the individual source. Coagulation and condensational growth processes tend to produce internally mixed particles.

2.3.1 Major components

49. Airborne particles contain both major and minor components. The major components, usually comprising at least a few per cent of the mass of particles, normally include the following:

- Sulphate – arises mainly as a secondary component from atmospheric oxidation of SO_2, although there may be a small primary component that arises from sea salt or mineral matter such as gypsum;

- Nitrate – normally present as NH_4NO_3, which results from the neutralisation of HNO_3 vapour by NH_3, or as sodium nitrate ($NaNO_3$), due to displacement of hydrogen chloride from NaCl by HNO_3 vapour;

- Ammonium – generally present in the form of ammonium sulphate ($(NH_4)_2SO_4$) or NH_4NO_3;

- Sodium and chloride – in sea salt;

- Elemental carbon – black, graphitic carbon formed during the high temperature combustion of fossil and contemporary biomass fuels;

- Organic carbon – carbon in the form of organic compounds, either primary, resulting from automotive or industrial sources, or secondary, resulting from the oxidation of volatile organic compounds (VOCs);

- Mineral components – crustal materials (rock and soil) are rich in elements such as aluminium, silicon, iron and calcium. These are generally present in coarse dusts that arise from, for example, wind-driven entrainment processes, quarrying, construction and demolition processes; and

- Water – water soluble components of airborne particles especially $(NH_4)_2SO_4$, NH_4NO_3 and NaCl – take up water from the atmosphere at high relative humidities, thereby turning from crystalline solids into liquid solution droplets. The transition occurs at a specific humidity known as the deliquescence point, at which dry crystals are exposed to increasing relative humidities (see Figure 2.3). There is, however, a hysteresis effect: upon a reduction in the relative humidity, water is lost only slowly and is retained at relative humidities well below the deliquescence point. This means that particles sampled from the atmosphere, even after drying at 40–50% relative humidity (as is customary before weighing), will still retain bound water, often representing a significant component of the mass.

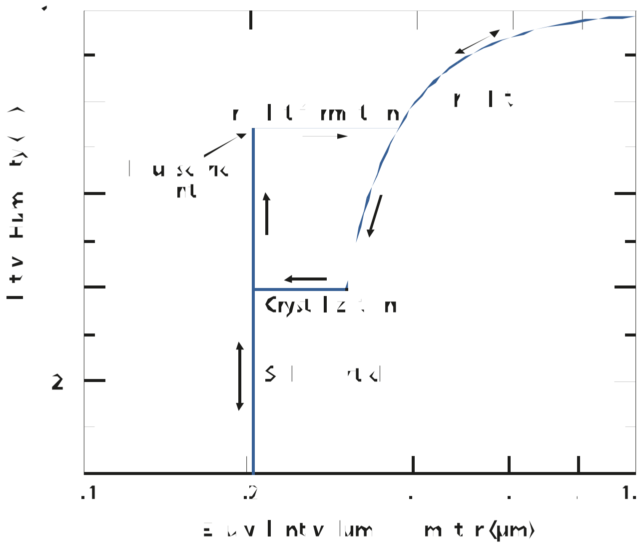
2.3.2 Minor components

50. Either in addition to or within some of the above major components there are many minor chemical components present in airborne particles. Their detection is often a function of the sensitivity of the analytical procedure. Minor components include the following:

- Trace metals – many metals such as lead, cadmium, mercury, nickel, chromium, zinc and manganese are used in metallurgical processes. Some occur as impurities or additives in fuels and others are used in industrial products. These and other uses cause emissions to the atmosphere, but concentrations are generally very small; and

- Trace organic compounds – although the total mass of organic compounds can comprise a significant part of the overall mass of particles, it is made up of a very large number of individual organic compounds, each of which is present at a very low concentration. Such organic compounds vary greatly in composition and include aliphatic and aromatic hydrocarbons, heterocyclics and oxygenates such as aldehydes, ketones and carboxylic acids. Generally speaking, organic compounds that arise directly from fuel combustion processes have a relatively high hydrogen to carbon ratio, whereas secondary organic compounds are more oxidised and polyfunctional carbonyl and carboxylic acids species are often present. To date, studies of specific organic compounds within airborne particles have mostly failed to account for more than 10% of the total mass of organic matter.

Figure 2.3 Equivalent volume particle diameter of a 10^{-14} g particle of NaCl as a function of increasing and then decreasing the relative humidity (based on Hinds, 1999).



2.4 Mass closure of airborne PM

51. The total mass concentration of particles in air can be determined by filtering a known volume of air and weighing the mass of particles collected by the filter. This is the principle behind both gravimetric techniques, such as the CEN reference method, and the TEOM instrument (see Chapter 5). In theory, it should be possible to analyse all of the chemical components within such a sample and, if all are accounted for, the sum of the masses of the individual components should equate to the mass of the sample determined by weighing. The concept of the sum of the components equalling the measured mass of particles is referred to as mass closure. In practice, mass closure for airborne particles has never been achieved since there are a number of components that cannot be measured adequately.

52. Specific problems in addressing mass closure include the following:

- As discussed above, under the usual conditions of weighing (standard protocols typically require humidities of 40% or 50%) particles retain some strongly bound water. There is no straightforward way of analysing the water content;

- Crustal materials have complex compositions and contain elements that cannot be readily analysed. For example, clay minerals in soils and rock have complex aluminosilicate structures and although measuring the aluminium and silicon content is relatively straightforward, the associated oxygen has to be estimated indirectly rather than analysed. Calcium and iron are also major components of some rocks and most soils, but they can be present in a variety of forms, including oxides and hydroxides (for iron) and sulphates and carbonates (for calcium). Without knowing the precise chemical form, it is difficult to use elemental analysis information for metals such as iron and calcium and empirical corrections are generally required; and

- As outlined above, it is not practicable to analyse all of the individual organic compounds and, therefore, techniques have been developed that analyse the two main forms of carbon: carbon present in elemental graphitic form and carbon present within organic compounds. Although no adjustment is needed to the mass of graphitic elemental carbon, it is necessary to adjust the mass of organic carbon upwards to allow for other elements such as hydrogen, oxygen and nitrogen present in the organic compounds. The recommended adjustment factors range from 1.2 for near-source environments to 1.8 for sites at which the majority of organic carbon is present as highly oxidised secondary components. Such factors are largely a matter of judgement and therefore open to significant uncertainty.

53. Attempts at mass closure for airborne PM have used a variety of approaches. The majority have used elemental analysis of a very wide range of components in order to analyse the major constituents. Inevitably, however, some reconstruction is needed to estimate the mass of crustal/soil particles, the organic compound mass from the organic carbon concentration and the quantity of bound water, which is often simply estimated by determining difference from the mass of the other components.

Table 2.3 Adjustment factors used in mass closure (from Harrison *et al*., 2003).

Analyte	Conversion to	Numerical factor
Sulphate	$(NH_4)_2SO_4$* hydrate	1.38 1.29
Nitrate (fine)	NH_4NO_3 hydrate	1.29 1.29
Nitrate (coarse)	$NaNO_3$ hydrate	1.37 1.29
Chloride	NaCl	1.65
Calcium	$CaSO_4.2H_2O$	4.30
Iron	Soil/road dust	5.50 (roadside) 9.00 (background) 3.50 (roadside increment)
Elemental carbon	Elemental carbon	1.00
Organic carbon	Organic compounds	1.30 (roadside) 1.40 (background) 1.20 (roadside increment)

54. A simplified approach developed within the UK is based upon the analyses and conversion factors given within Table 2.3 (Harrison *et al*., 2003). The basis of this method is to assume that sulphate is present as (NH4)2SO4; nitrate as NH4NO3 and NaNO3 in fine and coarse particles, respectively; and chloride as NaCl. As outlined above, a factor is used to convert the organic carbon measurement to a mass of organic compounds. Calcium is assumed to be present in the form of gypsum and iron is used as a tracer for other entrained crustal/road dust materials. The factors by which iron concentrations are increased are determined empirically from the data and increase between roadside samples and urban background, reflecting a greater enrichment of iron in traffic-related particles. Both sulphate and nitrate are assumed to be associated with strongly bound water and their masses are increased by an empirically estimated factor to allow for this. Use of this method has accounted for around 100% of measured particle mass and a high proportion of the variance in daily mass concentrations for both roadside and urban background locations.

2.5 Source apportionment

55. Source apportionment refers to the quantitative attribution of airborne pollutant concentrations to specific source categories. It, therefore, deals with particles measured in the air. This contrasts with source inventories, which quantify the amounts of pollutants emitted but take no account of either dispersion conditions and, therefore, effects on ground-level concentrations – or secondary formation. The results of source apportionment studies are key to targeting cost-effective mitigation measures at those pollutant sources making the greatest contributions to air pollution.

2.5.1 Dispersion modelling

56. One means of pollutant source apportionment is to construct a spatially disaggregated inventory of emissions of that pollutant and then model dispersion of the emissions. By such means the concentration attributable to each emission source can be estimated. This procedure can be very effective in the case of pollutants such as SO_2, which have well defined and quantified sources. However, for airborne particles, emissions inventories tend to be incomplete or uncertain and dispersion modelling alone is unlikely to provide a very good estimate of airborne particle concentrations and thus source apportionment.

2.5.2 Receptor modelling

57. Receptor modelling refers to the use of measured air quality data to achieve source apportionment of airborne material such as particles. A number of different approaches have been used for receptor modelling of airborne PM. These include the following:

- In its Third Report, in 1996, the QUARG estimated the proportion of airborne particles derived from road vehicle emissions by examining the correlations between PM_{10} and $PM_{2.5}$ and either NO_x or carbon monoxide (CO), both of which are assumed to arise almost wholly from road traffic emissions within urban areas. The assumption on which this work was based was sounder at the time of the QUARG work than it would be now: these days a greater proportion of urban emissions of NO_x and CO arise from non-traffic sources. Linear regression analysis gives rise to a relationship in the form:

  $$PM_{10} = mNO_x + \mathrm{c}$$

  The intercept, c, relates to that part of PM_{10} that does not vary with NO_x emissions and, therefore, is assumed not to arise from road traffic. The road traffic contribution is, therefore, determined by difference (PM_{10} – c).

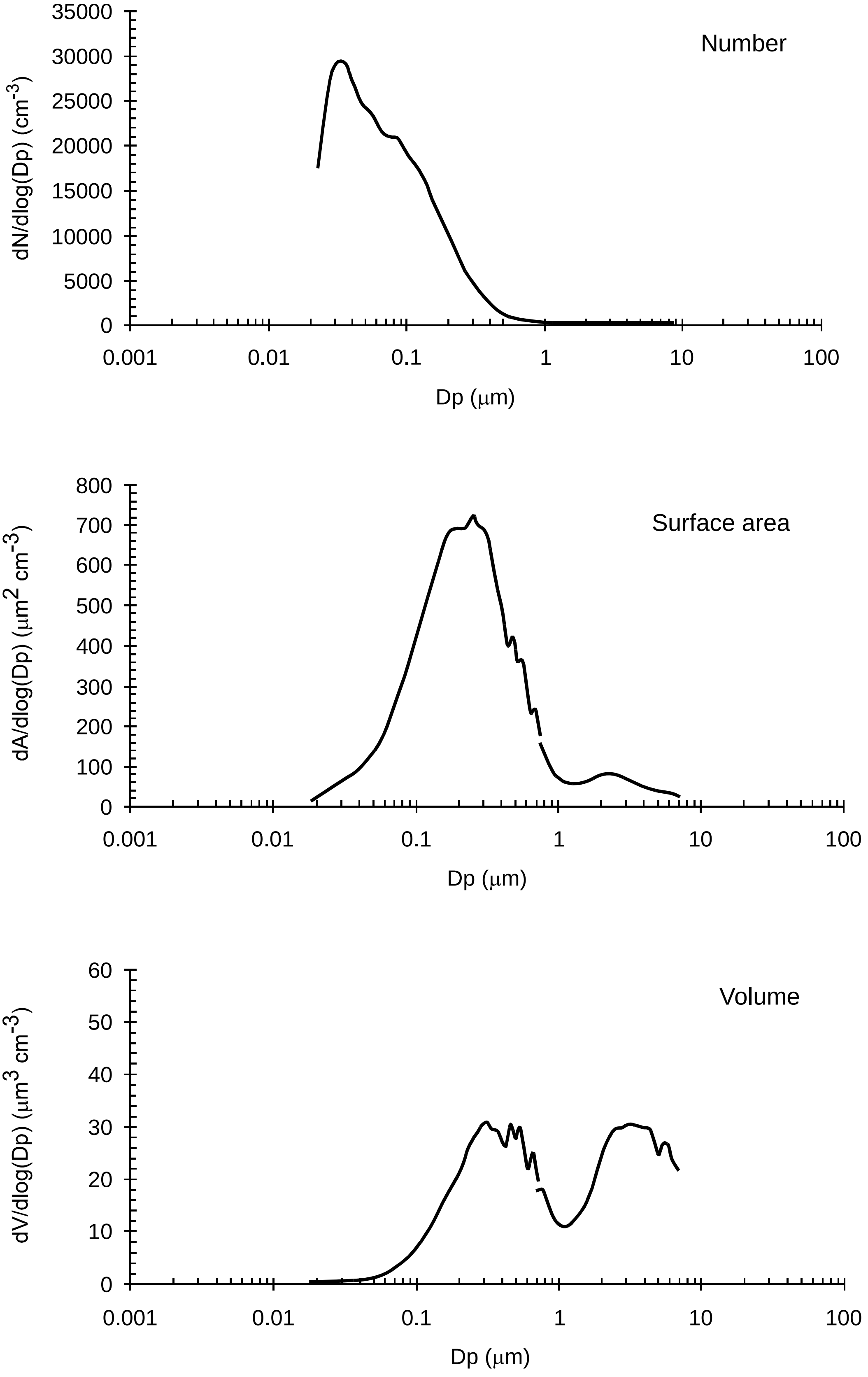
- Using major components as source indicators. The receptor modelling method adopted by APEG (1999) used sulphate as a surrogate for all secondary inorganic particles and then black smoke (later NO_x) as a surrogate for emissions from combustion sources that were largely but not solely road traffic. They then estimated scaling factors to account for the unmeasured part of the pollution from these sources. The third category of 'other', mainly coarse, particles was estimated by the difference. In the simplified mass closure method outlined in Section 2.4, this method is extended to include sulphate that arises mainly via secondary formation from SO_2 oxidation, nitrate from NO_x oxidation, elemental and organic carbon representing predominantly the emissions from combustion processes, sea salt and two types of crustal/soil/road dust particles indicated by calcium and iron. Furthermore, the organic to elemental carbon ratio can be used to distinguish between secondary and primary organic carbon.

- Multi-element analysis and multivariate statistical methods. There are a number of related source apportionment methods that depend upon the collection of a large number of sequential, normally daily, samples of airborne particles that are analysed for a wide range of chemical elements, often together with major ions and possibly some organic compounds (Hopke, 1985). The dataset is then analysed using a multivariate statistical method designed to recognise those components with a common temporal variation and with the capacity to aggregate the different chemical components into a smaller number of 'factors' that together account for a large proportion of the variance within the dataset. Thus, for example, sulphate, nitrate and ammonium are often identified within a single factor that corresponds to long-range transport of secondary components. Until recently road traffic emissions were generally identified through a factor heavily influenced by lead and bromine, reflecting the use of lead antiknock additives and brominated scavengers in petrol, a practice now discontinued in the UK. Although the method will aggregate individual chemical elements into factors with common sources, it requires expert judgement to associate the factors within given source categories. This is often dependent upon the association of trace elements with emissions from specific sources. For example, arsenic and selenium have typically been used as tracers for coal combustion sources, whereas vanadium and nickel indicate emissions from fuel oil combustion.

Although this technique has, in the past, generated relatively clear-cut quantitative information about the source apportionment of airborne particles, nowadays it appears to be far less effective. As a result of pollution control measures, trace elements have reduced greatly in concentration and now often arise in very small amounts from multiple sources. Additionally, elemental tracers such as lead, which in the past were very effective as source markers, are now no longer in use and inorganic elements do not provide a ready means of identifying road traffic emissions.

- Organic source tracers. Research in the United States (US) has demonstrated that certain trace level organic compounds can be very effective tracers of specific sources (Schauer *et al*., 1996). Thus, for example, hopanes and steranes, – complex substances present at low levels in petroleum fuels – are used as tracers of hydrocarbon fuel combustion (Fraser *et al*., 2003). Certain PAHs can be related to individual combustion sources, for example retene arises mainly from the combustion of wood. The application of such organic source tracers in a quantitative manner requires very detailed information on their abundance in emissions from all major source categories. Currently, they have only been applied in a detailed way to the source apportionment of PM in California; their use in Europe has yet to commence.

Figure 2.2: Particle size distribution measured in Birmingham



Chapter 3

What is causing the health effects of particles?

Key points

- Both short-term and long-term exposure to ambient levels of PM_{10} are consistently associated with respiratory and cardiovascular illness and mortality as well as other ill-health effects. The associations are believed to be causal.

- It is currently not possible to discern whether there is a threshold particle concentration below which there are no adverse effects on the whole population's health.

- The effects on health are likely to be dominated by exposure to particles in the long term. Using the health effects coefficients published by COMEAP (1998), estimates for the UK indicate that short-term exposure to the levels of PM_{10} prevalent in 2002 led to 6,500 deaths and 6,400 hospital admissions brought forward that year, although it is not possible to discern by what length of time the events were brought forward. Coefficients published by COMEAP (2001) also indicate that for each 1 $\mu g\ m^{-3}$ decrease in $PM_{2.5}$ over the lifetime of the current population of England and Wales between 0.2 and 0.5 million years of life will be gained (COMEAP, 2001), equivalent, on average, to 1.5–3.5 days per individual. These estimates assume both linearity in the relationship between particle concentration and health outcome and the absence of a threshold.

- The effects of particles on health are likely borne predominantly by susceptible subgroups of the population, such as those with pre-existing lung, heart or other disease and/or the elderly and children.

- The balance of evidence currently available suggests that it is combustion-derived components of PM_{10} – which are comprised predominantly of fine and ultrafine carbon-containing particles and may be enriched with trace metals or specific organic compounds – that are primarily responsible for the harmful effects.

- There is generally less evidence to connect secondary inorganic PM and coarse particles with adverse health effects. However, the latter, in particular, cannot be ruled out since certain sources of these particles may be enriched with components of putative high risk (for example, soluble trace metals). The coarse fraction also contains biological material such as pollen and may be proportionally enriched with endotoxin, both of which factors can lead to adverse health effects.

- The lack of information on quantitative relationships between adverse health effects and specific components of PM_{10} (if different from the relationship with total PM_{10}) makes it difficult to assess the impact of future changes to the PM_{10} mix, arising from strategies aimed at reducing PM_{10} overall, upon population health.

- Recently, the World Health Organisation (WHO) has recommended the development of air quality guidelines for $PM_{2.5}$ alongside the retention of measurement of PM_{10} for public health protection. Health-based guidelines for $PM_{2.5}$ should be derived with the use of $PM_{2.5}$ data since simple scaling of the PM_{10} limit value using the prevailing $PM_{2.5}/PM_{10}$ ratio does not lead to extra targeting of health benefits. WHO also recommended a re-evaluation of the value of black smoke as an indicator for traffic-related air pollution.

- Experimental evidence suggests that one mechanism by which particles exert their effects is by causing oxidative stress (probably via the generation of free radicals), which leads to inflammation.

3.1 Introduction

58. The evidence for adverse human health effects associated with both short-term (over several days) and long-term (over many years) exposure to ambient PM_{10} has been comprehensively reviewed elsewhere (COMEAP, 1998; WHO, 2000; EPAQS, 2001; WHO, 2003; USEPA, 2004). The adverse health effects include respiratory morbidity (wheeze, reduced lung function) and mortality; cardiovascular morbidity and mortality; and cancer. There is a widespread consensus that associations are causal.

59. In order to provide a context for the rest of the report, this chapter briefly reviews current understanding of the health effects of exposure to PM_{10} and its constituents in the UK. It does not draw new conclusions about the health effects of particles. The literature cited is representative, not exhaustive.

60. Current standards for ambient PM in the UK and elsewhere in Europe relate to PM_{10}, which consequently provides the vast majority of data against which adverse health effects have been quantified. However, ambient PM_{10} is physically and chemically diverse, as is described in Chapters 2 and 6, and it has generally not yet been possible to establish causal relationship(s) between the health effects and specific properties of PM_{10} that are more convincing than the relationship with total PM_{10} inhaled. Some of the chemical species contained within PM_{10} (for example, trace metals and polycyclic aromatic hydrocarbons) can exert considerable toxicity in their own right. However, exposure to PM causes adverse health effects that often exceed those that can be predicted on the basis of the toxicity of the known individual constituents. The review in this chapter of possible effects of chemical species within PM is in the context of this non-specific toxicity rather than the well-known toxicity of individual chemical components.

61. Epidemiological studies have formed the basis of PM standard setting because relating daily or longer term mass concentrations of particles to a wide range of health endpoints is relatively straightforward. Such studies often embrace the whole population and thus reflect the full range of individual sensitivity to inhalation of ambient concentrations of PM. However, the diverse sources and spatial and temporal variability of PM_{10} can complicate the accurate quantification of individual and aggregate-level exposure. A disadvantage of the empirical methodology of epidemiology is that considerable monitoring data and a large study population are required to yield sufficient statistical power to narrow the confidence intervals (CIs) of results. Consequently, epidemiology is inevitably measurement-led and a lack of epidemiological evidence for an effect or not of particular component(s) of PM_{10} often reflects a lack of sufficient data to test.

62. Toxicology can be used as an alternative approach to understanding the adverse health effects of PM and may involve *in vivo* animal exposures or *in vitro* studies. The advantage of a toxicological approach is control over exposure to individual components or mixtures, but doses are usually high and quantitative extrapolation to human health effects from ambient doses is difficult. In some studies, particle mixtures other than that found in ambient air have been used, and extrapolating from such studies to predict the effects of ambient particles is even more difficult. Nevertheless, toxicology can provide insight into the potential biological cause-effect pathways of specific PM_{10} components and subfractions.

3.2 The health impact of particles in the UK

63. Quantitative estimation of health impacts from exposure to PM assumes linearity in the relationship between exposure and responses as well as the absence of (or existence of a very low) threshold of exposure below which there is no effect.

64. COMEAP has used estimates of population exposure-response relationships from time-series studies to quantify the number of deaths or hospital admissions for respiratory disorders attributable to short-term exposure to PM_{10}. COMEAP (1998) estimated that at the concentrations that prevailed in 1996, exposure to PM_{10} in urban areas in the UK (excluding Northern Ireland) caused 8,100 deaths (all causes) to be brought forward and 10,500 hospital admissions for respiratory complaints to be either brought forward or additionally caused. Using the same COMEAP health-effects coefficients, but basing PM_{10} concentrations on the 2002 maps presented in Chapter 8, updated calculations indicate that short-term exposure to PM_{10} in 2002 in the UK caused 6,500 deaths to be brought forward and 6,400 hospital admissions to occur that were either brought forward or would not have otherwise have happened. Additional morbidity effects include the increased use of bronchodilators on poor air quality days.

65. For these acute responses it is not possible to establish either the extent to which death is brought forward or the split of hospital admissions between those that were brought forward and those that would not have occurred in the absence of exposure to PM_{10}. It may be that the major effect of short-term exposure to PM_{10} is to hasten the death of the elderly and the sick, in which case the impact of air pollution episodes on mortality is relatively small. However, in some cases deaths may be brought forward by weeks or months rather than days.

66. There are very few cohort studies[1] on the chronic health effects of particles in the UK/Europe from which to estimate the health effects from long-term exposure to particles. COMEAP (2001) has applied results from US cohort studies on $PM_{2.5}$, but it is important to stress that the impact of pollution could vary between countries because of differences in particle composition and population lifestyle.

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[1] In a cohort study, the exposure to particles, and the health status, is assessed over time for a subpopulation of specific individuals.

The long-term health effects appear to be dominated by cardiovascular rather than respiratory mortality. The quantitative estimate determined by COMEAP to be most likely, based on current knowledge, is a gain of 0.2–0.5 million life years for the population of England and Wales in 2000 for every 1 $\mu g\ m^{-3}$ decrease in annual mean $PM_{2.5}$ over its lifetime. (The range incorporates different assumptions in lag time between hazard reduction and health gain). This estimate corresponds to an average of 2.5 days gained per person for all 52 million people, or 25 days for 5 million people, or 4.5 months for 1 million people and so on, or a mixture of these. Although it is likely that effects are borne only by a subset of the population, the particular distribution of life expectancy gains across individuals cannot yet be identified.

67. The above data indicate that life expectancy gains from the reduction of exposure to particles in the long term are about tenfold greater than for a similar magnitude reduction in exposure over the short term. Uncertainties within the long-term estimates include undetected confounding[2], the lag time and the extent to which the long-term health outcomes are driven by the much higher exposures to PM_{10} air pollution in the past (Brunekreef and Holgate, 2002).

3.3 Evidence for adverse health effects from specific physical/chemical components of PM_{10}

3.3.1 Fine particles ($PM_{2.5}$)

68. Time-series epidemiological studies, in the UK and elsewhere, that use short (0–3-day) lag times have shown that $PM_{2.5}$ and black smoke are at least as predictive as PM_{10} for certain acute health endpoints.

69. In the West Midlands conurbation, analysis of hospital admissions by age over the period 1994–1996 showed evidence of associations between daily PM_{10}, $PM_{2.5}$ and black smoke and respiratory admissions in the 0–14 years age group (Anderson *et al*., 2001). The results for black smoke were more robust between different models than for PM_{10} or $PM_{2.5}$, whereas the coarse fraction ($PM_{10-2.5}$) showed smaller and less consistent associations. Recent updates to this study show stronger associations between adverse health effects and $PM_{2.5}$ and strengthen the conclusion that the active component resides mostly in the fine fraction.

70. In Edinburgh during 1981–1995, a significant positive association was found between black smoke (mean of previous 3 days) and daily all-cause and respiratory mortality in people aged 65 or over, in addition to a significant positive association between PM_{10} and cardiovascular hospital admissions also for people in this age group (Prescott *et al*., 1998).

71. As indicated in Section 3.2, there are no published data on the health effects of long-term exposure to $PM_{2.5}$ (or even to PM_{10}) in the UK. Of the US studies

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[2] A confounding factor is a condition or variable that is both a risk factor for disease and associated with an exposure of interest. This association between the exposure of interest and the confounder may make it falsely appear that the exposure of interest is associated with the disease.

used by COMEAP to estimate the health effects of long-term exposure in the UK, the Harvard Six Cities cohort study found $PM_{2.5}$ was significantly associated with cardiopulmonary and cardiovascular mortality (with a non-significant increased risk for $PM_{2.5}$ and lung cancer) (HEI, 2000). The American Cancer Society cohort study also showed a significant association between $PM_{2.5}$ and all-cause and cardiopulmonary mortality as well as lung cancer mortality, but no association between coarse particles ($PM_{15-2.5}$) and mortality (Pope *et al*., 2002) was found.

72. In the US, healthy human volunteers exposed for 2 h to average fine particle concentrations of 200 μg m^{-3} (inhaled as concentrated air particles) showed transient mild pulmonary inflammation, although no other indicators of respiratory symptoms or decline in pulmonary function (Ghio *et al*., 2000). Urban fine concentrated air particles also cause lung inflammation and injury in rats (Clarke *et al*., 1999). This work provides support for the assumed causality of the associations observed in epidemiological studies.

3.3.2 Ultrafine particles ($PM_{0.1}$)

73. Very few data on exposure to ultrafine particles ($PM_{0.1}$) exist on which to base epidemiology. Although a number of sites in the UK now have monitors for both total and size-resolved particle count there are probably still insufficient data for epidemiological analysis.

74. In studies with German and Finnish asthma patients, the exacerbation of symptoms had a stronger association with daily fluctuations of ultrafine than with fine particles. A study on daily mortality in Germany showed fine and ultrafine particles had comparable effects, although the fine particles had more immediate effects and the ultrafine particles showed effects between particle concentration and mortality that were delayed for a few days. The immediate effects were clearer in respiratory cases, whereas delayed effects were clearer in cardiovascular cases (Wichmann and Peters, 2000).

75. A number of *in vitro* and rat *in vivo* studies have shown that monodisperse ultrafine particles composed of, for example, carbon black, nickel or cobalt are capable of inducing relevant inflammation responses, in which the production of damaging reactive oxygen species (ROS) play a role (Dick *et al*., 2003).

76. To rationalise the cardiovascular effects associated with particle exposure, it has been hypothesized that, as well as initiating inflammation responses in the lung, ultrafine particles can pass through into the blood circulation and act systemically (Seaton *et al*., 1995). Measurements of the blood of volunteers who have inhaled radioactively labelled ultrafine carbon particles has recently provided evidence for this (Nemmar *et al*., 2002).

3.3.3 Coarse particles (PM_{coarse})

77. Some time-series studies have provided limited evidence that coarse particles are associated, independently of fine particles, with mortality and morbidity endpoints such as respiratory hospitalizations. It is likely that endotoxin is more closely associated with aged crustal coarse particles and contributes to some of the health effects of this fraction along with other biological material such as pollen and spores, which have known health effects. In addition, certain sources of coarse particles may be enriched with components of putative high risk such as trace metals (Section 3.3.5). Some coarse particles may contain agglomerations of fine particles, or fine particles or semi-volatile components adsorbed onto larger particles, that may disaggregate within the lung. Agglomerations of fine particles may also retain the high surface area and surface chemical activation of the subunits.

3.3.4 Total surface area of particles

78. A linear association between daily mortality rate and the estimated surface area concentration of coalescing particles has been demonstrated using historic UK black smoke time-series data and a simple model of the evolution of particle distributions (Maynard and Maynard, 2002). The analysis also indicated the existence of a threshold particle concentration below which particle mass and surface area concentrations were linearly related. Below this threshold (estimated to be ~35 μg m^{-3} for fine particles) mass concentrations may provide a good indicator of health effects and mask the influence of surface area, if the latter is the controlling property, whereas at high particle concentrations effects seemed to be more closely related to surface area. Present day fine particle concentrations in the UK are usually well below this threshold, but the size distribution of particles is unlikely to be uniform everywhere and consistent with this simple model, so the appropriateness of a surface area metric cannot be ruled out.

79. *In vivo* rat studies using monodisperse polystyrene particles with a diameter in the range of 64–535 nm have yielded a linear relationship between the surface area of particle dose instilled against different parameters of lung inflammation (Brown *et al*., 2001). These findings support the hypothesis that small particles, even of presumed low toxicity material such as polystyrene, can exert inflammatory activity in proportion to their large specific surface area.

3.3.5 Chemical constituents of particles

3.3.5.1 Sulphate, acidity and other major inorganic ions

80. In the West Midlands time-series analysis, health associations with particle sulphate were similar to those of $PM_{2.5}$ although, in both instances, clear effects were difficult to discern. In the US, the Harvard Six Cities cohort study also showed that particle sulphate was significantly associated with long-term cardiopulmonary and cardiovascular mortality. The similar associations for sulphate and $PM_{2.5}$ mean it is possible that sulphate is a surrogate for other components.

81. The evidence from toxicity studies is for low or undetectable toxic potency for sulphate and other major inorganic components of particles, such as nitrate and chloride (Schlesinger and Cassee, 2003).

3.3.5.2 Trace metals

82. Both toxicological studies and *in vitro* assays modelling inflammation have implicated transition metals (often, but not exclusively, the water soluble component) as being the causative agent(s) of cardiopulmonary injury in both healthy and compromised rats. Many of these studies have used real samples of PM_{10} or $PM_{2.5}$ and have shown that the effects are blocked by the addition of metal chelators to the samples. The balance of mechanistic evidence suggests that the inflammation is promoted by ROS generated by chemical cycling of the metals between different oxidation states.

83. A time-series epidemiological study in the UK showed no positive association between the daily trace metal composition of PM_{10} or $PM_{2.5}$ and respiratory and cardiovascular mortality and morbidity for one year of measured data (1999–2000) (Beverland *et al.*, 2002). The time-series of metal exposure was extended retrospectively for eight years by estimating daily metal concentrations in PM_{10} using the significant relationship of daily metal concentration with air mass back trajectory coordinates. For this 8-year dataset there was a significant association between some of the metal concentrations (for example, water-soluble and total iron, copper and zinc, water-soluble vanadium and total nickel) and cardiovascular admissions, although associations did not remain significant after adjusting for PM_{10} mass. The loss of statistical power is a consequence of a residual strong correlation between metal concentration and PM_{10} mass and limitations in the ability of air mass back trajectory coordinates to explain all variations in particle metal.

84. In the same study, the concentration of a number of trace metals, such as copper, analysed in PM_{10} (and in $PM_{2.5}$) correlated more strongly with the gravimetric concentration of black smoke than with PM_{10} or $PM_{2.5}$, indicating a source of these metals associated with fine primary combustion particles. This is a potential causal link, consistent with other observations that health effects are associated with combustion or traffic-related particle sources.

85. In another UK study, daily samples of PM_{10} taken from six sites with an anticipated variation in ambient particle composition were either instilled in rats *in vivo* or used in *in vitro* cell cultures (Lightbody *et al.*, 2003). Although markers of *in vivo* and *in vitro* inflammation were most strongly influenced by PM_{10} mass dose, analysis of the results based on PM_{10} composition showed that the mass of primary particles within PM_{10} (estimated using the Netcen source apportionment model, see Chapter 8) was a strong factor in determining potency, whereas secondary and coarse PM_{10} mass were not. Certain trace metal components – most notably water-soluble zinc, but also water-soluble manganese and the sum of water-soluble trace metal analysed – were also independently significant predictors of a number of toxicological endpoints.

3.3.5.3 Organic components

86. Carbon and organic compounds are major constituents of combustion-generated particles and these, together with secondary organic aerosol, comprise a substantial proportion of PM_{10}. Some organic compounds associated with PM from combustion sources, for example, PAHs and their oxy- and nitro-derivatives, exert inflammatory as well as mutagenic and carcinogenic effects. Oxygenated organics such as quinones can yield ROS. Plausible mechanisms therefore exist to link the organic components of PM_{10} with both acute and chronic adverse health effects. However, there are insufficient measurement data for individual organic components and their variability, in the UK or elsewhere, to attempt an epidemiological quantification of effects, if any.

3.3.6 Interactions between physicochemical properties of particles

87. Some toxicological studies have investigated the net effect of mixtures of model particles and chemical constituents. For example, the ability of monodisperse ultrafine carbon black particles to generate ROS *in vitro*, or rat lung inflammation *in vivo*, is synergistically enhanced by the presence of soluble iron or copper salts (Wilson *et al*., 2002). A mechanistic suggestion for the effect *in vivo* is that selective adsorption of lung surfactant onto carbon particles reduces rates of particle clearance from the lung and increases the effective exposure time. There is also some evidence that sulphate potentiates metal effects: increased metallic ion availability in an acidic environment is a possibility.

3.3.7 Particle source

88. Some epidemiological studies have investigated the health effects of particles categorised by source, rather than by specific physicochemical property. These generally use either multivariate models (for example, factor analysis) or geographical information systems (GIS)-based methods to apportion particles and thus exposure to different sources. Studies of this type have not yet been applied to UK data, although a recent study of the toxicology of PM_{10} from six different UK locations (Section 3.3.5.2) suggests that the concentration of particles, apportioned by modelling, as being combustion-related primary particles was a significant predictor of some toxicological endpoints. US studies have shown that daily mortality in six cities is significantly associated with concentrations of $PM_{2.5}$ apportioned by factor analysis to mobile and coal combustion sources but not to crustal sources (Laden *et al*., 2000).

89. A cohort study in the Netherlands indicated that traffic-related air pollution (as assessed by GIS data for the distance subjects lived from main road) was significantly associated with increased cardiopulmonary mortality (Hoek *et al*., 2002). The relative risk for living near a major road was 1.95 (95% CI: 1.09 – 3.52). A study in Sweden observed a relationship between motor vehicle emission (estimated by dispersion modelling) and lung cancer (Nyberg *et al.*, 2000). However, it is important to note that neither of these studies sought to separate the contribution of PM from other traffic-related pollution. A study in Amsterdam showed the relationship between mortality and black smoke was twice as steep for subjects on the main road network compared to subjects living elsewhere (Roemer and van Wijnen, 2001), suggesting traffic was associated with these effects.

90. Both types of source-oriented approach consistently identify an association between traffic-derived particles and adverse health effects. The observation that black smoke (generally taken as a current marker for traffic- and combustion-related primary PM) is associated with ill health is consistent with the outputs of these models.

91. Short duration (2 h) controlled human exposure studies with diesel exhaust have shown clear, although mild, local inflammatory effects in the respiratory tract as well as systemic effects in both healthy or asthmatic individuals. A small epidemiological study, mainly based on US railroad workers, has shown a small effect of diesel particles on lung cancer. The mechanism may be a combination of particle-induced oxidative inflammation and the carcinogenic potential of soot-attached PAH. Diesel exhaust does not of course consist exclusively of particles; animal tests vary in their indications of whether PM is the most important fraction of the diesel exhaust, although inflammation of airways has been shown in healthy human volunteers exposed to resuspended diesel exhaust particulate only (Nightingale *et al.*, 2000).

3.4 Summary of current understanding

92. Epidemiology has consistently demonstrated an association between adverse health effects and PM_{10}. A recent WHO meta-analysis of 33 time-series studies done in Europe gives an estimated 0.6% (95% CI: 0.4–0.8%) increase in total mortality effect for a 10 $\mu g\ m^{-3}$ increase in daily PM_{10} (WHO, 2004). The corresponding values for respiratory and cardiovascular deaths are 1.3% (0.5–2.0%) and 0.9% (0.5–1.3%), respectively. These values are similar to those reported previously in the APHEA (Air Pollution and Health, a European Approach) study in Europe and in the NMMAPS (National Morbidity, Mortality and Air Pollution Study) study in the US. For comparison, the COMEAP (1998) exposure-response coefficients used in the calculations for the UK given in Section 3.2 are 0.75% for increase in all-cause mortality and 0.80% for increase in respiratory hospital admissions.

93. The recent evidence from time-series studies is that the displacement of daily mortality and hospital admissions is not of just a few days. An increase in effects estimates with increasing duration of exposure to PM_{10} is consistent with the larger health impacts observed from long-term studies.

94. Despite their broadly consistent adverse health associations, the APHEA and NMMAPS studies both indicated some spatial variability in effects estimates between cities. These can be attributed to differences, *inter alia*, in mean temperature (leading to differences in domestic ventilation and proportion photochemical particles) and the proportion of traffic-related particles. Another source of variability is the distribution of exposures received by individuals making up the population.

95. The apparent general uniformity of time-series risk factors has been used to argue that individual components of PM are of secondary importance to total inhaled mass (Harrison and Yin, 2000). However, although it has generally been difficult to show convincingly that attributes other than size are more important determinants of ill health than others, this may simply reflect the dominance of measurements of PM by mass in the available data. The ability of studies to discriminate between effects of different fractions is also limited when there is a high degree of correlation between different measures of PM.

96. Nevertheless, the balance of evidence suggests overall that it is the combustion-derived components of PM_{10}, which are comprised predominantly of fine and ultrafine particles and may be metal and PAH (and other organic compound) enriched, that are primarily responsible for the harmful effects. These particles also have a high specific surface area. This component of fine/ultrafine carbonaceous particles have low optical reflectivity that likely also explains the associations of health effects with traditional black smoke measurements. Re-evaluation of the value of black smoke as an indicator for traffic-related air pollution has been recommended by WHO (2003).

97. The relationship between ultrafine particles and PM_{10} in ambient air is complex and consequently the assertion that ultrafine particles provide the underlying explanation for the generally consistent associations of PM_{10} with adverse health effects might appear to be a contradiction. A possible rationalisation of this is the considerable spatial and temporal variability of ultrafine particles and the consequent difficulty in correctly quantifying exposure to this component. Data from Germany show that although PM_{10} and $PM_{2.5}$ have declined with time, ultrafine particle concentrations have remained relatively constant and the smallest size fraction of ultrafine particles has continuously increased in the last decade (Wichmann and Peters, 2000). Particle number, and particle surface area, are metrics whose associations with health endpoints have yet to be properly explored, particularly in the UK.

98. There is generally much less compelling evidence to connect secondary inorganic particles (that is, ammonium and sodium nitrates and sulphates) and coarse particles with adverse health effects. Associations with sulphate have predominantly come from US studies where sulphate may have been a surrogate for particle acidity. The latter is less of an issue in the UK where particulate matter is generally well neutralised by ammonium.

99. Despite the evidence pointing to finer particles, health effects from coarse particles cannot yet be excluded.

100. Given the evidence emerging for the importance of exposure to traffic-related particles, the public health burden of exposure at such roadside ‘hotspots’ may be inadequately assessed by the characterization of population exposure using background urban concentrations.

101. An important issue is that of interactions, either between different PM components, and/or between different pollutants. In general, the evidence suggests that associations between PM and health effects are due to PM and that PM is not acting as a surrogate for something else. For example, results from many time-series studies are relatively insensitive to adjustment for a number of gaseous pollutants and human and animal exposure studies also show a direct effect of PM. On the other hand, in the APHEA study, adjustment for NO_2 reduced PM effect estimates by about half, suggesting that PM_{10} or $PM_{2.5}$ mass alone is not sufficient to represent fully the impact of complex air pollution mixtures (Katsouyanni *et al*., 2001). This may be due to a direct effect of NO_2, but could also be due to NO_2 acting as a surrogate for an unknown traffic-related pollutant. In the latter case, NO_2 may provide a better measure of the more toxic (traffic-related) component(s) of PM than does PM_{10}.

102. Mechanistically, the evidence suggests that in *vivo oxidative* stress caused by particles elicits inflammation and drives the respiratory and cardiovascular effects. In the case of ultrafine particles, this may include systemic effects from direct transfer into the vascular system. The mechanistic link to cancer is less clear but PM yields DNA adducts, which provide unequivocal evidence of exposure and, if the material analysed for adducts includes acids, may provide evidence of these genotoxic compounds reaching sites where genotoxic effects could be expressed.

103. The consistent demonstration of population health effects with PM_{10} indicates that PM_{10} is a relevant metric for air quality standards in spite of the important issues of PM_{10} composition and the relationship between ambient and personal exposure to PM_{10}. However, as part of their review of the health effects of particles for the CAFE programme (CAFE, 2004), WHO (2003) recommended the development of air quality guidelines for $PM_{2.5}$ alongside the retention of measurement of PM_{10} for public health protection. A health-based $PM_{2.5}$ limit value should be derived from $PM_{2.5}$ data since scaling the PM_{10} limit value according to the prevailing $PM_{2.5}/PM_{10}$ ratio assumes that both fractions have the same toxicity and leads to no extra targeting of health benefits. Furthermore, given that the coarse fraction of PM cannot currently be considered innocuous, development of a $PM_{2.5}$ target would need to be accompanied by health-based targets for the coarse fraction either indirectly via retention of a target for PM_{10} (as recognised by WHO) or directly via a target for PM_{coarse} instead of PM_{10}.

104. Without a better understanding of the relationships between specific components of PM_{10} and their health effects, it is difficult to assess the impact upon population health of future changes to the PM_{10} mix that may result from strategies to reduce PM_{10} overall.

105. There is general consensus that some health effects are dominated by those in susceptible subgroups, for example, the elderly, children, those with pre-existing lung or heart disease or (possibly also) diabetes. These groups are more likely to succumb to the very low extra inflammation they receive from exposure to particle pollution.

106. A consequence of the general agreement that no lower threshold is yet discernible in the exposure-response relationship for particles is that setting an air quality standard becomes a somewhat subjective judgement. This implies that an approach based on reducing the general exposure of the population to particles, rather than a limit value approach, which focuses on hotspots, should be more effective at maximising the total population health gain (CAFE, 2004).

Chapter 4

Sources of particles in the UK

Key points

- The source emissions relevant to atmospheric concentrations of particles are related to a complex multi-pollutant mixture that is present over a range of different particles sizes. These sources include primary, directly emitted, material and secondary material that is formed subsequently in the atmosphere. Particle sources can also be grouped into those originating from anthropogenic activities and those that are natural in origin.

- Total UK emissions of PM_{10} in 2001 were 180 kilotonnes (180 kt). Transport (in particular road transport) and industrial processes are the two most important sources of PM_{10} and each account for 27% of total UK emissions. Emissions from the domestic combustion sector (17% of the total) are also important.

- Considering the road transport sector, tyre and brake wear emissions in 2001 accounted for ~23% of the total road transport emissions. Resuspended road dust also makes an important contribution, one that is not accounted for in national estimates of UK total PM_{10} emissions. Recent work suggests that the resuspended component of PM_{10} can be as large as, and in some cases much larger than, exhaust emissions. Resuspended PM_{10} is, however, variable in magnitude and difficult to quantify with any certainty. In addition, both tyre and brake wear and resuspended PM_{10} appear to be much more important for heavy duty vehicles than for light vehicles such as cars. The type approval test for particulate emissions is only applicable to compression ignition engines. No legislation is currently in place to limit the particulate emissions from spark ignition engines. PM emissions associated with spark ignition engines are not well defined, but although they do not represent significant emissions in terms of mass, they can represent an important source of ultrafine particles.

- Type approval PM measurements for road vehicle emissions testing are based upon a gravimetric filter-based method that is measured at 52°C. No attempt is, therefore, made to include the volatile component.

- For road vehicle emissions testing, the emission sampling strategy can lead to the formation of sampling artefacts. Whereas various PM metrics could be measured at the point of formation or emission, limited work has been undertaken to characterise the changes in PM morphology between the point of emission and the ultimate receptor. The sampling strategy and instrument choice will dictate the measured PM emission value. International programmes across the EU go someway to provide standardised transient test protocols. Measurements undertaken outside these standardised in-service test protocols must be treated with caution.

- The shipping sector makes an important contribution to EU emissions of SO_2, which is an important particle precursor gas. Recent estimates suggest that by 2010 shipping will be equivalent to more than 75% of total land-based emissions of SO_2. Shipping emissions of SO_2 are also of local importance in the vicinity of major UK ports.

- The importance of each emission sector depends markedly on the particle size considered. For total UK emissions of coarse particles (PM_{10} – $PM_{2.5}$) in 2001, the industrial processes sector was most important (34%) followed by domestic combustion (19%). However, for particles smaller than $PM_{2.5}$, the road transport sector was the most important (39%), followed by industrial processes (21%). The road transport sector was even more important when particles smaller than $PM_{0.1}$ are considered: it accounts for 54% of total emissions.

- Between 1990 and 2000, total UK PM_{10} emissions, which have been quantified in the NAEI, have fallen by 42% from 309 to 178 kt. These reductions have been driven by a reduction in coal use in electricity generation and industrial processes. However, the decline in road transport emissions of 37% has also made a significant contribution to the overall decline.

- Total UK emissions of PM_{10} are expected to decline by 28% between 2000 and 2010, primarily due to a decline in power station and road transport emissions. Emissions of precursor gases are also expected to decline over this time period. SO_2 is expected to decline by ~52%, mostly as a result of the introduction of regulations to control the sulphur content of liquid fuels. The decline in emissions of NO_X by 30% is dominated by reductions in the road transport sector. For NMVOCs the estimated 28% reduction is mostly the result of reductions from industrial processes and further reductions from the road transport sector. Decreasing animal numbers and a reduction in fertilizer use is expected to reduce ammonia (NH_3) by 10%. Current projections suggest that total UK PM_{10} emissions will remain virtually unchanged between 2010 and 2020.

- There are important differences between the NAEI and the LAEI. Currently, the NAEI provides estimates for more sources than the LAEI, in particular emissions from vehicular tyre and brake wear and construction. However, increased use of local activity data in the LAEI is also likely to account for some of the differences reported.

- Current Department for Transport (DfT) projections forecast that diesel car sales will reach 42% by 2010, from 14% in 2000. The current NAEI forecast of 30% in 2010 means that emissions of PM_{10} will be higher than currently projected. Estimates suggest that UK urban emissions in 2010 will be 3.8% higher than currently projected. The effect on PM_{10} emissions is greater than that for emissions of NO_X, which are projected to be 1.5% higher on the same basis.

- At a local level, a preliminary analysis of the London Congestion Charging Scheme (CCS) estimates that exhaust emissions of PM_{10} have declined by ~12% in the charging zone as a result of reduced congestion and reduced flows of certain vehicle types. The 12% reduction is likely to underestimate the full benefit since the reduction in vehicle flows would also be expected to result in further reductions in PM_{10} resulting from tyre and brake wear as well as resuspended material, which have not yet been quantified.

- No legislation is in place to limit non-exhaust PM emissions through tyre, brake and clutch wear. Thus, given that tailpipe emissions are reducing, the relative contribution of non-exhaust emissions is increasing.
- Uncertainties in total national emissions of PM_{10} in 2001 range from –19% to +46% at the 95% CI. This range is considerably higher than that for the precursor gases: SO_2 (±3%), NO_X and NMVOCs (±8%) and NH_3 (+19% to –18%). It is also highly likely that the uncertainty estimates for particle size fractions less than PM_{10} will be higher than the uncertainty in PM_{10} emissions, although this uncertainty has not been quantified.
- A recent review of certain particle emissions by the NAEI has revised the estimates of particle emissions from important sources. Emissions from road vehicle tyre and brake wear have almost doubled, therefore increasing the non-exhaust particle emission rates. Emissions of PM_{10} from natural gas have been revised downwards, such that emissions are 25% of previous totals. The revision of the natural gas emission rates could significantly affect road source apportionment of particle emissions in urban areas. These revisions, together with others outlined in this report, will be implemented in the 2003 version of the NAEI.

4.1 Introduction

4.1.1 Scope

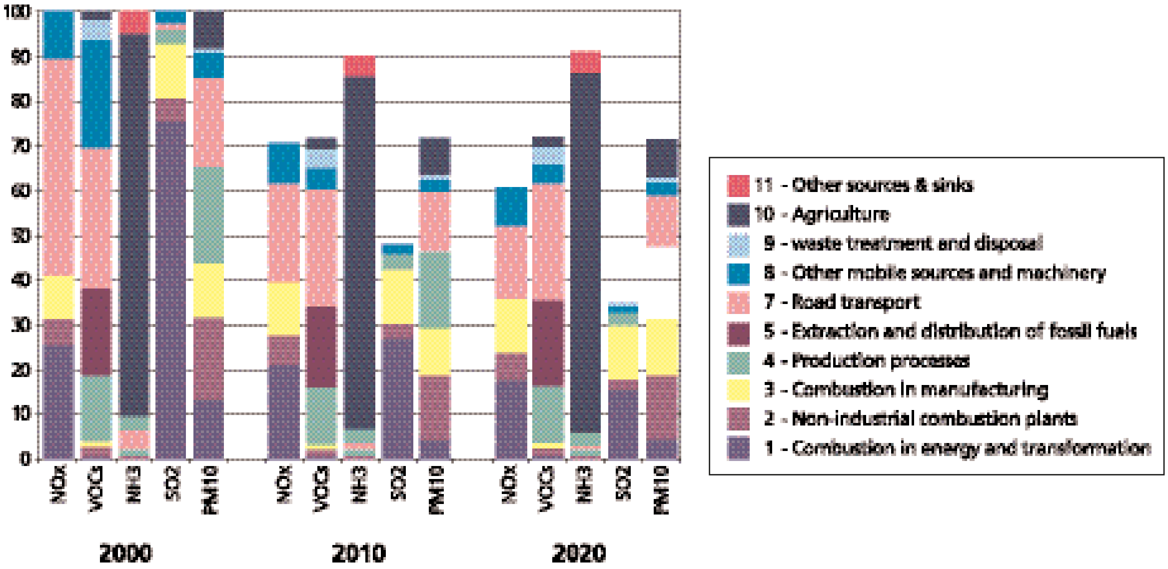
107. This chapter considers the sources of particle emissions in the UK and Europe. It focuses on the sources of emissions for which emission factors exist and is chiefly concerned with the emissions that are considered as part of national and local emissions inventories. Where available, the chapter highlights the characteristics of particle emissions from different source types, such as emissions rates and particle size distributions. Consideration is also given to both the emissions of important particle precursor gases, some of which are of natural origin, and the principal atmospheric chemistry routes that result in secondary particle formation.

108. There are many important natural sources of particles, which contribute significantly to ambient concentrations. Particles from these sources – for example, forest fires and wind blown resuspension – tend to be difficult to predict in terms of their magnitude, emission rate and the description of their temporal emission characteristics. Many sources of natural particles are essentially random in origin and are best described through atmospheric measurements and source apportionment studies. It is generally not possible to identify useful or meaningful emission rates for comparison with manmade sources. Nevertheless, emissions of these particles can, under certain conditions, make a large contribution to observed concentrations of particles and can be significantly more important than manmade primary particles that are generally well characterised in terms of their emission rate. The importance of some natural particle sources is discussed further in Chapter 6.

109. Key data have been considered from both UK and European particle emission inventories and these are presented as both emissions by source type and historical trend data. Where possible, emissions have been summarised by particle size fraction. National emission projections are also highlighted by source sector. Consideration has been given to the effectiveness of technical and non-technical measures to reduce particle emissions by highlighting different abatement technologies, such as vehicle particle filters, as well as the effects of different policies affecting particle emissions, such as the London CCS.

110. This chapter relies extensively on information contained in the NAEI. More detailed information relating to the methodologies for particle emissions inventory compilation and datasets can be found at the NAEI website at http://www.naei.org.uk. An overview of the principal primary and secondary sources of PM_{10} for 2000, 2010 and 2020 is shown in Figure 4.1.

Figure 4.1 UK emissions of PM_{10} and precursor gases relative to 2000 levels, based on a business-as-usual scenario.



4.1.2 Legislation and regulatory framework controlling emissions

4.1.2.1 Control of industrial sources of particles

111. Most industrial sources of particles are subject to control regimes set up by the Pollution Prevention and Control (England & Wales) Regulations 2000, the Pollution Prevention and Control (Scotland) Regulations 2000 and the Pollution Prevention and Control (Northern Ireland) Regulations 2003 which were introduced under the Pollution Prevention and Control Act 1999. Further information on these regulations can be found in the AQEG NO_2 report (AQEG, 2004) and a table listing how the major industrial sources of PM are covered by the different control regimes in England and Wales is available in Annex 1.

112. In general, the introduction of the regimes as a result of the Pollution Prevention and Control Act 1999 will not significantly change the types of conditions placed upon operators with regard to PM emissions from those issued under the older regimes set up by the Environmental Protection Act 1990.

113. For many processes where emissions of PM occur in a manner largely controlled through venting, the most important condition will be an emission limit, expressed in terms of a maximum allowable concentration of PM in the vented gases. For each process, emission limits are set by the regulators based on an assessment of what can be achieved using 'best available techniques not entailing excessive cost' (BATNEEC). In most cases the limits set are likely to correspond to those values suggested in guidance documents published by Defra and regulators. For example, the Environment Agency's guidance for combustion processes suggests that emission limits of 25 mg m^{-3} or less are achievable for coal-fired plants, depending upon the appliance type (Technical Guidance Note S3 1.01). Emission limits for most processes subject to LAPC are set at 50 mg m^{-3}.

114. Certain industrial processes are also subject to additional EC Directives, including waste incinerators and large combustion plants. The Waste Incineration Directive (2000/76/EC) replaces earlier directives and will set an emission limit of 10 mg m^{-3} for dust from nearly all UK incineration processes from 2006 onwards. In contrast, most plants are currently meeting emission limits of 30 mg m^{-3} or more.

115. The Large Combustion Plant Directive (2001/80/EC), which updated the earlier Directive 88/609/EC – introduced controls for combustion plants with an output at or above 50 MWth. New plants (those granted a licence after 1 July 1987) must comply with emission limits for PM, NO_X and SO_2. Existing plants must currently meet emission limits set out in the earlier directive but, from 2008 onwards, these plants will be required either to meet new emission limits or to comply with a 'national emissions reduction plan', which would achieve the same overall reduction in emissions. At the present time the Large Combustion Plant Directive is not transposed into UK law and the UK has still to decide how to achieve compliance. The latest UK Government statement indicates it hopes to adopt a modified approach that combines both these strategies, but whatever it decided in the end, individual plants may opt out and instead accept a limit on their operating hours for the period 2008–2015.

116. The Environmental Protection Act 1990 and the IPPC Directive also deal with emissions of the particulate precursor pollutants NO_X, SO_2 and NMVOC in a similar fashion to the controls placed on emissions of primary PM. As already mentioned, the Large Combustion Plant Directive introduced controls on NO_X and SO_2 from combustion plant with an output at or above 50 MWth. Additional legislation covering NMVOC emissions includes EC Directives on solvent use and petrol distribution.

4.1.2.2 Control of emissions from road vehicles

117. Particulate emissions from new diesel vehicle exhausts have been regulated by EU Directives since the early 1990s. The Directives set limits on emissions of particulates during Type Approval testing over specified vehicle or engine test cycles, some of which have changed in recent years in order to make the tests more representative of the performance of the vehicle on real road conditions. Since the first set of regulations – frequently referred to as Euro I – came into effect in 1992–1993, a succession of more stringent Directives have been brought in to further reduce particulate emissions from both light and heavy duty diesel vehicles, currently extending to tougher limits on new vehicles sold after 2008 (Euro IV vehicle standards (Official Journal, 1999)). Even tighter standards for new vehicles sold after 2010 (Euro V) are currently being considered by the EC.

118. There are currently no regulations on particulate emissions from petrol engines, but for conventional petrol engine technologies, particulate emissions are much lower than they are from diesel engines. However, emissions of NO_X and hydrocarbons, which are precursor pollutants to particulates in the atmosphere, are regulated for both petrol and diesel vehicles (AQEG, 2004). Relatively small amounts of the precursor pollutants NH_3 and SO_2 are emitted from vehicles, but their emissions are not regulated. Emissions of NH_3 can be affected by abatement technologies designed to control NO_X emissions, sometimes increasing their emissions (AQEG, 2004). Emissions of SO_2 are mainly dependent on the sulphur content of the fuels used.

119. The Type Approval limit values on PM, NO_X and hydrocarbons set by the various Directives for the different vehicle types since 1990 are summarised in Annex 2.

120. As well as emission limits for Type Approval testing of new vehicles and engines, the Roadworthiness Testing Directive 96/96/EC and its amendment 1999/52/EC sets limits during in-service testing of vehicles (the MOT). Rather than by mass, particulate emissions are limited according to monitoring by a free acceleration smoke opacity test.

121. The introduction of tighter vehicle emission standards has been accompanied by the introduction of tighter fuel quality standards. These have led to a continuous reduction in the sulphur content of petrol and diesel, which has been necessary to pave the way for the development of emergent exhaust treatment technologies successful in reducing primary particulate emissions and emissions of precursors. For example, most types of particulate trap technologies developed for diesel engine exhausts will only work using fuels with sulphur content below 50 ppm. The presence of sulphur in fuels seriously affects the performance of many abatement technologies. EU Directive 98/70/EC introduced successively tighter limits on the sulphur content of petrol and diesel. The maximum permitted level of 50 ppm (to be achieved by 2005) is already met by virtually all fuels supplied at UK filling stations. Directive 2003/17/EC has set a limit of 10 ppm on the levels of sulphur in petrol and diesel that can be sold from 2009. Even in the absence of particulate emission abatement devices fitted on diesel engine exhausts, reducing the sulphur content of diesel alone reduces particulate emissions by small amounts.

4.1.2.3 Control of emissions from other transport and machinery

122. There are no direct limits or controls on PM emissions from shipping, however there are controls on SO_2 emissions, which will also reduce emissions of PM. The Sulphur in Liquid Fuel Directive (1999/32/EC, see Section 4.1.2.4) does not cover marine heavy fuel oil but another international agreement (MARPOL) sets a limit of 1.5% for heavy fuel oil used in the North Sea, English Channel and Baltic Sea. A revision to the Sulphur in Liquid Fuel Directive has been proposed that would introduce the same or, in some instances, tighter limits into the Directive.

123. Emissions of NO_X, CO, unburned hydrocarbons and smoke from aircraft engines are subject to standards agreed by the International Civil Aviation Organisation (ICAO). These standards have been implemented in the Air Navigation (Environmental Standards) Order 2002.

124. Emissions of particulates, NO_X and hydrocarbons are regulated for 18–560 kW diesel engines used for non-road mobile machinery. These cover many different types of off-road machinery used in construction, mining, quarrying, industry and agriculture. Directive 97/68/EC defined emission limits (in g/kWh) to be implemented in two stages, between 1998 and 2003, depending on the power rating of the engine (Official Journal, 1998). Directive 2000/25/EC covered emission limits for agricultural tractors (Official Journal, 2000). These engines normally run on gas oil, a diesel fuel with higher sulphur content (currently 1500 ppm) than road diesel. As the sulphur content of this fuel is likely to be reduced, the effect this has on primary particulate emissions is not known. The EC has recently proposed an amendment to Directive 97/68/EC to cover diesel engines on railcars, railway locomotives and inland waterway vessels.

4.1.2.4 Control of emissions from other sources

125. The Environmental Protection Act 1990 banned the burning of crop residues in England and Wales with limited exemptions from 1993 onwards. Crop burning is still allowed in Scotland and Northern Ireland but is not common practice.

126. The Clean Air Act of 1956 introduced the concept of Smoke Control Areas, where domestic combustion of coal is prohibited and only designated smokeless solid fuels can be used. This Act has led to a major reduction in levels of PM in urban areas.

127. The Sulphur in Liquid Fuel Directive sets limits on the sulphur content of heavy fuel oil and gas oil burnt by sectors other than shipping.

4.1.2.5 National emissions ceilings

128. As well as regulations on emissions from specific sources, the UK is subject to certain national emission targets that must be achieved by 2010. In the mid-1990s, the United Nations Economic Commission for Europe (UNECE) started negotiating a multi-effect, multi-pollutant protocol on NO_X and related substances. This was aimed at addressing photochemical pollution, acidification and eutrophication. The Protocol to Abate Acidification, Eutrophication and Ground-Level Ozone was adopted in Gothenburg in December 1999 and was signed by the UK. The Gothenburg Protocol set emissions ceilings for nitrogen and sulphur oxides, ammonia and NMVOCs to be achieved by each country by 2010. The Gothenburg Protocol forms a part of the Convention on Long-Range Transboundary Air Pollution (UNECE, 1999).

129. Within the EU, the National Emissions Ceilings Directive set emissions ceilings for 2010 for each Member State for the same four pollutants listed in the Gothenburg Protocol (Official Journal, 2001). A number of Member States reduced their ceilings somewhat below the levels included in the Protocol. The UK reduced its ceiling for NO_X (as NO_2) emissions from 1181 kt set by the Gothenburg Protocol to 1167 kt set by the National Emissions Ceilings Directive. The UK ceilings for SO_2, NMVOCs and NH_3 set by the National Emissions Ceilings Directive are 585 kt, 1200 kt and 297 kt, respectively.

4.2 Sources of primary emissions

4.2.1 Estimation of emissions

130. Primary particle emissions can be estimated using several approaches, depending on the source type being considered. Often direct measurements of particles from different sources are available as well as information on how emissions vary in relation to the activity being undertaken. One example of this would be the road transport sector. Information is often available on the composition and size distribution of particles from sources within this sector. For other sectors data that characterise particle emissions are also available, but generally less information is available compared with the road transport sector. Often, therefore, emissions from a particular sector are derived using generic fuel-related factors. These factors have been derived for common fuel types such as coke, coal and fuel oil. Emissions from combustion processes are, in principle, straightforward to quantify and characterise, although often there is insufficient information available to do so.

131. Often it can be difficult to quantify primary emissions of particles. Many sources are often transient in nature, for example, emissions from fires or some agricultural emissions. Not only does the transient nature of these emissions make it difficult to quantify quantities of emissions released to the atmosphere, but the characteristics of the sources themselves can be very variable. In the case of agricultural or quarrying emissions, for example, the emissions can be dependent on other factors such as meteorology. This makes it very difficult to be specific about the quantity of material released to the atmosphere or the physical and chemical characteristics of the material. The difficulty in quantifying emissions from these sources also presents difficulties for dispersion modellers, who require emissions input data in the form of a mass release per unit time to derive ambient concentrations. Often, as highlighted in Chapter 6, it is necessary to rely on atmospheric measurements instead of directly inferring the concentrations from these sources.

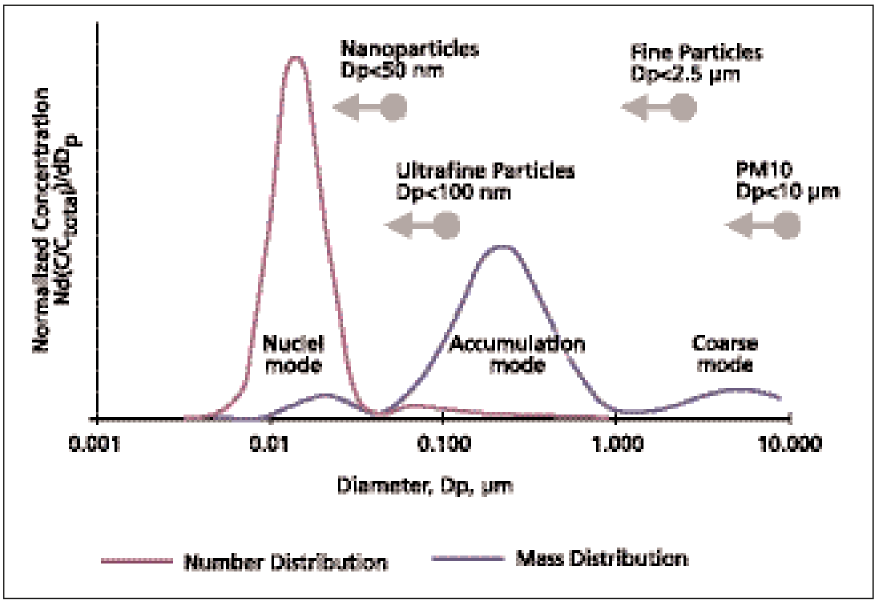
4.2.2 Transport

4.2.2.1 Particle formation mechanisms and definitions

132. An understanding of the mechanisms by which exhaust particles are formed is crucial to the interpretation of measurements and to the understanding of their limitations. The formation mechanisms and the ways in which particles are defined are closely linked.

133. Particles measured in diesel engine exhaust typically exhibit a trimodal distribution. The three distinct size modes are the nucleation mode (sometimes referred to as nuclei or nanoparticles), the accumulation mode and the coarse mode. Typical number-weighted and mass-weighted particle size distributions are shown in Figure 4.2.

Figure 4.2 Typical particle size distributions and definitions as measured in diesel vehicle exhaust (source: Kittleson, 1998).



134. The nucleation mode consists of primary particles that are composed mainly of lower volatility organics and sulphur compounds, but also some solid carbon and metallic particles. The nucleation mode has traditionally been defined as particles with a diameter of 5–50 nm. However, on the basis of the E-43 project in the US, Kittelson *et al*. (2002) have suggested that this range ought to be redefined as 3–30 nm. The accumulation mode ranges in size from ~30–500 nm. As its name suggests, this mode consists of particles formed by agglomeration of the smaller primary particles. The coarse mode consists of particles larger than 1,000 nm, which are formed by re-entrainment of material previously deposited on cylinder and exhaust system surfaces (Kittelson, 1998).

135. The general process of particle formation during combustion has been described in detail by Heywood (1988). Particle formation begins with the creation of carbonaceous material (soot) in the cylinder. Condensation reactions of gas-phase species such as unsaturated hydrocarbons and PAHs lead to the appearance of the first recognisable solid material (soot spherules) at combustion temperatures of between 1000 K and 2800 K. A phase of particle growth then follows inside the cylinder that includes surface growth of spherules by the adsorption of gas-phase components as well as by their coagulation and agglomeration. At temperatures above 500°C the particles are principally clusters of spherules, with individual spherule diameters of 15–30 nm. Once these clusters have left the cylinder they are then subject to a further mass addition process as the exhaust gases cool; as temperatures fall below 500°C, the particles become coated with adsorbed and condensed high-molecular weight organic compounds. Almost all of the particles found in the exhaust pipe before dilution are present as these carbonaceous agglomerates (accumulation mode particles), with a small amount of metallic ash and adsorbed materials.

136. However, the processes leading specifically to the presence of nucleation mode particles during measurement are not well understood. The nucleation mode can be viewed as a rather ephemeral and highly variable phenomenon. Most nucleation mode particles (up to 90%) are thought to originate from the condensation of lower volatility organics in the exhaust gas during exhaust dilution, rather than during combustion. However, nucleation is a non-linear process and the number of particles formed is very sensitive to the nature of the sampling conditions.

137. If a nucleation mode is observed during measurement – and this is not always the case – then the number of particles it contains will greatly exceed the number in the accumulation mode, although the nucleation mode particles contribute little to the total mass. The nuclei mode typically contains 0.1–10% of the particle mass and up to 90% or more of the particle number. Most of the mass is found in the accumulation mode, with the coarse mode typically accounting for 5–20%.

4.2.2.2 The regulatory test for exhaust emissions of PM

138. For exhaust emissions of PM[1], current Type Approval legislation in Europe only applies to heavy duty diesel engines and light duty vehicles powered by diesel engines. The legislation does not take into account the presence of the different particle modes and simply requires gravimetric measurement of the total mass of particles collected on a filter during the Type Approval test. The units used for reporting are g/kWh for heavy duty engines and g/km for light duty vehicles.

139. The legislation prescribes the use of a full-flow dilution system, known as a constant volume sampler (CVS), as the reference procedure for certification. The entire exhaust stream from the engine or vehicle is mixed with ambient air in a dilution tunnel, and a constant total air flow rate is maintained under all running conditions. The use of a constant air flow rate simplifies the calculation of mass-based emission factors. For light duty vehicles a single dilution tunnel is normally sufficient, but it is now customary to use a secondary dilution tunnel for heavy duty applications. This secondary dilution tunnel helps to accommodate the high exhaust flow rates from large-volume engines. Full specifications of this sampling system are given in the associated EU directive and are discussed further elsewhere (Moon and Donald, 1997; Stein, 2001; Kittelson *et al*., 1999).
For the sampling of PM, a portion of air is drawn from the dilution tunnel through a stainless steel probe configured with a 'Chinese hat'-type design facing into the exhaust stream; the gas is then passed through two series-mounted Teflon-coated glass fibre filters[2] that have been pre-weighed and preconditioned. A temperature of 51.7°C must not be exceeded at the sampling point. The exposed filter papers, after post-conditioning under controlled temperature and humidity conditions, are weighed using a microbalance. As with the sampling of PM in ambient air (see Chapter 5), the material collected on the filter includes not only solids but also liquid material, which would condense in the form of droplets at the test temperature, such as H_2SO_4 and high-boiling point hydrocarbons.

140. It is interesting to note that Switzerland has made an official statement saying that not only the mass but also the number of solid particles in the size range 20−300 nm must be measured and curtailed. This was in response to the Swiss legal requirement to minimize carcinogenic air pollutants using BAT, which currently includes particulate traps with efficiencies of >99%.

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[1] It is conventional to use the term 'particle' to refer to the airborne state and to use 'particulate' or 'PM' to refer to the material collected on a filter under the conditions of the Type Approval test.

[2] The second filter acts as a back-up to check for sample breakthrough.

On 1 September 2002, a trap retrofit regulation for diesels >37 kW used in all larger construction operations where traps must be VERT-verified (a verification system for particulate control devices derived from a research programme from 1994–2000, sponsored by Swiss, German and Austrian health authorities) to reach >95% filtration based on solid particle number definition came into force. Although Switzerland is free to regulate retrofit measures in highly polluted areas, Type Approval for new vehicles must be in accordance with EU regulations.

141. The limitations of the regulatory test method, in-service emissions testing and particle research programmes are discussed in Annex 3.

4.2.2.3 Particle size and number emissions by vehicle and fuel type and legislation class

142. Concerns have been raised that modern diesel engines, which emit a low mass of particles, have high emissions of particles in the nanometre size range. For example, in a study of the steady-state emissions of a (then) new technology diesel engine, Baumgard and Johnson (1996) observed a particle concentration of 10^9 cm^{-3}, around three orders of magnitude greater than expected. Bagley *et al*. (1996) found that there was a 15- to 35-fold increase in total particle number from a 1991 diesel engine compared with a 1988 engine, both running on very low-sulphur fuel. This was due to a 30- to 60-fold increase in the number of primary particles and occurred despite a substantial reduction in the mass of PM.

143. Ntziachristos and Samaras (2003) evaluated three diesel passenger cars manufactured under conditions of improving technology and operated on identical fuel. The vehicles selected corresponded to Euro 0 (pre1992), Euro I (1992 – 1996) and Euro III (2000 – 2005) emission standards. The cars were driven over a range of operational conditions, including three steady-state speeds (50, 90 and 120 km/h), one cold urban cycle, one hot urban cycle and an extra-urban cycle. A number of observations indicated emerging priorities for exhaust particle characterisation. PM reductions brought by technology improvements were not consistently accompanied by decreases in total particle number concentration and surface area. Hence, the introduction of additional metrics seems justified for more effective particle emission understanding and control. On the other hand, technology improvement was strongly apparent in the solid particle number concentration. This may provide an additional, more sensitive, measurement complementing PM mass emissions. Moreover, the median size of the solid particles was the least sensitive metric to either vehicle technology improvements or vehicle operation condition. This information may potentially be used to distinguish diesel particle emissions from other sources. Finally, all measurements referring to total particle emissions (except mass) showed a higher sensitivity to vehicle operation conditions rather than vehicle technology. This observation is consistent with the formation of nuclei mode particles that are not discerned from the simple PM measurements. Clearly, then, measurements of additional parameters – such as total particle concentration and surface area – measured according to well-defined test protocols are necessary to fully explore emission performance under varied operational conditions.

144. With respect to vehicle type it should be noted that although diesel engines emit higher concentrations of particles than gasoline engines, the total number of vehicle kilometres travelled by gasoline-powered vehicles in the UK is greater than that of diesels. Additionally, vehicles powered by spark ignition (SI) engines typically emit smaller particles than vehicles powered by diesel engines and are an important source of fine particles and nanoparticles. It has been estimated that two-thirds of the total fine particle mass occurs from SI engines (Latham *et al.*, 2001). New gasoline direct injection (GDI) engines emit much higher particle concentrations than conventional engines and may approach diesel levels under some conditions (Kittelson, 2000).

145. The EU Fifth Framework clustered projects ARTEMIS and PARTICULATES were designed to develop new exhaust emission factors for all transport modes. In particular the PARTICULATES project focussed on the development of a measurement protocol for in-service emissions of PM. One of the outputs of these projects was the provision of average speed-based emission factors for a range of particulate metrics – including particle size distributions, the number of particles and their surface area – for Euro I to Euro IV standards and for a range of fuel types and after-treatment devices. Results from this programme are expected in early 2005.

146. The measurement of various particulate metrics, in addition to mass, has highlighted the importance of sampling conditions. Diesel light duty vehicles generally emit $>10^{14}$ particles/km, measured as solid particles using an ELPI. Direct injection spark ignition engines emit 10–20-times fewer particles than the equivalent diesel-engined vehicle (Ntziachristos *et al*. 2004b).

147. Particulate emissions from heavy duty engines are higher than those associated with light duty engines. Total particle emissions associated with conventional Euro I to Euro III heavy duty engines are in the range 10^{14} particles/kWh. When operated with low sulphur fuels, vehicles equipped with PM traps generally produce very low mass emissions, low numbers of carbonaceous particles and low total numbers of particles. This effect of low sulphur fuels is greatest when associated with high speed and temperature operation (Thompson *et al*. 2004).

148. Several initiatives, including the PMP programme, are investigating the possibility of supplementing the PM Type Approval procedure with a standard based upon particle numbers. Whereas the PMP programme is expected to issue a recommendation during 2005, the Swiss BUWAL has proposed a standard of 10^{14} particles/km for light duty diesel vehicles, reducing to 10^{11} particles/km for those vehicles fitted with a diesel particulate trap. No proposed Type Approval standard has been set for heavy duty vehicles.

4.2.2.4 Estimating emissions from road transport

149. Emissions of PM and its precursors from road transport are derived using average speed-related emission factors in g/km. Within the UK, emission factors for vehicles at their normal operating temperature come from a large database of emission measurements held by TRL. These are combined with fleet composition and traffic activity data for different years on the national road network provided by the DfT. From this, the exhaust emissions are derived for each vehicle and road type.

150. The methods used for calculating emissions from road transport in an emissions inventory were described in detail in the AQEG report on NO_2. (AQEG, 2004). Only specific points related to the calculation of emissions of PM_{10} from road vehicles in the most recent (2002) version of the NAEI are commented on here.

Table 4.1 Exhaust emission factors for PM (g km^{-1}) for different road types used in the NAEI , based on data from TRL (Barlow *et al*., 2001).

Vehicle type	Euro class	Urban	Rural	Motorway
Petrol cars	ECE 15.04	0.0241	0.0151	0.0182
	Euro I	0.0029	0.0040	0.0089
	Euro II	0.0008	0.0014	0.0054
	Euro III	0.0008	0.0014	0.0054
	Euro IV	0.0008	0.0014	0.0054
Diesel cars	Pre-Euro I	0.154	0.131	0.167
	Euro I	0.057	0.049	0.077
	Euro II	0.051	0.037	0.058
	Euro III	0.032	0.023	0.036
	Euro IV	0.017	0.012	0.019
Petrol LGVs	Pre-Euro I	0.0300	0.0241	0.0696
	Euro I	0.0038	0.0066	0.0142
	Euro II	0.0009	0.0013	0.0048
	Euro III	0.0009	0.0013	0.0048
	Euro IV	0.0009	0.0013	0.0048
Diesel LGV	Pre-Euro I	0.287	0.243	0.358
	Euro I	0.080	0.079	0.195
	Euro II	0.082	0.081	0.200
	Euro III	0.057	0.057	0.141
	Euro IV	0.037	0.037	0.090
Rigid HGVs	Pre-1988	0.890	0.730	0.718
	88/77/EEC	0.426	0.350	0.344
	Euro I	0.220	0.180	0.177
	Euro II	0.153	0.124	0.123
	Euro III	0.111	0.089	0.089
	Euro IV (2006)	0.025	0.019	0.019
	Euro IV (2008)	0.025	0.019	0.019
Artic HGVs	Pre-1988	0.714	0.556	0.488
	88/77/EEC	0.626	0.487	0.428
	Euro I	0.577	0.474	0.466
	Euro II	0.399	0.323	0.321
	Euro III	0.288	0.233	0.231
	Euro IV (2006)	0.064	0.049	0.049
	Euro IV (2008)	0.064	0.049	0.049
Buses	Pre-1988	1.416	0.647	0.551
	88/77/EEC	0.614	0.281	0.239
	Euro I	0.311	0.169	0.167
	Euro II	0.203	0.117	0.116
	Euro III	0.147	0.084	0.084
	Euro IV (2006)	0.044	0.018	0.018
	Euro IV (2008)	0.044	0.018	0.018
Motorcycles (2 stroke)	All	0.04	0.04	0.04
Motorcycles (4 stroke)	All	0.12	0.12	0.12

Key: LGV, light goods vehicle; HGV, heavy goods vehicle.

4.2.2.5 Exhaust emission factors for PM

151. Surveys of road vehicle exhaust emission factors are on-going within the UK and elsewhere. The last revision to the UK database was undertaken by TRL on behalf of the DfT (Barlow *et al.*, 2001). The NAEI incorporates data from the TRL emission factor database for vehicles up to Euro II standards (that is, pre-2000 models). A substantial part of the measurements of Euro I and Euro II emission factors come from several test programmes funded by DfT and Defra and carried out at UK test laboratories between 1999 and 2001 (Barlow *et al.*, 2001). The measurements were made on dynamometer test facilities under various simulated real-road drive cycles.

152. Table 4.1 shows PM emission factors for each class of vehicle calculated from the emission factor equations at typical speeds on urban, rural and motorway roads. These emission factors are currently used in the NAEI. The vehicle types match the traffic activity data that are used in the national emission calculations. The emission factors in this table refer to vehicles running on ultra-low sulphur fuels, with a sulphur content of around 50 ppm, fuels which are currently available at most filling stations around the UK. In cases where the vehicles tested were running on diesel fuels with higher sulphur content, as would have been the norm in the 1990s when the emission measurements were made, the measured emission factors were corrected in deriving the factors shown in Table 4.1. The fuel correction factors were based on various studies and empirical equations relating exhaust emissions to fuel quality taken from the European EPEFE Programme (EPEFE, 1995).

153. Emissions from Euro III and Euro IV classes must be estimated through an examination of the proposed future legislation and the technical ability to meet these new limits. This is because either the emission standard has yet to be introduced at all or because insufficient numbers of vehicles have been tested, the Euro standard having only recently entered the fleet. As discussed in the AQEG report on NO_2 this estimation process can result in significant uncertainties in future emissions factors (AQEG, 2004).

154. Table 4.1 provides emission factors for Euro III and Euro IV vehicles, the latter referring to standards that will come into effect in 2006. These are based on scaling factors that are assumed to apply across all speeds relative to emission factors for Euro II vehicles. For diesel light duty vehicles, the scaling factors were estimated by the NAEI on the basis of current emission levels for Euro II vehicles and the amount emissions will need to be reduced to achieve the Type Approval limit values, taking into account differences in the drive cycles used and the contribution that cold start emissions will make to emission over the new 98/69/EC test cycle. Consideration was given to the emissions durability requirements of vehicles set in the Directive. This states that when using a deterioration factor of 1.2 to the emission factor of the vehicle when new, PM emissions from light duty diesel vehicles must be below the limit value after 80,000 km for Euro III vehicles and 100,000 km for Euro IV vehicles. The emission factors for in-service Euro III and IV vehicles are therefore taken to be a factor of 1.2 lower than the amount derived from the limit value when new and are then allowed to increase by a factor of 1.2 at a linear rate over 80,000 km (Euro III) or 100,000 km (Euro IV). The figures in Table 4.1 represent the value of the Euro III and IV emission factor for new vehicles. As no limit values are set by the Directives on PM emissions from petrol vehicles, emission factors for Euro III and IV vehicles are retained at Euro II levels.

155. For heavy duty vehicles, it was assumed that Euro II vehicles are currently at their emission limits and will be reduced no more than required to achieve the new emission limits for Euro III and IV, using scaling factors from the COPERT III database (EEA, 2000). COPERT III is the methodology and database for calculating road transport emission factors. It was developed by a group of experts for the European Environment Agency (EEA) and is recommended for national emission inventory reporting under CORINAIR.

4.2.2.6 Road transport activity data

156. To calculate UK emissions from road transport for the NAEI, the hot exhaust emission factors are combined with annual UK vehicle kilometre data for different vehicle types on different roads and data on the composition of the vehicle fleet by age, size and fuel mix. Further details are provided in the AQEG report on NO_2. (AQEG, 2004). The NAEI mainly draws on transport statistics published by the DfT and the Transportation Unit, DoE, Northern Ireland.

157. The NAEI makes an assumption that a small proportion of cars fitted with three-way catalysts fail each year. However, it is believed that current car models and future models are more durable than the first generation (Euro I) catalyst cars. This is as a result of on-board diagnostic systems fitted to modern vehicles and the more stringent durability requirements set out in the Directives. Although there are no definitive data available on catalyst failure, on the basis of discussions with experts in industry and at the DfT and evidence, for example, from studies analysing MOT test data, equipment warranty returns and car fleet maintenance records (DfT, 2002), the NAEI has adopted different failure rate assumptions for each Euro class as follows:

- Euro I cars: 5% per annum;
- Euro II cars: 1.5% per annum; and
- Euro III–IV cars: 0.5% per annum.

158. It is assumed that 95% of failed catalysts are repaired after the vehicle is three years old and reaches the age of MOT testing.

4.2.2.7 Cold start emissions

159. The excess emissions that arise when the vehicle is started with its engine cold or below its optimum operating temperature are calculated using a method taken from COPERT III. The procedure uses equations that take account of the effects of ambient temperature on the excess emissions and its effect on the distances travelled with the engine cold. The distances travelled with cold engines also depend on average trip lengths: the shorter the average trip lengths are, the greater the contribution of cold start emissions. The method calculates the ratio of cold to hot exhaust emissions for light duty vehicles that are used to calculate the overall cold start emissions from these vehicles.

160. Further details on the emission calculation methodology used in the NAEI for road transport are provided in the NAEI report (Goodwin *et al*., 2001).

4.2.2.8 Non-exhaust road transport PM

161. The PM generated by road transport activity can be categorised according to its mode of formation. It is generally assumed that fuel combustion is the primary mechanism by which particles are formed, and a considerable body of research on vehicle exhaust particulate emissions has been compiled. However, there are a number of other processes, involving mechanical abrasion and corrosion, that can also result in PM being released directly to the atmosphere. These processes include: tyre wear; brake wear; clutch wear; road surface wear; corrosion of chassis, bodywork and other vehicle components; and corrosion of street furniture and crash barriers.

162. The abrasion and corrosion processes can also lead to the deposition of particles on the road surface. The material that collects on the road surface, often referred to as 'road dust', may also contain exhaust particles, de-icing salt and grit deposited during winter maintenance and matter from a range of sources that are not related to road transport (for example, crustal and vegetative material and material from industrial/commercial/domestic activity). Road dust may subsequently be suspended or resuspended in the atmosphere as a result of tyre shear, vehicle-generated turbulence and the action of the wind. Clearly, the source apportionment and quantification of non-exhaust PM measured in the vicinity of roads is a rather complex task (Luhana *et al*., 2004).

163. Non-exhaust emission sources may contribute significantly to atmospheric particle concentrations. However, the data relating to the emission rate, size and composition of particles arising from such sources are far from comprehensive, and more accurate emission factors are required for emission models and inventories.

4.2.2.9 Road vehicle tyre wear

164. Relatively little data exists that quantifies the emission rates, size distribution and composition of vehicular tyre wear. Different experiments have, however, been carried out to quantify emissions of particles from tyre wear. These experiments have included the collection of samples close to roads and tests undertaken on dynamometers (rolling roads).

165. The NAEI uses g/km emission factors to estimate PM emissions from tyre and brake wear. However, the method used to estimate the emission factors for the 2002 NAEI has been revised from earlier versions of the inventory in light of a recent review of PM emissions from these sources carried out for the UNECE Task Force on Emission Inventories[3] and the EMEP/CORINAIR Emission Inventory Guidebook[4]. The review examined a number of experimental studies on the wear of tyre and brake material and the amount that becomes airborne. Tyre tread wear is a complex physiochemical process that is driven by the frictional energy developed at the interface between the tread and the road pavement (Veith, 1995). The rate of wear varies enormously depending on a combination of factors, including tyre characteristics, vehicle characteristics, road surface characteristics and vehicle operation. Typical wear rates of 10–90 mg km^{-1} per tyre have been reported, with 1–15% by mass of passenger car tyre wear material emitted as PM_{10}. A recent measurement campaign by TRL, involving six mileage accumulation vehicles, derived a typical factor of 74 mg km^{-1} for tyre wear (Luhana *et al*., 2004).

4.2.2.10 Road vehicle brake wear

166. During forced deceleration, vehicle brake linings are subject to large frictional heat generation and associated wear. This mechanical process generates particles from both the brake lining and the brake surface.

167. The wear rate of brake linings for passenger cars has been estimated to be around 10–20 mg vkm^{-1} and up to three- or four-times higher for heavy duty vehicles. Although there have been only limited studies, it appears that the proportion of material emitted as PM_{10} is higher from brake wear than from tyre wear (USEPA 1995; TNO, 1997). Typical PM_{10} emission factors of 1–9 mg vkm^{-1} have been reported for light duty vehicles and 20–40 mg vkm^{-1} for heavy duty vehicles. However, these same studies have concluded that the proportion of brake wear PM with an aerodynamic diameter of <0.1 µm could be as high as 33% of the total brake-wear PM emission.

4.2.2.11 Estimation of tyre and brake wear emissions

168. For both tyre and brake wear, the UNECE method provides emission factors for different vehicle types and provides speed correction factors. These imply higher emissions per km at lower speeds than at high speeds. For heavy duty vehicles, a load correction factor is also provided and tyre wear emissions depend on the number of axles.

169. Based on average speeds on UK roads, Table 4.2 shows emission factors for different vehicle types on different road types. Units are in g of PM_{10} per vehicle km, but in the case of tyre wear emissions from heavy duty vehicles, units are in g PM_{10} per axle km.

170. Use of these new emission factors in the NAEI has almost doubled the estimate of tyre and brake wear emissions in the UK in 2001 compared with previous estimates, mainly due to increased estimates for tyre wear emissions. However, the emission factors must still be regarded with a high degree of uncertainty.

4.2.2.12 Road vehicle resuspension

171. The resuspension of particles by road vehicles occurs through two principal mechanisms. First, particles can be resuspended from the wear and shear of tyres acting on the road surface. Second, particles can be resuspended as a result of the turbulence caused by moving vehicles. Since resuspended material is non-exhaust in origin, there are no direct measurements available that quantify emissions in terms of g km^{-1}. Most estimations of the importance of resuspension have been based on receptor modelling techniques at specific locations. In many cases, however, these studies suggest that vehicular resuspension is as important, if not more important, than exhaust emissions of particles from vehicles. The studies also indicate that HGVs are significantly more important in terms of resuspension compared with smaller vehicles such as passenger cars. Because the significance of resuspension is governed by many factors (for example, vehicle type, road surface condition and meteorological conditions), resuspended material is highly variable in terms of its source emission rate. It is therefore difficult to derive representative emission rates that can be expressed in terms of g km^{-1}.

Table 4.2 Average PM_{10} emission factors for tyre and brake in the UK.

		Tyre wear	Brake wear
Cars	Urban	0.00874 g km^{-1}	0.0117 g km^{-1}
	Rural	0.00680 g km^{-1}	0.0055 g km^{-1}
	Motorway	0.00579 g km^{-1}	0.0014 g km^{-1}
LGVs	Urban	0.01380 g km^{-1}	0.0182 g km^{-1}
	Rural	0.01074 g km^{-1}	0.0086 g km^{-1}
	Motorway	0.00915 g km^{-1}	0.0021 g km^{-1}
HGVs	Urban	0.00918 g axle km^{-1}	0.0510 g km^{-1}
	Rural	0.00737 g axle km^{-1}	0.0271 g km^{-1}
	Motorway	0.00608 g axle km^{-1}	0.0084 g km^{-1}
Buses	Urban	0.00937 g axle km^{-1}	0.0536 g km^{-1}
	Rural	0.00737 g axle km^{-1}	0.0271 g km^{-1}
	Motorway	0.00608 g axle km^{-1}	0.0084 g km^{-1}
Motorcycles	Urban	0.00376 g km^{-1}	0.0058 g km^{-1}
	Rural	0.00292 g km^{-1}	0.0028 g km^{-1}
	Motorway	0.00249 g km^{-1}	0.0007 g km^{-1}

There is also a potential difficulty with double counting in inventories because a proportion of particles assumed to be resuspended in origin might have already been accounted for elsewhere. Nicholson (2000) has estimated a mean PM_{10} emission factor of 40 mg km^{-1} for the UK based on the current UK mix of vehicles.

172. A comprehensive set of recent measurements undertaken in Nevada and North Carolina quantified the contributions from tyre wear, brake wear and resuspension for a range of conditions (Abu-Allaban *et al*., 2003). Measurements were made at eight locations and the emissions were quantified for light and heavy vehicles. Typically, exhaust particles were within ±30% at the different sites, whereas the other non-exhaust particles varied greatly between sites. The resuspended component depended on the vehicle speed, the degree of road silting and the proportion of heavy vehicles. Brake wear was only important on exit from a freeway. No evidence was found of tyre wear at any location. Furthermore, no statistically significant difference was found between the locations in Nevada and North Carolina for the non-exhaust particle emission rates calculated. Table 4.3 shows the key results from the study for PM_{10}. Of interest is the comparison between light and heavy vehicles since many of the particle components will depend on the specific conditions in Nevada and North Carolina, which have different climatic conditions compared with the UK. These results show that exhaust emissions from heavy vehicles are a factor of seven higher than those from light vehicles; the resuspended and brake-wear component is a factor of ten higher for heavy vehicles. The mean emission rate factor calculated in the Abu-Allaban study for heavy vehicles – 2.247 g km^{-1} is more than factor of five higher than the total non-exhaust particle emissions that were estimated for heavy vehicles on Marylebone Road (see Chapter 6).

Table 4.3 Emission factors for different sources of road vehicle PM_{10} (mg km^{-1}) (adapted from Abu-Allaban *et al.*, 2003).

Vehicle type	Total emission	Exhaust	Resuspension	Brake wear
Heavy	2570±1297	211±47	2247±617	124±71
Light	265±126	29±7	224±59	12±8

4.2.2.13 Other non-exhaust road vehicular emissions

173 Emissions from vehicles also arise from clutch wear and the corrosion of vehicular materials. Very little information exists on the likely composition and magnitude of these emissions, although these sources could make a non-negligible contribution to non-exhaust emissions.

4.2.2.14 Emission factors for other transport

174 Emissions are estimated from other transport sources including shipping, railway locomotives and aircraft. Emission estimates are based on the use of emission factors combined with either fuel-use statistics or, in the case of aircraft, numbers of take-off and landing cycles. Estimates for rail locomotives attempt to reflect the differences in emissions from different types, but estimates for aircraft and shipping are more simplistic. Further details of the emission estimation methodology for these transport sectors are given in the AQEG report on NO_2 (AQEG, 2004). Emission factors and emissions estimates from the most important non-road transport sources are shown in Table 4.4.

Table 4.4 Emission factors and emission estimates for PM_{10} from non-road transport.

Source	Emission factor	Emission in 2001 (kt)
Aircraft	0.204 kg t^{-1} fuel	0.2
Coastal shipping	1.07 kg t^{-1} gas oil	0.7
Fishing vessels	1.07 kg t^{-1} gas oil	0.1
Naval shipping	1.07 kg t^{-1} gas oil	0.3
Intercity trains	2.23 kg t^{-1} gas oil	0.6
Freight trains	0.816 kg t^{-1} gas oil	0.1
Agricultural vehicles	6.04 kg t^{-1} gas oil	2.0
Industrial off-road vehicles	4.10 kg t^{-1} gas oil	5.1
Airport support vehicles	3.93 kg t^{-1} gas oil	0.1
Garden machinery	8.70 kg t^{-1} DERV	0.1

4.2.2.15 Rail

175. Emissions from railways, that is, diesel locomotives, are estimated from fuel consumption estimates using fuel-related emission factors. Although total UK emissions from railways contribute in a minor way to the total, it is likely that they could be more important locally, for example, close to a rail terminus with a large proportion of non-electric trains.

4.2.2.16 Aircraft

176. Aircraft do not make a large contribution to total UK emissions of PM_{10}. The turbines used by aircraft tend to produce high concentrations of NO_X and low concentrations of PM_{10}. Even for large airports such as Heathrow, the contribution made by aircraft is not significant. Of more significance are the emissions from activities related to aircraft, such as those from airside vehicles. Dispersion modelling described in Chapter 8 also shows that PM_{10} emissions from aircraft do not make a large contribution to local concentrations of primary PM_{10}. There is however a lack of measurements related to aircraft emissions compared with other source sectors.

177. Within the EU ARTEMIS project, a new aircraft emission factor database and model is being developed. While this will largely draw upon existing emissions data, the project includes new emissions measurements from taxi and thrust operations, recorded at London Heathrow, Frankfurt, Vienna, Wroclaw Strachowice and Warsaw Okecie airports during the period 2001–2002.

4.2.2.17 Shipping

178. The shipping sector makes a small contribution to total PM_{10} emissions in the UK. Like rail, it is likely that shipping could be of significance at a local level close to busy ports. Emissions can arise from both the large diesel engines used to power ships and the auxiliary engines. Currently, most ships in UK waters use fuel that has a high sulphur content, which tends to increase emissions of particles. However, as discussed in Section 4.3, the most important impact that ships make towards particle emissions is likely to be their contribution to secondary aerosol through emissions of SO_2 and NO_X. Emissions from ships can be very transient in nature and their patterns of activity complex.

179. The result is that ships are now one of the biggest sources of SO_2 emissions in the EU. Recent research for the EC shows that by 2010, ship emissions of SO_2 are likely to be equivalent to >75% of all land-based emissions, including emissions from all transport modes, combustion plants and heating engines which burn liquid fuels (Entec, 2002).

180. Throughout the EU, ship emissions contribute 20–30% to the concentrations of secondary inorganic particles in most coastal areas (COM, 2002). For the reasons outlined above, the EC is seeking to reduce SO_2 and particle emissions from ships and this is now an important environmental priority.

4.2.2.18 Non-road mobile machinery

181. Emissions from off-road machinery cover a range of portable or mobile equipment powered by diesel or petrol engines. They include agricultural and forestry equipment, construction equipment, industrial machinery and portable generators, aircraft support vehicles and domestic house and garden equipment.

Further details are given in the AQEG report on NO_2 (AQEG, 2004). The emission factors used in the inventory, in combination with machinery population and usage estimates, are given in g PM kWh^{-1} for different types of equipment and engine sizes. Table 4.4 provides averaged emission factors for the UK population of machinery expressed in units of fuel used. These implicit fuel-based emission factors are taken from the 2001 NAEI.

4.2.2.19 Chemical composition of PM emissions from transport sources

182. Particulate emissions from diesel exhausts contain mainly elemental carbon and heavy hydrocarbons derived from the fuel and lubricating oil that may be partially volatile. The particulates contain a large proportion of the toxic PAHs found in diesel exhausts. A small proportion of the emitted particle mass is sulphate derived from the sulphur present in the fuel, although this component can be expected to decline as the sulphur content of fuel is reduced. The EPEFE Auto-Oil Programme (EPEFE, 1995) characterised the particulates emitted from diesel vehicles running on a number of different fuel formulations, but all had a sulphur content of around 400 ppm, about ten-times higher than current levels. The studies found 70–80% of the particulate mass was elemental carbon; 13–23% was heavy hydrocarbons from fuel and lube oil; and 7% was sulphate. The primary contribution to sulphate is, however, negligible compared with secondary sulphate formation.

183. Particulate emissions from tyre wear include elemental carbon; hydrocarbons, including PAHs; several metals, including zinc and iron; and other inorganic compounds containing sulphur, silicon and calcium. Zinc is particularly abundant in tyre material. Chemical markers used to attribute tyre wear debris include benzothiazole, 2-(4-morpholinyl) benzothiazole, n-cyclohexyl-2-benzothiazolamine, styrene butadiene rubber, zinc and extractable organic zinc.

184. A variety of metals and PAHs are present in the material of brake linings. Various organic compounds (including PAHs), elemental carbon and inorganic compounds containing silicon, barium, iron, magnesium, phosphorus and chlorine have been found in brake dust.

4.2.3 Stationary sources

185. This section considers emissions of PM from stationary sources in general and discusses methods for deriving emission estimates of PM_{10}. The emission estimation procedure and emission factors described in the following section relate to the 2001 version of the NAEI produced in 2003, the most up-to-date version used by modellers. Since then the NAEI has paid considerable attention to the estimation of particulate emissions from stationary sources, with particular regard to the emission factors used in the inventory. This review is on-going, but some of the main sources – where changes to the emission factors or estimation methods are recommended – are summarised later in this section.

4.2.3.1 Overview of stationary sources

186. PM is emitted by a wide range of sources, including most major industries as well as homes and commercial premises. Natural sources of particles are also important. The most significant sources are the combustion of fuels in power stations, industrial/commercial-scale plants and particularly small residential combustion units. Residential-scale combustion units tend to emit more PM per unit of fuel consumed compared to larger combustion plants used by industrial and commercial operations. This is due to the lower efficiency of combustion in domestic fires, which produces greater quantities of incompletely oxidised products in the waste gases, and the use of abatement systems on larger industrial plant to remove PM emissions.

187. Industrial processes are a second major source of PM emissions. These emissions generally occur as a result of one of three mechanisms:

(i) the grinding, crushing or abrading of larger materials, for example, cement clinker grinding;

(ii) the evaporation of volatile matter from heated materials, for example, process emissions from glass and non-ferrous metal furnaces; and

(iii) the entrainment of fine PM in gas streams, for example, spray drying of chemicals.

188. An important issue for PM emissions from many industrial processes is that a proportion of these emissions are from 'fugitive sources'. By this it is meant that emissions occur directly from a process into the workplace or outside atmosphere, without first being collected in by ducting or another container, and then being vented to atmosphere. The significance of fugitive emissions is that because they are not contained, they are difficult both to quantify and to abate.

189. Other, more minor, sources of PM include agricultural processes and waste incineration. Agricultural sources include the suspension of PM from agricultural land and entrainment of fine PM in vented gases from animal housing. Incineration of municipal solid waste, clinical waste, sewage sludge and chemical wastes are all very minor sources of PM, particularly so in recent years due to increasingly lower emission limits for persistent organic pollutants from incineration plants.

4.2.3.2 Derivation of emission estimates

190. Emission estimates for PM_{10} can be derived in one of two ways: with the use of national activity data such as fuel consumption data and emission factors or by the aggregation of emission estimates provided by process operators for individual sites, for example, via the Environment Agency's Pollution Inventory. The estimates provided by operators may be derived using emission factors and plant-specific activity data or may be based on stack monitoring or other measurements. The methodologies used by process operators in a given sector may not have been consistent and had not been detailed in the sources of these data. Nonetheless they are used in the NAEI, when available, because it is considered that they represent an improvement on estimates made using national activity data and emission factors. Furthermore, consistency should improve in emission estimates under the Pollution Inventory as Guidance Notes are published by the Environment Agency. A more detailed description of the methodology used for the 2001 NAEI for the most important sources of PM_{10} emissions is given in the following sections.

Table 4.5 Emission factors and emission estimates for PM10 emissions from industrial and commercial sources as used in the 2001 version of the NAEI produced in 2003.

Source	Emission factor	Emission in 2001 (kt)
Coal burnt at power stations	319 g t^{-1} coal	15.7
Fuel oil burnt at power stations	426 g t^{-1} fuel oil	0.4
Natural gas burnt at power stations	0.153 g $therm^{-1}$ gas	1.4
Autogenerators:		
burning coal	2500 g t^{-1} coal	4.0
burning gas	0.152 g $therm^{-1}$ gas	0.2
Oil and gas industry/refineries:		
burning fuel oil	516 g t^{-1} fuel oil	1.0
burning gas oil	1070 g t^{-1} gas oil	0.1
burning OPG	0.147 g $therm^{-1}$ OPG	0.2
burning gas	0.326 g $therm^{-1}$	0.3[b]
Iron, steel, chemical and other industry[c]:		
burning coal (other industry)[d]	2500 g t^{-1} coal	5.1
burning coke (other industry)[d]	288 g t^{-1} coke	0.1
burning fuel oil	1032 g t^{-1} fuel oil	0.8
burning gas oil	250 g t^{-1} gas oil	0.6
burning gas	0.326 g $therm^{-1}$	1.7[b]
burning LPG	0.295 g $therm^{-1}$	0.1
burning wood	440 g t^{-1} wood	0.3
burning waste oils	250 g t^{-1} oil	0.1
Commercial/public sector:		
burning coal	2500 g t^{-1} coal	0.7
burning fuel oil	1032 g t^{-1} fuel oil	0.3
burning gas oil	250 g t^{-1} gas oil	0.3
burning gas	0.326 g $therm^{-1}$	1.4[b]
Coke oven processes	43.8 g t^{-1} coke	0.2
Cement clinker production	165 g t^{-1} clinker	1.7
Coal burnt at lime works	4380 g t^{-1} coal	0.5
Natural gas burnt at lime works	0.342 g $therm^{-1}$ gas	0.1[b]
Fugitive emissions from lime works	336 g t^{-1} lime	1.0[b]

Source	Emission factor	Emission in 2001 (kt)
Sinter plant	250 g t^{-1} pig iron	2.5
Blast furnaces	115 g t^{-1} pig iron	1.1
Basic oxygen furnaces	74 g t^{-1} BOF steel	0.8
Electric arc furnaces	138 g t^{-1} EAF steel	0.4
Foundries	625 g t^{-1} castings	0.9
Other iron and steel sources	8.70 g t^{-1} pig iron	0.1
Ore stockpiles	104 g t^{-1} pig iron	1.0
Aluminium smelting	604 g t^{-1} metal	0.2
Anode baking	1740 g t^{-1} anode	0.4
Alumina production	1110 g t^{-1} alumina	0.1
Ammonia-based fertilizer processes	—	0.5
Titanium dioxide processes	—	0.1
Other chemical processes	—	0.7
Fletton brick manufacture	204 g t^{-1} bricks	0.2
Non-Fletton brick manufacture	360 g t^{-1} bricks	2.2
Primary lead/zinc production	474 g t^{-1} metal	0.1
Secondary aluminium processes	369 g t^{-1} metal	0.3
Other non-ferrous metal processes	[a]	0.1
Copper semis and alloys	192 g t^{-1} metal	0.1
Flat glass	428 g t^{-1} glass	0.3
Glass wool	1520 g t^{-1} glass	0.2
Wood products manufacture	433 g t^{-1} product	1.4
Asphalt manufacture	30 g t^{-1} asphalt	0.7
Cement batching	12.8 g t^{-1} cement	0.3
Other part B processes	[a]	4.6

Key: OPG, other petroleum gas; LPG, liquid petroleum gas. [a]Because of the highly varied nature of the processes included, no meaningful emission factor can be derived. [b]These figures are currently being revised. See Section 4.2.4. [c]It should be noted that these are implied, aggregate factors in some cases averaged across several industry sectors. [d]Coal and coke are not burned in the iron and steel industry – these figures refer to other industries.

4.2.3.3 Power stations

191. Estimates of PM emissions from most UK power stations are available via regulators: the Environment Agency publishes emission estimates in their Pollution Inventory for England and Wales; the Scottish Environment Protection Agency (SEPA) publishes emissions data intended for submission to the European Pollutant Emissions Register

(EPER); and the Department of the Environment (Northern Ireland) provides emissions data for power stations, directly for use in the NAEI. There are some gaps in the dataset – for example, for small, regional, generators – but emissions from these processes are estimated either by extrapolation of emissions for other sites or by the use of literature-sourced emission factors, depending upon the fuel being burnt. Emission factors and emission estimates for the most important fuels, included in the 2001 version of the NAEI, are shown in Table 4.5.

192. The emission estimates for power stations are assumed to be among the most reliable in the PM_{10} inventory. This is based on the fact that emission estimates are provided for individual sites by the process operators, who will have access to data on fuel characteristics, fuel consumption, abatement plant efficiency and emissions monitoring data.

4.2.3.4 Industrial and commercial combustion processes

193. The NAEI divides industrial and commercial combustion sources into a number of subsectors, the most important of which (in terms of fossil fuel use) are:

- autogenerators;
- refineries;
- iron and steel;
- oil and gas industry;
- other industry; and
- public services.

194. Emissions of PM for the above and other combustion sources are estimated largely using literature-based emission factors and national fuel consumption statistics. The exception is refineries, where emission estimates are based on estimates provided by process operators for inclusion in the Pollution Inventory and other sources maintained by regulators. Emission factors and emission estimates for the sub-sectors listed above and the most important fuels, included in the 2001 version of the NAEI, are shown in Table 4.5. It should be noted that the factors shown here are averaged across a range of industrial and combustion processes of various scales of operation. Detailed information on emission factors and calculations for specific sectors can be found in the NAEI annual reports. The NAEI provides an opportunity for feedback from industry sectors and has a programme of continuous improvement. The inventory uses data provided by the major iron and steel producers, but there are a number of smaller iron foundries operating smaller Part B processes that are included in the iron and steel emission sector. For these, emissions per tonne of fuel tend to be higher than from the larger plant. Little coal and coke is burned in the iron and steel sector.

195. The emission estimates for industrial and commercial combustion plants are considered more uncertain than those for power stations and have, since the 2001 version of the NAEI, been identified as requiring review and updating. The use of a single or a few emission factors for plants burning a particular fuel is overly simplistic and will be replaced with a more detailed treatment starting with the 2003 version of the NAEI. It is not yet possible to state what impact this will have on UK emission estimates for PM_{10}; however, it is likely that estimates for the generally large plant in sectors such as autogenerators and the iron and steel sector will decrease, whereas estimates for the generally small plants in sectors such as public services will increase.

4.2.3.5 Domestic combustion

196. Emissions from domestic combustion are estimated using literature-based emission factors and national fuel consumption data. Emission factors and emission estimates for the most significant sources, included in the 2001 version of the NAEI, are shown in Table 4.6.

Table 4.6 Emission factors and emission estimates for PM_{10} emissions from domestic combustion.

Fuel burnt	Emission factor	Emission in 2001 (kt)
Coal	10400 g t^{-1} coal	14.9
Anthracite	3594 g t^{-1} anthracite	4.0
Solid smokeless fuels	5600 g t^{-1} SSF	2.5
Wood	7900 g t^{-1} wood	7.1
LPG	0.393 g $therm^{-1}$ LPG	0.1
Natural gas	0.326 g $therm^{-1}$ gas	4.2

Key: SSF, smokeless solid fuel.

197. PM_{10} emission factors for domestic combustion and for gaseous fuels in particular are very uncertain and are currently the subject of review. Emission factors will depend upon the type of combustion device – for example, open fire or enclosed boiler – yet the current approach does not take account of these differences. Changes will, therefore, be made to the methodology and the emission factors used for the 2003 version of the NAEI. Some of the main findings of the recent review are summarised in Section 4.2.4.

4.2.3.6 Coke and SSF manufacture

198. Emissions from the manufacture of coke are reported separately for those emissions derived from fuels burnt and emissions from processes such as oven loading and discharging, product quenching and crushing. Emissions from the manufacture of SSF using low temperature carbonisation processes are reported under a single category covering both fuel-related and process emissions. The estimates for fuel combustion during coke manufacture are based on literature-based emission factors and national energy consumption data and are relatively small compared with emissions from processes. The process emission estimates for coke works and the emission estimates for SSF manufacture are derived from emissions data provided by process operators for inclusion in the Pollution inventory. The emission factors and emission estimates for the only significant source – coke oven processes – included in the 2001 version of the NAEI are summarised in Table 4.5.

199. The emission estimates for combustion of gaseous fuels are very uncertain, but the total estimate for coke ovens (that is, fuel combustion plus process emissions) is based on the emission estimates given in the Pollution Inventory so that the overall uncertainty in emission estimates for this source is equal to the uncertainty in the Pollution Inventory data. These data are provided by process operators and might therefore be expected to be robust, but PM_{10} emissions from coke ovens (and SSF manufacture) will include a fugitive component, which is difficult to quantify. On balance, therefore, it is assumed that emission estimates for both types of processes are more uncertain than the estimates for combustion processes.

4.2.3.7 *Cement and lime processes*

200. Emissions from the manufacture both of cement clinker and lime will include a contribution from the combustion of fossil fuels and also from 'process' sources such as volatilisation of some components of raw materials and dust generated during crushing and grinding operations. As with coke ovens, fugitive emissions occur and are difficult to quantify. Emission factors and emission estimates for significant sources in cement and lime works, included in the 2001 version of the NAEI, are shown in Table 4.5. Note that the figures for lime production in Table 4.5 are those used in the 2001 NAEI, but these have recently been revised downwards and are likely to be closer to 0.3 kt in 2001. These changes will appear in the 2003 version of the inventory.

201. Emission estimates for cement and lime works are generally expected to be of a quality similar to those for industrial combustion plants and are considered acceptable given the uncertainty of estimates elsewhere in the PM_{10} inventory (with the exception of the estimate for fugitive emissions from lime processes, which is being reviewed). The possibility of under-reporting of emissions from fugitive sources cannot be ruled out.

4.2.3.8 *Iron and steel industry*

202. Emissions of PM_{10} from iron and steel processes are estimated and reported under the following categories:

- sinter plants;
- iron and steel blast furnaces;
- basic oxygen furnaces;
- electric arc furnaces;
- iron and steel foundries;
- other iron and steel (including iron ore handling, slag processing, continuous casting and miscellaneous secondary processing); and
- stockpiles (emissions from stockpiles of feed materials).

203. These categories relate to emissions from these processes due to any combustion of fuels and due to 'process' emissions, such as from the charging and tapping of furnaces. Emissions from the combustion and process sources are, however, reported separately in the NAEI in the case of blast furnaces. Combustion-related emissions are estimated using literature-based factors, whereas process emission estimates are based mainly on data reported in the Pollution Inventory or provided directly by process operators for use in the NAEI. Emission factors and emission estimates for the most important sources from the iron and steel industry are shown in Table 4.5. As stated earlier, these factors are implied factors that include processes in smaller iron foundries where factors are higher per unit of output than larger plant. The figures in Table 4.5 refer to implied factors in 2000 carried forward to 2001. These are under review and are expected to be superseded in later versions of the inventory.

204. Many of the emission estimates for the iron and steel industry shown in Table 4.5 are considered very uncertain because of the scope for fugitive emissions from these processes, although they may be known with greater accuracy for larger iron and steel plant. The figures for blast furnaces, basic oxygen furnaces and electric arc furnaces are, perhaps, most reliable, whereas those for foundries and stockpiles are probably the least reliable. The uncertainty factors for these processes are under review.

4.2.3.9 Aluminium production

205. In the UK, most aluminium is produced by the pre-baked anode cell process, although one plant operated the older Soderberg Cell process until 2000. Emissions occur both from the smelting process and from the preparation of anodes by baking. Alumina was produced from bauxite ore at one plant in Scotland until 2002 and emissions occurred from the grinding and drying stages of the process. Emission factors and emission estimates are shown in Table 4.5 and emission estimates are assumed to be fairly reliable.

4.2.3.10 Chemical processes

206. Emissions from chemical processes are estimated based on data given in the Pollution Inventory (Environment Agency, 2002a) and are reported in the following categories:

- chemical industry (alkyl lead): emissions from the manufacture of alkyl lead petrol additives;
- chemical industry (ammonia-based fertilizer): emissions from manufacture of ammonium nitrate;
- chemical industry (carbon black): emissions from the non-fuel use of petroleum fractions in the manufacture of carbon black;
- chemical industry (chromium chemicals): emissions from manufacture of chromium-based chemicals;
- chemical industry (magnesia): emissions from the production of magnesia;
- chemical industry (phosphate based fertilizer): emissions from manufacture of superphosphate from phosphate rock;
- chemical industry (titanium dioxide): emissions from manufacture of titanium dioxide; and
- chemical industry: emissions from chemical processes not included elsewhere.

207. In general, activity data are not available for these processes; thus, although the emissions data can be used in the NAEI, it is not possible to derive meaningful emission factors. Emissions from the most important sources from chemical processes are shown in Table 4.5.

208. Emission estimates for chemical processes are likely to be fairly robust, although the possibility of under-reporting of emissions from fugitive sources cannot be ruled out.

4.2.3.11 Bricks, tiles, refractories and ceramics

209. Fletton bricks are manufactured from Lower Oxford Clay, which contains significant levels of sulphur-containing organic material. This organic material is burnt when the bricks are fired and contributes heat to the process, thus less fossil fuel is required. The combustion of this material does lead to emissions of CO_2, CO, SO_2, VOCs and PM, and these emissions have been estimated based on data available in the Pollution Inventory (Environment Agency, 2003a). Since these data will also include emissions from the burning of fossil fuels, an estimate is first made of fossil fuel use by Fletton brick producers and the emissions from these fuels are calculated. These figures are subtracted from the Pollution Inventory data to leave estimates of the emissions from the burning of the organic component of the clay alone.

210. Non-Fletton bricks are manufactured from clays other than Lower Oxford Clay. These clays do not contain significant quantities of organic material and the only emission of significance is likely to be PM, which is generated mainly during the firing of the bricks. Emissions of this pollutant are also likely during the manufacture of tiles, refractories and ceramics. Emissions of PM_{10} have been estimated using emission factors based on USEPA data, as recommended in Passant *et al*. (2002). Emission factors and estimates for significant sources from Fletton and non-Fletton brick manufacture are shown in Table 4.5.

211. Estimates for Fletton bricks are probably more robust than those for non-Fletton bricks, although the quality of both estimates is likely to be acceptable given uncertainties elsewhere.

4.2.3.12 Other processes

212. Although it is likely that almost all manufacturing processes give rise to some emissions of PM, the Pollution Inventory only contains estimates for those where emissions are expected to be significant. Currently, estimates are made for the following fifteen processes:

- primary lead/zinc production (ceased operation in 2003);
- secondary copper production (ceased operation in 2000);
- secondary lead production;
- secondary aluminium production;
- nickel refining;
- copper semis and alloys;

- zinc alloys;
- zinc oxide production;
- lead batteries manufacture;
- other non-ferrous metals;
- glass production (split into flat, container, domestic, lead crystal, continuous filament glass fibre, wool, frits, and special glass);
- asphalt manufacture (roadstone coating);
- cement batching;
- wood product manufacture; and
- other Part B (LAPC) processes.

213. Emission estimates for all processes except the last five are based on Pollution Inventory data (Environment Agency, 2002a), with the full time-series being completed by extrapolation of the Pollution Inventory data based on metal production or consumption data in the first seven cases and based on Department of Trade and Industry (DTI) indices of manufacturing output in the other three cases. Emission factors can be generated for those sectors where the Pollution Inventory data can be compared with published activity data for the sector. Emission factors for glass production are based on data reported in the Pollution Inventory in the case of frits, continuous filament glass fibre and glass wool and from data provided by the process operator for flat glass (Pilkington Glass Ltd, 2003). Information contained in the EMEP/CORINAIR Emission Inventory Guidebook (EMEP/CORINAIR, 1999) and the European IPPC Bureau's BREF note for glass manufacture (EIPPCB, 2000) was used for the other types of glass production. The emission factors for asphalt manufacture, cement batching and wood product manufacture are taken from the USEPA compilation of emission factors (USEPA, 1997). Other Part B processes cover a range of industrial processes including coal handling, metal processing, galvanising and coating. Emission estimates for these Part B processes are based on emission estimates for 'typical' processes – which were originally available from the UK Emission Factor Database at http://rsk.co.uk/ukefd/ – multiplied by estimates of the number of processes in the UK. Emission factors and emission estimates for the most important of these sources are given in Table 4.5. Although these sources may not be important nationally, they could be important local sources.

214. The estimates for other Part B processes in particular are subject to a very high level of uncertainty, although estimates for many of the sources are not as robust as those for combustion processes. The contribution of fugitive emissions could be underestimated.

4.2.3.13 Mining and quarrying

215. Emissions of PM_{10} from open-cast mining and quarrying were estimated using an USEPA (1997) factor of 0.1 g kg^{-1} of material throughput. This gives a very crude estimate of 20.6 kt in 2001, which does not take into account changes in technology and abatement over time. The NAEI methodology for this sector is currently being reviewed (see Section 4.2.4.3).

4.2.3.14 Construction

216. The emission estimates for construction are based on an emission estimate for 1990 presented in the third QUARG report (QUARG, 1996). Estimates for other years are extrapolated from this figure on the basis of the value of construction work done, giving a figure of 4.3 kt for 2001. This very crude estimate is currently being reviewed (see Section 4.2.4.3).

4.2.3.15 Waste incineration

217. The NAEI estimates emissions from the categories municipal solid waste (MSW) incineration, clinical waste incineration, sewage sludge incineration, animal carcass incineration and chemical waste incineration. The emission factors and emission estimates for the most important sources are shown in Table 4.7. The factor for MSW incineration is supported by an estimate of 38g/tonne waste based on site-specific data from waste management activities in a study carried out for Defra by Enviros Consulting and Birmingham University (www.defra.gov.uk/environment/waste). No emission factor is available for animal carcase incineration, since the source of data used for this sector only gives an estimate of total emissions from the sector (AEA Technology, 2002). Emission estimates for offshore flaring are provided by the United Kingdom Offshore Operators Association (UKOOA, 2002).

Table 4.7 Emission factors and emission estimates for PM_{10} from incineration and flaring.

Source	Emission factor	Emission in 2001 (kt)
MSW incineration	24.7 g t^{-1} waste	0.1
Clinical waste incineration	270 g t^{-1} waste	0.1
Chemical waste incineration	320 g t^{-1} waste	0.1
Animal carcase incineration	—	0.3
Offshore flaring	0.604 g m^{-3} gas flared	1.1

218. Emission estimates for incineration processes are likely to be fairly robust, whereas the reliability of the UKOOA figure for offshore flaring is likely to be less so. Dust emissions from flaring in other sectors of industry (chemicals, refineries) are not currently included in the NAEI but are unlikely to be as important as emissions from offshore flaring.

4.2.3.16 Agriculture

219. Emission sources include dust from agricultural activities such as ploughing, dust from animal housing and the use of fuels to provide heating. Straw is burned as a fuel but the *in situ* burning of stubble and crop residues has been prohibited since the mid-1990s, with limited exemptions. Emission factors and emission estimates for the most significant sources are shown in Table 4.8.

220. Emission estimates for agricultural operations must be considered among the most uncertain in the NAEI due to the fugitive nature of most of the emissions.

Table 4.8 Emission factors and emission estimate for PM_{10} from agriculture.

Source	Emission factor	Emission in 2001 (kt)
Agricultural activities	0.105 g t^{-1} cereals	2.0
Straw combustion	11000 g t^{-1}straw	2.2
Housed livestock:		
laying hens	19.5 g per head	0.5
broilers	58.8 g per head	6.6
other poultry	74.1 g per head	2.5
pigs	260 g per head	1.4
dairy cattle	130 g per head	0.4
other cattle	41.9 g per head	0.3

4.2.3.17 *Foot and mouth pyres*

221. During 2001 the UK suffered from an outbreak of foot and mouth disease. Due to the highly infectious and debilitating nature of the disease, action taken to contain the outbreak included the slaughter of over 6,000,000 farm animals with the subsequent need for disposal of carcasses. The slaughter included 4,200,000 animals for disease-control purposes and 2,300,000 animals for welfare reasons. Approximately one-third of the carcasses were disposed of by burning (including *in situ* burning, mass pyres and incineration). The burning of carcasses on mass pyres occurred between March 2001 and May 2001 and involved the burning of animals together with specified quantities of fuels. The fuels included wooden railway sleepers, kindling wood, straw, coal and diesel oil. Emissions of air pollutants occurred both due to the combustion of these fuels and the burning of the carcasses. Emissions of PM_{10} were estimated for inclusion in the Pollution Inventory. Table 4.9 shows the estimates and emission factors expressed as emissions per animal carcass.

Table 4.9 Emission factors and emission estimates for PM_{10} from foot and mouth pyres.

Source	Emission factor	Emission in 2001 (kt)
Cattle pyres	12.5 kg per carcass	3.0
Sheep pyres	0.682 kg per carcass	0.6
Pig pyres	3.01 kg per carcass	0.2

4.2.3.18 *Chemical composition of PM emissions from stationary sources*

222. Information on particle composition is not currently collected for the NAEI, so no data on the chemical composition of particles emitted can be provided. In general, however, it would be expected that emissions from small and inefficient combustion processes would largely consist of carbon, whereas emissions from larger combustion processes would contain some carbon, but would predominantly consist of inorganic matter deriving from the mineral content of the fuel. This may include metals in the fly ash. PM emissions from most industrial processes (for example, cement manufacture), quarrying and construction would be inorganic matter, often chemically similar to the raw materials or products of the process. These will usually contain a high proportion of alkaline earth metal compounds (for example, calcium).

4.2.4 A recent review of PM emission factors for industrial processes and combustion

223. The NAEI is currently carrying out a review of PM emission factors for a range of industrial processes, combustion, mining and quarrying and construction.

224. Analysis of the most significant industrial emitters of PM_{10} and smaller particle sizes indicates several important sectors:

- coal combustion and natural gas combustion for power generation public utility power stations or other industry subsectors;
- iron and steel sintering plants; and
- manufacture of non-Fletton bricks.

Each of these sectors and the manufacture of cement clinker have been reviewed.

225. For many sources, the emission factor reference documents generally refer to the USEPA publication AP42 for sources of raw data (often via several other references). Although AP42 is a valued reference, it is important to note that – within a number of sections – emission data have not been updated for many years. This is particularly relevant to PM size speciation profiles, some of which – despite advances in abatement technology – have not been updated for over a decade.

226. A brief summary of the main recommendations of the NAEI review and proposed changes to emission factors is given in the following sections.

4.2.4.1 *Review of industry processes*

227. The review of industrial emissions of PM_{10} has highlighted several issues including the fundamental question of what is PM_{10} when emitted from a process. In general, the NAEI includes emission estimates based on a ‘filterable’ PM_{10} release. However, the emission factors within the NAEI for natural gas combustion assessed in this task also include a ‘condensable’ fraction.

228. The USEPA defines two fractions to a PM_{10} emission from an industrial or combustion source. The filterable PM_{10} fraction in a stack emission can be determined using USEPA Method 201A and is essentially material that is smaller than nominally 10 μm (separated from larger PM using a miniature cyclone) and that can be retained by a filter at stack temperature. Condensable PM is also considered to be PM_{10} and is material collected downstream of the sampling filter from probe rinses and material collected within a chilled impinger train (USEPA Method 202).

229. The test procedure for condensable PM (Method 202) is an official USEPA test procedure, and emission inventories in the US are required to consider both filterable PM_{10} and the condensable PM_{10} emissions. However, the test method has been the subject of considerable debate over whether it provides data that are representative of condensable emissions. It is intended to replicate secondary formation of particulate by condensation downstream of the stack. However, although the sampling procedure lowers the temperature of the sampled gases, there is no concurrent dilution action. Interfering effects are also possible. Limited evidence from dilution tunnel experiments suggest that Method 202 may overestimate condensable emissions.

230. The NAEI incorporates PM_{10} data reported by process operators to the regulatory authorities. The regulatory authorities require operators of selected processes to report PM_{10} emissions for inclusion in the Pollution Inventory (or EPER) above a threshold of 10 t year ^{1}. This information is then incorporated into the NAEI. The methodologies adopted by process operators for estimating PM_{10} emissions includes a mixture of direct measurements, emission factors or factors applied to a total particulate emission.

231. The protocol for inventory reporting adopted by the Electricity Supply Industry Joint Environment Programme (JEP) is typical in that it applies a PM_{10} factor to the total particulate emission. The JEP protocol states that the Electricity Supply Industry total particulate emission data for coal-fired stations are based on continuous monitoring systems. These systems do not monitor the condensable fraction. Similarly, other operators may base emissions data on short-term measurements of particulate emission, which almost certainly do not include assessment of the condensable fraction. Conversely, other operators may include condensable material in their reports through the use of USEPA default emission factors for their process.

232. The scale of these inconsistencies and their impact on the NAEI is unknown. The consistency of Pollution Inventory/EPER reporting by process operators will be addressed in time by various initiatives, including guidance from the regulatory authorities or operator initiatives such as the JEP protocol.

233. Hence, methodology issues result in inconsistencies in the treatment of PM_{10} within the NAEI. The bulk of existing PM_{10} emission data in the NAEI are for 'filterable' PM_{10} and, hence, to ensure consistency within the NAEI, only filterable PM_{10} data should be considered. Default PM_{10} and other PM_{10} emission factors used in the NAEI for industrial and combustion sources should be reviewed to determine if any include a condensable component.

234. The review has concluded that for the manufacture of cement clinker, combustion of coal and iron and steel sintering processes, no changes to the current PM_{10} emission factors should be made, but further work is required to develop more robust particle size speciation data for these sources. Currently used USEPA speciation data are considered out-of-date and not appropriate for UK plants. For the manufacture of non-Fletton bricks, both a revision to the PM_{10} emission factor and measurements of size speciation data are recommended.

4.2.4.2 Review of emissions from the combustion of natural gas

235. Natural gas is a key fuel for industry and power generation. Combustion of natural gas is among the five highest emitters of particulates for several PM fractions. This is perhaps surprising given the inherently low potential for particulate emission from the combustion of refined gas. The scale of gas use is increasing and the significance of the sector is also increasing as other sources decline. The emission of PM from combustion of natural gas is also significant for the 'other industry' sector.

236. At present, the UK Electricity Supply Industry JEP inventory reporting protocol reports PM_{10} emissions from gas combustion as 0 or operators report emissions below the reporting threshold (10 t year^{-1}). However, the NAEI estimates UK emissions of PM_{10} from combustion of natural gas using emission factors and activity statistics. For the electricity supply industry, the NAEI default emission factor is currently 154 kg Mtherm^{-1}; however, the source of this factor is unclear. The default factor is about half the USEPA emission factor for total PM (filterable and condensable PM), which is currently used for the other industry and other gas combustion sectors within the NAEI (338 kg Mtherm^{-1}).

237. The review indicates that the USEPA emission factor for filterable PM from combustion of natural gas (84 kg Mtherm^{-1}) should be used by the NAEI in all sectors. Measurement of filterable PM_{10} at a power station combined cycle gas turbine for Defra and a smaller gas turbine for the Environment Agency as well as data for gas combustion published by EMEP indicate PM_{10} emission factors of a similar order of magnitude to the default USEPA factor. However, it should be noted that these emission measurements are likely to have a high uncertainty.

238. The proposed emission factor is approximately half that of the current 'default' factor used for electricity generation. However, it is traceable and consistent with PM_{10} emissions from combustion in other areas of the NAEI (that is, it is consistent with the definition of PM_{10} from other sources reported in the Pollution Inventory). Application of the filterable emission factor to combustion of gas in the domestic sector represents a 75% reduction in the emission factor currently used. The consequences of this change to estimates of low-level emissions of PM_{10} in urban areas can be expected to be significant and requires evaluation within the inventory.

4.2.4.3 Review of emissions from mining, quarrying and construction

239. A review of PM emissions from the mining, quarrying and construction sectors is still in progress. However, early indications are that the PM emission factors currently used in the NAEI for many of these processes may be too high and need to be reduced.

4.2.5 Fireworks and associated fires

240. Fireworks and the fires associated with bonfire night are important sources of particle emissions in the UK. It is known from ambient concentrations (see Chapter 8) that emissions from these sources can result in concentrations of PM_{10} that are greater than 50 µg m^{-3} for several days each year. These sources of particles can, therefore, be locally important, although no emission rate estimates are available to compare with other primary sources of PM. This source can also represent an important source of metals in the atmosphere, which are either contained in the fireworks themselves or originate from the combustion of materials that contain metals.

4.3 Secondary particle precursors

4.3.1 Origin of secondary particles

241. Secondary particles are generated by gas-to-particle conversion processes, following the formation of products of particularly low volatility or high solubility from the gas phase oxidation of emitted precursor gases. In addition to this, transfer of material from the gaseous to the particulate phase may also occur as a result of the reactions of gaseous species with aerosol or aqueous droplets, when the products of such reactions remain either wholly or partially in the condensed phase. In this section, the sources of secondary particle precursors are identified and a description is given of the chemical processes that convert these precursors into the major components of secondary particles, namely sulphate, nitrate, ammonium and organic.

4.3.2 Sulphate

242. The major precursor to sulphate aerosol formation in the UK is SO_2. As is described further in Section 4.4.3, SO_2 is mainly derived from the combustion of solid fuel and petroleum products; in the UK its major sources are power generation and industrial combustion processes.

243. As shown in Figure 4.3, the oxidation of emitted SO_2 can occur by both gaseous phase and aqueous phase routes. The predominant gas phase reaction for SO_2 is the reaction with the hydroxyl radical (OH):

$$\mathrm{OH + SO_2\ (+M) \rightarrow HOSO_2\ (+M)} \qquad (1)$$

where M is a third body, most commonly nitrogen (N_2). At a typical background OH concentration of 0.04 ppt, SO_2 is oxidized at a rate of about 0.3% h^{-1}, although the conversion rate is correspondingly greater at elevated OH concentrations consistent with a photochemical episode (that is, up to approximately 0.1 ppt). Gaseous H_2SO_4 is subsequently formed by the following sequence of reactions:

$$\mathrm{HOSO_2 + O_2 \rightarrow SO_3 + HO_2} \qquad (2)$$

$$\mathrm{SO_3 + H_2O\ (+M) \rightarrow H_2SO_4\ (+M)} \qquad (3)$$

244. In comparison with reaction (1), these reactions occur essentially instantaneously (typically on the timescales of 1 μs and 10 ms, respectively, in the boundary layer) and are consistent with OH-initiated oxidation, leading to quantitative conversion of SO_2 to gaseous H_2SO_4. Consequently, the fractional conversion of SO_2 to gaseous H_2SO_4 is determined by how efficiently the gas-phase oxidation route, initiated by the reaction with OH radicals, competes with the alternative oxidation or removal routes for SO_2. H_2SO_4 is sufficiently involatile to generate secondary particles by the processes of nucleation and condensation, as described in Section 2.1. The gas phase oxidation of SO_2 may, therefore, result in the formation of new particles or the growth of existing ones.

245. Although the gas phase oxidation of SO_2 can dominate under specific conditions (for example, summertime photochemical episodes), oxidation in the aqueous phase is generally more important. This is largely governed by how frequently polluted air masses encounter clouds and occurs at a rate of about 1% h^{-1}, on average, under UK conditions (Metcalfe *et al.*, 1995). As shown in Figure 4.3, the solvated form of SO_2 ($SO_2.H_2O$) is present in equilibrium with the ionic species HSO_3^- and SO_3^{2-}. In all these species, sulphur is in the +4 oxidation state; they are usually collectively denoted S(IV). Oxidation to the +6 oxidation state (S(VI) or 'sulphate') is promoted by the presence of aqueous phase oxidants and catalysts. As is described in detail elsewhere (Finlayson-Pitts and Pitts, 1999), dissolved hydrogen peroxide (H_2O_2) and ozone (O_3) are believed to be particularly important, although oxidation by O_3 becomes progressively less efficient with increasing acidity of the cloud droplets. The chemical processing of SO_2 by the cloud therefore leads to the oxidation of S(IV) to S(VI) and, upon evaporation, the resultant particles will contain sulphate. Unlike the gas-phase oxidation route, however, aqueous phase oxidation cannot result in new particle formation.

4.3.3 Nitrate

246. Nitrate aerosol is formed from the oxidation of emitted NO_X. As is described in detail elsewhere (AQEG, 2004), and in Section 4.4.3, NO_X is mainly derived from fossil fuel combustion, with its major sources in the UK being road transport and power generation.

247. As shown in Figure 4.4, nitrate aerosol is formed from the oxidation of NO_2, which is itself mainly derived from the oxidation of emitted NO. The predominant daylight oxidation route for NO_2 is initiated by its reaction with OH:

$$OH + NO_2 \ (+M) \rightleftarrows HNO3 \ (+M) \qquad (4)$$

which leads to the quantitative formation of gaseous HNO_3. At a typical background OH concentration of 0.04 ppt, NO_2 is converted to HNO_3 at about 5% h^{-1} by this reaction, although the conversion rate is correspondingly more rapid at elevated OH concentrations consistent with a photochemical episode (that is, up to ~0.1 ppt). HNO_3 is highly soluble and also shows a strong affinity for surfaces. It is, therefore, scavenged by existing particles and droplets to form nitrate aerosol, in competition with removal by deposition to the ground.

248. Nitrate aerosol is also generated via the formation of the higher oxide, N_2O_5, which occurs as follows:

$$NO_2 + O_3 \rightarrow NO_3 + O_2 \qquad (5)$$
$$NO_3 + NO_2 \ (+M) \rightleftarrows N_2O_5 \ (+M) \qquad (6)$$

249. This reaction sequence is very inefficient during the day and at higher levels of NO because the intermediate NO_3 radical photolyses and reacts with NO rapidly, leading to NO_X regeneration. Because N_2O_5 is in equilibrium with NO_3 (reaction 6), it also cannot persist at significant concentrations below these conditions. During the night, significant NO_2 to N_2O_5 conversion can occur, with the potential for nitrate aerosol generation from the reaction of N_2O_5 with H_2O on existing particle surfaces or aqueous droplets:

$$N_2O_5(g) + H_2O \ (+aerosol) \rightarrow 2HNO_3 \ (adsorbed or dissolved (7)$$

Figure 4.3 Major chemical processes involved in sulphate aerosol production.

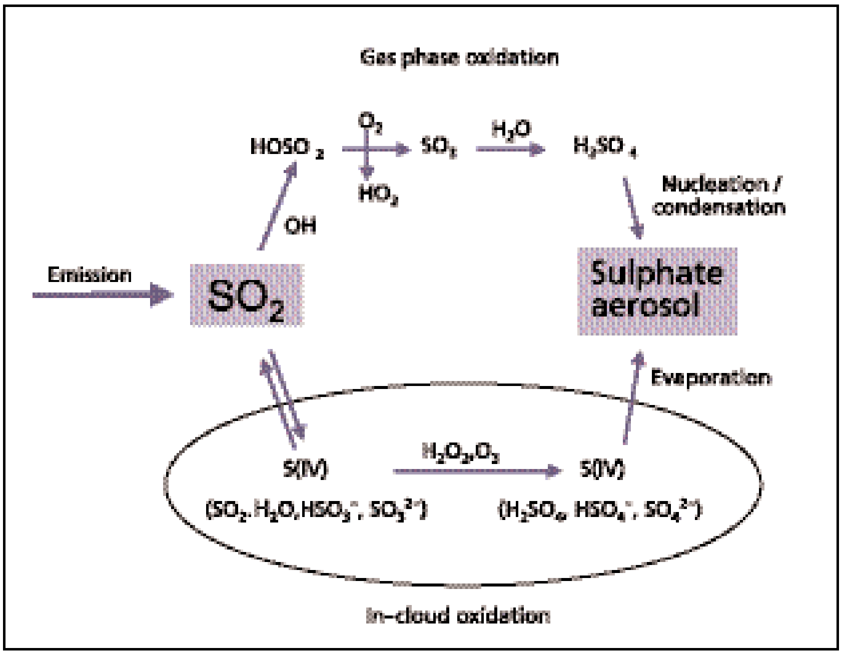
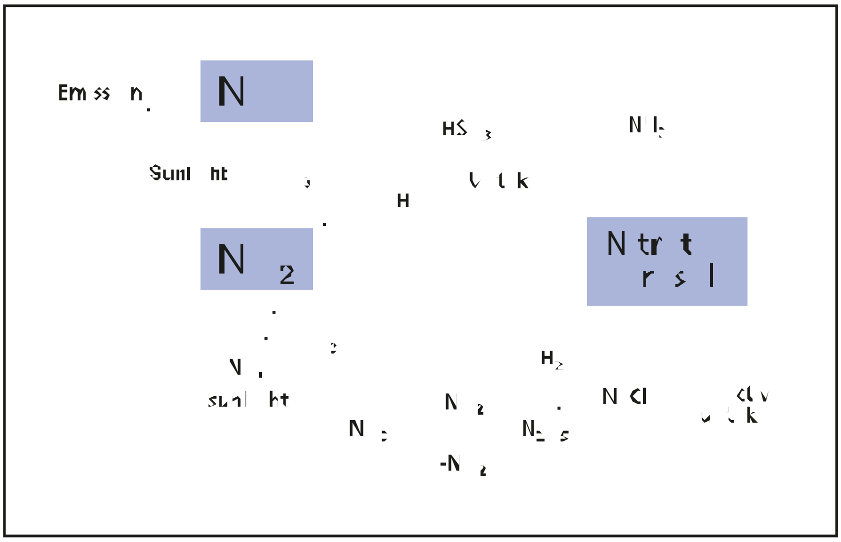


Figure 4.4 Major chemical processes involved in nitrate aerosol production.



250. The reaction of N_2O_5 with sea salt aerosol also leads to the formation of nitrate aerosol, with partial regeneration of NO_2 (and release of chlorine), as follows:

$$N_2O_5(g) + NaCl\ (aerosol) \rightarrow NaNO_3(aerosol) + ClNO_2(g) \quad (8)$$

$$ClNO_2 + sunlight \rightleftarrows Cl + NO_2 \quad (9)$$

251. The conversion of NO_2 to nitrate aerosol, via N_2O_5, potentially occurs at a comparable rate to that from OH-initiated oxidation during daytime. However, its efficiency can be reduced by the reaction of NO_3 with residual night time NO or with some organic compounds.

4.3.4 Ammonium

252. The precursor to ammonium aerosol formation is ammonia (NH_3). As is described further in Section 4.4.3, NH_3 emissions are dominated by agricultural sources, which are mainly due to the decomposition of urea and uric acid in livestock waste. A small, but increasing, contribution is also made by road transport, from cars fitted with three-way catalysts.

253. NH_3 is the most abundant alkaline gas in the atmosphere and, therefore, plays a significant role in neutralizing acids. It is efficiently taken up into acidic sulphate and nitrate aerosols, formed by the processes described above, leading to the formation of ammonium aerosol. As a consequence, NH_3 uptake also serves to regulate the acidity of aerosol and cloud droplets, which, in turn, influences the solubility and oxidation rate of species such as SO_2.

254. As is described in Sections 2.1 and 2.2, NH_3 also plays a role in the ternary nucleation of sulphate aerosols and reacts with gaseous HNO_3 to form particulate NH_4NO_3.

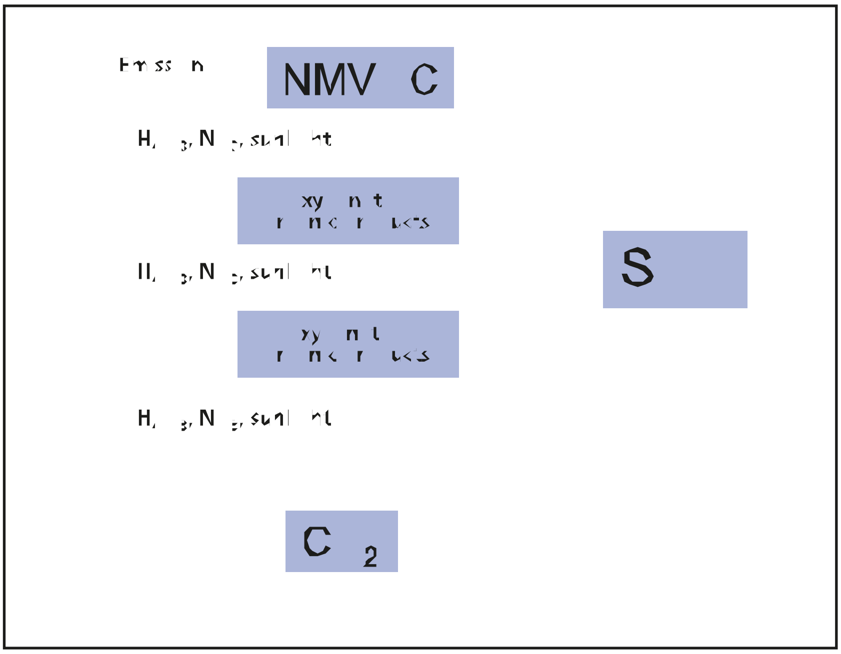
4.3.5 Organics

255. Secondary organic aerosol (SOA) is formed when the oxidation of emitted NMVOCs generates oxygenated products that undergo gas-to-particle transfer. As is described further in Sections 4.4.3 and 4.4.4, important precursor NMVOCs may be emitted from both anthropogenic and biogenic sources.

256. It is well known that hundreds of individual NMVOCs are emitted (Goodwin *et al.*, 2001), with contributions from many different classes (for example, alkanes, and aromatics). The gas phase oxidation of each NMVOC broadly follows the pattern illustrated in the simplified schematic in Figure 4.5. This shows that the oxidation may be initiated by one or more reactions, the relative importance of which depends on the structure of the given NMVOC and on the ambient conditions. The initial oxidation step leads to the production of a set of organic products containing polar oxygenated functional groups, such as aldehyde (–C(=O)H), ketone (–C(=O)–), alcohol (–OH), nitrate ($–ONO_2$) and carboxylic acid (–C(=O)OH) groups, which tend to make the products less volatile and more water soluble. Further oxidation may introduce additional functional groups, such that the ‘second generation’ set of products (and subsequent generations) may be of even lower volatility and higher solubility. However, the oxidation mechanisms also lead to fragmentation of the carbon chains to form lower molecular weight oxygenates (which are more volatile), and the gas phase oxidation ultimately converts all the carbon in the emitted NMVOCs into CO_2. This is in stark contrast, therefore, to the situation for SO_2 and NO_X, for which the ultimate oxidation products (sulphate and nitrate) are the forms that are most associated with the particle phase. SOA is, therefore, formed from the gas-to-particle transfer of partially oxidized organic material, which occurs in competition with further oxidation in the gas phase.

257. As a result of the complexity of the emitted NMVOC mixture and the oxidation chemistry, the atmosphere contains many thousands of structurally different organic oxygenates, which possess a wide range of properties (for example, reactivity, volatility and aqueous solubility) and, therefore, different propensities to undergo gas-to-particle transfer. The importance of emitted NMVOCs as SOA precursors is, therefore, very variable, with most emitted compounds being very inefficient at generating SOA (indeed, the vast majority of atmospheric organic material resides in the gaseous phase, even though the small fraction present in particles makes an important contribution to particle mass). However, certain classes of NMVOCs are more likely to lead to aerosol formation by virtue of their general high reactivity and types of oxidation product formed. Of particular significance are large, cyclic, unsaturated compounds: the oxidation products tend to be of comparatively high molecular weight and contain two or more polar functional groups, these products are, therefore, of lower volatility. Two such classes of emitted NMVOCs, which generate SOAs, are monoterpenes (for example, α and ß-pinene) and aromatic hydrocarbons (for example, toluene and xylenes), although other classes may also contribute. Monoterpenes are naturally emitted NMVOCs that are estimated to account for a substantial fraction (30–80%) of the UK biogenic total, with an important contribution made by coniferous trees (for example, Dore *et al*., 2003; EMEP-CORINAIR, 2004). Aromatic hydrocarbons are primarily of anthropogenic origin, accounting for about 15% of the UK anthropogenic NMVOC total (Goodwin *et al*., 2001). The majority of this derives from road transport emissions.

Figure 4.5 Simplified representation of NMVOC oxidation to CO_2 and formation of SOA as a by-product.



258. Monoterpenes undergo rapid photo-oxidation and reaction with O_3. Particularly significant oxidation products are bi- and multifunctional organic acids, such as pinic acid and pinonic acid. The results of chamber studies demonstrate that these acids play a major role in the SOA formation process (Yu *et al*., 1999a; Glasius *et al*., 2000; Kuckelmann *et al*., 2000); they have also been detected and quantified in ambient aerosols (Kavouras *et al*., 1999; Yu *et al*., 1999b; ODowd *et al*., 2002). Chamber studies of the photo-oxidation of aromatic hydrocarbons have also confirmed that SOA is formed (Odum *et al*., 1997; Kleindienst *et al*., 1999). Although the composition of SOA from aromatic oxidation is less well characterized, highly substituted cyclic products (for example, nitrocresols, catechols and quinones) and ring-opened bi- and multifunctional acids have been detected in the particle phase (Forstner *et al*., 1997; Jang and Kamens, 2001).

259. The formation of SOA from monoterpene and aromatic hydrocarbon oxidation in chamber experiments has generally been interpreted successfully in terms of the partitioning of semi-volatile oxidation products between the gaseous and particulate phases (Odum *et al*., 1996; Hoffmann *et al*., 1997; Seinfeld and Pankow, 2003), as described for PAHs in Section 2.2. However, it is also becoming clear that the partitioning organic species can be further stabilized in the condensed phase by association and polymerization reactions (Tobias and Ziemann, 2000; Kalberer *et al*., 2004), which may be enhanced in acidic aerosols (Jang *et al*., 2002), and it is probable that such processes play an important role in atmospheric SOA formation.

4.4 Emission estimates by source type

4.4.1 Directly estimated primary emissions

260. Figure 4.6 shows the contribution made by different source sectors by particle size for 2001 based on an analysis of data from the NAEI. In general the particles derived through combustion processes dominate the smaller size fractions. For example, road transport is responsible for only 18% of total PM_{10} emissions but 35% of PM_1 emissions. The converse is true for other sectors, such as production processes, where their contribution diminishes as the particle size fraction reduces. It is clearly important, therefore, to consider particle size when considering the contributions made by different source sectors. Further consideration is given to PM emissions by size distribution in the next section.

Figure 4.6 Contribution made by different source sectors by particle size.

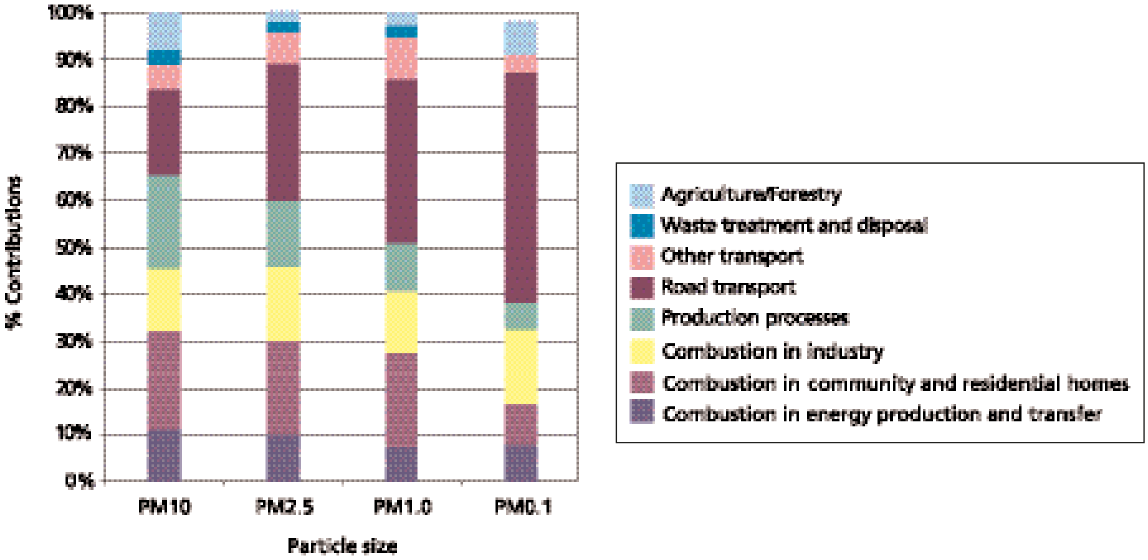


Table 4.10 Emission estimates for PM by size fraction in 2001.

Sector	Emissions in kt	Percentage by size fraction			
	PM_{10}	$PM_{(10-2.5)}$	$PM_{2.5}$	PM_1	$PM_{0.1}$
Electricity generation:					
Coal	15.7	56.1	43.9	19.1	8.3
Other fuels	2.0	5.0	95.0	90.0	40.0
Industrial/commercial combustion:					
Coal	5.4	35.2	64.8	29.6	7.4
Natural gas	3.7	0.0	100.0	100.0	51.4
Other fuels	6.8	45.6	54.4	32.4	14.7
Domestic combustion:					
Coal	12.6	61.9	38.1	31.7	2.4
Smokeless fuels	6.5	73.8	26.2	4.6	1.5
Wood	7.1	29.6	70.4	29.6	8.5
Natural gas	4.2	0.0	100.0	100.0	9.5
Other fuels	0.7	28.6	71.4	71.4	14.3
Industrial processes:					
Iron and steel	7.3	37.0	63.0	41.1	20.5
Cement and lime	3.3	46.4	53.6	25.0	10.7
Bricks, glass and ceramics	3.8	26.3	73.7	68.4	34.2
Quarrying	20.6	70.9	29.1	8.3	0.0
Construction	4.3	69.8	30.2	11.6	0.0
Other	9.0	40.0	60.0	51.1	18.9
Transport:					
Road transport – petrol	3.0	16.7	83.3	70.0	33.3
Road transport – diesel	27.4	10.2	89.8	85.0	50.0
Road transport – brake /tyre wear	8.9	44.9	55.1	10.1	7.9
Off-road vehicles	7.2	22.2	77.8	69.4	15.3
Other transport	1.9	10.5	89.5	84.2	15.8
Waste disposal:					
Foot and mouth pyres	3.8	60.5	39.5	31.6	2.6
Other	1.5	0.0	100.0	100.0	13.3
Agriculture	13.7	83.9	16.1	15.3	15.3
Total	180	43.3	56.7	41.2	17.2

Table 4.11 Contribution to UK PM emissions by source sector and size fraction in 2001.

Sector	Percentage by size fraction			
	PM_{10}	$PM_{2.5}$	PM_1	$PM_{0.1}$
Electricity generation:				
Coal	8.7	6.7	4.0	4.2
Other fuels	1.1	1.9	2.4	2.6
Industrial/commercial combustion:				
Coal	3.0	3.4	2.2	1.3
Natural gas	2.1	3.6	5.0	6.1
Other fuels	3.8	3.6	3.0	3.2
Domestic combustion:				
Coal	7.0	4.7	5.4	1.0
Smokeless fuels	3.6	1.7	0.4	0.3
Wood	3.9	4.9	2.8	1.9
Natural gas	2.3	4.1	5.7	1.3
Other fuels	0.4	0.5	0.7	0.3
Industrial processes:				
Iron and steel	4.0	4.5	4.0	4.8
Cement and lime	1.8	1.7	1.1	1.1
Glass, bricks and ceramics	2.1	2.7	3.5	4.2
Quarrying	11.4	5.9	2.3	0.0
Construction	2.4	1.3	0.7	0.0
Other	5.0	5.3	6.2	5.5
Transport				
Road transport – petrol	1.7	2.4	2.8	3.2
Road transport – diesel	15.2	24.1	31.3	44.3
Road transport – brake/tyre wear	4.9	4.8	1.2	2.3
Off-road vehicles	4.0	5.5	6.7	3.6
Other transport	1.1	1.7	2.2	1.0
Waste disposal:				
Foot and mouth pyres	2.1	1.5	1.6	0.3
Other	0.8	1.5	2.0	0.6
Agriculture	7.6	2.2	2.8	6.8

4.4.2 Emissions of $PM_{2.5}$ and fine PM

261. The NAEI includes estimates of emissions of $PM_{2.5}$, PM_1 and $PM_{0.1}$. These estimates are derived by applying a set of size distribution profiles to the NAEI estimates of PM_{10} emissions. Generally speaking, PM_{10} emissions from industrial combustion processes and road transport are considered to contain more fine material than, for example, mechanically generated PM_{10} from quarries and construction sites. Table 4.10 summarises the NAEI estimates for fine PM and Table 4.11 shows how the percentage contribution of major source sectors changes for each size fraction. This table shows how emissions from industrial combustion processes and road transport contribute an increasingly large proportion of UK emissions with decreasing size while emissions from quarries and construction become increasingly trivial with decreasing size. The contribution of diesel powered road vehicles increases from 15% for PM_{10} to 45% for $PM_{0.1}$.

262. The size distributions used in the NAEI are mostly taken from the USEPA Compilation of Emission Factors (known as AP-42). These size profiles are given for a specific level of control and it is not clear how appropriate these profiles are for the UK (although a critique of the NAEI size profiles is currently being prepared). In general, the proportion of fine material in PM_{10} emissions will increase as the level of abatement increases because abatement technologies preferentially collect the coarser particles ($<PM_1$). Thus, if the level of control in the UK is greater than that assumed in the USEPA profiles, then it is likely that emissions of PM_1 and $PM_{0.1}$ in particular may be underestimated. This will have significance for the time series of emission estimates for the finer particle sizes (Section 4.5.2), where these were based on particle size fractions that stay constant with time.

263. A quantitative uncertainty assessment of the inventory for fine particles has not been undertaken, but the uncertainties must be considered higher than those for the estimates of PM_{10} emissions. A current review has not revealed any additional information on size distribution profiles for sources other than road transport exhaust emissions. Recent data on modern petrol and diesel car exhaust emissions seem to indicate the proportion of PM_{10} emitted as $PM_{0.1}$ is considerably lower than the current inventory figure of 50% by mass.

264. The NAEI produces national maps of emissions of PM and its precursors in the UK. Although these will not replace detailed local inventories, they provide a valuable basis from which a local inventory can be built and validated and are used in modelling assessments required for national policy-making.

4.4.3 Emissions of particle precursors

265. As explained in Section 4.3, the pollutants NO_X, SO_2, NMVOCs and NH_3 all contribute to the formation of PM in the atmosphere. Emissions of these precursor pollutants arise from a number of anthropogenic sources and the main sectors contributing to their emissions in the UK in 2001 are illustrated in Figure 4.7, based on the 2001 NAEI.

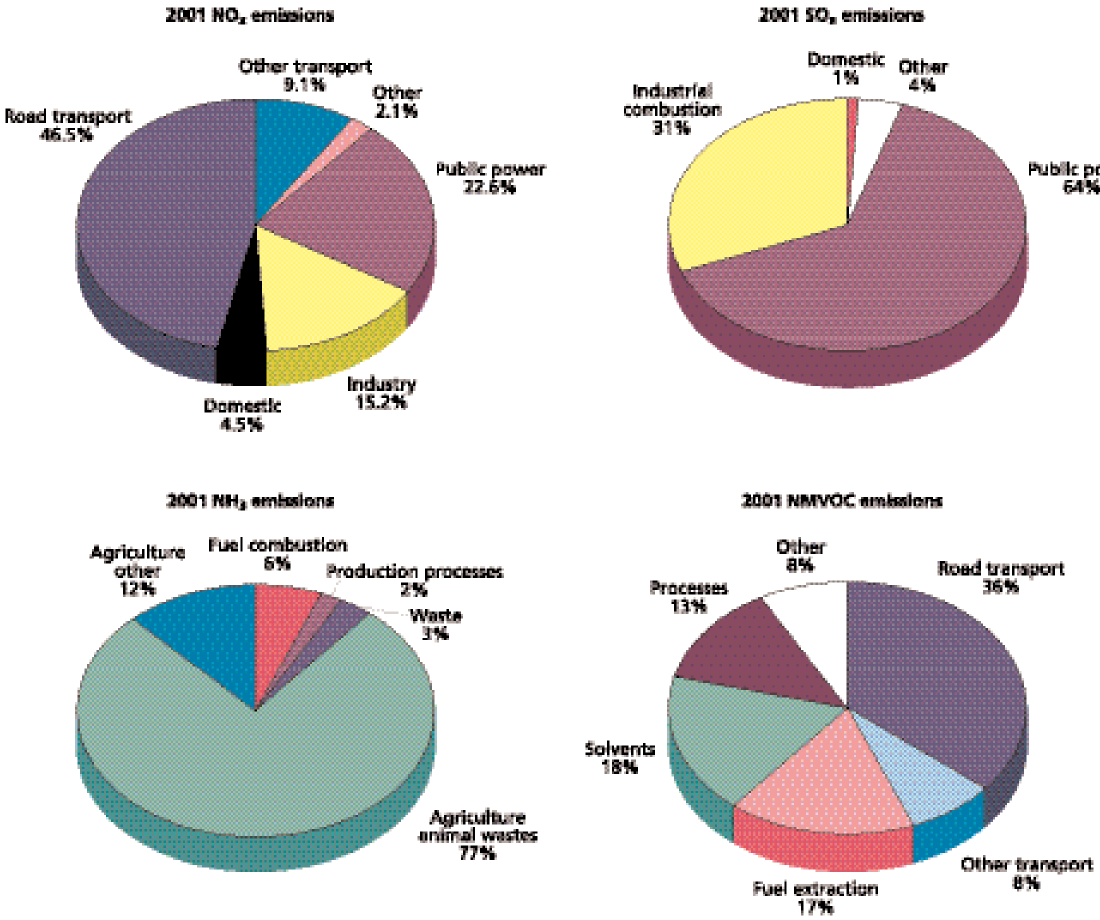
4.4.3.1 NO_X emissions

266. Emissions of NO_X occur mainly from combustion of fossil fuels and have been discussed in detail previously (AQEG, 2004). The most significant sources of NO_X are from road transport and public power generation. Emission occurs as a result of nitrogen present in the air and/or as a trace component in the fuel (for example, in coal) and the high temperatures of combustion. Emission estimates for NO_X from stationary sources are largely based on the use of emission factors combined with national energy statistics. However, emissions data provided by individual process operators for inclusion in the Pollution Inventory are also used, for example, in the cases of power stations and other large combustion plants. For these sources, emission estimates are considered to be of high quality. Area sources – which comprise smaller, but more numerous combustion sources (such as the domestic sector) – cannot be characterised to the same level of detail, and emission estimates are considered less accurate. Emissions from road transport are calculated using emission factors and transport activity data as discussed in Section 4.2.2.4 and in the report on NO_2 by AQEG (AQEG, 2004). Emissions of NO_X from road vehicles have been reasonably well quantified.

4.4.3.2 SO_2 emissions

267. Emissions of SO_2 also occur mainly as a result of the combustion of fossil fuels in which sulphur is naturally present as a trace component in varying amounts. Oxidation to SO_2 occurs during the high temperature combustion process. Sulphur is particularly prevalent in solid fuels such as coal. Consequently, the main consumer of coal – currently the electricity supply industry – is the most important source of SO_2. Total UK coal consumption in 2001 was 63.1 Mt, compared with the 105.9 Mt that were consumed 10 years earlier and the 154.7 Mt consumed 1970. Emissions have also been reduced through the adoption of measures to minimise emissions, including the use of lower sulphur content coals, as well as end-of-pipe abatement technology, such as the use of flue-gas desulphurisation (FGD) at coal-fired power stations. Emission estimates for SO_2 are generally derived by combining emission factors based on fuel sulphur contents with fuel use statistics. Some data provided by process operators for inclusion in the Environment Agencys Pollution Inventory are also used. Consequently, the quality of the inventory for SO_2 emissions is expected to be particularly high.

Figure 4.7 UK emissions of PM precursor pollutants from anthropogenic sources in 2001 by sector (data from the 2001 NAEI).



4.4.3.3 NMVOC emissions

268. Emissions of NMVOCs occur from a wider range of source types than for any other pollutant. The term NMVOC includes many hundreds of chemical species that are emitted to the atmosphere. Each of these has a different propensity to forming PM in the atmosphere based on its structure and chemical reactivity. The NAEI includes information on the species emitted from each source and this information has been published (Passant, 2002); it is also available at www.airquality.co.uk.

269. Major sources include domestic combustion, chemical processes, refineries, whisky production, production and distribution of liquid fossil fuels, industrial solvent use, domestic solvent use, combustion and evaporation of fuels from road and off-road vehicles. Emissions from combustion occur as a result of incomplete combustion of the fuel, reflecting the inefficiencies of the combustion process. Although these are important sources, non-combustion or fugitive sources made up nearly half of all UK emissions of NMVOCs in 2001. Total NMVOC emission estimates are generated via a number of approaches including use of literature-based emission factors, use of estimates provided by industry, estimates based on data held by regulators and emissions data reported in the Pollution Inventory and other inventories compiled by regulators. Fugitive emissions are more difficult to characterise compared to those arising from combustion sources and consequently their emissions estimates are associated with higher degrees of uncertainty.

4.4.3.4 NH_3 emissions

270. NH_3 emissions in the UK are dominated by emissions from agricultural animal wastes from the breakdown of urea and uric acid. Emission estimates for these processes are taken from the official UK NH_3 agricultural emissions inventory, which is compiled annually (Misselbrook *et al*., 2003). This inventory combines official UK agricultural statistics, such as livestock numbers, with emission factors from UK-based measurement programmes. The calculation methodology takes into account parameters such as nitrogen excretion rates and manure management techniques. However, NH_3 emissions are considered to be less well characterised than other pollutants. This is primarily because measurement of NH_3 from agricultural sources is considerably more uncertain than combustion emissions. Emissions from other sources are small, however, the contribution of road transport is growing due to the increasing use of three-way catalysts, which generate some ammonia as a by-product of the catalytic NO_X reduction process in the exhaust. This is discussed in detail in the AQEG NO_2 report (AQEG, 2004).

4.4.4 Biogenic emissions

271. The NAEI also provides estimates of emissions of the precursor pollutants from natural (biogenic) sources. In the UK, these mainly relate to NMVOC and NH_3 emissions, although forest fires and accidental fires can also be sources of NO_X. NMVOCs – in particular isoprene and monoterpenes – are emitted from several natural and agricultural sources such as forests. The emission rate depends on factors such as temperature, light intensity and relative humidity: it, therefore, displays seasonal and interannual variability and may increase substantially during hot summertime weather. Simpson *et al*. (1999) have estimated the total NMVOC emissions in the UK from forests, heathland, pastures and crops to be 178 kt for an average year, but this would be expected to vary with long-term meteorological trends and changes in land use. The NAEI currently adopts this figure, which it is assumed remains unchanged each year and is currently around 13% of anthropogenic UK sources of NMVOCs in 2001. This estimate is currently being reviewed.

272. Emissions of NH_3 occur naturally from humans and domestic and wild animals. The NAEI includes estimates from domestic and wild animals, human breath and sweat and nappies. These amounted to around 22 kt of NH_3 emitted in 2001, around 7.5% of anthropogenic sources including agriculture.

273. Emissions of NO_X occurring from forest and accidental fires are not currently estimated.

Table 4.12 Comparison of 2001 PM_{10} emissions by sector between the NAEI and the LAEI.

Sector	NAEI (t y^{-1})	LAEI (t y^{-1})	LAEI/NAEI (%)
Road transport exhaust	—	—	—
Major roads	—	—	—
Motorcycles	54	71	118
Cars	403	506	122
Taxis	—	97	—
Cars and taxis (combined)	403	603	150
Buses and coaches	116	51	44
LGVs	399	551	138
Rigid HGVs	306	355	116
Articulated HGVs	385	336	87
Minor roads	1104	189	17
Total road transport exhaust	2771	2157	78
Domestic	913	596	65
Other transport	342	204	60
Combustion in domestic, commercial, industry and agriculture	2024	1540	76
Brake and tyre wear	427	0	0
Construction	579	0	0
Total	6911	4517	65

4.4.5 Local emissions inventories

274. The general principles of inventory compilation for local inventories were considered in the AQEG NO_2 report (AQEG, 2004). Local inventories tend to cover specific urban conurbations compiled mostly by the London Research Centre. Particular attention was focussed on the LAEI, in part because London represents the largest urban conurbation in the UK, but also because there is significant dispersion modelling being undertaken for London. It is important, therefore, to consider the differences between the NAEI and LAEI since both have been used for predictions of current and future air quality concentrations and for assessments of source apportionment. Although there were differences highlighted in total NO_X emissions between the NAEI and the LAEI in the AQEG NO_2 report, there was broad agreement by sector. The LAEI typically estimated emissions of NO_X to be less than the NAEI.

275. The situation for particles is, however, more complex for many reasons, including the difficulty in estimating emission rates for some sectors highlighted in Section 4.2. There are currently two releases of the LAEI for 1999 and 2001.

276. The principal difference between the two inventories relates to the activity data used to derive total emissions; identical emission factors have been used for the majority of source sectors. Considering the 1999 LAEI, there are some important differences compared with the NAEI for particles. The most important of these was the assumption in the LAEI that particle emissions from natural gas combustion are negligible. Furthermore, the LAEI did not consider non-exhaust particles from road transport, including those from tyre and brake wear and particle resuspension. These are considered to be significant sources in the NAEI. Nevertheless, non-exhaust particles from road transport were not estimated in the LAEI 2001 version (LAEI, 2003). The Greater London Authority (GLA) is currently reviewing the available information on these sources with a view to incorporating them into the LAEI. Estimates for some sectors, for example, natural gas, were revised in the 2001 release of the LAEI. However, a recent review of PM_{10} emissions from natural gas combustion suggests that emissions should be 25% of current (2001) estimates (see Section 4.2.4.2). Referring to Table 4.12, this revision would reduce the contribution made by natural gas combustion to total London PM_{10} emissions in the LAEI from 20.1% to 5.9%. It is difficult to make a precise comparison between the NAEI and LAEI because of the different source sector definitions used. A summary of the emissions comparison for PM_{10} for 2001 is shown in the Table 4.12.

277. Beyond the differences outlined above, there are other differences that could have a significant bearing on the comparison between the NAEI and LAEI. Some of these differences are outlined below.

- The LAEI and NAEI use the same emissions factors, suggesting that the principal differences between the two inventories are related to emissions activity data.

- The LAEI calculates emissions from the major road network based on the measured hourly distribution of vehicle types.

- The LAEI uses detailed information on vehicle speeds (the ‘moving-car-observer’ method, where an instrumented car records vehicles speeds on a second-by-second basis on Londons major roads). Different vehicle speed estimates could account for significant differences between the two inventories.

- The LAEI uses London-specific vehicle age and technology information for important vehicle classes such as taxis and TfL buses. In the 2001 LAEI, 60%

of buses were assumed to have particle traps fitted, effectively reducing emissions by 95%. This assumption probably accounts for the differences in PM_{10} emissions between the two inventories shown in Table 4.12.

- Analysis of the vehicle activity and emissions totals shows that the LAEI and NAEI are similar in terms of their estimates of emissions on the major road network, that is, in terms of mass of emissions per unit distance travelled. Furthermore, the total vehicle kilometre estimates within the Greater London area are similar: 29.2 (LAEI) versus 30.9 (NAEI) billion vehicle kilometres. There are, however, differences in the treatment of minor roads. A larger proportion of minor roads are assumed in the NAEI compared to the LAEI since more roads are treated explicitly in the LAEI. Furthermore, the LAEI assumes a different vehicle stock for minor roads and vehicle speed, for example, significantly fewer HGVs and buses. It appears that the latter point accounts for the most important differences between the two inventories for km2 PM_{10} total emissions.
- There are differences in methodology and emission factors used for the calculation of emissions from rail and shipping.

278. Emissions of PM_{10} vary across London because of the patterns of use of different vehicle types in London. There are relatively higher numbers of buses and taxis in central London and this has an important effect on the sources of PM_{10} in this part of the city. Figure 4.8 shows that in 2001, taxis were responsible for an estimated 25% of total road transport emissions in central London, whereas buses were responsible for around 5%. The contribution from LGVs to PM_{10} emissions is important across London. The comparatively low figure for buses is the result of 60% of the bus fleet having particle filters fitted, which were assumed to reduce PM_{10} by 95%. It also should be noted that emissions from the newest buses in Londons fleet are lower still, however, and that this is not yet reflected in the NAEI. This bus fleet in London is newer and lower-emitting than the national bus fleet. This is accounted for in the NAEI, leading to a lower emission rate for buses in London than is apparent for urban areas in the UK as a whole.

279. Figure 4.9 shows the relative contributions of NO_X by area of London. Because particle filters have a very small effect on emissions of NO_X, buses contribute to a relatively higher proportion of NO_X emissions (19%) in central London. These spatial variations in emissions are an important consideration for source apportionment studies.

4.4.6 Changes in the 2002 NAEI

280. The latest version of the inventory (the 2002 NAEI) incorporates some improvements to the methodologies and modifications to activity data or emission factors for certain sectors. These changes have led to revisions in the inventory estimates for PM_{10} emissions produced in the 2001 NAEI, the version used for most of the air quality modelling and assessments. Overall, these have little effect on estimates of total UK emissions of PM_{10}, but the contributions from certain sectors have changed somewhat. The most important changes are summarised as follows.

Figure 4.8 Relative emissions of PM_{10} by area of London (LAEI, 2003).

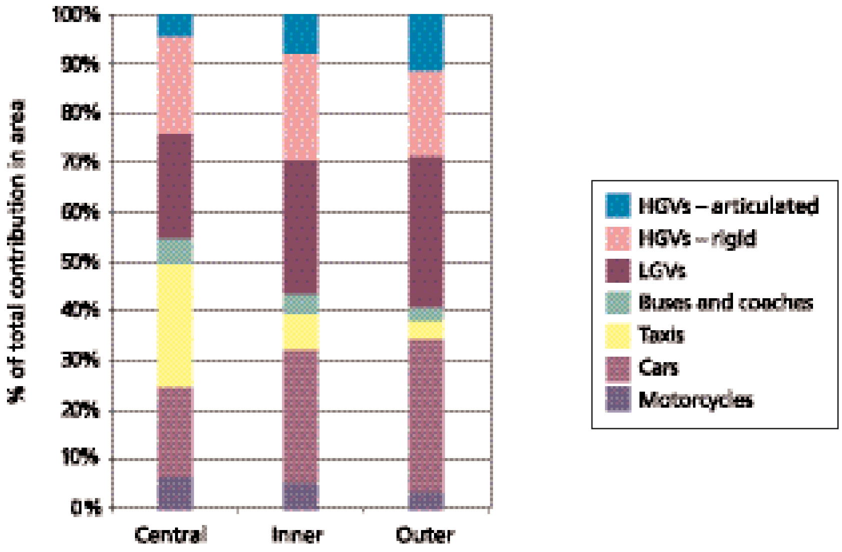
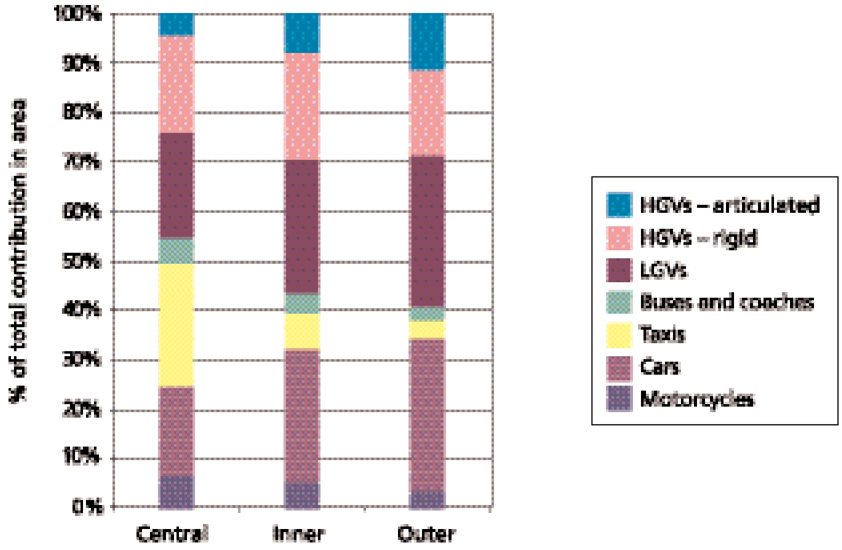


Figure 4.9 Relative emissions of NO_X by area of London (source: LAEI, 2003).



- Revisions to estimation methods for tyre and brake wear emissions. This was discussed in Section 4.2.2.11. The changes led to an increase in the estimate of 2001 emissions from 4.9 kt to 8.9 kt. The change is mainly due to changes in the estimate of tyre wear emissions.

- Revisions to the calculation of emissions from various industrial (mainly coal) combustion sources due to changes in the way emissions data from the Pollution Inventory are used in the NAEI and changes in some of the DTI data on fuel consumption by these sectors. These changes led to an overall reduction in PM emissions.

281. As mentioned in Section 4.2.4, in addition to having made these changes, the NAEI is currently carrying out a review of PM emission factors for a range of industrial processes, combustion, mining and quarrying and construction. The output from this review is expected to feed into the next version of the inventory to be compiled for release in 2005 (2003 NAEI).

4.4.7 Uncertainty in national emissions estimates

4.4.7.1 Quantification of uncertainties in emission estimates for sources covered in the inventory

282. Quantifying the uncertainties in the PM_{10} emission estimates is important if the trends in ambient concentrations and the contributions of different sources are to be understood. This also helps to prioritise future work by pointing to those areas where further information is required to improve the reliability of the inventory. The NAEI has made a detailed and quantified assessment of the uncertainties in the national emission estimates for PM_{10} and other pollutants in 2001 using a direct simulation approach, as described by Passant (2003) and AQEG (2004).

283. The procedure used corresponds to the IPCC Tier 2 approach discussed in the *Good Practice Guidance* (IPCC, 2000), as well as the Tier 2 method proposed in the draft *Good Practice Guidance for CLRTAP Emission Inventories*, produced for inclusion in the *EMEP/CORINAIR Guidebook on Emission Inventories*.

284. Uncertainties in the emission estimates arise from uncertainty in both the emission factor and activity rate used in the calculations. In some cases, emission estimates are supplied directly by process operators or regulators (for example, the Pollution Inventory) and an uncertainty is assessed on this. The main sources of uncertainty in emission inventory estimates are in the emission factors used. Uncertainty estimates for these are largely based on expert judgment, guided by the range in emission factors given by literature sources and elsewhere. Industrial trade associations and industrial process operators provide a final source of expert opinion. The report by Barlow *et al*. (2001) provided an indication of the range of measured emission factors for road vehicles. Table 4.13 gives a summary of the assumptions made for uncertainty for the major PM_{10} sources. The uncertainty factors are under review for some industry sources, e.g. iron and steel, where some industry contacts suggest the uncertainty factors are lower than shown in the table. It should be noted, however, that the uncertainty factors cover all parts of the given industry process.

285. The uncertainty associated with emission estimates for road and off-road transport, small industrial processes and smaller combustion processes is assumed to be greater than the uncertainty for emission estimates for large combustion plants and large industrial processes such as cement production. This is because the emission estimates for the former are all based on literature-based emission factors, whereas the emission estimates for the latter are all taken from estimates for individual plants, provided by process operators and at least some are based on measurements of emissions.

286. The uncertainty analysis gives an overall uncertainty in the annual total for UK PM_{10} emissions in the 2001 version of the NAEI of –19% to +46% at the 95% CI for the mean value emission figure of 191 kt (that is, 155–279 kt). This range is considerably wider than for SO_2 (±3%) and NO_X and NMVOCs (both ±8%). It should be noted that the uncertainty estimates quoted here only refer to uncertainty in the national annual emission totals, not to emissions from particular sources. Neither do they reflect uncertainties in emissions at a particular time or location. The result may appear lower than expected and is a result of a statistical

Table 4.13 Parameters used in the analysis of uncertainty in the PM_{10} inventory for 2001.

Source	**Uncertainty expressed as limits relative to the mean value**	
	Emission factor	**Activity data**
Combustion:		
Coal and fuel oil-fired power stations	±20%	NA
Other coal combustion	30–300%	±1%
Refinery combustion of fuel oil	±40%	±4%
Domestic wood	50–200%	±30%
Domestic anthracite	30–300%	±1%
Domestic SSF	30–300%	±3%
Natural gas (all sectors)	30–300%	±2%
Agricultural use of straw as fuel	30–300%	±50%
Industrial processes:		
Quarrying	10–1000%	NA
Construction	10–1000%	NA
Non-Fletton bricks	20–500%	±30%
Steel production processes	30–300%[a]	±1%
Chipboard manufacture	20–500%	±30%
Part B processes	20–500%	NA
Cement production	±40%	NA
Road transport:		
Petrol cars/vans	0–100%	±10%
Diesel cars – cold start	±60%	±40%
Diesel vans – cold start	±60%	±60%
Diesel cars/vans – other	±40%	±10%
HGV/buses/coaches	±60%	±5%
Brake wear	10–1000%	±10%
Other transport:		
Off-road vehicles	50 – 200%	±80%
Waste disposal:		
Foot and mouth pyres	50 – 200%	±5%
Agriculture:		
Arable farming	±80%	NA
Livestock housing	±50%	NA

Key: NA, not applicable because emission estimate is taken directly from the Pollution Inventory or from other sources. [a]This covers all parts of the production process. Industry contacts suggest this range is too high: this is under review.

interpretation of roughly similar contributions from several different emission sources each with moderately high uncertainties. It is worth noting that IIASA concluded a very similar level of uncertainty to the UK's figures in their independently derived estimate of UK emissions of NO_X using a similar statistical approach (Suutari *et al*., 2001).

287. A key source analysis was also undertaken following IPCC Tier 2 methodology to identify the major contributors to inventory uncertainty (Passant *et al*., 2002). Key sources will often, but not always, be those sources that contribute most to national emissions. For PM_{10}, a number of key sources were identified. These were:

- quarry processes;
- domestic combustion of coal, wood and anthracite;
- construction processes;
- coal-fired power stations and autogenerators;
- road vehicle brake wear;
- gas oil use in off-road vehicles and machinery;
- industrial burning of coal;
- domestic gas combustion;
- livestock housing; and
- Part B processes (vehicle respraying, timber processes and so on).

288. The results of the uncertainty analysis apply to estimates for the year 2001 and the 2001 version of the NAEI only. One issue that should be considered is that since emissions are believed to be declining from many sectors as a result of legislation, it is vital that the emission factors and other data used in the NAEI are updated regularly to reflect any reductions in emissions that are occurring. This will require continuous and detailed research to be carried out to provide evidence for any reductions, thereby allowing the emission factors used for the annual NAEI emission estimates to continue being up-to-date. A case in point is the road transport emission factors.

4.4.7.2 Sources omitted in the inventory

289. The statistical method used in the uncertainty analysis is useful for investigating the likely impact of uncertainties in emission estimates. However, it is not helpful for identifying missing sources in the inventory and the impact this has on uncertainties. A study of uncertainty in the NAEI identified a number of potential sources of PM emissions that have been previously omitted from the NAEI, although some of these sources have been included in the current 2002 version of the NAEI (Passant, 2003). Emissions from the omitted sources were not expected to be important, it being considered unlikely that their inclusion would increase the UK estimate of PM_{10} emissions by more than a few per cent. They could be more important at particular times at a local level. The sources still omitted from the NAEI are:

- flaring of waste gases at oil refineries and at chemical works;

- use of chemical process wastes as fuels;
- use of natural gas as a fuel by railway operators and processes for the manufacture of ammonia;
- use of coke by the iron and steel industry but excluding coke use in coke ovens, sinter plant and blast furnaces;
- fuel use by military aircraft;
- natural fires, for example, forest fires, and accidental fires, such as house fires and car fires;
- small-scale burning of waste including garden and other bonfires; and
- burning of wastes produced during well testing on offshore oil and gas installations.

290. In some cases insufficient data are currently available to enable emissions to be estimated. In other cases (including the third, fourth and fifth items in the above list) it will be possible to include emission estimates in future versions of the NAEI. Of the remaining sources, the priorities for estimation of emissions should be natural and accidental fires and small-scale burning of waste. It is recommended that research be done to collect the data necessary for estimating emissions from these sources.

291. Uncertainty assessments were also undertaken for emissions of black smoke, which were more uncertain than PM_{10} emissions (+76 to –50 %). Also of relevance is the uncertainty of important particle precursor emissions.
The uncertainty in the UK totals are:

NO_X ±7%, SO_2 +4 to 3% and NH_3 +19 to –18%.

Overall there is more uncertainty associated with primary particle emissions compared with the particle precursor emissions.

4.4.7.3 Local emissions in critical areas and ratio to NO_X

292. The limit values and objectives place specific emphasis on the characterisation of local emissions at hotspots, in particular busy roads. However, due to the uncertainties in traffic flow characteristics, and the potential contribution from road traffic resuspension, quantification of emissions at critical urban locations such as these can give rise to substantial problems.

293. It is useful to carry out a comparison of PM_{10}:NO_X ratios for both emissions and concentrations, and this has been carried out for the Marylebone Road (kerbside) and London North Kensington (background) sites for 2001. Table 4.14 overleaf summarises the NO_X and PM_{10} concentrations at each site.

Table 4.14 Concentrations of NO_X and PM_{10} at Marylebone Road and Kensington (2001).

	NO_X (µg m^{-3})	PM_{10} (µg m^{-3}, TEOM)
Marylebone Road	337	33
North Kensington	75	20
Increment	262	13

294. The PM_{10}:NO_X ratio of the increment in concentrations is 0.05 (mass basis), which can be used to represents an estimate of the ratio of road traffic emissions from Marylebone Road. Detailed emissions calculations from the LAEI suggest a ratio of 0.063 (mass basis) for 2001. The LAEI emissions estimate includes only exhaust emissions, and does not include emissions from tyre and brake wear or from resuspension.

295. These values can be used in a dispersion model to predict the PM_{10} concentration that is due to vehicle emissions alone, assuming that the model is able to produce accurate estimates of NO_X. On this basis, a dispersion model would predict 262*0.063 = 16.5 µg m^{-3} PM_{10}. This 16.5 µg m^{-3} can then be added to the background concentration of 20 µg m^{-3} (measured at North Kensington) to yield a total of 36.5 µg m^{-3}.

296. The predicted PM_{10} concentration of 36.5 µg m^{-3} is greater than the measured concentration of 33 µg m^{-3} but does not include the additional contributions from tyre and brake wear and resuspension. Inclusion of these other sources would lead to a predicted concentration significantly higher than the measured value. On this basis that there appears to be an important discrepancy between PM_{10}:NO_X ratios based on emissions data and ambient concentrations at this location. Further analysis of other sites is required to determine whether there is a similar conclusion, and if so what is the explanation.

4.5 Time series of emissions

4.5.1 UK emissions

4.5.1.1 Stationary fuel combustion

297. The decline in UK emissions of PM_{10} with time shown in Figure 4.10 is primarily due to the reduction in coal use and the more extensive use of natural gas. In Figure 4.10, the category 'Other' includes emissions from off-road transport, waste treatment and disposal as well as emissions from agriculture. Emission estimates for the resuspension of dust from roads are not typically included in officially reported UK emissions (as the source is not a primary source). However, for completeness, the emission estimates from this source are included. Domestic emissions (a subset of commercial, residential and institutional combustion) have fallen from 222 kt in 1970 (41% of the total emission) to 33 kt in 2001 (18% of the total). Similar reductions were observed from the public power and industrial combustion sectors. In the UK, fuel switching from coal to gas has been driven by the Clean Air Acts and price differentials. However, domestic coal can still be a significant source in some smaller towns and villages, in Northern Ireland and in areas associated with the coal industry.

298. There has also been a drive to control particulate emissions from large industrial sources. So in addition to fuel switching, reductions can also be attributed to the installation of abatement equipment (and to a lesser extent increases in plant efficiencies). There are a number of less well-pronounced features evident in Figure 4.10. Emissions in 1984 were significantly lower than previous years due to the miners' strike. Considerably less coal was used for both electricity generation and domestic heating, which was reflected by a marked decrease in PM_{10} emissions. For a number of other years, elevated emissions from the domestic sector can be attributed to particularly cold winters.

4.5.1.2 Production processes

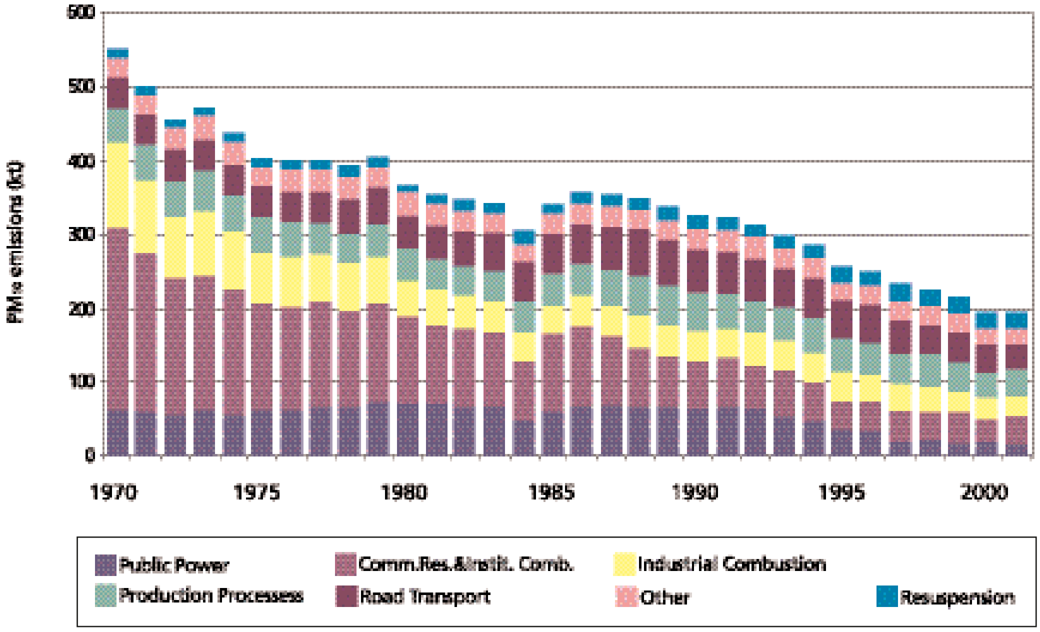
299. These include industrial sources not associated with combustion, that is, the production of metals, cement, lime, coke and chemicals as well as bulk handling of dusty materials, construction, mining and quarrying. Emissions from these sources are difficult to quantify due to the contribution of fugitive emissions. This also makes it inherently difficult to abate emissions and helps to explain why there has been little decrease with time during 1970–2001.

4.5.1.3 Road transport

300. Emissions from road transport (exhaust and tyre and brake wear) decreased by 38% between 1990 and 2001 in spite of a continuous increase in traffic over this period. The decrease is mainly due to the penetration of petrol cars fitted with catalysts in the fleet, the phasing out of leaded petrol and the fleet penetration of newer heavy duty diesel vehicles meeting tighter emission standards. Although emissions of PM from petrol cars are small compared with diesel vehicles, the large number of petrol cars without catalysts in the fleet in the late 1980s meant they made a significant contribution to the PM inventory. Their contribution has declined as lower PM-emitting cars with catalytic converters penetrated the fleet in the 1990s, replacing non-catalyst cars. Furthermore, PM emissions from non-catalyst cars running on leaded petrol are estimated to be around a factor of three times higher than emissions from non-catalyst cars running on unleaded petrol. So the gradual phasing out of leaded petrol during the 1990s helps further explain the trends in emissions from petrol cars. The downward trend in road transport emissions of PM has been tempered by the fact that emissions from diesel vans and cars have been increasing over this period. Although European emission standards have been reducing PM emissions from these vehicles over the 1990s, this has been offset by a large growth in diesel car and van activity between 1990 and 2001. The increase in emissions appears to be reaching a peak now as the impact of tighter Euro standards starts to dominate.

301. The time series of UK PM_{10} emissions is shown in Figure 4.10 split by source sector.

Figure 4.10 Time series of UK PM_{10} emissions.



4.5.2 Emission trends by particle size: the 2001 NAEI time series

302. Typically, particles arising from sources such as quarrying and other processes are the coarsest. The combustion of solid fuels also results in relatively coarse particles, compared to the fine particulate from gas combustion. As a result, these sectors give rise to relatively large emissions for PM_{10}. For the finer particulate emissions, road transport becomes the dominant source. Emissions from gas combustion give rise to fine particulate material; however, the small magnitude of particulate emissions gives rise to relatively lower emissions from these sources.

303. The uncertainties in the emission estimates and the time series of the finer particle size ranges must be considered to be high and more uncertain than the estimates for PM_{10}. As discussed in Section 4.4.2, it is not clear how appropriate the size profiles used in the inventory are for the UK and how they are effected by abatement. Due to the lack of information, the inventory assumes that the size fractions for each source remain unchanged with time, implying that abatement has no effect on the particle size distribution. This may not be the case, as many particulate emission controls for stationary sources may be more effective for the coarser particles than the finer ones ($<PM_1$). This means that applying constant particle size fractions to the inventory for PM_{10} emissions from each source could lead to underestimations of the finer particle size emissions. A critique of the NAEI PM size fractions is currently being prepared. Figure 4.11 indicates the emissions for different particulate size fractions by sector, based on the 2001 NAEI.

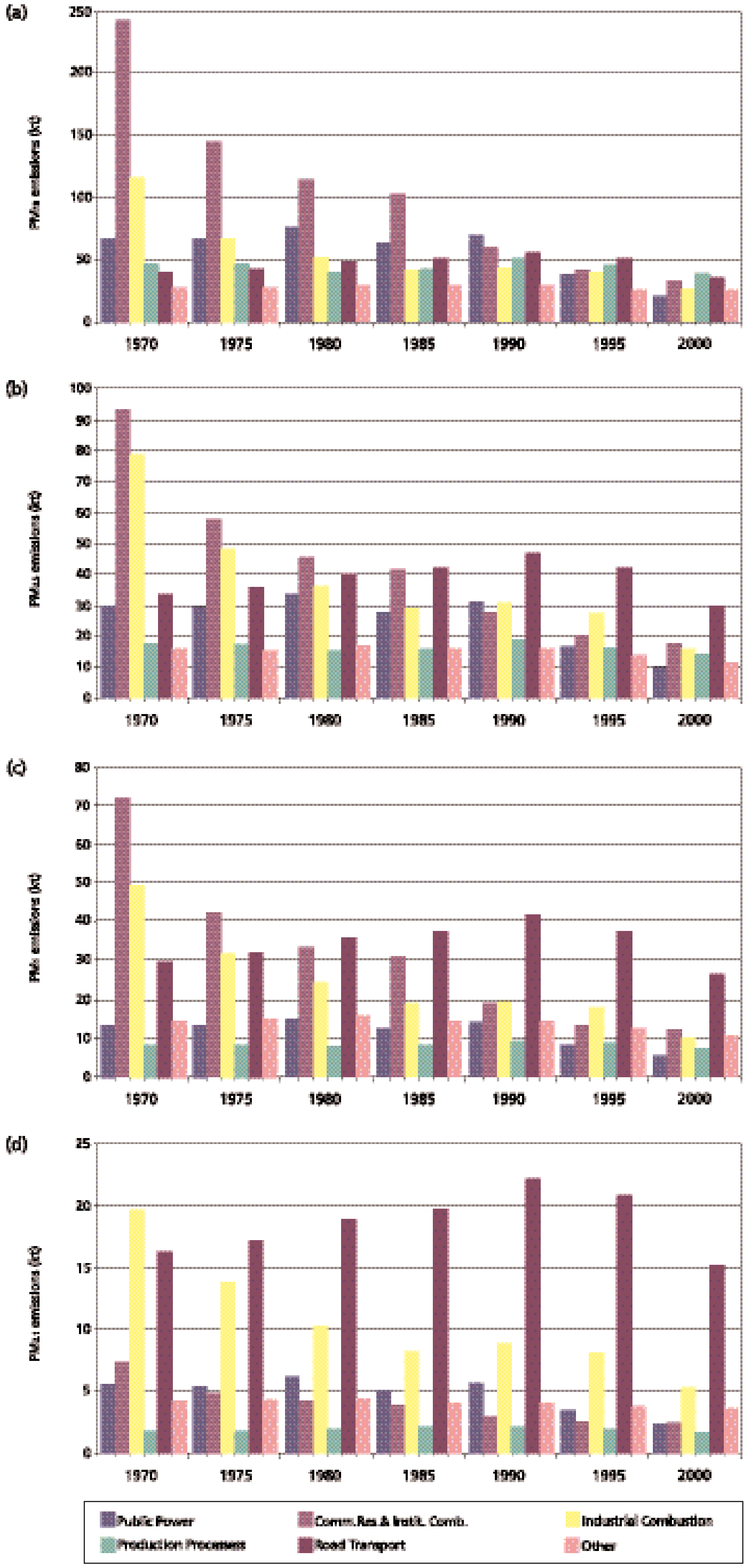
4.5.3 Emissions by fuel type

304. This section highlights the emissions of PM through the combustion of major fuel types and particle size (PM_{10} and $PM_{0.1}$). Two major source sectors are considered: public power and commercial, residential and institutional combustion.

4.5.3.1 Public power

305. Historically, emissions of PM have been dominated by emissions from coal, as shown in Figure 4.12. Despite the large reduction in emissions from coal use,

Figure 4.11 Time trends in PM emissions for (a) PM$_{10}$, (b) PM$_{2.5}$, (c) PM$_1$ and (d) PM$_{0.1}$ (source: NAEI, 2001).



these emissions are still large compared to those from other fuels. For the smaller particle sizes there is less impact from coal combustion and an elevated impact from gas combustion.

4.5.3.2 Commercial, residential and institutional combustion

306. Emissions from this sector follow a similar pattern to power generation, that is, the sector is dominated by coal combustion, as shown in Figure 4.13. However, the combustion of gaseous fuels is more important for fine particle emissions compared with the power generation sector. In this sector there is a relatively high contribution from wood, straw and MSW. Emissions from the combustion of wood are more important to total emissions of PM_{10}, whereas the combustion of straw is particularly important at $PM_{0.1}$. Emissions from MSW are small.

4.5.4 Trends in PM_{10}–$PM_{2.5}$ emissions

307. Figure 4.14 shows the trends in PM_{10}–$PM_{2.5}$ from 1990 to 2001 for total UK emissions by source sector. The most significant reductions have been for the energy production and industrial combustion sectors, which have seen reductions of 75% and 46% in emissions between 1990 and 2001, mostly as a result of a reduction in the use of coal in these sectors. Overall the reduction in PM_{10} $PM_{2.5}$ has been 44%.

Figure 4.12 Time series trends in PM emissions from public power for (a) PM_{10} and (b) $PM_{0.1}$.

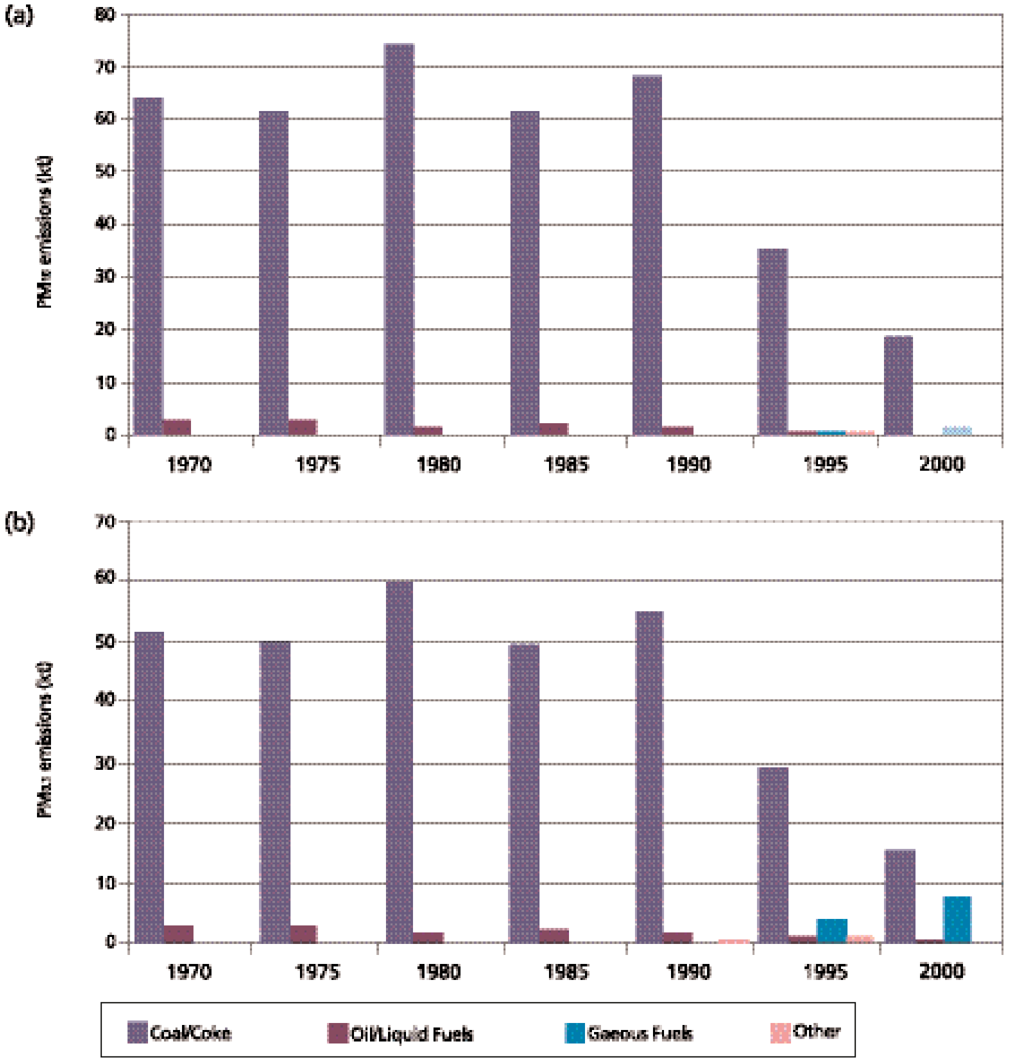


Figure 4.13 Time series trends in PM emissions from commercial, residential and institutional combustion for (a) PM_{10} and (b) $PM_{0.1}$.

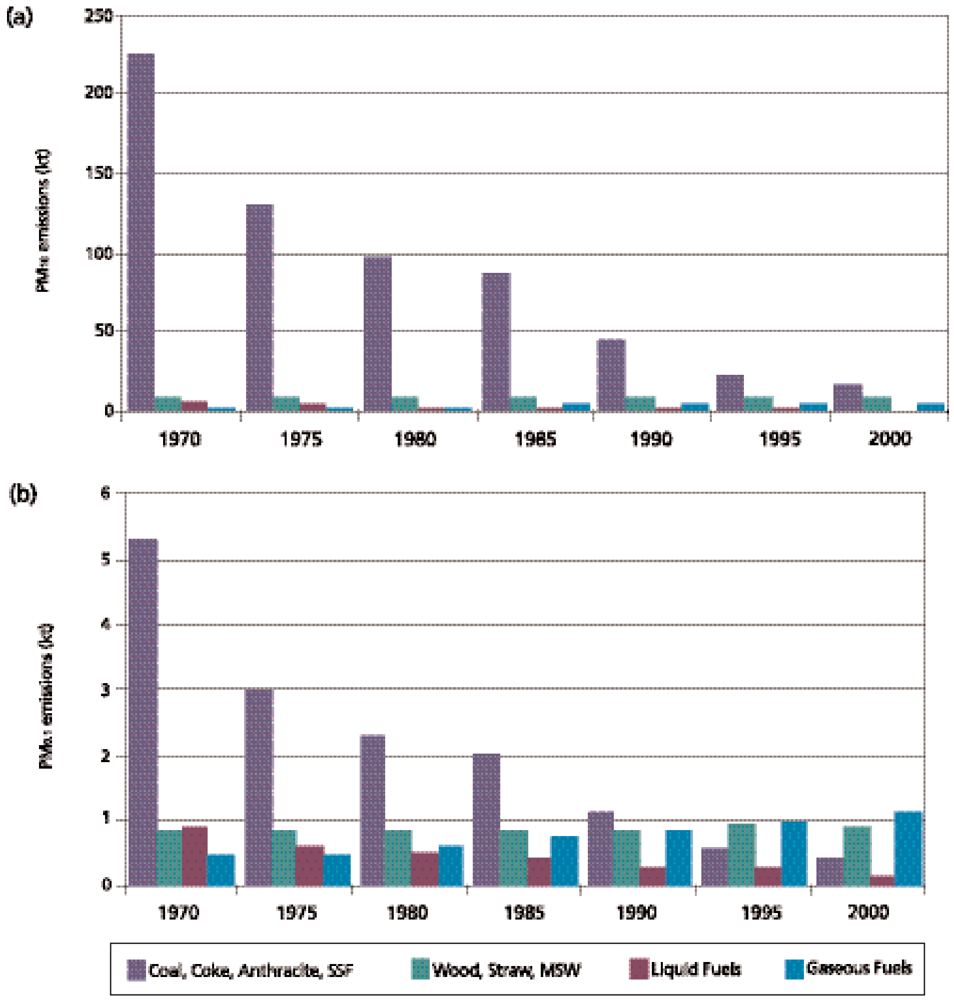
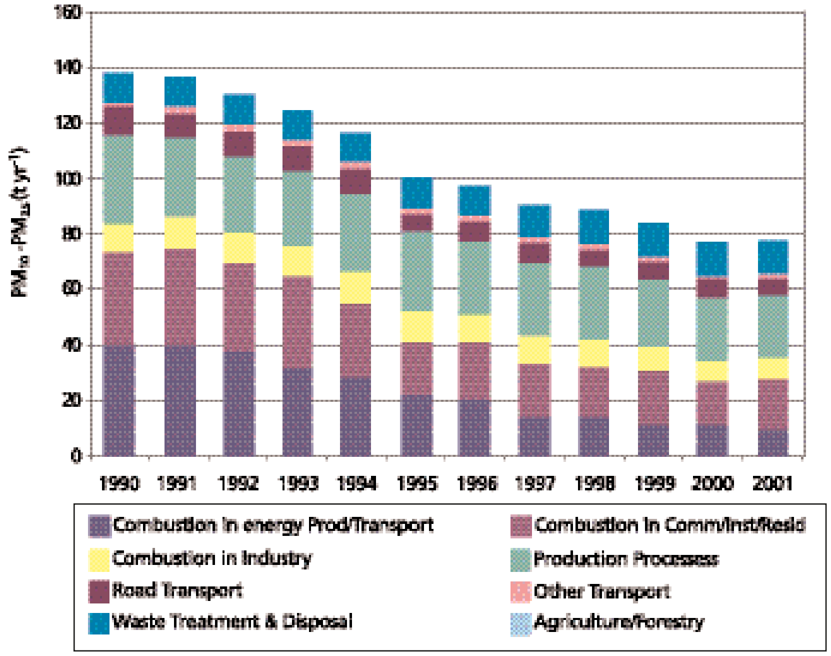


Figure 4.14 Trends in PM_{10}–$PM_{2.5}$ from 1990 to 2001.



4.5.5 Trends in UK precursor emissions

308. This section discusses the temporal trends in UK emissions of the precursor pollutants to PM in the atmosphere, as indicated by the 2001 NAEI.

4.5.5.1 Temporal trends of NO_X emissions

309. Since 1990 there has been a 39% reduction in total NO_X emissions; however, this decrease in emissions has not been constant (Figure 4.15). Up to 1984 the NO_X emission profile was relatively flat with small peaks in 1973 and 1979, as seen previously for CO_2; these peaks were due largely to the cold winters during those years. However, from 1984, emissions rose markedly as a result of the growth in road traffic reaching a peak in 1989. Since 1989, total emissions have declined by 40% as a result of a 51% reduction from power stations and 40% decrease from road transport. The decrease in road transport emissions since 1990 has been due mainly to the introduction of catalytic converters on petrol cars and stricter regulations on emissions from diesel vehicles.

310. Emissions from power stations were relatively constant in the 1970s, but have been declining since 1979. In the 1980s this was due to the increased use of nuclear power and an increase in the average efficiency of thermal power stations. However, since 1988, the electricity generators have adopted a programme of progressively fitting low-NO_X burners to their large coal-fired stations. The introduction of modern combined cycle gas turbine (CCGT) plants burning natural gas in the early 1990s and the decreased share of coal for power generation in favour of gas further reduced NO_X emissions from this sector up until 1999. Emissions of low NO_X gas turbines are lower than those of pulverised coal-fired plants, even when these are fitted with low-NO_X burners. However, this trend was broken in 2000 when output from coal-fired stations grew again at the expense of nuclear power production. Further growth in electricity output from coal-fired stations occurred in 2001, this time at the expense of gas, possibly due to the increased price of natural gas supplies. The trend in NO_X emissions has been discussed in detail (AQEG, 2004). Emissions from industry have declined by 26% since 1990, again mainly due to the decline in coal use in favour of gas.

Figure 4.15 Trend in UK total NO_X emissions by sector.

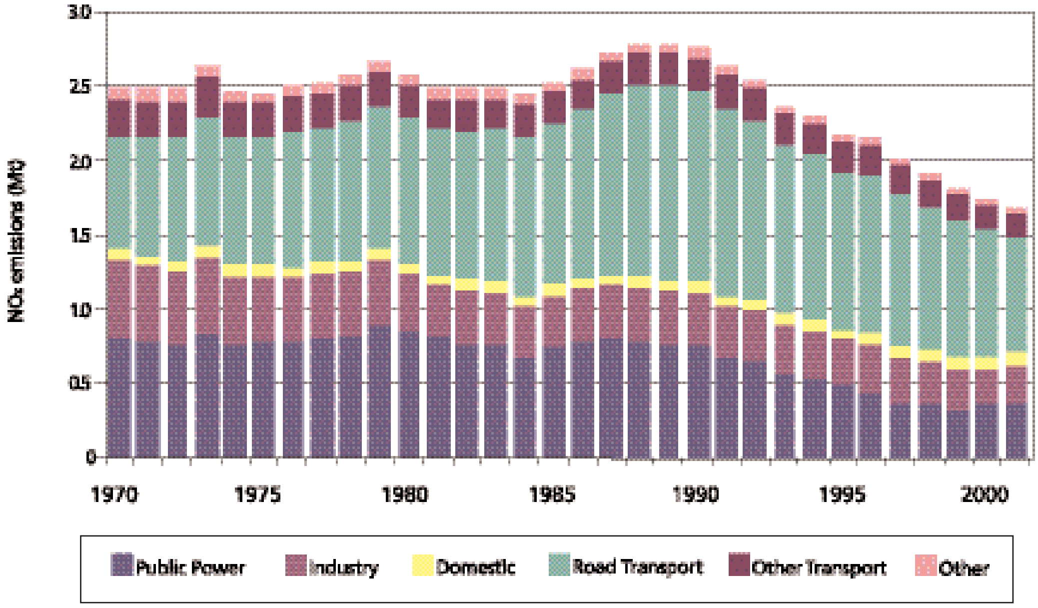
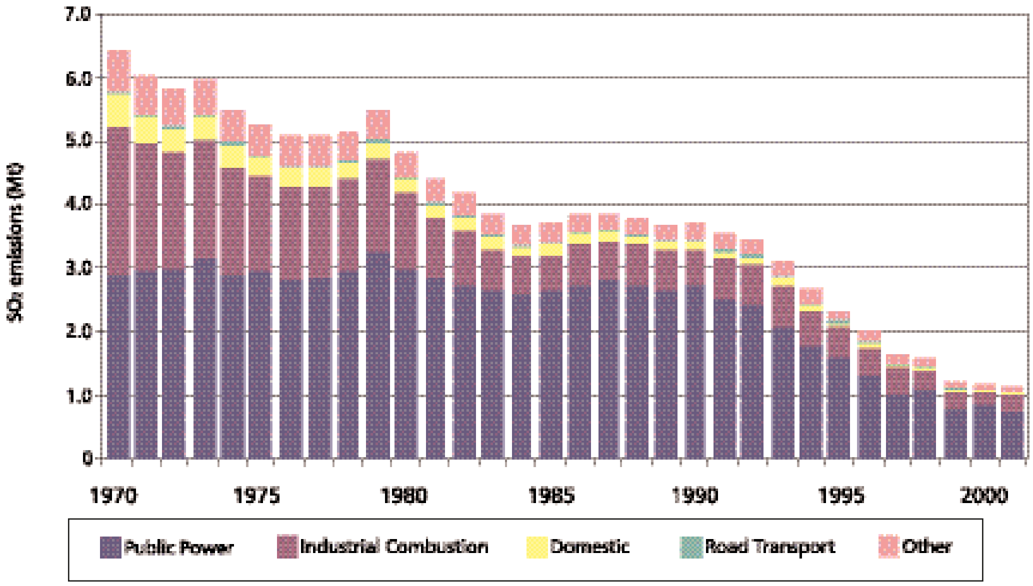


Figure 4.16 Trend in UK total SO_2 emissions by sector.



4.5.5.2 *Temporal trends of SO_2 emissions*

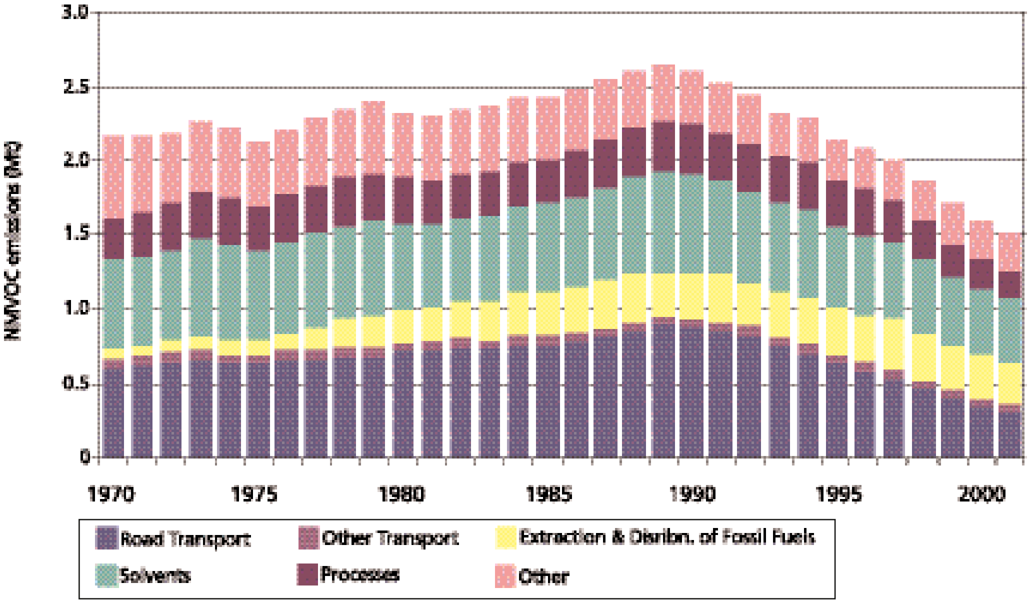
311. Since 1970 there has been a substantial overall reduction of >83% in SO_2 emissions (Figure 4.16). Emissions have been falling fairly steadily over this period. Power stations are the largest contributors to SO_2 emissions, accounting for 64% of UK emissions in 2001. However, emissions from power stations have fallen by 74% since 1970. During the 1980s, the decrease was due to the increased use of nuclear power at the expense of coal and an increase in average efficiency of thermal power stations. Since 1990, this decline has accelerated because of the increase use of CCGT stations and other gas-fired plants. CCGTs are more efficient than conventional coal and oil stations and have negligible SO_2 emissions. Most recently the FGD plants constructed at Drax and Ratcliffe power stations have had a significant effect on emissions.

312. SO_2 emissions from industry fell by 90% from 1970 to 2001. This was due to a decline in the use of coal and oil in favour of natural gas for combustion as well as a decline in the energy intensive iron and steel industry and other heavy industries. Emissions from domestic and other commercial/institutional sectors have declined substantially during the period 1970–2001, reflecting the major changes in fuel mix from oil and coal to gas.

4.5.5.3 *Temporal trends of NMVOC emissions*

313. Emissions of NMVOCs have fallen by 42% since 1990, when they were near their peak, as shown in Figure 4.17. The road transport sector has been mainly responsible for this fall. Emissions from road transport have decreased by 66% since 1990, mainly due to the penetration of cars fitted with three-way catalysts, which are very efficient at reducing NMVOC emissions from exhausts. The fitting of carbon canisters on cars and the decline in the volatility of summer fuel blends have reduced evaporative emissions from vehicles, also contributing to the fall in NMVOC emissions from road transport. Emissions from solvent use have decreased by 37% since 1990. This has been partly due to emission controls, technological changes and reduced manufacturing output in some sectors. Emissions from the domestic use of solvents have also declined due to a trend

Figure 4.17 Trend in UK total NMVOC emissions by sector.



towards formulating products such as paints and aerosols with lower solvent contents. Emissions from various production processes have fallen by 46% since 1990. This has been mainly due to reductions in emissions from petroleum refineries and the chemical industry. Emissions from food and drink manufacture have risen slightly over this period. Emissions from offshore oil and gas have increased since 1990 and are now contributing 11% of total NMVOC emissions.

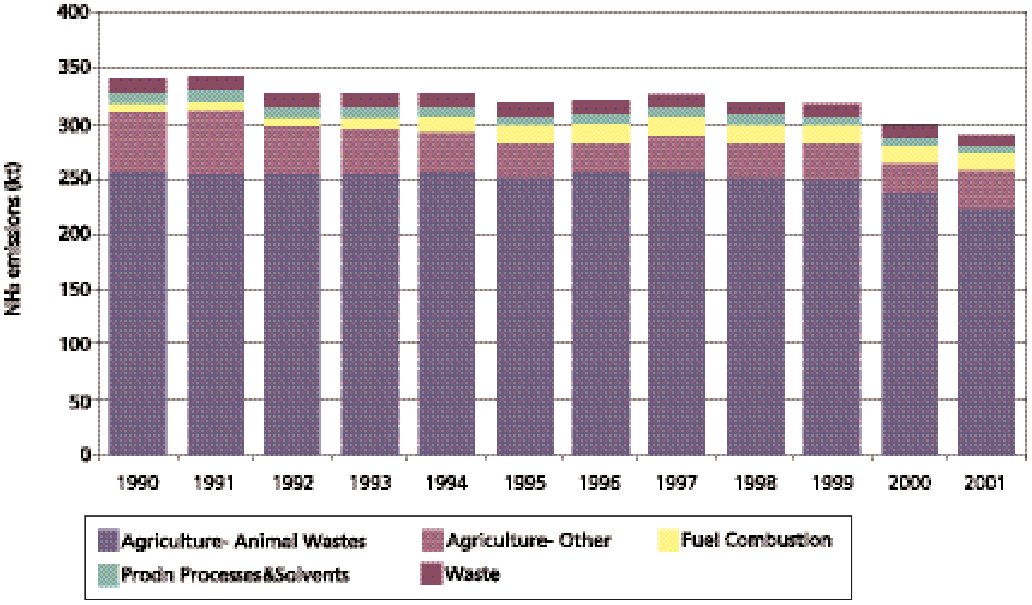
4.5.5.4 *Temporal trends of NH_3 emissions*

314. Ammonia emissions have fallen by 15% since 1990, as shown in Figure 4.18. Emissions are dominated by agricultural sources with emissions from livestock and their wastes comprising 77% of the total emission. These have been decreasing as animal numbers have been declining. During 2001 the foot and mouth outbreak had a significant impact on the animal livestock numbers. However, a degree of restocking occurred after the outbreak had finished. As a result, across an annual period, the impact of the foot and mouth outbreak does not stand out as being a particularly strong feature in the time series. The other agricultural sources contributing to NH_3 emissions include fertiliser use, crops and decomposition of agricultural vegetation. A decline in fertiliser use has led to a reduction in emissions from this sector. The non-agricultural sources are diverse and contribute 11% to the total emissions. However, emission estimates for these sources are very uncertain due to a lack of data. Emissions of ammonia from road transport, although relatively small, are increasing as a result of the increasing number of three-way catalysts in the vehicle fleet.

4.5.6 Trends in the PM_{10} to NO_X ratio

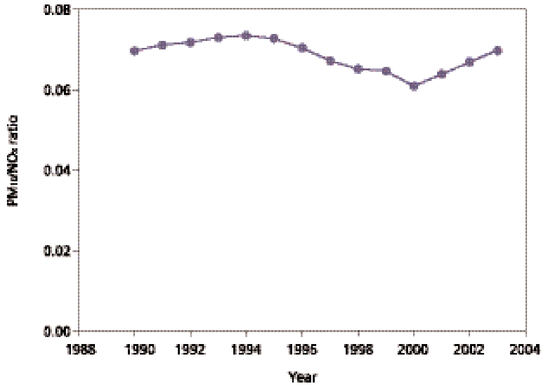
315. It can be useful to consider trends in the ratio of pollutants since they can provide some information on the validity of emissions inventories compared with atmospheric measurements. The ratio between PM_{10} and NO_X is important because this ratio is used in some receptor-based modelling approaches. Figure 4.19 shows the ratio of PM_{10}:NO_X since 1990 based on the NAEI 2002 methodology for calculating road transport emissions. The PM_{10} emissions include

Figure 4.18 Trend in UK total NH_3 emissions by sector.



exhaust emissions, tyre and brake wear, but no resuspension estimate. The 2002 NAEI has revised the estimates of particle emissions of tyre and brake wear and assumes that tyre wear is more important than previous inventory estimates. Figure 4.19 shows how the ratio declined from 1994 to 2000, but since 2000 has begun to increase again. The upward trend since 2000 mostly results from the increased purchase and use of diesel cars, which has kept urban PM_{10} emissions almost constant during 2000–2003, whereas emissions of NO_X have declined.

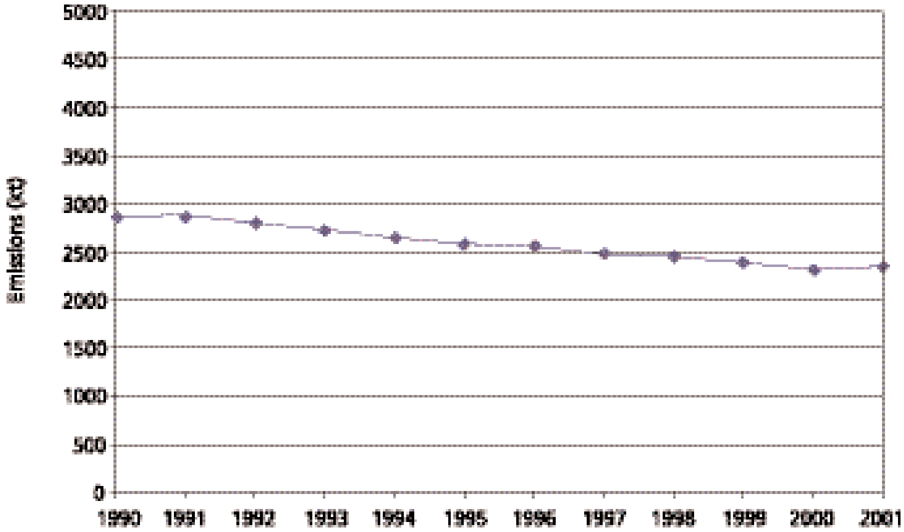
Figure 4.19 PM_{10}:NO_X emissions ratio for urban road transport emissions using the NAEI.



4.5.7 Emissions of primary particulates in Europe

316. An indication of European emissions and time trends for PM_{10} and particulate precursor species can be obtained from the national and sectoral emissions data officially reported by countries to UNECE/CLRTAP. A detailed analysis of European emission sources and time-series trends has also recently been prepared by the EC's CAFE Working Group on Particulate Matter (CAFE WG, 2003). The following analyses are based on the most recent data available, which were reported in 2003. Where countries did not report data, estimates of primary PM_{10} national totals were obtained from the Auto-Oil II programme (European Commission, 2000) for the EU15 countries (the 15 countries that were EU members prior

Figure 4.20 European PM_{10} emissions 1990–2001.



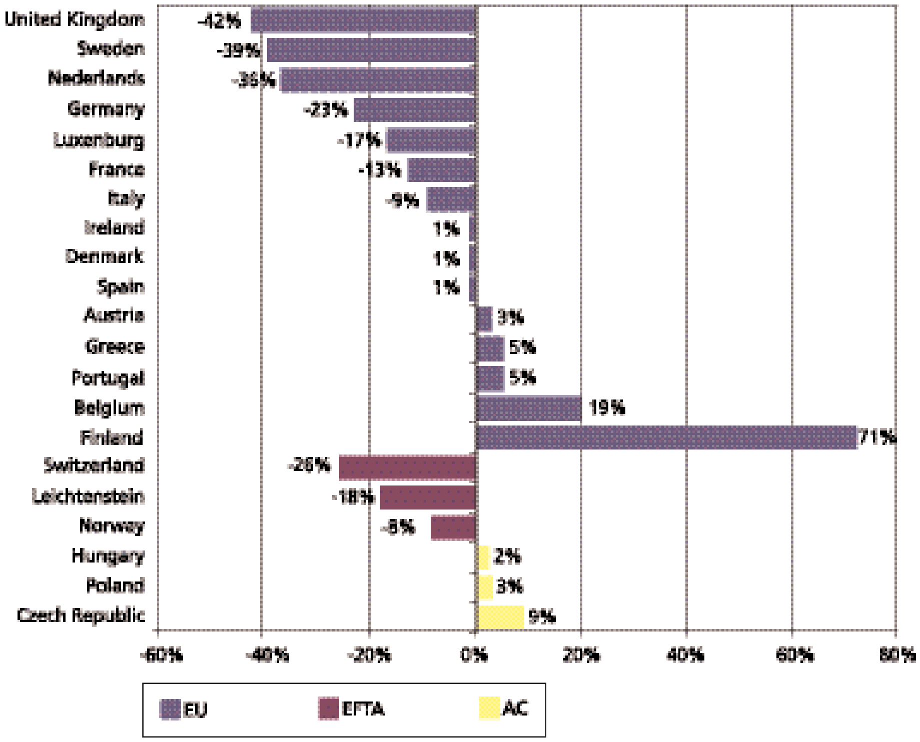
to the 2004 expansion) and from CEPMEIP (2001) for other countries. Gaps in reported time-series data have been filled using the methodology employed by the European Topic Centre on Air and Climate Change (ETC-ACC). Figure 4.20 illustrates the reduction in emissions of PM_{10} during 1990–2001 for the EU15 countries.

317. Emissions across all countries have decreased by 15% between 1990 and 2001. The vast majority of this decrease is primarily due to reductions that have occurred in the EU15, where emissions have decreased by 18% during this period. Emissions from the accession countries are around 21% of the EU15 emissions in 2001. However, this is based on an estimate for 1995 and the time series of emissions from the accession countries (and levels in 2001) is not certain due to lack of reported data. The EFTA4 countries (Liechtenstein, Norway and Switzerland only – Iceland did not report emissions) show only a relatively small level of emissions compared to the other country groupings. In 2001, emissions from this region accounted for 3% of the European total. Figure 4.21 shows the percentage change in PM_{10} emissions on a country-by-country basis. As the figure shows, most EU countries showed a significant decrease in primary PM_{10} emission reductions over the period 1990–2001.

318. The reasons why the Finnish emissions have increased steeply over this period are not currently known, and the emissions reported by Finland to UNECE/CLRTAP submission in 2003 are consistent with the values reported in previous years (such as 2002). In comparison with other European countries, the UK has shown the largest percentage decrease during this period (–42%) and, in absolute terms, the second largest emissions decrease among European countries (–131 kt), the first largest being Germany (–264 kt).

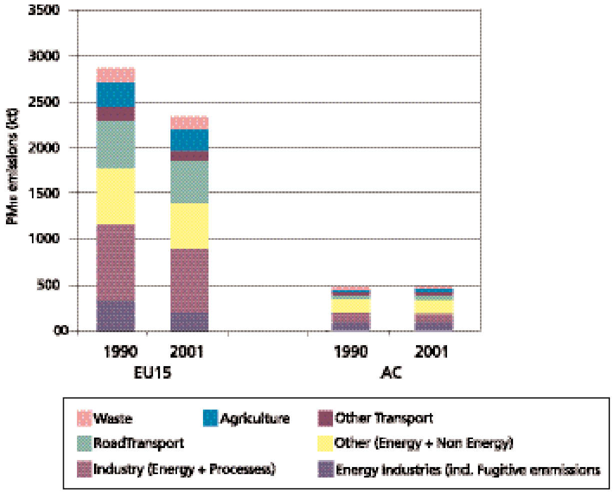
319. In the EU15, PM_{10} emissions are dominated by sectors that include road transport, industry and 'other' (for example, fuel combustion in the residential sector) (Figure 4.22). In the accession countries, the largest sectoral contribution of PM_{10} emissions comes from the 'other' (energy and non-energy) sector, with only a small contribution from road transport. This difference is probably due to the smaller vehicle fleets, the greater relative use of solid fuels (in comparison to natural gas in power generation) and the lower use of effective abatement technologies in the power generation sector.

Figure 4.21 Percentage change in PM_{10} emissions by country, 1990–2001.



320. Sectors contributing most to the reduction in PM_{10} emissions between 1990 and 2001 have been the energy industries and industry sectors and road transport (Figure 4.22). In energy industries fuel switching to natural gas and the use of and improvements to technologies such as electrostatic precipitators have contributed to the decrease in emissions observed. In the road transport sector, tighter PM_{10} emission regulations have been introduced in Europe for heavy duty vehicles and for cars.

Figure 4.22 Emissions of PM_{10} by sector, 1990 and 2001.



4.5.8 Emissions of PM precursor gases in Europe

321. Figure 4.23 illustrates the European (EU15, EFTA4 and AC10) emission trend between 1990 and 2001 for the particulate precursor species SO_2, NO_X, NH_3 and NMVOCs (AC10 refers to the ten accession states that joined the EU in 2004). The most significant reduction in emissions during this period was for SO_2, which was predominantly due to fuel switching from high sulphur solid and liquid fuels

Figure 4.23 Emissions of particulate precursor gases 1990–2001 in Europe (EU15, AC10 and EFTA4 countries).

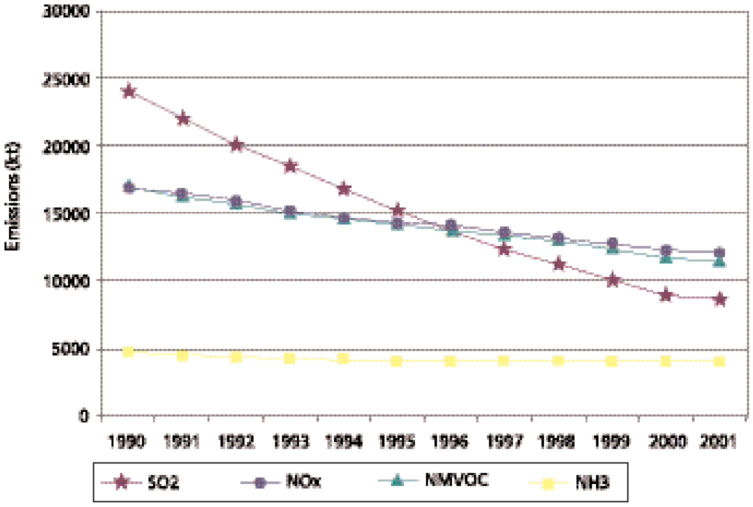


Figure 4.24 Emissions of NO_X by sector, 1990 and 2001.

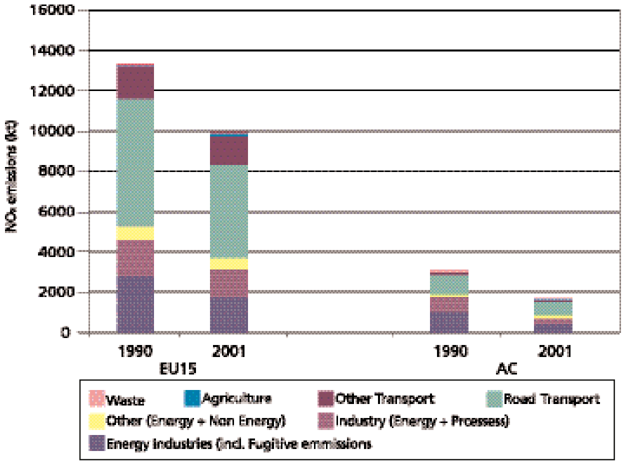


Figure 4.25 Emissions of SO_2 by sector, 1990 and 2001.

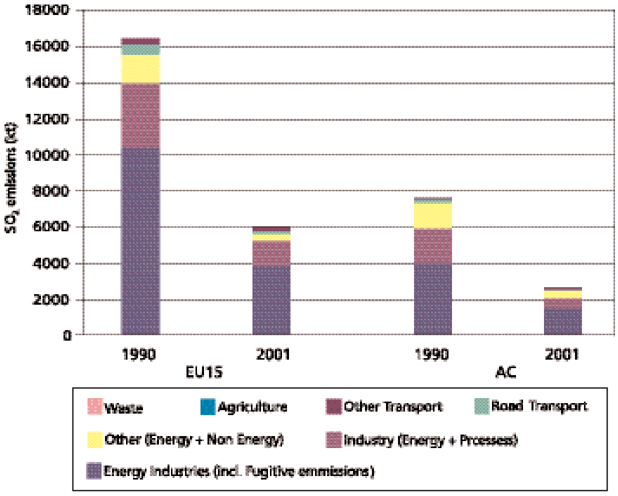


Figure 4.26 Emissions of NMVOC by sector, 1990 and 2001.

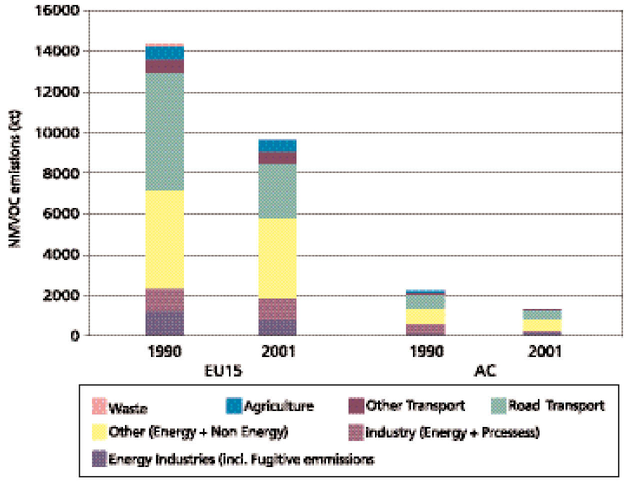
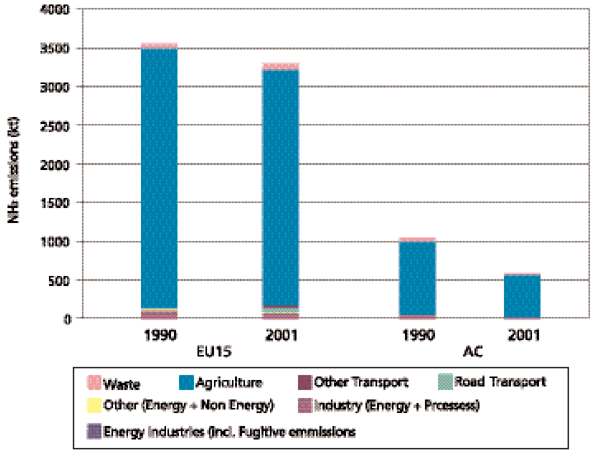


Figure 4.27 Emissions of NH_3 by sector, 1990 and 2001.



to natural gas in the energy industries and industrial and domestic sectors. The construction of new, more efficient power plants coupled with the use of low sulphur coal and FGD have also contributed to the lower emissions observed. The relative sector contribution to emissions for each of the respective particulate precursor species in 1990 and 2001 are shown in Figures 4.23 to 4.26.

322. To allow UK emissions to be placed in context with those from other European countries, Table 4.15 shows the emissions of PM_{10} together with those of the particulate precursor species in 2001, the most recent year for which data is available.

4.5.9 UK emission projections

323. Projections of UK emissions are compiled by the NAEI to enable comparisons with international commitments and to aid in the development of national strategies and policies. The 2001 base year projections were recently produced for the four National Emission Ceiling Directive Pollutants and PM_{10}. To produce a projection each activity is linked to an activity driver. The majority of these drivers have been obtained from either Energy Paper 68 (EP68) or Oxford economic forecasting (OEF) or, as is the case for road transport, vehicle kilometre and fleet data. In addition to changes in activity-influencing emissions, improvement in abatement measures – often the result of legal requirements – will reduce emissions. Therefore, the emission factors, where relevant, have been varied to account for this.

Table 4.15 Emissions of primary PM_{10} and particulate precursor species, 2001 (in kt).

Country grouping	Country	PM_{10}	NO_X	SO_2	NMVOCs	NH_3
AC	Cyprus	0.6	22	52	14	9
AC	Czech Republic	33	332	251	220	79
AC	Estonia	33	38	92	33	9
AC	Hungary	30	184	401	166	66
AC	Latvia	13	42	13	59	12
AC	Lithuania	20	55	49	71	50
AC	Poland	305	909	1564	577	313
AC	Slovakia	41	106	129	89	28
AC	Slovenia	13	57	66	39	19
AC	Iceland	—	28	27	10	—
AC	**Total**	**490**	**1772**	**2643**	**1277**	**586**
EFTA4	Liechtenstein	0.1	0.3	0.1	0.6	0.2
EFTA4	Norway	64	221	25	385	25
EFTA4	Switzerland	24	92	21	147	68
EFTA4	**Total**	**88**	**313**	**46**	**533**	**92**
EU15	Austria	48	199	37	232	54
EU15	Belgium	77	323	176	259	85
EU15	Denmark	22	207	26	125	102
EU15	Finland	53	220	85	152	33
EU15	France	550	1411	610	1674	779
EU15	Germany	887[a]	1592	650	1606	607
EU15	Greece	41	331	485	267	74
EU15	Ireland	19	125	131	90	123
EU15	Italy	204	1373	758	1336	451
EU15	Luxembourg	5	17	3	13	7
EU15	Netherlands	51	423	89	268	143
EU15	Portugal	21	397	301	492	108
EU15	Spain	115	1404	1425	1533	384
EU15	Sweden	69	248	57	303	54
EU15	**UK**	**178**	**1680**	**1125**	**1336**	**290**
EU15	**Total**	**2342**	**9950**	**5960**	**9687**	**3294**

Source: UNECE/CLRTAP (2003) with additional data for PM_{10} taken from Auto Oil II (European Commission, 2000) for EU15 countries and from AC countries from CEPMEIP (2001), where values were not reported.
[a]Note: In 2003 Germany did not report values for PM_{10} emissions to UNECE/CLRTAP and so the value of 887 kt for emissions of PM_{10} in Germany is taken from the Auto-Oil II programme (European Commission, 2000), as was done for other EU countries that did not report. However, Germany did report a total suspended particle(TSP) emission of 251 kt for 2000 (from which an estimate of 175 kt PM_{10} can be derived). There is a clear discrepancy between these two emission estimates for PM_{10}.

324. For road transport, a fairly detailed emission forecasting approach is used, adopting the latest traffic forecasts reflecting current Government policies on transport, fleet turnover and the penetration of vehicles meeting the tougher European vehicle emission directives and the impact these will have on emissions from in-service vehicles.

4.5.9.1 Assumptions for non-road transport sources

325. For most of the sectors, other than road transport, the main activity driver is forecasts in fuel consumption according to the central growth/high fuel price scenario in EP68, provided by the DTI. These estimates incorporate an assumed growth in economic activity of about 2.5% per year and the continuation of current trends towards greater use of natural gas and cleaner technologies (DTI, 2000). The industrial and service share sector projections have been based on analysis by OEF using the 1998 version of their UK industry model. The OEF model is not currently used to forecast beyond 2010. To project beyond 2010, OEF assumes that industries will have adjusted to their long-term growth paths by 2010 and that they will continue on these paths.

326. Agricultural forecasts have been provided by Defra on a sector-by-sector basis. The estimates exclude possible effects of common agricultural policy (CAP) reform on animal numbers and farm practice.

327. At the present time the Large Combustion Plant Directive is not transposed into UK law and it is still not decided how the UK is going to do this. The EP68 analysis was based on an assumption that plants without FGD will make use of the 20,000 h opt-out. For plants with FGD, no Large Combustion Plant Directive controls have been applied.

328. Other regulations that have been taken into account include the Sulphur Content of Liquid Fuels Regulations 2000 and the IPPC Directive.

329. In general, the projections are based on a number of assumptions: that measures are introduced when required by legislation and not earlier; and that all operators comply with this legislation and there is no improvement in environmental efficiency other than in response to legislation. Therefore, the projections produced tend to be conservative and are likely to over estimate the actual emissions.

4.5.9.2 Assumptions for road transport sources

330. The base emission projections for road transport are currently from a 2001 base year and are calculated from a combination of road traffic activity projections and knowledge of the expected emission characteristics of the vehicle fleet in the future. The changes in emission factors for vehicles in the projections are largely driven by the legislative emission standards set in the European Directives for Euro III and Euro IV vehicles applying to conventional petrol and diesel powered vehicles. This means that no further improvements in new vehicle emissions occur beyond the limits set for introduction in 2006 for light duty vehicles and 2008 for heavy duty vehicles. In other words, the penetration of advanced vehicle types and technologies currently under development, such as fuel cell, hybrids and other electric-powered vehicles are not considered in the base projections, neither are alternative fuels such as compressed natural gas (CNG).

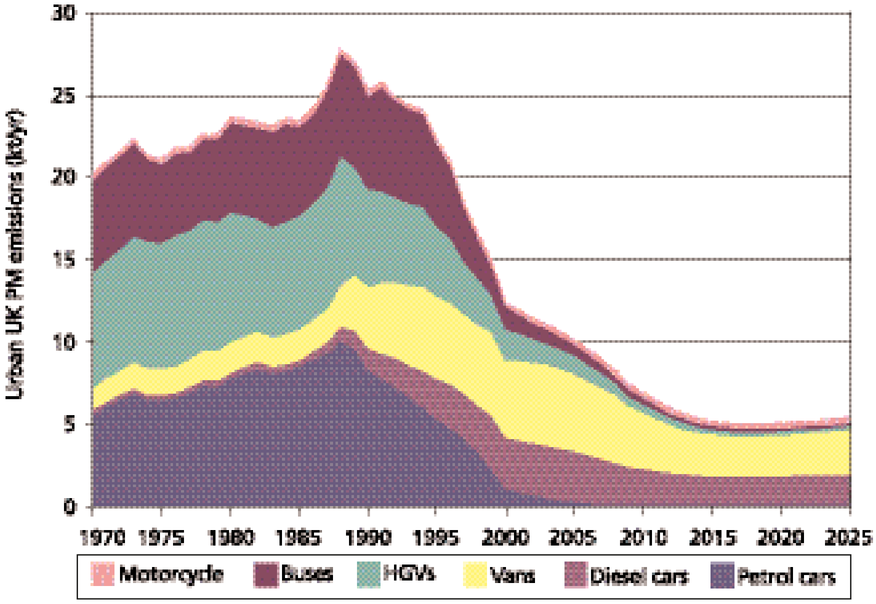
331. The projections are based on emission factors for vehicles up to and including Euro IV standards as shown in Table 4.1. The assumptions and methods for calculating the road transport emission projections were explained in detail in the AQEG report on NO_2 (AQEG, 2004). However, there have been several important changes made to some of the key assumptions since that report was published and these are described in this section.

- The projections now use the latest set of figures from DfT in 2003 on traffic growth, taking account of the impact of the Ten Year Plan for Transport. The mean of the high and low demand traffic growth scenarios is used for traffic in 2010. Projections beyond 2010 use the figures from DfT's older set of 1997 National Road Traffic Forecasts, which provided vehicle km data in 5-year intervals from 1996 to 2031. The vehicle kilometre data in the National Road Traffic Forecasts are rescaled to the Ten Year Plan figures for 2010.

- The rate of penetration of diesel cars in the fleet is important in the emission projections for PM. The 2001 base projections assumed a growth of diesel car sales from actual levels of 18% of all new car sales in 2001 to 30% by 2010.

- As described in Section 4.1.2.2, detailed account is taken of the emission durability requirements of the Directive covering Euro III and IV light duty vehicles and consequently changes in the emission degradation rate for new vehicles assumed for the 2001 NAEI projections were made. Changes to the assumptions made concerning the percentage of failed catalyst cars in the fleet were also described earlier.

Figure 4.28 Time series of PM_{10} emissions from UK urban road transport by vehicle type.



- London-specific bus fleet forecast data from TfL are incorporated into the national emission projections, taking account of measures introduced by London Transport Buses to reduce emissions from the bus fleet in London.
- The NAEI projections now take account of the introduction of sulphur-free fuels (with a sulphur content below 10 ppm) required under Directive 2003/17/EC. The Directive requires their introduction by 1 January 2009, but the NAEI projections assume their penetration into the UK market sooner than this, starting at 15% in 2005 to 100% by 2009. The sulphur-free fuel penetration rates used in the NAEI are supplied by the DfT. The effect of these sulphur-free fuels on emissions is based on studies carried out by SENCO (2000).

4.5.9.3 Baseline PM emission projections for urban UK road transport

332. Figure 4.28 and Table 4.16 demonstrate the trends in urban UK road transport emissions of PM_{10} by vehicle type from 1970 to 2025 as forecast by the 2001 NAEI.

Table 4.16 Urban UK exhaust emissions of PM_{10} (kt) from road transport projected to 2025.

	1970	1980	1990	1995	2000	2005	2010	2015	2020	2025
Petrol cars	5.71	7.83	8.39	5.39	1.16	0.31	0.15	0.13	0.14	0.14
Diesel cars	0.28	0.38	1.28	2.41	3.04	3.09	2.17	1.77	1.75	1.82
Vans	1.33	1.84	3.73	5.03	4.76	4.74	3.45	2.53	2.51	2.74
HGVs	6.96	7.85	5.92	4.28	1.82	1.07	0.51	0.27	0.22	0.23
Buses	5.69	5.47	5.67	5.03	1.36	0.70	0.37	0.23	0.18	0.17
Motorcycles	0.20	0.33	0.29	0.21	0.23	0.26	0.27	0.29	0.30	0.32
Total	**20.18**	**23.70**	**25.27**	**22.34**	**12.36**	**10.16**	**6.92**	**5.21**	**5.10**	**5.42**

333. Urban exhaust emissions have declined by 51% from 1990 to 2000 mainly as a result of the reduction in emissions from the large fleet of petrol cars due to the phasing out of leaded petrol and the introduction of cars fitted with three-way catalysts. However, the tighter standards on emissions of PM from diesel vehicles introduced in the 1990s have also contributed to this decline. The fact that emissions from diesel cars and vans appear to have reduced less rapidly than emissions from large diesel HGVs and buses is because of the faster growth in diesel car and van activity over this period somewhat offsetting the gains achieved by technology improvements.

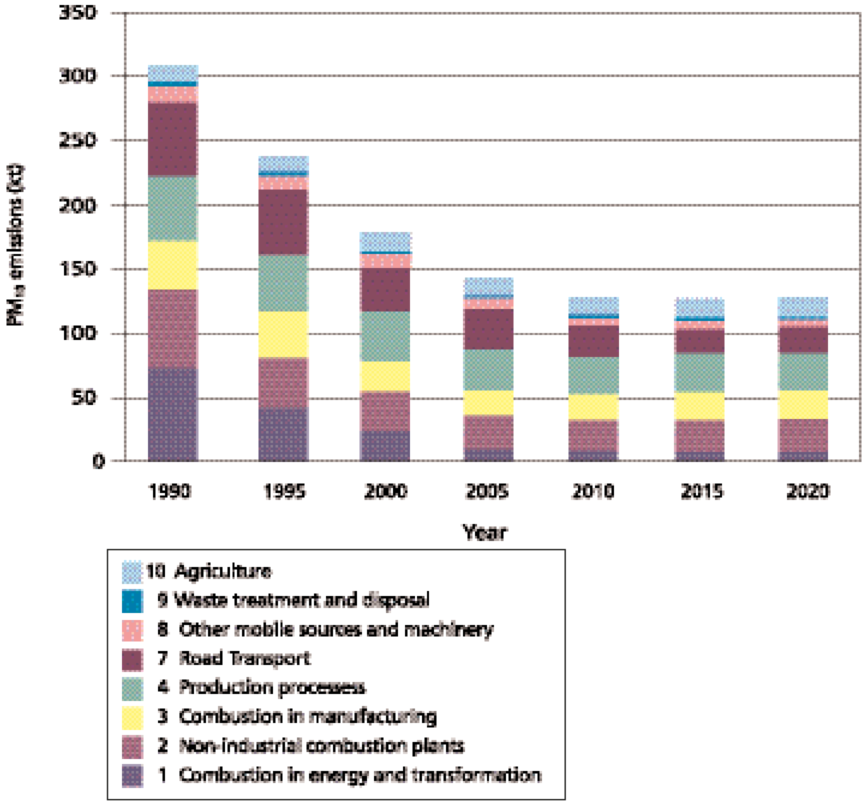
334. Emissions are expected to continue to decline until around 2018 due to the growth of lower-emitting diesel vehicles in the fleet that comply with tighter emission regulations. After this date, without any further improvements in technology beyond those necessary to meet Euro IV standards, emissions start to rise again as the continuous growth in traffic starts to offset the gains achieved by the penetration of cleaner vehicles in the fleet.

335. By 2010, urban exhaust emissions are predicted to reduce by 44% compared to 2000 levels and by 2020 by 59% compared to 2000 levels. By 2010 the largest PM emitters are expected to be diesel cars and vans due to their increased growth in activity, although this is tempered by the fact that new vehicles entering the fleet will be lower emitting. Very tight standards on PM emissions from larger diesel vehicles (HGVs and buses) ensure the reduction in emissions from these vehicles continues. The introduction of cleaner diesel fuels with lower sulphur content will also contribute to the decrease in PM emissions.

4.5.10 Baseline PM emission projections for all UK sources

336. Figure 4.29 shows the time series of PM_{10} emissions in the UK from 1990 projected to 2020 at 5-year intervals. Emissions are expected to decline by 28% between 2000 and 2010. This is primarily due to the fitting of FGD at power stations and reductions in the road transport.

Figure 4.29 Historic and projected UK PM_{10} emissions.



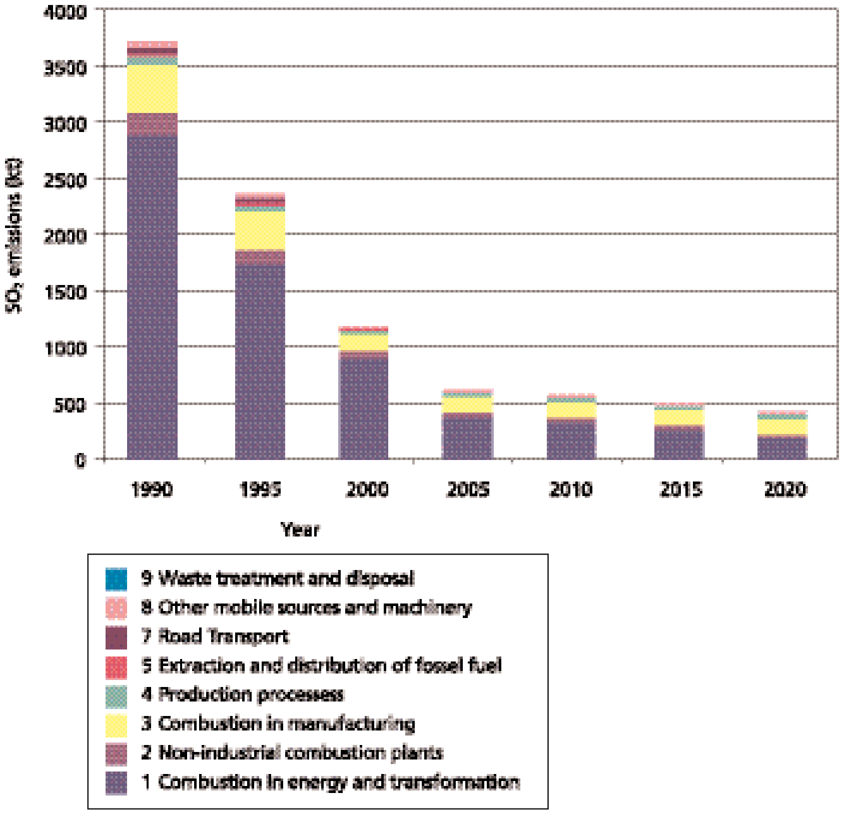
4.5.11 Baseline emission projections for PM precursor pollutants from all UK sources

4.5.11.1 SO_2

337. Figure 4.29 shows the time series of SO_2 emissions in the UK from 1990 projected to 2020 at 5-year intervals. Emissions of SO_2 are predicted to decline by approximately 52% between 2000 and 2010. This is largely due to the Sulphur Content of Liquid Fuels Regulations 2000 (1999/32/EC). For gas oil, a sulphur content of no more than 0.2% was required by 2003 and no more than 0.1% by 2008. For fuel oil, the requirement was that from 2003 the sulphur content of the oil must be less than 1%. In addition, a large decline is seen in SO_2

emissions from the power station sector. This is because a switch away from coal to lower sulphur fuels such as gas and the fitting of FGD at further coal-fired power stations are forecast.

Figure 4.30 Historic and projected UK SO_2 emissions by sector.



4.5.11.2 NO_X

338. Figure 4.31 shows the time series of NO_X emissions in the UK from 1990 projected to 2020 at 5-year intervals. UK NO_X emissions are predicted to decline by ~30% between 2000 and 2010. NO_X emissions from the road transport sector are forecasts to decline by 54% as a result of vehicles meeting tighter emission standards penetrating the UK fleet. A decline is also expected in the power station sector as coal consumption with higher NO_X emissions per joule of energy is forecast to decline, whereas natural gas consumption with lower NO_X emissions per joule of energy is expected to increase.

4.5.11.3 NMVOCs

339. Figure 4.32 shows the time series of NMVOC emissions in the UK from 1990 to 2020 at 5-year intervals. NMVOC emissions are predicted to decline by ~28% between 2000 and 2010. This is largely a result of a reduction in emissions from industrial processes due to more stringent emission controls resulting from the Solvent Emission Directive (1999/13/EC). In addition NMVOC emissions from road transport are expected to decline. This is as a result of an increasing number of vehicles meeting tighter emission standards requiring catalytic converters, fuel switching from petrol to diesel cars, lower volatility of summer petrol fuel blends and better evaporative control systems on cars leading to lower fuel evaporative losses.

Figure 4.31 Historic and projected UK NO_X emissions by sector.

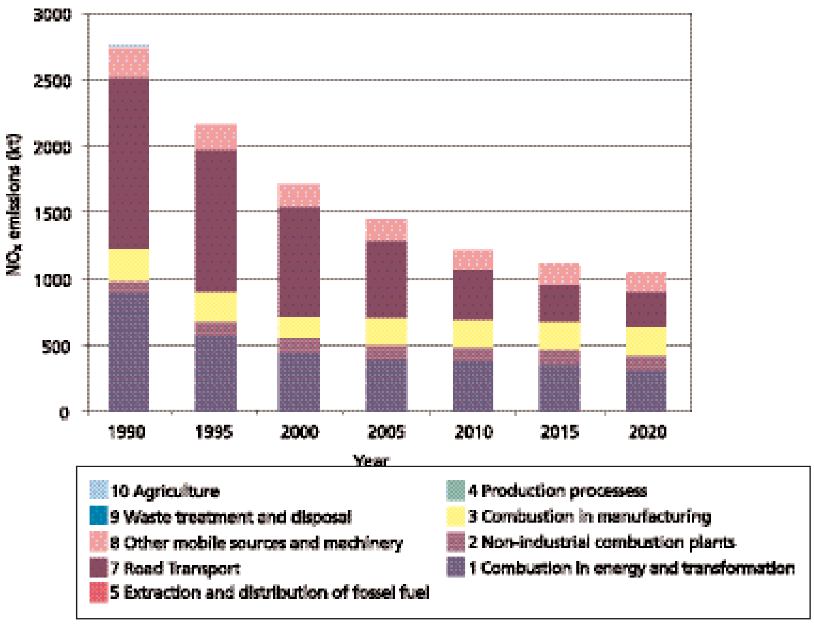
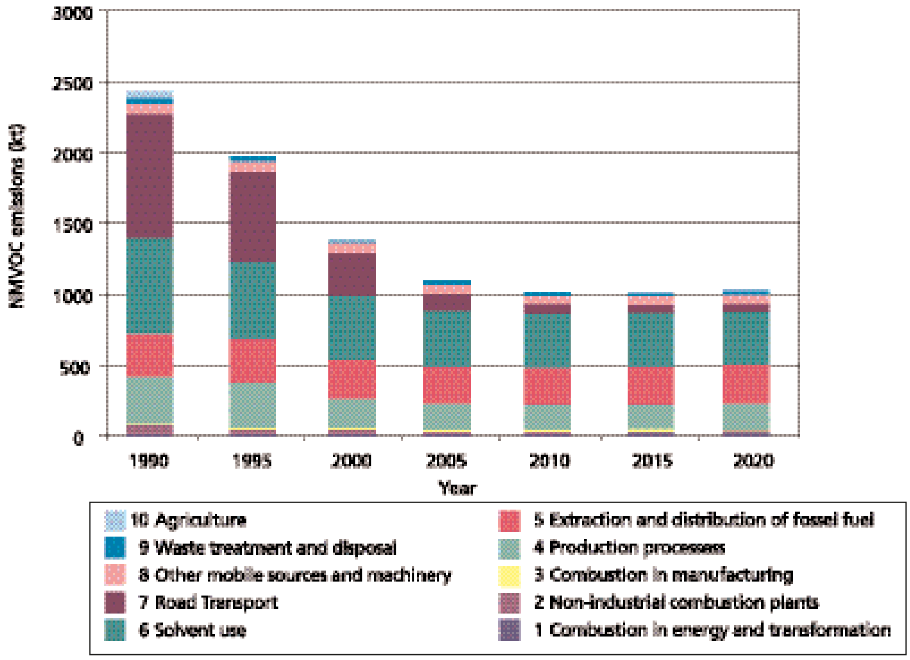


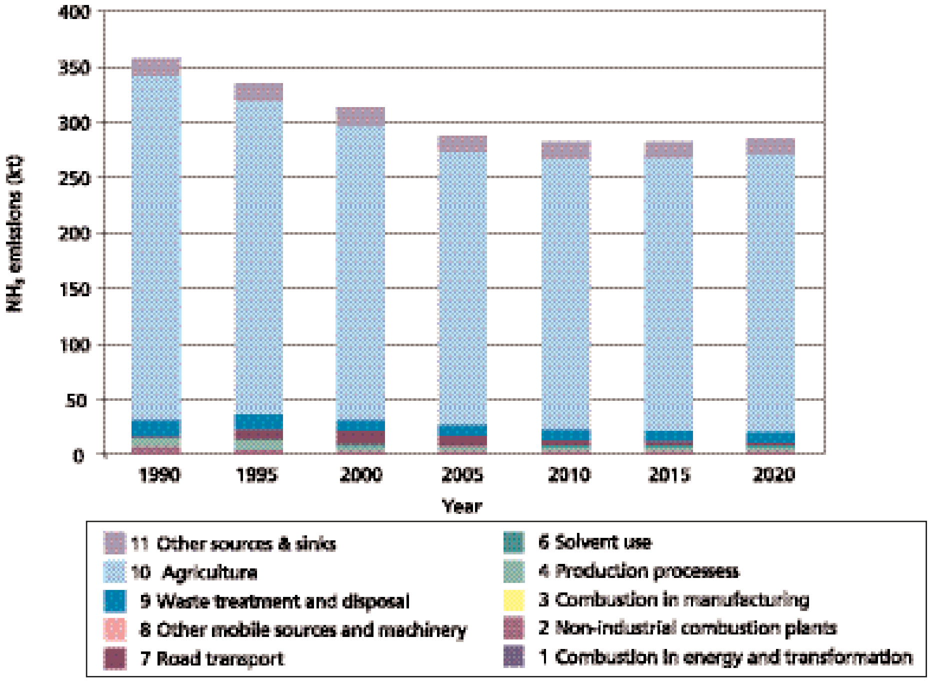
Figure 4.32 Historic and projected UK NMVOC emissions by sector.



4.5.11.4 NH_3

340. Figure 4.33 shows the time series of NH_3 emissions in the UK from 1990 to 2020. In 2000, 85% of NH_3 emissions were from agriculture. A 10% decline in total NH_3 emissions is predicted between 2000 and 2010. This is largely due to decreasing animal numbers and a decline in fertiliser use. NH_3 emissions from road transport increased between 1990 and 2000. This was caused by the increased use of catalytic converters. Post 2000 NH_3 emissions from road transport are predicted to decline due to the increasing number of diesel vehicles and improvements in technology.

Figure 4.33 Historic and projected UK NH_3 emissions by sector.



4.6 Emissions controls and abatement technologies

4.6.1 Abatement of industrial emissions of particles

341. Strategies for reducing emissions of PM from industrial sources may be subdivided into three general approaches:

- process changes;
- process management; and
- end-of-pipe arrestment.

A detailed description of abatement methods is beyond the scope of this report, but this section gives a brief overview of some of the main control strategies for industrial emissions of PM. Further information can be found in the IPPC BREF documents that can be found at http://eippcb.jrc.es/pages/FActivities.htm and in the *Environment Agency Guidance Notes*.

342. Not all of the approaches will be available for each source of particles but all three can contribute to significant reductions in national emissions of PM.

343. Process changes involve modifications to raw materials, process technology, operational parameters or products, which reduce or prevent emissions occurring from a process. Examples of process changes range from relatively simple measures such as burning ‘cleaner’ fuels or switching fuels used in an existing plant or modifying individual pieces of equipment such as burners (to improve combustion efficiency) through to more fundamental changes such as replacing complex plant with less polluting technology, for example, replacing a coal-fired power station with a CCGT power station.

344. Process management refers to improvements to the operation of an existing process leading to reduced emissions. Process management measures can be very 'low-tech', for example, ensuring that spillages of dusty materials from processes are cleaned up rapidly before they can be suspended by wind, keeping doors closed to minimise fugitive dust emissions or ensuring that process equipment is maintained in good condition. At the other end of the scale, computerised control of large combustion processes can ensure efficient combustion and potentially lower emissions of PM. Monitoring of PM abatement plant efficiency in order to prevent high emissions occurring during periods of equipment failure is also an important process management option.

345. Both process change and process management methods have been applied by UK industry and are particularly important for those sources where fugitive emissions are significant. Both strategies have made a major contribution to reducing particle emissions over the past three decades, however, the scope for further reductions via these means is probably fairly limited (except in the case of major changes in process technology). Currently, the main option for reducing emissions of particles is the use of end-of-pipe arrestment technology to recover, or occasionally destroy, dust in waste gas streams. The four most important technologies are electrostatic precipitators (ESPs), fabric filters, scrubbers and cyclones, with the first three being most important for large industrial (Part A) processes and cyclones used as the main abatement measure mostly for smaller (Part B) processes.

346. ESPs work by using a high-voltage electrical current to impart an electrical charge to PM passing through the device and then collecting the charged particles on collection plates. The collection plates are periodically rapped or vibrated to dislodge the PM into a hopper. ESPs typically have two or three stages of collection. ESPs can operate at relatively high temperatures and can treat both wet and dry waste streams. Their performance is affected by the electrical properties of the dust and collection efficiency is markedly lower for finer particles (especially <1 µm).

347. Fabric filters employ a fabric membrane that is gas permeable but traps PM. The layer of dust that builds up on the fabric also acts as a filter and increases the efficiency of collection but must be removed periodically to control the pressure drop over the filter. Fabric filters can operate to moderately high temperatures but are sensitive to moisture. They are very efficient even for fine PM. Dry or semi-dry absorbents can be added to the waste gas upstream of a fabric filter and, being collected with the PM, can remove gaseous pollutants such as SO_2 or mercury. Ceramic and other sinter materials have been developed for filtration of gases in high temperature applications.

348. Wet scrubbing involves the entrainment of dust particles in a liquid. The liquid can be water or an aqueous solution of an acid, alkali or oxidising agent where gaseous pollutants are also being abated. Various designs exist for the contact stage of scrubbers and venturi scrubbers are the most efficient at collection of PM but are less good at absorbing gaseous pollutants. Generally, therefore, the choice of scrubber design will depend upon whether the control of gaseous or particulate pollutants is more important. Wet scrubbing techniques can be used for wet and dry waste streams and are able to treat high temperature waste streams. They are less efficient at collecting fine particles. Wet FGD is an example of a wet scrubbing technology used for control of gaseous pollutants that will capture PM as well. However, because wet FGD systems cannot handle high levels of PM, a primary abatement device such as an ESP must be fitted upstream.

349. Cyclones are inertial separation devices and can achieve high collection efficiencies for larger particulate material (>PM_{10}). For particles smaller than 10 µm and particularly below 2.5 µm their efficiency is substantially poorer than ESPs and bag filters. They are rarely applied as final abatement on major industrial (Part A) processes but are used as a first stage abatement device. The technology is still widely used in some industries that generate coarse and/or 'difficult' PM (for example, wood machining processes and food processes). Although the emissions may be small in a national context, such processes may have low height discharges and other factors that may make the emissions more significant for local air pollution.

350. ESPs are the dominant technology for utility boilers and other large combustion plants and are also widely used in industrial processes such as cement and lime kilns. Fabric filters are also common for the control of industrial process emissions and are increasingly important for combustion processes as well. Fabric filters can be regarded as the likely candidate technique for most new industrial processes, although ESPs can also be considered BAT for some industrial sectors, e.g. for cement and lime kilns. The use of wet scrubbing techniques is declining. However, they are still important where waste streams containing mixed gaseous and PM emissions must be treated. Cyclones are too inefficient to be used alone for large industrial sources of PM, but are still common for smaller processes.

4.6.2 Abatement of emissions from road vehicles

351. During the 1990s, reductions in particulate emissions from diesel vehicles were mainly achieved through improvements to engine design and fuel systems. These were further aided by a simultaneous reduction in the sulphur content of diesel fuels from levels close to 2000 ppm in the early 1990s to around 50 ppm by the end of the decade.

352. However, to meet the even more stringent particulate emission standards coming into effect, diesel vehicles – especially the larger ones – will need exhaust after-treatment processes to further reduce emissions. Many vehicles will be fitted with some form of diesel particulate filter (DPF) on the exhaust, as these are extremely efficient at reducing emissions of particles, including the ultrafines.

353. There are a number of DPF designs on the market, all generally requiring some form of continuous or periodic filter regeneration capability built into them. DPFs capture particulates in the exhaust stream through a combination of surface-type and deep-bed filtration mechanisms. The most commonly used type of DPF in the UK are those that convert NO_X to NO_2 in the exhaust stream and use the NO_2 to continuously oxidise the particulates held on the filter, thus regenerating the trap. The system uses a ceramic wall-flow filter to trap particulates. An oxidation catalyst converts NO_X in the exhaust to NO_2 upstream of the filter. The efficiency of these DPFs at reducing PM emissions is typically around 90%. A potential problem is the release of a higher fraction of NO_X as NO_2 from the exhaust unless the system is optimised to reduced NO_2 slippage. The negative implications of primary NO_2 emissions from diesel exhausts were discussed in the AQEG report on

NO_2 (AQEG, 2004). This type of DPF requires use of ultra-low sulphur fuel (<50 ppm), but this fuel has been freely available in the UK since around 2001, well ahead of legislative requirements mandating its introduction by 2005. Some new diesel vehicle models are now equipped with this type of DPF. DPFs of this nature have also been developed for retrofit application, and schemes such as the Clean UP programme – run by the Energy Savings Trust – promote and offer grants towards the cost of fitting DPFs on diesel vehicles.

354. Fuel additives have also been developed to enable the regeneration of DPF systems. These are usually organometallic fuel additives, which oxidise the particulates held on the filter through a catalytic process (Cook and Richards, 2002). The most common additives use iron and cerium, but others have used sodium and strontium.

355. Diesel oxidation catalysts have been fitted on some vehicles for some time, especially HGVs and buses and other vehicles in 'captive fleets'. These help to reduce emissions of CO and hydrocarbons, but also reduce the soluble organic fraction of diesel particulates. However, they will also oxidise SO_2 present in the exhaust leading to the generation of sulphate particulates, which can lead to an increase in mass emissions of PM from vehicles running on fuels with high sulphur content. This is less of a problem with fuels of very low sulphur content, and modern oxidation catalysts are optimised to minimise CO and hydrocarbon emissions with low SO_2 conversion.

356. Without any after-treatment technology, the decrease in sulphur content of fuel alone will slightly reduce PM emissions from diesel vehicles. Directive 2003/17/EC mandates the provision of sulphur-free fuels (<10 ppm) by 2009, but the uptake of this fuel in the UK is expected to commence earlier than this date. It is estimated that running a diesel vehicle on sulphur-free fuel leads to around a 5% reduction in PM emissions relative to emissions from the same vehicle running on current ultra-low sulphur grade fuel (~50 ppm sulphur).
The introduction of ultra-low sulphur diesel itself led to a 510% reduction in PM emissions compared to emissions from vehicles running on 500 ppm sulphur diesel, typical of the grade of fuel sold in the mid to late 1990s.

357. Water-diesel emulsion fuels have also shown significant reductions in both emissions of PM and NO_X. The uptake of this fuel in the UK has been limited so far.

358. Switching vehicles from running on diesel to alternative transport fuels such as LPG and CNG led to significant reductions in PM emissions. The uptake of these fuels has not been significant in the UK, but targeted fleets have the potential to switch to CNG, with significant benefits in terms of reduction in PM emissions.

359. Among the precursor pollutants, transport only makes a major contribution to emissions of NO_X and NMVOCs. Emissions of these pollutants have been significantly reduced over the last decade as a result of the fitting of three-way catalysts on petrol vehicles. Tighter reductions on emissions of these pollutants from both petrol and diesel vehicles will lead to further reductions in the future. Measures for reducing NO_X emissions were discussed in detail elsewhere (AQEG, 2004). Emissions of NMVOCs have also been reduced by the fitting of carbon canister devices on petrol cars to curb evaporative emissions of fuel vapour from vehicles, and the reduction in the volatility of fuel blends sold during the summer months, as required by European directives. Controls at filling stations have also helped to reduce emissions during the refuelling stages.

360. Transport is not a significant source of ammonia, but emissions have been increasing with the fleet penetration of cars with catalytic converters, as ammonia is a by-product of the NO_X reduction process on the catalyst. However, better catalyst and engine management systems are expected to tackle this problem, leading to lower emissions from cars in the future (AQEG, 2004). Selective catalytic reduction using urea to control NO_X emissions from heavy duty diesel vehicles can lead to ammonia slip, but again systems can be optimised to minimise emissions (AQEG, 2004).

361. Road transport makes a very small contribution to emissions of SO_2 in the UK; emissions are being further reduced with the uptake of fuels with lower sulphur content.

4.7 Impact of policies on particle emissions

4.7.1 The impact of increased diesel car penetration

362. There has been a rapid rise in the sale of new diesel cars in the UK in recent years. Sales increased from 14% in 2000 to almost 24% by the end of 2002 and 27% by 2003. It is believed that the reason for this rapid increase is that motor manufacturers, in order to meet the Voluntary Agreement on CO_2 emissions, have been making diesel cars more attractive through reducing the previous cost penalty over commensurate petrol cars. Additionally, the performance and driveability of diesel cars has improved to the extent that the stigma attached to them by motorists in the past has been diminishing. Diesel car sales in the UK are still lower than the rest of Europe, but are expected to catch up, although still not reach parity. The current forecast of 30% diesel car sales by 2010 used in the NAEI emission projections, originally set in 2002, now looks doubtful and the DfT and Society of Motor Manufacturers and Traders (SMMT) now forecast sales in the UK will reach 42% by 2010.

363. The NAEI has modelled the sensitivity of projections for PM_{10} and NO_X emissions from UK road transport in response to increased diesel car sales relative to the original base case assumption of 30% diesel car sales by 2010. Emissions were calculated for a growth in car sales to 42% by 2010 (the DfT/SMMT estimate), 35% by 2010 and 50% by 2010. The results for road transport emissions in urban areas are shown in Table 4.17. PM emissions are more sensitive to the increase in diesel car sales than NO_X emissions. By 2015, PM emissions would be almost 10% higher for the 42% scenario than is currently forecast for that year; for the 50% scenario, PM emissions would be 15% higher than is currently forecast.

364. Table 4.18 shows how increases in diesel car penetration rates affect the predicted rate of decline in PM and NOX emissions for each year relative to 2000. It shows that whereas a growth in diesel car sales to 30% in 2010 is associated with a 57% reduction in PM10 emissions compared to 2000 levels by 2015, that reduction becomes 53% if diesel car sales reach 42% by 2010.

Table 4.17 Urban UK exhaust emissions in kt from road transport for different diesel car penetration rates. Base case assumes 30% sales by 2010.

		2000	2005	2010	2015	2020	2025
PM	Original base case	12.08	9.95	6.82	5.19	5.09	5.41
	35% diesel by 2010	12.08	9.98	6.94	5.40	5.36	5.70
	42% diesel by 2010	12.08	10.01	7.08	5.70	5.73	6.10
	50% diesel by 2010	12.08	10.02	7.17	5.96	6.14	6.55
NO_X	Original base case	274.1	196.6	129.5	95.5	89.4	91.5
	35% diesel by 2010	274.1	197.3	130.3	96.5	90.5	92.7
	42% diesel by 2010	274.1	197.9	131.4	97.8	92.0	94.2
	50% diesel by 2010	274.1	198.1	132.0	99.0	93.7	96.1

Table 4.18 Percentage change in urban UK exhaust emissions from road transport relative to emissions in 2000 for different diesel car penetration rates. Base case assumes 30% sales by 2010.

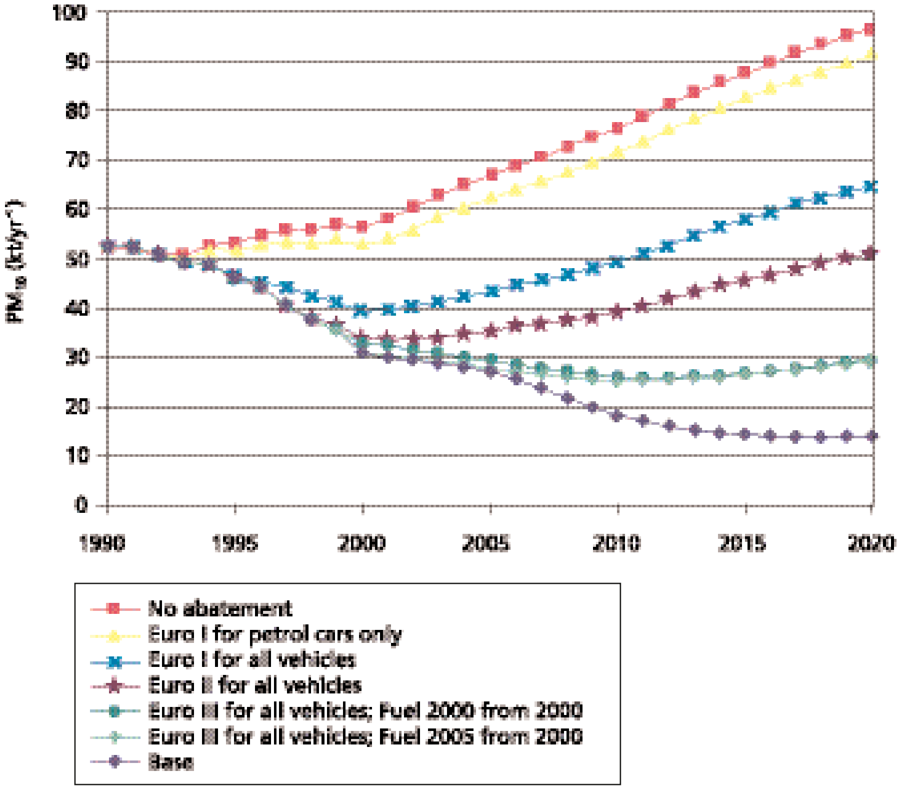
		2000	2005	2010	2015	2020	2025
PM	Original base case	0%	18%	44%	57%	58%	55%
	35% diesel by 2010	0%	17%	43%	55%	56%	53%
	42% diesel by 2010	0%	17%	41%	53%	53%	50%
	50% diesel by 2010	0%	17%	41%	51%	49%	46%
NO_X	Original base case	0%	28%	53%	65%	67%	67%
	35% diesel by 2010	0%	28%	52%	65%	67%	66%
	42% diesel by 2010	0%	28%	52%	64%	66%	66%
	50% diesel by 2010	0%	28%	52%	64%	66%	65%

4.7.2 An evaluation of policies and regulations affecting emissions from the road transport sector in the UK

365 Netcen carried out a study to evaluate the impact of air quality policy over a period of major change – the last decade – to help provide information on which policies have been successful and which have not. The study helped to improve our knowledge of the quality of modelled outputs and showed how the response to legislation can sometimes be different to that which was originally anticipated. The study focused on the road transport sector, in particular the Euro standards on vehicles and fuel standards and how they affected emissions and air quality.

366 The analysis considered the emission benefits of individual policies and combined policies compared to a scenario of no policies, the 'no abatement' scenario. The no abatement scenario assesses what would have happened in the absence of the policies (that have since been implemented). It is based on the conditions in 1990, but takes account of the economic/activity growth over the period. It should be pointed out that the study involved projections from an older version of the inventory (2000 NAEI), with slightly less up-to-date assumptions than those generally discussed in this report. The absolute emission reductions for PM_{10} and NO_X are shown in Figures 4.34 and 4.35. In general, the greatest single benefit has been achieved by the Euro I emission standard. However, the pattern of behaviour is different with regard to the effectiveness of the Euro standards on these two pollutants.

Figure 4.34 Effect of Euro emission standards on trends in UK PM_{10} emissions from road transport.

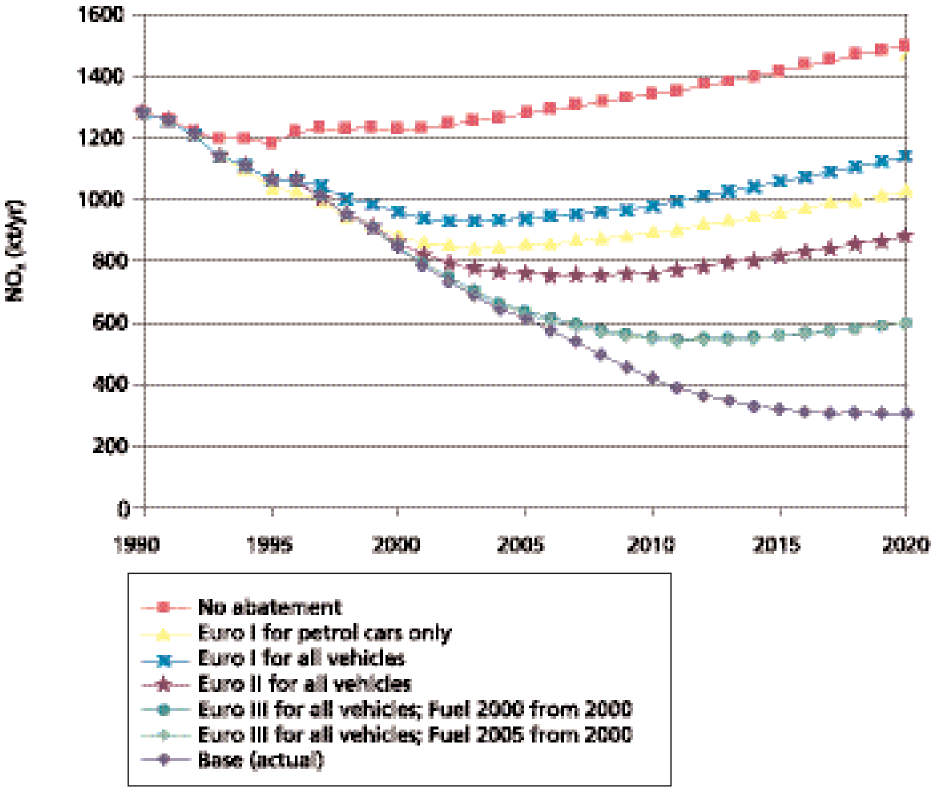


367. For NO_X, a large proportion of the improvements seen are from the introduction of Euro I technology to petrol vehicles (around a 30% improvement compared to 'no abatement'). However, the NO_X emission factors for Euro I heavy duty vehicles were higher than for pre-Euro I vehicles (this issue was discussed elsewhere (AQEG, 2004)). Consequently, the Euro I emission standards appear to be detrimental for NO_X emissions from diesel vehicles. This spreads the benefits of NO_X reductions more evenly over Euro I to IV standards. The analysis here shows that the approximate (cumulative) benefits of successive technical standards are 30% reduction for Euro I, 40% for Euro II, 60% for Euro III and 70–80% for Euro IV, relative to the 'no abatement' scenario.

368. A different pattern emerges for PM_{10}. The effect of introducing Euro I technology to petrol vehicles was small (7%). The largest benefits arose from the introduction of Euro I technology to diesel vehicles (30%), but there were also progressive improvements through the Euro standards: compared to the 'no abatement' scenario, the cumulative reductions were 50% for Euro II, 65–70% for Euro III and 75–85% for Euro IV.

369. These conclusions are important because when all emissions are considered, it is the Euro I standard (for petrol cars) that seems to have provided the dominant benefit. However, when the focus is switched to the two primary pollutants of most concern for local air quality management, NO_2 and PM_{10}, the benefits are attributable to all the Euro standards applied to all vehicles.

Figure 4.35 Effect of Euro emission standards on trends in UK NO_X emissions from road transport.



370. This pattern is slightly different to that anticipated at the start of the study. A previous evaluation (undertaken in 2001 with an older set of emission factors) showed a much greater overall benefit attributed to Euro I in reducing NO_X emissions and lower benefits from later Euro standards. The new emission factors, therefore, show more benefits attributable to later Euro standards. This mitigates against the expected effect of diminishing returns, that is, as emission standards are tightened, it becomes progressively more difficult to get large emission reductions at reasonable cost. It should be stressed, however, that the Euro III and Euro IV factors are not based on measurement data (whereas values for Euro I and II are) and that there are, therefore, some uncertainties in these conclusions and the predicted outcomes.

4.7.3 Low emission zones

371. A feasibility study that considered the effects of a low emission zone (LEZ) in London has recently been carried out; its aim was to quantify the benefits of reducing road vehicle emissions of NO_X and PM_{10} (Watkiss *et al*., 2003). The study highlighted the difficulty in determining which vehicles would be affected by the scheme for several reasons; these included the reliability of emission factors, the costs of enforcement of the scheme and the impacts on the operators of vehicle fleets. Nevertheless, three recommended schemes were identified. It was stated that by 2006/7 all lorries buses and coaches that are Euro II (that is, vehicles built before late 2001) or older would need to fit abatement equipment (for example, a particle filter) in order to enter the zone, but Euro III or Euro IV vehicles would be allowed to enter. In 2010, two schemes emerged as the best options. The first would require all heavy vehicles that are Euro III (that is, those built before late 2006) or older to fit pollution control equipment in order to enter the zone but would allow Euro IV vehicles to enter. The second would in addition restrict vans and taxis more than 10 years old from entering the zone. The study recommended the scheme should be enforced within the Greater London boundary.

372. It was assumed that vehicles fitted with pollution control equipment used continuously regenerating traps (CRTs). It was further assumed that these traps reduced total PM_{10} emissions by 95%. For 2007 it was calculated that total road transport emissions of PM_{10} would be reduced by 9.0% compared with baseline 2007 emissions. For 2010 it was estimated the LEZ would reduce PM_{10} emissions by 19.0% – by 23% if vans and taxis were included, – compared with baseline 2010 emissions.

4.7.4 The London Congestion Charging Scheme (CCS)

373. The London CCS started in February 2003. The CCS aims to reduce road traffic congestion in central London by charging each vehicle £5 to enter the zone on weekdays during 7.00 am – 6.30 pm (excluding bank holidays). The CCS zone covers approximately 22 km^2 of central London (<1.5% of the area covered by the Greater London boundary). Certain important vehicle types, such as buses and taxis, are exempt from the charge, as are vehicles using alternative fuels such as gaseous fuels and electricity. An initial assessment of the impact of the scheme has been undertaken to determine the effect on emissions and the concentrations of NO_X, NO_2, PM_{10}, CO_2 and fuel use (Barratt *et al*., 2004); the following summary is based on that assessment. Overall, the scheme has been at the high-end of TfL's expectations in terms of reducing traffic congestion (TfL, 2003).

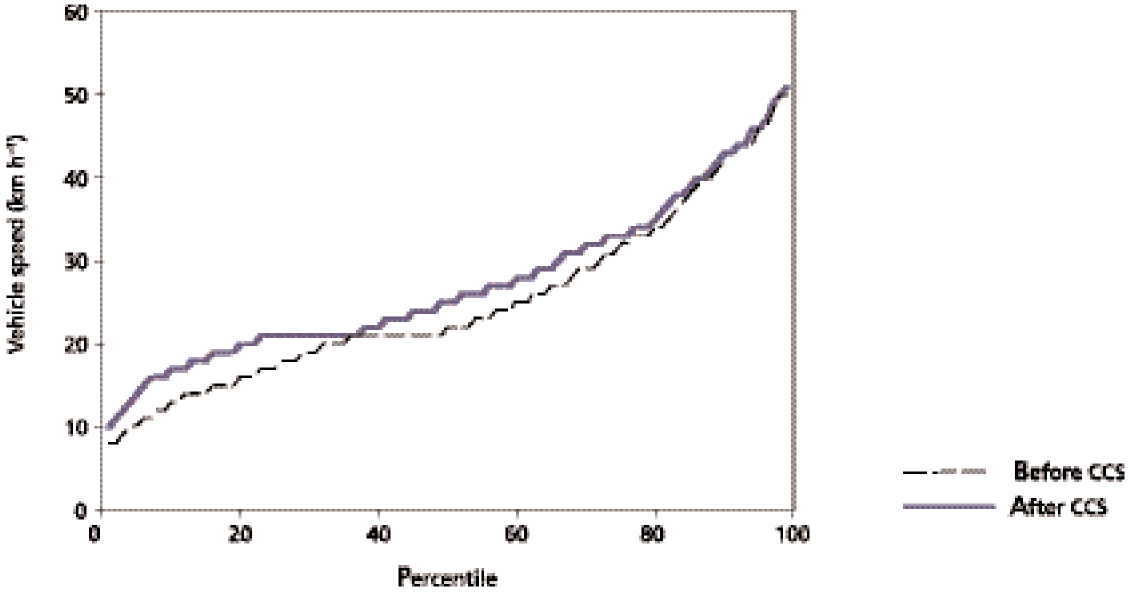
374. Table 4.19 summarises the changes in emissions brought about by the introduction of the CCS by vehicle type (Beevers and Carslaw, 2005). These emissions were calculated using the currently available emission factors described in Section 4.2.2.4 and London-specific vehicle type and age profiles, which are also used in the LAEI. Use was made of detailed vehicle flow and speed data that were collected as part of the CCS monitoring programme. Changes in the total emissions of PM_{10} depend on the vehicle type considered and vary between a +5.0% increase for motorcycles and a –22.4% for rigid HGVs.

Table 4.19 Effect of congestion charging on calculated annual emissions of PM_{10} (% change).

	Motor-cycles	Taxis	Cars	Buses and coaches	LGV	Rigid HGVs	Artic HGVs	Total
CCS zone	5.0	3.3	–36.1	–13.4	–14.8	–22.4	–20.8	–11.9
Inner ring road	6.0	–4.6	–2.3	–0.7	4.0	–6.7	–6.0	–1.4

375. One important outcome from the analysis is the effect of vehicle speed. Because vehicle speeds in central London are low, any increase in vehicle speed caused by reduced congestion can have a large effect on calculated emissions. For example, taxi flows increased by 13% in the CCS zone but total emissions of PM_{10} only increased by 3.3%. The small increase is a result of the increased vehicle speed on roads in the CCS zone, which offsets the increased flows of both taxis and buses. Overall, the effect of increased vehicle speed has reduced PM_{10} emissions in the zone by around a factor of two compared with the reduction of vehicle flows. Figure 4.36 shows the percentile speed distribution of hourly vehicle speeds on major roads before and after the introduction of the CCS and highlights that the most significant changes in speed have occurred in the lower speed range below 20 km h^{-1}. It should also be stressed that the analysis might underestimate the benefit of the CCS in terms of the total reduction in PM_{10} emissions because no account has been taken of the potential reduction in tyre and brake wear and resuspended PM_{10} resulting from a reduction in vehicle flows (particularly of HGVs). As is discussed in Section 4.2.2.12, resuspended material from the road surface (and tyre and brake wear) can be important sources of PM_{10}.

Figure 4.36 Percentile vehicle speed distribution for hourly speed data on major roads before and after the introduction of the CCS.



Chapter 5

Methods for monitoring particulate concentrations

Key points

- The measurement of concentrations of PM in ambient air is not straightforward. There are a variety of techniques available for measuring mass concentrations, but due to the complex nature of PM, the method that is selected can significantly influence the result.

- The TEOM analyser is widely used for monitoring PM_{10} concentrations in the UK. This continuous sampler provides real-time data with a short time resolution that can be used for providing public information. For the TEOM to be used as a USEPA equivalent method for PM_{10} measurement, a default adjustment factor (1.03 * TEOM reading + 3 $\mu g\ m^{-3}$) must be applied. All TEOM analysers in the UK, both for PM_{10} and $PM_{2.5}$, are set up with this default factor included.

- Due to the need to eliminate the effect of changing humidity on the mass measurement, the TEOM is required to maintain the sample filter at an elevated temperature which may lead to losses of semi-volatile species such as ammonium nitrate. This has led to reported differences in concentrations of PM between the TEOM and the European reference method for PM_{10} measurement, which uses an unheated, filter-based gravimetric sampler.

- The European reference method for PM_{10} measurement, EN 12341, does not fully address the issue of semi-volatile losses and so can give inconsistent results. The draft reference method for $PM_{2.5}$ measurement, prEN 14907, is intended to reduce this problem and will also provide the basis for a revised PM_{10} measurement reference method.

- Intercomparison trials between the European reference method and various other methods, including the TEOM, are being carried out in the UK. As an interim measure, a default scaling factor (also known as correction factor) of 1.3 is currently applied to all TEOM PM_{10} data in the UK national networks in an attempt to account for these losses.

- If the reference method were changed so that only the non-volatile component of the particle mass were measured, this would lead to great improvements in the consistency and practicality of measurements for regulatory purposes. These 'core particulate' measurements could be supplemented by specific monitoring techniques to determine the volatile and other particle mass components as required.

- Monitoring of PM_{10} concentrations, based largely on the TEOM analyser, is carried out on a national scale on behalf of Defra and the Devolved

Administrations and at a regional or sector-specific scale on behalf of local authorities, the Highways Agency and the power generation industry. These networks are subject to rigorous quality assurance and quality control procedures.

- Data from 240 PM_{10} monitoring sites operating in 2003 – including sites at kerbside, roadside, urban background/centre, industrial and rural/remote locations – have been collated within a spreadsheet pro forma for this report Fifteen of these sites have co-located $PM_{2.5}$ analysers.

- National networks also exist to measure concentrations of black smoke, PAHs, heavy metals and major ions (such as sulphate, nitrate and chloride).

- As part of a research programme funded by Defra and the Devolved Administrations, continuous measurements of particulate nitrate and organic and elemental carbon, together with total and fractionated particle number concentrations (predominantly ultrafine particles), are carried out at a number of sites in the UK.

- The monitoring of PM in the UK is dominated by PM_{10} mass measurements. Only limited data are available for particle number, size, surface area and morphology.

5.1 Introduction

376. The complex nature and composition of airborne PM has been described in detail in Chapter 2. In the absence of a confirmed toxicological route for the effects of particles on human health, the relative importance of the different components of the particle mass is, as yet, unknown. This presents a significant challenge for the measurement of particulate concentrations in the ambient atmosphere and, to a large extent, the method selected for the collection and determination of particle mass determines the mass concentration that is subsequently reported. For this reason it is essential to state precisely the method by which the reported measurements have been made.

5.2 Overview of PM monitoring methods

5.2.1 Introduction

377. There is a variety of monitoring methods available for the measurement of mass concentrations of PM in ambient air. These include both direct reading instruments, which provide continuous measurements of particle concentrations, and filter-based gravimetric samplers that collect the particulate material onto a filter, which must then be weighed subsequently in a laboratory.

378. Commonly used methods in the UK for the mass measurement of PM in ambient air include:

- filter-based gravimetric samplers (including the European reference sampler);

- Tapered Element Oscillating Microbalance (TEOM) analysers;

Box 5.1 Units of measurements

With gaseous pollutants it is possible to express concentrations as an amount fraction – the ratio of pollutant molecules to the total number of air molecules – for example, with units of nanomoles per mole (nmol/mol) or, more commonly, parts per billion (ppb).

This is not possible for PM, and measurements are always given in units of particulate mass per unit volume of air (typically µg m^{-3}). When these units are used without specifying the temperature and pressure of the air, the same ‘packet’ of air will have a different concentration as these properties of the air change. The European legislation for PM requires that the air volume used must be at the same ambient air temperature and pressure as at the time of sampling.

In practice this means that appropriate corrections need to be made if the flow rate used to calculate the sampled volume is not based on the actual volume of sampled air.

As is described in more detail within this chapter, different measurement techniques, although nominally measuring the same PM, may treat the airstream in different ways, leading to significantly different results. For clarity, all mass measurements of PM_{10} and $PM_{2.5}$ in this report are expressed as µg m^{-3} for both gravimetric and TEOM analysers. If any ‘correction factor’ has been applied to the data, this is explicitly stated. For example, where data from TEOM analysers have been ‘corrected’ by applying a factor of 1.3, these data are reported as µg m^{-3} (TEOM * 1.3).

- ß-attenuation analysers;
- optical analysers;
- black smoke method; and
- personal samplers.

379. A summary of the various advantages and disadvantages of each method is provided in Table 5.1. Annex 4 provides a summary of the PM_{10} and $PM_{2.5}$ measurement techniques that are used in the monitoring networks across Europe.

378. Alongside these measurement techniques, which directly or indirectly monitor particulate mass, other types of measurements are routinely made.

381. Various methods are available to quantify the soiling aspect of nuisance dust. These are discussed briefly in Section 5.6.

382. Ultrafine particle measurements quantify the number of particles below ~100 nm. These are described in Section 5.7. Chemical analysis of the PM can be carried out for a very wide range of components and chemical properties using many analytical techniques. Within the UK, PAHs, such as benzo[a]pyrene; metals, especially lead, nickel, arsenic and cadmium; and major ions, especially sulphate, are analysed routinely, generally in the PM_{10} size fraction. These measurements are described in Section 5.8.

5.2.2 Size-selective inlet heads

383. Both gravimetric and automated instruments, apart from those that use optical methods, rely on a size-selective inlet to exclude unwanted larger particles before the concentration of PM is quantified.

384. The selection is achieved using aerodynamic principles: a convoluted route forces the larger particles to move outside the path followed by smaller particles and they can then be prevented from travelling on to the filter. Designs use either an impactor, where the larger particles collect on to a flat plate, or a cyclone, where they collect on the inner surface of a ring.

385. Impactor designs need more regular cleaning to prevent the build up of piles of dust that would change the cut-off characteristic and more regular application of grease to prevent the larger particles bouncing off and continuing on to the filter.

386. No design can have an infinitely sharp cut-off, with all particles below 10 µm, for example, being let through and all particles above that size rejected. Inlets are designed so that 50% of particles of the critical size are rejected, but the curve of percentage rejected versus particle size varies between designs. Cyclones generally have a less sharp cut-off curve than impactors, and the reference design is, therefore, usually an impactor.

387. The cut-off characteristics depend on the speed of the air passing through the inlet, so that control and calibration of the flow is necessary for the correct size fraction as well as for the correct sampled volume. Some specific inlets are mentioned in Section 5.3.1.

5.3 PM_{10} monitoring methods

5.3.1 Filter-based gravimetric samplers

388. The EU First Air Quality Daughter Directive (1999/30/EC) specifies that measurements of PM_{10} should be carried out using the reference method, as defined in European Standard EN12341. This standard refers to three sampling devices that may be used:

- superhigh volume sampler – the WRAC (Wide Range Aerosol Classifier);
- high-volume sampler – the HVS PM_{10} sampler (68 $m^3\ h^{-1}$);
- low-volume sampler – the LVS PM_{10} sampler (2.3 $m^3\ h^{-1}$).

389. Each of these samplers consists of a PM_{10} sampling inlet that is directly connected to a filter substrate and a regulated flow controller. Following completion of the sampling period, the PM_{10} mass collected on the filter is determined gravimetrically. The filter is conditioned at 20°C and 50% relative humidity prior to weighing.

390. The WRAC sampler is usually regarded as a 'primary standard' and is not suitable for deployment in the general ambient environment. Within the UK, the low volume sampler (often referred to as the kleinfiltergerat or KFG) has been most widely used.

391. It is important to emphasise that the PM_{10} concentration measured by the reference method is simply that – it is not an absolute measure of PM_{10} mass in the atmosphere. The reference method will be subject to both positive (for example, due to an increase in particle-bound water) and negative artefacts (for example, due to loss of semi-volatile compounds) during sampling.

392. A number of samplers based on the low-volume sampler have been designed to allow automatic filter changing after each 24-h period, so that the instrument does not need to be visited each day. This includes a system that incorporates eight separate PM_{10} sampling heads connected to a central pump via a solenoid switching system (PNS-X8, IND) and a sequential sampler that uses an automatic exchange mechanism (SEQ47/50, Leckel). Although these types of samplers are more convenient to operate, there is the potential for an unquantified loss of semi-volatile compounds if the filter is exposed to the fluctuating conditions in the ambient environment following the end of sampling period.

393. There is a wide range of other filter-based gravimetric samplers available that are based on principles similar to the reference method. Within the UK, the Partisol sampler (R&P) has been used in a wide number of studies. The instrument operates at a lower sampling volume ($1\ m^3\ h^{-1}$) than the LVS reference sampler, and incorporates a USEPA-approved PM_{10} size selective inlet. The instrument is available as both a single filter sampler (Partisol 2000) and as a 16-day sequential sampler (Partisol 2025).

394. A number of UK research workers have also used the MiniVol sampler (Airmetrics). This is a portable sampler operating at $0.3\ m^3\ h^{-1}$ and is powered from a rechargeable battery, allowing it to be easily deployed in the field. The sampler may be configured to collect either PM_{10} or $PM_{2.5}$ samples depending upon the impactor that is selected. Mass determination is by subsequent weighing of the filter.

5.3.2 TEOM analysers

395. The TEOM analyser is widely used in both the UK and throughout the rest of the world for measuring continuous concentrations of PM. The instrument is based on the principle that the frequency of oscillation of a glass, tapered tube (element) changes by an amount that is directly proportional to the mass of the tube. Therefore, any change in mass of the tube, due to the deposition of particles onto a small filter affixed to one end, will result in a change in the resonant frequency – this change is proportional to the additional mass.

396. Air is sampled at $16.7\ l\ min^{-1}$ through a USEPA-approved PM_{10} size selective inlet. A sample splitter divides the flow, carrying $3\ l\ min^{-1}$ to a 16 mm PTFE-coated quartz filter. Both the manifold inlet and tapered element are maintained at 50°C to minimise errors associated with the evaporation and condensation of water on the filter.

397. In order for the TEOM to be used as a USEPA-equivalent method for PM_{10} measurement, a default adjustment factor ($1.03 * \text{TEOM reading} + 3\ \mu g\ m^{-3}$) must be applied to the raw data. This adjustment factor was derived to account for moisture equilibration differences between the TEOM and the HI-vol sample media. The adjustment factor was determined at sites where non-volatile PM

Table 5.1 Summary of advantages and disadvantages of principal PM sampling methods.

Technique	Advantages	Disadvantages	Estimated precision[a]
Filter-based gravimetric samplers	The reference method for PM_{10} specified in the EU First Daughter Directive.	High operating costs. Time resolution of the measurement is limited to 24-h. Reporting requirements of the EU First Daughter Directive cannot be met and results can only be provided some days after the sample was collected.	±2µg m^{-3}
TEOM analysers	Provide real-time data with short time resolution (<1 h) that can be used for public information. Improved precision compared to the reference method.	Preheated air stream causes a greater loss of semi-volatiles compared to the reference method. High capital cost.	± 0.5 µg m^{-3}
ß-attenuation analysers	Provide real-time data with short time resolution (<1 h) that can be used for public information.	If a heated inlet is used some semi-volatile material may be lost. Unheated samplers may suffer from interference due to the presence of water. Analyser contains a radioactive source.	±3 µg m^{-3}, but depends on analyser type
Optical analysers	Portable and often battery operated. Can measure several size fractions simultaneously.	Depends on assumptions about particle characteristics, which may vary from place to place and time to time.	Depends on analyser type
Black smoke	Simple, robust, inexpensive and easy to maintain. Long time series of existing data.	Measures an index rather than a gravimetric concentration. Calibration factor not appropriate to the current mix of pollution sources. Time resolution of the measurement is limited to 24-h.	±2 µg m^{-3}, may be higher at current typical concentrations
Personal samplers	Portable samplers that can easily be deployed in the field and used to determine personal exposure to particulate concentrations.	As above, depending on measurement method used. All personal exposure analysis is very labour-intensive.	According to technique employed, as above

[a] The precision stated refers to when the sampler is used to determine PM_{10} mass over an averaging period of 24-hours.

dominated and is intended to reflect the filter character more than the PM. It is understood that USEPA has no general policy on the use of this empirical adjustment factor for $PM_{2.5}$ measurements. All TEOM analysers in the UK measuring both PM_{10} and $PM_{2.5}$ are currently set up with this default adjustment factor included. In addition, TEOM analysers within the UK networks are set to report concentrations corrected to 293K and 101.3 kPa.

398. Due to the need to eliminate the effect of changing humidity on the mass measurement, the TEOM is required to maintain the sample filter at an elevated temperature. This has led to reported differences in concentrations of PM between the TEOM and the European reference sampler (Allen *et al*., 1997; APEG, 1999; Ayers *et al*., 1999; Soutar *et al*., 1999; Salter and Parsons, 1999; Cyrys *et al*., 2001; Williams and Bruckmann, 2001). This is largely attributed to the loss of volatile species such as ammonium nitrate. As an interim measure, a default 'scaling factor' (also known as correction factor) of 1.3 is currently applied to all nationally reported TEOM PM_{10} data in the UK as recommended by the EC Working Group on Particulate Matter (2001).

399. To attempt to overcome these problems, two innovations to the TEOM have become available in recent years. A TEOM analyser equipped with a Sequential Equilibration System (SES) incorporates a Nafion dryer, which reduces the relative humidity of the sample stream. This allows the instrument to operate at the lower temperature of 30°C, thereby reducing the loss of volatile components.

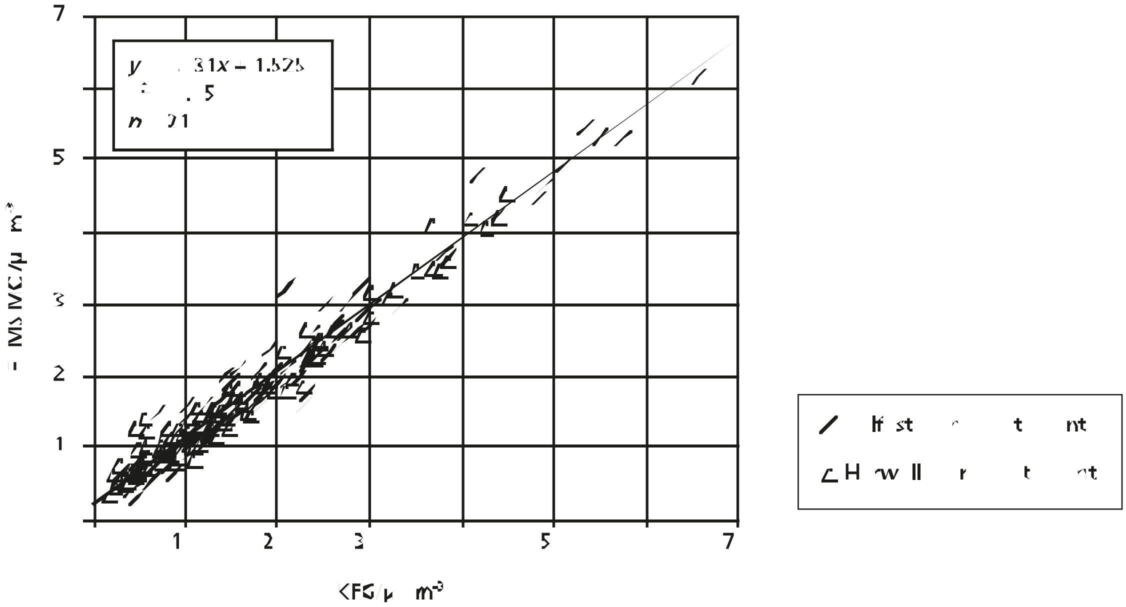
400. A more recent development, the Filter Dynamics Measurement System (FDMS) independently measures the volatile component of the incoming air sample. The incoming air stream passes through a size selective inlet and a SES dryer before it is alternately switched every six min between the TEOM microbalance sensor unit and a purge filter system held at 4°C. The purge filter effectively removes aerosols from the sample stream before it passes to the sensor unit. The system then adjusts the final mass concentration by reference to any mass change that occurred during the purge cycle, for example, if a decrease in mass was measured during the purge cycle, then this would be added back to the mass measurement reported with particle-laden air.

401. FDMS analysers have been deployed on a trial basis at two locations in the UK: at Belfast (from 1 October 2003) and Harwell (from 20 November 2003). Figure 5.1 shows a comparison of the 24-h mean PM_{10} concentrations measured by the FDMS with a co-located reference sampler (KFG) up until May 2004. Regression analysis was performed using an orthogonal approach. These data form a preliminary assessment, and there is an intention to include FDMS analysers in the detailed equivalence programme described in Section 5.4.3.

5.3.3 ß-attenuation analysers

402. ß-attenuation particle analysers are the most widely used method for measuring ambient PM_{10} concentrations in national networks across Europe (see Annex 4). However, they are less widely employed within the UK networks, where use of the TEOM analyser is more common (Section 5.10.). The ß-attenuation analyser uses measurement of the reduction in intensity of ß particles passing through a dust-laden filter. This reduction in intensity is due to absorption of ß particles by the dust collected (and the filter material). The relationship between radiation absorbed and mass of dust collected closely follows an exponential relationship

Figure 5.1 Comparison between FDMS and reference KFG samplers at Belfast and Harwell.



that is reasonably independent of the chemical composition of typical PM found in the atmosphere. Some ß-attenuation monitors are calibrated using foils based on quartz dust.

403. Although the exact configuration varies from analyser to analyser, the sample is collected onto a filter or filter tape and a radioactive source (typically carbon 14, krypton 85 or promethium 147) is positioned above the sample, with a detector below to measure the radiation absorbed. In some units the particle mass can be monitored continuously, whereas in others the sample is collected for a discreet period (typically 30 or 60 min) before the mass is determined. Absorption due to the clean filter is accounted for either by the sequential measurement of clean and exposed filters or using a dual chamber arrangement.

404. In addition to the mass measurement, measurement of the volume of air sampled is necessary to calculate particle concentration. The determination of volume is normally achieved by the use of flow controllers and flow measurement devices, as in other automatic monitors.

405. Particle size selection is determined by the inlet and the flow rate used. Hence, ß-attenuation analysers can measure total suspended particles (TSP), PM_{10} or $PM_{2.5}$ (or other size fractions), depending on the inlet head used. Although the filter material in ß-attenuation monitors is not generally heated, in some analyser's configurations the inlet system is heated to reduce relative humidity in the sample and, therefore, minimises the water content of the aerosol mass. However, this is likely to lead to loss of semi-volatile material, as in the TEOM.

406. Studies of intercomparisons of ß-attenuation analysers and gravimetric samplers have been reviewed (EC Working Group on Particulate Matter, 2001). The report concluded that the factor relating ß-attenuation analysers to the reference gravimetric sampler varied from study to study and depended on whether the sample was heated or not, the analyser calibration and the season (winter or summer). However, in general, the factor was close to one in systems without heated inlets and approximately 1.3 with heated inlets during the winter season and closer to one in the summer season.

407. Monitoring with an unheated ß-attenuation analyser (the Met One beta-Attenuation Monitor, or BAM 1020) is carried out by a number of local authorities in the UK. Although no systematic long-term collocation studies comparing the BAM, TEOM and reference sampler have been carried out, consistent evidence shows that the BAM measures significantly higher PM_{10} concentrations than the reference sampler or the TEOM * 1.3 (see Annex 5).

408. This effect is also demonstrated in Section 6.2.3, where both TEOM * 1.3 and BAM data have been used to generate a concentration surface across London. It is evident that the concentrations generated by the BAM measurements are anomalous compared with the TEOM * 1.3 data.

5.3.4 Optical analysers

409. Optical particle monitors utilise the interaction between airborne particles and visible, infrared or laser light.

410. Nephalometer and/or transmissometers are commonly used in the USA to determine visibility loss due to airborne particulate material, especially in national park areas. Transmissometers operate over a long open path of 1–10 km and measure total light extinction by determining the loss of light (due to scattering and absorption of the intervening atmosphere) from an artificial light source of known luminescence. Nephalometers, which operate over a short closed path, measure light scattering, which is responsible for the majority but not all of the total light extinction.

411. Of more relevance to measuring particle concentrations, some newer light scattering devices use laser light scattering to both count and size classify PM. The advantage of this approach is that a single analyser with a TSP inlet can monitor particles of a range of sizes, including PM_{10}, $PM_{2.5}$ and PM_1, simultaneously. However, the particle counts need to be converted to mass by calculation using a range of assumptions whose validity may vary due to the nature of the particles sampled. In an attempt to overcome this problem, some analysers also collect the particles sampled onto filters for subsequent gravimetric analysis. These results can then be used to 'calibrate' the optical results for the particular application.

412. Often these analysers are small, lightweight and battery operated and hence are readily portable and very useful for short-term measurements and 'screening' of a number of locations to build up knowledge of particle distribution in both space and time.

413. Optical monitors are not used in the national monitoring networks in the UK, although particle counters – which utilize optical techniques (see Section 5.6) – are used in research programmes for Defra and the Devolved Administrations. Portable optical particle monitors are often used in screening studies and for workplace and indoor particle monitoring programmes.

5.3.5 Black smoke method

414. The methodology for monitoring particulate material as black smoke was developed more than 50 years ago, when domestic smoke was a major cause of particle pollution in the UK. The method is fully described in ISO 9835. PM is collected on a filter paper over a 24-h period and the darkness of the stain is measured by a reflectometer. The inlet size (30–50 mm diameter), sample tube (<8 m of 6.5 mm PVC tube) and the sample flowrate (2 m^3 per day) define the fraction of material collected, and this is generally taken as approximately PM_4. The monitoring equipment is relatively simple, robust and inexpensive.

415. The 'darkness' of the stain is measured on a standard reflectometer. The darkness reading is converted to a measure of particulate material in the atmosphere by reference to one of two standard calibration curves: the OECD curve and the British Standard curve. The calibration curves convert reflectance measurements to $\mu g\ m^{-3}$ of black smoke, which can then be converted to black smoke concentration ($\mu g\ m^{-3}$)[1].

416. In practice, however, no meaningful universal relationship can be derived between reflectance and black smoke since the relation can change from place to place and from time to time. The 'concentrations' of black smoke derived from this method should, therefore, be considered to be an index of black smoke rather than an absolute measure in mass terms. The precision of the method is high, typically about 2 $\mu g\ m^{-3}$ for concentrations >5 $\mu g\ m^{-3}$. However, many concentration measurements in the UK are now lower than this and are likely to be less precise.

417. Despite the shortcomings of the methodology, many epidemiological studies still find clear links between black smoke concentrations and the effects on human health (see Chapter 3). In addition, there is a vast body of black smoke measurement data in the UK and throughout the world, stretching back to at least 1961. In the UK, over 1000 monitoring sites operated during 1962–1980; at the present time, 123 sites are still in operation. This provides a rich source of research data. Section 5.10.2 gives further information on the UK Black Smoke Monitoring Network.

5.3.6 Personal samplers

418. An increasingly wide range of measurement techniques for various metrics of PM are being applied to personal exposure analysis. The requirement of exposure analysis is that sampling should be from the breathing zone of an individual, which extends approximately 30 cm from their nose and mouth, and moves with them from one microenvironment to the next. Portable equipment ranges in size and weight from passive badge samplers (Brown *et al*., 1995; Wagner and Leith, 2001), although these techniques have not been very widely used, to high-volume pumps, 16 l min^{-1} from a pump weighing ~3 kg (Adams *et al*., 2001a). Low-volume active sampling is used for integrated measurement over daily or longer averaging times (Jantunen *et al*., 1998), whereas high-volume sampling allows shorter times to be investigated, such as commuting periods (Adams *et al*.,

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[1] The British Standard curve is used within the UK, except when results are reported to the EC, when they are reportedin OECD units, as required by Directive 80/779/EEC(British Standard= OECD Standard * 0.85).

2001b). Particle size selection for these filter sampling techniques is achieved by conventional impactor or cyclone techniques or specially designed more lightweight methods such as the conical inhalable sampler using treated foam to remove particles larger than 10 or 2.5 µm at a given low or high flow rate. Condensation particle counters (for example, TSI P-Trak) and optical particle counters (for example, Grimm) are portable and may be used for personal exposure measurement. They also have the advantage of providing time resolutions as high as 1 Hz, if required, to identify the frequency, duration and severity of peak exposures and higher moments of the personal exposure distribution function.

5.4 Comparison of PM_{10} monitoring methods

5.4.1 Introduction

419. The reference method describes a uniform approach to the measurement of PM_{10} concentrations, but provides a number of practical difficulties for monitoring, particularly within national networks. Filter-based gravimetric samplers rely heavily on human resources for servicing, particularly if the filters are exchanged on a daily basis, and for subsequent weighing of the filters. The quality assurance must be more strictly enforced, simply due to the additional potential pathways for error (for example, sample handling, transfer, storage and weighing). In addition, as the filters have to be taken to the laboratory for weighing, there are several days delay (as a minimum) between sample collection and reporting. This is disadvantageous to the protection of public health and contravenes the reporting requirements of the Daughter Directive.

420. For these reasons, monitoring of PM in the UK has been largely founded on the TEOM analyser (as described in Section 5.10). Studies to investigate the relationship between the TEOM analyser and the reference method are ongoing and are described in Section 5.4.3 below.

5.4.2 EU Working Group paper on equivalence

5.4.2.1 Background

421. For all air pollutants covered by European Ambient Air Directives, the Directives specify a reference method for carrying out the measurements. These methods are, in principle, set out in sufficient detail to meet the Data Quality Objectives of the Directives within European Committee on Standardisation (CEN) standards, many of which exist in draft form and are due for publication in 2005. PM_{10} measurement is an exception to this, as the CEN standard EN 12341 was published in 1998 (European Committee on Standardisation (CEN), 1998), before the relevant Ambient Air Directive, and it, therefore, does not explicitly address the Data Quality Objectives for PM_{10}.

422. At the same time as specifying a reference method, the Ambient Air Directives also contain the statement:
'A Member State may use any other method which it can demonstrate gives results equivalent to the above method.'

423. For PM_{10} this is extended by the statement:
'... or any other method which the Member State concerned can demonstrate displays a consistent relationship to the reference method. In that event the results achieved by that method must be corrected [scaled] by a relevant factor to produce results equivalent to those that would have been achieved by using the reference method.'

424. For non-reference methods for all pollutants and, in the case of PM_{10} also for non-reference methods with a scaling factor, there is a need to be very clear about what is meant by results being 'equivalent' to those of the reference method. The EC set up a Working Group to produce a guidance document covering this issue. A final draft document was published in 2004 (EC Working Group on Guidance for the Demonstration of Equivalence, 2004).

5.4.2.2 The equivalence procedure for PM_{10} in EN 12341

425. The CEN standard EN 12341 contains a procedure for determining whether non-reference instruments are equivalent to the reference method. However, the procedure states that it is not to be used to determine the equivalence of automated methods. It also uses relatively crude statistical procedures to assess field trial data, and the equivalence criteria are not directly linked to the Data Quality Objectives. It is, therefore, not greatly helpful for current or future purposes of complying with the Ambient Air Directive. It is expected to be replaced in due course with an equivalence procedure based on the EC Guidance Document, described below.

5.4.2.3 The equivalence procedure for PM_{10} in the draft EC Guidance Document

426. The draft EC Guidance Document on Equivalence (EC Working Group on Equivalence, 2003) has the general aims of determining whether candidate (non-reference) methods are capable of fulfilling the Data Quality Objectives (especially measurement accuracy/uncertainty) and of ensuring that only suitable methods are employed for official reporting purposes with sufficient ongoing quality control to ensure that the Data Quality Objectives are met during routine use.

427. In principle, equivalent methods should not require less rigorous implementation than the reference method, whose requirements are set out in detail in CEN standards.

428. For PM (both PM_{10} and $PM_{2.5}$), the draft EC Guidance Document specifies two approaches for determining whether a candidate method is equivalent, depending on whether the method differs in minor or major ways from the reference method. Where minor variations – such as non-standard storage of filters or non-standard filter weighing conditions within manual methods, otherwise similar to the reference method – are proposed they can be evaluated by laboratory trials that determine the worst case effect on the measurements.

429. Major variations, such as any automatic methods, must be evaluated from field trials involving side-by-side reference and candidate instruments. The draft document proposes:

- two reference instruments;
- two candidate instruments;

- operation of the candidate instruments as they would be at a standard field site;
- at least four distinct field comparisons containing at least 40 measurement days each; and
- adequate coverage of the full range of expected measurement conditions in the field – PM concentration, PM composition, temperature, humidity and wind speed.

However, the draft guidance document also allows that data from previous parallel measurements with two reference instruments may be taken into account (obviating the need for simultaneous measurements of both two reference and candidate instruments) where that previous data is obtained under equivalent conditions to those given in the Guidance document.

430. Equivalence can only be declared for the range of conditions covered during the field tests. For instruments without a scaling factor, it is determined by quantitative tests covering:

- agreement between the duplicate reference instruments (to provide a check that they have been used correctly);
- agreement between the duplicate candidate instruments (to check repeatability);
- the regression slope between the average reference measurement and the average candidate measurement (which must be close to one); and
- the uncertainty of individual candidate measurements – a combination of repeatability, the uncertainty of the reference method, deviation of the regression line from ideal slope and intercept and the deviation of individual data points from the regression line – which is compared to the Data Quality Objectives.

431. These tests must be passed for the entire set of measurement data, and also for the distinct comparisons separately, so that anomalous performance in one set of conditions will not be masked by other more favourable conditions.

432. In line with the special allowance for scaling factors to be applied to PM instruments, all the candidate data may be adjusted for a constant offset, a constant factor or both, which would then be applied in the same way to all data collected for official monitoring. After data have been scaled, the tests listed above must be passed, again both for the entire dataset and for the distinct comparisons separately. In addition, an extra term relating to the correction is added to the uncertainty of the individual measurements.

5.4.3 The UK PM_{10} intercomparison exercise

433. An intercomparison study between the TEOM analyser and the reference sampler commenced in 1999 (Casella Stanger, 2004). It is important to note that this study predated the draft EC Guidance Document described above.

434. The study focused on six monitoring sites within the UK Automatic Urban and Rural Network (see Section 5.9), representative of the geography, particle concentrations and sources in the UK. These were:

- Marylebone Road (Kerbside – London);
- Thurrock (Urban Background – SE England);
- Harwell (Rural – Oxfordshire);
- Port Talbot (Industrial – Wales);
- Glasgow Centre (Urban Centre – Scotland);
- Belfast Centre (Urban Centre – Northern Ireland).

435. The conclusions of the study are that the TEOM analyser generally reports lower PM_{10} concentrations than the reference sampler. The ratio of TEOM and reference sampler concentrations varies from site-to-site and from season-to-season. The lowest seasonal ratios (close to unity) are consistently observed at Port Talbot (presumably influenced by local emissions from the nearby steelworks) and the highest ratios (approximately 1.4 to 1.5) are observed at Thurrock. In general, higher ratios are observed in the winter than in the summer.

436. The application of the 'default' 1.3 scaling factor that is currently applied to TEOM data in the UK provides a variable level of agreement with the reference sampler. The default factor provides a reasonable result for many sites when long-term averages are considered, but does not give an accurate indication of the number of exceedence days.

437. Regression analysis (based on Reduced Major Axis) indicates R^2 values generally within the range of 0.6 to 0.9 (examples are shown in Figure 5.2), although the relationships are both site and seasonally dependent. The poorest relationship has been consistently observed at Harwell, with R^2 values in the range of 0.36 to 0.63. An analysis of time series data indicates that, in general terms, the TEOM and the reference method compare reasonably well on a day-to-day basis for much of the time, but measured levels may diverge significantly particularly when concentrations are elevated.

438. Although the intercomparison study meets the general requirements of the draft EC Guidance Document in terms of the range of expected environmental conditions and the number of field comparisons, the study has not included side-by-side reference and candidate instruments. It has, therefore, not been possible to determine the ‘between-instrument’ uncertainties. However, in order to provide an initial indication as to how the TEOM analyser might perform, and to inform the requirements for any future work, the data from the intercomparison study have been evaluated using the statistical approach in the draft Guidance Document and assumptions[2] for 'between-instrument' uncertainties.

439. The expanded uncertainty of the full dataset following scaling is 30.5%, which exceeds the requirement for measurement uncertainty (25%) stated in the

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[2] For the KFG reference sampler this was assumed to be the maximum allowable uncertainty of ≤ 1 µg m^{-1}.

Daughter Directive, and the TEOM analyser fails the test. This conclusion must be considered with the following points in mind:

- The reference sampler used in this study was fitted with an automatic filter switching mechanism to avoid daily site servicing. Although this was consistent with general network operations, there is the potential for losses of semi-volatile material post-sampling, as the filters remain exposed in the ambient environment for up to seven days.

- Filters were transported to the laboratory by post, which may have incurred further losses of semi-volatiles.

- Given the potential for these losses, it has not been possible to conclude whether the stringent criterion for the 'between-instrument' uncertainty of the reference sampler has been met.

440. In light of these results, Defra started a more detailed evaluation exercise in 2004, which will include a wide range of co-located candidate samplers. It is expected that preliminary results will be reported in late 2005.

441. Notwithstanding the outcome of the proposed evaluation exercise, it is noted that if the reference method were changed so that only the non-volatile component of the particle mass was determined, this would lead to a great improvement in both the consistency and practicality of measurements of PM for regulatory purposes. In the event that the semi-volatile, or any other mass component not within this 'core particulate' were deemed important, then these could be measured by additional specific monitoring techniques at a representative number of sites.

5.5 Measurement of $PM_{2.5}$ (and PM_1) concentrations

5.5.1 Methods of measurement

442. The $PM_{2.5}$ fraction of PM differs from the PM_{10} fraction solely in the size of the particles included. The fundamental difference in methods for $PM_{2.5}$ compared to PM_{10} measurement in most cases is, therefore, just the design of the size-selective inlet. The methods used for PM_{10}, described in Section 5.2 can, therefore, be adapted for use with $PM_{2.5}$ with no other changes[3]. The exceptions to this are instruments that use optical methods, which determine the size fraction by means other than a size-selective inlet. However, optical instruments for $PM_{2.5}$ are essentially similar to those for PM_{10}, with commercial instruments capable of monitoring the different size fractions in a rapid cycle.

443. The size-selective inlets for $PM_{2.5}$ measurements are generally designed to operate at the same sampling flow rate as those for PM_{10}, so that the instruments can be converted for the different size fractions with minimal effort.

444. In practice, the measurement of $PM_{2.5}$ differs from PM_{10} because there will be a smaller quantity of PM to measure, and the coarse fraction, which tends to consist of windblown dust and other non-volatile material, is excluded. Both these factors make accurate $PM_{2.5}$ measurement more difficult than PM_{10} measurement.

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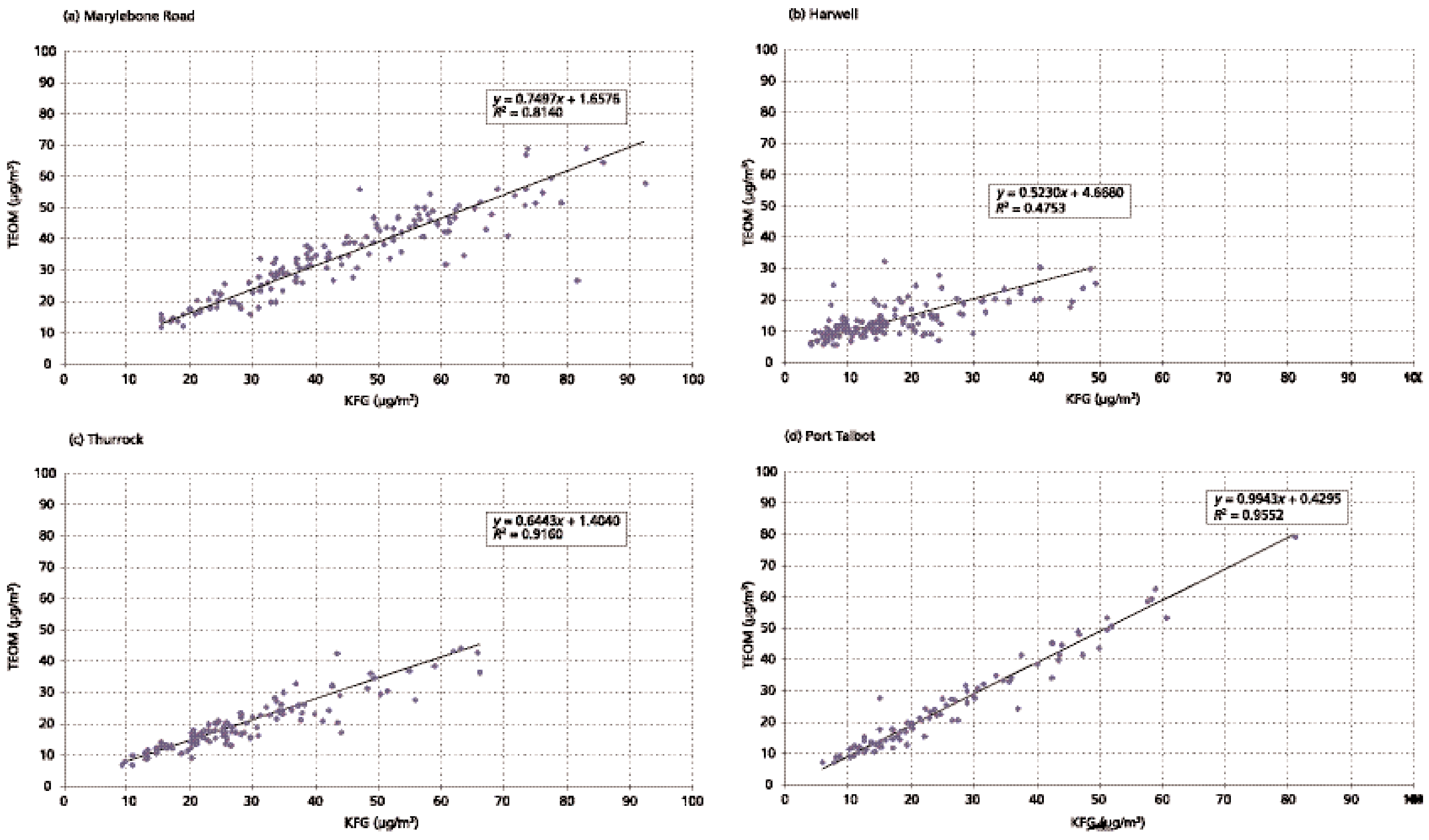
[3] As described in Sections 5.3.2, all TEOM analysers measuring $PM_{2.5}$ are set up with the USEPA default adjustment factor (TEOM *1.03 +3 µg m^{-3}).

Figure 5.2 Regression (Reduced Major Axis) plots for the four sites used in the UK intercomparison study (October 1999 to March 2000). At the mean TEOM concentrations measured at each site in 2002, the regression equations below represent gravimetric:TEOM factors of 1.27 (Marylebone Road), 1.20 (Harwell), 1.45 (Thurrock) and 0.99 (Port Talbot).



445. Instruments for measurement of PM_1, a yet smaller size fraction, follow the same principles as those for $PM_{2.5}$ and PM_{10} measurement.

5.5.2 Development of CEN reference method for $PM_{2.5}$

446. The European Ambient Air Directives require Member States to make measurements of $PM_{2.5}$ and the EC mandated the standardisation body CEN to specify the reference method to be used. In line with the European reference method for PM_{10} measurement, the method was required to be based on the collection of the $PM_{2.5}$ fraction of ambient PM on a filter, with mass determination by weighing.

447. The standard will, therefore, include one or more designs of size-selective inlet, with associated parameters for flow control; specifications for conditioning, transporting and weighing the filters; additional quality assurance and quality control requirements; and a methodology for calculating the measurement uncertainty. It will also contain a procedure for determining whether non-reference methods give equivalent results to those of the reference method, which will closely follow the EC Guidance Document on Equivalence.

448. Apart from the different inlet design(s), the major operational differences from the PM_{10} reference method, EN 12341, are expected to involve stricter measures to minimise losses of semi-volatile PM from the filter during and after sampling and other filter artefacts affecting the measured mass, issues that were not fully appreciated when EN 12341 was written.

449. The draft standard prEN 14907 was completed in December 2003 and is passing through a process of consultation, revision and voting by Member States. The process is coordinated through national standardisation bodies, BSI in the UK.

450. During the drafting of the standard, a major series of field trials involving candidate reference instruments – operated under the proposed stricter conditions of the new standard – together with selected automatic instruments, was carried out. Nine locations across Europe were used, including Teddington in SW London. This location also included a TEOM FDMS instruments. The initial results of these trials have not yet been published.

5.6 Measurement of nuisance dust

451. Nuisance dusts may be generated by a wide range of activities, including road traffic, construction and demolition work, mineral extraction processes and industry. These emissions can cause soiling of surfaces such as car bodywork and window ledges and can give rise to considerable public concern. There is no precise definition of nuisance dust, or how to measure it, and no standard criterion for assessment has been established.

452. A variety of monitoring techniques have evolved. These involve the determination of the mass of dust deposited over a given area or determination of the soiling of a surface by reduction in surface reflectance or gloss. A more detailed description of these methods is given elsewhere (for example QUARG,1996).

5.7 Measurement of ultrafine particles

5.7.1 Particle number measurements

453. Ultrafine particles are commonly defined as those particles that are <0.1 µm in diameter. Such particles contribute very little to particle mass, but contribute significantly to the total number of particles. It is common practice to measure ultrafine particles in terms of the particle number concentration. The conventional light scattering instruments normally used to measure particle number concentrations cannot detect ultrafine particles. Within the UK, the Condensation Particle Counter (CPC, TSI) has been used in a number of studies. The CPC is based on the principle that supersaturated vapour will condense on small particles. The sampled aerosol passes through a chamber that is saturated with *n*-butyl alcohol vapour and then to a cooled condenser where the alcohol condenses onto the particles, which causes them to grow in size. The particles are then counted in an optical detector. Depending upon the configuration of the CPC, the analyser is capable of measuring particles within the range of 0.003–2.0 µm. Within the UK, measurements within the range ~0.007–2.0 µm have been routinely carried out.

454. Size-fractionated particle number concentrations may be carried out using a Scanning Mobility Particle Sizer (SMPS, TSI). The SMPS uses a ^{85}Kr ß particle source to impart a bipolar equilibrium charge on the polydisperse aerosol before passing to the electrostatic classifier of a differential mobility analyser (DMA), where the particles are separated according to their electrical mobility. By varying the air flow rate and the voltage in the DMA, the monodisperse aerosol exiting the classifier can be controlled to within a narrow, specified size range. The monodisperse aerosol then enters a CPC where the particle concentration is determined. The size ranges that can be measured using the SMPS are dependent upon the equipment configuration and user-defined parameters. Within the UK, measurements within the range 11–450 nm have been routinely carried out.

5.7.2 Measurement of particle surface area

455. The measurement of particle surface area is not straightforward, principally because airborne particles have a complex morphology. Two main approaches have been used. The first method attaches labelled atoms to the particle surface and then measures the mass transfer. The principal drawback of this method is that the rate of attachment within a polydisperse aerosol is not directly related to the particle surface area: there will be a transition from the molecular bombardment regime for particles smaller than the mean free path of the gas molecules to a diffusion-limited regime for the larger particles. The epiphaniometer is based on this approach and attaches gaseous ^{211}Pb atoms onto the aerosol particles, which are then collected onto a filter and determined by an α-counter. The measured signal is proportional to the exposed Fuchs surface of the aerosol particles (Fuchs, 1964).

456. The second method is to use measurements of particle number size distributions, which may then be converted to surface area based on assumptions regarding particle geometry. This may, for example, include measurements carried out using the SMPS for the smaller particles, in tandem with a suitable instrument for measuring the larger particle size fractions (such as the Aerodynamic Particle Sizer, TSI).

5.8 Measurement of metals, PAHs and major ions

457. Samples of PM collected using the techniques described in Section 5.2 can be subjected to chemical analysis for the subsequent determination of metals, PAHs and major ions (most commonly sulphate). These days, measurements of metals and sulphate are based on the PM_{10} size fraction, although historically, monitoring of these species in the UK were based on the M-type sampler[4]. PAH monitoring is currently based on use of a modified pesticide sampler, which also has a cut-off of approximately PM_{15}. When the 4th EU Daughter Directive is agreed and implemented, it is likely that the network will need to change to PM_{10}-based sampling.

5.8.1 Metals

458. Particles containing the analytes nickel, arsenic, cadmium and lead are sampled by drawing a known volume of ambient air through a cellulose filter held within a high- or low-volume sampler. The filter is dissolved in a solution of HNO_3 and H_2O_2 by microwave digestion in a closed vessel. This sample is then analysed for metals content by inductively coupled plasma-mass spectrometry (ICP-MS).

459. ICP-MS is a technique that employs an argon plasma to ionise a sample as it is introduced. The ions produced are then analysed by a mass spectrometer. The ICP-MS analyses all elements of interest effectively simultaneously and can also automatically correct for molecular and isobaric interferences.

5.8.2 PAH

460. Samples are collected using polyurethane foam plugs and filter papers to capture PAHs in the volatile and non-volatile phases, respectively. The fortnightly samples are extracted using dichloromethane and then bulked to give a single sample. The concentrations of 32 individual PAHs are determined from the quarterly samples using gas chromatography with mass spectrometric detection.

5.8.3 Major ions

461. Samples are collected onto filters (generally cellulose acetate) with subsequent aqueous extraction and analysis by ion chromatography. In some sampling systems, a denuder is incorporated into the sample stream to remove potential interference from acid gases.

5.9 Continuous monitoring of speciated PM

462. Conventionally, measurements of speciated PM are carried out by subsequent chemical analysis of the exposed filter. However, an increasing number of direct-reading analysers are now available, which permit continuous measurements for

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[4] The performance of the M-type sampler is strongly windspeed-dependent, although at average windspeeds for the UK, the sampling is close to that of a PM_{10} sampler.

a range of particulate species including nitrate, sulphate and elemental/organic carbon. These instruments have the advantage of providing measurements over much shorter time resolutions (1 h or less) and can provide valuable data in tracking diurnal trends and pollution episodes.

463. Carbon particulate analysers (for example, Series 5400, R&P) are based on a thermal CO_2 analysis technique similar to that used in many laboratories. Ambient air passes through a size-selective inlet and the particulate material is collected onto a cartridge contained within a temperature-regulated oven. Differentiation between the organic and elemental carbon component is achieved by oxidising the sample at an intermediate temperature (340°C for organic carbon) and a final high burn temperature (750°C for elemental carbon). The analyser is capable of providing data with a 2-h time resolution.

464. Several analysers are available for the continuous analysis of particulate nitrate and sulphate (Series 8400N, R&P; 9000A Ambient Ion Monitor, URG). Ambient air is drawn through a size-selective inlet and then passes through a denuder to remove interfering acid gases. To achieve a high collection efficiency of very small secondary aerosol, the particles are then enlarged by passage through a humidifier or saturation chamber. The final determination is then by flash volatilisation and measurement of NO_X or SO_2 or by injection into an ambient ion detector.

5.10 The UK monitoring networks

465. Measurement of the mass of PM in the atmosphere (mainly as PM_{10}, but including some $PM_{2.5}$ monitoring) is carried out on both a national and regional scale using a range of monitoring techniques. National networks are also established to monitor black smoke and ultrafine particles and to collect and analyse particulate material for the determination of PAHs, metals and sulphate concentrations. These networks are described in the following sections.

5.10.1 Monitoring networks for PM_{10} and $PM_{2.5}$

466. Networks for monitoring PM are operated at both national level, on behalf of Defra and the Devolved Administrations, and at regional and sector-specific levels. The vast majority of these sites monitor PM_{10}, although a few also monitor $PM_{2.5}$. The TEOM analyser is by far the most common method of measurement in the UK, although some sites use sequential gravimetric samplers and some attenuation analysers.

467. The locations of all sites that were operational in December 2003 and have provided PM data for this study are shown in Figure 5.3. The structures of the various monitoring networks are summarised in Table 5.2.

468. The national research network provides additional monitoring of PM_{10} and $PM_{2.5}$ with a range of gravimetric samplers at nine sites that are part of the AURN and LAQN National Monitoring Networks (see Section 5.9.1.3).

5.10.1.1 UK networks

469. The UK networks are comprised of monitoring sites within both the AURN and the LAQN. The AURN contains a number of monitoring sites that are directly

Table 5.2 Summary of PM monitoring networks (2003).

Networks	Managment unit	Quality assurance and control responsibility	Number of PM_{10} monitoring locations	Number of $PM_{2.5}$ monitoring locations
UK networks				
AURN	Cesella Stanger	Netcen	66	3
LAQN[a]	Kings College ERG	Netcen	5	1
Regional networks				
LAQN				
KAQM	King's College ERG	NPL/ERG	64	5
HBAPMN	King's College ERG	NPL/ERG	18	0
SAQSG	King's College ERG	NPL/ERG	14	0
	King's College ERG	ERG	8	0
Other networks				
Birmingham	Birmingham CC	Birmingham CC	4	1
Calibration Club	Netcen	Netcen	41	2
Highways Agency[b]	TRL	TRL	5	2
Heathrow Airport[c]	Netcen	Netcen	2	0
JEP/Innogy	Casella Stanger	Casella Stanger/NPL	2	1
JEP/Powergen	Power Technology	Power Technology	2	0
Defra/Developed Administrations research	Casella Stanger	Casella Stanger	9	8

Key: KAQN, Kent Air Quality Network; HBAPMN, Hertfordshire and Bedfordshire Air Pollution Monitoring Network; SAQSG, Sussex Air Quality Steering Group; Birmingham CC, Birmingham County Council; Kings College ERG, Kings College Environmental Research Group This table summarises information from the PM monitoring networks used in this report. It is not a complete list of all networks in the UK. [a]A number of sites in this network have been affiliated into the national networks. [b]The Victoria Street site was operated for the Highways Agency up to 2001. From 2003 onwards, this site has been operated for TfL. [c]The Heathrow Airport site is also part of the LAQN and the Heathrow Harlington site is also part of the AURN.

funded by Defra and the Devolved Administrations, together with additional sites that are owned and operated by other organisations (mainly local authorities) and which have been affiliated into the network. The LAQN consists solely of affiliated local authority monitoring sites. The first PM_{10} measurements at six sites in the UK commenced in 1992 and through the years, the number of monitors has steadily increased to the present number of 71. Initially all sites used TEOM analysers, although one ß-attenuation analyser was introduced into the network in 1998[5]. During 2001–2002 additional sites were established in order to meet the requirements of the First Daughter Directive and sequential gravimetric analysers were installed at these seven new PM_{10} monitoring sites. Thus, although some sites have been operating for 10 years or more and can provide invaluable information on long-term trends over this period, data are limited to a considerably shorter period at the majority of monitoring sites. In 2003, four automatic $PM_{2.5}$ monitoring sites (see Section 5.9.1.4) were incorporated into the national network.

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[5] A Met One ß-attenuation monitor (BAM 1020), which has an unheated inlet.

Figure 5.3 The Location of (a) PM_{10} and (b) $PM_{2.5}$ 2003 monitoring sites.

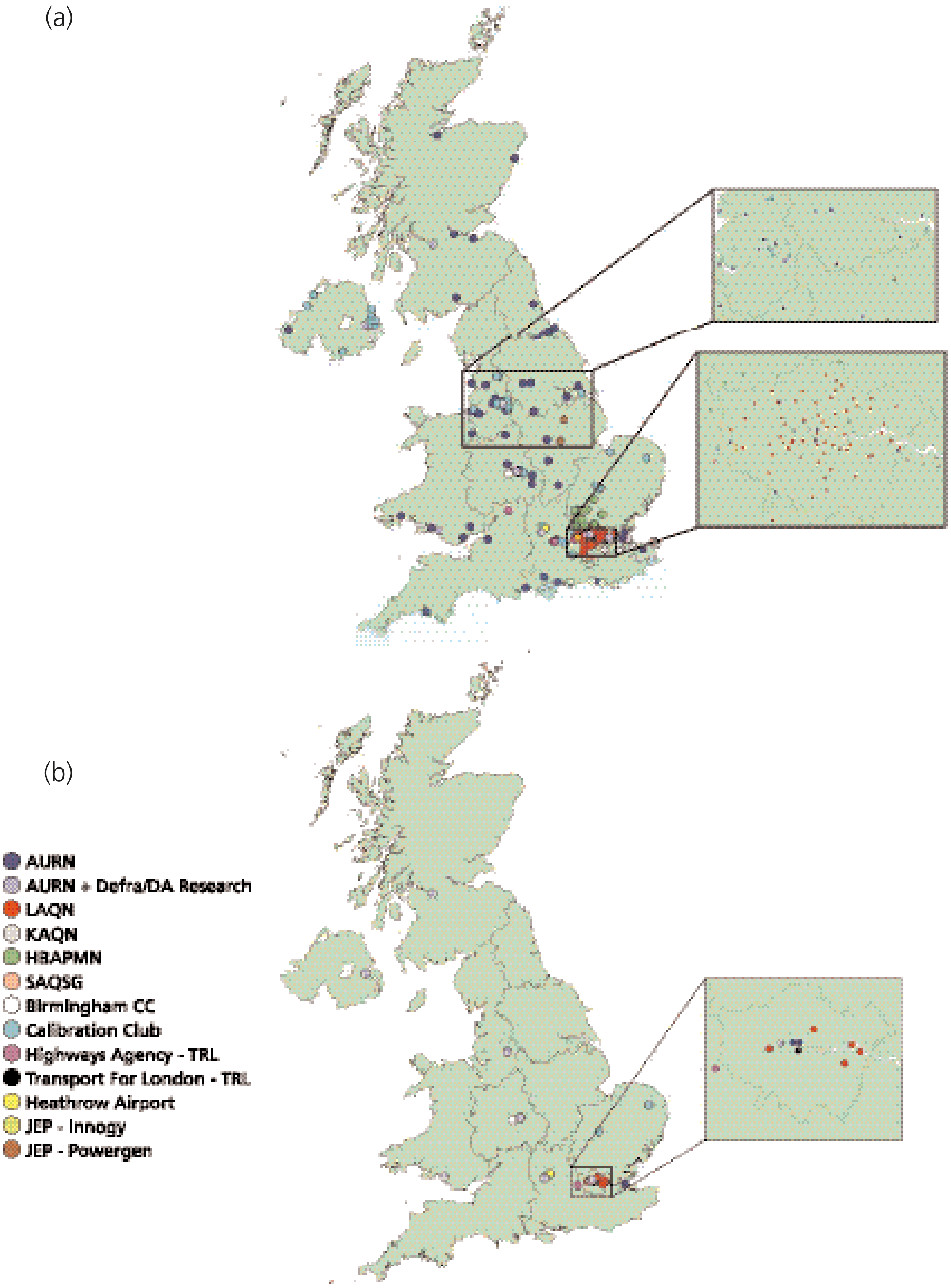

Table 5.3 Summary of PM_{10} monitoring stations in the UK national networks (2003).

Site type	AURN	LAQN
Kerbside	1 (0)[a]	2 (1)
Roadside	7 (0)	1 (0)
Urban	23 (0)	1 (0)
Background Urban centre	23 (1)	0 (0)
Suburban	4 (0)	1 (0)
Industrial	4 (0)	0 (0)
Rural/remote	4 (2)	0 (0)
Total	66 (3)	5 (1)

[a] The number of sites with co-located $PM_{2.5}$ analysers is shown in parentheses.

470. Monitoring stations are classified according to their location; a summary of the PM monitoring sites is provided in Table 5.3. Details of the site-type definitions used in the UK are provided in Annex 3 of the AQEG report on NO_2 (AQEG, 2004). Further details on the national monitoring network sites and measurement data can be found at www.airquality.co.uk.

471. Automatic data are collected from the network sites on an hourly basis and, after initial validation, are disseminated to the public and the media via the Air Quality Communications Unit. Daily data from the sequential gravimetric samplers are available after the filters have been collected from the analyser and returned to the laboratory for conditioning and weighing.

472. There is a relatively small percentage of roadside and kerbside monitoring sites in the national network (~10%). This is because the networks were developed primarily to monitor urban and rural background pollution, to which most of the population is exposed. However, in recent years, there has been increased interest in roadside pollution levels and a much higher percentage of the more recent sites added to the network have been in this site category. It is possible that many more roadside sites will be required if some of the currently proposed revisions to the EU First Daughter Directive are agreed; however, at present these are still under discussion within the EU.

5.10.1.2 Regional networks

473. Regional networks are operated or coordinated by King's College ERG, DoE Northern Ireland and the Welsh Assembly Government, on behalf of local authorities. The operation of the ERG networks mirrors that of the national networks, with routine data polling and validation and subsequent quality assurance and control to provide ratified datasets. A summary of the site types

Table 5.4 Summary of PM_{10} monitoring stations in regional UK networks (2003).

Site type	LAQN[a]	KAQMN	HABAPMN	SAQSG
Kerbside	5	2	—	5
Roadside	34 (2)[b]	8	4	—
Urban Background	16(1)	6	10	3
Urban Centre	—	—	—	—
Suburban	9 (2)	1	—	—
Industrial	—	—	—	—
Rural	—	1	—	—
Total	64 (5)	18 (0)	14 (0)	8 (0)

[a] These sites are not part of the UK network (see Table 5.3). [b] The number of co-located $PM_{2.5}$ sites is given in parentheses.

within these networks is provided in Table 5.4. In Northern Ireland the majority of local authority air quality monitoring sites are part of the Netcen calibration club and have been included under this heading in the tables in this section and in the data proforma.

5.10.1.3 Other networks

474. Birmingham City Council operates a number of PM_{10} and $PM_{2.5}$ monitoring sites. The site at Hodge Hill has one of the longest time series of $PM_{2.5}$ data in the UK: it commenced in 1994. These data have been made available to AQEG.

475. Some local authority monitoring stations throughout the UK form part of the Calibration Club network within which data handling and data quality assurance and control are undertaken centrally by Netcen, on behalf of the individual authorities.

476. Other monitoring networks are orientated towards monitoring air quality close to particular emission sources. Examples of such monitoring networks used in this report are those operated by the power generators, Heathrow Airport Ltd and the Highways Agency (monitoring sites close to very busy roads). As with national and regional networks, these other networks have recognised procedures for data quality assurance.

477. In addition, Defra and the Devolved Administrations operate a research network of particle measurements with a range of gravimetric samplers. These measurements are not part of the AURN and the data are not fully ratified. The network is primarily intended to investigate the relationship between gravimetric samplers and automatic particle analysers that are operated within the AURN (see Section 5.4.3).

478. A summary of the site types within these other networks is given in Table 5.5.

Table 5.5 Summary of PM_{10} monitoring stations in other UK networks (2003).

Site type	Birmingham	Calibration Club	Highways Agency/TfL	Heathrow Airport Ltd[a]	JEP/ Innogy[b]	JEP/ Powergen	Defra/ Devolved Administrations research[c]
Kerbside	—	—	1 (1)[d]	—	—	—	1 (1)
Roadside	3	21 (2)	4 (1)	—	—	—	—
Urban background	1 (1)	15	—	—	—	—	3 (2)
Urban centre	—	—	—	—	—	—	4 (4)
Suburban	—	—	—	—	—	—	—
Industrial	—	4	—	—	—	—	—
Rural	—	—	—	—	1 (1)	2	1 (1)
Special:							
Rural kerbside	—	—	—	—	—	—	—
Suburban roadside	—	—	—	—	—	—	—
Airport	—	1	—	2	—	—	—
Total	4 (1)	41 (2)	5 (2)	2 (0)	1 (1)	2 (0)	9 (8)

[a] Data have been provided for more sites for earlier years. [b] The Heathrow Airport site is also part of the LAQN; the Heathrow Harlington site is also part of the AURN. [c] Data for PM_{10} and $PM_{2.5}$ are from a range of gravimetric samplers. [d] The number of co-located $PM_{2.5}$ sites is given in parentheses.

5.10.1.4 $PM_{2.5}$ monitoring networks

479. Monitoring within the UK national networks is supplemented by measurements of $PM_{2.5}$ to fulfil the requirements of the First Daughter Directive and to support policy development. Some regional and other networks also include $PM_{2.5}$ monitoring, as indicated in Tables 5.2 to 5.5. In the Defra/Devolved Administration research network, $PM_{2.5}$ monitoring is primarily undertaken with filter-based gravimetric methods (Partisol), with the instruments equipped with $PM_{2.5}$ Sharp-Cut Cyclone (SCC) heads. In each case, the $PM_{2.5}$ instrument is located with a PM_{10} instrument of identical type, so that relevant information on the fine ($PM_{2.5}$) and coarse (PM_{10}–$PM_{2.5}$) fractions can be determined. At two sites, Harwell and Marylebone Road, PM_{10} and $PM_{2.5}$ monitoring is undertaken with both gravimetric and TEOM analysers.

480. A summary of the $PM_{2.5}$ monitoring sites is provided in Table 5.6.

Table 5.6 $PM_{2.5}$ monitoring sites (2003).

Network	Site	Site type	TEOM	Partisol
AURN	London, Bloomsbury	Urban centre	✓	
	Rochester (Stoke)	Rural		
AURN and Defra/Devolved Administration research	Marylebone Road	Kerbside	✓	✓
	Belfast Centre	Urban centre		✓
	Birmingham Centre	Urban centre		✓
	Glasgow Centre	Urban centre		✓
	London North Kensington	Urban background		✓
	Manchester Piccadilly	Urban centre		✓
	Port Talbot	Urban background		✓
	Harwell	Rural	✓	✓
LAQN	Bexley 2– Belvedere	Suburban	✓	
	Bexley 3 – Thamesmead	Suburban	✓	
	Ealing 2 – Acton Town Hall	Roadside	✓	
	Greenwich Bexley 6 – A2 Falconwood	Roadside	✓	
	Hackney 4 – Clapton	Urban background	✓	
Birmingham CC	Birmingham Hodge Hill	Urban background	✓	
Calibration club	Norwick Golding Place	Roadside	✓	
	South Cambs Bar Hill	Roadside	ß-guage	
Highways Agency	Victoria	Kerbside		✓
	M25	Roadside	✓	
JEP/Innogy	Rosehurst Farm	Rural	✓	

5.10.2 Black smoke monitoring network

481. The Smoke and SO_2 Network was set up in 1961 and was then known as the National Survey. It constituted the backbone of the UK's air pollution monitoring during a period when the main pollutants of concern were smoke and SO_2. Monitoring sites were equipped and operated by participating organisations, primarily local authorities, who participated at their own expense. By the mid-1960s there were over 1,200 sites and the network size remained relatively stable until the early 1980s. Levels of smoke and SO_2 decreased substantially throughout the 1960s and 1970s, and in 1982 the National Survey underwent a major reorganisation. The total number of sites was reduced to around 470 and existing rural sites became the separate Rural Network.

482. The Network has continued to operate in this format, although the number of monitoring sites has gradually reduced to 123 at the end of 2003.

483. All daily data for the smoke network back to 1961 are available on www.airquality.co.uk and earlier data are available in printed annual reports.

5.10.3 PAH monitoring network

484. PAH concentrations were originally monitored as part of the Toxic Organic Micropollutant Network on behalf of Defra and the Devolved Administrations. Four on-going sites were established in the early 1990s (Hazelrigg, London, Manchester and Middlesbrough) to monitor long term trends of dioxins, PCBs and PAHs. Two rural sites (High Muffles and Stoke Ferry) were added in 1996 to investigate more fully rural concentrations of these species.

485. More recently, the UK PAH monitoring network, operated on behalf of Defra and the Devolved Administrations, was established. This is comprised of 25 sites in total: the six sites mentioned above and a further 19 sites located in areas of high domestic fuel use, roadside locations, major urban areas and close to a selection of major industrial plants. These sites monitor a range of PAH compounds as follows:

- at all sites: acenaphthene, fluorene, phenanthrene, anthracene, pyrene, benz[a]anthracene, chrysene, benz[b]fluoranthene, benzo[a]pyrene and benzo[ghi]perylene;

- additionally at London and Middlesborough: acenaphthylene, fluoranthene, benzo[k]fluoranthene, indeno[123,cd]pyrene and dibenzo[ah/ac]anthracene;

- additionally at Manchester and Hazelrigg: fluoranthene/methylphenanthrene, dibenzo[ah/ac]anthracene/benzo[k]fluoranthene and coronene;

- since 2000 the following extra PAHs have been measured at sites other than Hazelrigg, High Muffles, Manchester and Middlesbrough: anthanthrene benzo[e]pyrene, benzo[b]naph[2,1-d]thiophene, benzo[c]phenanthrene benzo[k]fluoranthene, benzo[j]fluoranthene, coronene, cyclopenta[c,d]pyrene, dibenzo[ae]pyrene, dibenzo[ah]pyrene, dibenzo[ai]pyrene, dibenzo[al]pyrene, fluorene, 5-methyl chrysene, methylphenanthrenes/methyl anthracenes/4,5-methylene phenanthrene and retene.

486. A detailed analysis of the data for benzo[a]pyrene was undertaken in 2001 in support of the development of the UK National PAH objective within the UK Air Quality Strategy (Coleman *et al*. 2001) and data are available at www.airquality.co.uk.

5.10.4 Heavy metals monitoring networks

487. The monitoring of a range of up to 16 elements in urban areas is undertaken by the multielement monitoring network. This commenced in 1976 with 20 sites in 17 towns and cities throughout the UK. After two years, the monitoring was reduced to five sites: London Brent, Glasgow, central London[6], Leeds and Motherwell. These sites remain in operation today (see below for current details),

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[6] Minster House, Vauxhall 1976 to 1990; Bridge Place, Victoria 1990 to 2000; Horseferry Road, Westminster from June 2002 onwards.

with the addition of a site in Cromwell Road in London in 1990. Additionally, one rural site, at Eskdalemuir in Scotland, was added to the multielement network in 2000.

488. Since 2003, monitoring of PM_{10} particulate trace elements, including heavy metals, has been conducted at a network of ten rural sites across the UK, with weekly sampling and analysis of filter samples using ICP-MS for trace elements and ion chromatography for major elements. Details of the network can be found at www.heavymetals.ceh.ac.uk.

489. Monitoring of particulate lead has historically been undertaken in the lead-in-petrol survey: this commenced in 1985 with two rural and four urban sites. Following the ban on sales of leaded petrol in 2000, the network was reduced to two urban sites – Manchester and Cardiff – where monitoring still continues.

490. When the first EC Directive on Lead in Air was introduced in 1982, a number of short term surveys of lead at industrial premises were undertaken. From the results of these surveys, three industrial locations were selected for long-term monitoring (eight sites in total). These monitoring sites have been in operation from 1985/6 to the present day.

491. When the new Daughter Directive considerably reduced the limit value for lead in air concentrations, a similar 12-month screening exercise of industrial premises was carried out in 2000. In preparation for the introduction of the 4th Daughter Directive on arsenic, cadmium, nickel and mercury, measurement of these elements was also included. In 2002, monitoring at a number of these sites recommenced at five locations: Avonmouth, Hallen Village (Avonmouth), Swansea, Sheffield and Runcorn.

492. In 2003, the operation of all of these networks, with the exception of the Rural Trace Element Network, has been rationalised into a single integrated programme, with a common sampling and analysis structure. Monitoring at all sites is now conducted on a weekly basis, using Partisol 2000 samplers equipped with PM_{10} size selective inlets, and subsequent extraction and analysis conducted using techniques consistent with the CEN WG14 proposed reference method.

493. Data for all these survey are available at www.airquality.co.uk, with the exception of the industrial monitoring undertaken during 2000, which are available at www.stanger.co.uk/airqual/metals.

5.10.5 Major ions and elemental and organic carbon in PM

494. Sulphate in particulate material was monitored at eight rural sites (Eskdalemuir, Stoke Ferry, Lough Navar, Barcome Mills, Yarner Wood, High Muffles, Strathvaich Dam and Glen Dye) as part of the UK Acid Deposition Monitoring Networks. Monitoring started in the early to mid-1980s at most sites, but at Eskdalemuir, monitoring started in 1973. In November 2001, three of the sites were closed (Stoke Ferry, Strathvaich Dam and Glen Dye). Four of the remaining sites are also part of the UK automatic Urban and Rural Monitoring Network - Yarner Wood, Lough Navar, High Muffles, Eskdalemuir.

Table 5.7 Ultrafine particle monitoring sites.

Site	Site type	CPC	SMPS
Marylebone Road	Kerbside	✓	✓
London Bloomsbury	Urban centre		✓
Belfast Centre	Urban centre	✓	
Birmingham Centre	Urban centre	✓	
Glasgow Centre	Urban centre	✓	
London North Kensington	Urban background		
Manchester Piccadilly	Urban centre	✓	
Port Talbot	Urban background	✓	
Harwell	Rural		✓

495. The results are reported and analysed in the reports for the Network (Hayman *et al.*, 2003) and data are available at www.airquality.co.uk.

496. Rural sulphate measurements are also undertaken as part of the Nitric Acid Monitoring Network. This network has been in operation since September 1999 and provides data on nitric acid, particulate nitrate and other species (gaseous sulphur dioxide and hydrogen chloride and particulate sulphate, chloride, sodium, magnesium and calcium), as part of the UK Acid Deposition Monitoring Programme. Measurements are made on a monthly basis at 12 rural locations using the CEH DELTA denuder and filter system. The denuder and filter samples are extracted and analysed using ion chromatography. Information on the monitoring programme and the measurements can be found in Hayman *et al.*, 2003 or at http://www.edinburgh.ceh.ac.uk/cara/networks.htm.

497. Daily urban sulphate in particulate measurements were initially undertaken at 20 sites throughout the UK during the period 1976–1978. From 1978, monitoring continued at five sites for a further year. In 1979, four of these sites were closed and a new site opened at the central London Laboratory in Vauxhall Bridge Road. Monitoring at the site in Brent continued until 1980 to provide some overlap with the new site in Vauxhall Bridge Road. This site moved a short distance to Bridge Place in 1990. Data from this site continued until 1996, although data capture was generally poor. Particulate material was collected on Whatman 40 filter on a daily basis at a nominal flow rate of 3 l min^{-1}. The filters were returned to the laboratory for analysis. The data are available at www.airquality.co.uk.

498. Measurements of daily mean sulphate, nitrate and chloride have been carried out on behalf of Defra and the Devolved Administrations since September 2001 at four sites (Marylebone Road, London North Kensington, Belfast Centre and Harwell). Samples are collected using Partisol 2025 PM_{10} samplers, with subsequent laboratory analysis. Measurements of continuous elemental and organic carbon are also carried out at these four sites, together with measurements of continuous nitrate at Harwell and Belfast.

5.10.6 Ultrafine particles

499. Monitoring of both total and fractionated particle number concentrations are carried out on a routine basis at a number of national network sites. This network is summarised in Table 5.7.

5.11 Measurement uncertainty and quality assurance control

500. The general principles of quality assurance and control applied to national air quality monitoring networks were described in the AQEG report on NO_2 measurements (AQEG, 2004). These principles, including those detailed in site operator's instruction manuals (Netcen, 2003), 6-monthly site audit visits and data ratification procedures apply equally to the measurements of particulate material, but there are significant differences in the detail.

501. In contrast to the measurement of gaseous species, no reference material is available for ambient particle samplers and, hence, no routine calibrations of the analysers can be performed. At the 6-monthly audit visits, the flow rate through the analyser size selective inlet is measured and, in the case of the TEOM analysers, the flow rate through the filter is also measured. Also, in the case of TEOM analysers, the tapered element calibration constant is determined using preweighed filters. Both the determination of flows and the tapered element calibration constant are undertaken using UK Accreditation Scheme (UKAS)-accredited methods.

502. For the gravimetric samplers used in the national network, the weighing of the filters is also undertaken using UKAS-accredited method during the process of data ratification. Any unusual data are checked carefully and additional information such as analyser status outputs, measurements of other pollutants at the same site, measurements of particulate material at nearby sites, known local effects such as nearby industry or bonfires and so on are investigated to assess whether the data are valid and reliable. The hourly data in the Air Quality Information Archive are marked as provisional until this process of data ratification is complete.

503. As discussed above, there are no reference materials available for calibration of ambient particle analysers to establish traceability of the measurements to metrology standards. The quality assurance and control procedures in use for the UK Network measurements ensure high quality data for PM, as measured by a standard TEOM instrument. For EU reporting purposes, the uncertainty of the data needs to comply with the Data Quality Objectives that refer to PM as measured by the reference method, which can be significantly different. Calculating this uncertainty, is therefore, not straightforward. This issue, which does not arise for measurements of other air pollutants, is being addressed by the EU Working Group on Equivalence (described in Section 5.3.2) and by other European initiatives.

Chapter 6

What do the measurements tell us?

Key points

- Although annual mean PM_{10} concentrations are highest at roadside sites followed by urban background and lowest at rural sites, there is significant overlap between concentrations within adjacent site classifications. The gradient in PM_{10} between roadside, urban background and rural sites is considerably lower than that for NO_X, reflecting a greater regional background of PM_{10}.

- Colocated measurements of daily PM_{10} and $PM_{2.5}$ concentrations show that these are strongly correlated. Correlations of PM_{10} and $PM_{2.5}$ with black smoke are considerably weaker and lower in summer than winter. There is a tendency for $PM_{2.5}$ to PM_{10} ratios to increase with increasing $PM_{2.5}$ and for the ratio of coarse particles to PM_{10} to increase with mean daily windspeed, suggesting the importance of sea salt or resuspended crustal dusts.

- Current concentrations of lead at roadside and urban background sites in large conurbations are well below the limit values and objectives, whereas the concentrations of arsenic, cadmium and nickel are within the proposed EU assessment thresholds. All four metals show significant elevation adjacent to some industrial metallurgical sites.

- Several metals, most notably iron, manganese and copper show a distinct gradient in their concentration within airborne particles between roadside (highest), urban background and rural sites (lowest concentration).

- Concentrations of PAHs have declined at UK sites through the 1990s. Concentrations of benzo(a)pyrene were below the proposal EU target value of 1 ng m^{-3} in 2002 at all but one of the UK monitoring sites. Concentrations were below the UK air quality objective of 0.25 ng m^{-3} at 15 of the 19 sites for which data are available.

- Black smoke is a good indicator of primary combustion particles. The strongest correlations with PM_{10} are at sites in Northern Ireland, where domestic coal burning is still an important influence on air quality.

- Continuous measurements of elemental and organic carbon have been made at four UK sites. Mean concentrations declined from kerbside to urban background to rural sites with a stronger gradient for elemental than organic carbon. Analysis of data from the London Marylebone Road site shows diurnal profiles consistent with a substantial contribution from road traffic. Although elemental carbon is strongly correlated with the diurnal profile in heavy duty traffic, suggesting a source in diesel emissions, organic carbon is more closely related to the diurnal cycle of light duty vehicles. Analysis by wind direction again showed a strong relationship between elemental carbon and NO_X, whereas for organic carbon there is a strong correlation with the concentration of PM_{10}.

- A time series of weekly 24 h samples of PM_{10}, organic and elemental carbon at a rural site in Scotland show that the ratio of organic to elemental carbon increases substantially in the summer months, consistent with the influence of secondary organic compounds at this time of year.

- Sodium chloride derived from marine emissions is readily measurable at inland sites, its concentration relating to the distance which the air mass has travelled over the sea in the preceding 24 h.

- A mass closure model has been devised that is capable of accounting for a very high proportion of the variance in daily gravimetric $PM_{2.5}$ and PM_{10} concentrations. It contains the following components: ammonium sulphate, ammonium nitrate (in fine particles), sodium nitrate (in coarse particles), sodium chloride, elemental carbon, organic compounds, calcium (representative of sources such as construction and demolition dust), iron (representative of suspended soil and road dust and of non-exhaust traffic emissions) and bound water. The results of application of the model show a strong roadside increment in organic and elemental carbon from traffic exhaust emissions and iron-rich particles derived from non-exhaust traffic emissions.

- Although the fine fraction of particles at urban background sites is dominated by ammonium sulphate, ammonium nitrate, elemental carbon, organic compounds and bound water, the coarse fraction comprises primarily iron-rich dusts, calcium-containing dusts, sodium chloride, sodium nitrate and organic matter.

- The correlation of daily PM_{10} concentrations between two sites in Edinburgh and between two sites in Glasgow is very significant and, although lower, remains significant between Edinburgh and Glasgow, showing the importance of regional transport and regional meteorological influences on PM_{10} concentrations.

- An analysis of the data from the many sites measuring PM_{10} within London shows that those sites located away from traffic exhibited an average annual PM_{10} concentration in 2002 of 19.9 $\mu g\ m^{-3}$, which was 4.6 $\mu g\ m^{-3}$ above the regional background of 16.5 $\mu g\ m^{-3}$. The traffic-influenced sites show an annual average PM_{10} concentration of 36.0 $\mu g\ m^{-3}$ or 9.3 $\mu g\ m^{-3}$, with the regional background subtracted (all TEOM data). Regional background is a substantial proportion of the PM_{10} measured within London.

- The relatively few available data showing the concentration of PM_{10} as a function of distance from a major highway indicate a roadside elevation of about 30% above the background concentrations. Roadside increments of PM_{10}, $PM_{2.5}$ and PM_{coarse} at heavily trafficked sites in London and Birmingham show average roadside increments in PM_{10} varying between 8.7 $\mu g\ m^{-3}$ and 15.6 $\mu g\ m^{-3}$ (gravimetric). The roadside increment was highest on a heavily trafficked road with high buildings to one side, when the wind direction causes a recirculation of air across the road towards the sampler.
  Up to around half of the PM_{10} increment can be comprised of coarse particles, although the coarse particle increment in most cases is appreciably lower than the $PM_{2.5}$ increment.

- UK-wide maps of the spatial distribution of secondary sulphate, nitrate and ammonium show a gentle gradient from highest concentrations in the south and east of England to lowest concentrations in Northern Scotland and Northern Ireland.

- When data are normalised according to the number of sites in the network, there is very substantial month-to-month variation in the number of exceedences of 24 h average concentrations of 50 µg m^{-3}, emphasising the episodicity of PM_{10} concentrations.

- Specific components of PM_{10} also show episodicity, with nitrate showing a very high ratio of the 90th percentile concentration to the annual mean. Nitrate tends to show more episodicity than sulphate and was subject to a very large excursion in concentration in the spring of 2003.

- There are episodic contributions to PM_{10} in the UK caused by atmospheric transport of Saharan dust. This could lead to hourly excursions in PM_{10} concentration of over 200 µg m^{-3} (TEOM) with about half of the PM_{10} being in the form of fine particles. The frequency of significant Saharan dust episodes in the British Isles is one to two episodes per year.

- Concentrations of sea salt in the atmosphere also show considerable episodicity. Major episodes cross the country with a frequency of one to five times each year, generally in the autumn and winter months, causing PM_{10} excursions in excess of an hourly mean of 40 µg m^{-3} (TEOM) at coastal sites. There is a clear dependence of such events upon the strength of the wind.

- Biomass-burning fires remote from the UK can also provide a substantial source of elevated PM_{10} concentrations. During one such event in Western Russia, hourly mean PM_{10} concentrations across Northern England exceeded 100 µg m^{-3} (TEOM) and persisted for several hours.

- PM from construction activities can be a cause of short-term episodic peaks in PM_{10} concentrations. Such events tend to be rather localised and are highly dependent on current construction site activity. Annual mean concentration increments fall rapidly with distance, approaching background at about 100 m from the source.

- Correlations of PM_{10} with other traffic-generated pollutants such as NO_X, CO and benzene are strongest at roadside sites. At London Marylebone Road, concentrations of PM_{10}, $PM_{2.5}$, PM_{coarse} and particle number concentration are all higher on weekdays than on Sundays, despite there being little difference in overall traffic volume. This appears to be explained by the reduction in heavy duty traffic on Sundays.

- In addition to local effects, there are synoptic scale effects on average PM_{10} concentrations, with air masses arriving from the European continent showing a higher average concentration of PM_{10} and $PM_{2.5}$ than air masses with other origins.

- Abnormally high secondary PM concentrations in 1996 and 2003 were clearly associated with a high contribution of easterly wind directions corresponding to transport of air from the European mainland.

- An analysis of coarse and fine particle concentration data from London Marylebone Road clearly demonstrates a strong association with the volume of heavy duty traffic. Regression analysis indicates that the contribution of heavy duty vehicles is on average about 15-times greater per vehicle than for light duty vehicles. Traffic activity is responsible for by far the major part of the coarse particle concentrations at Marylebone Road.

- The activity of heavy duty traffic in causing coarse particle emissions and resuspension appears to be responsible for frequent daily mean concentrations in excess of 50 μg m^{-3} at a site in east London.

- Studies of major power stations have shown that primary PM emissions contribute typically about 0.25 μg m^{-3} to the annual mean PM_{10} concentrations. Evidence for rapid new particle formation in the plumes of coal- and oil-fired power stations is lacking.

- The investigation of the PM_{10} concentrations around Port Talbot is an illustration of how ambient monitoring, in combination with meteorological data, may be used to check on emissions, especially emissions from point sources. Such studies are a useful tool in site investigation. The technique was able to identify changes in ambient levels following the closure of furnace 5 in November 2001.

- Particle number concentrations are strongly influenced by road traffic, and concentrations at Marylebone Road far exceed those at central urban background locations. The diurnal profile of particle number concentration at urban sites follows that typical of traffic-generated pollutants.

- Measurements of particle size distributions show an elevation in concentrations for the diameter range 11–700 nm at London Marylebone Road compared to Bloomsbury, whereas over the majority of the size range concentrations at Bloomsbury well exceed those at the Harwell rural site.

- Traffic-generated particles with a diameter <20 nm are typically formed during mixing of the exhaust plume with ambient air by nucleation and condensation processes: this has a major influence on particle size distributions at roadside and urban sites.

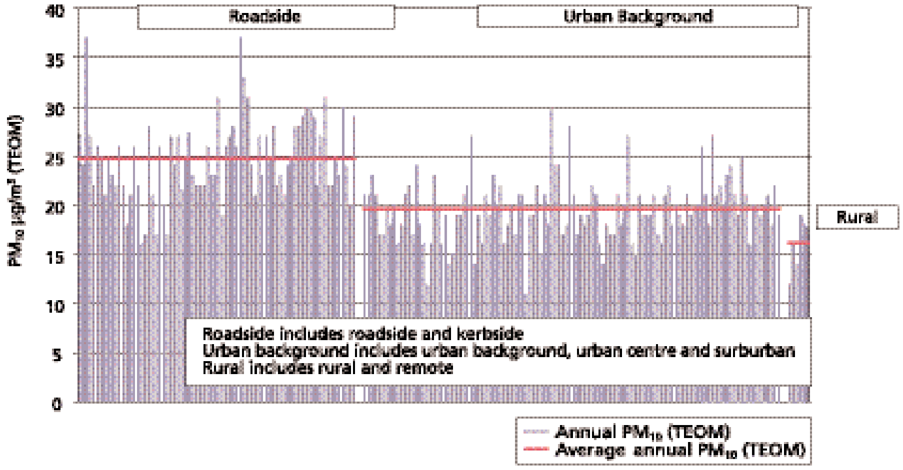
- Particle size distribution data from Harwell show clear evidence of the formation of new particles through homogeneous nucleation processes, which is having a significant influence upon particles of <30 nm diameter. It appears that the events responsible for new particle production are favoured by the low pre-existing surface area of particles in air from relatively clean trajectory directions.

6.1 Overview of PM mass concentration data

6.1.1 Introduction

504. To give a general indication of PM_{10} concentrations throughout the UK, Figure 6.1 shows the annual average PM_{10} for 2003 at 196 sites using TEOM

Figure 6.1 Roadside, urban background and rural annual average PM_{10} TEOM concentrations in 2003.



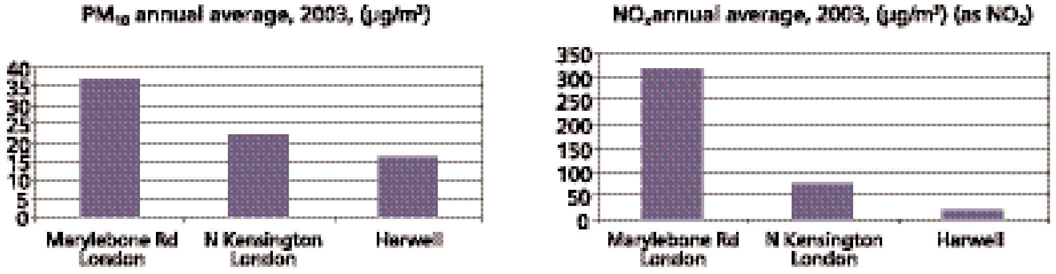
monitors, grouped into roadside, urban background and rural site locations. The average concentration for each site type is also shown. It is evident from the plot that although roadside concentrations are, on average, higher than urban background levels, which are in turn higher that rural levels, there is considerable overlap on an individual site basis. Individual site data appear in the pro forma within this report.

505. Figure 6.2 shows that, for three sample sites, the difference between PM_{10} concentrations at roadside, urban background and rural sites in much less than for NO_X.

506. At the North Kensington urban background site, NO_X concentrations are more than four-times lower than at Marylebone Road, whereas PM_{10} concentrations are less than a factor of two lower. In addition, whereas rural PM_{10} concentrations are only slightly lower than urban concentrations, NO_X concentrations are more than a factor of three lower.

507. This pattern reflects the more even distribution of PM_{10} concentrations across the whole of the UK, which arises from the wide diversity of PM_{10} sources and the contribution of secondary particulate material, as discussed in Chapter 4 and elsewhere.

Figure 6.2 Comparison of average NO_X and PM_{10} concentrations at roadside, urban background and rural sites.



6.1.2 Relationships of PM metrics

508. Daily PM_{10}, $PM_{2.5}$ and black smoke reflectance were measured at a single location in urban background air in Edinburgh for 1 year (September 1999 to September 2000). The PM samples were collected with a manual gravimetric sampler and black smoke reflectance was converted to concentration units using the British standard reflectance calibration curve. Mean 24 h PM_{10} and $PM_{2.5}$ for the year of measurement were 15.5 and 8.5 µg m^{-3} (gravimetric). For PM_{10}, the annual mean and the 98th percentile of 33.9 µg m^{-3} (gravimetric) – approximately equivalent to the threshold for seven exceedences per year – were both within the Air Quality Strategy Objectives for Scotland. For $PM_{2.5}$, the 98th and 90th percentiles of daily means (the latter equating to 35 exceedences annually) were 21.1 and 15.3 µg m^{-3} (gravimetric), respectively. The annual mean black smoke equivalent concentration was 6.6 µg m^{-3}.

509. Figure 6.3 shows that daily concentrations of PM_{10} and $PM_{2.5}$ were strongly correlated ($r^2 = 0.75$), reinforcing the strong relationship between $PM_{2.5}$ and PM_{10} in UK urban background air, noted previously by EPAQS.

510 The distribution of daily ratios of $PM_{2.5}$:PM_{10}, black smoke:PM_{10} and black smoke:$PM_{2.5}$, divided into summer and winter, are shown in Figure 6.4. The median daily ratio of $PM_{2.5}$:PM_{10} was 0.52 (interquartile range: 0.44–0.62), showing that PM_{10} at this site is approximately equally apportioned, on average, between fine and coarse particles. The narrow interquartile range is a consequence of the strong correlation between $PM_{2.5}$ and PM_{10}. The median (and interquartile ranges) for daily ratios of black smoke:PM_{10} and black smoke:$PM_{2.5}$ were 0.42 (0.27–0.59) ($r^2 = 0.18$) and 0.80 (0.51–1.09) ($r^2 = 0.27$), respectively. The poorer correlation and much wider variability in these latter ratios reflects the fact that black smoke is a strong indicator for a specific source contributor to PM (combustion-derived dark particles), rather than being a measure of all particles.

511. Figure 6.4 also shows that daily ratios of black smoke:PM_{10} and of black smoke:$PM_{2.5}$ were significantly lower in summer than in winter, reflecting seasonal factors, particularly the higher winter concentrations of black smoke caused by poorer dispersion of emissions from ground-level sources.

512. There was a significant trend for the proportion of $PM_{2.5}$:PM_{10} to increase with increasing $PM_{2.5}$ (Figure 6.5), but less strongly with increasing PM_{10}. This indicates a trend for variations in daily PM_{10} to be driven more by variations in $PM_{2.5}$, superimposed on a more constant contribution from the coarse fraction PM_{coarse} (of ~10 µg m^{-3} (gravimetric) in these Edinburgh data).

513. The ratio of coarse particles to PM_{10} increased with mean daily windspeed (Figure 6.6), consistent with the dominant coarse component source being wind-blown or resuspended dust or sea salt.

6.1.3 Relationship between $PM_{2.5}$ and PM_{10}

514. Measurements of PM_{10} and $PM_{2.5}$ were obtained at four pairs of roadside and background sites at Elephant and Castle, High Holborn and Park Lane in London and Selly Oak in Birmingham using a manual gravimetric sampler. The data obtained at the roadside and background sites at Elephant and Castle are shown in Figures 6.7 and 6.8, respectively. Data obtained at the other sites were similar

Figure 6.3 Scatter plot of 1 year (September 1999 – September 2000) of 24 h PM_{10} and $PM_{2.5}$ in urban background air in Edinburgh.

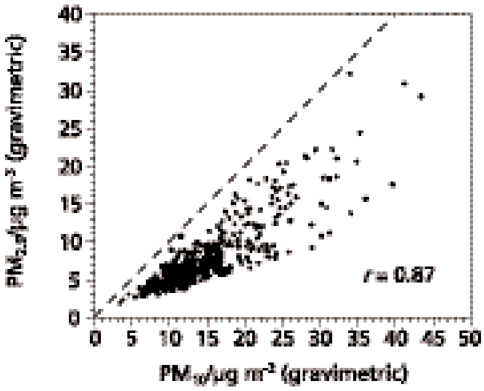


Figure 6.4 Distribution of 1 year of 24 h PM_{10}, $PM_{2.5}$ and black smoke ratios in urban background air in Edinburgh.

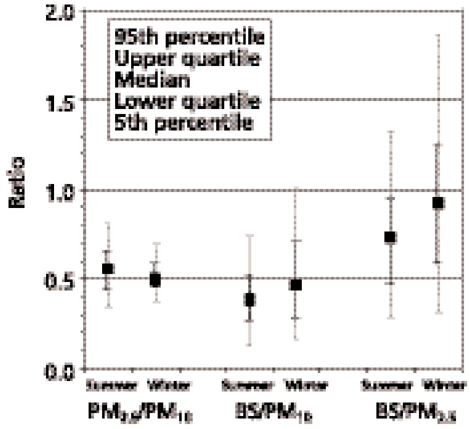


Figure 6.5 Ratio of $PM_{2.5}$: PM_{10} with $PM_{2.5}$, for 24 h measurements in urban background air in Edinburgh.

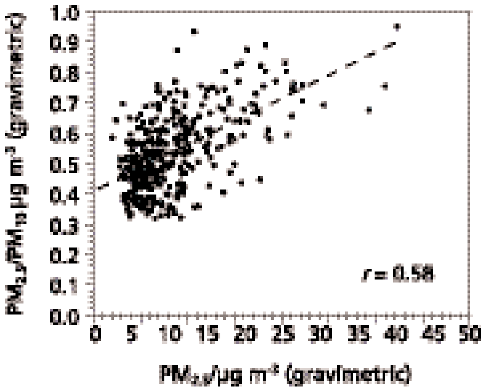


Figure 6.6 Ratio of PM_{coarse}: PM_{10} with mean windspeed for 24 h measurements in urban background air in Edinburgh.

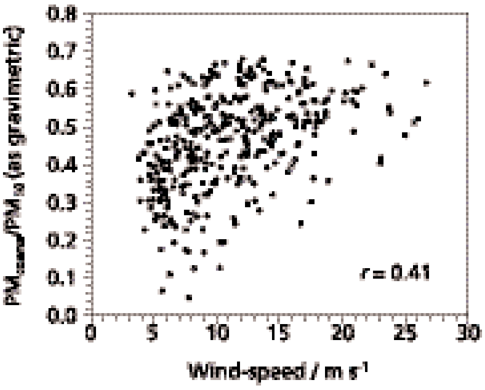


Figure 6.7 Elephant and Castle $PM_{2.5}$ versus PM_{10} roadside site.

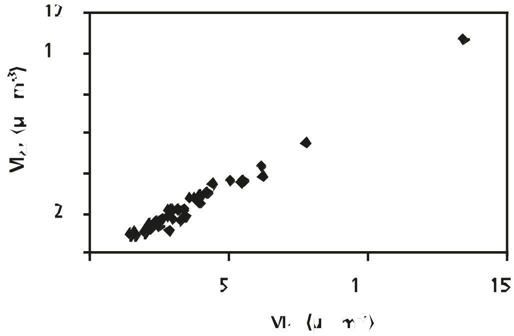
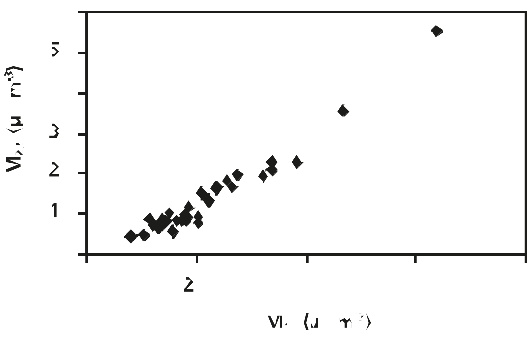


Figure 6.8 Elephant and Castle $PM_{2.5}$ versus PM_{10} background site.



to those obtained at Elephant and Castle, having a generally curvilinear form with an increasing gradient at larger values. This is consistent with the higher values of PM_{10} (and $PM_{2.5}$) being driven by episodes of high levels of $PM_{2.5}$ from distant sources.

515. The coefficients and correlation coefficients of lines fitted to the data using a reduced major axis (RMA) regression of $PM_{2.5}$ upon PM_{10} for each site are shown in Table 6.1.

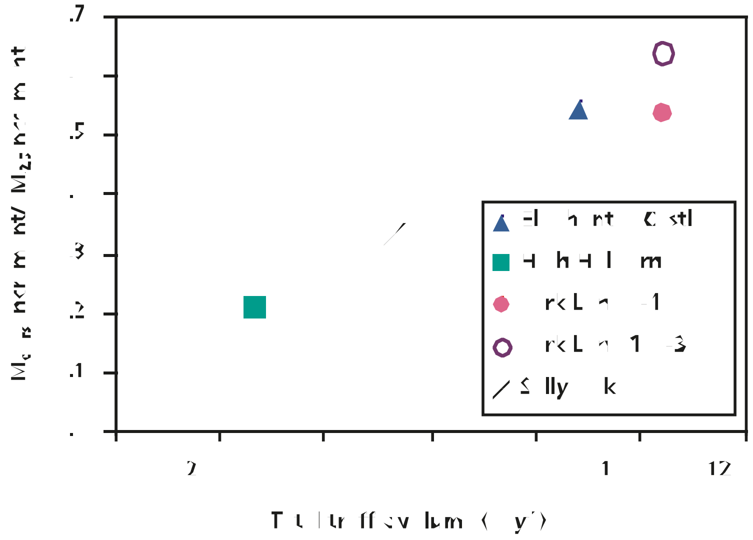
Table 6.1 Linear reduced major axis regressions of $PM_{2.5}$ on PM_{10}.

Site	Site type	RMA fit to $PM_{2.5}$ versus PM_{10} ($y = PM_{2.5}$, $x = PM_{10}$)	r^2 coefficient
Elephant and Castle	Roadside	$y = 0.81\,x - 5.46$	0.97
	Background	$y = 0.87\,x - 5.31$	0.94
High Holborn	Roadside	$y = 0.83\,x - 5.94$	0.88
	Background	$y = 0.86\,x - 6.73$	0.92
Park Lane (all measurements)	Roadside	$y = 0.72\,x - 2.38$	0.96
	Background	$y = 0.86\,x - 3.97$	0.92
Selly Oak	Roadside	$y = 0.79\,x - 3.71$	0.93
	Background	$y = 0.74\,x - 2.82$	0.93

516 The most heavily trafficked roadside sites have lower values for the gradient of the $PM_{2.5}$ versus PM_{10} relationship. Conversely, the corresponding background sites also have the highest values of gradient. Lower gradients indicate a higher contribution of PM_{coarse} at the heavily trafficked sites.

517. The mean of the ratio of the PM_{coarse} increment to the $PM_{2.5}$ increment at each location is shown plotted against the traffic volume at the roadside site in Figure 6.9. Again, the evidence is of a larger proportion of PM_{coarse} at the more heavily trafficked sites. The Park Lane data are split according to wind direction.

Figure 6.9 Ratio of roadside increments versus traffic volume.



6.1.4 Specific components of airborne particles

6.1.4.1 *Metals: network data*

518. A description of the UK monitoring networks for heavy metals is provided in Chapter 5. A detailed analysis of trends for specific metals is given in Chapter 7. A summary of limit values, assessment thresholds and air quality objectives applicable to heavy metals appears in Table 6.2 below.

Table 6.2 EU limit values and assessment thresholds and National Air Quality Strategy Objectives for metals.

Pollutant		Concentration	Date to be achieved by
Lead	Limit value	0.5 μg m^{-3}	01/01/05
	Objective	0.5 μg m^{-3}	31/12/04
	Objective	0.25 μg m^{-3}	31/12/08
Arsenic	Assessment threshold	6 ng m^{-3}	NA
Cadmium	Assessment threshold	5 ng m^{-3}	NA
Nickel	Assessment threshold	20 ng m^{-3}	NA

519. Metal concentrations measured in 2002 within the UK multi-element network and the UK industrial network are described in Table 6.3.

520. Concentrations at roadside and urban background sites in large conurbations are well below the limit values and objectives for lead and below the proposed assessment thresholds for arsenic, cadmium and nickel. Significantly higher concentrations of iron measured at roadside sites compared with urban background sites may reflect the contribution from particle resuspension or brake wear.

521. Concentrations of arsenic, cadmium, nickel and lead are significantly higher at monitoring sites in close proximity to metal processing industries, depending upon the precise nature of the process. Although levels of lead are below both the limit value and 2004 objective, they are currently equivalent to the 2008 objective at two locations. Concentrations of cadmium and nickel exceed the proposed assessment threshold at four locations.

6.1.4.2 *Metals: other data*

522. A summary of the concentration of trace metals measured in PM_{10} and $PM_{2.5}$ in urban air in the UK is shown in Table 6.4.

523. The apportionment of metal between $PM_{2.5}$ and PM_{coarse} and between water-soluble and non-water-soluble fractions in these two size fractions, for the data from Edinburgh is shown in Figure 6.10. Figure 6.10b shows that the proportion of metal in PM_{10} that is contained within $PM_{2.5}$ varies widely, ranging from only 13% (titanium), 17% (iron) and 21% (nickel) to 63% (zinc), 72% (vanadium), 82% (arsenic), 84% (lead) and 95% (cadmium), on average. The proportion of each metal that was water-soluble in each size fraction also varied widely.

Table 6.3 Metal concentrations measured in the UK in 2002.

(a) Multi-element network sites.

Site name	**Site type**	**Metal concentration (ng m^{-3})**										
		As	**Cd**	**Ni**	**Pb**	**Pt**	**V**	**Zn**	**Cr**	**Cu**	**Fe**	**Mn**
London Brent	Roadside	1.7	0.9	2.9	22	0.011	3.8	34.2	7.2	24.3	810	10.7
London Cromwell Road	Roadside	1.5	0.3	4.4	27	0.009	3.8	45.7	11.8	49.2	1102	12.6
Central London	Urban background	1.9	0.4	2.2	22	0.006	3.9	35.3	4.7	21.6	598	9.2
Leeds	Urban background	1.9	0.5	2.9	43	0.006	2.8	46.2	7.8	14.3	448	11.8
Glasgow	Urban background	1.3	0.2	1.6	15	0.005	1.4	28	5.3	12.7	314	5.0
Motherwell	Urban background	1.4	0.4	1.3	13	0.006	1.2	15.1	5.2	7	224	4.1
Eskdalemuir	Rural	0.7	0.1	0.6	3	0.005	0.7	4.1	4.6	1.7	27.8	0.8

(b) Industrial network sites.

Site name	**Metal concentration (ng m^{-3})**			
	As	**Cd**	**Ni**	**Pb**
IMI, Walsall	1.9	2.2	2.1	40
BZL, Avonmouth	—	8.3	—	—
BZL, Hallan	4.8	13.1	2.0	250
Corus, Rotherham	2.7	—	—	—
INCO, Swansea	—	—	28.9	—
CEM, West Bromwich	1.7	0.7	—	—
Walkers Galvanizing, Walsall	1.6	—	—	—
White Rose, Leeds	1.7	—	3.1	—
Bruhl, Sandwell	2.5	—	—	—
Sidney Smith, Stourbridge	2.0	1.2	—	—
Britannia, Wakefield	—	—	—	250
Avesta, Rotherham	2.4	0.9	16.2	50
Brookside Metals, Willenhall	1.8	5.1	2.0	190
Elswick, Newcastle	1.7	3.7	1.9	160

Table 6.4 Comparison of total metal concentrations in urban PM samples from the UK. (The years denote the period during which samples were collected.)

Component (ng m^{-3})	PM_{10}			TSP			$PM_{2.5}$	
	Edinburgh 1999–2000[a]	**Birmingham 2000–2001[b]**	**Birmingham 1992[c]**	**Glasgow 2000[d]**	**London 1999[d]**	**Liverpool 1994–1996[e]**	**Edinburgh 1999–2000[a]**	**Birmingham 1992[c]**
Ti	3.7	—	16.6	—	—	—	0.4	11.9
V	1.1	—	7.4	1.7	4	7.3	0.7	2.4
Cr	1.6	—	12.6	8.9	2	2.1	0.5	6.2
Mn	2.9	6.4	16.3	6.9	12	8.4	0.7	64
Fe	183	204	301	399	870	340	27.6	187
Ni	3.4	2.5	4.8	4.8	5	3.0	1.0	1.9
Cu	4.9	12.0	38.7	12	21	21	1.4	8.5
Zn	13.3	29.9	353	30	41	36	7.5	55.6
As	0.4	—	5.7	—	—	—	0.30	1.5
Cd	0.34	0.51	—	0.4	—	—	0.4	—
Pb	14.1	27.4	91	—	—	43	13.6	17.1

[a] Median value (n = 366), urban background air, analysis by acid-digest and ICP-MS (Heal *et al*., 2004). [b] Mean value (n = 30), roadside, analysis by acid-digest and ICP-MS (Harrison *et al*., 2003). [c] Mean value (n = 55), analysis by PIXE (Harrison *et al*., 2000).
[d] Mean value (n = 12), analysis by XRF and INAA (DEFRA, 2003a). [e] Geometric mean value (n = 60), analysis by ICP-MS (Chester *et al*., 2000).

For example, the proportion of water-soluble titanium and iron was ≤10% in both $PM_{2.5}$ and PM_{coarse}, whereas ≤74% of vanadium, zinc and lead in $PM_{2.5}$ was water-soluble. For the PM_{10} fraction as a whole, >50% of the metal was water-soluble for vanadium, zinc, arsenic and cadmium. For all 11 metals quantified, a greater proportion of metal was water soluble in the fine fraction than in the coarse fraction.

524. On average, for the data from Edinburgh, the 11 trace metals constituted 1.8% of daily PM_{10} mass (maximum daily proportion 9.0%) and 0.8% of daily $PM_{2.5}$ mass. The PM_{coarse} fraction of PM_{10} was, therefore, almost a factor of four more enriched with these metals (median daily value 3.1%) than the $PM_{2.5}$ fraction. These proportions varied significantly with season, with coarse particles being more enriched with trace metal in summer and vice versa for fine particles. A greater proportion of the trace metal in $PM_{2.5}$ (~30%) was water-soluble than in PM_{10} (~15%).

525. Although iron is the dominant trace metal, its relative contribution to the sum of the measured trace metal concentration varies widely with both size and water-soluble fractions (Figure 6.11). The contributions of total iron to the sum of total metal for the 11 trace metals measured in PM_{10}, $PM_{2.5}$ and PM_{coarse} are 80%, 51% and 89%, respectively, whereas the contributions of water soluble iron to the sum of water-soluble metal are only 43%, 19% and 64%, respectively. In fact, within $PM_{2.5}$, the amount of water-soluble lead and zinc both exceed that of iron. Across all the particle size fractions, over 90% of the water soluble or total mass of metals analysed is always contributed by iron, copper, zinc and lead. Titanium, vanadium, chromium, manganese, nickel, arsenic and cadmium each only constitute <2%, on average, of the total metal measured in each size fraction. The greater water solubilities of vanadium and manganese mean these two metals are relatively more important contributors to water soluble metal, although both still constitute <5% each (Figure 6.11).

526. A comparison of the proportion of trace metal in 24-h samples of PM_{10} from six sites across the UK is shown in Figure 6.12. The values of total PM_{10} (TEOM) concentrations are shown in the first column of the figure. The main feature of the data is for the proportion of most of the trace metals in PM_{10} to be roughly comparable across the sites measured (even though the total airborne concentration of each metal will vary with total mass concentration of PM_{10}). There are some minor exceptions, presumably indicating enhanced local sources of a particular element at a particular receptor. For example, copper is relatively more abundant in PM_{10} at the Marylebone Road location and is relatively less abundant as size of urban area decreases; zinc is more abundant in Birmingham, but this can be attributed to local industrial point sources; vanadium is relatively more abundant in Belfast.

6.1.4.3 PAHs

527. Chapter 5 outlines the network for PAH monitoring in the UK and the method of measurement. A total of 32 PAH species are measured at some or all of the 25 monitoring locations currently in operation. However, this brief analysis of the data will concentrate on the compound benzo[a]pyrene. EPAQS has recommended setting a national standard for PAHs using annual average

Figure 6.10 Median concentrations of trace metals in one year of 24-h samples of PM_{10} in background air in Edinburgh, subdivided into $PM_{2.5}$ and PM_{coarse} and further subdivided into water-soluble and non-water-soluble metal. (a) Absolute concentrations and (b) relative proportions.

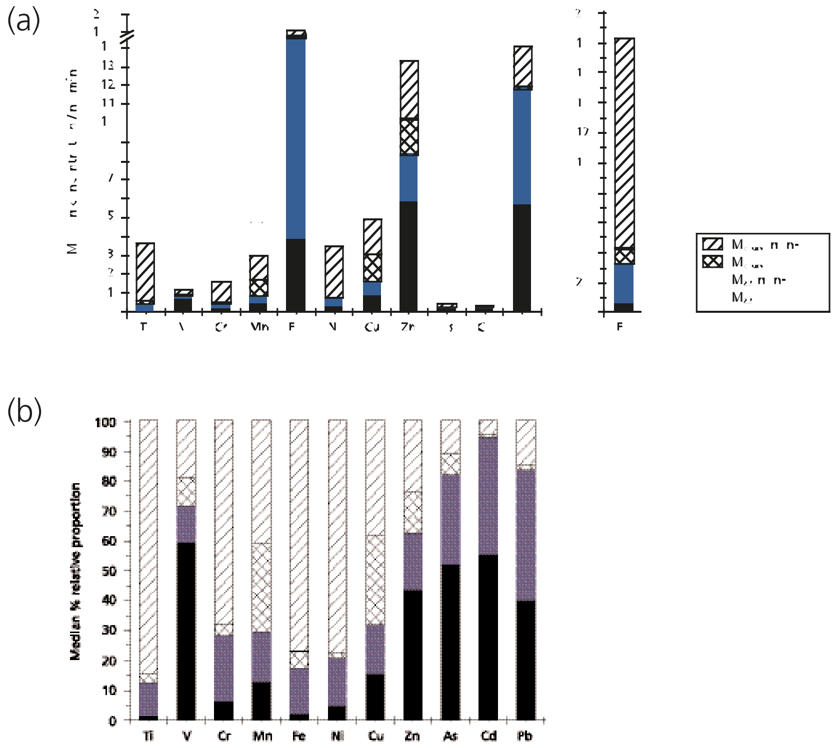
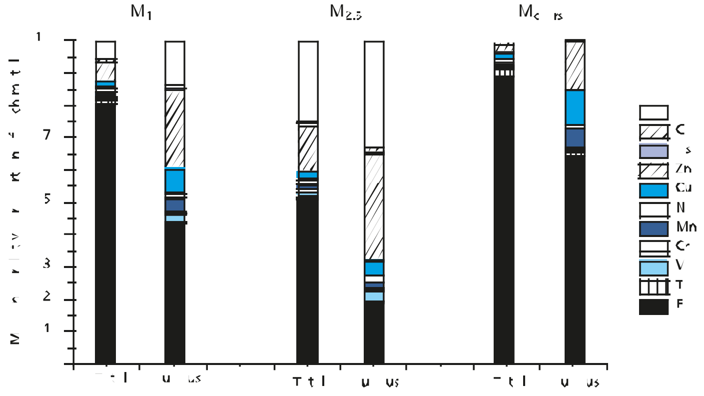


Figure 6.11 Relative contribution of each metal to the sum of 11 trace metals measured in PM_{10}, $PM_{2.5}$ and PM_{coarse} in background air in Edinburgh both water-soluble and total metal. (Data are median values of n = 366 daily samples.)



concentrations of benzo[a]pyrene as a marker for the human health effects of PAHs. EPAQS has recommended an annual average concentration of 0.25 ng m^{-3} as a guideline. The current draft proposal for the 4th EU Daughter Directive proposes an assessment threshold for benzo[a]pyrene of 1 ng m^{-3}.

528. Emissions of benzo[a]pyrene in the UK calculated within the NAEI show that there has been a reduction in emissions from just under 70 t in 1990 to under 10 t in 2001. In 2001 the two largest sources, accounting for about 65% of total emissions, were residential sources and other sources (mainly natural sources).

Figure 6.12 Proportion of metal (ng µg–1) in 24-h samples of PM_{10} from six sites in the UK. Although all samples are from the period September 1999–September 2000, the data presented are not all entirely concurrent.

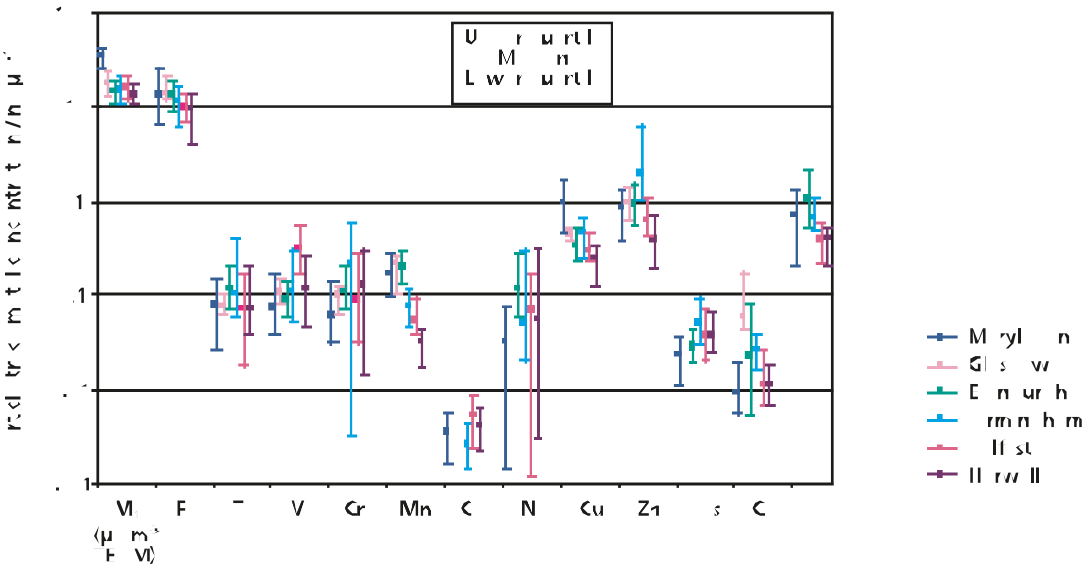
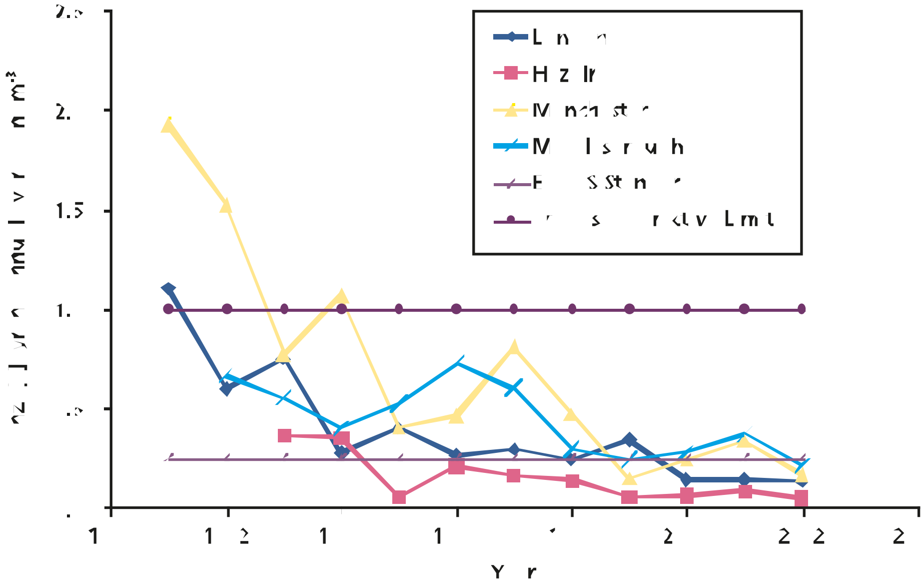


Figure 6.13 Annual average benzo[a]pyrene concentrations at four long-running sites in the UK.



529. Annual average concentrations of benzo[a]pyrene at the four long-running sites show a similar downward trend (Figure 6.13). Concentrations at the urban sites – London and Manchester – have decreased considerably and more than at the industrial site in Middlesbrough and the rural site at Hazelrigg in Lancashire.

530. **Figure 6.13** also indicates the EPAQS standard value of 0.25 ng m^{-3} and the proposed EU target value of 1 ng m^{-3}. Table 6.5 gives a summary of the benzo[a]pyrene annual average concentrations at all sites for 2000–2002. It shows that concentrations at all except one site are below the EU target value, but that the EPAQS standard was equalled or exceeded at four sites in 2002.

Table 6.5 Summary of annual mean benzo[a]pyrene concentration at all sites 2000–2002.

Site	Mean annual benzo[a]pyrene concentration (ng/m^3)		
	2000	2001	2002
Ashington	0.17	0.2	NA[a]
Belfast	—	**0.37**	0.13
Birmingham	—	0.16	0.13
Bolsover	**0.25**	**0.28**	0.24
Bromley	—	—	**0.25**
Glasgow	0.12	0.12	0.12
Hazelrigg	0.06	0.083	0.048
High Muffles	0.04	0.05	0.043
Holyhead	0.11	0.15	NA[b]
Kinlochleven	**2.3**	**0.34**	**0.38**
Leeds	—	0.16	0.18
Lisburn	**0.93**	**0.96**	**0.66**
London 2a	<0.14	0.14	0.13
Manchester	0.24	**0.34**	0.17
Middlesbrough	**0.28**	**0.37**	0.21
Newcastle	—	0.11	0.12
Newport	**0.35**	**0.36**	0.19
Port Talbot	**0.59**	**0.4**	**0.34**
Scunthorpe	1.2	**0.34**	1.4
Speke	—	—	0.14
Stoke Ferry	0.087	0.09	0.083

Note: Concentrations in bold are at or above the EPAQS recommended annual average of 0.25 ng m^{-3}. [a]Not available due to loss of samples during analysis. [b]Sampling not undertaken in quarter 2 of 2002 due to health and safety concerns.

531. More detailed analyses of benzo[a]pyrene concentrations up to 2000 are available elsewhere (Coleman *et al*., 2001). An assessment of benzo[a]pyrene atmospheric concentrations in the UK to support the establishment of a national PAH objective (AEAT/ENV/R/0620 August 2001) are available at www.airquality.co.uk.

6.1.4.4 Black smoke

532. The black smoke method measures the reflectance of particles collected on a filter and is a measure of the elemental carbon content of the particles, which arises mainly from coal and oil combustion and automotive diesel emissions. Measurements of black smoke have been used previously as an indicator for primary combustion PM_{10} particles using the APEG receptor model (APEG, 1999). More recent analyses using this model, such as those presented in Chapter 8, have used measured NO_X concentrations in preference to black smoke. This is because NO_X measurements are much more commonly colocated with PM measurements compared to black smoke measurements. The precision of the black smoke method is also less good at very low concentrations (see Chapter 5).

533. Black smoke measurements can, however, still be useful indicators of PM mass concentrations in areas where high concentrations are strongly influenced by domestic coal burning. Table 6.6 lists PM_{10} and black smoke monitoring sites in the UK that are either collocated or less than 1000 m apart. Mean concentrations of PM_{10}, black smoke and rural sulphate for available data between January 2001 and March 2003 are also listed in the Table 6.6 along with the intercept and regression coefficients from the APEG receptor model. Rural sulphate concentrations have been taken from the nearest site.

534. Strabane Spring Hill Park is the site with the highest measured concentrations of both black smoke and PM_{10} and the highest correlation coefficient. This suggests that black smoke is a good indicator of primary combustion PM_{10} from domestic coal burning at this site. This site also has a higher black smoke regression coefficient, suggesting that the particles are less dark in colour than at many of the other sites examined. The receptor model results suggest that primary combustion PM contributed 55% of the total PM_{10} mass at this site. Note, however, that PM_{10} measurements at this site were carried out using a ß-attenuation monitor; all other sites used TEOM.

535. The values of the intercept and regression coefficients are generally as expected, but the correlation coefficient is only >0.5 at 5 of the 15 sites examined. This suggests that the combination of black smoke and sulphate measurements are relatively poor indicators of PM_{10} concentration at many of the sites. The correlation coefficients are low and intercepts high at the Glasgow and Wigan Leigh sites. The black smoke coefficient is unusually high at the Wrexham site, where the PM_{10} monitoring site is at the roadside.

536. Figures 6.14 and 6.15 illustrate the better agreement between the PM_{10} and black smoke monitoring data at Strabane Spring Hill Park than in Belfast.

6.1.4.5 Elemental and organic carbon

537. Measurements of elemental and organic carbon within PM_{10} were made at Marylebone Road, North Kensington, Belfast and Harwell during 2002 and 2003 using a Rupprecht & Patashnick Series 5400 ambient carbon particulate monitor. Measurements were made for 3-h periods.

538. The mean values of elemental and organic carbon and PM_{10} for each site are given in Table 6.7. Substantially higher concentrations of elemental and organic carbon and PM_{10} were observed at Marylebone Road (a kerbside site within a street canyon) than at the other sites. The ranking of the sites was the same for organic carbon and PM_{10} but in the case of elemental carbon higher concentrations were observed at the urban centre site (Belfast) than at the urban background site (North Kensington).

Table 6.6 PM_{10} and black smoke measurements by monitors colocated within 1000 m from January 2001 to March 2003. (Statistical data from the APEG receptor model are shown.)

PM_{10} monitoring site	Black smoke monitoring site	Distance between PM_{10} and black smoke monitors (m)	PM_{10} mean (μg m^{-3})	Black smoke mean (μg m^{-3})	Sulphate mean (μg m^{-3})	Number of Observations	r^2	Intercept	Black smoke coefficient	Sulphate coefficient
Strabane Springhill Park (BAM)	Strabane 2	Colocated	40	26	1.1	151	0.86	11.22	0.86	6.36
Nottingham centre	Nottingham 20	320	20	9	2.2	274	0.68	8.98	0.89	1.44
Newcastle centre	Newcastle 27	400	13	8	1.4	710	0.61	6.74	0.41	2.63
Newry Monaghan Row	Newry 3	Colocated	21	7	1.2	489	0.58	12.41	0.58	3.78
Bradford centre	Bradford 6	360	20	12	1.8	570	0.54	9.31	0.48	2.85
Wrexham (roadside)	Wrexham 10	540	23	4	1.9	200	0.49	9.79	2.41	2.37
Edinburgh centre	Edinburgh 25	810	20	11	1.3	365	0.44	9.26	0.66	2.94
Leicester centre	Leicester 19	920	17	15	2.2	268	0.37	7.64	0.30	2.31
Bolton	Bolton 24	860	16	8	1.8	540	0.35	7.71	0.44	2.36
Stoke centre	Stoke-on-Trent 20	860	16	10	1.8	509	0.35	7.71	0.44	2.36
Belfast centre	Belfast 45	990	19	7	1.2	591	0.34	10.60	0.41	4.26
Manchester Piccadilly	Manchester 11	540	25	14	1.8	672	0.27	13.07	0.60	2.21
Wigan Leigh	Leigh 4	Colocated	18	2	1.9	574	0.19	13.40	0.29	2.29
Glasgow kerbside	Glasgow 20	810	24	5	1.6	201	0.18	16.17	0.38	3.84
Glasgow centre	Glasgow 20	630	18	5	1.6	200	0.17	12.66	0.13	2.86

Figure 6.14 Daily means of black smoke and PM_{10} (Strabane 2 versus Strabane Springhill Park).

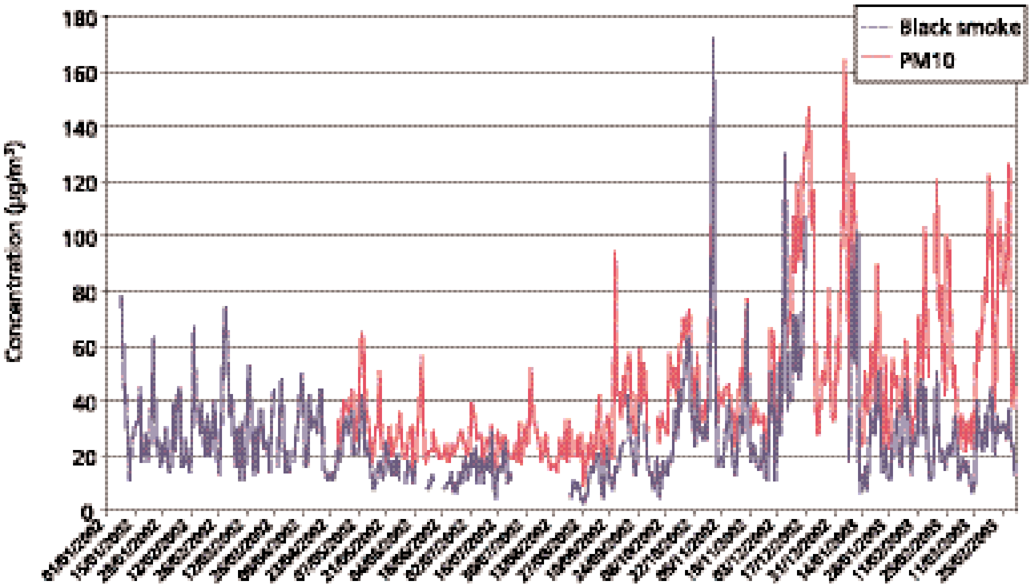


Figure 6.15 Daily mean black smoke and PM_{10} (Belfast 45 versus Belfast centre).

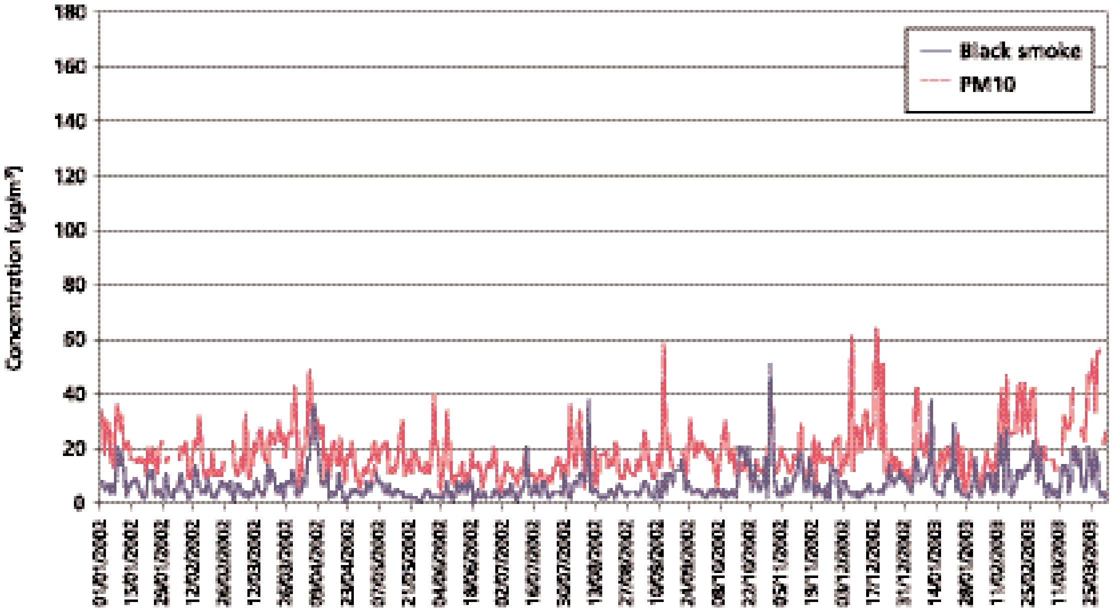


Table 6.7 Mean concentrations (and standard errors) of elemental and organic carbon and PM_{10}, measured at each of the four sites.

Site	Type of site	Elemental carbon ($\mu g\ m^{-3}$)	Organic carbon ($\mu g\ m^{-3}$)	PM_{10} ($\mu g\ m^{-3}$)
Marylebone Road	Kerbside	3.66 (0.04)	4.63 (0.03)	37.22 (0.30)
North Kensington	Urban background	0.98 (0.05)	2.45 (0.08)	24.69 (0.71)
Belfast	Urban centre	1.18 (0.03)	2.19 (0.03)	17.61 (0.23)
Harwell	Rural	0.66 (0.04)	1.94 (0.05)	13.43 (0.16)

539. The seasonal diurnal profiles of elemental and organic carbon concentrations obtained at Marylebone Road and Belfast are shown in Figures 6.16 and 6.17, respectively. No winter data were available for Belfast and due to instrumentation problems the autumn data for Marylebone Road were only available on alternate sampling periods. At both sites, minimum concentrations occurred during the 03:00 h to 06:00 h sampling period, with higher concentrations during the afternoon. At Belfast, higher concentrations of both elemental and organic carbon were seen in the evening during the autumn – possibly a result of domestic space heating.

Figure 6.16 Marylebone Road – seasonal diurnal profiles of elemental and organic carbon. (The error bars show standard error in the mean.)

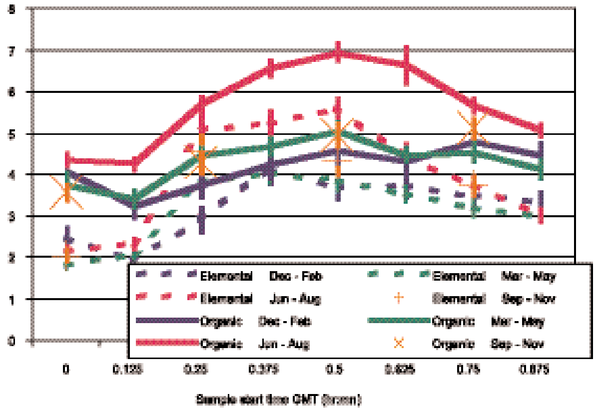
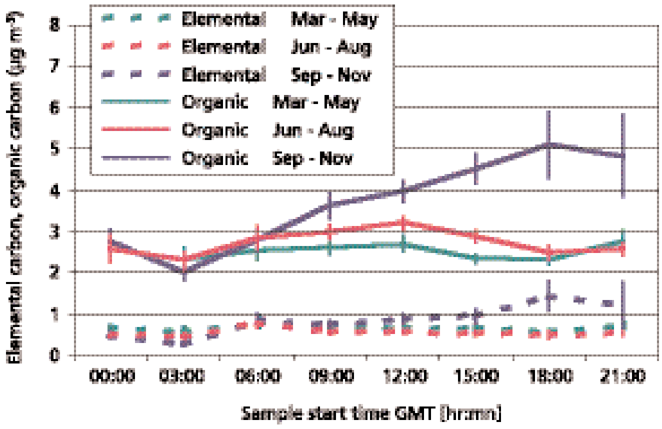


Figure 6.17 Belfast – seasonal diurnal profiles of elemental and organic carbon. (The error bars show standard error in the mean.)



540. The normalised diurnal profiles of elemental and organic carbon at Marylebone Road are shown in Figure 6.18. A much more rapid increase in concentration at the start of the working day was apparent in the case of elemental carbon, with a more rapid decline in the evening than occurs for organic carbon. The trends in elemental and organic carbon diurnal profiles show some similarity with the diurnal profiles of heavy duty and light duty traffic volumes, respectively (Figure 6.19).

541. Variation in the normalised concentrations of elemental and organic carbon and PM_{10} with wind direction at Harwell is shown in Figure 6.20. Wind direction was measured at Benson airfield, some 25 km to the north-northeast. The profiles of the different parameters are similar except for peaks in elemental carbon at 70°

Figure 6.18 Marylebone Road 2003 – normalised diurnal profiles of elemental and organic carbon. (The error bars show the standard error in the mean.)

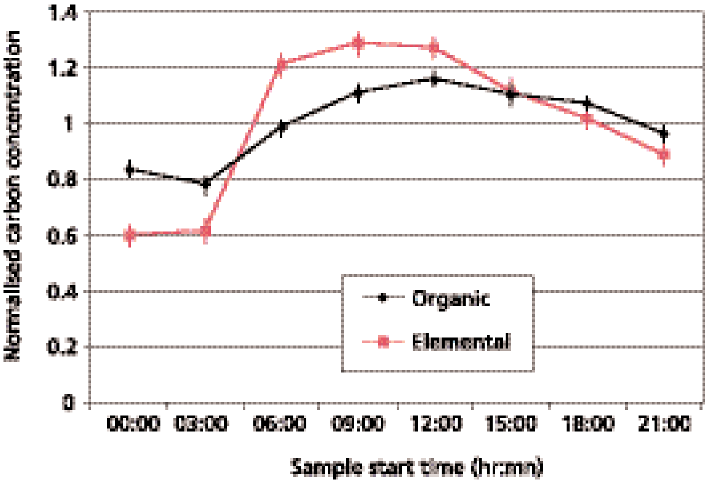
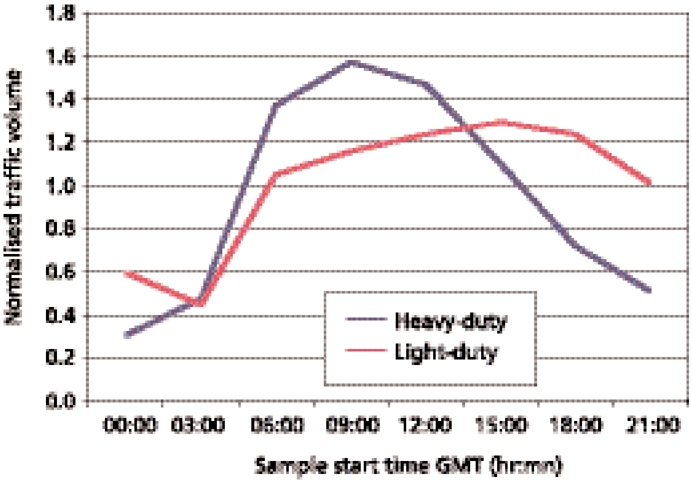


Figure 6.19 Marylebone Road – diurnal traffic profile (July 1998 to December 2001).



and 130° and at 130° in organic carbon. Observations on site identified these as the directions of local construction sites where the exhaust from site equipment was presumably the cause of the increased concentrations of carbon.

542. The corresponding plot of elemental and organic carbon and PM_{10} against wind direction for Marylebone Road is shown in Figure 6.21. Although there is a strong similarity between the profiles for organic carbon and PM_{10} (with the exception of a small peak in the PM_{10} around 90°), the profile for elemental carbon is significantly different. The profiles of organic carbon and PM_{10} are generally symmetrical about 170° wind direction with similar higher concentrations when the wind is from a range of southerly directions. These results are consistent with the cross street component of wind direction being reversed at street level in the street canyon. This carries material from vehicles onto the sampling station, which is on the southern side of Marylebone Road and is aligned approximately 80–260°.

Figure 6.20 Harwell – comparison of elemental and organic carbon and PM_{10} with wind direction.

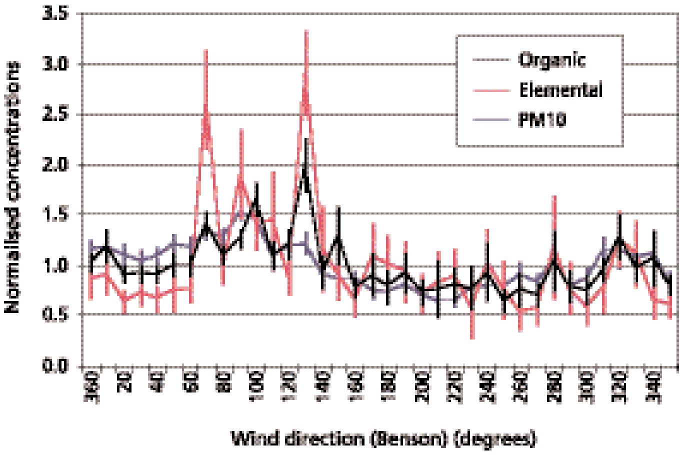
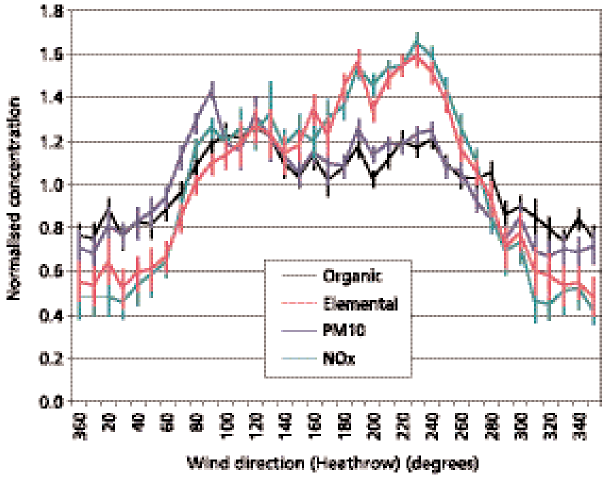


Figure 6.21 Marylebone Road – comparison of elemental and organic carbon, PM_{10} and NO_X with wind direction.



543. The normalised concentrations of elemental and organic carbon and NO_X are also plotted against wind direction in Figure 6.21. The similarity between the profiles of elemental carbon and NO_X is notable – although (as with organic carbon and PM_{10}) concentrations are higher when the wind is from a southerly direction, the highest concentrations occur in southwesterly winds. Other pollutants measured at Marylebone Road follow these two basic profiles of normalised concentration with wind direction. The profile of $PM_{2.5}$ follows those of organic carbon and PM_{10}, whereas the profiles of particle number and carbon monoxide follow the elemental carbon and NO_X profiles.

544. Monitoring of traffic speeds on Marylebone Road shows a much larger reduction in speed during periods of higher traffic volume in the westbound traffic than occurs in the eastbound traffic. This may be ascribed to the presence of a pedestrian light-controlled crossing within 100 m or a major light-controlled junction 300 m to the west of the sampling site, causing the westbound flow of traffic to come to a halt. The variable length queues for these features, which may at some times stretch to the east of the sampling point, will provide an additional source of exhaust-generated material, with greater quantities being detected when the wind direction has a westerly component.

Figure 6.22 Weekly 24-h values of PM_{10}, organic and elemental carbon and organic:elemental carbon ratio, Bush Estate.

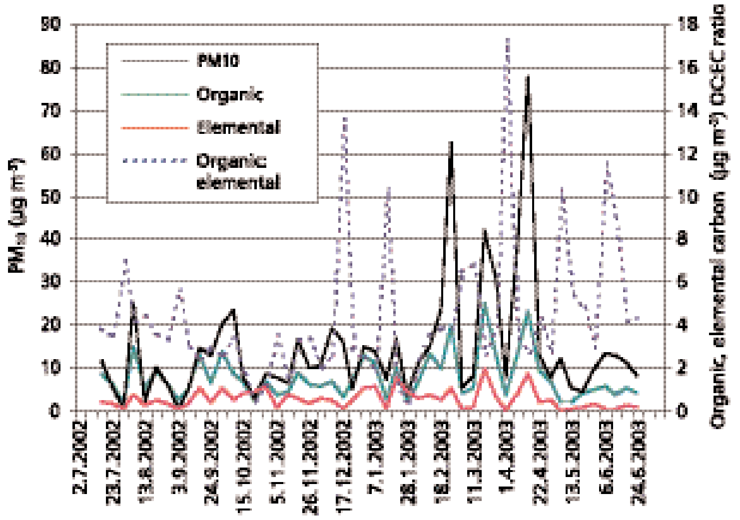
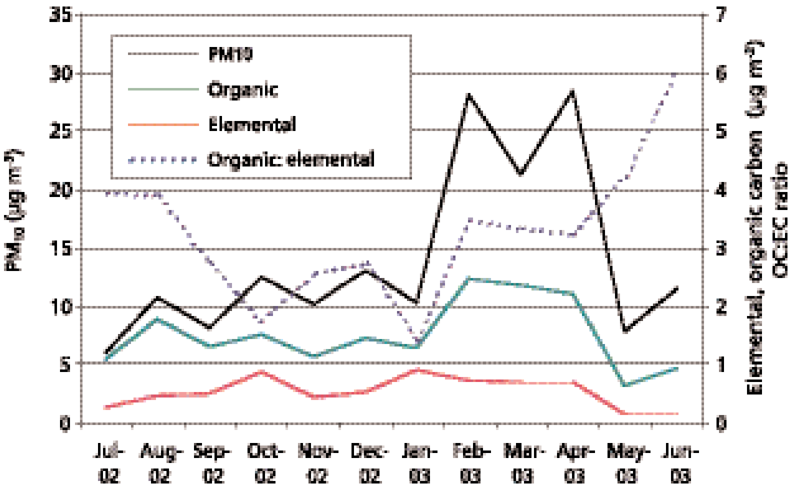


Figure 6.23 Monthly values of PM_{10}, organic and elemental carbon and organic:elemental carbon ratio, Bush Estate.



545. Weekly 24-h samples of PM_{10}, organic and elemental carbon have been collected at a rural site at Bush Estate in southern Scotland (EMP/CCC, 2004). The weekly samples appear in Figure 6.22 and as monthly means in Figure 6.23. The latter clearly shows an increased organic to elemental carbon ratio in the summer months, attributable to a higher proportion of secondary organic carbon at this time of the year. To convert organic carbon concentrations to the mass of organic matter, a factor of 1.4 is typically used.

6.1.4.6 Airborne chloride measurements at inland sites

546. During the measurement of the chemical components of airborne particulate over 24-h periods at three pairs of roadside and background sites in London and one pair of sites in Birmingham, it was observed that the chloride ion concentration tended to be higher when the sulphate and nitrate ion concentrations were low and vice versa. There were no significant differences between the mass of chloride at the roadside and background sites.

547. Air mass trajectories at the 950 mb level (obtained from British Atmospheric Data Centre), with end times at the middle of the sampling period, were used to calculate the distance over sea that the air mass had travelled during the previous 24 h. A simplified Western European coastline was assumed for this purpose. The assumed coastline, and examples of the 24-h trajectories of maritime and continental air masses, are shown in Figure 6.24.

548. The mass of chloride is plotted against the distance travelled over sea in the previous 24 h is shown in Figure 6.25. The distance travelled over sea in the previous 24 h is equivalent to the speed of the air over the sea multiplied by the proportion of that period spent over the sea. Although the data are scattered there is a clear trend of increasing chloride concentration with increasing distance travelled over the sea. The use of a 24-h trajectory to calculate distance over the sea gave a better correlation than other time periods. The equations for plots of chloride concentration against distance over sea for various periods fitted by least squares are presented in Table 6.8.

Table 6.8 Relationship of chloride concentration (μg m^{-3}) to distance travelled over the sea (km).

Period over sea		**Relationship between chloride concentration and distance travelled over sea**	r^2
Previous 24 h	0 to –24 h	$y = 0.0025\,x + 0.0995$	0.43
Preceding 24-h period	–24 to –48 h	$y = 0.0015\,x + 0.0994$	0.28
Previous 48 h	0 to –48 h	$y = 0011\,x + 0.0591$	0.39

6.1.5 Mass closure model for airborne particulate at roadside and background sites

549. Ninety-seven roadside and background samples obtained during the Traffic Management and Air Quality (TRAMAQ) study were quantitatively analysed for iron, calcium, sulphate, nitrate, chloride, elemental carbon and organic carbon. These analytes were chosen as representative of the major components of airborne particulate matter and factors were derived to calculate the mass of the component present.

- Iron: representative of suspended soil and road dust. Factors of 5.50 for the roadside samples and 9.00 for the background samples were chosen to account for other components of the dusts.

- Calcium: representative of construction dust. No difference was seen in the concentrations of calcium at the roadside and background sites. A factor of 4.30 was used to convert the mass of calcium to gypsum ($CaSO_4.2H_2O$).

- Sulphate: representative of ammonium sulphate, a major secondary component of airborne particulate. After subtracting a proportion of the

Figure 6.24 Relationships between chloride concentration and distance travelled over sea during various time periods.

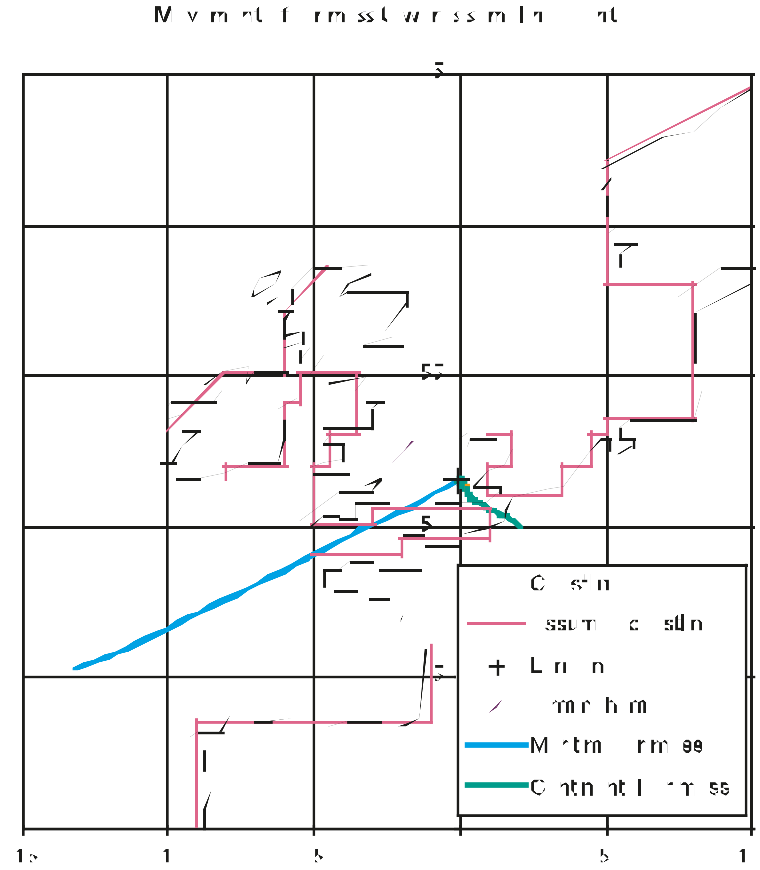
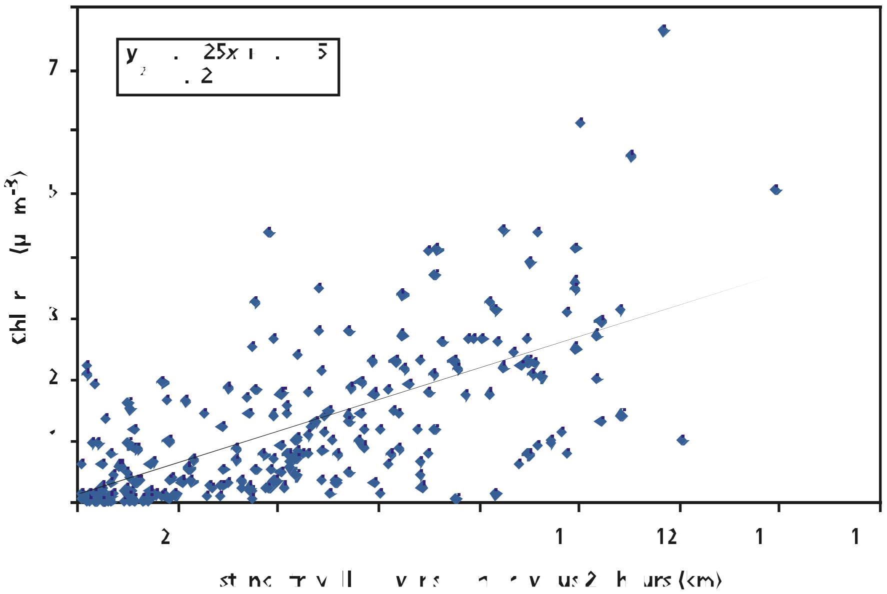


Figure 6.25 Concentration of chloride measured versus distance air mass travelled over sea in previous 24 h.



mass of calcium to allow for the presence of calcium sulphate, a factor of 1.38 was used to convert the mass of the remaining sulphate to ammonium sulphate.

- Nitrate: representative of ammonium nitrate and sodium nitrate, other major secondary components of airborne particulate. A factor of 1.29 was used to calculate the mass of NH_4NO_3 from the mass of the fine fraction of nitrate, and a factor of 1.38 was used to calculate the mass of $NaNO_3$ from the mass of the coarse fraction of nitrate.

- Chloride: representing marine aerosol. A factor of 1.65 was used to convert the mass of chloride to NaCl.

- Elemental carbon: resulting from combustion processes. It is a form of graphitic carbon and, therefore, a factor of 1.00 was used.

- Organic carbon: representing a combination of natural organic material and combustion products. Factors of 1.30 and 1.40 were used respectively to convert the roadside and background masses of organic carbon to the masses of organic compounds.

- Bound water: sulphates and nitrates are highly hygroscopic. A factor of 0.29 was used to calculate the mass of bound water from the masses of ammonium sulphate, ammonium nitrate and sodium nitrate.

This scheme is explained in more detail elsewhere (Harrison *et al*., 2003).

550. With the exception of the factors for iron, organic carbon and bound water, all the factors chosen were derived from the molecular weights of the compounds. The factors were used to calculate the total masses of PM_{10}, PM_{coarse} and $PM_{2.5}$ for each set of measured masses at the roadside and background sites and for the mass differences between the sites. Reduced major axis regression lines were fitted to the calculated and measured mass data and the factors chosen for iron, organic carbon and bound water were adjusted to obtain the most advantageous fitted line (gradient 1:1; intercept = 0).

551. The equations of the fitted lines are presented in Table 6.9 along with the squared correlation coefficients.

552. The calculated values of PM_{10} and PM_{coarse} at the roadside are plotted against the measured values in Figures 6.26 and 6.27, respectively. The fitted lines are good representations of the data. In the case of PM_{coarse} there is a greater scatter of the points about the line as would be expected from the lower value of the squared correlation coefficient.

553. The model, although simple, accounts for a large part of the variance in the mass concentration and encompasses the major components of the airborne particulate. The different factors chosen for iron and organic carbon at the roadside and background sites are compatible with there being higher concentrations of iron from mechanical wear and organic carbon from vehicle exhausts at the roadside sites. Trace metals and PAHs, discussed above, although

Table 6.9 Regression equations for lines fitted to the calculated and measured masses.

	Site	RMA regression equation	r^2	*n*
PM_{10}	Roadside	$y = 1.00\,x + 1.33$	0.92	97
PM_{coarse}	Roadside	$y = 1.00\,x + 0.19$	0.73	97
$PM_{2.5}$	Roadside	$y = 1.00\,x + 0.97$	0.93	97
PM_{10}	Background	$y = 0.99\,x + 2.50$	0.84	97
PM_{coarse}	Background	$y = 1.00\,x + 0.58$	0.61	97
$PM_{2.5}$	Background	$y = 1.01\,x + 1.52$	0.90	97
PM_{10}	Difference	$y = 1.08\,x - 0.96$	0.79	97
PM_{coarse}	Difference	$y = 0.99\,x - 0.11$	0.66	97
$PM_{2.5}$	Difference	$y = 1.19\,x - 1.52$	0.72	97

of potential importance in toxicological terms, are minor components, not analysed directly in the model. The PAHs comprise a component of the organic carbon, whereas the trace metals, unless soil-derived, are not accounted for explicitly, but represent only a very small proportion of the mass.

Figure 6.26 Relationship of calculated PM_{10} and weighed PM_{10} at roadside sites.

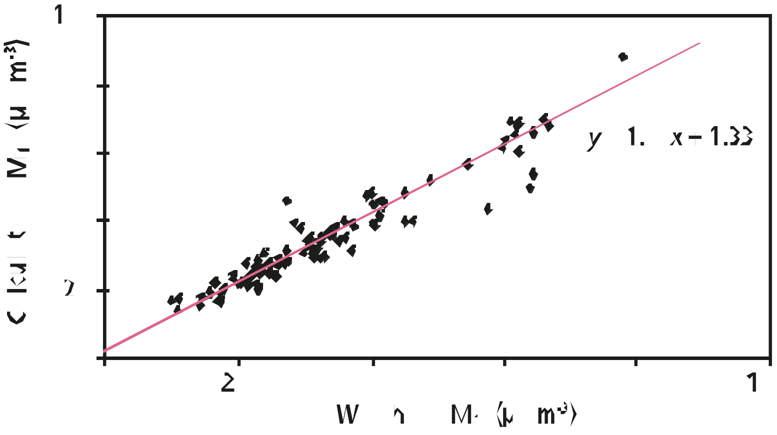
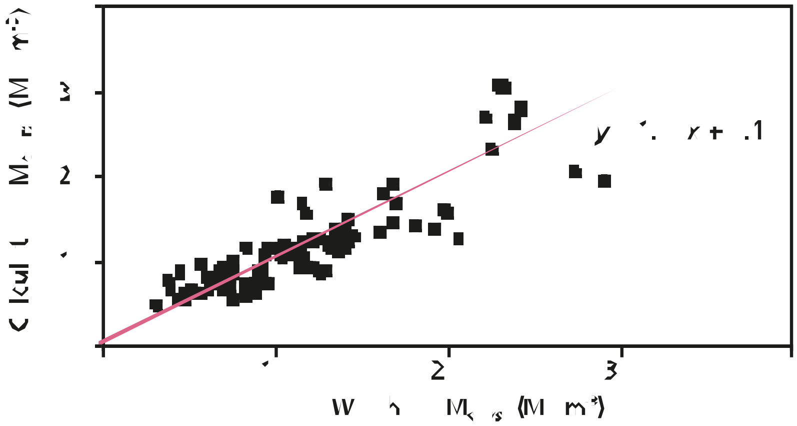


Figure 6.27 Relationship of calculated PM_{coarse} and weighed PM_{coarse} at roadside sites.



6.1.6 Chemical composition of roadside and urban background particles and the roadside increment

554. The mean chemical components of the airborne particulate calculated in the TRAMAQ study at the roadside and background sites are shown in Figures 6.28 to 6.31 for both coarse and fine PM fractions. The mean masses of each component are shown on the figures. Iron-rich dust is the largest component of both the roadside and background coarse PM, with a larger mass at the roadside sites. The masses of the other components are similar at both sites. Within the fine fraction, similar masses of ammonium nitrate, ammonium sulphate and bound water are present at both roadside and background sites, with the sum of organic compounds and elemental carbon making up two-thirds, and one-third of the of the total fine particulate at the roadside and background sites respectively.

Figure 6.28 Chemical components (μg m^{-3}) at roadside sites – coarse fraction.

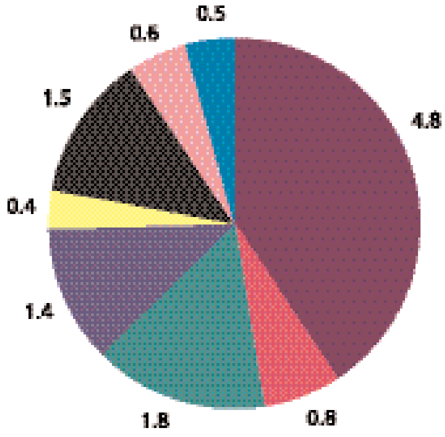


Figure 6.29 Chemical components (μg m^{-3}) at background sites – coarse fraction.

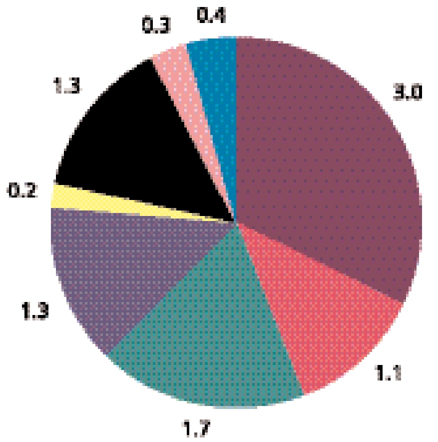


Figure 6.30 Chemical components (μg m^{-3}) at roadside sites – fine fraction.

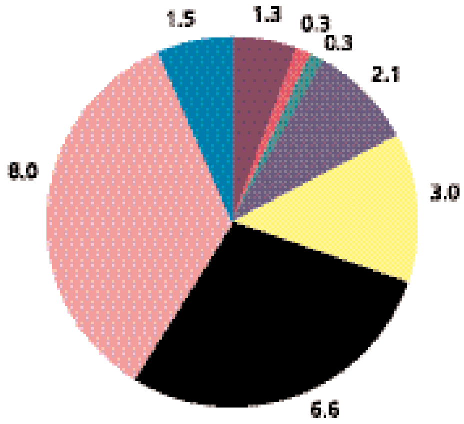
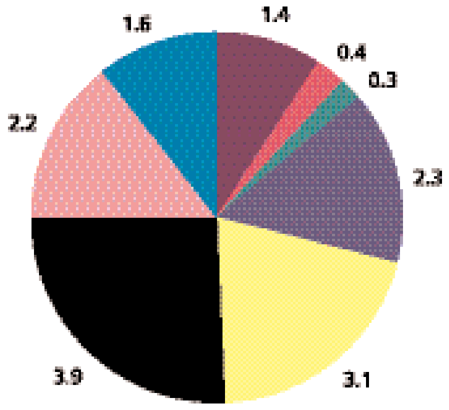


Figure 6.31 Chemical components (μg m^{-3}) at background sites – fine fraction.



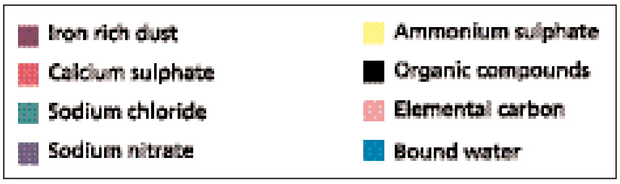


Figure 6.32 Roadside – background difference (µg m^{-3}) – coarse fraction.

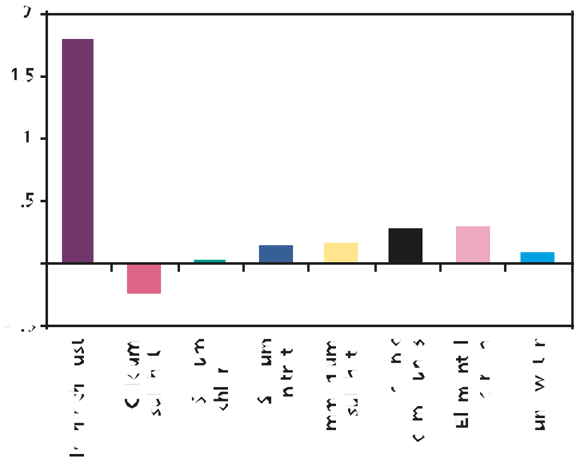
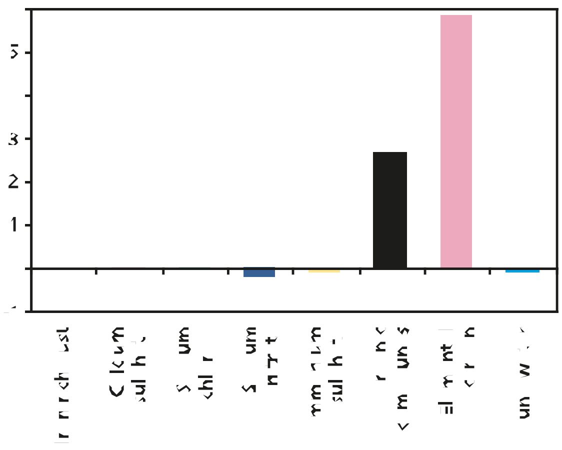


Figure 6.33 Roadside – background difference (µg m^{-3}) – fine fraction.



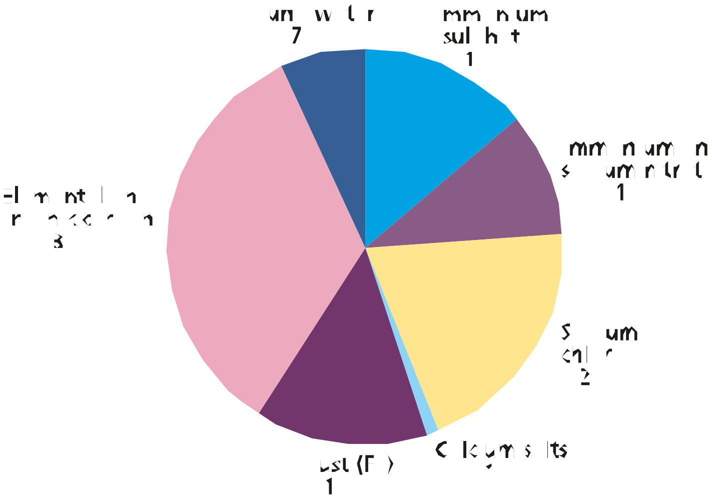
555. The differences between the calculated masses of the components of the airborne particulate matter in the coarse and fine fractions are shown in Figures 6.32 and 6.33. The difference in the coarse material is dominated by the iron-rich dust, with smaller contributions from sodium nitrate, ammonium sulphate, organic compounds and elemental carbon. Calcium sulphate was higher at the background sites than at the roadside sites. This may be due to localized sources of calcium sulphate, possibly from construction activities. In the fine fraction the difference between the roadside and background sites is dominated by the organic compounds and elemental carbon.

6.1.7 Apportionment of urban background PM$_{10}$ in Glasgow using mass-closure model

556. The mass closure model developed by Harrison *et al*. (2003) was applied to measurements of chemical composition of PM$_{10}$ at an urban background site in Glasgow. The 182 sets of measurements were nearly all from 24-h samples (although some samples were 48 h) collected between August 1999 and September 2000. In this work, measured nitrate could not be apportioned between ammonium or sodium nitrate, so a single multiplier of 1.33 was applied to the mass of nitrate measured, which is an average of the multipliers of 1.29

and 1.37 required to scale nitrate to ammonium and sodium nitrate, respectively. In addition, carbon was not apportioned between elemental and organic, and a single multiplier of 1.3 was used to give best agreement in regression of aggregated mass-closure mass on measured PM_{10}. Figure 6.34 shows the mean chemical apportionment for this site. The mean PM_{10} was 18.9 µg m^{-3} (gravimetric). Approximately one-third, on average – that is, ~7 µg m^{-3} (gravimetric) – of PM_{10} at this urban background site in Glasgow is constituted by primary mechanically (wind)-generated particles, that is, dust and sea salt. The relatively high proportion of sea salt is due to Glasgow's west coast location. Approximately one-third of the PM_{10} is due to secondary inorganic aerosol. The remaining one-third is carbonaceous aerosol, although this cannot be apportioned between primary elemental carbon from local sources and organic carbon either from local sources or as longer range secondary organic aerosol.

Figure 6.34 Apportionment of urban background PM_{10} in Glasgow using the mass closure model of Harrison *et al*. (2003). Proportions are means of *n* = 182 samples.



6.2 Spatial distribution of PM

6.2.1 Spatial distribution of PM_{10} within and between Edinburgh and Glasgow

557. Measurements of daily PM_{10} at two separate urban centre sites in Edinburgh and at two separate urban centre sites in Glasgow provide evidence of a reasonably strong spatial correlation in background PM_{10} for both cities, see Figures 6.35 and 6.36 ($r = 0.64$ and $r = 0.56$ for Edinburgh and Glasgow, respectively).

558. The value of the correlation coefficient ($r = 0.50$) between the mean of the PM_{10} from the two sites in Edinburgh and the mean of the PM_{10} from the two sites in Glasgow for almost 1 year's measurements is almost as strong as the correlation in PM_{10} within each city (Figure 6.37). Although these two cities are separated by only ~80 km, these data indicate that for roughly comparable cities in the UK, a significant proportion of variability in PM_{10} is controlled by regional-scale sources and synoptic meteorology.

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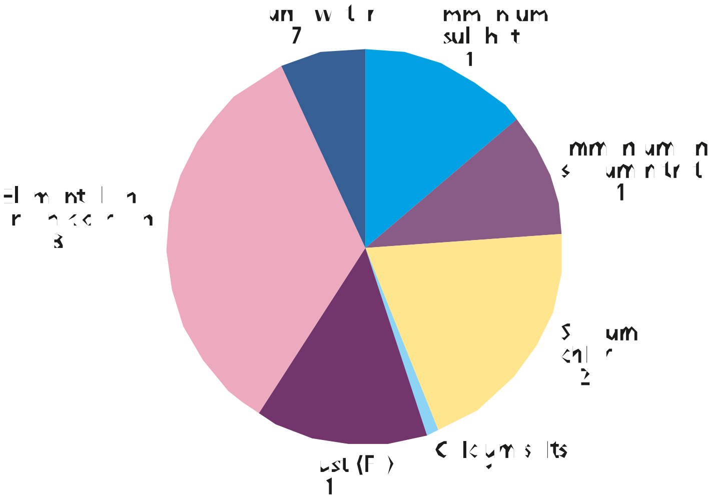

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Figure 6.35 Correlation between daily PM_{10} at two sites in Edinburgh (1999–2000).

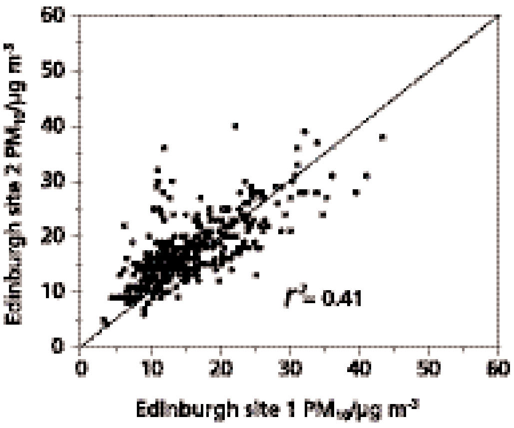


Figure 6.36 Correlation between daily PM_{10} at two sites in Glasgow (1999–2000).

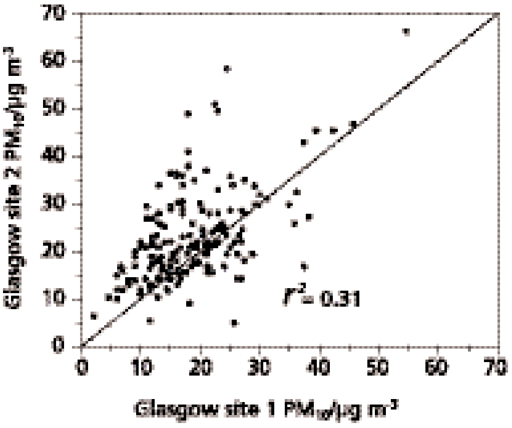
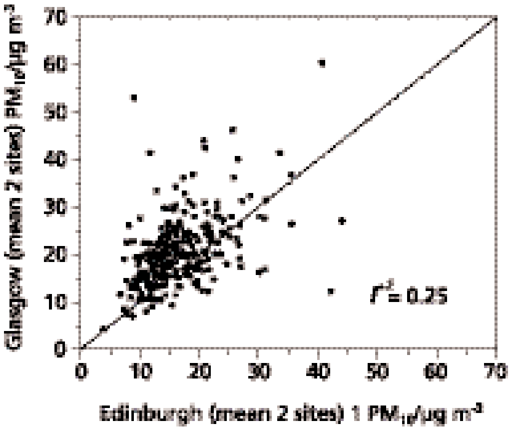


Figure 6.37 Correlation between mean daily PM_{10} at two sites in Edinburgh and two sites in Glasgow (1999–2000).

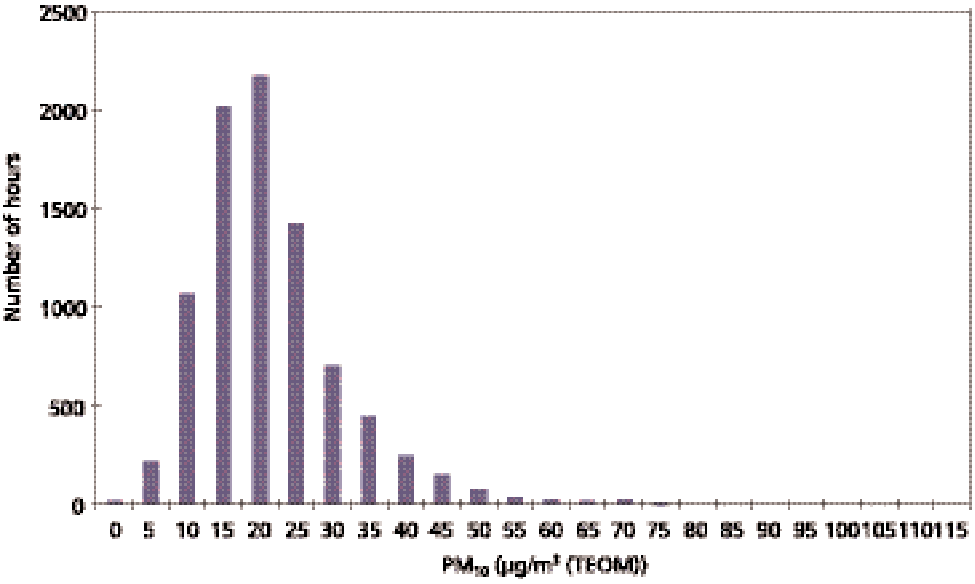


6.2.2 Regional distribution of PM_{10} and $PM_{2.5}$ concentrations across London

559. An analysis of the frequency distribution of the hourly PM_{10} concentrations observed at the London North Kensington site during 2002 (see Figure 6.38) reveals the presence of a long tail of hourly values that reach to levels that are over five-times the median concentration. In contrast, the frequency distribution shows that concentrations that are one-fifth of the median concentrations are relatively rare. A simple explanation of the relative absence of low concentrations would be that there is a background concentration below which hourly values are infrequent. Examination of the frequency distributions observed at other sites in the LAQN shows that this is a universal feature.

560. To understand better the nature of the background concentration observed in the frequency distribution at the London North Kensington site, a detailed analysis was made of the spatial distribution of the PM_{10} observations compiled in the LAQN. Two transects, 20 km-wide, were constructed through the London conurbation in a north-south and a east-west direction, and the PM_{10} concentrations were plotted for those sites that fell within the transects. Figure

Figure 6.38 The frequency distribution of the hourly mean PM_{10} concentrations observed during 2002 at the London North Kensington site.



6.39 shows the distribution of PM_{10} concentrations in the north-south transect and Figure 6.40 in the east-west transect. Each transect passed through central London and all data were the annual mean concentrations for 2002.

561. Broadly speaking, PM_{10} levels rose towards the middle of each transect for both sites away from traffic and those influenced by traffic and fell off towards the ends of the transects. There is clear evidence within both these plots for a background concentration upon which the London contribution stands. In Figures 6.39 and 6.40, the presence of such a background is indicated by the straight line set at 16.5 µg m^{-3}.

562. The value of 16.5 µg m^{-3} for the regional background across London was chosen as follows. From the LAQN, ten sites were selected that were located on the periphery of London and away from the immediate influence of traffic. The sites were: Crawley background, Dacorum background, East Hertfordshire background, Hertsmere background, Horsham background, Mole Valley 1, North Hertfordshire background, Sussex mobile, Luton background and Thanet Airport. The mean of the sites was calculated for each hour of 2002, and an hourly time series of background concentrations was constructed. The annual mean of this synthetic time series was 16.5 µg m^{-3}.

563. This background concentration was then subtracted from each hourly value for each site in the LAQN to obtain a time series of background-corrected values. The 47 sites that were sited away from traffic exhibited an average annual PM_{10} concentration of 19.9 µg m^{-3} and of 4.6 µg m^{-3} when the hourly background concentrations were subtracted. The sites that were under the influence of traffic showed an average annual PM_{10} concentration of 26.0 µg m^{-3} and of 9.3 µg m^{-3} when the background was subtracted. On this basis it appears that a substantial fraction of the observed PM_{10} concentrations observed at sites within the LAQN can be attributed to the regional background and that a relatively smaller fraction was due to emissions within the London conurbation.

Figure 6.39 Annual PM_{10} concentrations for 2002 for those LAQN sites within a 20 km-wide transect that passes north-south through the London conurbation.

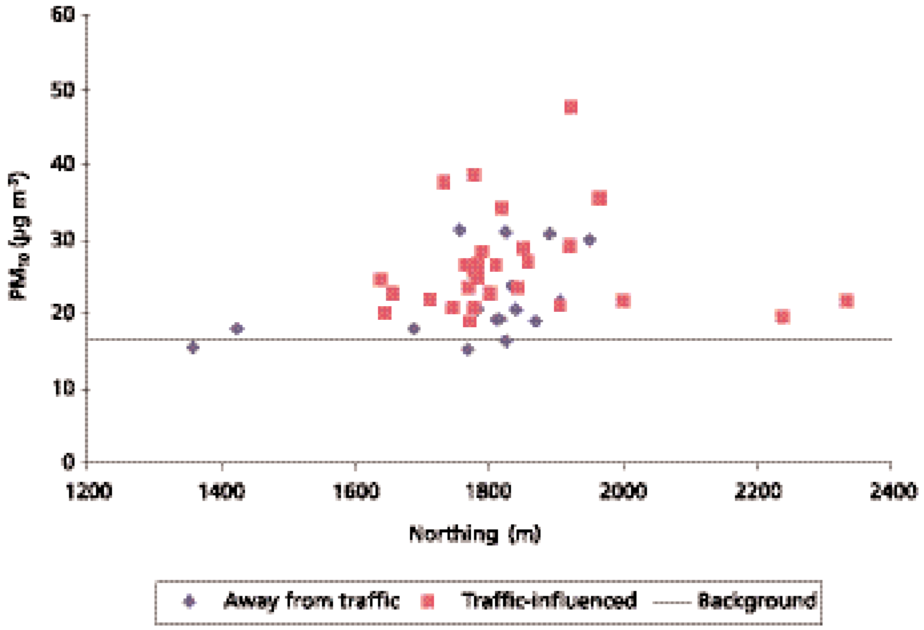
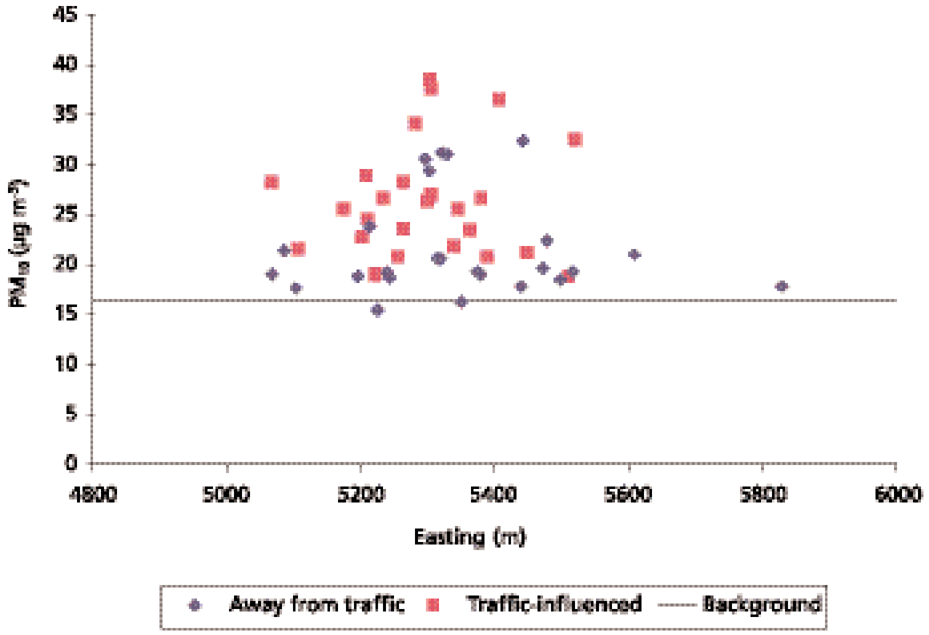


Figure 6.40 Annual PM_{10} concentrations for 2002 for those LAQN sites within a 20 km wide transect which passes east-west through the London conurbation.



564. The corresponding analysis for $PM_{2.5}$ is hampered by the availability of $PM_{2.5}$ monitoring sites but the main principles still apply. The frequency distribution of the hourly $PM_{2.5}$ concentrations observed at the Ealing roadside site show the presence of a background below which observed concentrations are infrequent. Two sites were selected for the attribution of a background concentration for each hour of 2002 and this value was subtracted from each of the $PM_{2.5}$ observations at the LAQN sites. The background sites selected were the London Bexley and Rochester sites and an annual mean background concentration of 10.9 µg m^{-3} was inferred. The annual mean of the four LAQN sites monitoring

$PM_{2.5}$ in 2002 was 15.2 µg m^{-3}, 3.9 µg m^{-3} with the background subtracted. On this basis it was concluded that the regional background contributed a substantial fraction of the $PM_{2.5}$ concentrations observed across London.

6.2.3 Background concentration surfaces of PM_{10} in London

565. The spatial distribution of city-scale background concentrations of PM_{10} was assessed through interpolation of monitored data for selected sites in and around London. The use of monitoring data to investigate spatial patterns in concentrations is useful for complementing the more detailed spatial representations derived from air quality models. Interpolated maps of monitoring data tend to give a broad impression of the pattern of concentrations in a particular area. In this case, the maps estimate concentrations at background locations only. They do not include contributions from specific local sources, particularly roadside or kerbside locations, which would tend to increase concentrations from those shown in this section. Surfaces have been produced for annual average concentrations in 2003 and 2002 and for shorter averaging periods for two contrasting PM_{10} episodes during February 2003 and August 2003.

6.2.3.1 Annual mean background concentrations in London

566. Interpolated surfaces were generated using ordinary kriging with 25 TEOM sites from the London and the Southeast networks. Sites were discarded from the analysis if they were classified as kerbside or roadside or if there was a data capture rate of less than 60%. Due to a relatively uneven spatial distribution of the location of urban background sites, it was necessary to include suburban and urban centre sites in the analysis (see Figure 6.41 and Table 6.10). Monitored annual means for 2003 ranged between 23–32 µg m^{-3}. The standard scaling factor of 1.3 was applied, but it is recognised that this may not adequately account for the true local geographic variability in concentrations (Green *et al*., 2001).

Figure 6.41 Locations and siting characteristics of monitoring sites used to estimate the spatial distribution of annual mean concentrations.

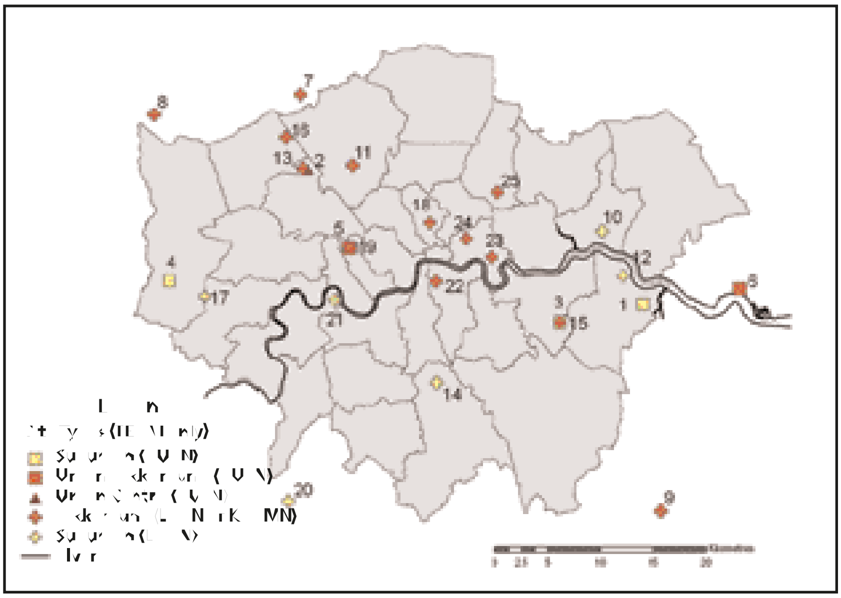


Table 6.10 Summary of 2003 PM monitoring data from sites shown in Figure 6.47.

Number	Name	Max daily mean TEOM* 1.3 ($\mu g\ m^{-3}$)	Date of max daily mean	Days >50 $\mu g\ m^{-3}$	Data capture (%)	Annual mean TEOM* 1.3 ($\mu g\ m^{-3}$)
1	London Bexley	79	26/03	33	87	27
2	London Brent	81	22/02	24	86	27
3	London Eltham	79	26/03	24	89	27
4	London Hillingdon	129	01/11	28	80	30
5	London North Kensington	77	08/08	28	88	29
6	Thurrock	135	09/04	38	88	31
7	Hertsmere (Borehamwood)	70	20/04	20	88	25
8	Three rivers background	75	22/02	27	78	23
9	Sevenoaks 2 – Greatness	77	29/03	14	91	23
10	Barking and Dagenham 2 – Scrattons Farm	78	20/03	43	89	32
11	Barnet 2 – Finchley	70	10/08	23	90	26
12	Bexley 2 – Belvedere	78	15/04	30	88	27
13	Brent 1 – Kingsbury	81	22/02	25	86	26
14	Croydon 3 – Thornton Heath	79	26/03	17	86	26
15	Greenwich 4 – Eltham	78	26/03	24	91	27
16	Harrow 1 – Stanmore background	72	11/08	16	85	23
17	Hounslow 2 – Cranford	77	22/02	20	88	26
18	Islington 1 – Upper Street	72	08/08	25	91	27
19	Kensington and Chelsea 1 – North Kensington	77	08/08	29	90	29
20	Mole Valley 2 – Lower Ashstead	73	20/04	14	91	25
21	Richmond 2 – Barnes Wetlands	86	16/04	33	90	27

Number	Name	Max daily mean TEOM* 1.3 (µg m^{-3})	Date of max daily mean	Days >50 µg m^{-3}	Data capture (%)	Annual mean TEOM* 1.3 (µg m^{-3})
22	Southwark 1 – Elephant and Castle	81	15/04	31	91	30
23	Tower Hamlets 1 – Poplar	168	15/04	42	88	31
24	Tower Hamlets 3 – Bethnal Green	72	10/08	28	85	27
25	Waltham Forest 1 Dawlish Road	66	20/04	16	69	25

567. Figure 6.42 shows the predicted distribution of annual mean PM_{10} concentrations in 2003. The overall root mean square (RMS) error for the predicted data was 2.2 µg m^{-3}, and an assessment of predicted against monitored data showed that there is a tendency for lower concentrations to be overpredicted and higher concentrations to be underpredicted. There were no background locations estimated to exceed an annual mean of 30 µg m^{-3} and also no locations with an estimated annual mean less than 23 µg m^{-3}.

568. As might be expected, Figure 6.42 shows a broad pattern of relatively high background concentrations through the centre of the city. It also suggests that there is a tendency for higher concentrations to occur towards the east of the city and lower concentrations towards the northwest and south. It should be noted that the limited number of background stations to the south of the conurbation may give additional uncertainty to the results presented here. Although the highest concentrations were estimated to occur in the northeast, this area is associated with relatively large error estimates (Figure 6.43).

569. Figure 6.44 shows estimated background annual mean PM_{10} concentrations for 2002. Here the annual mean concentrations are generally lower, with a lower RMS error of 1.5 µg m^{-3}. In 2002, there were no background locations estimated to exceed 28 µg m^{-3} and a number of areas to the south were estimated to be lower than 23 µg m^{-3}. It should be noted that the areas estimated to have a background concentration of 23 µg m^{-3} or lower tended to be associated with a relatively high degree of uncertainty (standard errors were ~1.5–2.0 µg m^{-3} (Figure 6.45)).

570. The results from both years show a similar pattern of background concentrations across London. The associated error maps are also broadly comparable and suggest that any additions to the existing network of background stations may be most beneficial in the south and northeast of the conurbation. The use of basic interpolation routines and background data, as in this example, would always tend to produce a more generalised map compared to the output of dispersion models that use emissions inventory data. The east-west gradient in concentrations suggested by the interpolated surfaces is not reflected in the maps produced from more detailed modelling studies (see Chapter 8). Since dispersion modelling of detailed emissions inventory data is the most reliable means of estimating the spatial distribution of annual mean PM_{10} concentrations,

Figure 6.42 Annual mean PM_{10} concentrations across London in 2003 interpolated from background sites. (The number of days with a daily mean >50 $\mu g\ m^{-3}$ is also given for sites included in the study.)

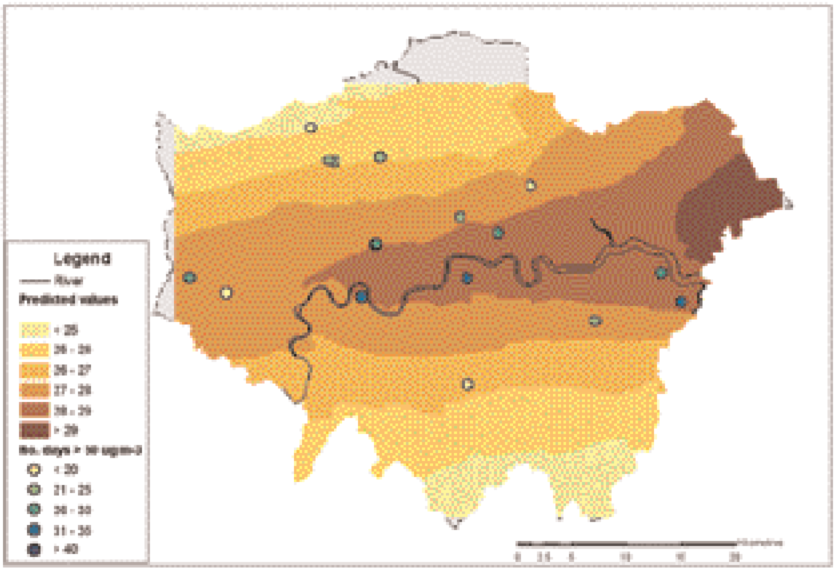
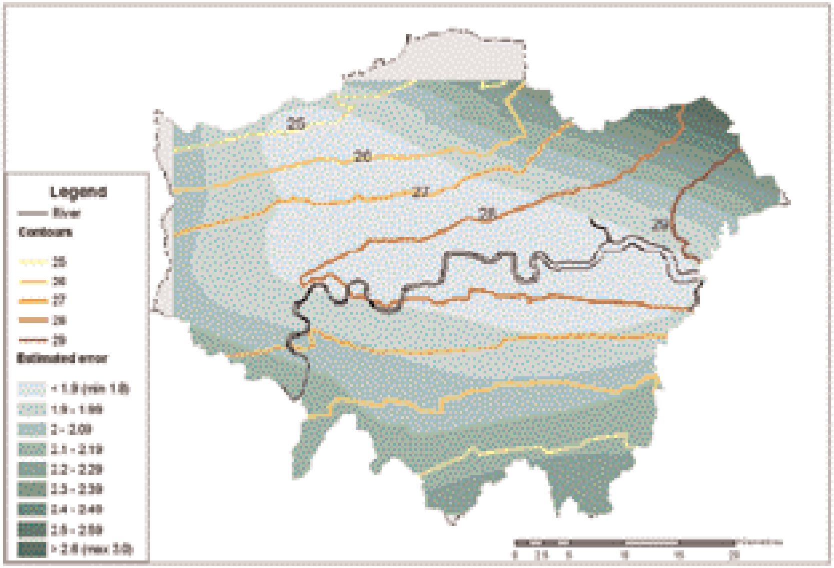


Figure 6.43 Error estimates associated with predicted PM_{10} annual average concentrations for 2003. (Predicted contours are shown for reference.)



Figures 6.42 and 6.44 should be treated with caution. Rather than the maps reflecting true geographic variability, observed patterns from the interpolated surfaces could be associated with data artefacts linked to the specific siting characteristics of the stations shown in Table 6.1. These might include differences in the nature of local sources and their relative proximity to individual sites. More sophisticated interpolation-based approaches would be required to remove some of these uncertainties.

571. Maps generated from a dataset including both ß-attenuation monitor and TEOM monitoring stations are not reproduced in this section due to the uncertainty in generating comparable annual means. Following the application of the 1.3 scaling factor to TEOM sites, the highest concentrations were still clearly associated with

Figure 6.44 Annual mean PM_{10} background concentrations across London in 2002 interpolated from background sites. (The number of days with a daily mean >50 µg m^{-3} is also given for sites included in the study.)

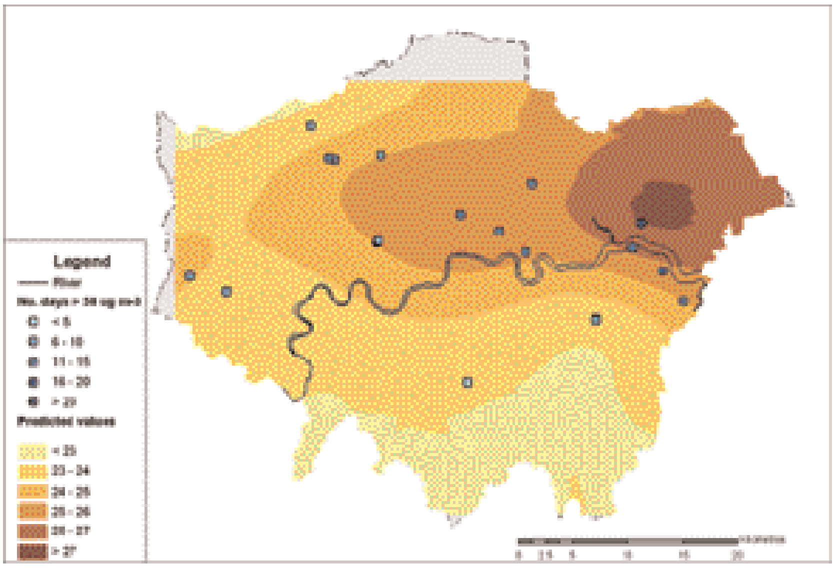
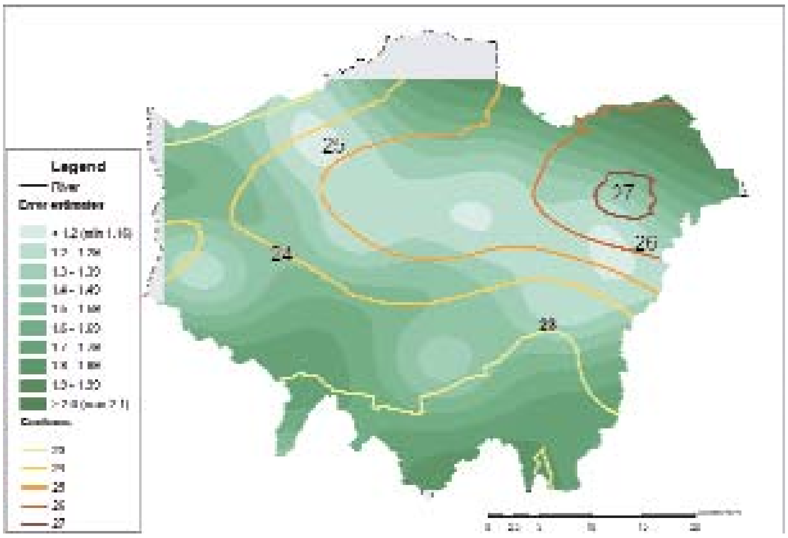


Figure 6.45 Error estimates associated with predicted PM_{10} annual average concentrations for 2002. (Predicted contours are shown for reference.)



ß-attenuation monitor sites. The top three annual mean concentrations for 2003 were 35 µg m^{-3} (Haringey 2 Priory Park), 36 µg m^{-3} (Redbridge 1 Perth Terrace) and 38 µg m^{-3} (Lambeth 3 Loughborough Junction) – considerably larger than the highest TEOM annual mean of 32 µg m^{-3} (Barking and Dagenham 2 – Scrattons Farm). This issue is discussed further in Annex 4.

6.2.3.2 Spatial distributions of PM_{10} in London under episode conditions

572. Surfaces showing the spatial distribution of concentrations associated with two particular episodes (labelled A and D on Figure 6.46), which occurred in London during 2003, have also been produced. Episode A occurred during the period 17th to 23rd February and was characterised by a high secondary particulate contribution from continental sources but with an important primary component.

Episode D was especially dominated by secondary particulate from continental sources and was a period of very high temperatures and photochemical activity that led to elevated ozone concentrations (Fuller, 2003). (See Chapter 8 for further discussion of the episode.)

573. Unscaled hourly concentration data were provided for 33 TEOM sites from the London and Southeast air quality networks and used to generate scaled 24-h means for the periods of interest. The data for February have been ratified but the data for August are a mix of ratified and non-ratified data. Due to incomplete data, not all of the sites may be included in each of the surfaces produced.

574. Figure 6.47 shows that during the February episode, the highest concentrations tended to be localized to the urban area of Greater London, with relatively high concentrations also estimated for North Kent. For some of the days, for example the 19th and 21st, there was also some evidence of relatively high concentrations on the southern coast, although there was a higher degree of error for these areas due to estimates being based on few data points (Figure 6.48).

575. The spatial pattern of the August episode (Figure 6.49 upper) appears to have had a wider impact over the entire Southeast region. Many of the surfaces show less variability in the error estimates than was the case for the February episode (Figure 6.49 lower). Although there is still some evidence of a distinct London effect, there are very high values estimated for the area north of the city (on 8th and 10th) and towards the south coast (on 11th).

Figure 6.46 Daily mean PM_{10} monitored at a typical London background site Kensington and Chelsea 1 – in 2003 (Fuller, 2003).

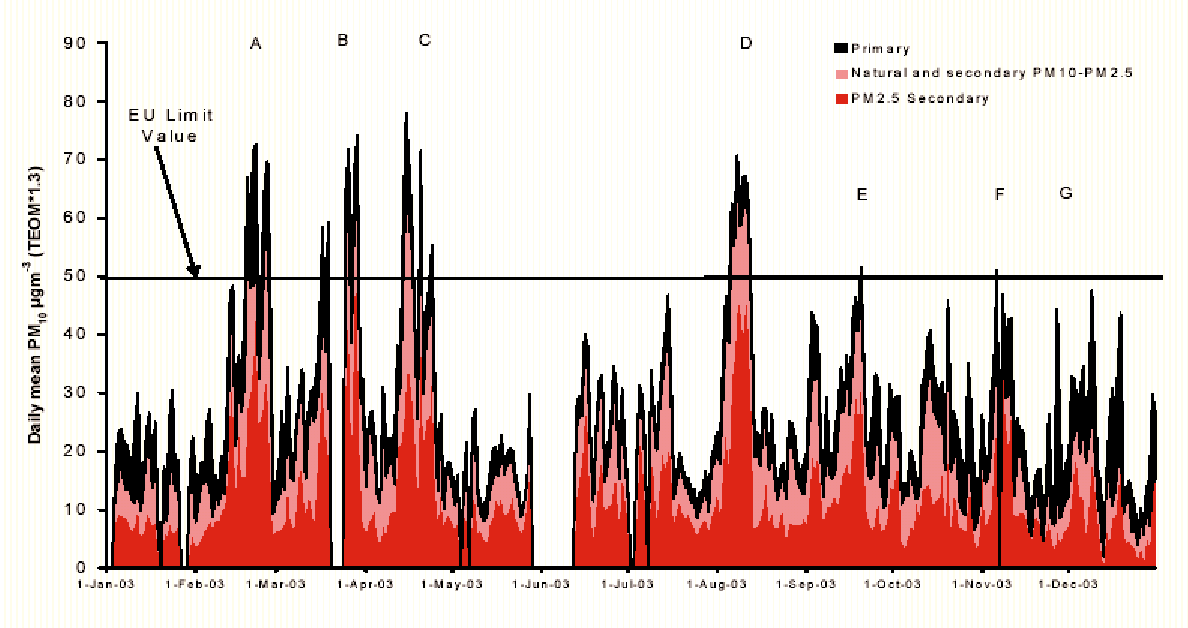


Figure 6.47 Estimated spatial distributions of daily mean concentrations of PM_{10} during 18th to 23rd February 2003.

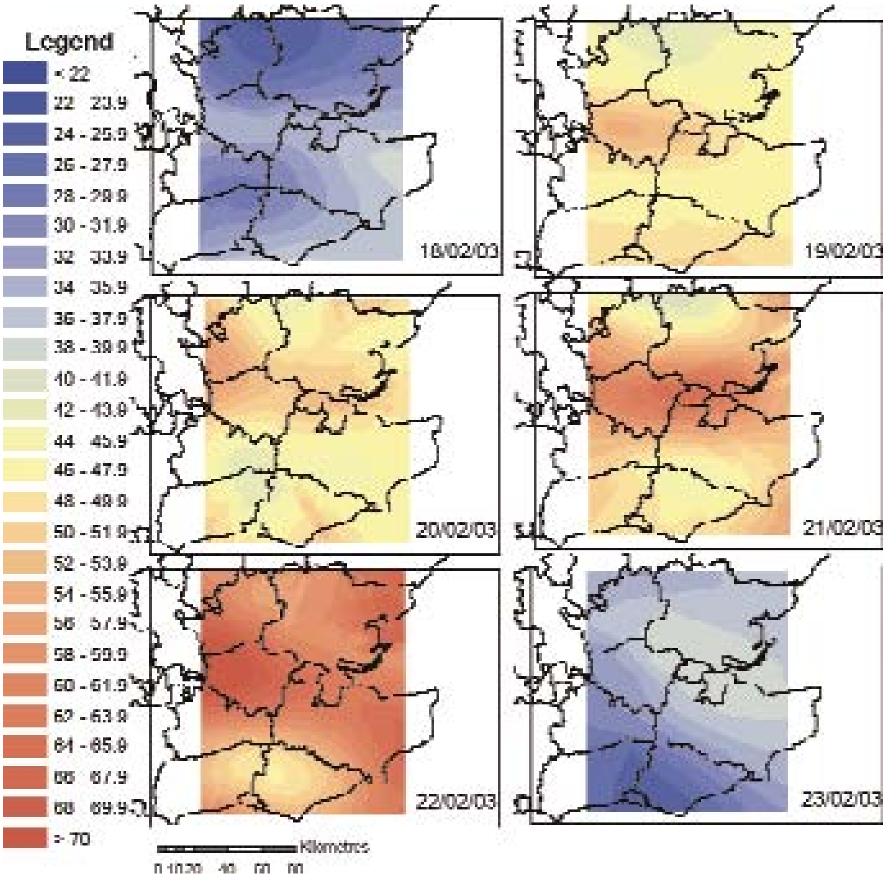


Figure 6.48 Estimated spatial distribution of prediction errors during 18th to 23rd February 2003 (μg m^{-3}).

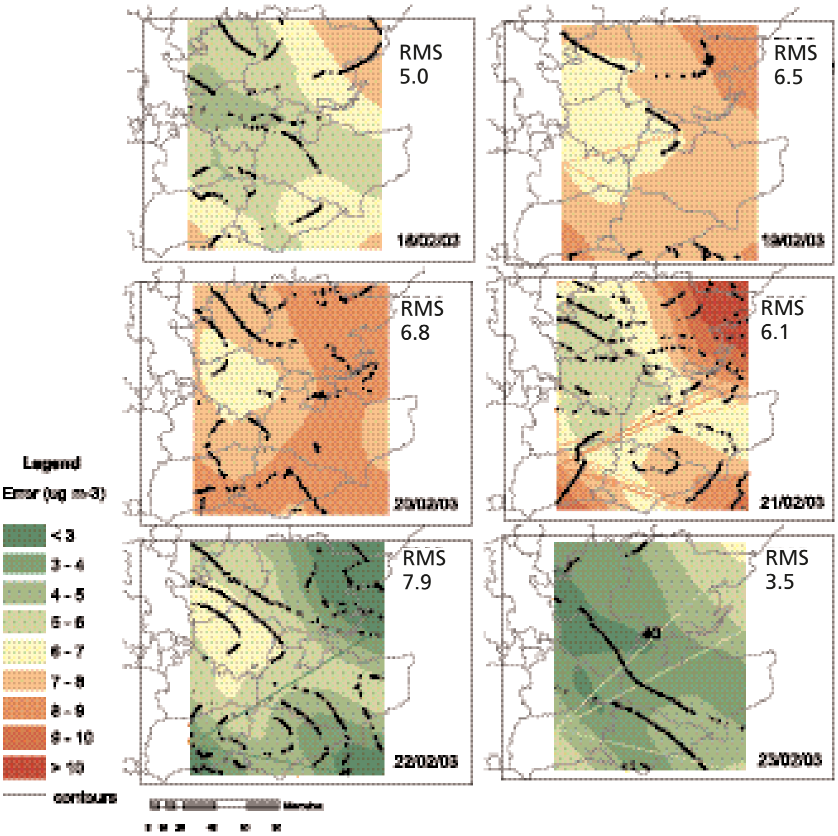
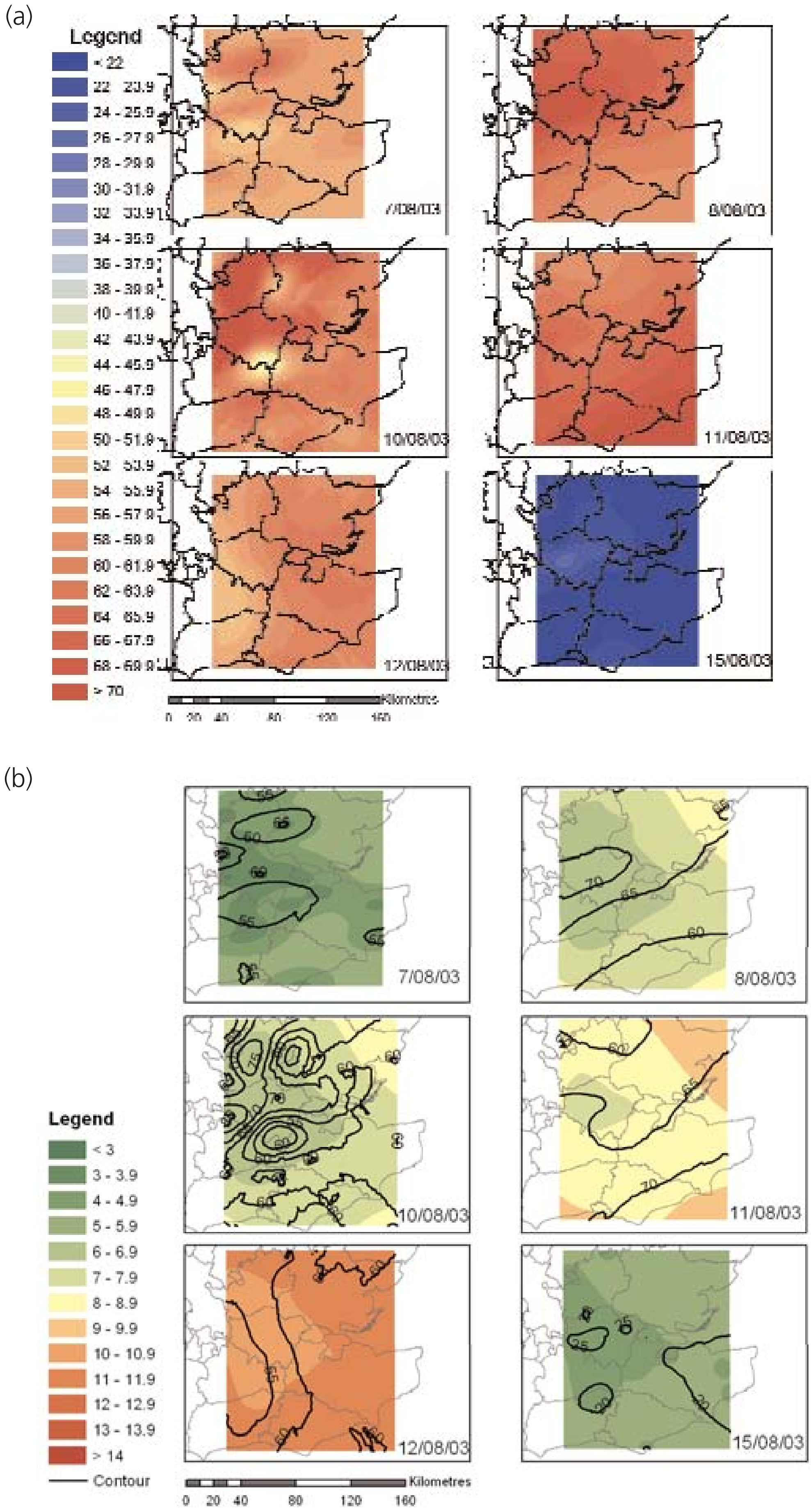


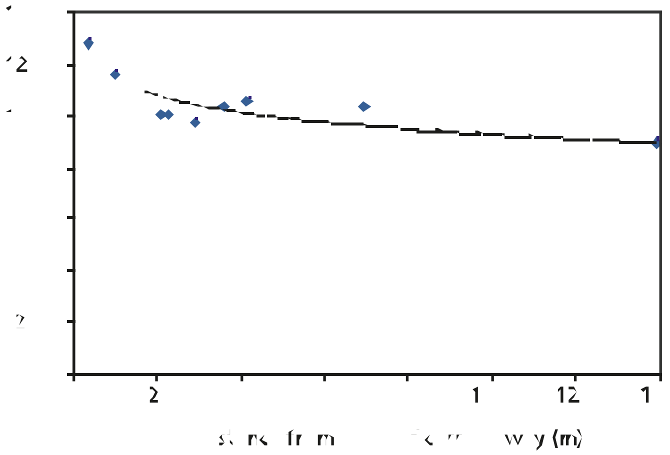
Figure 6.49 (a) and (b) Estimated spatial distributions of daily mean concentrations of PM_{10} during 7th to 15th August 2003 (µg m^{-3}).



6.2.4 Roadside concentration distribution

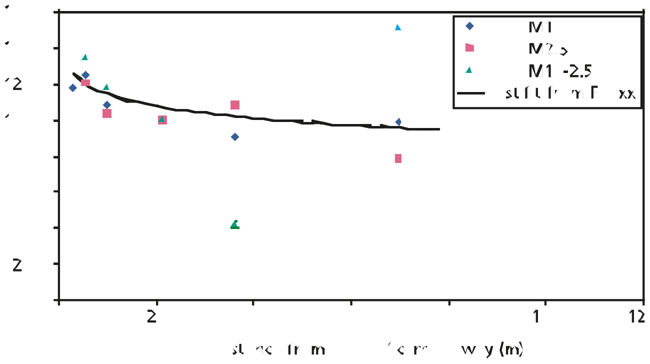
576. The spatial variation of PM_{10} concentrations alongside roads has not been studied in great detail. Only one study has been identified that shows the way in which concentrations decline away from roads. This study involved the use of MiniVol samplers alongside the M25 west of London (Hickman *et al.*, 2002). (REF to be supplied to list) These are gravimetric samplers that were used to collect daily samples for a total of 28 days split between November 1998 and March 1999. Comparison of the results with a TEOM sampler at the same location showed good agreement with TEOM * 1.3 values. Concentrations were measured on a transect both east and west of the motorway. The combined results for both transects, normalised to 100% at ~20 m from the edge of the carriageway, are illustrated in Figure 6.50. There is only evidence of a significant increase above the background within about 30 m of the carriageway. Beyond 20–50 m from the edge of the road, concentrations will be essentially indistinguishable from the local background, taking account of measurement uncertainty and the normally high background contribution to measured roadside concentrations.

Figure 6.50 PM_{10} gravimetric concentrations measured on transects away from the M25 motorway, normalised to 100% at ~20 m from the edge of the carriageway. (The data points have been fitted using a logarithmic relationship.)



577. An additional survey was carried out to the west of the M25 motorway for 14 days in September 2000. On this occasion measurements were also made of $PM_{2.5}$. The ratios of $PM_{2.5}$:PM_{10} ranged from 0.63 to 0.81, averaging 0.71,

Figure 6.51 PM_{10}, $PM_{2.5}$ and PM_{coarse} (gravimetric) concentrations measured on a transect away from the M25 motorway, normalised to 100% at ~20 m from the edge of the carriageway. (The line is the best-fit relationship from Figure 6.50.)



showing no systematic trend with distance. The results normalised to 100% at ~20 m from the edge of the carriageway are presented in Figure 6.51. Values for PM_{coarse}, calculated by difference, are also shown. The values in this figure are for the 10 days during which full datasets were available. Also shown in Figure 6.51 is the best-fit line taken from the earlier period, when more data were available (that is, from Figure 6.50). The same general pattern is seen as for the 1998–1999 monitoring, except that PM_{coarse} is a much noisier signal.

6.2.5 Roadside increments of PM_{10}, $PM_{2.5}$ and PM_{coarse}

578. As part of the TRAMAQ study, Partisol Plus model 2025 dichotomous manual gravimetric samplers were used to collect simultaneous roadside and background samples at four different locations – Elephant and Castle, High Holborn and Park Lane in London and Selly Oak in Birmingham – with instruments being moved from site to site over a number of cycles to capture samples over all four seasons. The roadside sites were chosen on the kerbside of heavily trafficked roads, and the background sites were within 1 km of the roadside sites but at least 200 m from any busy road. The masses of the samples collected were determined in both fine ($PM_{2.5}$) and coarse ($PM_{2.5–10}$) fractions, and the PM_{10} mass was calculated from the masses of the two fractions. Fuller details of the sites and measurements are given in Harrison *et al*. (2004).

Table 6.11 The means (and standard errors) of the roadside, background and difference measurements of PM_{10}, PM_{coarse} and $PM_{2.5}$ measured at each location. (Units are gravimetric µg m^{-3}.)

	Roadside			**Background**			**Difference**		
	PM_{10}	**PM_{coarse}**	**$PM_{2.5}$**	**PM_{10}**	**PM_{coarse}**	**$PM_{2.5}$**	**PM_{10}**	**PM_{coarse}**	**$PM_{2.5}$**
Elephant and Castle	36.8 (3.6)	12.5 (0.9)	24.2 (2.9)	21.2 (1.9)	8.2 (0.5)	13.0 (1.6)	15.6 (2.0)	4.4 (0.6)	11.2 (1.4)
High Holborn	38.3 (1.9)	12.3 (0.7)	25.9 (1.6)	28.7 (1.9)	10.8 (0.6)	17.9 (1.6)	9.6 (0.6)	1.5 (0.4)	8.0 (0.3)
Park Lane (EW)	63.3 (3.9)	21.3 (1.6)	42.0 (3.7)	27.9 (3.6)	9.3 (0.9)	18.6 (3.5)	35.3 (1.3)	12.1 (0.9)	23.3 (1.2)
Park Lane (WW)	22.8 (2.0)	8.2 (0.6)	14.6 (1.6)	15.7 (1.3)	5.7 (0.4)	10.0 (1.1)	7.1 (0.9)	2.5 (0.3)	4.6 (0.6)
Selly Oak	25.1 (1.2)	9.1 (0.4)	16.0 (0.9)	16.4 (1.0)	7.0 (0.3)	9.4 (0.7)	8.7 (0.4)	2.1 (0.2)	6.6 (0.3)

579. The mean values of PM_{10}, PM_{coarse} and $PM_{2.5}$ measured at the roadside and background sites at each of the four locations are presented in Table 6.11. The means of the differences between the individual measurements at the roadside and background sites are also shown. Due to the wide variation in the differences for the Park Lane data, it is separated into two groups: easterly winds (EW), when the wind direction was between 0° and 180° and westerly winds (WW). Wind data were obtained from Heathrow Airport.

580. The roadside concentrations of PM_{10}, PM_{coarse} and $PM_{2.5}$ are plotted against their respective values at background sites for each of the four locations in Figures 6.52 to 6.55. Lines are fitted by the reduced major axis (RMA) regression due to the similar uncertainty in the values of concentration at both the roadside and background sites (Ayers, 2001). The equations for the lines and values of the squared correlation coefficient are given in Table 6.12.

Table 6.12 Results of regression analysis of roadside (*y*) versus background (*x*) concentrations.

Location		RMA regression	r^2
Elephant and Castle	PM_{10}	$y = 1.94\,x - 4.32$	0.88
	PM_{coarse}	$y = 1.79\,x - 2.14$	0.57
	$PM_{2.5}$	$y = 1.81\,x + 0.65$	0.91
High Holborn	PM_{10}	$y = 1.03\,x + 8.78$	0.90
	PM_{coarse}	$y = 1.27\,x - 1.33$	0.62
	$PM_{2.5}$	$y = 1.00\,x + 8.07$	0.96
Park Lane (EW)	PM_{10}	$y = 1.10\,x + 32.57$	0.90
	PM_{coarse}	$y = 1.79\,x + 4.72$	0.86
	$PM_{2.5}$	$y = 1.07\,x + 22.07$	0.90
Park Lane (WW)	PM_{10}	$y = 1.55\,x -1.52$	0.91
	PM_{coarse}	$y = 1.45\,x - 0.07$	0.75
	$PM_{2.5}$	$y = 1.39\,x + 0.63$	0.91
Selly Oak	PM_{10}	$y = 1.24\,x + 4.70$	0.88
	PM_{coarse}	$y = 1.15\,x + 1.05$	0.76
	$PM_{2.5}$	$y = 1.31\,x + 3.68$	0.90

581. It is notable that for High Holborn the regression lines for PM_{10} and $PM_{2.5}$ have a smaller gradient (approximately 1:1) and larger intercept than is the case for Elephant and Castle or Selly Oak. The constant increment of PM_{10} and $PM_{2.5}$ at all background concentrations at High Holborn indicates that the increment at this location is not determined by the factors that influence the level of the background concentrations. The High Holborn sites are within central London street canyons with building heights greater than the cross street distance between buildings. The Elephant and Castle and Selly Oak sites are considerably more open and meteorological factors are much more likely to affect particulate matter concentrations.

582. At Park Lane the nature of the regression depended upon the wind direction. In easterly winds the gradients of the PM_{10} and $PM_{2.5}$ regression equations are close to unity and the intercepts are large and positive, similar to the results obtained at High Holborn. In westerly winds the results at Park Lane are similar to those seen at the Elephant and Castle and Selly Oak locations. The Park Lane roadside site is on the eastern side of the road with a substantial building (The Intercontinental Hotel) at the rear of the footway. On the opposite side of the road is Hyde Park and an absence of buildings. During easterly winds it might be expected that a wind flow regime similar to that experienced in a street canyon would occur with winds passing over the top of the building and establishing a rotating flow in the lee of the building giving rise to the higher concentrations seen in Table 6.11, as the reverse flow brings PM from the direction of the road. No such rotating flow would be seen during westerly winds. While making these comparisons it should be noted that the number of measurements made during easterly winds at Park Lane was small (nine) and that the results are heavily influenced by individual measurements.

583. The effects of wind speed on PM_{10} concentrations at Park Lane are shown in Figure 6.56. In westerly winds there is a general decrease in concentration with wind speed, consistent with greater dilution of PM occurring in higher wind speeds. In easterly winds PM_{10} tends to increase with higher wind speeds, presumably as the rotor effects become stronger and the reverse flow draws more air onto the instruments from the roadway.

Figure 6.52 Elephant and Castle – roadside concentration versus background concentration.

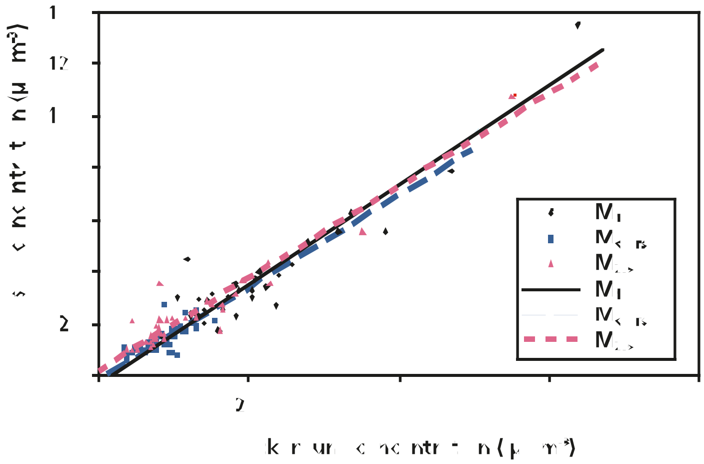


Figure 6.53 High Holborn – roadside concentration versus background concentration.

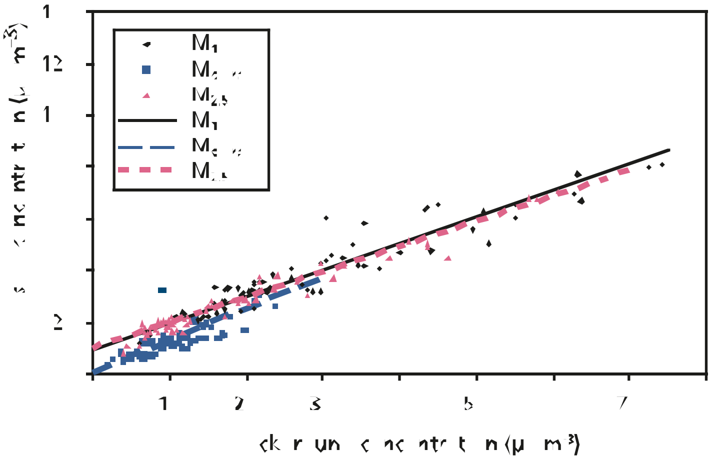


Figure 6.54 Park Lane – roadside concentration versus background concentration.

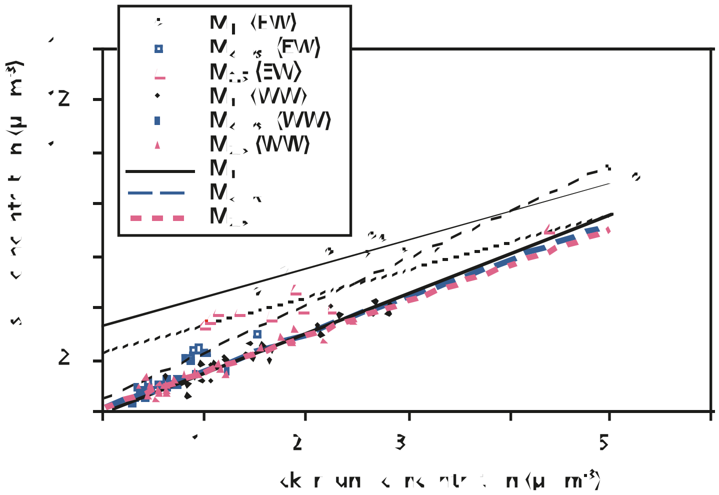


Figure 6.55 Selly Oak – roadside concentration versus background concentration.

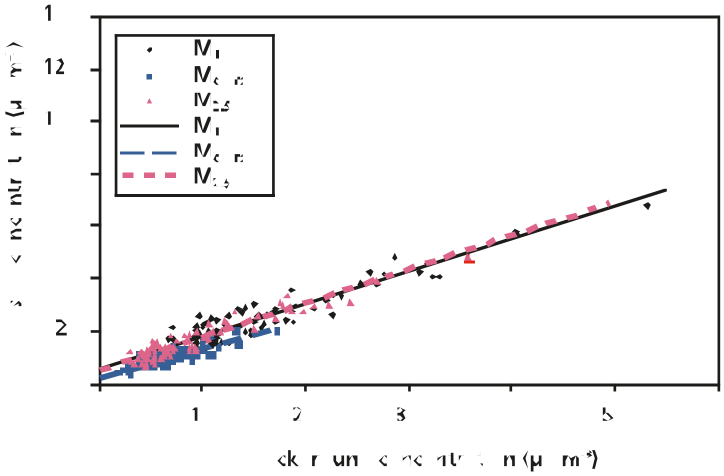
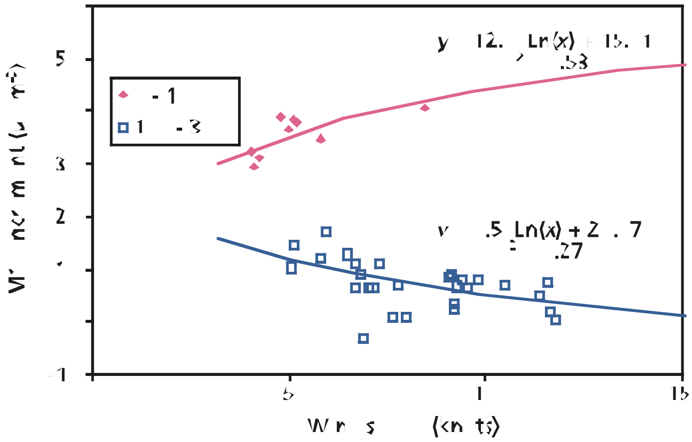


Figure 6.56 Park Lane – effect of windspeed on PM_{10} increment.



6.2.6 Spatial distribution of secondary constituents across the UK

584. Maps of sulphate, nitrate and ammonium (Figures 6.57 to 6.59) have been derived from measurements made by the Centre for Ecology and Hydrology (CEH). The sulphate and nitrate maps have been interpolated from monthly measurements at a network of 12 rural sites. The sulphate and nitrate maps have a southeast to northwest gradient. Ammonium shows similar gradients to those seen for nitrate.

Figure 6.57 Annual mean sulphate concentration (μg m^{-3}) for 2002.

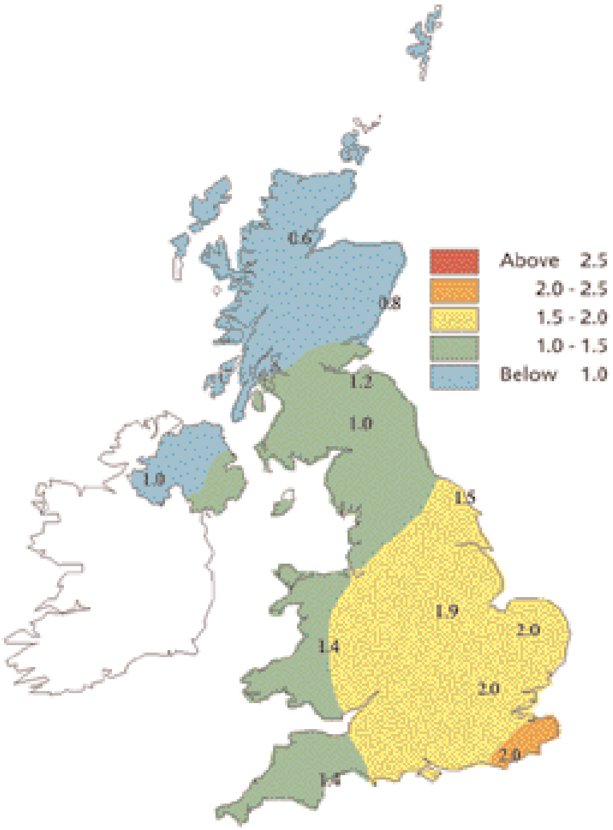


Figure 6.58 Annual mean nitrate concentration (μg m^{-3}) for 2002.

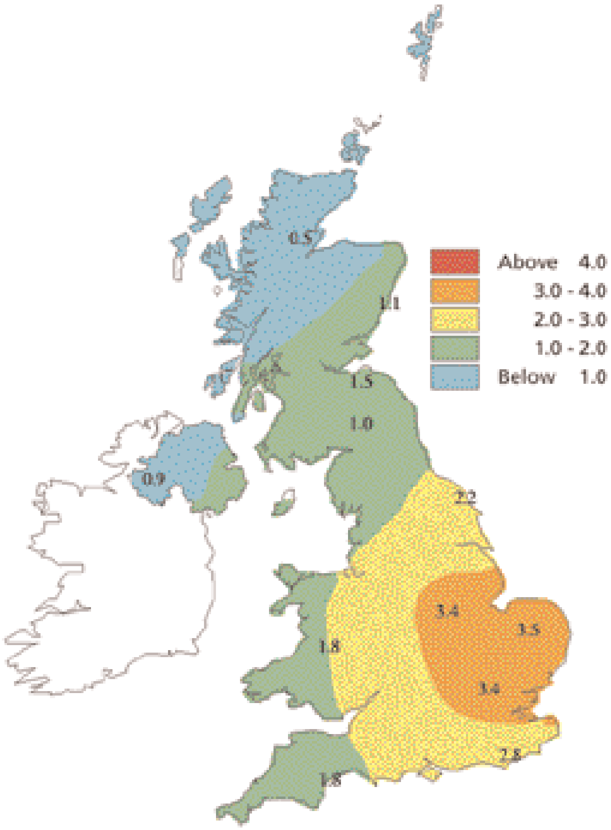


Figure 6.59 Annual mean ammonium concentration (μg m^{-3}) for 2002.

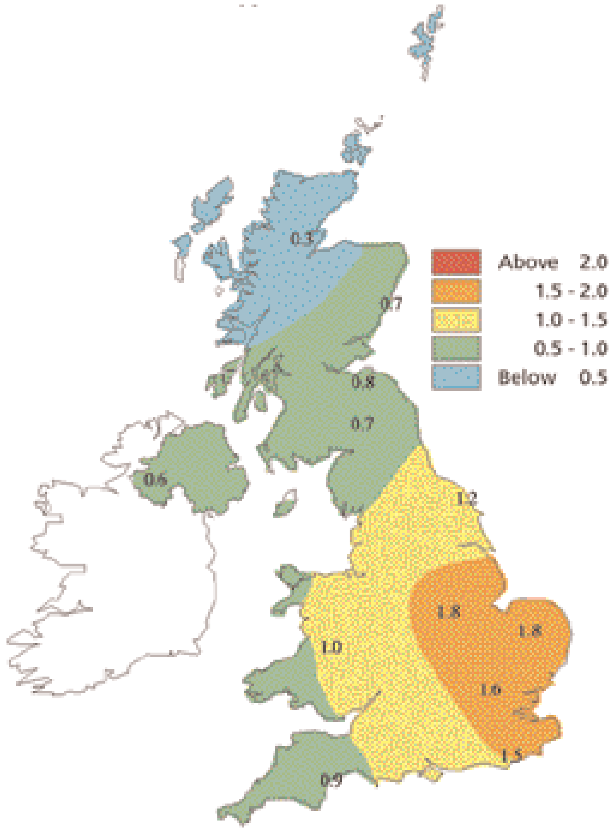
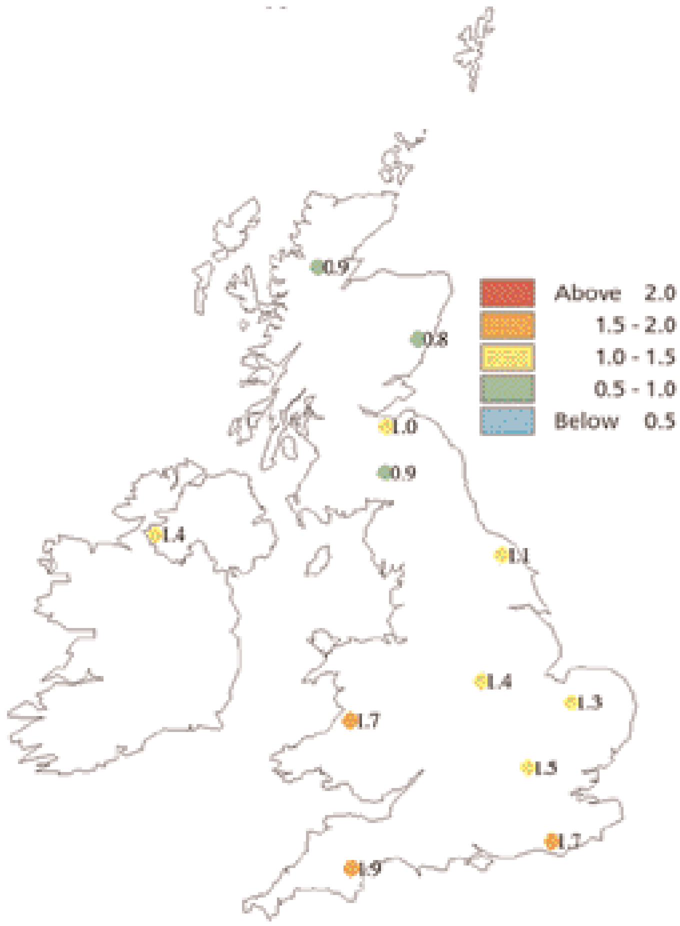


Figure 6.60 Annual mean chloride concentration (μg m^{-3}) for 2002.



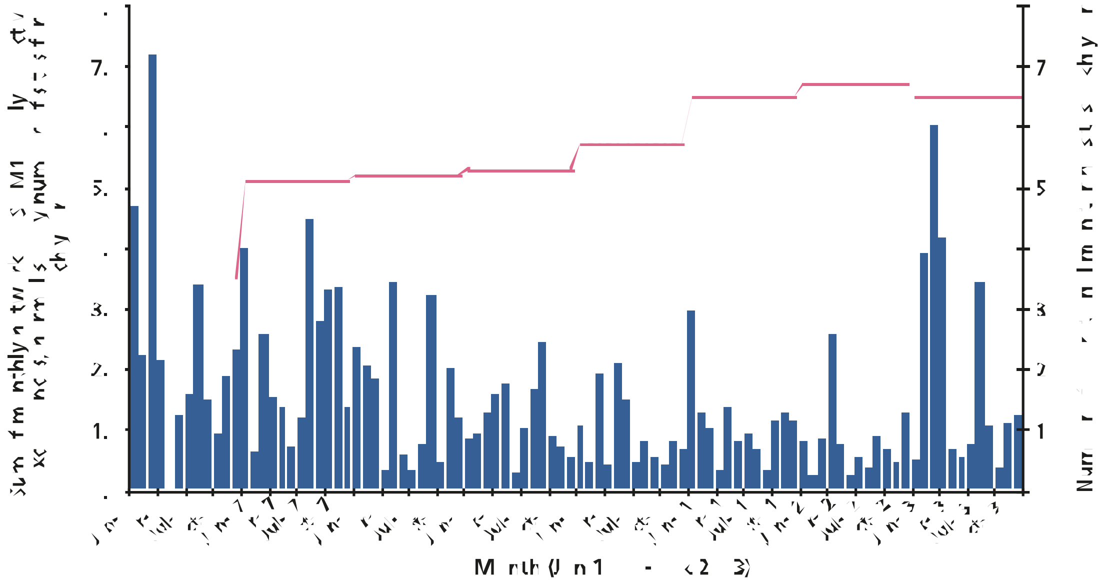
585. Chloride concentrations have also been measured at these sites. Chloride concentrations have more small-scale spatial variation than the secondary sulphate and nitrate concentrations. In particular the 12-site network does not provide enough information on the gradients in chloride concentrations close to the coast. The spatial pattern of chloride concentrations is also more variable from year to year. Chloride concentrations have, therefore, not been interpolated in Figure 6.60. In 2002 there are indications of a southwest to northeast gradient.

6.3 Episodicity of particle concentrations

6.3.1 Monthly exceedences of air quality objective concentrations

586. Figure 6.61 shows the number of exceedences of 50 µg m^{-3} (TEOM * 1.3) measured each month between January 1996 and December 2003 at national network sites. There was an increase in the number of monitoring sites over this period, so the number of sites operational during each year is also shown. The total number of measured exceedences tended to increase as the number of sites in the network increased, since an exceedence can occur at several sites on the same day. Figure 6.61 shows the number of monthly exceedences divided by the number of operational sites in each year. This should provide an estimate of the number of exceedences that is reasonably independent of the number of sites.

Figure 6.61 Number of daily PM$_{10}$ AQS exceedences summed across AURN by month (1996–2003).



587. Figure 6.61 shows that there is a considerable month-to-month variation in the number of exceedences. The number of exceedences has generally declined since 1996 but the number of exceedences during 2003 was unusually high. The episodes seen in Figure 6.61 had a range of causes. There were notable winter secondary PM episodes in January and March 1996 and in early 2003. The photochemical episode that happened in August 2003 is also clearly shown. Episodes were also caused by poor dispersion of primary pollutants, such as during the autumn of 1997. Long range transport dust events such as those that occurred in March 2000 and early 2003 are also shown.

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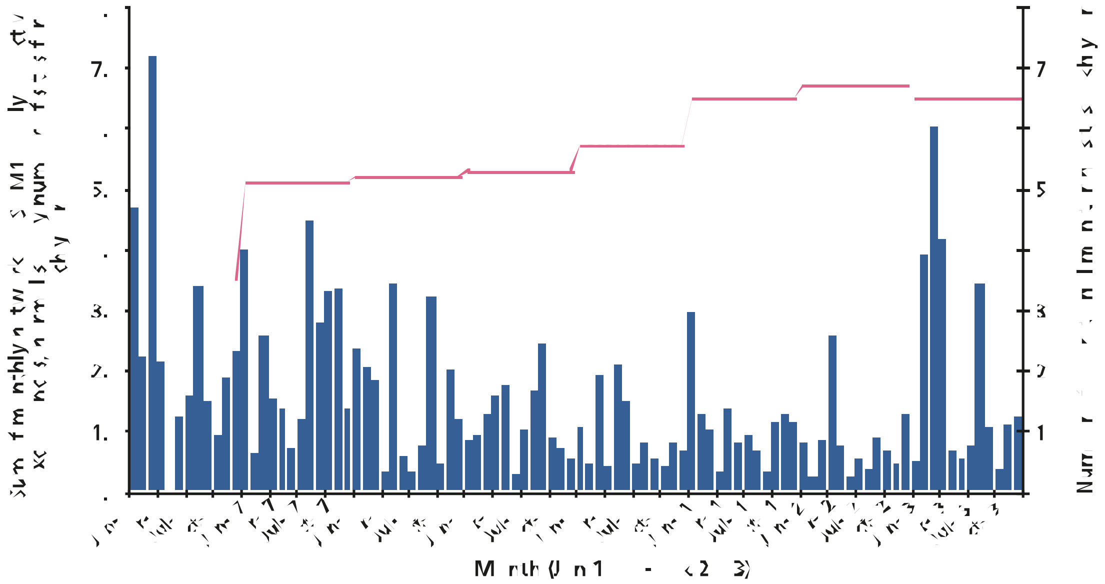

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Table 6.13 Annual number of AQS exceedences normalised by the number of sites in each year.

Year	Number of exceedences
1996	29
1997	27
1998	19
1999	14
2000	11
2001	13
2002	10
2003	24

588. Table 6.13 lists the annual number of exceedences per site summed from the monthly counts in Figure 6.61. There were more exceedences in 2003 than in any year since 1997.

6.3.2 Episodicity of specific components

589. Short-term excursions of primary emissions (such as fireworks and bonfires) influence the UK's ability to meet the current EU limit value for 24 h PM_{10}, but the episodicity of individual chemical components may have an important influence also. Thus, the frequency distributions of concentrations of PM_{10} and three of its major components are examined at Belfast and Harwell (see Table 6.14).

590. Episodicity is expressed in terms of the ratio of the higher percentile values (90% and 99% in this example) to the mean concentration. Table 6.14 shows that at the 90%ile level, chloride, nitrate and sulphate all have greater episodicity than PM_{10} and, therefore, make greater contributions than some other components to high PM_{10} concentrations. The same is true at the 99%ile level (except for chloride in Belfast), but nitrate shows far greater episodicity than either chloride or sulphate: this component may make a substantial contribution to very high PM_{10} concentrations.

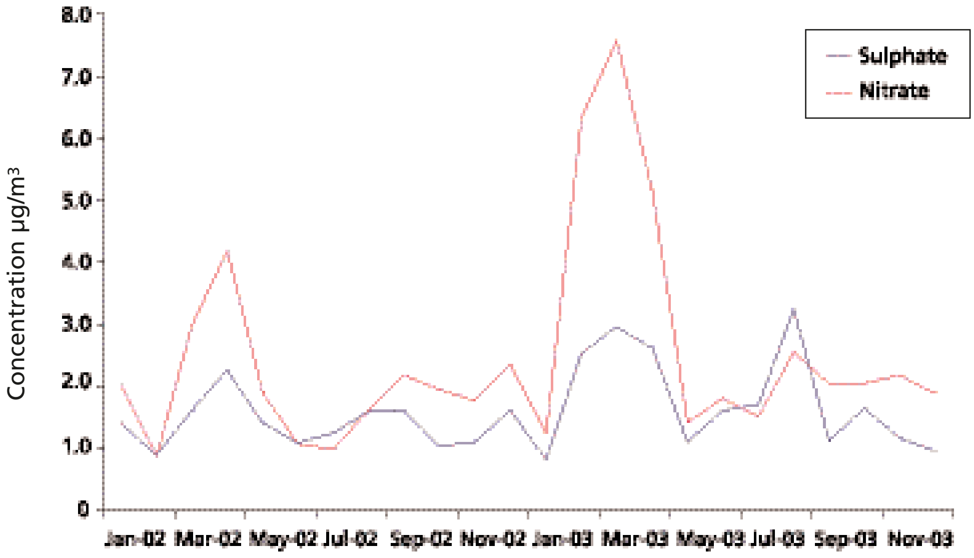
591. Figure 6.62 indicates monthly levels of sulphate and nitrate from CEH, averaged over the network. There were notable rises in sulphate and nitrate levels during the spring seasons of both 2002 and 2003, with nitrate exhibiting the more marked increases. Levels peaked in April of 2002 and in March of 2003. This rise was particularly marked for the 2003 data. The increase in sulphate and nitrate associated with the August 2003 heatwave can also be observed in Figure 6.62. Sulphate at this time rose to exceed the monthly concentrations of that spring, whereas nitrate levels remained significantly lower than the dramatic rises seen earlier in the year.

Table 6.14 Statistical properties of 24-h air quality data for Belfast and Harwell.

Site	Period	Component	Mean	90%ile	99%ile	90%ile ÷ mean	99%ile ÷ mean
Belfast	23/10/01–30/06/03	PM_{10}[a]	17.5	29.0	56.7	1.66	3.24
		Chloride	1.92	3.90	6.04	2.03	3.14
		Nitrate	2.22	5.19	21.56	2.34	9.71
		Sulphate	2.33	5.02	8.54	2.15	3.66
Harwell	17/11/01–30/6/03	PM_{10}[a]	12.91	21.00	37.22	1.63	2.88
		Chloride	1.05	2.59	4.58	2.47	4.36
		Nitrate	3.07	7.95	21.24	2.59	6.92
		Sulphate	2.49	5.10	9.66	2.05	3.88

[a] Measured by TEOM.

Figure 6.62 Monthly concentrations of sulphate and nitrate across the network during 2002–2003.



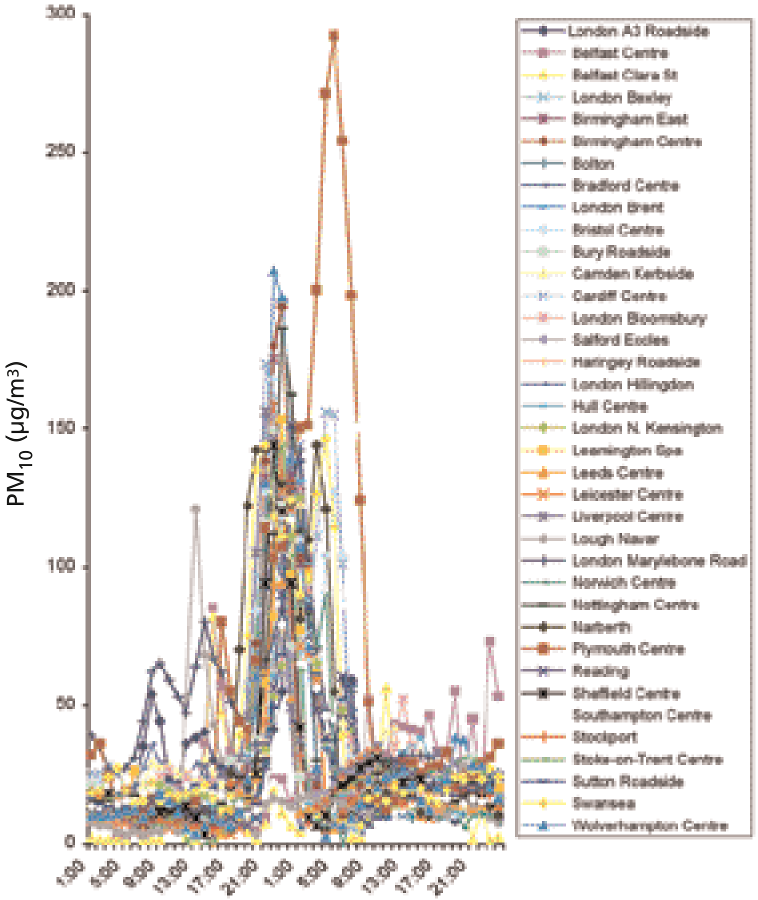
6.3.3 Saharan dust episodes

592. On a global scale, wind-blown dust is the largest natural source of particles after sea spray, with a source strength of about 2150 Tg yr^{-1} (IPCC, 2001). Dust source regions are mainly deserts, dry lake beds and semi-arid desert fringes.
The atmospheric lifetime of dusts depends on particle size: large particles are removed quickly from the atmosphere by gravitational settling, whereas sub-micron particles can have atmospheric lifetimes of weeks. Long-range transport of Saharan dusts occurs across the Mediterranean Sea into southern and central Europe (Rodriguez *et al*., 2001) and across the tropical Atlantic Ocean into the Caribbean (Rajkumar and Siung Chang, 2000) and into both North and South America (Prospero *et al*., 1981). Long-range transport of Saharan dust into the British Isles has been identified (Stevenson, 1969) and red dust deposits are regularly reported following rainfall in air masses originating over the Sahara.

593. Figure 6.63 shows the observed values of PM_{10} at all the AURN monitoring sites in Northern Ireland, Wales and England (excluding the northeast of England) on the 2nd and 3rd March 2000. Significant peaks were observed at all sites except those in the northeast of England and Scotland, whether rural, suburban, roadside or kerbside sites. No corresponding peaks were observed in NO_x. $PM_{2.5}$ levels followed the PM_{10} levels at the London Bloomsbury and Rochester sites where peak concentrations were in the range 40–60 µg m^{-3}, about one-half of the PM_{10} concentrations observed at these sites. The highest hourly mean peak PM_{10} concentration observed was 292 µg m^{-3} at Plymouth, the highest concentration observed at that site since measurements began there. The peak concentrations generally reduced across the British Isles from west to east, with maximum concentrations observed at eastern sites of ~100 µg m^{-3}. The rise in PM_{10} concentrations from their baselines occurred some 2–3 h earlier in the west, indicating rapid west-east transport. The end of the episode occurred later in the south, resulting in a longer episode in the south and west and a shorter episode in the north and east of the British Isles.

594. The Meteorological Office dispersion model NAME has been used to study the origins of the high particle concentrations observed on the 2nd and 3rd March 2000 (Ryall *et al*., 2002). Assuming a source in Africa, the model is able to explain the timing of the PM_{10} episodes at the different sites across the British Isles with close correspondence. An eruption was reported from Mount Hekla in Iceland on 26th February 2000 and this was also followed using the NAME model. From the model results, it was clear that the observed PM_{10} peaks on 2nd and 3rd March could be clearly associated with Saharan dust rather than volcanic ash from Mount Hekla.

Figure 6.63 PM_{10} concentrations observed during 2nd and 3rd March 2000.



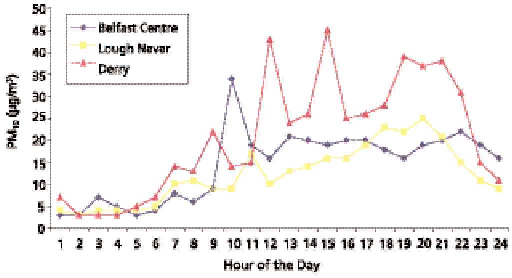
595. Ryall *et al*. (2002) considered the frequency of such dust episodes in the British Isles and predicted the occurrence of just one or two episodes per year with PM_{10} concentrations exceeding 50 µg m^{-3}. In contrast, Rodriguez *et al*. (2001) identified up to 23 events resulting in daily mean concentrations exceeding 50 µg m^{-3} in eastern and southern Spain.

6.3.4 Sea salt aerosol

596. On a global scale, sea spray is the largest natural source of particles with a source strength of 3,300 Tg yr–1 (IPCC, 2001). Sea salt particles cover a wide diameter size range of 0.05–10 µm; the majority fall into the 1–16 µm size range. They are generated by various physical processes, especially the bursting of entrained air bubbles during whitecap formation, resulting in a strong dependence on wind speed (IPCC, 2001).

597. Because of the long exposed Atlantic Ocean coastline, sea spray is an important source of coarse particles in the British Isles. Figure 6.64 shows the PM_{10} concentrations observed at three locations in Northern Ireland during 4th May 1998. Concentrations reached 40 µg m^{-3} at the Derry site, which was the site most exposed to the strong winds. Lough Navar is an inland site and the peak concentrations were somewhat lower. The levels observed at the Belfast centre site were about one-half of those observed at the coastal site.

Figure 6.64 Hourly PM_{10} measurements at three sites in Northern Ireland during 4th May 1998.



598. In most years, gales are more generally observed over the British Isles during autumn and winter, and the high winds that caused the sea salt episode shown in Figure 6.64 are a relatively uncommon during May. Figure 6.65 shows the PM_{10} concentrations observed during the period 25th to 26th November 2000. Early winter gales are a more common occurrence and this example recorded a gust of 78 knots at the Mumbles near Swansea. The peak PM_{10} concentrations reached 50–70 µg m^{-3} at exposed sites in Wales and Southwest England. Elevated levels were also observed at the Port Talbot site, and the sea salt episode was clearly detectable despite the strong local particle emissions from industrial activities in the local area.

599. Severe and strong gales are reported with a frequency of between one- and about five-times during each year and they are likely to bring sea salt episodes to exposed coastal locations with a similar frequency. It is important to determine how far inland these episodes penetrate and whether they have a wider impact on air quality than in the coastal fringe. Figure 6.66 shows the PM_{10} concentrations recorded at a number of sites in the southern part of the British Isles during 10th and 11th December 2000. A depression travelled in a northeasterly direction across Ireland and Scotland, with severe gales on its southern flank. The wind gusted at up to 80 knots at the Mumbles near Swansea and at Cranwell in Lincolnshire. Peak PM_{10} concentrations reached 50–60 µg m^{-3} at the more exposed sites, such as Blackpool and Swansea. However, there were a significant number of urban sites in the path of the gales that reached 30–40 µg m^{-3} during this episode that could not be classified as especially exposed. These included sites in Birmingham, Reading and London. Indeed, there is evidence that the PM_{10} concentrations observed at the London Marylebone Road, Camden and Haringey roadside sites were also influenced by the sea salt episode.

Figure 6.65 Hourly mean PM_{10} concentrations observed at five sites during 25th and 26th November 2000.

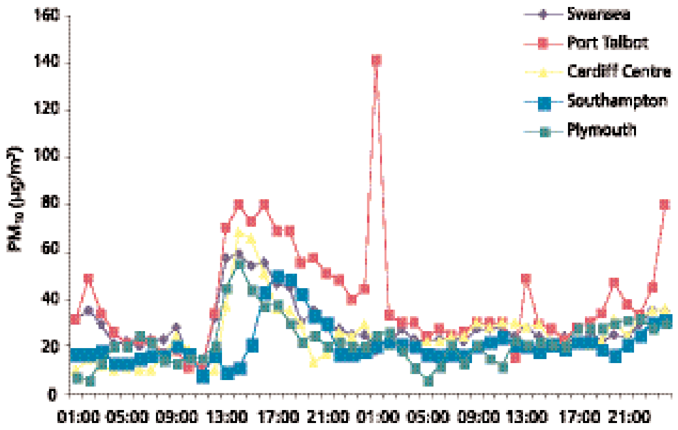
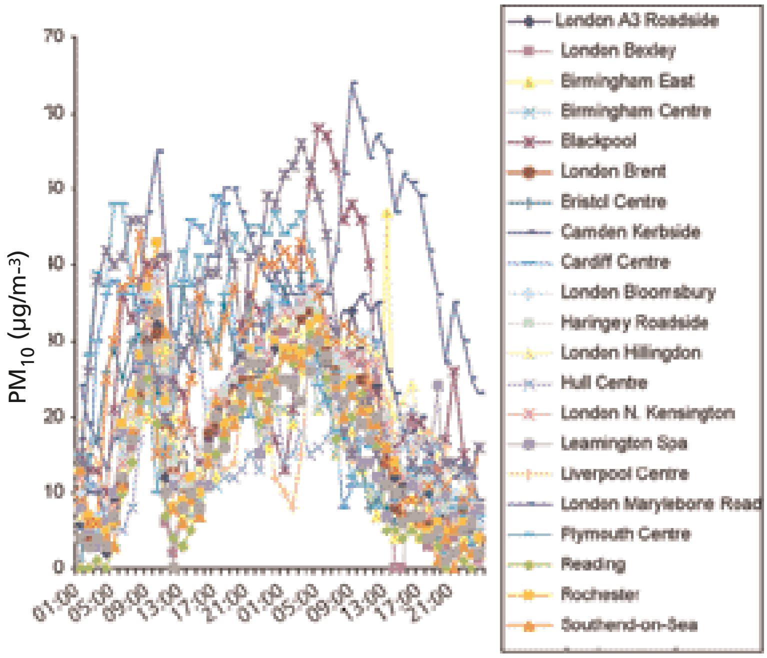


Figure 6.66 Hourly mean PM_{10} concentrations observed during 10th and 11th December 2000.

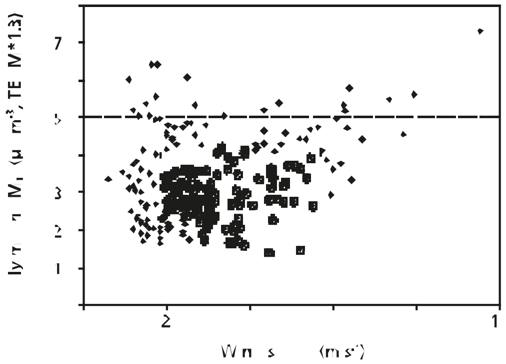


600. It is concluded that there are natural episodes of coarse sea salt particles in the UK and they occur with a frequency of between one- and five-times each year. They generally occur during the autumn and winter months and strongly influence the most exposed coastal sites, bringing peak hourly PM_{10} concentrations to in excess of 40 µg m^{-3}. Urban background sites in the paths of severe gales, far removed from the coastline, will experience hourly peak PM_{10} concentrations that are about one-half of the coastal values.

601. An analysis of PM_{10} data from a monitor sited on the coastal fringe of Sussex has also highlighted distinct sea salt episodes. Elevated concentrations of PM_{10} have been measured at the Lewes monitoring site during high wind speeds that result in daily mean concentrations above 50 µg m^{-3} (all data reported as TEOM * 1.3). During the eight months from September 2002 to May 2003, five such daily means were measured. The Lewes site is located in a roadside location on top of cliffs at Peacehaven in East Sussex, around 250 m from the sea. Figure 6.67 shows the relationship between daily mean PM_{10} and wind speed measured at the coast. Two groups of daily means above 50 µg m^{-3} can be seen. One group of means occur at low wind speeds and is due to primary and secondary PM_{10}. A second group of daily means above 50 µg m^{-3} occurs at higher wind speeds (and, therefore, rougher sea conditions) with an apparent wind speed dependency. The PM_{10} wind speed dependency at the Peacehaven site contrasts with the measurements from inland sites, such as the Horsham roadside site, shown in Figure 6.68, which is located ~30 km from the sea. At Horsham, daily mean concentrations above 50 µg m^{-3} are confined to low winds speeds, with PM_{10} concentration being inversely proportional to wind speed.

602. Source apportionment of daily mean PM_{10} at Peacehaven has been undertaken using measurements of NO_x at nearby Brighton and Hove roadside sites to provide an estimate of local primary NO_x at Peacehaven to eliminate this source from the measurements shown in Figure 6.67. Local primary PM_{10} has an inverse relationship to wind speed and makes a negligible contribution to PM_{10} concentrations during high wind speeds.

Figure 6.67 Daily mean PM_{10} at Lewes (Peacehaven) as a function of wind speed.



603. The PM_{10} coastal increment (Lewes – Horsham) is shown in Figure 6.69 and exhibits strong wind speed dependence. The coastal increment reached a maximum of >50 µg m^{-3} at the highest wind speeds. Several negative artefacts in Figure 6.69 suggest that the analysis may not be accurate for these low wind speed conditions. This is probably due to the distance between the monitoring sites and the greater prevalence of inversion conditions at the Horsham site, which is located in the Sussex Weald. Further measurements made during winter 2003–2004 will enable a better characterisation of this coastal effect, including comparison of TEOM and 'gravimetric' measurements.

Figure 6.68 Daily mean PM_{10} at Horsham as a function of wind speed.

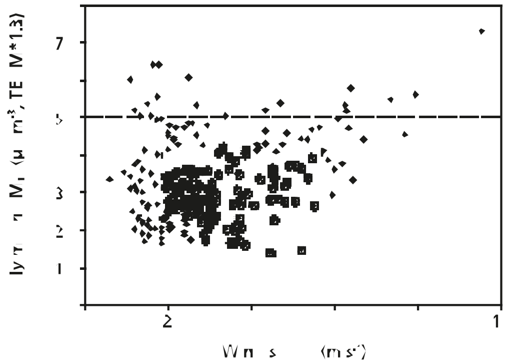
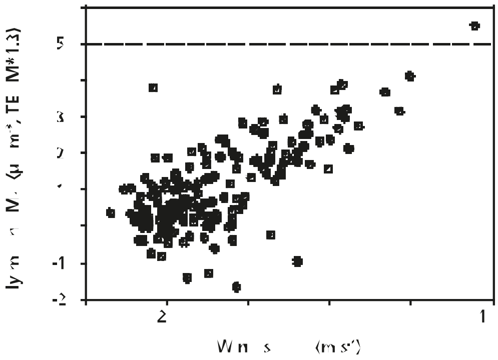


Figure 6.69 PM_{10} daily mean coastal increment as a function of wind speed.



6.3.5 Biomass-burning fires as a source of particles

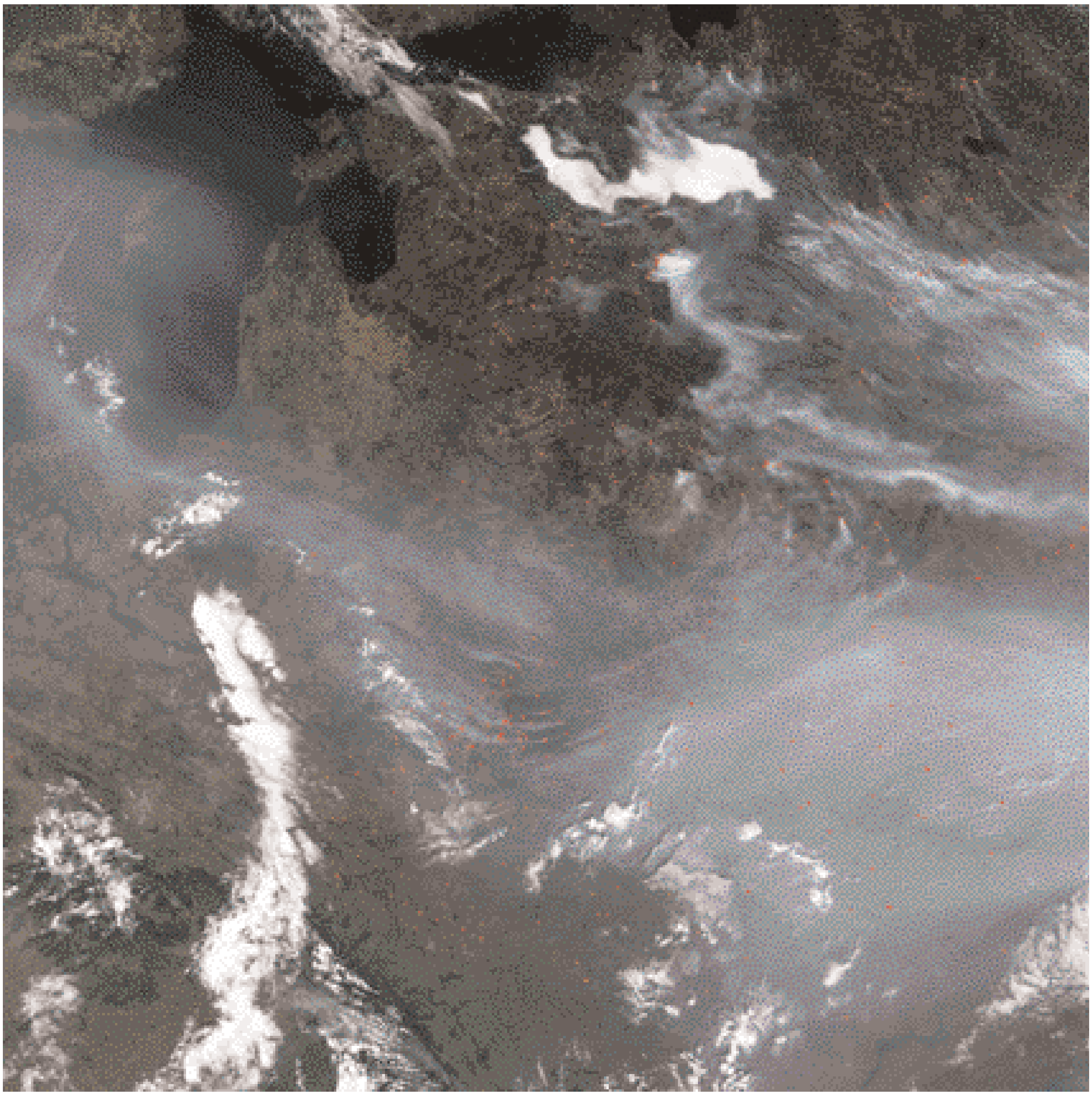
604. Towards the end of August 2002, reports were received of extensive forest fires in Western Russia and consequential poor air quality in Moscow due to smoke. Around the end of August/early September elevated PM_{10} levels were reported in Helsinki where levels reached close to 100 µg m^{-3}. Figure 6.70 presents a satellite image from the Moderate Resolution Imaging Spectroradiometer (MODIS) rapid response team at NASA-GSFC: http://earthobservatory.nasa.gov/NaturalHazards, which clearly shows the wide extent of the forest fires on 4th September 2002 within the region 28° to 40° E, 52° to 60° N.

605. The smoke-laden air masses were transported in a westerly direction and crossed the North Sea during the 11th and 12th of September. PM_{10} peaks were observed at all the automatic monitoring stations in the British Isles at some time during these two days. Figure 6.71 presents the PM_{10} data for 17 monitoring locations in northeastern and northwestern England. Peak hourly concentrations here were reported on the 12th of September and were in the range 70–125 µg m^{-3}. Peaks were also seen on the 11th September, but these were generally smaller in magnitude ~40–80 µg m^{-3} and their timings showed less coherence between the sites than they did on the 12th.

606. Two sites were operational in Scotland at Edinburgh centre and Glasgow centre. These sites showed peaks during the morning and late evening of the 11th and substantial peaks of >100 µg m^{-3} during the afternoon of the 12th. Four sites were operating in Northern Ireland and they showed no evidence of the pollution event on the 11th. Peaks of >100 µg m^{-3} were observed during the afternoon of the 12th in Belfast and of about 80 µg m^{-3} at Lough Navar. Two peaks were observed at the

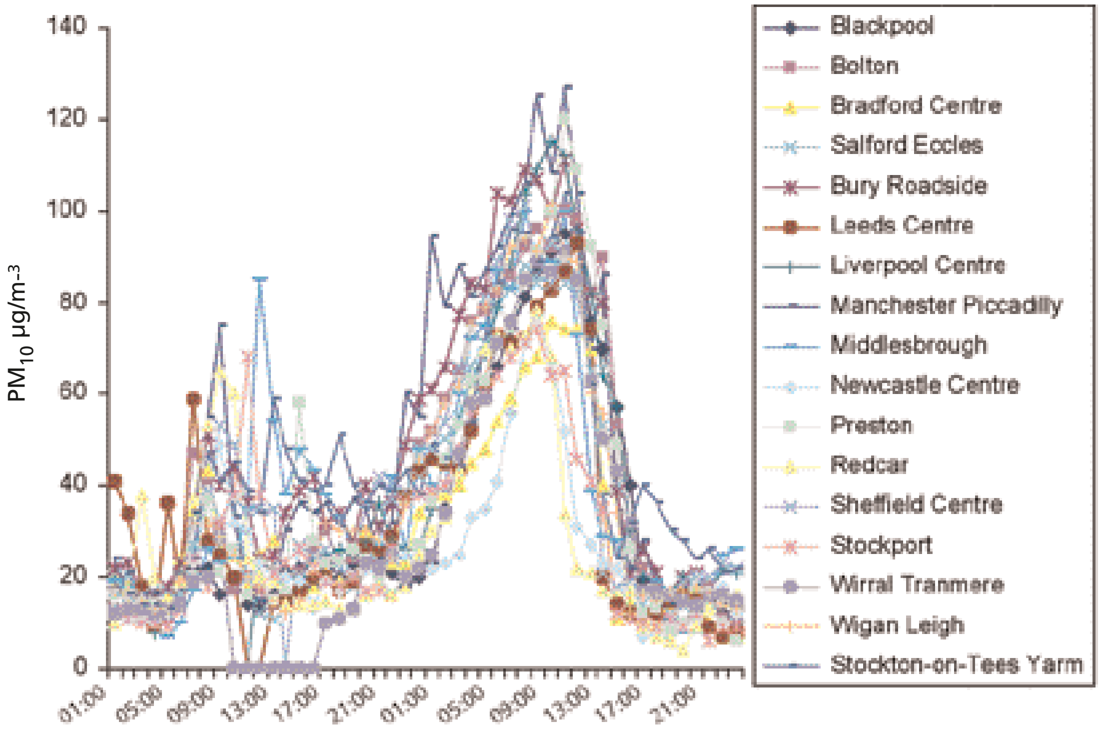
Welsh sites Cardiff Centre, Cwmbran and Port Talbot and reached over 50 µg m^{-3}. The rural site Narberth in the west of Wales showed peaks that arrived significantly later than the other Welsh sites and with much reduced magnitudes. The smoky air masses reached London late on the evening of the 11th and again during the afternoon of the 12th. These peaks were similar in magnitude, both being ~60–70 µg m^{-3} at the seven AURN sites. Sites in the Midlands showed a morning peak on the 11th and elevated concentrations throughout the morning into the early afternoon of the 12th, which reached 40–80 µg m^{-3}.

Figure 6.70 Satellite image of western Russia from MODIS for the 4th September 2002, covering the region 28° to 40° E, 52° to 60° N and showing the extent of the forest fires.



607. The Meteorological Office Lagrangian dispersion NAME model was run for 10 days for each 3-h period from midnight on 11th September 2002 to midnight on 14th September 2002 for three locations: London, Lough Navar and Edinburgh. Global meteorology was used, which has a horizontal resolution of ~60 km and 32 vertical levels. A total of 45,000 air parcels were released over each 3-h period of interest between the ground and 80 m. Air parcel position information was collected on a ~60 km horizontal by 80 m vertical grid extending from 40°W to 60°E and 20°N to 80°N. The dispersed and transport material was assumed to be inert and, therefore, no account was taken of dry or wet deposition, sedimentation or chemical changes. The amount of material released was arbitrarily set to 1 g/s.

Figure 6.71 Hourly mean PM_{10} concentrations monitored at 17 locations in northeast and northwest England during September 11th and 12th 2002.

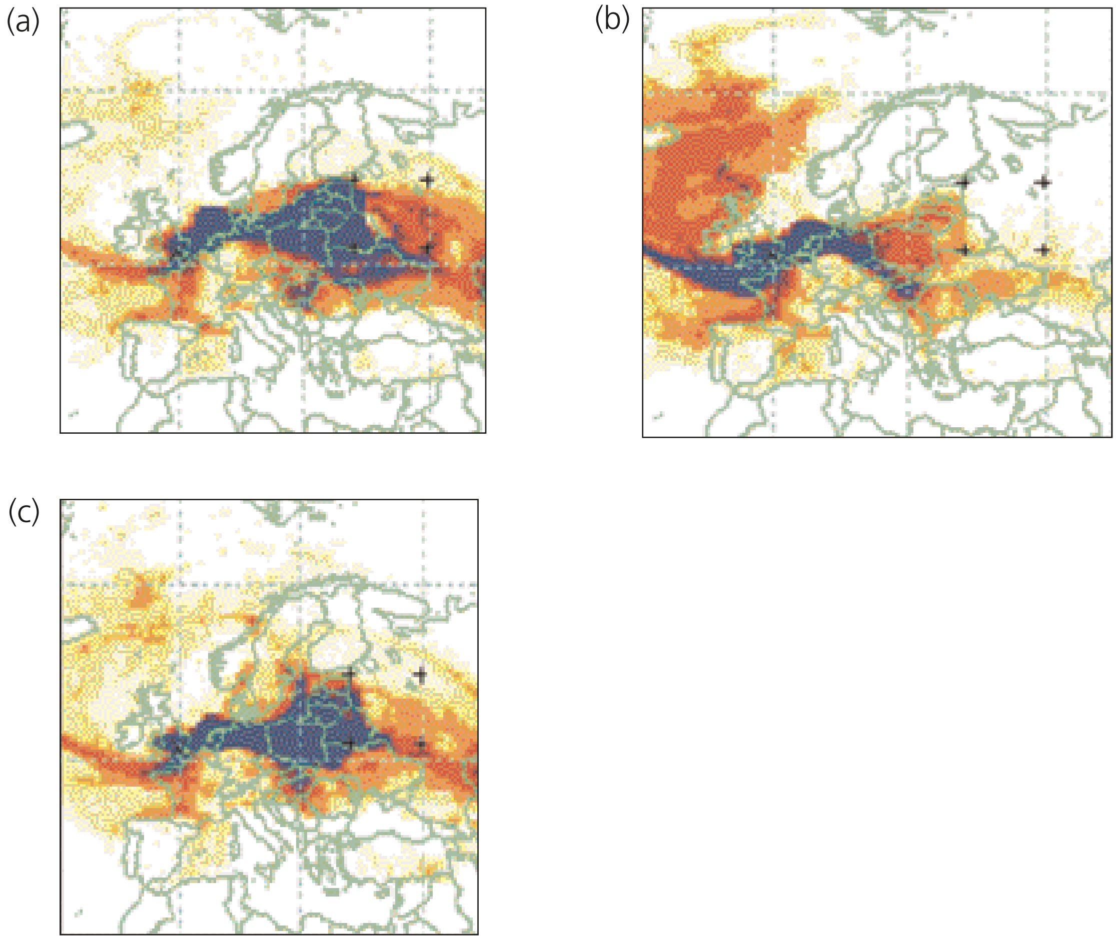


608. Figure 6.72 shows the modelled air origin maps for London for the 3-h periods ending at (a) 00Z 12/09/02, (b) 06Z 12/09/02 and (c) 12Z 12/09/02. The four corners of the domain encompassing the fires identified in the satellite images have been marked (+) in each plot and the location of the receptor has also been highlighted (x). The plot is contoured with darker shades representing greater contributions.

609. Figure 6.73 shows the time series of observations of PM_{10} and the modelled contribution from the domain encompassing the fires (see Figure 6.72) at London, Edinburgh and Lough Navar. The observations shown for London are the average PM_{10} values measured at the Bloomsbury and North Kensington sites.

610. The double peak structure (Figure 6.73a) in the London observations is reflected in the modelled concentrations. The air origin maps for the times of the two peaks and the low point in-between are shown in Figure 6.72. The map of the 3-h period marking the low point (Figure 6.72b) shows that the origin of the air was more diffuse, with significant contributions from the Atlantic. The observations were uniformly elevated by background levels of 15–30 µg/m3, reflecting the normal diurnal pattern of PM_{10} concentrations at London during an easterly flow. The second observed peak for the period 09:00 to 12:00 h will have a significant extra burden from local traffic sources compared to the first peak at 21:00 to 24:00 h. The timings of the peaks in the observations are identical to those modelled, namely at the 3-h periods centred on 22:30 h 11/9/02 and 10:30 h 12/9/02.

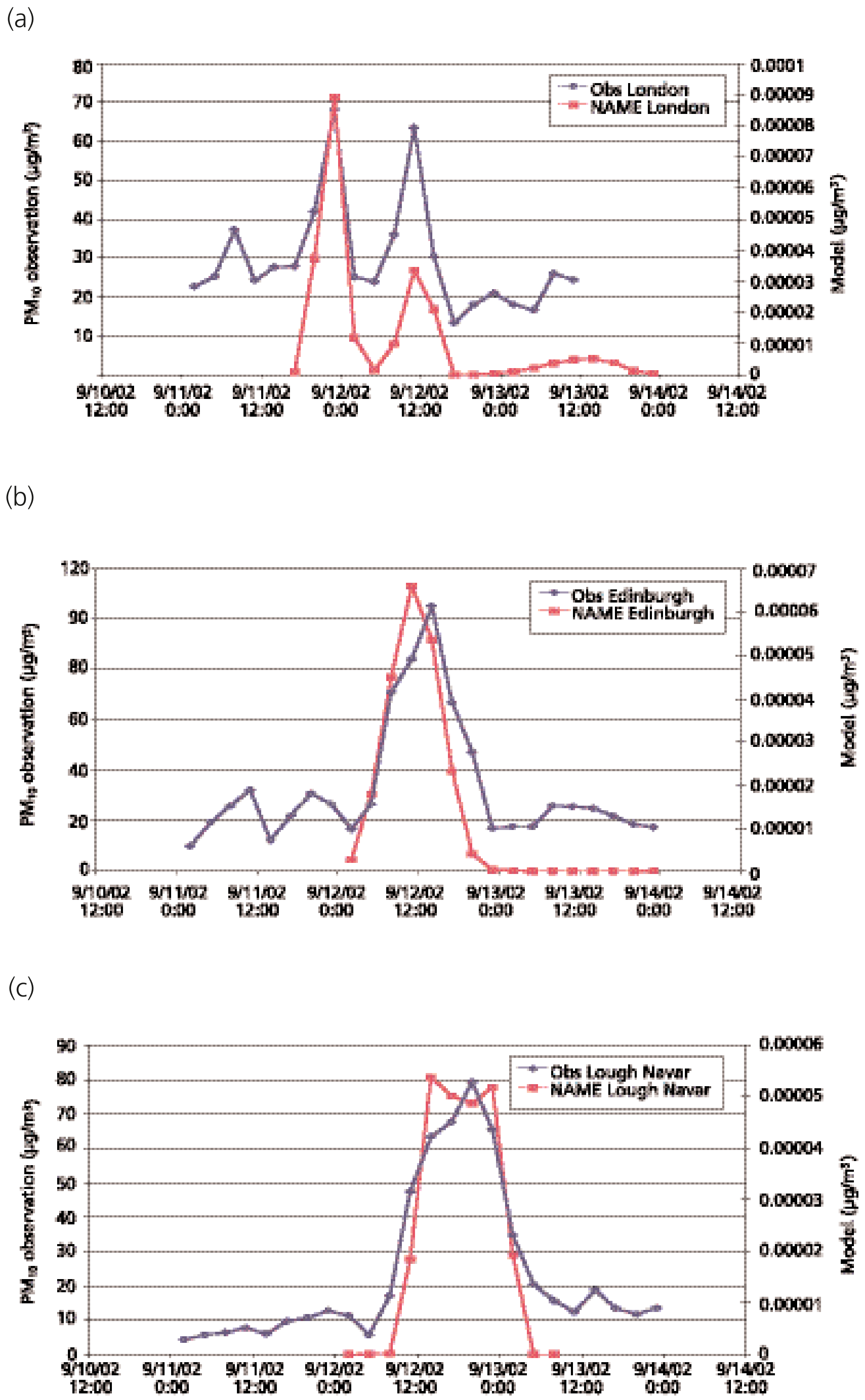
Figure 6.72 Ten-day back-attribution maps for London, 3-h periods ending (a) 00z 12/09/02, (b) 06z 12/09/02 and (c) 12z 12/09/02.



611. The start times of the episodes at the three locations are significantly different. At London it began at 18:00 to 21:00 h on 11th September; at Edinburgh it was 03:00 to 06:00 h on 12th September and at Lough Navar the episode did not start until –09:00 to 12:00 h on 12th September. The observed episode start times were all mirrored by the modelled arrival times of air from the identified fire domain, as shown in Figure 6.73.

612. The conclusions drawn from the above analysis of the PM_{10} pollution episode on the 11th and 12th of September 2002, is that particles can clearly travel distance of over 1000 km across Europe from Western Russia into the British Isles. Furthermore, natural sources of particles from biomass burning can produce elevated hourly PM_{10} concentrations of >100 µg m^{-3} at sites within the UK.

Figure 6.73 Time series of observed background PM_{10} and modelled air concentration from the estimated area on fire in Western Russia at (a) London (b) Edinburgh and (c) Lough Navar. (The model results correspond to a notional emission rate of 1 g/s, as the true source strength is unknown.)



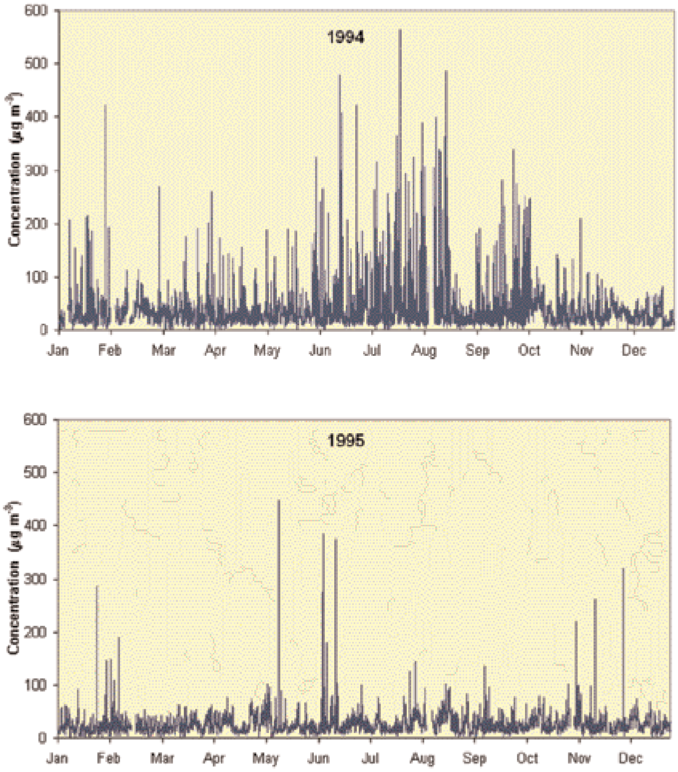
6.3.6 PM from construction activities

613. Dust is generated from a number of activities on a construction site, the principal sources being: vehicle movement over dry unpaved ground; the handling of dusty materials; the cutting of concrete/stone; and the demolition of buildings. Dust is also generated by material tracked out of the site onto local roads. Typically 15–45% of construction dust is emitted as PM_{10}. Nationally, construction is not considered to be an important source of ambient PM (see Chapter 5)), but locally this can be different. Unfortunately there have been few reported studies of PM concentrations near to construction activities, so it is difficult to be certain about impacts on local concentrations. These impacts would, in any case, be expected to vary from location to location, depending on the size and nature of the works, the weather conditions, and the stringency with which mitigation measures are applied. They will also be of a temporary nature, although the impacts could increase public exposure over a period of several years for major construction sites.

614. One opportunity to investigate the impact very close to a construction site arose at the Cardiff centre AURN monitoring site in 1994. This city centre site is located in a pedestrianised area away from traffic. In 1994 a construction programme was started on a retail unit located alongside the monitor, 5 m away at the nearest point. The works lasted a full year and involved demolition, ground works, laying of concrete foundations, erection of a steel frame and concrete floor slabs, wall and roof construction, then finishing. A significant impact on PM_{10} (TEOM) concentrations was observed, as illustrated in Figure 6.74. (It is appropriate to use raw TEOM values, as the particles are unlikely to be volatile and the 1.3 scaling factor is unlikely to be appropriate). During 1994 there were 54 days when PM_{10} (TEOM) 24-h concentrations exceeded 50 µg m^{-3}, with a maximum 24-h value of 96 µg m^{-3}, compared to 12 days in 1995 and a maximum of 82 µg m^{-3}. The greatest impact was on 1-h PM_{10} (TEOM) concentrations, with 89 h >200 µg m^{-3} in 1994, compared to 11 h in 1995. The impact on annual mean values was, however, much smaller, with PM_{10} (TEOM) values of 31 µg m^{-3}, 34 µg m^{-3} and 25 µg m^{-3} in 1993, 1994 and 1995, respectively. An analysis of the events showed that the 1-h exceedences of 200 µg m^{-3} nearly all occurred between 08:00 and 17:00 h, that is, during working hours (Kukadia *et al*., 1998). The greatest number of high 1-h concentrations was during the summer months; it is not clear how much this was due to drier weather or the nature of the work activities at that time.

615. The national monitoring activities have provided a further opportunity to examine the effects of construction works on PM concentrations. This occurred when demolition activities took place close to the Marylebone Road site in London, between July and November 1999 (Charron & Harrison, 2004). During this period and more especially in September, very high concentrations of PM_{10} (TEOM) were recorded, with hourly peaks of up to 800 µg m^{-3}. The main contribution to these PM_{10} peaks was made by PM_{coarse}, although coincident peaks of $PM_{2.5}$ (TEOM) were also recorded (Figure 6.75). This resulted in many hourly $PM_{2.5}$:PM_{10} ratios being lower than 0.5. All the high PM_{coarse} concentrations occurred during the working period (08:00 to 16:00 h) on weekdays, and the very high concentrations occurred for short periods of few hours or less. High concentration events did not appear to depend on wind speed or rainfall conditions and occurred on days with low or high wind speeds as well as on rainy or dry days. All high concentrations occurred when the synoptic wind was between 70° and 270°, which corresponds to wind direction associated with the southerly vortex in this canyon-like location (see Section 6.2.5) and parallel to the street. (Most of them occurred during southerly vortex situations.)

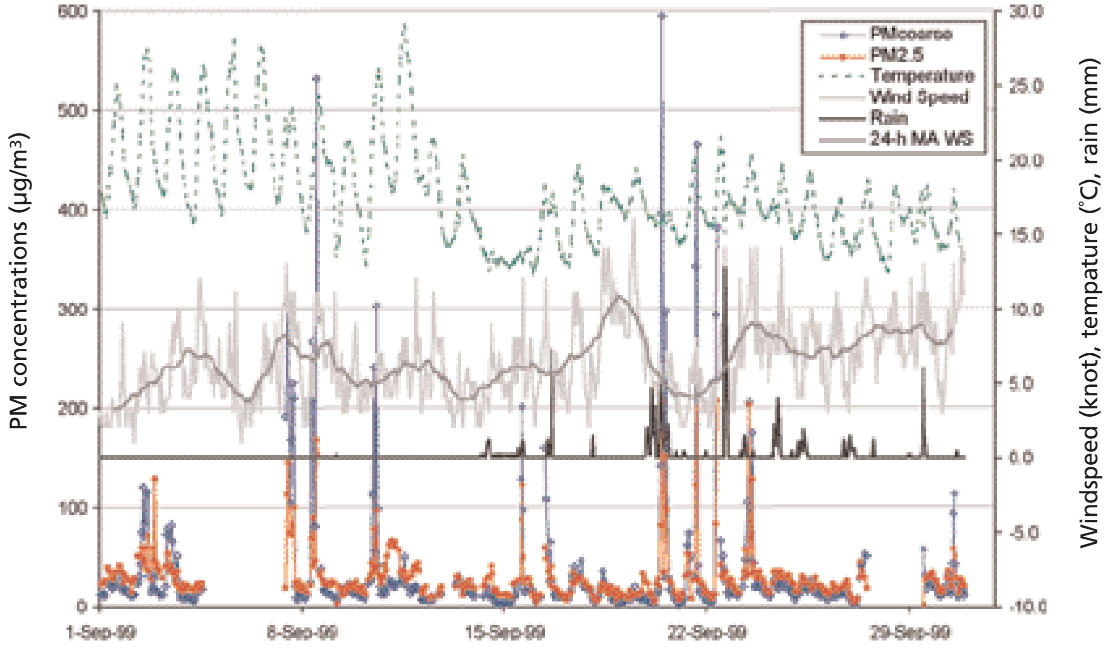
Figure 6.74 PM_{10} (TEOM) 1-h concentrations at the Cardiff centre AURN site: construction occurred in 1994, no construction occurred in 1995.



616. During the period of July to November 1999, the very high hourly concentrations had a significant impact on daily PM_{10} concentrations. There were 27 days when PM_{10} (TEOM) 24-h concentrations exceeded 50 μg m^{-3}, with a maximum 24-h concentration of 139 μg m^{-3}; compared to 6 days and a maximum of 57 μg m^{-3} and 16 days and a maximum of 87 μg m^{-3}, respectively, from July to November 1998 and 2000. From July to October 1999, the agreement between daily PM_{10} (TEOM) data and filter-based mass measurements (KFG) was good (median ratio of KFG:TEOM was 1.04), showing that particles were mainly made of non-volatile material. Although the impact of demolition activities on $PM_{2.5}$ concentrations was less spectacular than the impact on PM_{coarse} concentrations (PM_{coarse}: 44 h >100 μg m^{-3}; $PM_{2.5}$:19 h >100 μg m^{-3}), it is nevertheless still significant. From July to November 1999, there were 8 days when $PM_{2.5}$ (TEOM) 24-h concentrations exceeded 40 μg m^{-3} with a maximum of 54.8 μg m^{-3}, which compares with 1 day and a maximum of 41.9 μg m^{-3} and 3 days and a maximum of 50.2 μg m^{-3}, respectively, from July to November 1998 and 2000.

617. Local fugitive PM_{10} at Marylebone Road during 1999 has been modelled using a source apportionment technique (Fuller & Green, 2004). This shows that the local fugitive PM_{10} alone was sufficient to cause the daily mean PM_{10} at the site to exceed 50 μg m^{-3}, reaching a maximum daily mean concentration of 133 μg m^{-3}.

Figure 6.75 Hourly $PM_{2.5}$ and PM_{coarse} concentrations (TEOM), wind speed (including 24-h moving average), temperature and rain from 1st to 30th September 1999 at Marylebone Road.

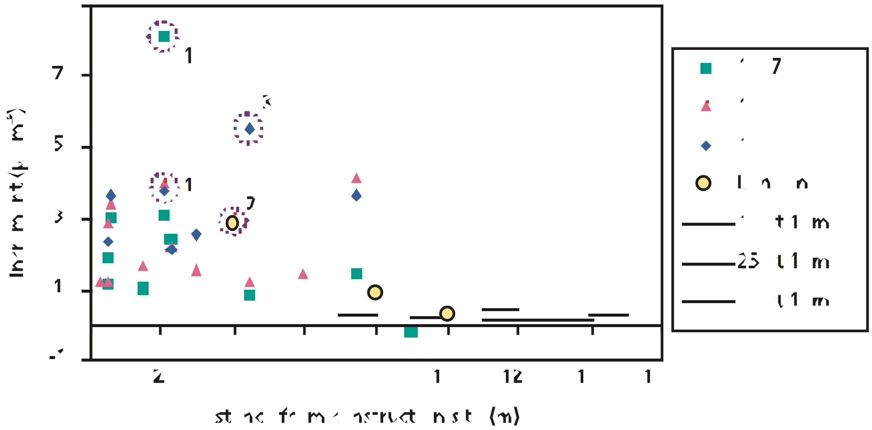


618. Monitoring data from sites across London and Southeast England have been examined for the period 1999–2001 to identify PM_{10} incidents that are considered to be unrelated to road traffic (Fuller & Green, 2004). Incidents have been attributed to local fugitive PM sources on the basis of brief high measurements of PM_{10}, using the criterion of 15-min mean concentrations in >250 µg m^{-3}. PM_{10} measurements from 81 TEOM sites during the 3 years were examined, using this criterion, to estimate the likely frequency of local fugitive incidents (many of which were likely to be related to some form of construction activity, but not exclusively so). This dataset of 156 site years included 134 daily means >50 µg m^{-3} that might have been linked to fugitive sources. This represents an average of 0.9 days per site per year. Furthermore, the analysis suggests that fugitive sources might have been associated with 5% of the 2,470 daily means >50 µg m^{-3} and that they occurred at almost half (37/81) of the monitoring sites, including all types of site from kerbside to rural. Two sites that measured more than ten daily means associated with fugitive sources during the 3-year period were Kensington and Chelsea 2 (19 days) and Marylebone Road (45 days). Marylebone Road measured more than 7 such days in each of the 3 years.

619. Turning to more specific studies of construction sites, results are available for monitoring around a major construction programme in the centre of downtown Boston, Massachusetts (Dolan *et al*., 2000). PM_{10} (gravimetric) was monitored at 16 locations within 90 m of an eight-lane underground roadway being constructed through the city. Monitoring using Minivol gravimetric samplers was carried out as part of a programme to control dust impacts. The programme involved monitoring 2 days a week for the 5 months of June to October over the 3 years of the project. The results are summarised in Figure 6.76, in which the increment above background is plotted against distance from the site (results for two monitoring sites that were alongside roads with well over 100,000 vehicles per day, as well as being close to the construction site, are not included).

Concentrations clearly increase closer to the construction site. Distances are from the boundary of the site, not the dust source. In some instances the dust source is material tracked out onto local roads, thus the source of the dust can be closer to the monitor than shown (see the caption for Figure 6.76). The results show clear evidence that mean PM_{10} concentrations can be significantly elevated by construction dust, potentially by some 10–50 µg m^{-3} over 5 summer months within 90 m of a major construction site. They also show that the movement of dump trucks along off-site roads spreads the PM_{10} impact away from the site. Separate studies were carried out following a number of incidents when 24-h PM_{10} (gravimetric) concentrations exceeded 150 µg m^{-3} – the USEPA standard at the time. These showed that additional control measures to reduce airborne dust, including improved cleaning of roads, were effective in reducing PM_{10} concentrations, confirming the role of construction dust in producing the higher levels close to the construction site.

Figure 6.76 PM_{10} increment in relation to background versus distance from boundary of a major construction site. (1997–1999 refers to the summer mean PM_{10} (gravimetric) for the site in Boston, Massachusetts, 1997–1999; London refers to the annual mean PM_{10} (TEOM) for the site in London. See text for discussion of the lines. (1) This site was close to a road being used by over 600 dump trucks a day in 1997. Lorries were idling alongside the monitor. (2) This site was five m from the road used by site lorries and 40 m from site entrance. (3) This site was affected by dirt being tracked out by dump trucks. (4) There is no explanation for the high levels recorded at this site; the lines are fitted assuming 10 µg m^{-3} at 10 m, 25 µg m^{-3} at 10 m and 40 µg m^{-3} at 10 m.)



620. Monitoring of PM_{10} using TEOM instruments has recently been carried out at three locations near to a major construction site in London. Results for a 1 year period have been made available with permission of the developer and the local authority (Symonds, 2004). During this period major earthworks were under way on the site of around 0.75 km^2. Annual mean and daily exceedences of 50 µg m^{-3} are set out in Table 6.15. TEOM units have been retained, as the excess above the background is unlikely to contain volatile particles, thus a factor close to 1.0 would be expected to apply to this increment. The results show that there can be a significant impact close to the site, but the impact 100 m away seems to be minimal (the site at 100 m is not downwind of the predominant wind direction). The highest concentrations are found near to the site entrance, where vehicle activity is generating significant amounts of dust.

621. The results for the London construction site are also shown as an increment above the background in Figure 6.76. The lines in this figure are derived using the relationship shown for the modelled decline in concentrations with distance from a fugitive source (Defra, 2003a). This model was normalised to a concentration of one at 10 m from the source. A good fit would not be expected, as the monitoring sites are not located on a transect away from as single source, also the distance is from the boundary of the site, not the source of the dust, nevertheless the pattern of the monitoring seems to be consistent with the modelled relationship.

Table 6.15 Measured PM_{10} (TEOM) concentrations around a major construction site in London.

Monitoring location	Distance from site boundary	Annual mean μg m^{-3} (TEOM)	Daily exceedences of 50 μg m^{-3} (TEOM)
Site 1	40 m from site entrance to east of site and five m from road affected by dirt being tracked out of the site.	46	122
Site 2	80 m from northeast boundary of site	27	26
Site 3	100 m from site boundary to south in residential area	21	7
Site 4	Local authority background site ~7 km to north	18	4

622. At a smaller scale, a study by Upton & Kukadia (2002) involved PM_{10} monitoring at three locations around a 0.65 ha development site within a town centre. The site housed an old industrial building, which was demolished before removing 1 m of contaminated soil, after which the site was in-filled and levelled. The work took place between September and March, thus not over the season when dust would be at its worst (dust is more likely during drier summer months, May to August). One monitoring site was located a few metres from the boundary of the site, to the north, the other two were 185 m to the southwest and 130 m to the east of the site. No impacts were detected at the two distant monitors, but at the site boundary concentrations were elevated by ~3 μg m^{-3} during internal stripping of the buildings (over 4 weeks), 11 μg m^{-3} during demolition (over 8 weeks) and 5 μg m^{-3} during earth-working operations (over 8 weeks). This period of works is equivalent to an increase in annual mean of 2.7 μg m^{-3} at the site boundary. There was one exceedence of the 50 μg m^{-3} objective at the site boundary, but none at the other two locations. It is not surprising with increases of such small magnitudes at the site boundary that no effects are seen at the more distant monitoring sites. This was the case even at times when the winds were blowing from the site towards the more distant monitors. Dust was controlled in a rigorous manner at this site, due to its town centre location, so it may not be typical of other sites.

623. In conclusion, it is clear that construction activities can give rise to elevated PM_{10} concentrations, both in terms of annual mean and 24-h exceedences of 50 µg m^{-3}. This is certainly the case within 10 m of relatively small construction sites and may extend out to 50–100 m from major construction sites. It would seem that beyond about 100 m, construction sites are likely to make only a small contribution to local PM_{10} concentrations. The evidence is that vehicle movements are a key source of the PM_{10}, including byway or resuspended material tracked out of the site onto public roads. This track out of dust and dirt could extend the area of significant impact further than 100 m from the site boundary. The situation occurring at Manor Road, Bexley in London, where dust and dirt are being tracked out from light industrial sites onto the local roads by lorries, is similar (see Section 6.4.7). The study of monitoring sites across the southeast of England suggests that there might be fairly widespread contributions to PM_{10} made by fugitive, in many cases construction-related, sources and that these incidents can contribute to the number of days with concentrations >50 µg m^{-3}. No specific studies have been identified that show the contributions construction activities can make to $PM_{2.5}$ concentrations, although the results for the Marylebone Road site suggest that $PM_{2.5}$ concentrations are affected by construction activities, although to a lesser extent than PM_{coarse} concentrations.

6.4 Use of correlations, concentrations or chemistry to elucidate particle sources

6.4.1 Comparison of PM concentrations with measurements of other pollutants

624. Measured PM_{10} and $PM_{2.5}$ concentrations have been compared with the measured concentrations of other air pollutants at Marylebone Road and Harwell. Figures 6.77 to 6.79 show comparisons of hourly PM concentrations during 2002 with NO_x, benzene and CO. PM concentrations are more closely correlated with NO_x than with benzene or CO. This is consistent with the expected emission factors for road vehicles, which are higher for PM and NO_x for diesel vehicles and higher for benzene and CO for petrol vehicles. It confirms that traffic emissions are an important PM source at this site. The emissions of PM_{10} are greater than the emissions of $PM_{2.5}$ (the gradient of the line is greater), although this analysis does not directly inform as to whether the additional emissions of PM_{coarse} are exhaust or other emissions.

625. The gradients for PM_{10} and $PM_{2.5}$ are much more similar at the rural site at Harwell (Figures 6.80 and 6.81) and the correlation coefficients are lower because this site is less strongly influenced by traffic sources. Figures 6.84 and 6.85 show the relationships between PM concentrations and ozone. The daily maximum running 8-h ozone concentration has been compared with daily mean PM concentrations (hourly PM and ozone concentrations tend to be anti-correlated because primary PM emissions are correlated to NO emissions and thus anti-correlated with measured ozone concentrations). Figure 6.82 shows that there is some tendency for high PM concentrations to be associated with high ozone concentrations in the summer at Harwell, which reflects the photochemical generation of secondary particles. This is not the case in winter when the weak anti-correlation suggests that the higher PM days tend to be associated with lower ozone concentrations, when poor dispersion leads to a build up of primary

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Figure 6.77 Marylebone Road hourly NO_x against PM_{10} and $PM_{2.5}$ (TEOM) in 2002.

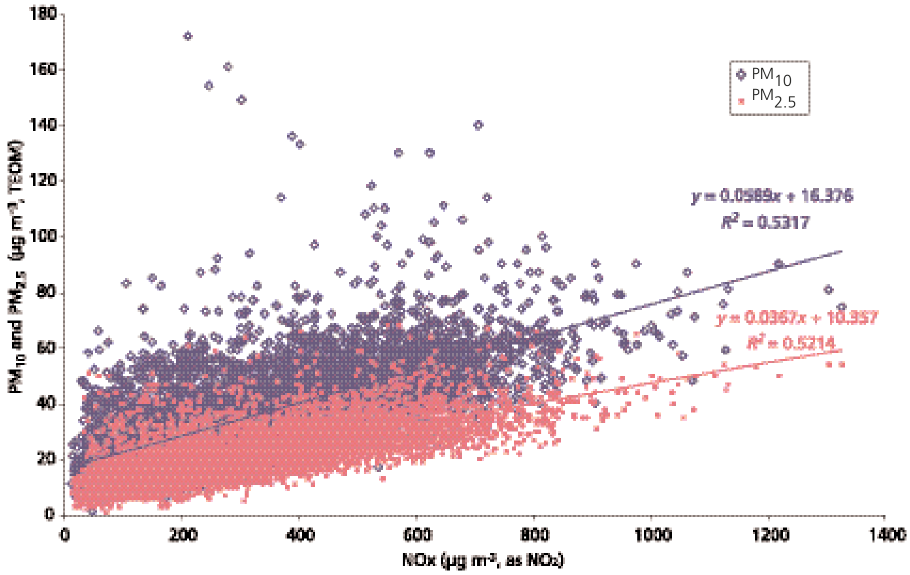
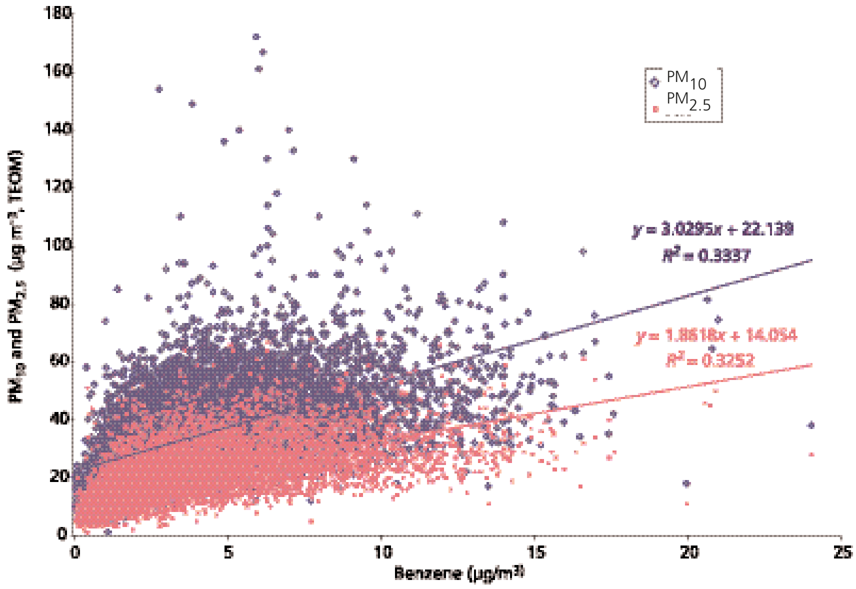


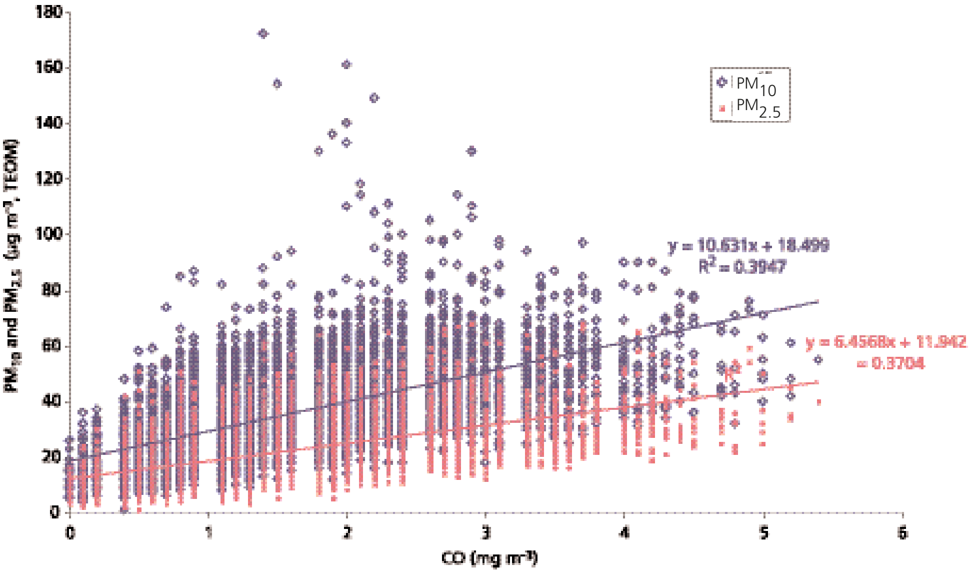
Figure 6.78 Marylebone Road hourly benzene against PM_{10} and $PM_{2.5}$ (TEOM) in 2002.



particles and more efficient titration of ozone with NO. Figure 6.83 shows that the titration of ozone with locally emitted NO dominates measured ozone concentrations at Marylebone Road leading to PM and ozone concentrations being anti-correlated in both winter and summer.

626. Figures 6.84 and 6.85 show the diurnal variation of PM and other pollutant concentrations at these two sites in 2002. These plots confirm the conclusions of the scatter plot analysis. There is also evidence that NO_x and PM emissions at Marylebone Road are greatest during the morning peak and that CO and benzene emissions are greatest during the evening peak. This is consistent with a greater proportion of goods vehicle activity during the morning, as is shown in Figure 6.86.

Figure 6.79 Marylebone Road hourly CO against PM_{10} and $PM_{2.5}$ (TEOM) in 2002.



627. The diurnal variation of the measured CPC particle counts is also included in Figure 6.84 (for the period January 2001 to March 2002). The diurnal variation in the particle count is more similar to that for PM and NO_x than to the diurnal variation for CO and benzene. Figures 6.87 and 6.88 show the correlation between and PM and CPC particle count and CPC particle count and NO_x concentrations at Marylebone Road. The particle count appears to be more closely correlated with NO_x than with PM.

628. In the final analysis in this section, Figure 6.89 and 6.90 show the correlation between the roadside increments of hourly PM concentrations at Marylebone Road in 2002 with the roadside increments of NO_x and CO concentrations (the roadside increment is the measured roadside concentration with the measured value from a nearby background site subtracted. In this case we have chosen London Bloomsbury, where both PM_{10} and $PM_{2.5}$ concentrations are measured.) The correlation coefficients for this roadside increment analysis are not necessarily better than for the total roadside concentrations due to the uncertainties associated with the subtraction of background concentrations, but the intercepts are smaller and the gradients of the lines should be more directly comparable with emission factors.

Figure 6.80 Harwell hourly NO_x against PM_{10} and $PM_{2.5}$ (TEOM) in 2002.

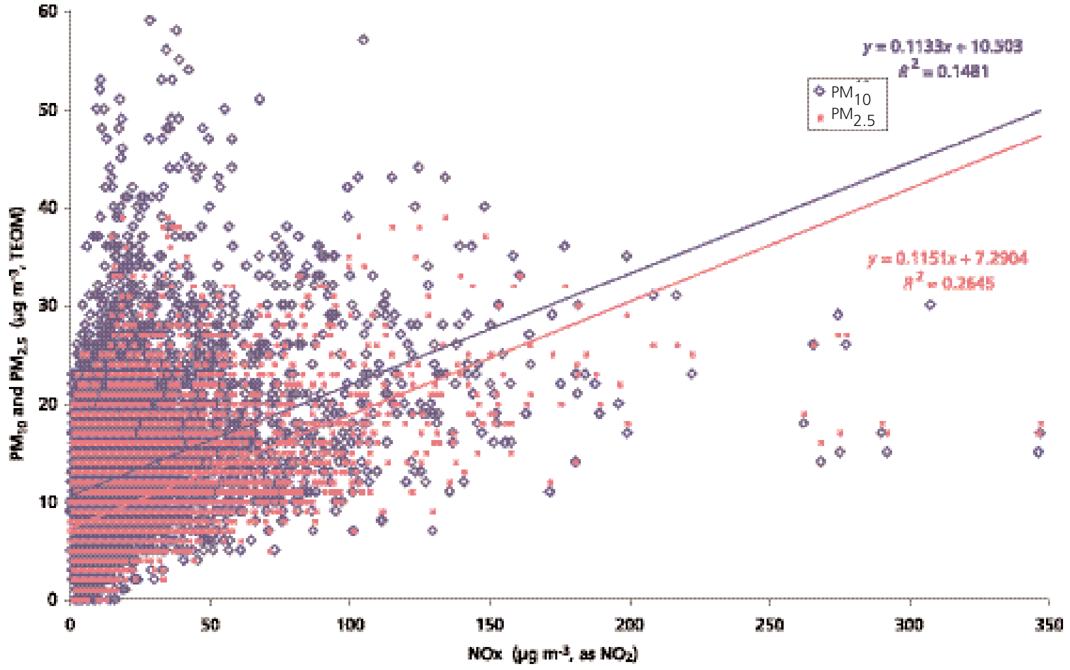


Figure 6.81 Harwell hourly benzene against PM_{10} and $PM_{2.5}$ (TEOM) in 2002.

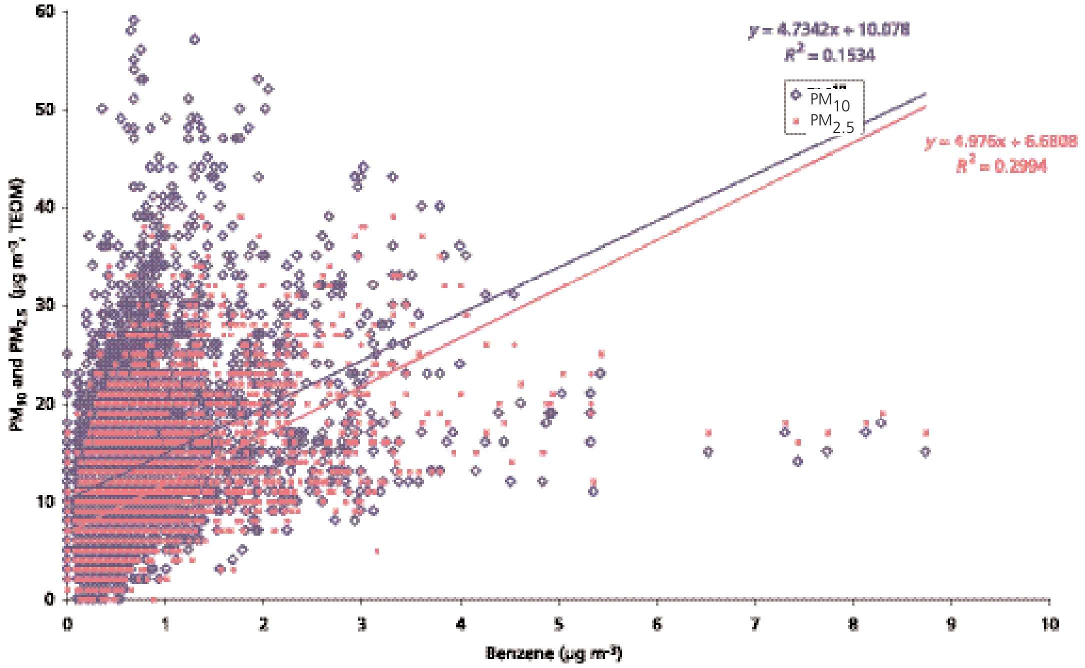


Figure 6.82 Relationship of PM_{10} and $PM_{2.5}$ with daily maximum running 8-h mean ozone; Harwell 2002 and 2003.

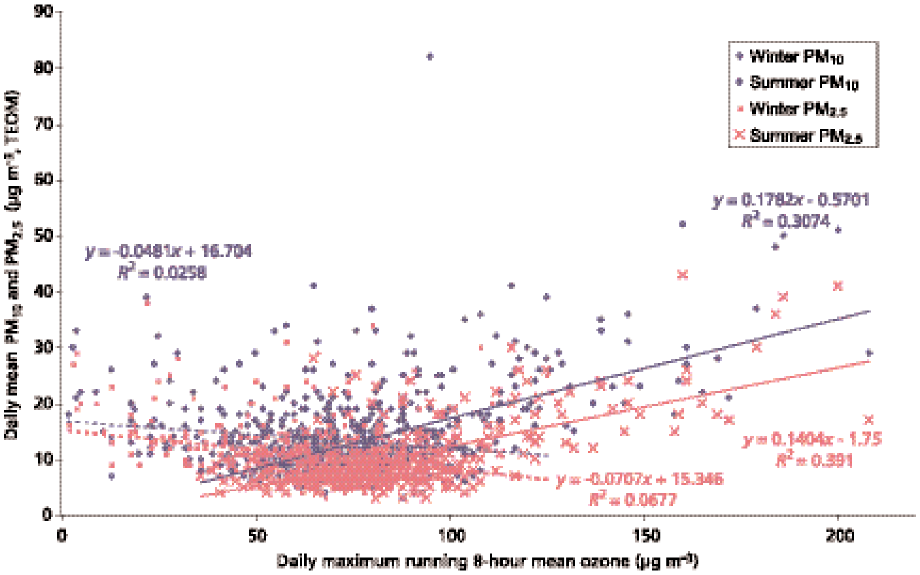


Figure 6.83 Relationship of PM_{10} and $PM_{2.5}$ with daily maximum running 8-h mean ozone; Marylebone Road 2002 and 2003.

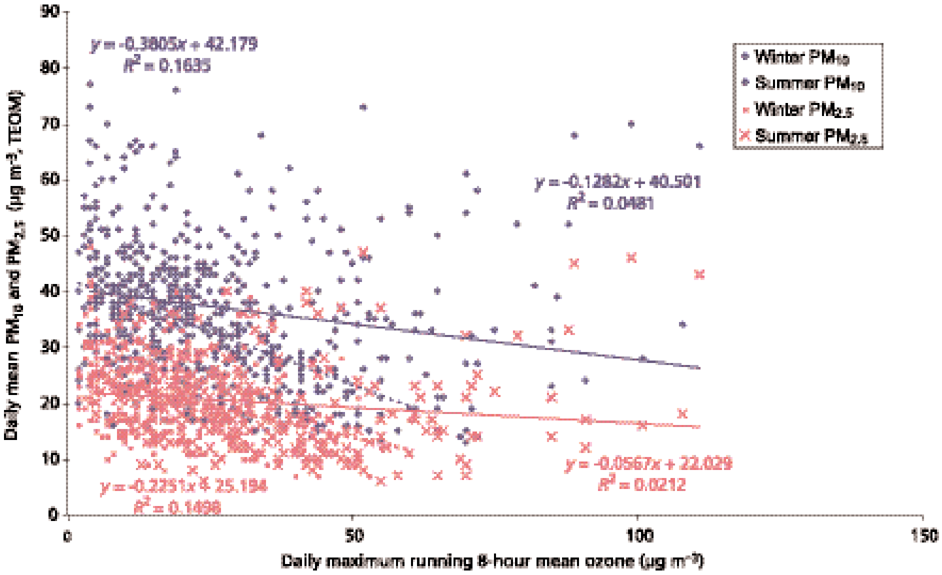


Figure 6.84 London Marylebone Road diurnal concentrations normalised by average.

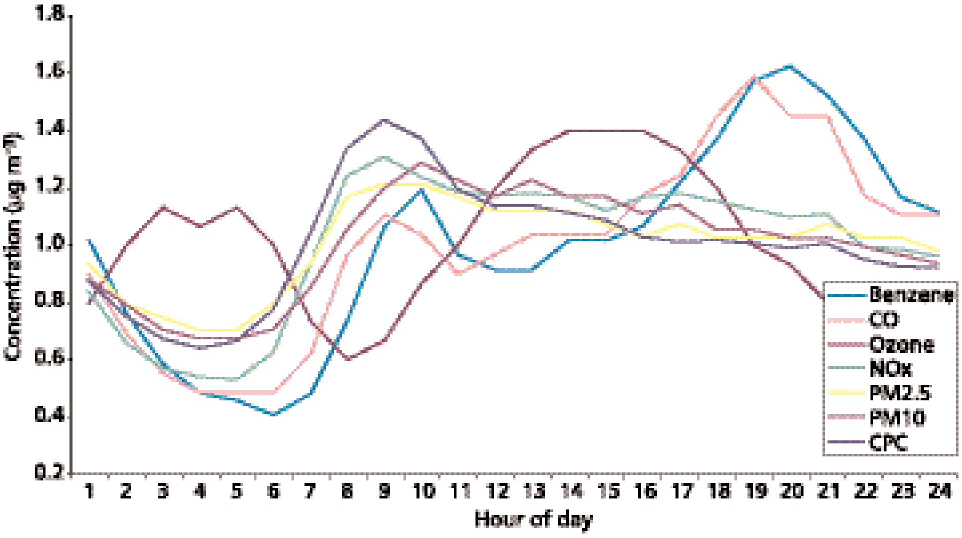


Figure 6.85 Harwell diurnal concentrations normalised by average.

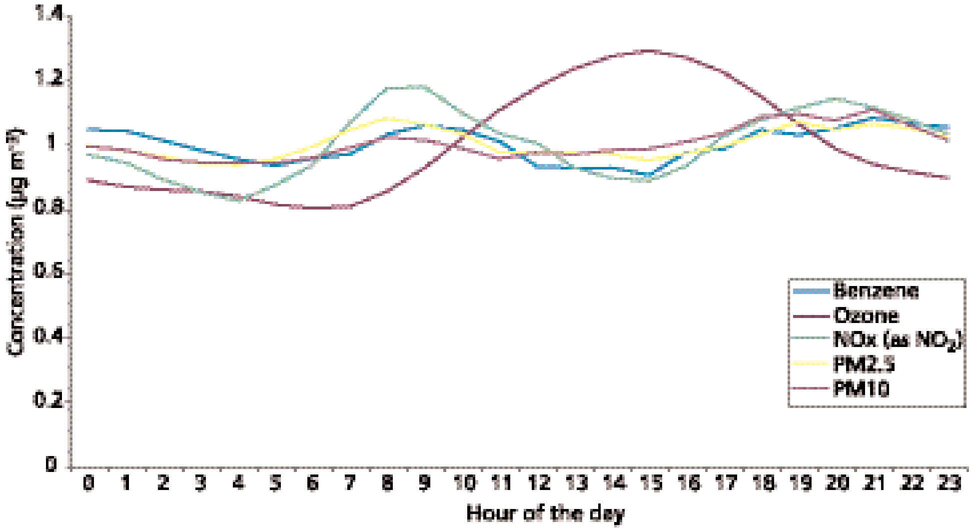


Figure 6.86 London Marylebone Road diurnal traffic flow data, normalised by average, local time.

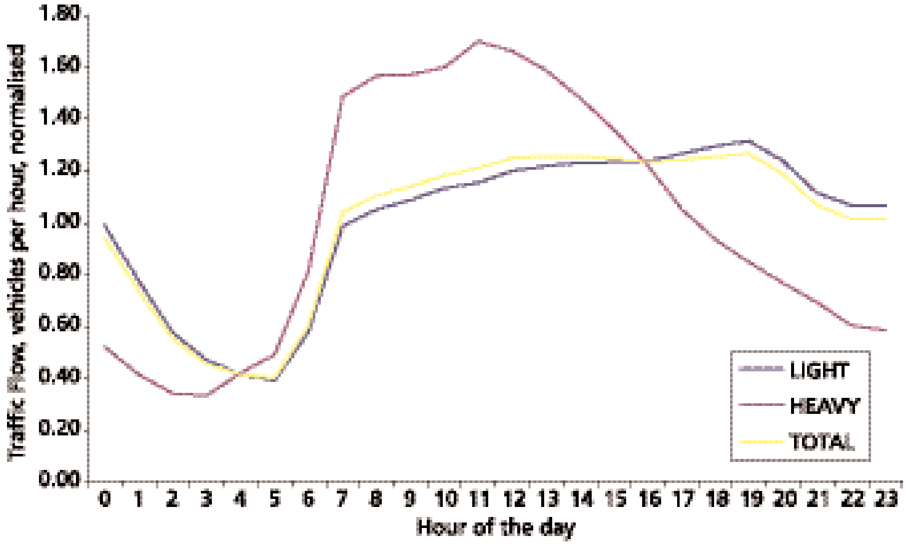


Figure 6.87 PM_{10} and $PM_{2.5}$ versus CPC at Marylebone Road, January 2001 to March 2002.

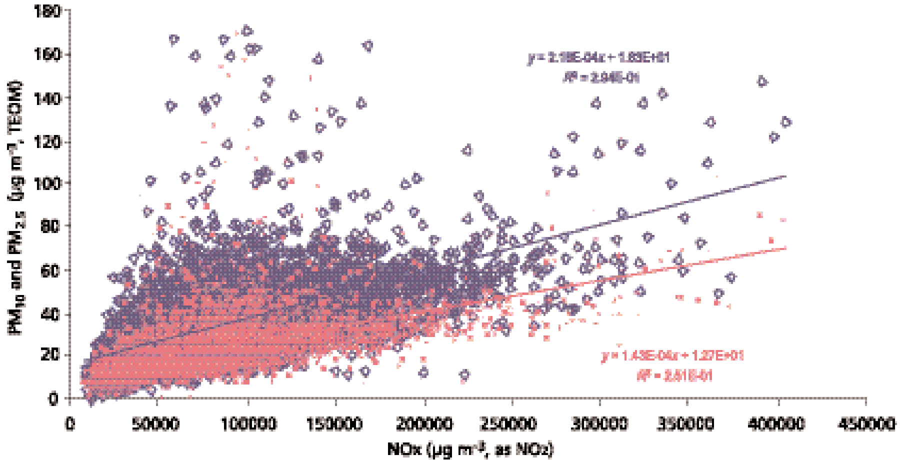


Figure 6.88 CPC versus NO_x (paired hourly data) at Marylebone Road.

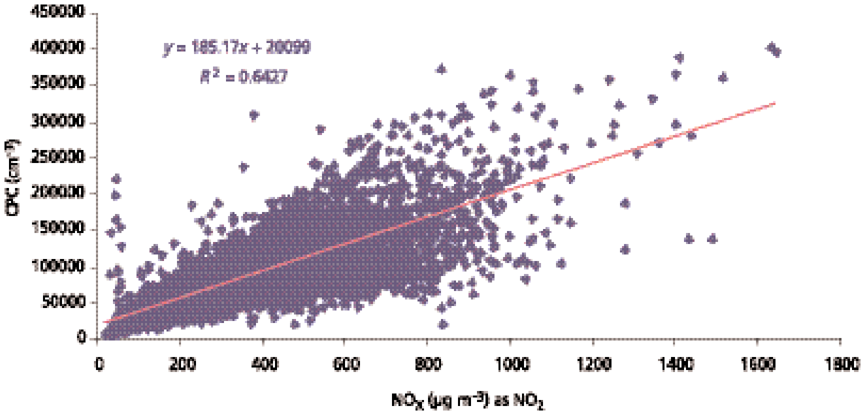


Figure 6.89 Marylebone Road PM_{10} and $PM_{2.5}$ increment against NO_x.

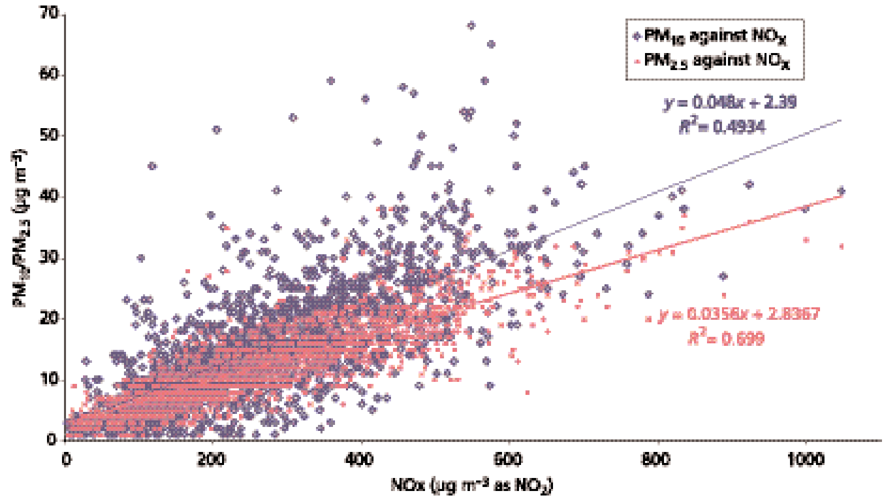
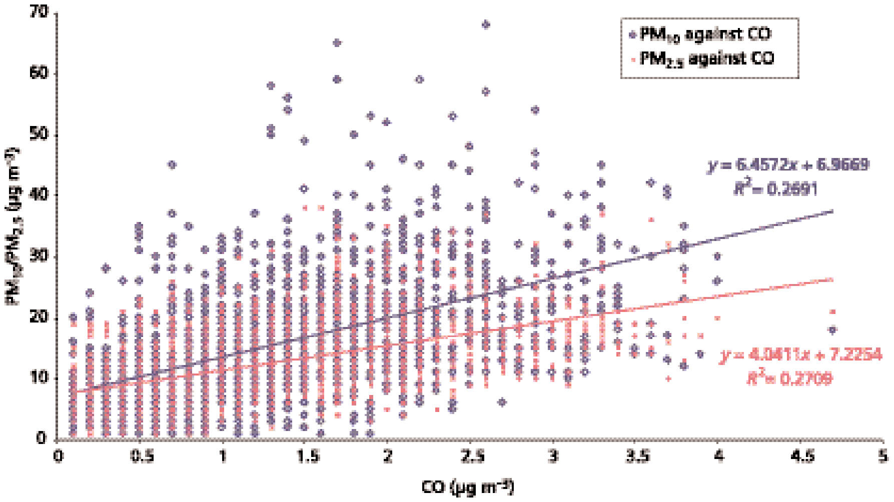


Figure 6.90 Marylebone Road roadside PM_{10} and $PM_{2.5}$ increment against CO.



629. Table 6.16 shows the gradients determined in this comparison of hourly roadside increments and the ratios of the NAEI estimates of road traffic exhaust emission for Marylebone Road in 2001 and for total UK urban traffic emissions in 2001. The measured gradients are lower for $PM_{2.5}$ than for PM_{10}. The implied ratio of PM_{10} and NO_x traffic emission at Marylebone Road is somewhat lower than the NAEI estimates for this road but similar to the ratio for total UK urban emissions. This is a little unexpected, as it suggests that exhaust emissions are sufficient to account for the measured roadside increment of PM_{10}, without any contribution from non-exhaust traffic emissions. The observed ratio of PM_{10}:CO concentrations is lower than suggested by the comparisons of traffic emissions.

Table 6.16 Empirically determined gradients and ratios of emissions (dimensionless).

	$PM_{2.5}:NO_x$	$PM_{2.5}$:CO	$PM_{10}:NO_x$	PM_{10}:CO
Marylebone Road measurements 2002	0.036	0.0040	0.048	0.0065
Marylebone Road emission 2001	—	—	0.055	0.0116
UK urban emissions 2001	—	—	0.046	0.0079

6.4.2 Weekday/weekend differences at traffic-influenced sites

630. Table 6.17 shows the results of a comparison of PM concentrations at Marylebone Road on weekdays (Monday to Friday) and Sundays. Measured PM concentrations were higher at Marylebone Road on weekdays than on Sundays. The weekday-Sunday difference was greater for PM_{10} than for $PM_{2.5}$ and was similar for both Partisol and TEOM measurements. The percentage difference was also very consistently ~20% for the different monitoring methods and $PM_{2.5}$, PM_{10} and PM_{coarse} fractions. The weekday-weekend difference was smaller at the nearby background sites (North Kensington and London Bloomsbury) but was still evident (North Kensington was chosen as the most appropriate background site for this analysis, but TEOM $PM_{2.5}$ measurements are not available at this site.) The weekday-Sunday difference for the roadside increment was therefore somewhat smaller than for the total roadside concentration but generally larger in percentage terms. The weekday-Sunday difference for particle count for the period January 2001 to March 2002 was also calculated for comparison and was ~25%. Interestingly, the total traffic count was similar on weekdays and Sundays at Marylebone Road but there was more light traffic on Sundays than on weekdays and the count for heavy traffic was much lower on Sundays. This suggests that heavy traffic had the strongest influence on the difference in measured concentrations between weekdays and Sundays.

Table 6.17 Average PM concentrations for weekdays and Sundays during January 2002 to October 2003 at Marylebone Road (µg m^3).

	Weekdays	Sundays	Weekdays-Sundays difference
$PM_{2.5}$ (Partisol)	29	23	5 (19%)
PM_{10} (Partisol)	47	39	8 (18%)
$PM_{2.5}$ (TEOM)	22	17	4 (20%)
PM_{10} (TEOM)	38	30	8 (20%)
PM_{coarse} (Partisol)	17	13	4 (22%)
PM_{coarse} (TEOM)	16	13	3 (21%)
North Kensington $PM_{2.5}$ (Partisol)	20	18	2 (10%)
North Kensington PM_{10} (Partisol)	28	24	3 (12%)
London Bloomsbury $PM_{2.5}$ (TEOM)	14	12	2 (14%)
North Kensington PM_{10} (TEOM)	21	19	3 (14%)
North Kensington PM_{coarse} (Partisol)	8	7	2 (19%)
North Kensington PM_{coarse} (TEOM)	7	6	1 (16%)
Roadside increment $PM_{2.5}$ (Partisol)	9	6	3 (32%)
Roadside increment PM_{10} (Partisol)	18	14	4 (25%)
Roadside increment PM_{coarse} (Partisol)	8	7	2 (18%)
Roadside increment $PM_{2.5}$ (TEOM)	7	5	2 (31%)
Roadside increment PM_{10} (TEOM)	16	11	5 (29%)
Roadside increment PM_{coarse} (TEOM)	9	7	2 (26%)
CPC (cm^3) (January 2001 to March 2002)	1,325,265	961,576	36,3689 (27%)
Light traffic (count per day)	67556	69699	–2143 (–3%)
Heavy traffic (count per day)	9337	4093	5244 (56%)
Total traffic (count per day)	76896	73478	3418 (4%)

631. The weekday-Sunday difference was a similar magnitude to Marylebone Road at Bury roadside and M25 Staines (Table 6.18). The total traffic counts was much lower at Bury roadside on Sundays than on weekdays; the difference was rather smaller at M25 Staines. The variation in rural sulphate concentrations by day of the week was also investigated but no consistent patterns were found.

Table 6.18 Average PM concentrations for weekdays and weekends January 2002 to December 2003 at Bury roadside and M25 Staines (µg m^3).

	Weekdays	Sundays	Weekdays-Sundays difference
Bury roadside PM_{10} (TEOM)	26	20	6 (23%)
Bury roadside total traffic (counts per day)	146,869	97,456	49,413 (34%)
M25 Staines $PM_{2.5}$ (TEOM)	14	12	3 (18%)
M25 Staines PM_{10} (TEOM)	24	18	5 (23%)
M25 Staines PM_{coarse} (TEOM)	9	7	3 (30%)
M25 Staines total traffic (counts per day)	197,849	167,410	30,438 (15%)

6.4.3 Influence of air mass back trajectory and weather type on PM concentrations

632. Daily concentrations of PM_{10} in Edinburgh for the period 1992–2000 vary significantly with the geographical sector of the back trajectory calculated for the arriving air mass (Figure 6.91). Trajectories were categorized objectively by statistical hierarchical clustering using mean distance squared and average linkage clustering on the vectors of the 30 variables comprising: distance north of Edinburgh; distance south of Edinburgh; and pressure level at 12, 24, 36…120 h prior to arrival at Edinburgh. There was no *a priori* expectation for the 'best' number of clusters into which to classify the trajectories, which was determined by examining plots of RMS and r^2 as a function of number of trajectories for major discontinuities in slope. Overall, five major clusters emerged, together accounting for >94 % of the total number of trajectories. Median PM_{10} on days when air mass originated from continental Europe or from local circulation about the British Isles (23 and 18 µg m^3 (TEOM), respectively) were significantly elevated compared with days when air mass originated from Atlantic SW, Atlantic W and the Arctic (15, 15 and 16 µg m^3 (TEOM), respectively). This means that air-mass source can be associated with increases of up to 50%, on average, of receptor PM_{10} in the UK.

Figure 6.91 Apportionment of 24 h PM_{10} concentrations in Edinburgh (1992–2000) by geographical sector of air mass back trajectory. (Cluster 1, Local British Isles; Cluster 2, Atlantic SW, Cluster 3, European; Cluster 4, Arctic; Cluster 5, Atlantic W. The box edges define lower and upper quartiles; lines extend to 5th and 95th percentiles.)

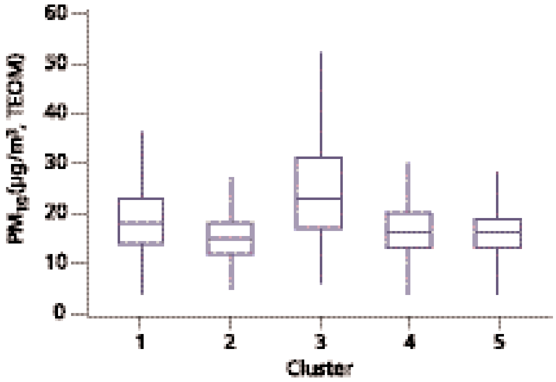
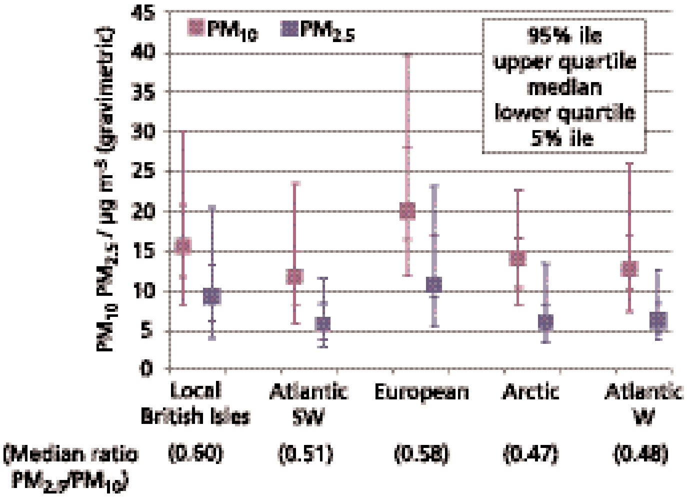


Figure 6.92 Apportionment of 24 h PM_{10} and $PM_{2.5}$ concentrations in Edinburgh (1999–2000) and $PM_{2.5}$:PM_{10} ratio by geographical sector of air mass back trajectory.



633. Data for colocated PM_{10} and $PM_{2.5}$ for the shorter period 1999–2000 show a similar, but accentuated, trend for $PM_{2.5}$ (Figure 6.92). For this more recent period, median PM_{10} on days when air mass originated from continental Europe or from local circulation about the British Isles (20 and 16 µg m^3 (gravimetric), respectively) was still significantly elevated compared with days when air mass originated from Atlantic SW, Atlantic W and the Arctic (12, 13 and 14 µg m^3 (gravimetric)). Likewise, $PM_{2.5}$ was significantly elevated (by 3–4 µg m^3 (gravimetric) or >50%, on average) on days with air masses from continental Europe and the British Isles. Importantly, the ratio of $PM_{2.5}$ to PM_{10} was also significantly elevated on these days, as shown by the numbers at the bottom of Figure 6.92. This demonstrates the influence in the UK of sources of anthropogenically derived fine particles, independent of local scale air quality management areas.

634. Similar findings were noted by Buchanan *et al*. (2002), who divided data according to the classification of daily regional scale synoptic weather maps into Jenkinson weather types to show that PM_{10} in Edinburgh was significantly higher for anticyclonic, southerly and southeasterly Jenkinson weather types, indicative of air flow from continental Europe.

6.4.4 Analysis of trends in PM concentrations by wind direction

635. Figures 6.93 and 6.94 show an analysis of the trend in mean PM_{10} concentration at London Bexley by wind direction. An examination of the mean concentrations in each 10° bin has indicated that the concentration when the wind is from directions between 4° and 120° (east) were generally higher than the mean of all wind directions (due to the elevated secondary PM concentration in easterly winds). Previous analysis of particulate sulphate measurements at rural sites in the UK has shown that concentrations are highest for easterly flow (RGAR, 1999). Concentrations when the wind was from the southwest (190° to 270°) were generally lower than the mean. The mean of concentrations when the wind was from all other wind directions was generally close to the mean for all directions.

636. The unusually high secondary PM concentration in 1996 and 2003 are clearly shown in Figure 6.93. Concentrations in the southwesterly direction showed a decline from 1994 to 1998 and were then relatively constant. Concentrations from the east decline to 2000 and then increased. Figure 6.94 shows the contributions from the wind directions to the annual mean concentrations. The contribution from the east was highest in 1996, 1997 and 2003. The contribution from the southwest was reasonably constant throughout. Figure 6.95 shows the frequency of wind directions within these categories. The frequency of 'other' directions was relatively consistent throughout the period. The frequency of wind from the southwest was high in 1998, 1999 and 2000 and the frequency of winds from the east was correspondingly low in these years. 1996 and 2003 were the years with the highest concentration in easterly winds and the greatest frequency of winds from this direction.

637. Figures 6.96 to 6.98 show a similar analysis for Belfast Centre. Mean concentrations for the easterly direction were relatively constant between 1999 and 2003 but the contribution to the annual mean from this wind direction was much greater in 2003, due to the greater prevalence of easterly winds. The frequency of winds from the southwest decreased over the period leading to a decline in the contribution from the southwesterly direction seen in Figure 6.97. Overall the frequency of winds from the east was lower than in London.

638. Figures 6.99 to 6.101 show the results of the analysis for the Birmingham Hodge Hill site. The PM_{10} concentration in winds from the east shows a decline over the period but a rise between 2002 and 2003. Concentrations when the wind was from the east were particularly low in 1999. The PM_{10} concentration for southwest winds also shows a decline. The relative frequencies of the wind directions at this site were reasonably consistent with those for Bexley with less winds from the east in 1998, 1999 and 2000 and a higher frequency of winds from the southwest in these years. The contribution from winds from the east to annual mean PM_{10} concentration was highest in 1995, 1996 and 1997, with a rise from 2002 to 2003.

639. $PM_{2.5}$ concentrations have also been measured at Birmingham Hodge Hill and this enables the analysis of the concentrations measured for different wind directions to be carried out for both $PM_{2.5}$ and PM_{coarse}, in addition to PM_{10}. Figures 6.102 and 6.103 show the results for $PM_{2.5}$. Once again concentrations were greater for winds from the east. The trends with time were similar to those for PM_{10} but the overall decline in concentrations was less steep. Figures 6.104 and 6.105 show the results for PM_{coarse}. The PM_{coarse} concentrations showed a decline for all wind directions. The PM_{coarse} concentrations showed less variation in concentration with wind directions than $PM_{2.5}$. The decline in annual mean $PM_{2.5}$ concentrations at Birmingham Hodge Hill has, however, not been consistently observed across the other UK monitoring sites, as is illustrated in Figure 6.106.

Figure 6.93 Mean PM_{10} concentration by wind direction at London Bexley ($\mu g\ m^3$).

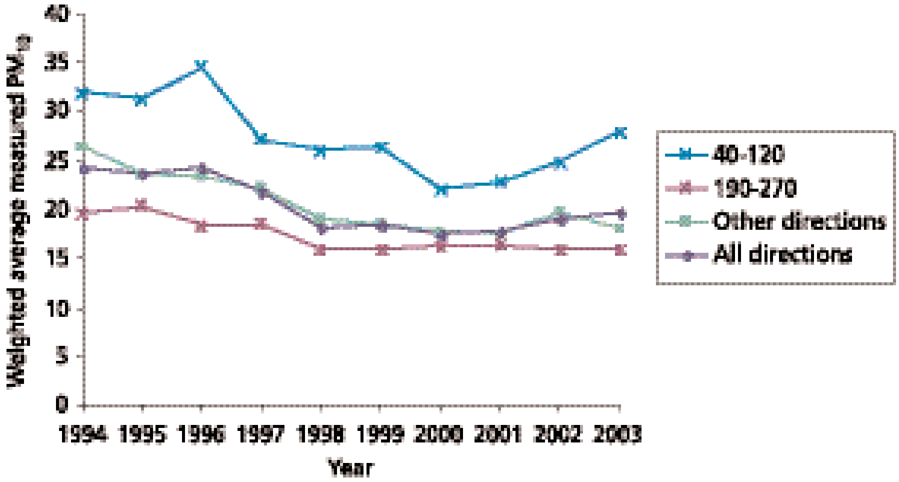


Figure 6.94 Bexley time series PM_{10} contribution by year and direction.

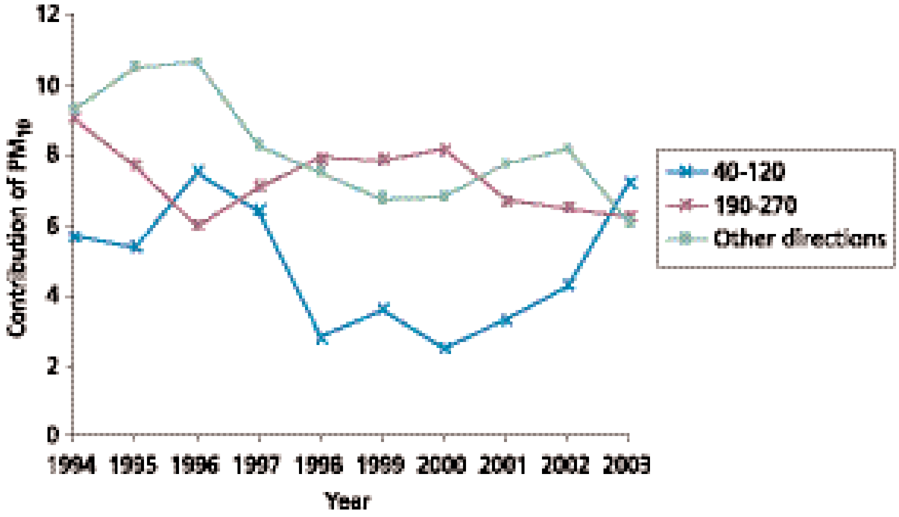


Figure 6.95 Bexley time series of frequency of wind direction by direction.

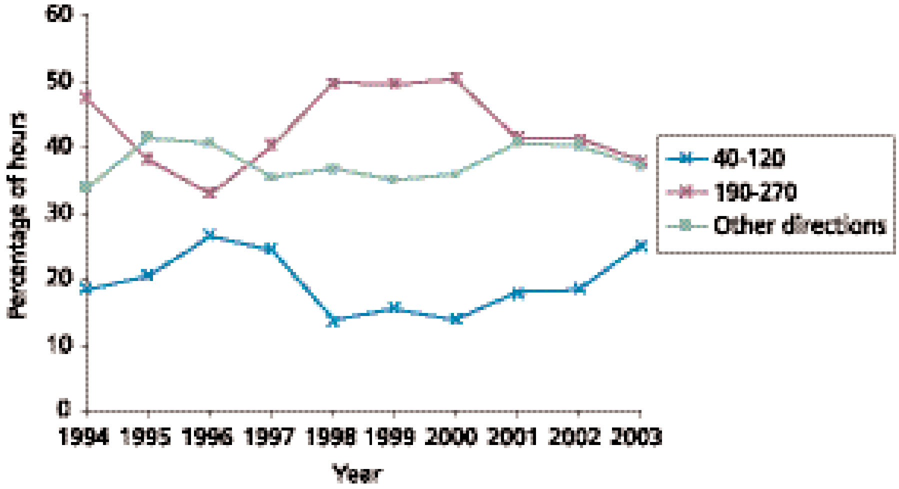


Figure 6.96 Mean PM_{10} concentration by wind direction at Belfast centre (µg m^3).

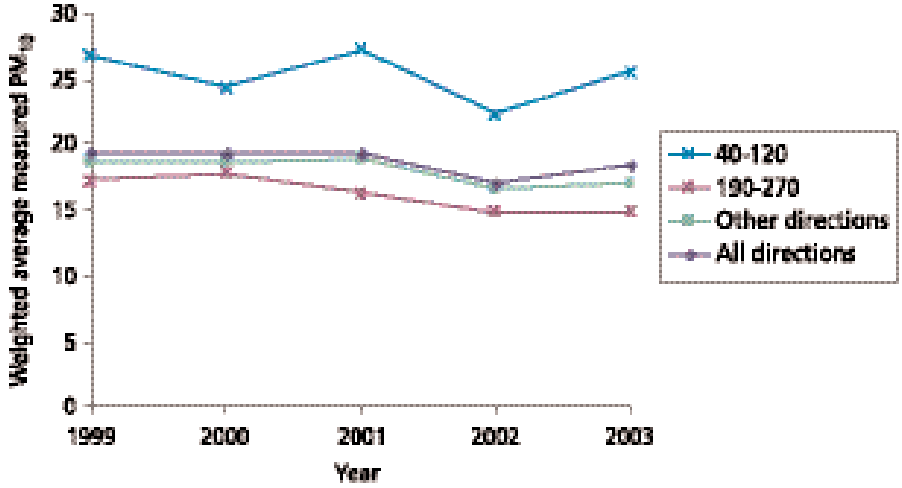


Figure 6.97 Belfast centre time series PM_{10} contribution by year and direction.

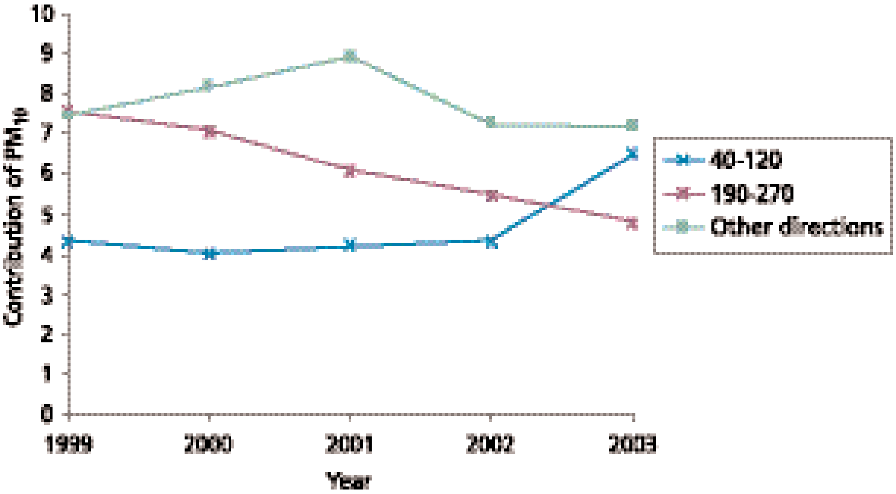


Figure 6.98 Belfast centre time series of frequency of wind direction by direction.

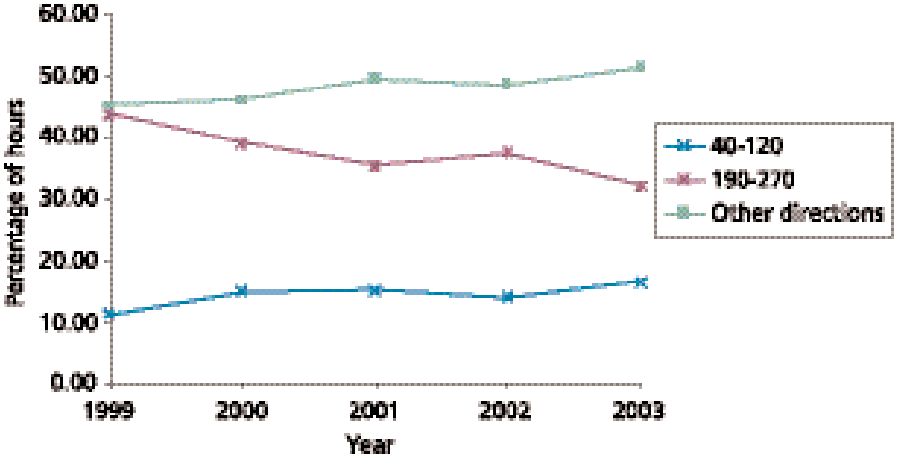


Figure 6.99 Mean PM_{10} concentration by wind direction, Birmingham Hodge Hill (µg m^3).

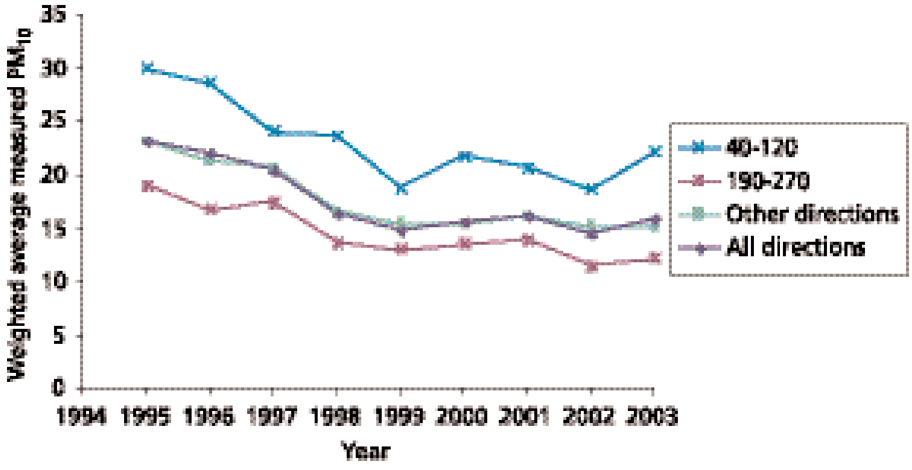


Figure 6.100 Birmingham Hodge Hill time series PM_{10} contribution by year and direction.

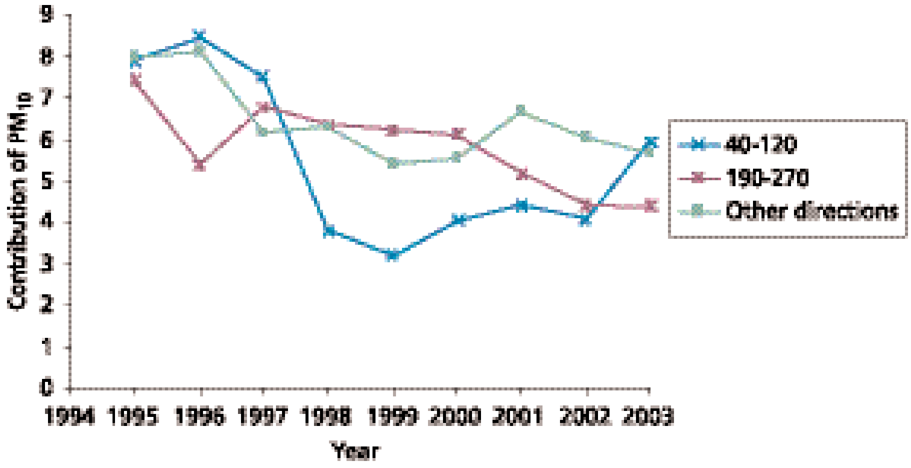


Figure 6.101 Birmingham Hodge Hill time series of frequency by wind direction.

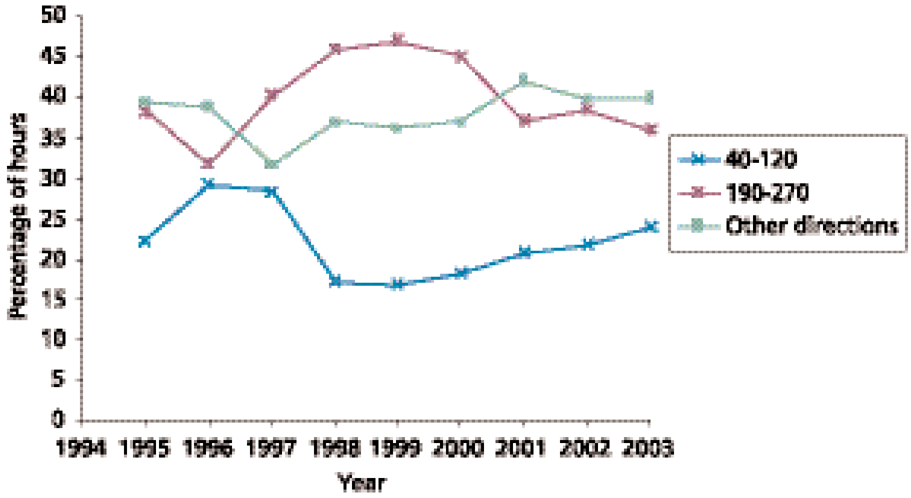


Figure 6.102 Mean $PM_{2.5}$ concentration by wind direction, Birmingham Hodge Hill (µg m^3).

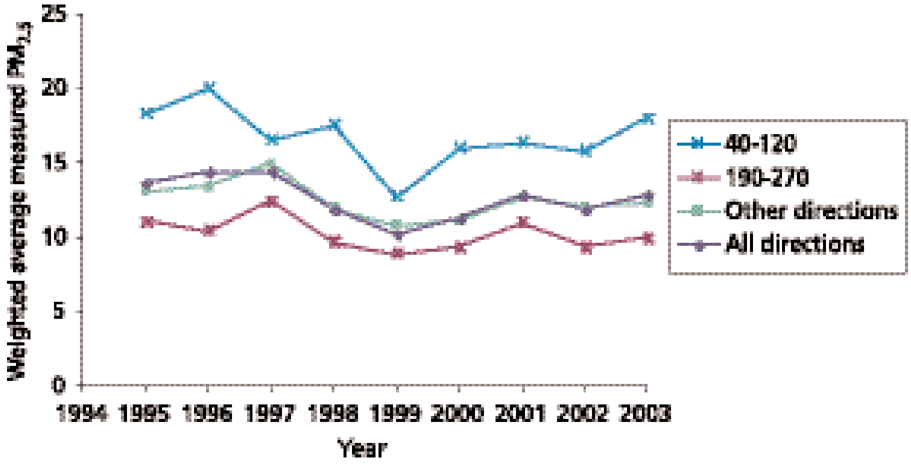


Figure 6.103 Birmingham Hodge Hill time series PM_{10} contribution by year and direction.

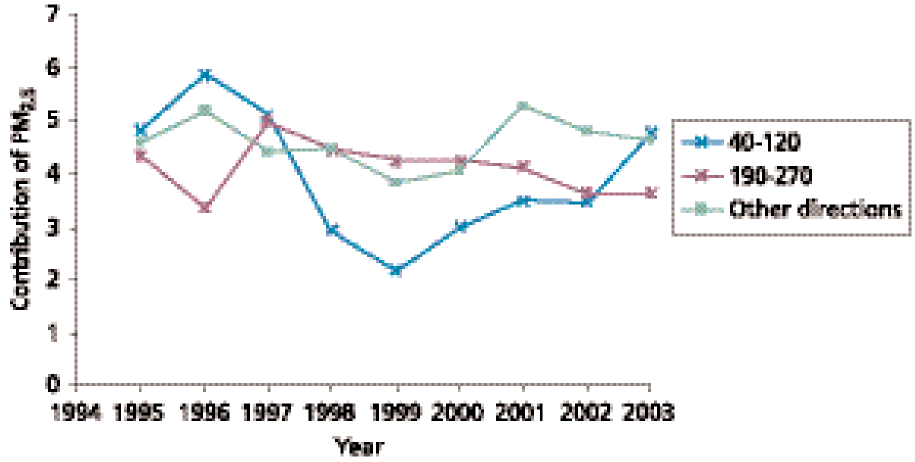


Figure 6.104 Mean PM_{coarse} concentration by wind direction, Birmingham Hodge Hill (µg m^3).

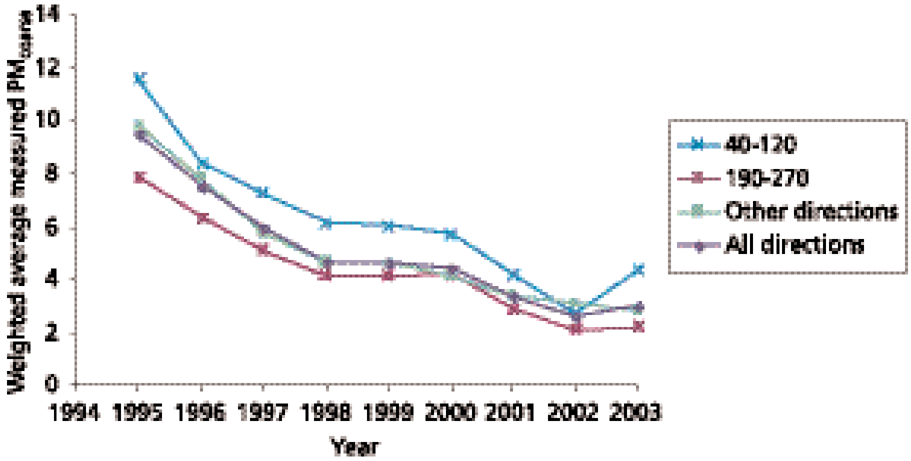
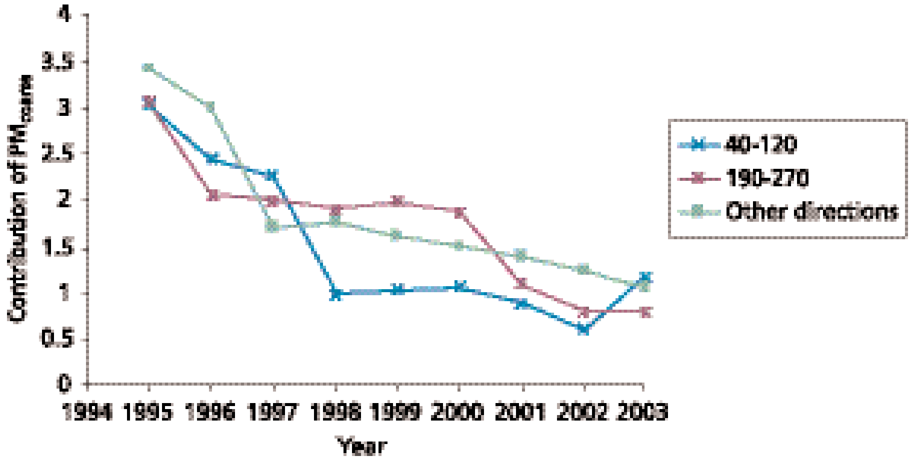


Figure 6.105 Birmingham Hodge Hill time series PM_{coarse} contribution by year and direction.



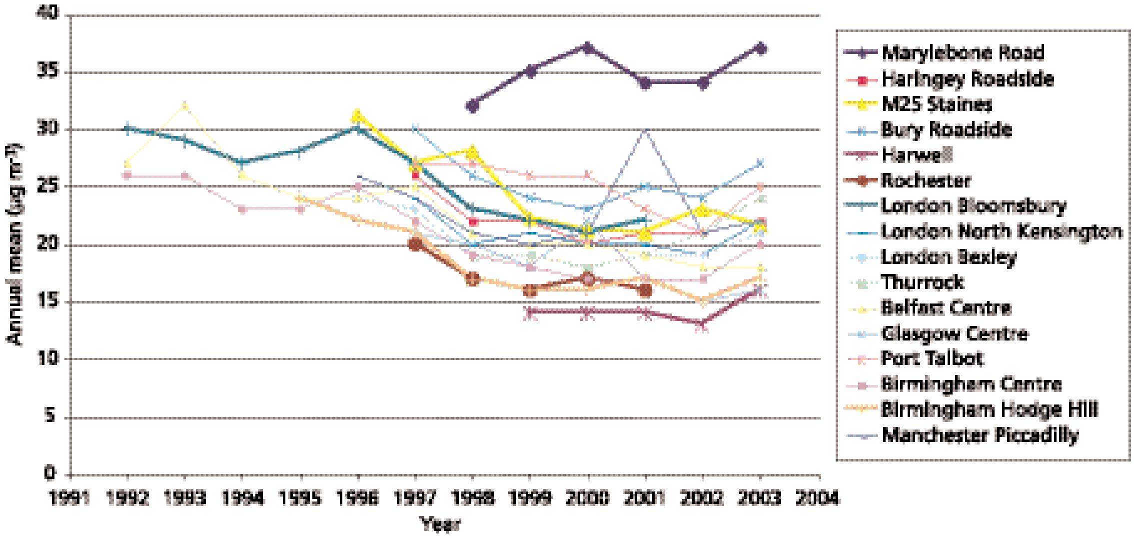
6.4.5 Coarse particles at Marylebone Road: sources and concentrations

6.4.5.1 Concentrations measured at Marylebone Road

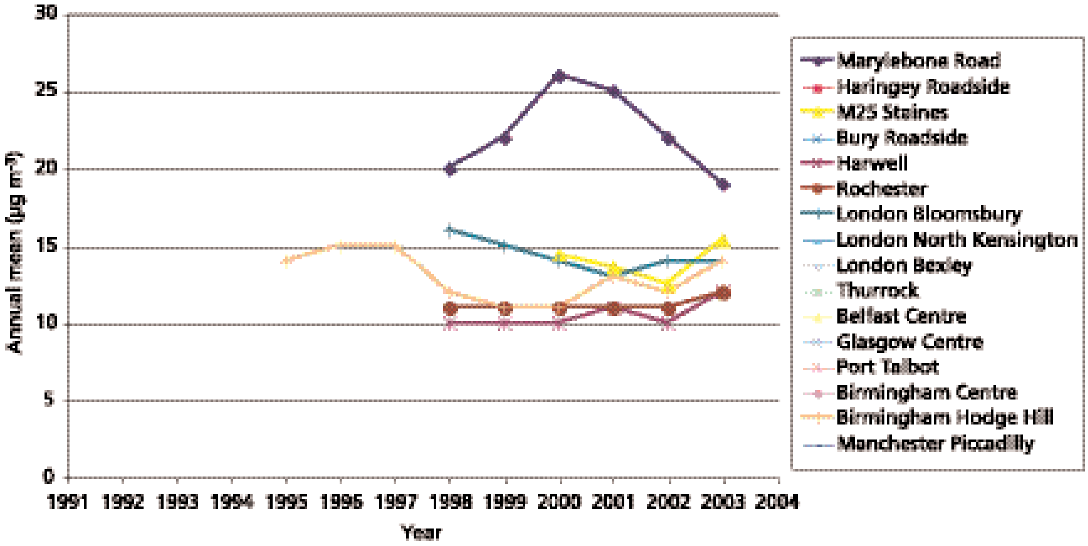
640. As shown in Figure 6.107, hourly concentrations of PM_{coarse} measured at Marylebone Road from July 1998 to August 2001 were highly variable. They ranged from 1 to 595 µg m^{-3} and followed lognormal distributions. High particle mass concentrations in comparison to urban background sites are measured in Marylebone Road. More than 25% of hourly PM_{coarse} concentrations were above 15 µg m^3. Some very high concentrations (above 100 µg m^{-3}) were occasionally measured. They were generally associated with demolition or construction activities. No seasonal variation was observed at Marylebone Road. This suggests that there is no significant temporal variation of influential meteorological parameters and sources of particulate matter at Marylebone Road on a seasonal timescale, or that the effects of different factors cancel one another.

Figure 6.106 TEOM measurements for (a) PM_{10}, (b) $PM_{2.5}$ and (c) PM_{coarse} during 1991–2003.

(a)



(b)



(c)

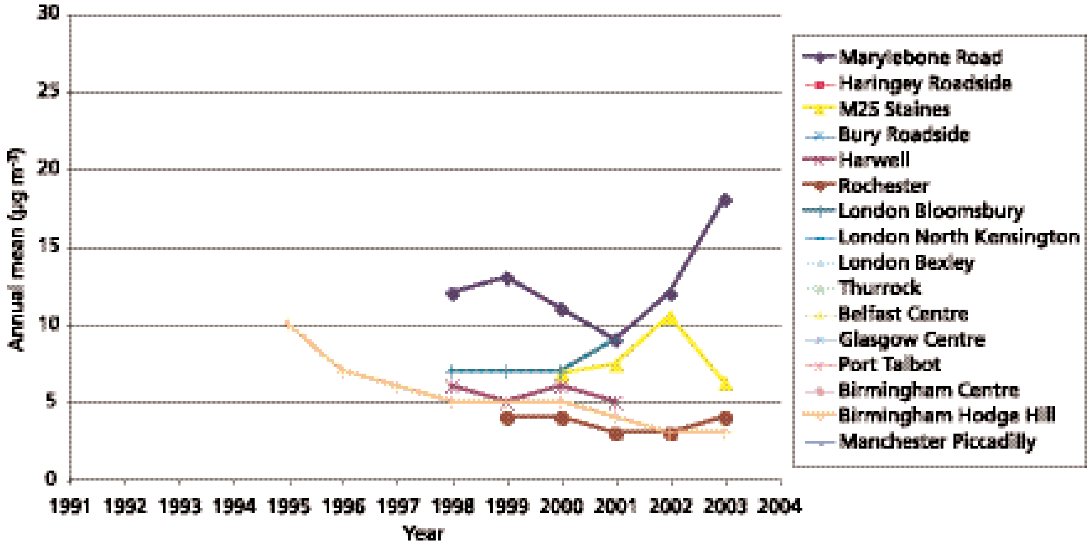
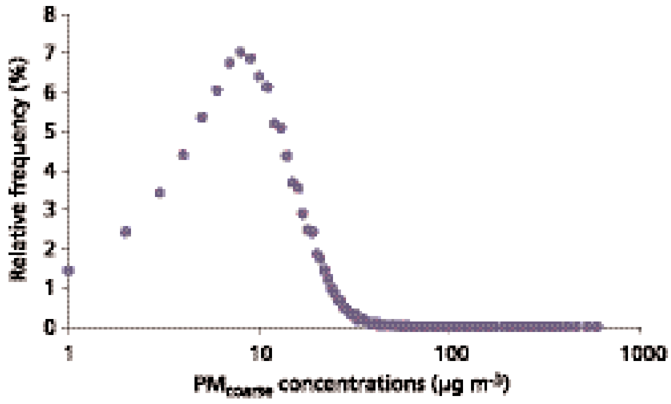


Figure 6.107 (a) Basic statistics for $PM_{2.5}$ and PM_{coarse} concentrations measured at Marylebone Road and (b) relative frequencies of PM_{coarse} concentrations (logarithmic scale) when each size bin is 1 µg m^3 in width. (Hourly data from July 1998 to August 2001 (*n* = 22,268).)

(a)

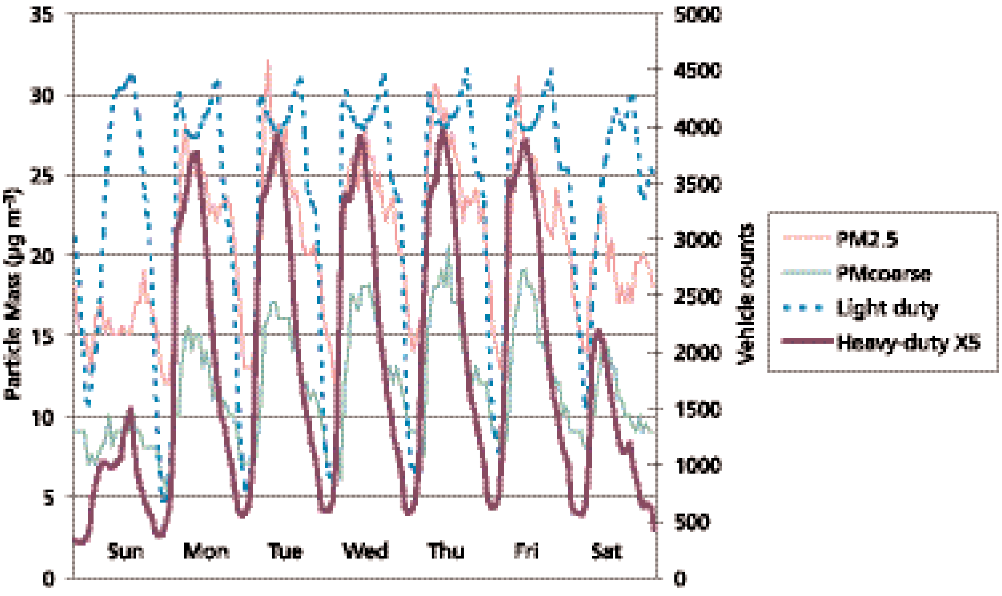
	$PM_{2.5}$ (µg m^3)	**PM_{coarse} (µg m^3)**	**PM_{10} (µg m^3)**
Minimum	1	1	3
Maximum	330	595	800
Median	21	10	32
Arithmetic mean	23.3	12.7	36.0

(b)



641. Figure 6.108 presents the median weekly cycles of $PM_{2.5}$ and PM_{coarse}. The median weekly cycles of light duty traffic (cars and motorcycles) and of heavy duty traffic (lorries, buses, coaches) are also represented in this figure. Very similar patterns of traffic are observed from one week to another, and the median weekly cycles of traffic densities well represent the single weekly cycles. The median weekly cycle of $PM_{2.5}$ and PM_{coarse} concentrations at Marylebone Road shows similar strong daily and weekly variations that correlate with those of heavy duty traffic. Higher concentrations are measured during the daytime on work days (median >15 µg m^3 for PM_{coarse}) and much lower concentrations during night-time and weekends. During the night, the median concentration of PM_{coarse} is <8 µg m^3.

Figure 6.108 Median weekly cycles of $PM_{2.5}$ and PM_{coarse} concentrations and median weekly cycles of light duty and heavy duty vehicle counts.



642. The median $PM_{2.5}$:PM_{10} ratio at Marylebone Road was ~0.66 for the whole period of study, that is, on average one-third of the mass of the PM_{10} is within the PM_{coarse} fraction. Similar ratios are reported elsewhere (Ruellan and Cachier, 2001; Manoli *et al*., 2002). This median mass ratio $PM_{2.5}$:PM_{10} at Marylebone Road is much lower than reported mass fractions from exhaust emissions (in the APEG report it was 0.8 for non-catalyst petrol exhaust, 0.9 for catalyst petrol exhaust and 0.9 for diesel exhaust (APEG, 1999). This suggests that there is an excess of coarse particles in comparison to exhaust emissions.

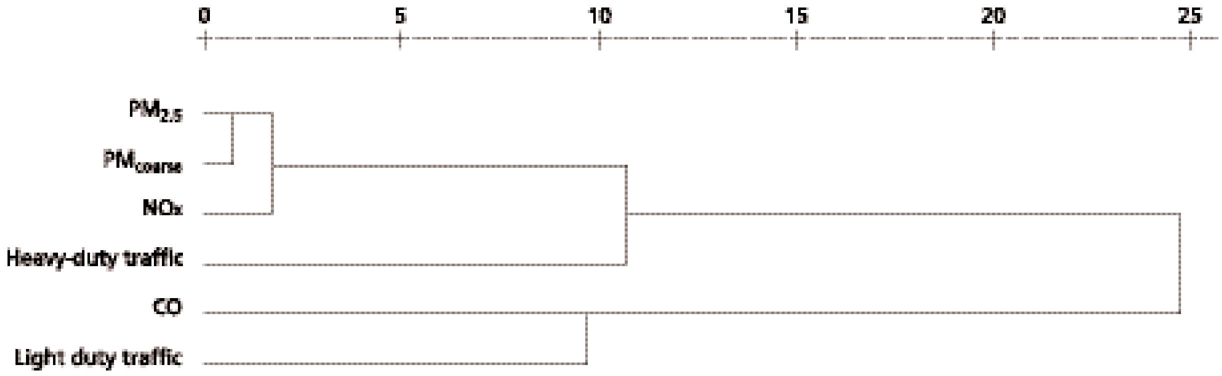
6.4.5.2 Local sources of PM_{coarse} at Marylebone Road

6.4.5.2.1 Traffic

643. The weekly cycle of PM_{coarse} suggests that PM_{coarse}, as well as $PM_{2.5}$, is related to the traffic. The highest concentrations are measured during the midday period of the work days when the heavy duty traffic is at a maximum level and reaches 20% of the total traffic flow. The lowest concentrations are measured on Sundays and a minor peak in concentration is seen on Saturday mornings. Both $PM_{2.5}$ and PM_{coarse} cycles are clearly closer to the heavy duty traffic cycles than to the light duty traffic cycle (see Figure 6.108). The stronger wind and air turbulence during the day would lead to the same PM_{coarse} concentration patterns on work days as on weekends if the coarse particulate matter were only mechanically induced by the wind; this is clearly not the case. A high Spearman rank correlation coefficient is found between the PM_{coarse} and the heavy duty traffic ($r = 0.80$). A high Spearman rank correlation coefficient is also found between both the $PM_{2.5}$ and the heavy duty traffic ($r = 0.77$).

644. The relationship between traffic data and particle data was also examined using a hierarchical cluster analysis (Wards' method and square Euclidean distance as a similarities measure). This multivariate method separates observations into groups according to their similarities and differences. The median weekly cycle of concentrations is used to smooth the variability of concentrations due to the variability of the meteorological parameters. Only the concentrations measured when the wind direction favoured the measurement of on-road emissions were included in the analysis, that is to say, the following wind directions were considered: (i) wind parallel to the street (accumulation of on-road emissions) and (ii) from southerly directions (vortex favouring the measurement of on-road emissions). The two tracers, NO_x and CO were simultaneously examined. The results obtained with this method are presented in Figure 6.122.

Figure 6.109 Dendogram representing the result of a hierarchical cluster analysis (Wards' method, square Euclidean distances as a similarities measure).



645. According to Figure 6.109, two clusters are clearly identified:

- $PM_{2.5}$, PM_{coarse} and NO_x concentrations are clustered with the heavy duty traffic.
- CO concentrations are clustered with the light duty traffic.

646. These results confirm that PM_{coarse} concentrations as well as $PM_{2.5}$ concentrations are associated with the heavy duty traffic (lorries, coaches and buses). The clustering of NO_x with the heavy duty traffic and of CO with the light duty traffic (mainly petrol vehicles) gives confidence in the results. The heavy duty traffic is mainly comprised of diesel vehicles that have high emission factors for particle mass; this result was expected for the fine fraction of particulate matter. The clustering of the coarse PM with the heavy duty traffic suggests that the substantial turbulence induced by the heavy duty traffic is responsible for a greater magnitude of resuspension of particles from the road than the light duty traffic. Another explanation is the larger amounts of dust emitted by the heavy duty traffic due to stronger abrasion processes such as tyre wear and brake linings. Sternbeck *et al*. (2002) found a strong correlation between particulate barium (which is a common element in brake linings and is not used as a fuel additive in Sweden) and heavy duty traffic, suggesting that heavy duty vehicles are responsible for larger brake wear particulate emissions than light duty vehicles. Large vehicles have higher brake wear emission rates for airborne particles than small vehicles and these emissions would include both mechanically generated coarse particles and the formation of finer particles due to high temperatures at the brake/rotor interface (Garg *et al*., 2000). These results are in agreement with other studies that have shown a link between coarse PM and traffic flow (Janssen *et al*., 1997; Ruellan and Cachier, 2001; Gehrig and Buchmann, 2003).

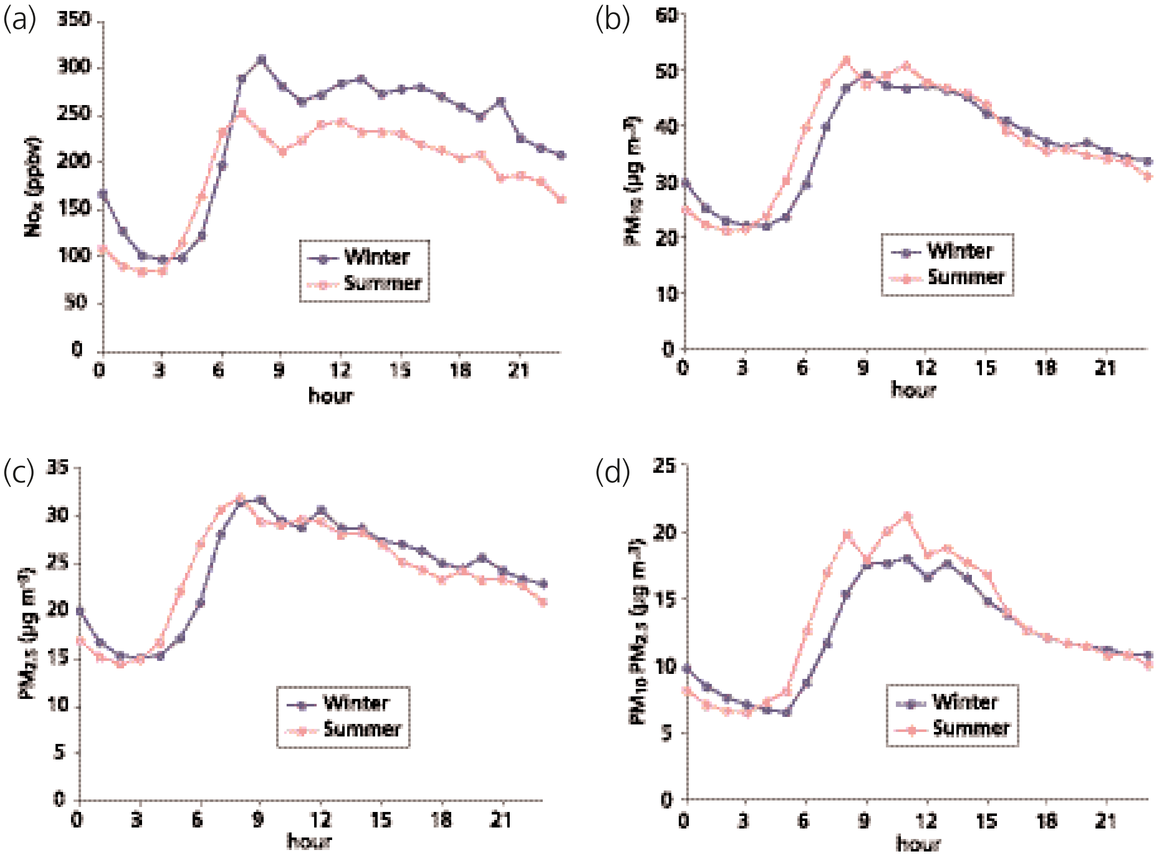
6.4.5.2.2 Construction and demolition activities

647. Analyses of the Marylebone Road data in regard of construction and demolition activities appear in Section 6.3.6.

6.4.6 Estimation of the non-exhaust road transport particle emissions at Marylebone Road

648. Data from the Marylebone Road kerbside site have been analysed to determine the diurnal variations in NO_x and different PM components. Consideration of hourly means of the coarse fraction shows that it is highly variable on this timescale and has a poor relationship with other pollutants such as NO_x or vehicle flows. Because of its highly variable nature, longer term averages are required to reveal how it varies against other parameters. Consideration of Figure 6.110 highlights several important features. First, there is a time lag of 1 h between winter and summer diurnal plots of all pollutants, including the coarse fraction. This time lag is indicative of road traffic activity sources where the lag is observed when Greenwich Mean Time (GMT) changes to British Standard Time (BST). These results strongly indicate that the majority of the coarse fraction observed at Marylebone Road is related to road traffic. Another feature of Figure 6.110(d) is that there is very little difference between PM concentrations in summer and winter, which, unlike NO_x, is clearly lower in summer due to increased dispersion. The similarity between the magnitude of the summer and winter coarse fraction suggests that either the source strength is higher in summer or the dispersion processes are different compared with NO_x.

Figure 6.110 Summer and winter diurnal variations in (a) NO_x, (b) PM_{10}, (c) $PM_{2.5}$ and (d) PM_{10}-$PM_{2.5}$ at Marylebone Road (May 1998–December 2002).



649. The diurnal variation in the coarse fraction shown by Figure 6.110(d) is different from the variation in NO_x and $PM_{2.5}$, indicating different origins for these pollutants.

650. A consideration of how the coarse fraction varies over these periods strongly suggested a link with the flow of HGVs since the coarse fraction is lowest during weekend periods (particularly Sundays) and the diurnal variation through each day was similar to that for HGVs. A multiple regression was carried out to determine whether the observed variation by hour of the day and day of the week of the coarse fraction could be related to other parameters such as other pollutants and vehicle flows. The multiple regression took the form:

$$PM_{coarse} = a * LGV + b * HGV + c.$$

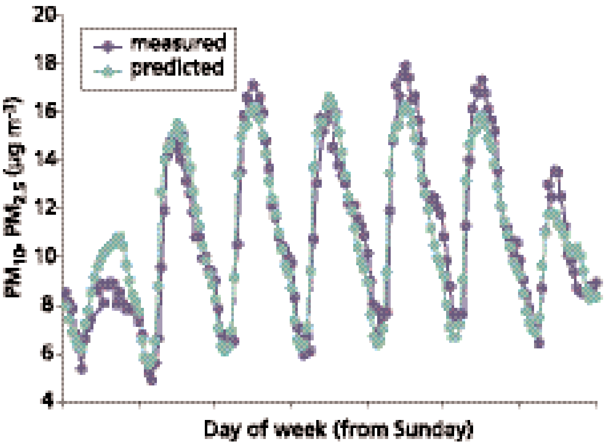
Where LGV and HGV are the hourly mean flows of LGVs and HGVs by day of the week; a, b and c are constants to be derived from the regression. The results from the multiple regression are shown in Table 6.19. These results suggest an HGV emission factor approximately 15-times higher than LGVs. The statistics for the regression also suggest that all calculated constants are significant at the 95% CI, but the relationship with HGVs is much stronger than LGVs. The multiple regression also yielded a high r^2 value of 0.88. The intercept of 3.9 µg m^{-3} is for the coarse fraction that cannot be related to vehicular activity and is presumably related to wind-blown PM and other sources of coarse particles in the urban environment.

Table 6.19 Results from the multiple regression.

	Coefficients	Standard error	*t* statistic
Intercept *c*	3.932492	0.299579	13.1
a	0.00085	0.000112	7.6
b	0.012306	0.000534	23.0

651. Figure 6.111 shows the comparison between the measured and predicted coarse fraction. Overall the agreement is very good, but predicted values tend to be too high on Sundays. One likely explanation for the poor comparison on Sundays is the use of broad categories of vehicle types, which mask important changes for individual vehicle types. For example, an analysis of manual count traffic data in London suggests that rigid and articulated vehicles are ~30% of their weekday flows on Sundays, whereas the continuous traffic count data suggests a value of 43%. It is likely the traffic counter at Marylebone Road categorizes other non-rigid or articulated HGVs as heavy vehicles.

Figure 6.111 Comparison between measured and predicted coarse particles at Marylebone Road (May 1998 to December 2002).

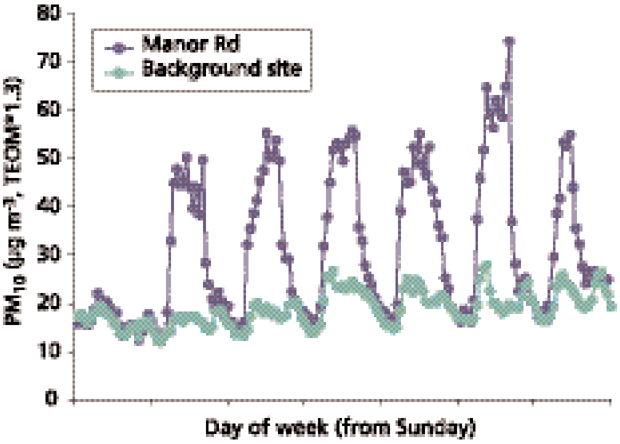


652. The results above can be used to estimate the emissions factors in g km^{-1} for coarse particles from light and heavy vehicles. Emission rates of NO_x have already been calculated for light and heavy vehicles using Marylebone Road (see the AQEG NO_2 report). A multiple regression of light and heavy vehicles with NO_x suggests that over the period covered by the dataset, heavy vehicles were responsible for ~150 µg m^{-3}, for a mean emission rate of 9.52 g km^{-1}. Using these data with the results from the coarse fraction multiple regression it is possible to estimate the emission rates for light and heavy vehicles for coarse particles. These results suggest a mean emission rate for heavy vehicles of 0.415 g km^{-1} and 0.028 g km^{-1} for light vehicles. The mean rate for the mix of vehicles using Marylebone Road is 0.067 g km^{-1}. The coarse particle emission rates for light and heavy vehicles are very similar in magnitude to the estimated exhaust emissions, suggesting that at this location particle resuspension and exhaust emissions are similar in magnitude.

6.4.7 Particle resuspension at the Manor Road site, London

653. Manor Road lies in Erith in the north of the London Borough of Bexley. Manor Road provides access to an industrial area with vehicles having to pass through a residential section of the road. The roadside TEOM situated on Manor Road consistently exceeds the 2005 EU limit value. The TEOM has measured between 78 and 131 daily means above 50 µg m^3 (TEOM * 1.3) per year since installation in 1999. Consequently, Manor Road has been declared an AQMA. Hourly mean PM_{10} at the site shows a distinct diurnal and day of week elevation above background, with Manor Road PM_{10} being elevated during working hours on weekdays and on Saturday mornings. This is shown in Figure 6.112.

Figure 6.112 Diurnal and day of week mean PM_{10} at Manor Road, compared to background, during July to October 2001.



654. Analysis of the measurements of elevated PM_{10} and NO_x above background suggests that the elevated PM_{10} is due to traffic using Manor Road. The effective PM_{10}:NO_x ratio at the site is over ten-times higher than the average factor at other sites in London. Manor Road is often visibly silted from material largely carried onto the roadway from a waste transfer site and vehicles using Manor Road often carry open and dusty loads (Fuller *et al.*, 2001; Bexley Council, 2003). It is likely that resuspended PM_{10} and perhaps dust lifted directly from vehicles accounts for the disproportionately high PM_{10} at this location. Similar elevated PM_{10} levels have been measured at the Hastings 1 site in East Sussex where over 50 daily means >50 µg m^3 per year were recorded during 2002 and 2003. Again visible road silting is also present near the site, arising from access to a nearby landfill. Although further investigation is required, it is likely that similarly elevated local PM_{10} may arise at other locations with substantial road silting. Such sites may include other landfills and waste transfer sites and also large construction sites and exits from unmade roads.

655. Examination of the elemental composition of individual particles sampled from roadside air at Manor Road and sampled from the surface of Manor Road (Jansz *et al.*, 2004) confirms that the airborne particles are similar to the material on the road during times when the roadside PM_{10} concentration is elevated. In wind directions where roadside PM_{10} concentrations at Manor Road are similar to those at nearby background sites, the airborne particle composition is different to that on the road surface.

656. Deposit gauge measurements at varying distances from the road and the waste transfer station confirm the raised levels of airborne particulate matter are associated with the road and not airborne dispersion of fugitive emissions directly from the waste transfer site.

657. Detailed examination of the time series shown in Figure 6.112 potentially provides some indication that the non-exhaust emissions are raised from the road surface by all vehicles, not just from the soiled wheels or spilling of load of the vehicles exiting the waste transfer station. The daily opening and closing hours of the waste transfer station and the different diurnal and weekly variability of total traffic flow on Manor Road allows the respective contributions of the two parts of the vehicle fleet to be separated using modelling, including analysis of the temporal variability of the sediment loading on the road surface. This modelling is currently in progress (Jansz *et al*., 2004).

6.4.8 Contribution to ambient particle concentrations from power stations

658. The power generators as part of their Joint Environmental Programme (JEP) undertake routine, continuous monitoring at a number of rural sites in the vicinity of coal-fired and oil-fired power stations. These sites provide a valuable supplement to the limited number of monitoring sites in the national network. PM_{10} and $PM_{2.5}$ monitoring results using TEOMs have been summarised in a report (Webb *et al*., 2001). This report has been updated by a more recent report on monitoring in 2001 (Webb *et al*., 2002). There is one site at Cliffe in the vicinity of Yorkshire power stations, two monitoring sites at Bottesford and Thorney in the Trent valley and two sites at Hall Farm and Wingham in the Thames Valley. Given that major industrial sources, whose emissions are regulated by the Environment Agency, are a small fraction of total national primary PM_{10} emissions (~10%) and that releases from tall chimneys are likely to be diluted near the ground, industrial sources are generally not thought to make a large contribution to primary particulate matter concentrations at ground level. Using co-located SO_2 measurements as an indicator of the presence of a power station plume, from measurements from the network it was concluded that the primary particulate emissions contributed much less than 10% of the daily PM_{10} threshold of 50 $\mu g\ m^{-3}$ and ~0.25 $\mu g\ m^{-3}$ to the annual mean PM_{10} concentration.

659. Power stations and other industrial processes can also make a contribution to PM through the conversion of gaseous releases of SO_2 and NO_x to secondary particulate. Studies of the formation of secondary particles in plumes containing high concentrations of acid gases have been reviewed (Griffiths and Hill, 2000; Wright and Cocks, 2001). The particles would initially be in the ultrafine range and combine together with other particles to form fine agglomeration mode particles. To supplement the reviews and theoretical understanding, field studies have been attempted. One set of measurements of ultrafine particulates in power station and urban plumes used a differential mobility particle sizer (DMPS) during three episodes (Griffiths *et al*., 2002). Power station source attribution is difficult; using SO_2 and NO_x as plume markers and considering correlation coefficients, estimates of the contribution of a power station plume to particle numbers at ground level were made. In another study two particle size analysers (APS and ELPI) were installed near Didcot Power Station and operated for one month (Brooke and Wright, 2002). The particle size distributions from the two instruments were rather different. In conclusion, there is no clear evidence for rapid particle formation in the vicinity (out to some 100 km) of coal- and oil-fired power stations under typical British atmospheric conditions.

6.4.9 Industrial sources: Port Talbot case study

660. There are few industrial sources that give rise to significant PM_{10} contributions. One that has been recognised from the early days of PM_{10} monitoring is the Corus steel works at Port Talbot. A monitoring site was established in 1997 in the town, which lies just to the east-northeast of the works. Frequent exceedences of 50 µg m^{-3} as a 24-h concentration (TEOM * 1.3) were measured, which are comparable to the London Bloomsbury urban centre site (Table 6.20). The results for this site illustrate the difficulties associated with the default TEOM scaling factor. A KFG reference gravimetric sampler was subsequently operated alongside the TEOM analyser at this site for three years, showing a relationship which was close to unity. The TEOM results for Port Talbot are, therefore, shown in Table 6.20 as unscaled as well as scaled to the default 1.3 factor. This factor of 1.0 at Port Talbot is consistent with a dominant contribution of non-volatile particles being emitted from the steel works. With this factor, there would be no exceedences of the objective.

Table 6.20 PM_{10} concentrations in Port Talbot and the urban centre site at London Bloomsbury during 1997 and 2003.

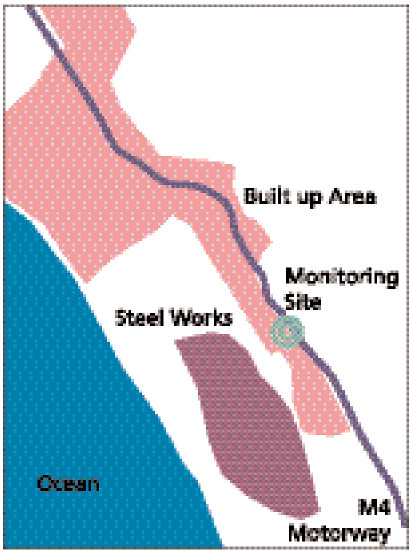
	1997	1998	1999	2000	2001	2002	2003
Exceedences of 50 µg m^3							
Port Talbot days >50 µg m^3 (TEOM)	22	28	26	27	18	5	16
Port Talbot days >50 µg m^3 (TEOM * 1.3)	59	59	62	56	39	24	43
London Bloomsbury days >50 µg m^3 (TEOM * 1.3)	43	20	20	11	17	(21)[a]	(14)[a]
Annual mean							
Port Talbot annual mean (TEOM)	27	27	26	26	23	21	24
Port Talbot annual mean (TEOM 1.3)	35	35	34	33	30	28	32
London Bloomsbury annual mean (TEOM 1.3)	35	30	28	28	29	n/a[a]	n/a[a]

[a] There was low data capture for 2002 and 2003 at this site.

661. On the basis of the monitoring carried out near the steel works, and using the default 1.3 scaling factor for the TEOM, Neath Port Talbot County Borough Council declared an AQMA covering the residential areas to the north and east of the steel works. In order to inform its Review and Assessment work and subsequent Action Plan, the Council commissioned various studies to identify the sources of the high PM_{10} concentrations (Neath Port Talbot CBC, 2000). The monitoring site lies 75 m from the M4 motorway and 700 m from the steel works, both these sources being in opposite directions (Figure 6.113).
It was, therefore, necessary to establish the relative roles of the motorway and the steel works and within the steel works to identify the relative roles

of the different processes and/or fugitive PM from the various stockpiles on site. The first investigations involved analysis against wind direction (Figure 6.114). This showed that elevated PM_{10} concentrations, above 50 µg m^3 (TEOM * 1.3) were associated with winds from the southeast through to west-northwest and, in particular, with southwest and west-southwest winds. The higher CO concentrations were associated in particular with the peak PM_{10} directions of southwest and west-southwest winds. This was consistent with the steel works being the most important contributor to elevated PM_{10}, with the sources widely distributed across the site, although with one or more major sources to the southwest and west-southwest of the monitoring site, which were likely to be combustion related given the associated carbon monoxide elevation. The motorway appeared to be influencing the nitrogen dioxide concentrations, perhaps increasing them by some 10–20 µg m^{-3} (assuming a background for non-motorway directions of around 20 µg m^{-3}). This pattern has been seen in each year of monitoring.

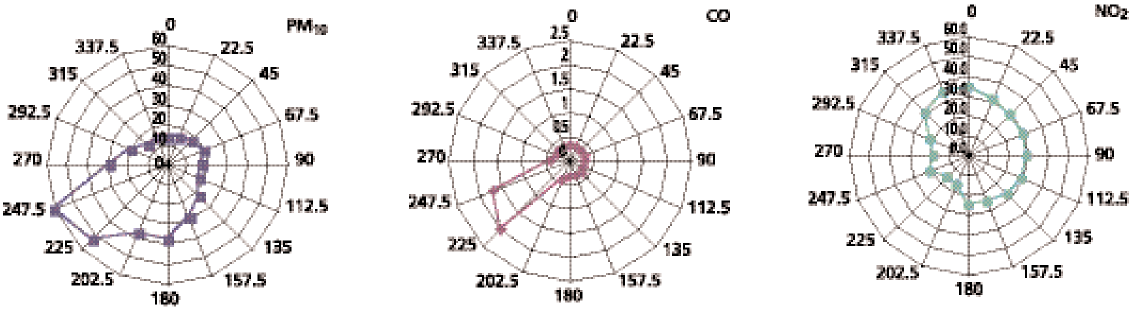
Figure 6.113 Layout of Port Talbot monitoring site in relation to the steel works and the M4 motorway.



662. Samples were also collected using an ACCU sampler attached to the TEOM. This was set up to collect samples on eight different filters according to the wind direction. Samples were then analysed for chemical composition and examined for particle morphology using an analytical transmission electron microscope. The particle examination identified an abundance of spherical particles associated with the winds from the south and southwest. These particles were compared to samples of the emissions from the coke works, the BOS plant, the sinter plant and the blast furnaces. The semi-quantitative chemical analysis showed that iron oxide dominated in the ambient samples. The composition profile and the particle morphology of the ambient samples provided the best match with the blast furnace particles.

663. Confirmation of the role of the steel works processes came from the unfortunate incident that took place on 8 November 2001, when an explosion occurred in furnace 5 (Kent and Willis, 2004). Hourly average concentrations, which had shown frequent peaks of 50 150 µg m^{-3}, were essentially eliminated after the accident (Figure 6.115). Table 6.20 shows that annual average concentrations and

Figure 6.114 PM_{10} (TEOM * 1.3), carbon monoxide and nitrogen dioxide concentrations at Port Talbot AURN site 1989.



24-h exceedences of 50 μg m^{-3} were also much lower in 2002 than 2001. The steel works is though still contributing to PM concentrations measured locally, as discussed below.

664. A directional analysis of 2003 size fractionated monitoring data for the AURN site at Port Talbot provides further insight into industrial concentrations. Daily concentrations of PM_{10} and $PM_{2.5}$ have been measured using a Partisol (gravimetric) sampler, which has also allowed PM_{coarse} concentrations to be calculated. A KFG, reference PM_{10} (gravimetric) sampler, has also been used, and the site is equipped with a TEOM sampler. The directional analysis of the 24-h samplers will inevitably smooth the pollution rose, as the 24-h concentration is applied to each hour of the day. The PM_{10} (TEOM) pollution rose shows a pattern similar to that previously shown (Figure 6.116), with emphasis on winds from the southwest; however it is notable that the PM_{10} (gravimetric) pollution rose measured with the KFG sampler (Figure 6.117) shows a much more even distribution, further illustrating the problems with the TEOM. The problem arises because of loss of volatiles from the PM coming from the northeast, broadly the direction of the motorway on the basis of the nitrogen dioxide pollution rose (Figure 6.114), but also the direction for imported air from England and continental Europe. The TEOM is under-reading by ~40% for the winds from the northeast, whereas from the southwest there is a virtual 1:1 agreement. The latter is consistent with mineral particles from the steel works.

665. The $PM_{2.5}$ and PM_{coarse} results from the Partisol sampler (Figure 6.118) show that the particles from the east are dominated by $PM_{2.5}$, whereas from the southwest the PM_{coarse} particles dominate. This would suggest that the steel works is predominantly contributing coarse PM. This is a significant observation if the air quality criterion is to be changed from PM_{10} to $PM_{2.5}$, as it shows that less attention would be paid to industrial PM emissions from a steelworks.

666. The emerging technique of single particle aerosol mass spectrometry is yielding useful insights into the sources and behaviour of airborne particles. Two complementary mass spectrometry techniques have been deployed for the characterisation of aerosols in the UK atmosphere. One instrument, referred to as an Aerosol Time-of-Flight Mass Spectrometer (ATOFMS) provides real time size and mass spectral information from which chemical composition can be inferred on individual atmospheric particles. The other technique using an instrument referred to as an Aerosol Mass Spectrometer (AMS) provides real-time size resolved information on the concentrations of specific non-refractory components of airborne particles.

Figure 6.115 Hourly average PM_{10} (TEOM) concentrations from August 2001 to January 2002 at Port Talbot. (An explosion took place on 8 November 2001 at the steel works.)

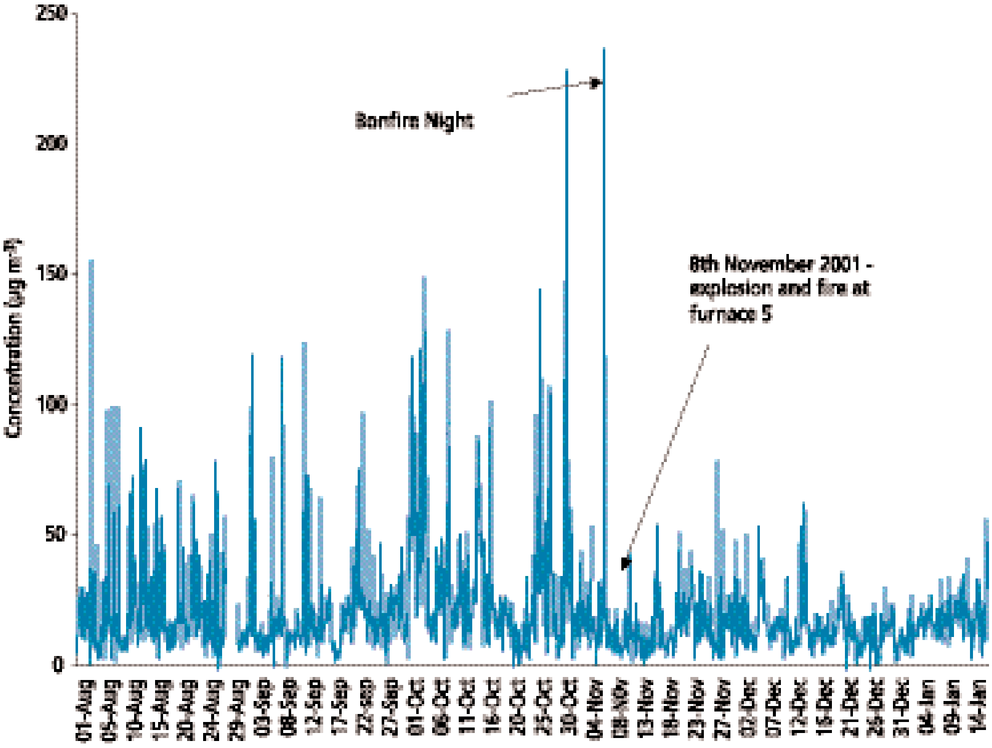
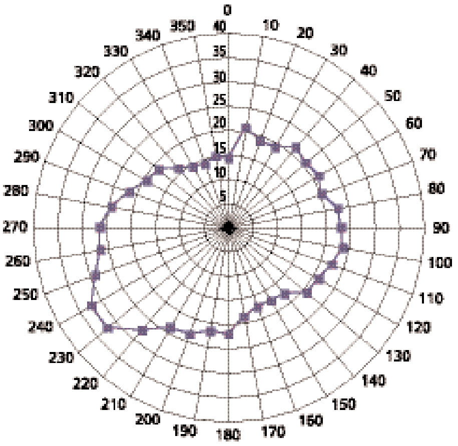


Figure 6.116 PM_{10} (TEOM) at Port Talbot in 2003, by wind direction.



667. Beddows *et al*. (2004) deployed an ATOFMS instrument at Eskdalemuir, a rural location in southern Scotland. A total of 51,980 particles in the size range 0.3–7.4 µm were detected. The individual particle mass spectra were analysed for the presence of individual elements and of ion fragments characteristic of specific molecular composition. A wide range of chemical species was identified with an abrupt change in composition coincident with a transition from a continental to a maritime airmass. Whilst the technique has great potential for application to source apportionment of particles in polluted atmospheres, there are major issues of data interpretation to be addressed before fully quantitative data are obtainable.

Figure 6.117 PM_{10} (gravimentric, KFG) at Port Talbot in 2003, by wind direction.

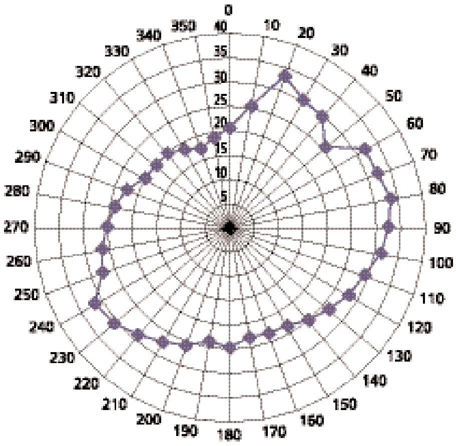
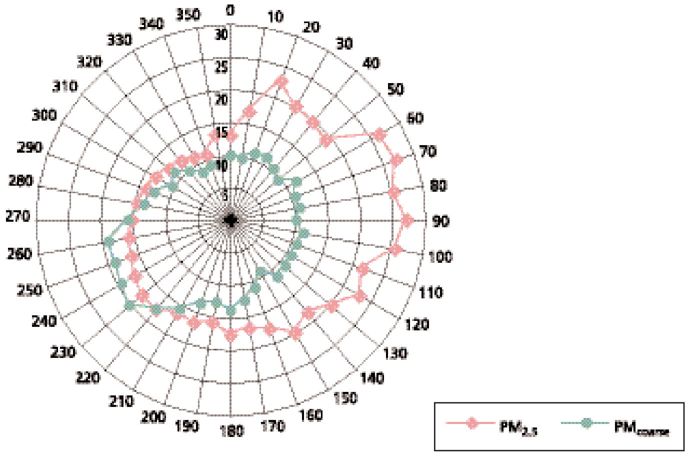


Figure 6.118 $PM_{2.5}$ and PM_{coarse} (Partisol) at Port Talbot in 2003, by wind direction.



668. The AMS instrument has been deployed by Allan *et al*. (2003) for the measurement of fine particle chemical composition in Edinburgh and Manchester. The results from the AMS are more readily quantifiable than those from the ATOFMS, but the instrument is limited to measuring a smaller number of chemical components. The most powerful application of the instrument is to the high time resolution measurement of the size distributions of chemical components such as sulphate, nitrate and carbon. By careful interpretation of the mass spectral data, the instrument allows differentiation between primary and secondary organic carbon components of airborne particles. Allan *et al*. (2003) reported that the observed nitrate, sulphate and organic activity appeared to be confined to two modes during all campaigns. The lower diameter mode had a mass peak at an aerodynamic diameter of 100–200 nm and mainly consisted of organic chemicals with little oxidation which appeared to originate from motor vehicle emissions. The larger diameter mode appeared at various diameters between 300–800 nm and contain nitrate, sulphate and organic chemicals with a notable degree of oxidation, as would be expected from secondary aerosol.

6.5 Particle number: concentrations and size distributions

6.5.1 Particle number concentrations

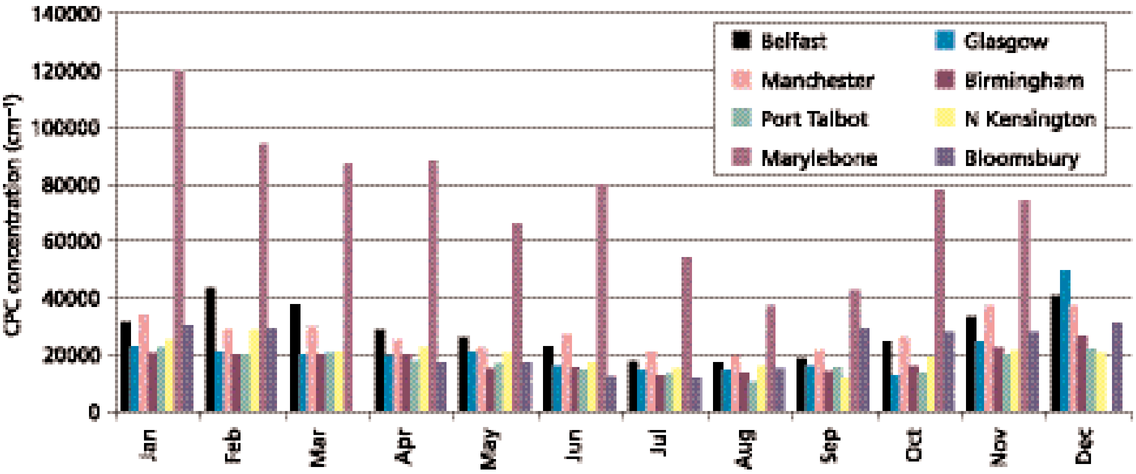
669. Particle number concentrations have been measured using TSI model 3022A Condensation Particle Counters at eight sites. This instrument measures the number of particles in the size range of ~7 to 2000 nm. Data are collated over 15-min periods. Seven instruments were progressively introduced from early 2000 with one instrument being located at Marylebone Road up until March 2002 and then transferred to Bloomsbury. The sites used, their type and location, and the periods of instrument operation are presented in Table 6.21.

Table 6.21 Operating sites of CPC instruments.

Site	Type of site	OS location	Operating period
Belfast centre	Urban centre	J 339744	Jan 2000 – present
Glasgow centre	Urban centre	NS 589650	Feb 2001 – present
Manchester Piccadilly	Urban centre	SJ 843983	Jun 2000 – present
Birmingham centre	Urban centre	SP 064868	Feb 2000 – present
Port Talbot	Urban background	SS 780882	Jan 2000 – present
North Kensington	Urban background	TQ 240817	Apr 2000 – present
Marylebone Road	Kerbside	TQ 281820	Jan 2001 – Mar 2002
London Bloomsbury	Urban centre	TQ 302820	Apr 2002 – present

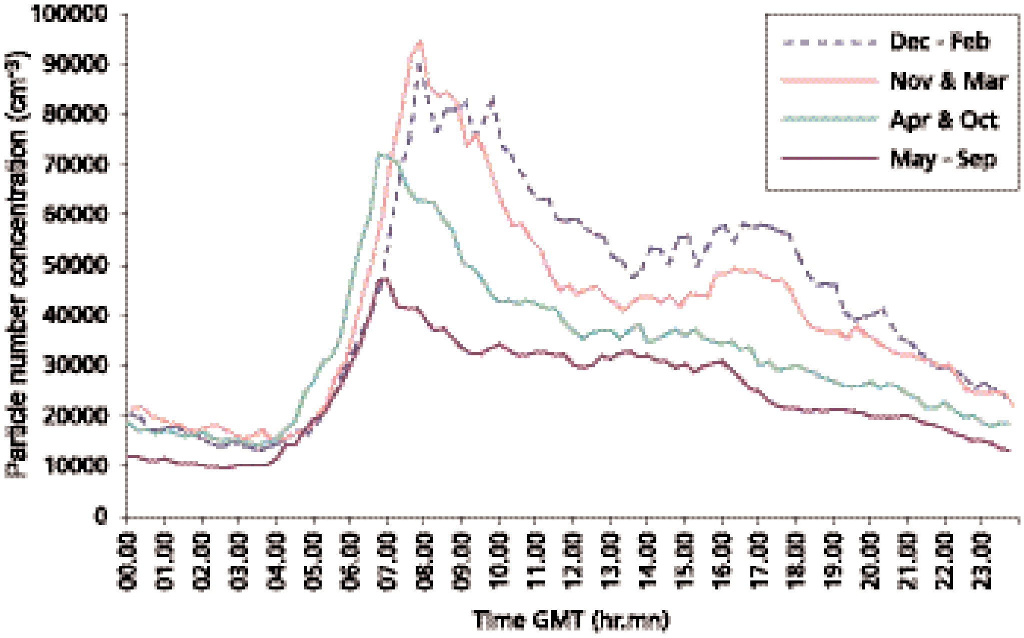
670. The monthly means of particle number concentration at each of the sites are shown in Figure 6.119. No data were collected at the Marylebone Road or Bloomsbury sites during the months of December and March, respectively, due to operational reasons. The data for all sites show the same general trend with minimum concentrations occurring during the summer months and higher concentrations during the winter. Concentrations measured at the Marylebone Road (kerbside) site are substantially greater that those measured at the other sites.

Figure 6.119 Monthly mean particle number concentration at eight sites 2000–2003.



671. Diurnal profiles of particle number concentration were similar to those of particle mass measured in urban areas with peak concentrations occurring around the morning commuting period and some evidence of a secondary peak during the evening commuting period. Diurnal profiles of particle number concentration obtained during weekdays (Monday to Friday) in Belfast during 2002 are shown in Figure 6.120. The data are divided into four groups consisting of the months before and after the changes of clock time on, respectively, the 31 March and 27 October (March and November – including 28 to 31 October); the months after and before these changes of clock time (April and October – excluding 28 to 31 October); the winter months; and the remaining spring and summer months. The 15-min time base of the data makes it possible to distinguish the earlier peak in concentrations at 06:45 GMT, rather than 07:45 GMT, when clock times are brought forward during the summer. The summer trend of lower particulate number concentrations seen in Figure 6.119 is also evident in Figure 6.120. Little evidence of a morning peak in concentration was seen in data collected on Sundays.

Figure 6.120 Diurnal weekday CPC profiles, Belfast 2002.



672. Particle number concentration data (CPC) and concurrently measured PM_{10} and NO_X data collected during 2000–2002 were separated by wind direction data provided by the British Atmospheric Data Centre for a nearby meteorological station. To allow comparison between the different pollutants, the normalised mean concentration for each wind direction was plotted against wind direction. The results for Birmingham are shown in Figure 6.121: the inner circle represents the mean concentration over all directions and the outer circle represents twice the mean. Meteorological data for Birmingham are from Elmdon (Birmingham Airport),12 km to the east of the Birmingham centre air quality measurement site. Figure 6.121 shows a strong peak at 110° from the measurement site with the normalised concentration of particle number at this direction increasing by a greater amount than the increase in normalised PM_{10} concentration, but by a lesser amount than the increases in the normalised NO_X concentration. There may also be some increase in the normalised concentrations in the 30°–40° and 130–140° directions. The major peak at 110° may be associated with local sources in the city centre or with longer range sources.

Figure 6.121 Birmingham normalised concentration versus wind direction.

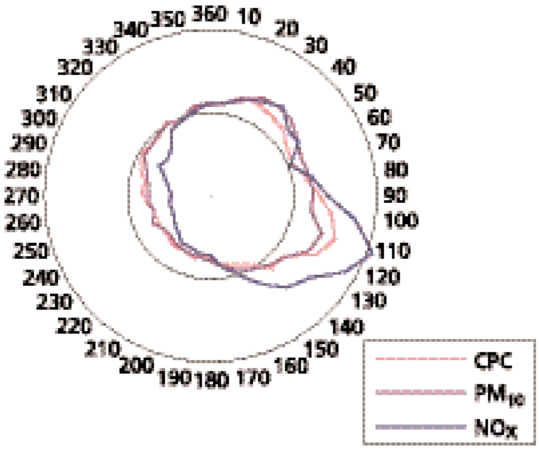


Figure 6.122 Port Talbot normalised concentration versus wind direction.

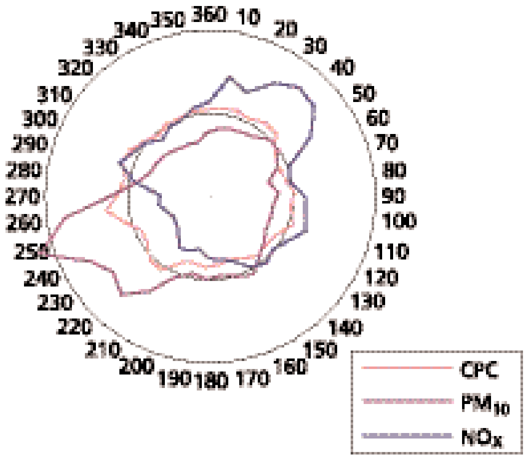
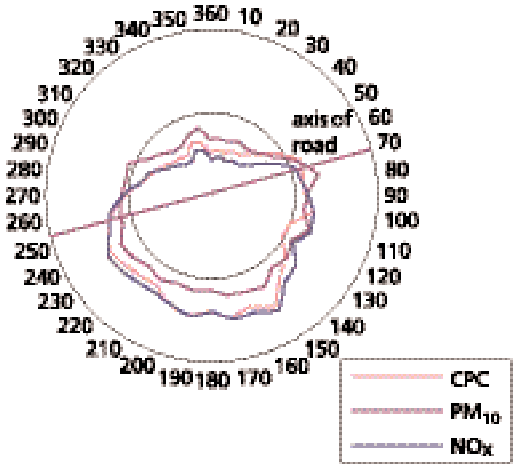


Figure 6.123 Marylebone Road normalised concentration v wind direction.



673. The normalised particle number (CPC), PM_{10} and NO_X concentration data for Port Talbot are plotted against wind direction in Figure 6.122. In this case meteorological data are from Mumbles Head, 15 km to the west. There is little variation in the normalised particle number concentration unlike the raised levels of NO_X when the wind is from the northeast or the raised levels of PM_{10} when the wind is from the west-southwest. The raised levels of NO_X can be associated with the M4 motorway carrying 50,000 to 55,000 vehicles on a typical weekday, 75 m away to the northeast. The raised levels of PM_{10} can be associated with a large steel works that is ~700 m away to the west and south and is a likely source of larger fugitive dust, which will affect the measure of PM_{10} more than it does particle number (see also Section 6.4.9).

674. Figure 6.123 shows the normalised particle number (CPC), PM_{10} and NO_X concentrations measured at Marylebone Road plotted against wind direction. Wind data are from Heathrow Airport 23 km to the west-southwest of the site. The Marylebone Road sampling site is on the southern kerb of the road, which is aligned on an axis of 75°–255°, within a street canyon, with buildings of a height similar to the distance between buildings on either side of the street. The normalised concentrations of all pollutants are higher when the wind is from southerly directions under which conditions the airflow crossing the top of the street canyon can be expected to induce an opposite flow at street level. Two patterns of normalised pollutant distribution are evident in Figure 6.123. The normalised PM_{10} concentration is similar at all directions when the wind direction is from the south side of the street. The normalised concentrations of particle number and NO_X show higher concentrations when the wind is between 150–250° than when the wind direction is between 80–140°. These two patterns of distribution are apparent for other pollutants, with carbon monoxide and elemental carbon following the NO_X distribution and $PM_{2.5}$ and organic carbon following the PM_{10} distribution. There is a major road junction to the west of the sampling site and it may be that stationary traffic queuing to pass through this junction produces greater quantities of certain pollutants.

6.5.2 Particle number size distributions at a rural (Harwell), urban (Bloomsbury) and roadside (Marylebone Road) sites

675. Datasets from February and March 2002 for Harwell, Bloomsbury and Marylebone Road have been analysed and appear in Figures 6.124–6.126.

Figure 6.124 Median particle number size distributions for the period February to March 2002 at the three sites.

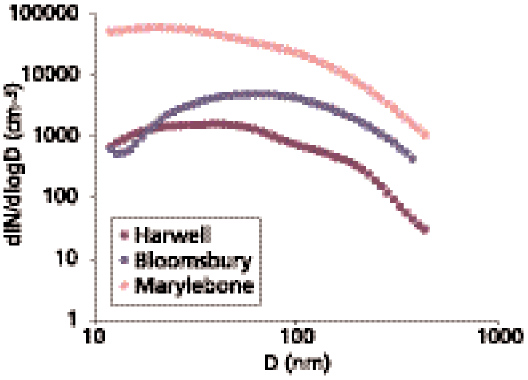
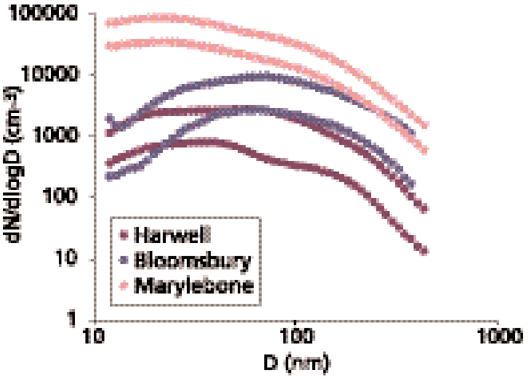


Figure 6.125 The 25 and 75 percentiles of particle number size distributions for the period February to March 2002 at the three sites.



676. The median number size distributions measured at the three sites are so different that the better scale to represent ∂N/∂logD is a logarithmic scale (Figure 6.124). As expected, particle numbers are higher at the Marylebone Road site, strongly influenced by on-road emissions that have a large impact on particle numbers and higher at the urban background of Bloomsbury than Harwell.

677. The difference between numbers of particles below 100 nm is larger between the roadside site and the urban background than between the urban background and the rural site. Additionally, median normalised particle counts of particles below 20 nm at Harwell are higher than those measured at Bloomsbury. This suggests that dilution of urban vehicular sources is not the only reason for differences between sites. There are actually two explanations for this. Firstly, freshly emitted particles below 100 nm have a short lifetime in the atmosphere due to coagulation and other processes. Their number rapidly drops with the distance from the road. New particle formation at Harwell is occasionally responsible for high concentrations of very small particles (see Section 6.5.5).

678. In comparison, for particles above 300 nm the difference between the roadside site and the urban background is smaller than the difference between the urban background and the rural site.

679. Marylebone Road shows considerably higher concentrations than the other sites. The difference is larger for particles below 100 nm and very large for particles below 40 nm. The difference reaches a factor of about 100 for particles below 20 nm. This confirms that on-road vehicle emissions are a very important source of particles and especially of very small particles.

680. The difference between median particle number size distributions at Bloomsbury and Harwell is quite constant for particles above 70 nm; but is weaker and weaker for particles below 70 nm, and median normalised numbers of particles below 20 nm are higher at Harwell than at Bloomsbury. This is probably the consequence of new particle formation at Harwell responsible for occasional bursts of small particles while new particle formation at Bloomsbury is probably very limited because of the large pre-existing particle surface area that prevents homogenous nucleation.

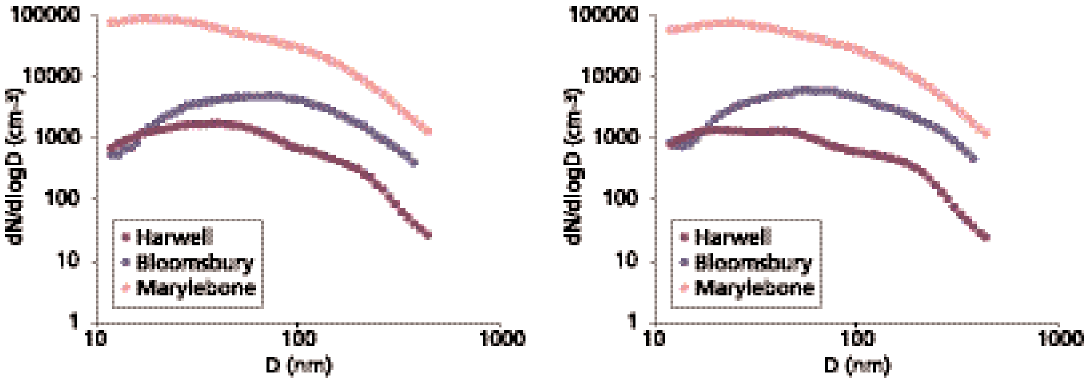
681. Examination of the median particle number size distributions for different periods of the days does not show any considerable difference indicating that the intra-site variability is much lower than the inter-site variability (Figure 6.126).

682. Figure 6.125 presents the 25 and 75 percentiles of the particle number size distributions: 50% of the particle number size distributions measured are between these two series of values. This indicates the spread of particle number size distributions.

683. The 25th percentile of the particle number size distribution at Marylebone Rd and the 75th percentile of the particle number size distribution at Bloomsbury show similar normalised counts of particles above 300 nm. This shows that within the >300 nm size range, the less polluted conditions at Marylebone Road are not dissimilar to the more polluted at Bloomsbury, despite the large differences under average conditions.

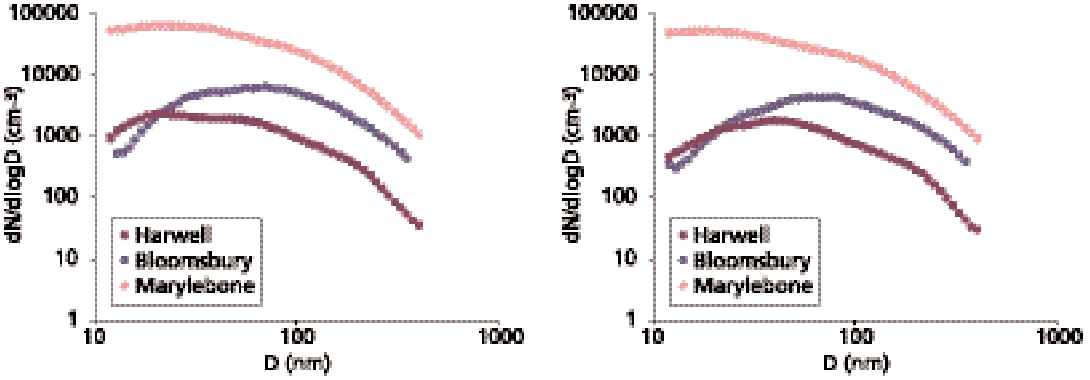
684. The 25th percentile of the particle number size distribution for Bloomsbury and the 75th percentile of the particle number size distribution for Harwell show similar normalised counts of particles above 50 nm, while there is almost a factor 10 between the two sites for particles below 50 nm. This shows that the most polluted conditions at Harwell are similar to the less polluted conditions at Bloomsbury, except for small particles.

Figure 6.126 Median particle number size distributions for the period February to March 2002 at the three sites as a function of the time of the day.



(a) Mornings (05:00–10:00 h)

(b) Midday period (11:00–16:00 h)



(c) Evenings (17:00–21:00 h)

(d) Nights (22:00–04:00 h)

6.5.3 Particle formation from vehicle emissions at Marylebone Road

685. In contrast to the other particle size ranges and to the particle mass ($PM_{2.5}$, PM_{coarse}), the numbers of the smallest particles (below about 30 nm) are not strongly correlated with the traffic intensity. The highest concentrations of the smallest particles occur in the morning of the weekdays and during the Friday to Saturday and Saturday to Sunday nights of the weekend (Figures 6.127 and 6.128). This corresponds to the beginning of the morning rush hour of weekdays and, at the weekend, to intense leisure traffic during the night. Interestingly, during the weekdays, the decrease of the particle number from the morning rush hour to the afternoon rush hour is associated to a simultaneous shift of the mode of the particle size distribution from ~23 nm to ~31 nm as well as an increase of the mean diameter of ultrafine particles (11–100 nm) from 35–39 nm.

686. Another particular feature of these very small particles is their unusual behaviour with meteorological parameters. Wind speed does not seem to influence their concentrations, whereas an increase of wind speed significantly reduces the concentrations of the other particle fractions. Also worthy of note, the wind speed generates a shift of the size distribution to finer diameters and an increase of the relative number (Figure 6.129). Rain events are associated with higher particle numbers and drought periods are associated with much lower particle numbers. In addition, particle mass increases with the duration of the drought period due to accumulation in the atmosphere in the absence of washout.

Figure 6.127 Weekly median cycles of integrated particle numbers from 11 to 30 nm, particle surface area, heavy duty traffic and light duty traffic.

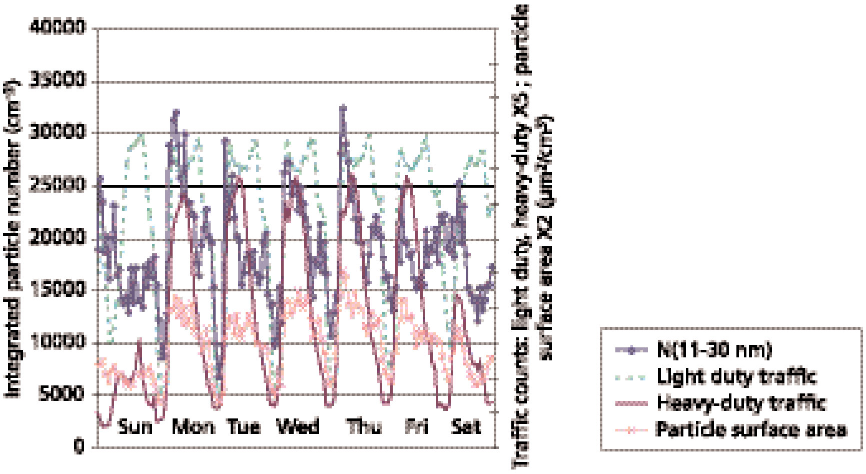


Figure 6.128 Weekly median cycles of integrated particle numbers from 30 to 100 nm and from 100 to 450 nm, heavy duty traffic and light duty traffic.

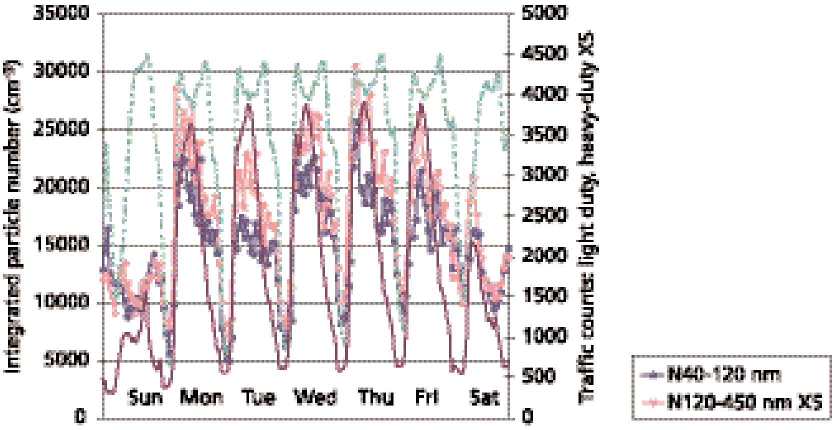
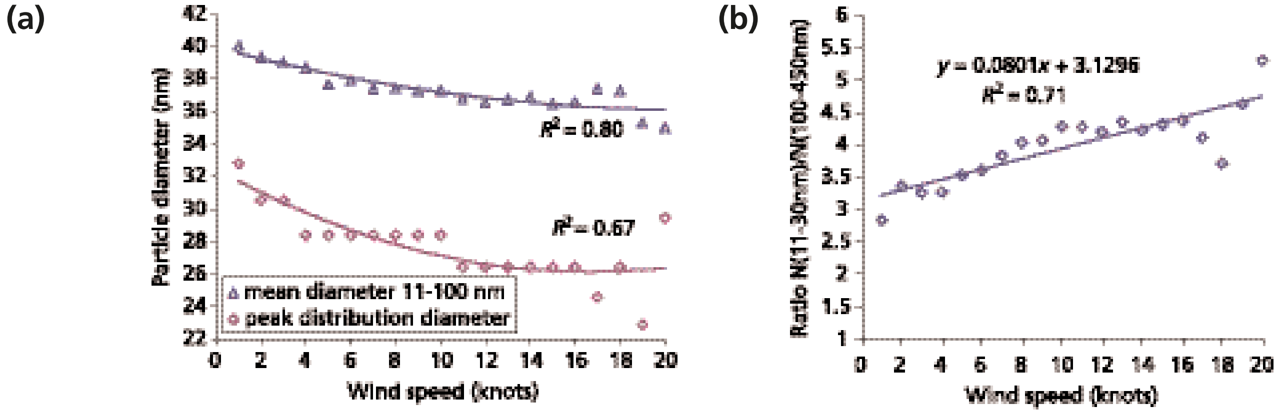


Figure 6.129 Relation between wind speed and (a) the particle diameters (b) the ratio of integrated particle counts from 11 to 30 nm to integrated particle counts from 100 to 450 nm.



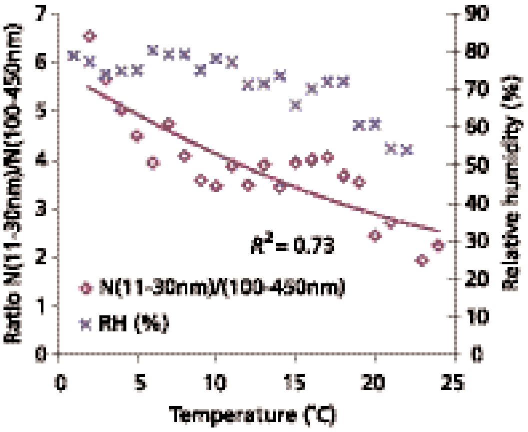
687. These observations clearly indicate that some factors favour the occurrence of the small particles in the early morning and during the night. They also suggest that 'cleaner atmospheres' created by stronger winds and rain events favour the occurrence of high numbers of ultrafine particles. This suggests that these particles are not directly emitted by motor vehicles but formed during the cooling and dilution of vehicle exhaust in the atmosphere of Marylebone Road. On the other hand, the evolution of the particle diameter and the lower number of ultrafine particles during the midday and the afternoon suggests that the high on-road emissions lead mainly to the growth of particles via condensation processes that would be more favourable than new particle formation during these parts of the day. The possible coagulation of the small particles formed in the early morning is also in agreement with the evolution of both particle diameters and numbers.

688. After dilution in the atmosphere, the exhaust gases are cooled and diluted; the saturation ratio of gaseous compounds of low volatility reaches a maximum. Then, two major processes are possible: firstly, the nucleation and formation of new particles, and secondly, condensation onto existing particles. The occurrence of these two processes will depend on the particle surface area available for the condensation of the semi-volatile species. Small pre-existing aerosol concentrations favour both the production of new particles and their growth to detectable sizes in the atmosphere (Kulmala *et al*., 2000). In contrast, high concentrations of pre-existing particles both promote the condensation of the semi-volatile vapours and disfavour the growth of fresh nuclei and their survival from high coagulational scavenging (Kerminen *et al*., 2001). The large amounts of semi-volatile vapours from vehicle exhausts in the early morning, associated with low pre-existing particle surface area (~300–500 $\mu m^2 cm^{-3}$), would favour both production of new particles and their growth to detectable sizes (>11 nm). During daytime, when the particle surface area ranges from 800 to 1100 $\mu m^2 cm^{-3}$, condensation of the emitted condensable gases onto existing particles is likely to be the most favourable process.

689. The mixing of two air parcels with different temperature and different relative humidities increases nucleation rates significantly (Nilsson and Kulmala, 1998) and this is comparable with the mixing of hot exhaust vapours with cool air. More efficient mixing (by a stronger wind) with colder air is, therefore, expected to improve nucleation rates. More efficient mixing by stronger winds would also lessen the possibility of particle coagulation and new particle self-coagulation, which would in turn have an impact on their diameters.

690. Some meteorological parameters such as low temperatures and high relative humidity favour the formation of new particles (Easter and Peters, 1994). In the morning and during the night the temperature is lower and the relative humidity higher. The relative number of particles ranging from 11 to 30 nm measured during the morning rush hour is strongly influenced by the temperature (Figure 6.130). Lower temperature favours the relative number of small particles (ratio to the other size ranges). This strong dependence on the temperature is further support for the idea that these particles are not a primary emission but are formed during the cooling and dilution of the vehicle exhaust. In contrast, no obvious relation has been found with relative humidity despite the diurnal anti-correlation of temperature and relative humidity. Higher water contents in the atmosphere are expected to favour homogeneous binary nucleation of sulphuric acid and water (Easter and Peters, 1994), whereas ternary nucleation involving ammonia is expected to be independent of relative humidity (Korhonen *et al*., 1999), and the possible nucleation from organic compounds might not be influenced by relative humidity.

Figure 6.130 Relation between the temperature and the ratio of integrated particle counts from 11 to 30 nm to integrated particle counts from 100 to 450 nm and the relation between the relative humidity and the temperature.

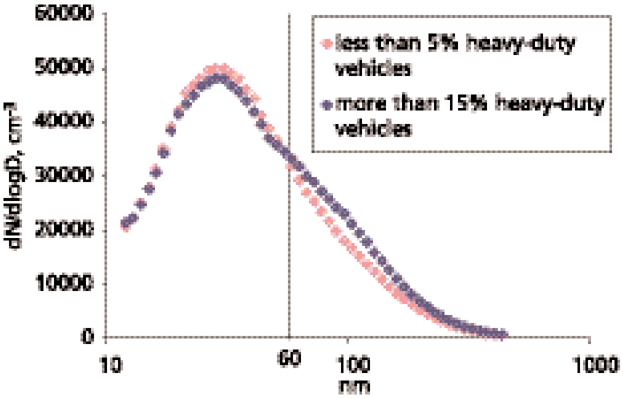


6.5.4 NanoSMPS data at Marylebone Road

691. Measurement data from 25 September 2003 to 18 November 2003, comprising 7,767 10-min particle number size distributions ranging from 5 to 184 nm, have been analysed using a nanoSMPS, which measures smaller particle sizes than in the earlier work.

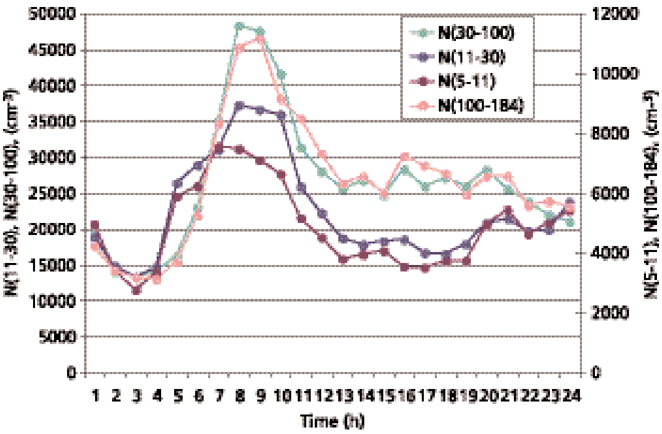
692. The peaks in the size distributions range from 20 nm to 30 nm, which is in good agreement with the previous work and other roadside studies. Another mode around 60–80 nm seems to be hidden by the largest mode around 25 nm. This is apparent in earlier measurements only when the proportion of heavy duty vehicles exceeds 15% (Figure 6.131).

Figure 6.131 The 1998 to 2000 average size distributions from Marylebone Road SMPS data.



693. The pattern observed on Figure 6.132 confirms that particles below 11 nm are from vehicle emissions (the same figures for the weekends are not presented because of the small datasets). The daily cycle of particles ranging from 5 to 11 nm is quite similar to that for particles ranging from 11 to 30 nm. On the other hand, particles ranging from 30 to 100 nm and particles ranging from 100 to 181 nm show similar patterns to one another. Concentrations of the former increase from 03:00 h to reach their maxima at 07:00 to 08:00 h and then decrease from 10:00 to 18:00 h. Then, concentrations of both particles ranging

Figure 6.132 Median daily integrated particle numbers for weekdays.



from 5 to 11 nm and particles ranging from 11 to 30 nm increase slightly to reach a second maximum around 22:00 h. Larger particles (above 30 nm) do not show this second maximum (their concentrations decrease from the morning to the night) and, additionally, a shift between particles below 30 nm and above 30 nm can be seen in the morning.

694. In agreement with earlier work, during weekdays, the decrease of the total particle number from the morning rush hour to the afternoon rush hour is associated with a simultaneous shift of the mode of the particle size distribution from about 21 nm to about 35 nm as well as an increase in the median diameter from 26 nm to 37 nm.

6.5.5 Homogenous nucleation events at Harwell

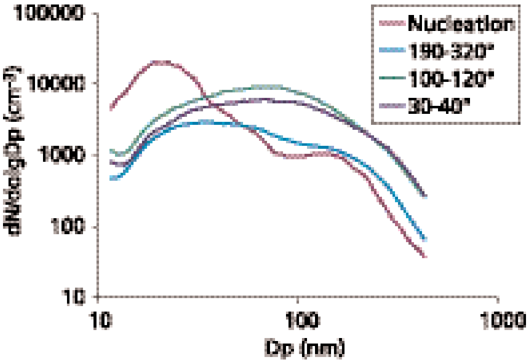
695. The term homogenous nucleation refers to the process of gases condensing to form wholly new particles.

696. High particle number concentrations at Harwell are often associated with plumes from the nearby A34 road (about 100° to 120° direction). Few events involving significant particle numbers that can be attributed to photochemically induced homogenous nucleation events are observed at the Harwell rural site. Observed particle number size distributions suggest that most events may have occurred upwind of the Harwell site. All events occurred during the warm period of the year.

697. Figure 6.133 represents median particle number size distributions corresponding to four different situations at Harwell: (i) possible homogenous nucleation events that occurred in 2000; (ii) winds blowing from 100º–120° directions, that is, downwind of the closest part of the A34 road, 2 km away; (iii) winds blowing from 190º–320° directions, that is, clean air masses; (iv) winds blowing from 30º–40° directions, that is, downwind of Didcot power station (with a possible influence of traffic).

698. The Aitken mode generally dominates at Harwell: the highest numbers are generally measured within the 30–100 nm particle diameter range. Similar median total integrated particle numbers (11–450nm range) are observed for the A34 traffic influence and during (or downwind) events of homogenous nucleation

Figure 6.133 Four median particle number size distributions at Harwell: for homogenous nucleation events (midday hours) and for certain wind directions: 30°– 40° (Didcot power station + possible influence of traffic); 100–120° (the closest parts of the A34 road); 190–320° (clean air masses conditions). Log-log scale.

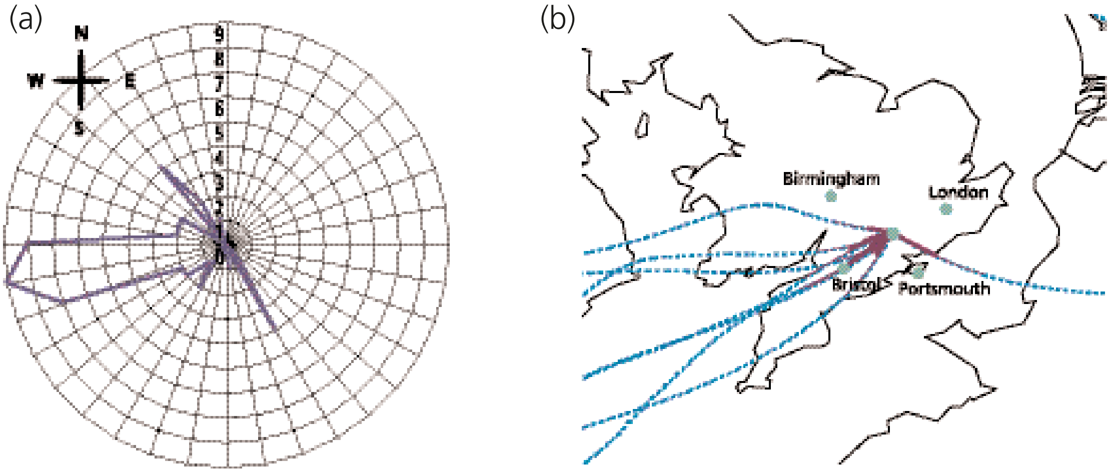


(medians of about 7000 cm^{-3}). However, very different particle number size distributions are measured. The median particle size distribution for homogenous nucleation events shows two modes: one around 20 nm and another one above 100 nm. The median particle number size distributions for anthropogenic influences (from the A34 road and from the Didcot power station) show one single large mode around 70 nm. The one for clean conditions shows a mode around 35 nm and a second mode above 100 nm. In such clean conditions, particle number size distributions were generally bimodal or trimodal.

699. The events of homogeneous nucleation that occurred at Harwell in 2000 have been examined. New particle formation events occurred on cleaner days and especially the most spectacular bursts of particles (days with lower particle surface area, lower concentrations of particles ranging from 100 to 450 nm and lower concentrations of pre-existing particles ranging from 30 to 100 nm). The influence of the particle surface area is consistent with competition between homogeneous nucleation and condensation of vapour onto existing particles. Both the formation and the growth of new particles are favoured by small pre-existing aerosol concentrations, whereas high particle surface areas promote the condensation of the semi-volatile vapours and disfavour the survival of new nuclei from coagulation scavenging (Kulmala *et al*., 2000 ; Kerminen *et al*., 2001). A low particle surface area is possibly not the only influential parameter since high particle surface areas were measured on days of nucleation events at other sites (Harrison *et al*., 2000; Birmili and Wiedensohler, 2000).

700. The examination of wind direction and backward trajectories shows that in 2000, new particle formation often occurred when winds blew from westerly directions, bringing clean air masses from the northern Atlantic Ocean over rural areas, except 18th June (see Figure 6.134). Conversely, easterly winds are generally associated with higher particle numbers at Harwell, in consistent with the location of most of the local anthropogenic sources (the A34 road and London to the east-southeast; Didcot and Didcot power station to the northeast). Table 6.23 shows that the weather is significantly better on days of homogeneous nucleation. Events of homogenous nucleation occurred on days with higher global daily insolation and low cloud cover. In common with the Melpitz site (Eastern Germany; Birmili, 1999), these days had also lower relative humidity, slightly higher temperature and higher winds speeds.

Figure 6.134 (a) Wind direction associated with nucleation events in the year 2000 (in number of hours per direction). (b) Possible pathways of air masses associated with nucleation events and possible areas where nucleation had occurred upwind of Harwell. Backward trajectories end at noon at Harwell (from the British Atmospheric Data Centre).



701. Mode diameters generally observed at Harwell suggest that events of homogenous nucleation had possibly occurred upwind of the Harwell site. Figure 6.134b represents the possible pathways of air masses coming to Harwell on days of nucleation and the possible areas where nucleation had occurred. The possible positions where nucleation had occurred are highlighted on the figures in red (assuming a minimum growth rate of 4 nm/h) and in bold red (assuming a minimum growth rate of 10 nm/h). The length of the highlighted pathways also depends on the wind speed and on mode diameters observed at Harwell and assumes that newly formed particles have grown from about 1 nm initial diameter.

Table 6.23 Median values for nucleation events in the year 2000. Particle surface area (PSA) in $\mu m_2/cm_3$, integrated particle numbers for the 11–30, 30–100, 100–450 nm ranges in cm^{-3}; O_3, SO_2, and NO_X in ppb, wind speed (WS) in knots (0.515 $m\ s^{-1}$), total cloud amount (TCA) in Oktas (0,1,2: fine; 3,4,5: fair; 6,7,8: cloudy), T in °C, RH in %, global irradiation amount (GIA) in $kJ\ m^{-2}$. For the SO_2, zero concentration is assumed below the detection limit (DL) of the instrument.

	PSA	**11-30 nm**	**30-100 nm**	**100-450 nm**	**O_3**	**SO_2**	**NO_X**	**WS**	**TCA**	**T**	**RH**	**GIA**
Nucleation	61	6556	1977	345	39	1	2	11	3: fair	15.6	53	20,300
All data, midday time	103	878	1898	761	32	1	6	9	7: cloudy	14.0	68	10,300
All data	112	716	2118	818	27	< DL	6	7	7: cloudy	11.1	84	10,300

Chapter 7

What are the main trends in particulate matter in the UK?

Key points

- The decline in coal use during the second half of the 20th century has led to declines in the rural concentrations of a wide range of trace elements in suspended PM.

- The phase-out of lead in petrol has exerted a significant influence on particulate lead concentrations throughout the UK from the 1980s onwards. Particulate lead concentrations have declined by more than an order of magnitude over the last 20 years in urban locations.

- Annual mean black smoke concentrations in London appear to have fallen by about a factor of 50 over the 80 years during which black smoke records have been kept, largely as a result of the phase-out of coal burning.

- The average black smoke concentration over the 197 available sites fell from 14.5 µg m^{-3} in 1990 to 7.1 µg m^{-3} in 2002. The annual trend was –0.67 µg m^{-3} per year or –5% per year. Over this same period, black smoke emissions are estimated to have declined by –6.8% per year.

- Black smoke levels declined at the Northern Ireland sites by –1.1 µg m^{-3} per year or –5% per year. The trends would indicate that substantial progress is being made with smoke control and that, if maintained, black smoke levels in Northern Ireland will, within the next 6 years or so, become comparable to those in the rest of the UK.

- Between 1990 and 2002 London sites showed some of the smallest black smoke trends of all the regional groupings: only –0.5 µg m^{-3} per year or –2.8% per year due to a significant contribution to black smoke emissions from diesel road traffic.

- Particulate sulphate levels in rural locations are falling with highly statistically significant annual trends that lie in the range of –4 and –9% per year. Over the period from 1990 to 2002, particulate sulphate levels appear to have fallen by more than a half, with the UK average concentration falling from 1.2 µg S m^{-3} to 0.5 µg S m^{-3}. The observed trends in particulate sulphate levels are significantly smaller than the decline in UK SO_2 emissions over the same period, which averaged about –11% to –12% per year. This may point to an increasing fraction of the emitted SO_2 being oxidised and present in the atmosphere as particulate sulphate.

- Across the UK, 48 PM_{10} monitoring sites have long enough records for trend determination and all but three of the sites show downwards trends in annual mean PM_{10} concentrations, with about half being highly statistically significant. The steep decline in urban PM_{10} levels observed from 1992 to 1999 has given way to a flattening out and a slight increasing trend from 2000 to 2003.

- From 1997 to 2003, network average annual mean PM_{10} concentrations declined from 23.4 µg m^{-3} to 20.5 µg m^{-3}, indicative of a trend of about –0.5 µg m^{-3} (–2%) per year. Some sites in the London area have exhibited trends distinctly smaller than the network average or close to 0.

- The observed downwards trends in PM_{10} reported for rural sites are significantly larger than those trends that could be explained solely by the observed trends in particulate sulphate over the same period. There must, therefore, be significant proportions of primary or other secondary PM in the PM_{10} size range, reaching these rural sites, and these components must also have been showing significant downwards trends.

- There is only one long-running time series of $PM_{2.5}$ measurements in the UK and these are for Hodge Hill, Birmingham. Annual mean $PM_{2.5}$ concentrations over the 7-year period 1995–2002 have shown a statistically significant downwards trend of –0.4 µg m^{-3} (3%) per year. On the basis of the observed trends in particulate sulphate at Stoke Ferry, a downwards trend of –0.4 µg PM_{10} m^{-3} per year (by scaling up to $(NH_4)_2SO_4$ from sulphur) would be anticipated, close to that observed at the Hodge Hill site. This would point to the trends in secondary sulphate particles as the main driving force behind the $PM_{2.5}$ trends at the Hodge Hill site.

- Over the 7-year period, PM_{coarse} levels at the Hodge Hill, Birmingham site have shown a highly statistically significant downwards trend of –0.7 µg m^{-3} (–13%) per year.

- Normalised monthly mean PM_{10} trends within the LAQN have shown a 35% decrease by 2000 relative to 1st January 1996, followed by a period of substantial increase so that by the end of 2003, PM_{10} levels have declined by only 19%. The normalised trends of PM_{10} and NO_x are diverging increasingly because the normalised trends in NO_x are monotonically downwards.

- Particulate cadmium and copper trends during 1990 and 2002 in the multi-element network have shown highly statistically significant downwards trends at Glasgow, London Cromwell Road and Motherwell. Chromium trends have been upwards at London Brent and Glasgow. Trends in manganese have been downwards at five sites, but London Brent has shown no overall trend. Nickel and vanadium have shown downwards trends at all six sites.

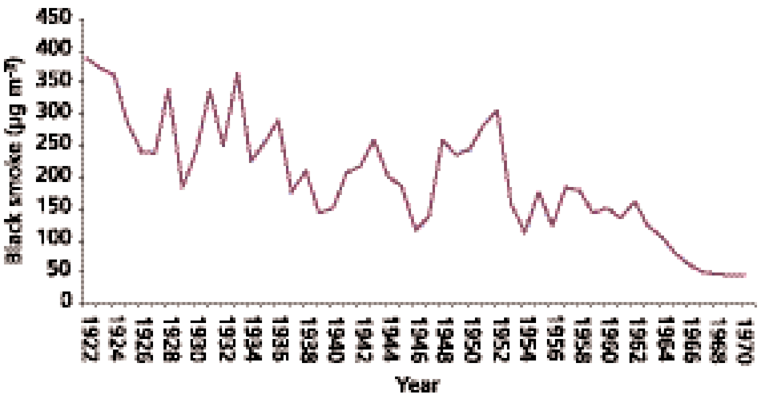
- Particulate iron levels have shown highly statistically significant downwards trends at Motherwell, Leeds and London Cromwell Road from 1990 to 2002. The upwards trend in particulate iron observed at the London Brent site could indicate a steady increase in non-exhaust traffic emissions due to traffic. This could explain why the observed PM_{10} trends at the London Brent and London Marylebone Road were significantly smaller than those observed elsewhere in the UK.

7.1 Historical setting

702. Particulate monitoring began over 80 years ago using the black smoke method and has continued through to the present day, making it the most heavily and continuously monitored of all the urban pollutants. Figure 7.1 presents the time series of black smoke measurements made at the Kew Observatory from 1922 through to the end of the 1960s, when the site closed. Black smoke appears to have been declining throughout the five decades during which these measurements were made, a decline which has been barely influenced by the passage of the Clean Air Act in 1956. The downwards trend must have been influenced by economic factors such as the change of London from a manufacturing to a trading centre, the substitution of coal burning for electricity and town gas and, more recently, the use of oil and natural gas instead of coal. Similar trends in changes to black smoke levels will have been seen in all the major population and industrial centres throughout the UK.

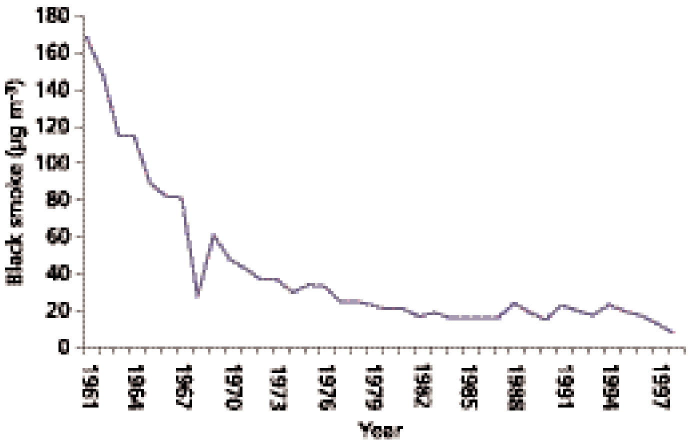
703. Figure 7.2 shows how black smoke concentrations have continued to decline since 1961 through to the present day. Marked downwards trends have been seen during the 1990s in London; these are the subject of a more detailed analysis below. Again, similar trends during the 1960s to 1990s have been reported across the UK. The network of black smoke monitoring sites reached its peak during the 1960s with over 1200 monitoring sites. The network has been shrinking in size during the 1990s but complete records are still available for 197 sites. Although absolute measurements using the black smoke method are of limited validity, they still serve as a useful indicator of long-term trends in suspended PM. Current (2002) annual mean black smoke concentrations in London are now about 6–9 µg m^{-3} compared with historic (1922) levels of about 380 µg m^{-3}, a decrease of about a factor of 50 over the 80 years of the measurement record.

Figure 7.1 Black smoke concentrations at the Kew Observatory, London between 1922 and 1970.



704. The decline in coal use during the second half of the 20th century has influenced not only black smoke levels but has also led to declines in the concentrations of a wide range of trace elements in suspended PM. Table 7.1 shows an example of the trends in iron aerosol at four rural locations during 1972–1981 and 1982–1991 (Cawse *et al*., 1994). Similar trends have been reported for other trace elements at these locations, including particulate antimony, arsenic, caesium,

Figure 7.2 Black smoke concentrations in Lambeth, London between 1961 and 1998.



calcium, cerium, chromium, cobalt, copper, europium, indium, lanthanum, magnesium, manganese, nickel, potassium, samarium, scandium, selenium, silver and zinc.

Table 7.1 Rural concentrations of particulate iron at four locations between 1972 and 1991, showing mean concentrations and annual trends (Cawse *et al*. 1994).

Location	Mean	Concentrations (ng m^{-3})		
	1972–1981	Trend (% change per year)	Mean 1982–1991	Trend (% change per year)
Harwell, Oxfordshire	285	–6.0	172	3.3
Styrrup, Nottinghamshire	682	–11.2	460	–5.2
Trebanos, Glamorgan	372	–9.1	318	NST
Wraymires, Cumbria	242	NST	98	–4.2

NST, no significant trend.

705. In addition to the decline in coal usage, the phase-out of lead in petrol has exerted a significant influence on particulate lead concentrations throughout the UK from the 1980s onwards. Particulate lead concentrations have declined by more than an order of magnitude over the last 20 years in urban and rural locations. Levels have fallen from 770 ng m^{-3} in 1980 at the London Brent site to 22 ng m^{-3} in 2002, whereas at the rural site Chilton, Oxfordshire the fall has been from 110 ng m^{-3} to 8 ng m^{-3}. At the remote rural site, Eskdalemiur in Dumfries and Galloway, levels have fallen from 29 ng m^{-3} to 3 ng m^{-3}.

7.2 Trends during the 1990s and onwards

7.2.1 Black smoke trends

706. Trend analysis has been performed on a dataset of annual mean black smoke concentrations from 1990 to 2002 obtained from the longest running 197 UK sites. Each time series was subjected to two types of statistical analysis. The first tested for the presence of a monotonic increasing or decreasing trend with the nonparametric Mann-Kendall test; the second estimated the slope of a linear trend with the nonparametric Sen's method (Salmi *et al*. 2002). Of the 197 times series analysed, 15 showed upwards trends and 184 showed downwards trends. A total of 66 black smoke sites showed statistically significant trends at the 99.9% level of significance, 56 at the 99% level, 56 at the 95% and 9 at the 90% level, that is, 161 sites showed trends that were highly statistically significant. None of the upwards trends were highly statistically significant.

707. The network average black smoke concentration fell from 14.5 µg m^{-3} in 1990 to 7.1 µg m^{-3} in 2002. The trend was –0.67 µg m^{-3} (–4.8%) per year and this was highly statistically significant at the 99.9% level. Over this same period, according to the NAEI, UK national total black smoke emissions declined at –6.8% per year, a somewhat higher rate compared with the observed trend across the black smoke network.

708. The network average necessarily hides large variations across the UK and so the sites were assembled into regional groups to allow a more detailed analysis of the trends in black smoke levels. Table 7.2 summarises the trends in annual mean black smoke concentrations for the different regions across the UK, ranked in order of their 1990 average concentrations. The regional grouping with the highest annual mean concentrations at the start of the trend period was that encompassing the 38 sites identified as being located in the Yorkshire, Derbyshire, Durham and Lothian coalfields. These sites showed the largest downwards trends of all and almost all were highly statistically significant. These trends averaged –1.55 µg m^{-3}(–7.3%) per year and were large enough to bring this category from the most polluted regional grouping down to fourth most polluted. The phase-out of coal burning in these coalfield communities has clearly been associated with significant downwards trends in black smoke levels. The observed downwards trends agree closely with those reported in UK national total black smoke emissions over this same period in the NAEI.

709. Black smoke levels declined at the Northern Ireland sites by –1.1 µg m^{-3} (–5.3%) per year, see Table 7.2. These sites were ranked the second most polluted regional grouping at the start of the trend period and the most polluted at the end. Two-thirds of the observed trends were highly statistically significant. The trends indicate that substantial progress is being made with smoke control and that, if maintained, black smoke levels in Northern Ireland will, within 6 years or so, become comparable with those in the rest of the UK.

710. The West Midlands sites were the third most polluted regional grouping at the start of the trend period and, despite showing highly statistically significant downwards trends, remained with this ranking at the end of the period. The phase-out of coal combustion has halved black smoke levels, but they still remain significantly above the UK network average, presumably because of contributions from other sources, road transport and industry.

Table 7.2 Summary of the analysis of trends in the annual mean black smoke concentrations monitored in different regions of the UK between 1990 and 2002, showing the annual mean concentrations in 1990, the annual trends and the numbers of sites showing highly statistically significant trends for each region.

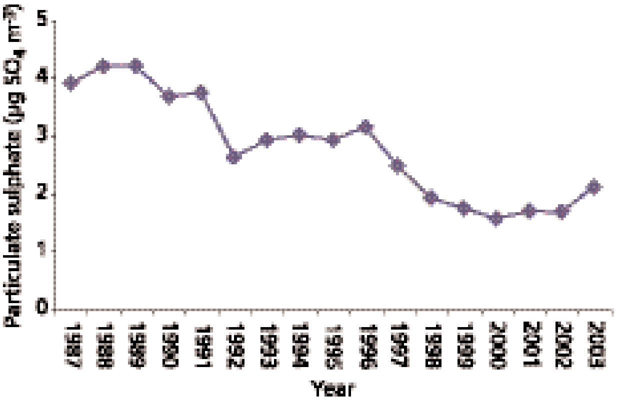
	Annual mean 1990 ($\mu g\ m^{-3}$)	Annual trend 1990–2002		Number of sites				
		($\mu g\ m^{-3}$ per year)	(% per year)	Total	Statistical significance of annual trends			
					$P < 0.001$	$P < 0.01$	$P < 0.05$	$P < 0.1$
Coalfields	21.2	–1.55	–7.3	38	23	11	2	—
Northern Ireland	19.0	–1.10	–5.3	12	4	4	—	—
West Midlands	18.1	–1.13	–5.4	7	2	2	—	1
Northwest	14.5	–0.70	–4.6	35	15	9	—	3
Northeast	14.2	–0.66	–4.3	24	7	8	2	—
London	13.7	–0.52	–2.8	12	3	1	2	—
East Midlands	13.4	–0.70	–5.1	10	3	2	—	1
Southwest	11.4	–0.53	–4.5	4	1	2	—	1
Scotland	11.2	–0.69	–6.1	17	6	4	4	1
Southeast	10.7	–0.43	–3.9	11	2	1	3	1
Wales	9.2	–0.41	–4.5	5	—	—	3	—
Suburban and rural sites	4.7	–0.02	0.3	22	—	1	4	1
Network average or total	14.5	–0.67	–4.8	197	66	45	20	9

711. The London sites showed some of the lowest trends of all the regional groupings, only –0.5 $\mu g\ m^{-3}$ (–2.8%) per year. This had the effect of taking the London region from close to the middle of the rankings at the start of the trend period to the second most polluted region after Northern Ireland at the end. This situation appears to have resulted from the phase-out of coal burning having taken place largely before the start of the trend period, leaving a significant contribution from other sources, particularly road transport. The most polluted London sites (Lambeth, Acton, London City, Ealing, Greenwich and Ilford) have shown downwards trends of between –6.7% per year and –3.7% per year. In contrast, the least polluted sites, with annual mean black smoke concentrations in the range 6–9 $\mu g\ m^{-3}$, have shown no significant trends during the period from 1990 to 2002. Since these levels are considerably higher than background black smoke levels (2 $\mu g\ m^{-3}$ at Lerwick), the presence of a road traffic contribution is presumably the explanation for the lack of observed trends.

7.2.2 Particulate sulphate trends

712. Particulate sulphate levels have been falling steadily across Europe in response to a reduction in regional SO_2 emissions. Figure 7.3 shows the trend in the UK network average for the period 1987–2003 and indicates a substantial downwards trend. The annual average concentrations up to 2001 were calculated for the eight NETCEN Acid Rain Monitoring Network sites, extended to 2003 using the 12-site CEH network. Table 7.3 summarises the trend data from the rural particulate sulphate monitoring carried out as part of the Acid Rain Monitoring Network. UK particulate sulphate levels are falling with highly statistically significant annual trends that lie in the range of between –4.3% and –9.3% per year. Over the period 1990–2002, particulate sulphate levels appear to have fallen by more than half, with the UK average concentration falling from 3.7 µg SO_4 m^{-3} to 1.6 µg SO_4 m^{-3}.

Figure 7.3 Trends in the annual average particulate sulphate concentrations between 1987 and 2003.



713. The observed trends in particulate sulphate levels are somewhat smaller than the decline in UK SO_2 emissions over the same period, which averaged about –11% to –12% per year, and those of the EC by –10% per year. This would point to an increasing fraction of the emitted SO_2 being oxidised and present in the atmosphere as particulate sulphate. This may have resulted as a by-product of the decreased NO_x emissions and hence increased photochemical oxidation rate for SO_2 to sulphate aerosol.

714. This apparent increase in the oxidising capacity has been observed across Europe. Figure 7.4 shows the long-term measurements of sulphur dioxide and particulate sulphate at the EMEP GB02 site at Eskdalemuir, Dumfries and Galloway during the period 1977–2002. Although SO_2 concentrations have fallen by a factor 7, those

Figure 7.4 Long-term trends in sulphur dioxide and particulate sulphate (µg S m^{-3}) observed at Eskdalemuir, Dumfries and Galloway between 1977 and 2002.

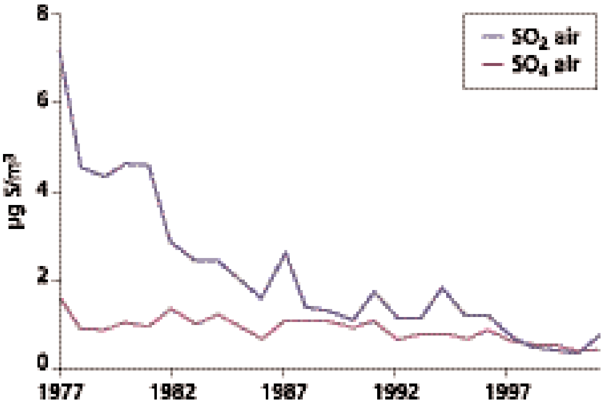


Table 7.3 Annual mean particulate sulphate concentrations at UK rural locations in 1990 and 2002 and their annual trends between1990 and 2002.

	Annual mean (μg S m^{-3})		Annual trend		
	1990	2002	μg S m^{-3} year 1	% per year	Statistical significance
Eskdalemuir	0.94	0.46	0.043	4.5	$P < 0.001$
Stoke Ferry[a]	1.66	0.72	0.093	5.4	$P < 0.001$
Lough Navar	0.81	0.40	0.046	4.9	$P < 0.01$
Barcombe Mills	1.48	0.71	0.076	4.7	$P < 0.01$
Yarner Wood	1.21	0.58	0.064	5.0	$P < 0.01$
High Muffles	1.33	0.59	0.068	5.0	$P < 0.001$
Strathvaich Dam[a]	0.87	0.32	0.044	5.9	$P < 0.01$
Glen Dye[a]	1.52	0.46	0.049	5.3	$P < 0.01$

[a] Data and trends refer to 1990–2001.

of particulate sulphate have fallen only by a factor 2. The fraction of airborne sulphur present as particulate has thus increased from ~0.18 to 0.33, a near doubling over the 25-year period.

7.2.3 PM$_{10}$ trends

715. There are 12 long-running PM$_{10}$ monitoring sites across the UK that allow trend determination over the period up to 2003; the time series of their annual mean PM$_{10}$ concentrations are shown in Figure 7.5. All trend analyses for PM$_{10}$ have used unscaled TEOM data. Also shown in Figure 7.5 is the average concentration for these long-running sites. Clear downwards trends are apparent in all cases. All these sites and their average showed highly statistically significant downwards trends over the period up to 2003. The observed downwards trends were between –0.3 and –1.7 μg m^{-3} per year or between –1.7 and –10.5% per year, see Table 7.4. The 12-site average annual mean concentration has shown a highly statistically significant downwards trend of –0.9 μg m^{-3} year 1 or –4.4% year 1. Interestingly, Figure 7.5 shows that the strength of this downwards trend has not remained constant over the 1992 2003 period. The downwards trends appears to have slowed significantly and there is a strong hint that it may have been reversed during 2000 to 2003.

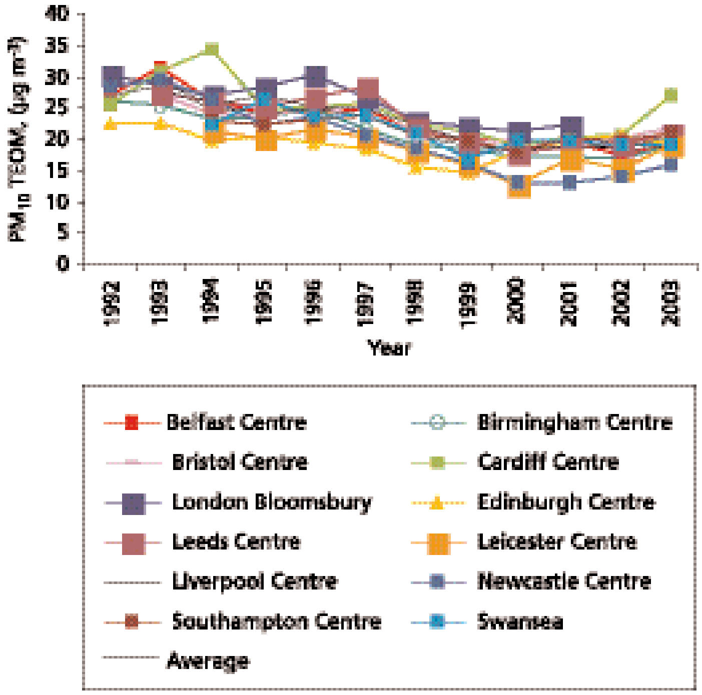
716 Across the UK, 48 PM$_{10}$ monitoring sites have long enough records for trend determination during the 1997–2003 period and longer. The available trend data are summarised in Table 7.4, which gives the length of the monitoring record, the annual mean PM$_{10}$ concentration in 2003, the annual trend in μg m^{-3} per year or percentage change per year and the statistical significance of the observed trend. All but three of the sites show downwards trends in annual mean PM$_{10}$ concentrations, with just under half being highly statistically significant. The range of the observed downwards trends was from more than –14% per year to less than +3% per year. The network average annual mean PM$_{10}$ concentrations declined from 23.4 μg m^{-3} to 20.5 μg m^{-3}, indicative of a trend of about 0.5 μg m^{-3} year^{-1} or –2% per year. Highly statistically significant downwards trends

Table 7.4 Annual mean PM_{10} concentrations and their recent trends, with their statistical significance at 48 locations, ranked in order of percentage change per year. (All analyses were performed on unscaled TEOM data.)

Site	**First year**	**Last year**	**Annual mean 2003 ($\mu g\ m^{-3}$)**	**Annual trend**		**Statistical significance**
				$\mu g\ m^{-3}$ $year^{-1}$	**% change $year^{-1}$**	
Stoke-on-Trent centre	1997	2003	11	–1.5	–14.0	$P < 0.01$
Reading	1997	2003	12	–1.5	–12.8	$P < 0.05$
Newcastle centre	1992	2003	16	–1.7	–10.5	$P < 0.001$
Stockport	1996	2003	15	–1.3	–8.4	$P < 0.01$
Liverpool	1993	2003	19	–1.2	–6.1	$P < 0.01$
Sheffield centre	1995	2003	21	–1.2	–5.9	$P < 0.05$
Norwich centre	1997	2003	18	–1.0	–5.8	—
Glasgow centre	1996	2003	16	–0.9	–5.7	$P < 0.05$
London A3 roadside	1997	2003	26	–1.4	–5.6	—
Leeds centre	1993	2003	21	–1.1	–5.0	$P < 0.05$
Birmingham centre	1992	2003	19	–1.0	–5.0	$P < 0.001$
Belfast centre	1992	2003	19	–0.9	–5.0	$P < 0.001$
Derry	1997	2003	18	–0.9	–4.9	—
Redcar	1997	2003	21	–1.0	–4.6	—
London Bloomsbury	1992	2001	23	–1.1	–4.6	$P < 0.01$
Glasgow kerbside	1997	2003	24	–1.0	–4.3	—
Narberth	1997	2003	14	–0.6	–4.1	—
Salford Eccles	1992	2003	18	–0.7	–4.1	—
Leicester centre	1994	2003	19	–0.7	–3.9	$P < 0.05$
Belfast Clara St	1994	2003	22	–0.8	–3.5	$P < 0.05$
Bristol centre	1993	2003	22	–0.7	–3.4	$P < 0.01$
Bradford centre	1997	2003	21	–0.7	–3.3	—
Swansea	1994	2003	19	–0.6	–3.3	$P < 0.05$
Middlesbrough	1995	2003	21	–0.7	–3.2	—
Nottingham centre	1996	2003	20	–0.6	–3.2	—
Cardiff centre	1992	2003	27	–0.8	–2.9	$P < 0.1$
Manchester Piccadilly	1995	2003	22	–0.6	–2.9	–
Port Talbot	1997	2003	24	–0.7	–2.8	$P < 0.05$
Camden kerbside	1996	2003	27	–0.8	–2.8	—
Leamington Spa	1996	2003	21	–0.6	–2.7	—
Haringey roadside	1996	2003	22	–0.5	–2.3	—

Site	First year	Last year	Annual mean 2003 ($\mu g\ m^{-3}$)	Annual trend		Statistical significance
				$\mu g\ m^{-3}\ year^{-1}$	% change $year^{-1}$	
Hull	1994	2003	22	–0.5	–2.1	—
Plymouth centre	1997	2003	17	–0.4	–2.1	$P < 0.1$
Rochester	1996	2003	19	–0.4	–2.0	—
Bury roadside	1997	2003	27	–0.5	–1.9	—
Southampton centre	1994	2003	21	–0.4	–1.9	—
London N. Kensington	1996	2003	22	–0.4	–1.8	—
Edinburgh centre	1992	2003	19	–0.3	–1.7	$P < 0.1$
London Hillingdon	1996	2003	23	–0.4	–1.6	—
Wolverhampton centre	1995	2003	19	–0.3	–1.6	$P < 0.1$
Bolton	1997	2003	18	–0.2	–1.3	—
Birmingham East	1993	2003	17	–0.2	–1.2	—
Thurrock	1996	2003	23	–0.2	–0.8	—
London Brent	1996	2003	20	–0.1	–0.7	—
London Eltham	1996	2003	21	–0.1	–0.3	—
London Marylebone Road	1997	2003	37	0.0	0.0	—
Lough Navar	1996	2003	12	0.1	1.1	—
Scunthorpe	1997	2003	30	0.9	3.0	—

Figure 7.5 Time series of annual mean PM_{10} concentrations at the long-running urban background sites in the UK between 1992 and 2003.



during 1997–2003 were found at eight sites: Belfast centre, Birmingham centre, Glasgow centre, Plymouth centre, Port Talbot, Reading, Stockport and Stoke-on-Trent centre. The observed trends fell in the range of –0.4 to –1.5 µg m^{-3} per year. Slight upwards trends were reported for the Birmingham East, Edinburgh centre, Manchester Piccadilly, Middlesbrough and Scunthorpe sites.

717. The London conurbation appears to be showing somewhat different behaviour compared to the other network sites with regard to PM_{10} trends during the 1997–2003 period. At one site, London A3 roadside, the observed trend was higher than the 48-site trend. However, at all others either no trend was found (London Brent, London Eltham and London Marylebone Road) or small downwards trends in the range –1.0 to –1.8% per year were found (Camden kerbside, Haringey roadside, London Hillingdon, London North Kensington and Thurrock).

718. Urban PM_{10} trends have appeared to slow down during the period 2000–2003 compared to the 1993–1999 period. At some urban sites there is some evidence for increasing annual mean PM_{10} concentrations during 2000–2003. Bearing in mind that the year 2003 appears to have been influenced by some large-scale regional pollution episodes, it is an issue whether the trends during the 2000–2003 period have been overly influenced by 2003. The residuals in the trend analyses have been examined to see if they significantly concurred with the other years. For 33 of the sites in Table 7.4 it was apparent that 2003 did not stand out as particularly high in terms of the 2000–2003 period. However, for 15 sites, that is, about one-third, the year 2003 stood out as significantly different from the overall trend. There was, however, no clear spatial pattern in the distribution of the sites that showed the year 2003 as an outlier.

719. The rural sites at Narberth in Wales and Rochester in Kent show PM_{10} downwards trends of –0.6 µg m^{-3} per year and –0.4 µg m^{-3} per year, respectively. It is possible that the observed PM_{10} trends at these sites are driven by the simultaneous trends in particulate sulphate described in Section 7.2.2 above. The particulate sulphate trends estimated for Narberth and Rochester, accordingly, are –0.06 µg S m^{-3} per year and –0.08 µg S m^{-3} per year or –0.25 and –0.33 µg PM_{10} m^{-3} per year (by scaling up to $(NH_4)_2SO_4$ from S). It is clear, therefore, that the observed downwards trends in PM_{10} reported for these rural sites are larger (by factors of 1.2 to 2.5) than those trends that could be explained solely by the observed trends in particulate sulphate over the same period. There must therefore be significant proportions of primary or other secondary PM in the PM_{10} size range, reaching these rural sites and these proportions must have been showing significant downwards trends.

720. The rural site at Lough Navar, Northern Ireland shows the least PM_{10} trend of three rural sites, Lough Navar, Narberth and Rochester. The levels monitored at this site are the lowest reported for any of the 48 sites and have remained consistently in the range 9.4 to 9.9 µg m^{-3} for 5 of the 8 years that the site has been operational. This site might have been expected to show some evidence of a downwards trend if only from the decreasing long range transport of particulate sulphate, as shown by the Eskdalemuir site. The annual means for 2000–2003 have been consistently higher than the 9.4–9.9 µg m^{-3} range and it is clear from the residuals in the trend analysis that this reflects a reversal in trend rather than the influence of year-on-year variability caused by 2003.

721. Strong downwards trends are thus a widely observed feature of urban PM_{10} levels in the UK during the 1990s, before the apparent levelling off during the 2000–2003 period. Average PM_{10} concentrations are declining at about –4.4% per year, as indicated by the 12 long-running urban background sites during the 1990s, and this is close to the trend in PM_{10} emissions that have, over the same period, declined at about –5% per year, according to the NAEI.

722. Although PM_{10} emissions have continued to decline strongly from 1997 to 2003, this is not the case for urban PM_{10} concentrations. Only 12 of 48 sites achieved annual percentage trends during the 1997–2003 period that approached those shown by PM_{10} emissions. The 48-site average showed a trend of –2.2% $year^{-1}$, one-half of the trend shown by the estimated PM_{10} emissions over the same period. The PM_{10} emission inventory is therefore not giving a clear indication of the likely origin of the observed slowing up in the downwards trends in urban PM_{10} levels observed during 2000–2003.

7.2.4 $PM_{2.5}$ trends

723. There is only one long-running time series of $PM_{2.5}$ measurements in the UK and these are for Hodge Hill, Birmingham, a suburban background site operated by Birmingham City Council. Annual mean $PM_{2.5}$ concentrations were found to be 14 µg m^{-3} during 1995 and 12 µg m^{-3} in 2002. Over the 7-year period, $PM_{2.5}$ levels have shown a statistically significant (at the 90% level of significance) downwards trend of –0.4 µg m^{-3} (–3.1%) per year.

724. According to the NAEI, emissions of $PM_{2.5}$ have declined by –5.2% per year over the period 1995–2001. This decline in emissions is somewhat larger than the observed decline in $PM_{2.5}$ concentrations in Birmingham. This would suggest that primary emissions are not the main source of $PM_{2.5}$. Particulate sulphate is likely to be a component of the $PM_{2.5}$ fraction and is likely to be declining at about –0.09 µg S m^{-3} per year, based on the Stoke Ferry observations shown in Table 7.3. On this basis, a downwards trend of –0.4 µg $PM_{2.5}$ m^{-3} per year (by scaling up to $(NH_4)_2SO_4$ from S) would be anticipated, close to that observed at the Hodge Hill site. This would point to secondary sulphate particle trends as the main driving force behind the $PM_{2.5}$ trends at the Hodge Hill site and not primary emission trends.

Table 7.5 Observed trends in annual mean $PM_{2.5}$, PM_{10} and PM_{coarse} concentrations at four AURN sites between 1998 and 2003.

Site	$PM_{2.5}$ trends (µg m^{-3} $year^{-1}$)	PM_{10} trends (µg m^{-3} $year^{-1}$)	PM_{coarse} trends (µg m^{-3} $year^{-1}$)
London Marylebone Road	0.0	0.5	0.5
London Bloomsbury	–0.4	—	—
Harwell	0.3	0.0	0.0
Rochester	0.0	0.5	0.0

725. Co-located measurements of PM_{10} and $PM_{2.5}$ concentrations have been made at four AURN sites from 1998 onwards and their time series are just long enough to enable comments to be made about trends. Table 7.5 indicates that annual mean $PM_{2.5}$ concentrations have shown no overall trend at the London Marylebone Road and Rochester sites, an increasing trend at the Harwell site and a decreasing trend at the London Bloomsbury site, although none of these trends was highly statistically significant. The observed downwards trend in $PM_{2.5}$ at London Bloomsbury of –0.4 µg m^{-3} per year corresponds closely with the downwards trend of –0.4 µg $PM_{2.5}$ m^{-3} per year anticipated at Stoke Ferry on the basis of the observed trend in particulate sulphate.

7.2.5 PM_{coarse} trends

726. There is only one site in the UK for which long-running time series are available for both PM_{10} and $PM_{2.5}$ using co-located instruments, and this is the Hodge Hill, Birmingham site referred to in Section 7.2.4 above. Annual mean PM_{coarse} concentrations were monitored and found to be 9 µg m^{-3} in 1995 and 3 µg m^{-3} in 2002. Over the 7-year period, PM_{coarse} levels have shown a highly statistically significant (at the 99% level of significance) downwards trend of –0.7 µg m^{-3} (–13%) per year. There are no other time series that can be used to corroborate such a large and significant trend in particulate levels. The corresponding trend in PM_{10} levels observed at this site is –1.1 µg m^{-3} (–4.9%) per year. This is somewhat higher than the trends for the Birmingham centre and Birmingham East sites of –1.0 and –0.6 µg m^{-3} per year or –3.7 and –2.8% per year, reported for the same time period. On this basis, it would appear that at the Hodge Hill, Birmingham site, much of the observed trend in PM_{10} levels has been caused by the decline in PM_{coarse} levels.

727. The strong downwards trend in PM_{coarse} levels could point to coarse fraction secondary nitrate particles as the main source of the observed downwards trend in PM_{coarse} levels at the Hodge Hill site, through the influence of the halving of European regional scale NO_x emissions. An alternative explanation could be that there has been a steep decline in PM_{coarse} emissions during 1995–2002. The NAEI shows PM_{coarse} emissions declining by –4.4% $year^{-1}$ over during 1995 to 2001, thus providing some explanation for the large observed trend at the Hodge Hill, Birmingham site.

728. Table 7.5 indicates that at the four AURN sites from 1998 to 2003, annual mean PM_{coarse} concentrations have exhibited increasing trends at the London Marylebone Road and no trends at the rural sites (Harwell and Rochester), although none of these trends was highly statistically significant.

7.2.6 Ultrafine particle number trends

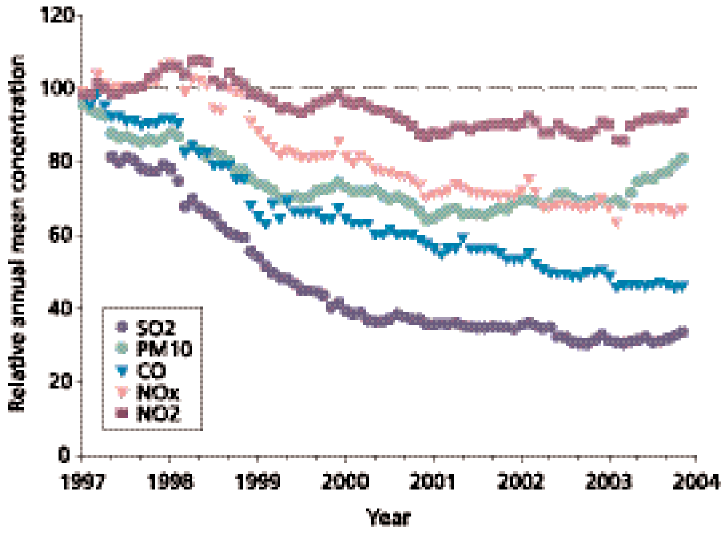
729. Particle number count is dominated by the ultrafine particle size range. An analysis of UK particle number data is presented in Chapter 6, but the time-series of measurements are insufficient to be able to comment on the long-term trends. Elsewhere in Europe, particle number measurements made each winter from 1991/1992 to 2000/2001 in Erfurt, Germany, have shown that, whereas the mass concentration of fine particles decreased substantially over this period (by 75%), the total number of ultrafine particles remained fairly constant. This was the consequence of a significant shift in the size distribution towards smaller sizes, with a twofold increase in the number concentration between 10 and 30 nm (Kreyling *et al.*, 2003). In Helsinki, Finland, annual mean particle number concentration (for the 8–400 nm size range) declined by about 25% between

1997 and 2003 (Hussein *et al*. 2004). The very limited evidence, therefore, suggests that average ultrafine particle number concentrations are declining and shifting to smaller sizes, but are likely to show a different trend to PM_{10}. It is important to remember that particle number concentrations exhibit wide fluctuations in short-scale spatial and temporal variability about longer term means.

7.2.7 Normalised monthly mean PM_{10} trends within the LAQN

730. Figure 7.6 shows the trends in different pollutants in London from the beginning of 1996. Annual data have been averaged at monthly intervals and hence show a start date of 1997. Concentrations have been normalised to 100 at the beginning of the dataset to highlight the relative change in the concentrations of the different species. A range of site types and sites mostly from the LAQN has been used for the different species, depending on data availability and the need for a consistent dataset since 1996. Most pollutants have shown a steady decline in concentration since 1996. The most significant reductions have been observed for SO_2 and CO, where concentrations have declined by over 50% between 1996 and 2003. For PM_{10}, the annual mean concentration declined rapidly until 1999, but has seen a significant increase during 2003 due to the frequency of pollution episodes in that year. Since 1996 PM_{10} concentrations have declined by 19% by the end of 2003.

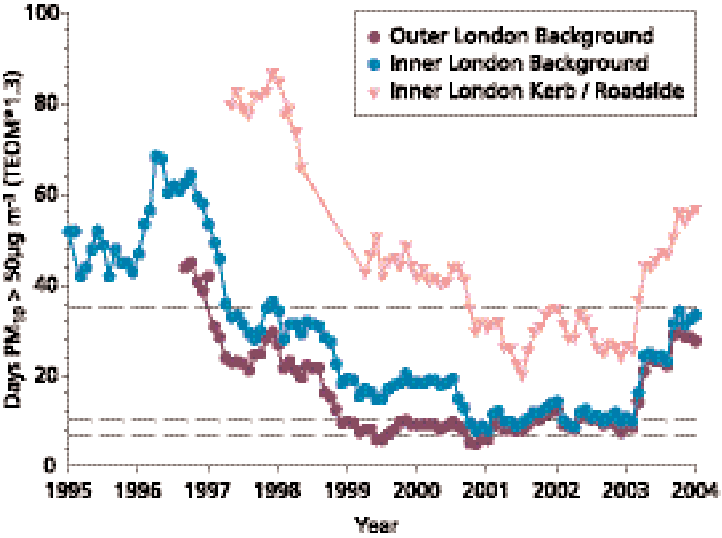
Figure 7.6 Relative annual mean concentration (monthly intervals) for a selection of monitoring sites in London. (Data for 2003 are provisional.)



731. The sites in the LAQN, therefore, seem to be showing the same behaviour in PM_{10} trends as the other urban background sites throughout the UK. Annual mean PM_{10} trends were strongly downwards during the 1990s but flattened out during 2000–2003 and have shown a slight increase during 2003. Figure 7.6 shows that the PM_{10} and NO_x curves followed each other closely during 1997 2000 and then began to diverge increasingly from 2001 onwards. This divergence has been maintained through 2003, a year associated with large regional pollution episodes.

732. Figure 7.7 shows how the daily mean concentrations >50 µg m^{-3} have varied at three different locations in London. These data have been compiled in a similar

Figure 7.7 Days when the daily mean PM_{10} concentration was >50 μg m^{-3} (TEOM * 1.3) for different categories of monitoring sites in London. Annual data have been averaged at monthly intervals.



way to that used for Figure 7.6. It is clear that there is significant interannual variability in this statistic, which highlights the importance of different episodes in different years. Of note are the episodes during 1996 and 2003. The most recent pollution episodes during 2003 resulted in the inner London roadside and kerbside sites exceeding the 35 days objective, with the inner London background sites being very close to this limit. Also shown in Figure 7.6 are the levels of the different limits and objectives shown at 35, 10 and 7 days.

7.2.8 Trends at sites close to motorways

733. TRL operates two long-running sites, on behalf of the Highways Agency, by the side of the M4 and M25 motorways. The M4 site measures PM_{10} and the M25 site measures both PM_{10} and $PM_{2.5}$. During 1996–2003 the annual mean PM_{10} concentrations showed downwards trends of –0.33 and –0.85 μg m^{-3} $year^{-1}$ at the M4 and M25 sites, respectively, although neither were highly statistically significant. Interestingly, both sites showed strong downwards trends at the start of the period, which flattened out and then showed signs of an increase during 2000–2003, as exhibited by many urban background and roadside sites.

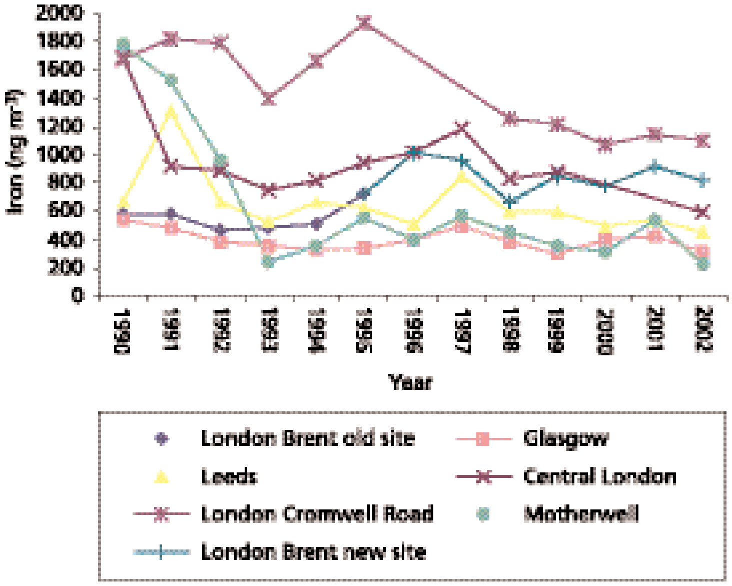
734. During 1999–2003 there were no significant trends in the annual mean PM_{10}, $PM_{2.5}$ or PM_{coarse} concentrations at the M25 site.

7.2.9 Trends at the multi-element sites

735. Up to eight trace elements are monitored at six sites in the multi-element network. During 1990 to 2002, highly statistically downwards trends were observed for cadmium at the Glasgow, London Cromwell Road and Motherwell sites, but no trends were observed at the Leeds, Central London and London Brent sites. Highly statistically significant upwards trends were reported for chromium at the London Brent and Glasgow sites. Upwards trends were also observed for chromium at the Leeds, London Cromwell Road and Motherwell sites and downwards at the Central London site, but these were not highly statistically significant. Highly statistically significant downwards trends were observed for copper at the Glasgow, London Cromwell Road and Motherwell sites. Manganese showed highly statistically significant downwards trends at five of the six sites, with no trend detected at the London Brent site. Nickel and vanadium showed highly statistically significant downwards trends at all six sites.

736. Iron is an important trace element in its own right. It is also an important indicator species in source attribution analyses. It is appreciably enriched in roadside air and appears to be associated with the non-exhaust emissions from road traffic (Harrison *et al.*, 2003, 2004). Its origin is probably brake wear and corrosion products, although the relative amounts of direct emissions and non-exhaust traffic emissions are not known. It is also present in soils, but at much lower abundance. Figure 7.8 plots the time series for annual mean iron concentrations at the six multi-element sites for the period 1990 to 2002. Levels at the Motherwell site declined quickly during the early 1990s following the cessation of industrial activities close to the site. The Leeds and Glasgow sites showed steady declines throughout the 1990s. The behaviours observed at the three London sites were characteristically different from each other. The London Cromwell Road and Central London sites showed steady declines as seen for the Leeds and Glasgow sites, but the London Brent site showed a steady increase, although the site location was changed in 1995. Table 7.6 shows that this steady increase at the London Brent site is highly statistically significant.

Figure 7.8 Time series of the annual mean iron concentrations observed at the multi-element sites between 1990 and 2002.



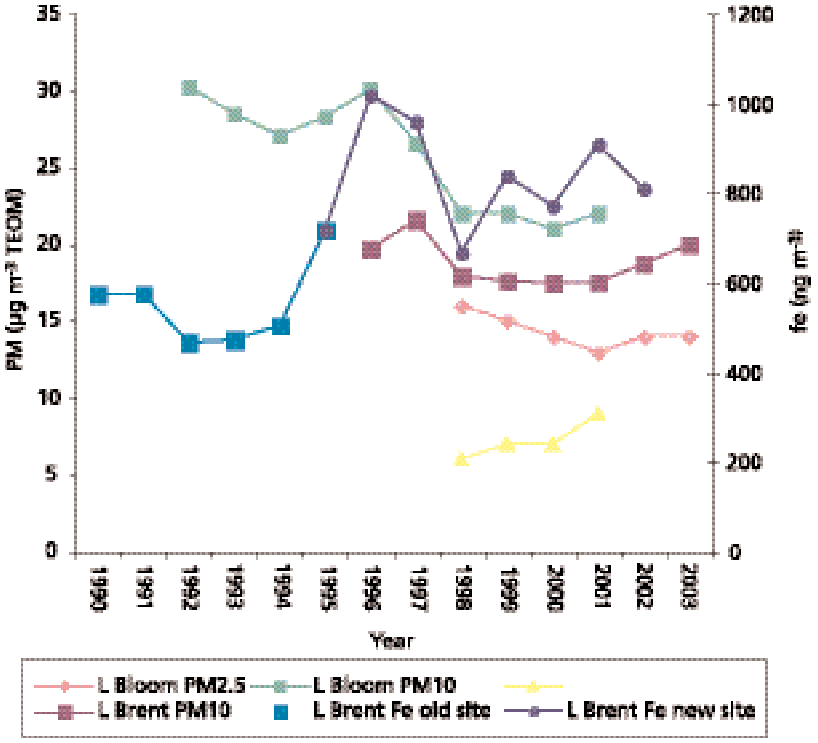
737. Figure 7.9 shows the time series for particulate iron and PM_{10} at London Brent from 1990 to 2003. Also shown are the PM_{10}, $PM_{2.5}$ and PM_{coarse} measurements for London Bloomsbury. The iron at London Brent shows a steady upwards trend with a major excursion during 1995 to 1997, although the site location was changed in 1995. The PM_{10} levels also show elevated levels during 1996–1997 and steady upwards trends thereafter exactly analogous to the behaviour shown by particulate iron. PM_{10} and particulate iron were both high during 1996–1997, which was marked by a number of intense regional pollution episodes.

Table 7.6 Details of the observed trends in the annual iron concentrations at the multi-element survey sites from 1990 to 2002.

Site	Start	Finish	Annual mean in 2002 (ng m^{-3})	Annual trend (ng m^{-3} $year^{-1}$)	Annual trend (% change per year)	Statistical significance
London Brent[a]	1990	2002	810	32.6	4.0	$P < 0.05$
Central London	1990	2002	313	22.0	7.0	—
Glasgow	1990	2002	448	7.4	1.6	—
Leeds	1990	2002	598	17.1	2.9	$P < 0.05$
London Cromwell Road	1990	2002	1102	52.7	4.8	$P < 0.05$
Motherwell	1990	2002	224	69.9	31.2	$P < 0.05$

[a] Site change in 1995.

Figure 7.9 Time series for particulate iron and PM_{10} at London Brent and of PM_{10}, $PM_{2.5}$ and PM_{coarse} at London Bloomsbury, between 1990 and 2003.



738. The highly statistically significant upwards trend in iron observed at the London Brent site could indicate an upwards trend in non-exhaust traffic emissions due to increasing traffic levels. It could also explain why the observed PM_{10} trends at the London sites, particularly the London Brent and London Marylebone Road sites, were significantly smaller than those observed elsewhere in the UK.

Chapter 8

Modelling of particulate matter

Key points

- Models for predicting PM concentrations are essential for assessing pollutant levels at locations without monitors, for making future projections and for investigating the potential of different air pollutant mitigation policies.

- In view of the many different sources and the diverse chemical make-up of particulates in the atmosphere, no model is able to fully describe all the processes related to their generation and spatial variation in concentration. However, a range of different types of model of varying sophistication are currently available for predicting concentrations for different spatial scales and averaging times.

- Models routinely used for scales up to the national scale are either empirically based or make use of dispersion algorithms or a combination of the two. The empirical models are generally fast to run, make full use of available data and are able to provide both source apportionment and predictions of annual means. However, they are not able to provide reliable daily peak concentrations nor are they easily transferable to areas with few observations. In addition, both estimated mass concentration and source apportionment depend critically on the measurement technique employed.

- Dispersion models are able to predict concentrations at smaller averaging times and at greater spatial resolution. They can provide source apportionment for each emission category defined within the relevant emission inventory. However, they make less direct use of available data and may require increased run times. Dispersion models are not able to account for the variation of roadside concentration with road type unless they consider local effects such as surface roughness changes, vehicle-induced turbulence, exhaust height, presence of street canyons and so on. For this reason, empirical models have made use of roadside adjustment factors; however, these are based on limited data and little quantitative understanding of the processes involved.

- Both types of models have significant uncertainties connected with the residual largely coarse component, which is not modelled explicitly. The different empirical models are at variance in their estimates of the background coarse component as shown by the source apportionment, whereas dispersion models generally add a constant based on monitoring data to take account of this contribution. Neither approach can be projected forward in a satisfactory manner. The modelling of the traffic-related coarse component is also poorly formulated.

- Regional models continue to be developed by including more sophisticated parameterisations of the physical and secondary inorganic chemical processes. These models are now used routinely to investigate future projections and emission scenarios for policy development across the UK and Europe. Because of the large areas considered, their spatial resolution is much lower so that they are only able to predict background concentrations.

- The national model and London-based calculations suggest that in a typical meteorological year, the annual average PM_{10} will generally achieve the 2005 limit value (40 µg m^{-3}) in 2005, but that there are likely to be widespread exceedences of the Stage II indicative limit value of 20 µg m^{-3} in 2010, especially in urban areas. The extent of these exceedences varies significantly according to the model employed. Exceedences of the daily average will be close to the limit value in major urban areas, especially London in 2005; however, the extent of exceedence of the 2010 daily limit value depends markedly on whether this parameter is calculated from annual mean or directly from daily averages, as in dispersion models.

- Regional models are able to reproduce the main observed features of distributors of particulate sulphate, nitrate and ammonium across Northwest Europe and the southeast-northwest gradient across the British Isles.

- $PM_{2.5}$ modelling has been calculated both nationally and for London. However, the reliability of these models is currently difficult to determine because of the wide divergence in measurements of the different monitoring devices and lack of agreed conversion (or scaling) factors.

8.1 Introduction

739. This chapter presents details of local, urban and regional models used for calculating PM concentrations in the UK. It also includes maps and tables of calculated PM concentrations of relevance to compliance with PM_{10} air quality limits and policy development in the UK. Although monitors can give information about PM concentrations at specific points, they can only give limited information about the spatial distribution of concentration, source apportionment and future concentrations. Models are therefore an essential tool for assessing future policy and for understanding the physical and chemical processes that determine PM concentrations and trends.

740. A fully comprehensive model for PM would include source modelling, particle condensation and accumulation models, advection/dispersion/deposition and chemical transformation models on a range of spatial scales from local to long range. The compilation of such a model is not scientifically viable at the moment because some important processes (for example, non-exhaust traffic emissions) are not well understood compared to other processes; it would not be practically useful either. However, a great number of models do exist for different aspects of the problem. These tend to focus on specific scale ranges or physical/chemical processes and either ignore other important processes/effects or take account of them through simple assumptions or parameterisation and/or use appropriate monitoring of PM concentrations to complement the modelling. It is not possible to describe or even mention the full range of models available and so prominence is given to the models that are informing policy development and assisting with air quality assessment in the UK. For convenience these are presented in four separate categories, although sometimes – for certain aspects of the models – the distinctions between the different categories are not clear cut.

741. The models that have been used routinely to calculate concentrations (both current and projected) across the UK are the APEG Receptor Model and the Mapping Model developed by Netcen. This first group of models is semi-empirical, relying to a large extent on monitored data, but they also include a local dispersion modelling component. The second group of models covers those designed for calculating concentrations at high spatial resolution across urban areas. These vary in the relative extent to which they rely on either dispersion modelling and/or monitoring data as the basis for their calculations. The third major group are the regional or long range models. These cover greater spatial scales at lower spatial resolution than the urban models. These models require meteorological fields as input and include both Eulerian and Lagrangian models for advection and dispersion. They may also include advanced routines for droplet condensation/accumulation and chemical transformation. The remaining category consists of road emission models and finally there is some discussion of a miscellaneous set of models, including particle accumulation models, particle source models and other models/modelling studies prominent in Europe or North America but which have not been used for policy development in the UK.

742. This chapter starts by presenting each of these model groups in turn, including validation and case studies as appropriate. Section 8.3 compares and contrasts the different modelling approaches to important effects/contributions. Section 8.4 presents model outputs mainly from the first three groups described above. This section includes current and future projections, at both national and urban scales, maps and receptor point output and detailed source apportionment; where possible, appropriate comparisons are made between different models. Finally, Section 8.5 includes discussion of model uncertainty before conclusions and recommendations are made in Section 8.6.

8.2 Current modelling and mapping methods

8.2.1 National models – Receptor and Mapping

8.2.1.1 Site-specific source apportionment using the APEG receptor model

743. This model – developed for APEG (APEG, 1999; Stedman *et al*., 2001a) – calculates the source apportionment of PM_{10} and $PM_{2.5}$concentrations at locations where concentrations of these pollutants are measured. The basic assumption is that the daily average PM concentration can be considered as three distinct components, namely:

- primary combustion PM;
- secondary PM; and
- 'other' PM.

744. It is further assumed that the primary PM is related directly to co-located NO_X measurements and that secondary PM can be determined from rural sulphate measurements. Then a regression analysis is carried out for the calendar year of monitoring data for each site to determine the coefficients A and B as follows:

$$\text{[measured PM}_{10}\text{ (µg m}^{-3}\text{, TEOM)] = A [measured NO}_X\text{ (µg m}^{-3}\text{, as NO}_2\text{)]}$$
$$\text{+ B [measured sulphate (µg m}^{-3}\text{)] + C (µg m}^{-3}\text{, TEOM).}$$

Note that the analysis presented here is generally based on TEOM instruments.

745. The contribution from primary PM is further subdivided into the contributions from each emission sector (traffic, domestic, industry and so on). The contribution from individual point sources is calculated using ADMS 3.1.

746. The contribution from secondary PM is divided into the contribution from sulphate and nitrate by assuming that all of the sulphate is present as ammonium sulphate and that the remainder of the secondary PM within the secondary fraction determined from the regression analysis is present as nitrate. Thus, all of the site-to-site variation in the relationship between sulphate and secondary PM concentrations is assumed to be due to variation in the nitrate component, as is apparent from the results for 2002 listed in Tables 8.6–8.13. Note the nitrate concentration is greater for gravimetric PM_{10} measurements than for TEOM PM_{10} measurements and very low for TEOM $PM_{2.5}$ measurements, as might be expected due to the losses of ammonium nitrate for the TEOM measurement and the relatively larger size ranges of sodium nitrate particles.

747. The site-specific source apportionment has been combined with temporal trends in emissions to calculate site-specific projections both backwards and forwards in time, as described in the AQEG NO_2 report (AQEG, 2004). The primary PM concentration from each source sector has been projected according to published emission trends for each sector. The contributions from sulphate and nitrate particles have been projected forward by assuming that sulphate and nitrate concentrations will follow the predicted emissions reductions required to meet the National Emissions Ceilings Directive targets. The trends in previous years have been derived from the trends in measured rural mean sulphate and nitrate across the UK. The concentration of ‘other’ particles (the residual) for which emissions cannot currently be specified is assumed to remain unchanged in all years. The projected total annual mean PM concentration from all sources is calculated as the sum of the contributions in each year.

8.2.1.2 Netcen mapping model

748. The Netcen mapping model (Stedman *et al*., 2003) is designed to calculate UK-wide maps of annual average pollutant concentration at sufficient speed that extensive scenario testing can be conducted. The background maps provide concentrations at 1-km resolution, and a single representative concentration for each road segment is used to represent a roadside increment.

749. The maps of background concentrations are made up of the following components:

- large point sources of primary particles;
- small point sources of primary particles;
- area sources of primary particles;
- secondary particles; and
- the residual component (usually be dominated by particles within the PM_{coarse} fraction).

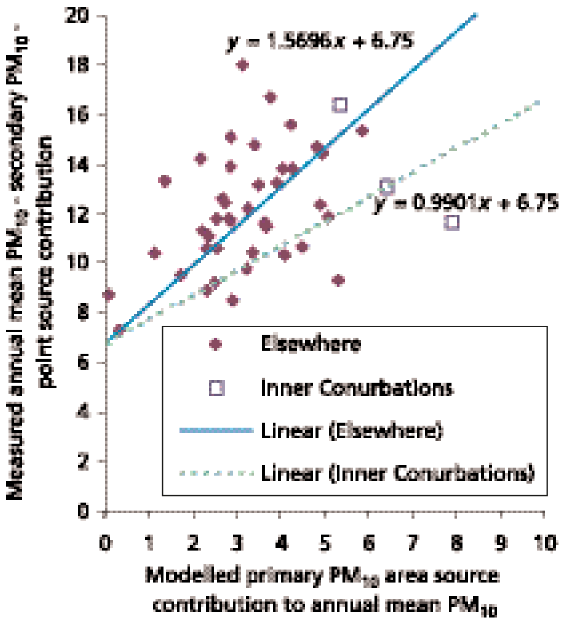
8.2.1.2.1 Contributions from point sources

750. The contribution to annual mean PM_{10} concentrations from 58 PM_{10} point sources with >200 tonnes per annum emission, is modelled using ADMS 3.1 and sequential meteorological data for 2002 from Waddington. Concentrations of PM_{10} point sources with less than 200 tonnes per annum release are modelled on the assumption that they all disperse in a similar manner i.e., the sources are assumed to have the same height, buoyancy and momentum (Stedman *et al*., 2003).

8.2.1.2.2 Contributions from area sources

751. Figure 8.1 shows the calibration of the area source model. The modelled large point and small point source and mapped secondary PM_{10} has been subtracted from the measured annual mean PM_{10} concentration at background sites. This is compared with the modelled area source contribution to annual mean PM_{10} concentration calculated with ADMS 3.1 using emissions from a 33 km x 33 km square in which the receptor point is at the centre. It is assumed that all area sources disperse with initial buoyancy or momentum and in identical meteorological conditions in this case 10-year average meteorological data for 1993–2002 from Heathrow have been used. Calibration plots are shown in µg m^{-3} (TEOM), since TEOM measurements have been used to calibrate the models. Following the same approach that used for NO_X, the monitoring sites are split into two groups: ‘inner conurbations’ and ‘elsewhere’, but the scatter is large. It is clear from the figure that the calibration relationships are much weaker than the corresponding relationships for NO_X (AQEG, 2004). This is due to the smaller contribution of local area sources to ambient PM concentrations compared with regional contributions relative to NO_X. Robust relationships have been found for NO_X and the same method has, therefore, been applied to PM. The modelled area source contribution is multiplied by the relevant empirical coefficient to calculate the calibrated area source contribution for each grid square in the country.

Figure 8.1 Calibration of a PM_{10} area source model.



8.2.1.2.3 Secondary particle contributions

752. Secondary particles are assumed to consist of sulphates and nitrates only, as in the receptor model. A map of secondary PM_{10} particle concentrations across the UK is calculated from rural measurements of sulphate and nitrate concentrations by interpolation onto a 20 km x 20 km grid. Sulphate and nitrate particle concentrations were measured on a monthly basis at 12 rural sites using a denuder method during 2002 (CEH, 2003).

753. Sulphate is assumed to be present as ammonium sulphate, and sulphate concentrations were multiplied by 1.354 to take the presence of the counter ion into account. The mean value of the APEG receptor model coefficient B, described above and relating secondary PM_{10} concentrations to sulphate concentrations in 2002, was 2.71, averaged over 11 background monitoring sites. A comparison of interpolated sulphate and nitrate concentrations at these locations indicates that a scaling factor for nitrate concentrations of 1.0 is equivalent (along with a sulphate scaling factor of 1.354) to the sulphate to nitrate ratio implied by the coefficient derived from the APEG receptor model. (The equivalent B factor derived from the interpolated concentration fields at these 11 locations was 2.76.) TEOM instruments are subject to partial losses of the more volatile particle components, such as ammonium nitrate. This may explain why a scaling factor of greater than 1.0 is not required to take account of the counter ions associated with the measured nitrate concentrations.

8.2.1.2.4 Coarse and other particles not included explicitly in the modelling

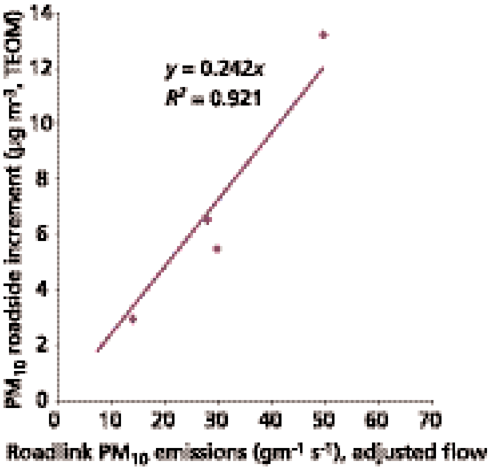
754. A constant residual particle concentration c above of 8.8 µg m^{-3} (gravimetric) (6.75 µg m^{-3}, TEOM) is the final component of the mapped PM_{10} concentration at background locations. This represents emissions of particles more especially in the larger size or coarser range and includes other sources such as wind-blown dusts, sea salt and agricultural activities, which are not generally included in emission inventories.

8.2.1.2.5 Roadside concentrations

755. The annual mean concentration of PM_{10} at a roadside location is assumed to be made up of two parts: the background concentration (as described above) and a roadside increment dependent on traffic flow:

$$\text{roadside increment } (\mu g\ m^{-3}, \text{TEOM}) = 0.242\ (g\ m^{-1}\ s^{-1})$$
$$*\ \text{road link emissions (adjusted traffic flow).}$$

Figure 8.2 Calibration of PM_{10} roadside increment mode.



756. The NAEI provides estimates of PM_{10} emissions for major road links in the UK for 2001 (Dore *et al.*, 2003) and these have been adjusted to provide estimates of emissions in 2002. Figure 8.2 shows a comparison of the roadside increment of annual mean PM_{10} concentrations at roadside or kerbside national automatic monitoring sites with PM_{10} emission estimates for the individual road links alongside which these sites are located. Emissions were adjusted for annual average daily traffic flow using the method described in Section 8.3.2.

8.2.1.2.6 Verification of mapped values

757. Figures 8.3 and 8.4 show comparisons of modelled and measured annual mean PM_{10} concentration in 2002 at both background and roadside monitoring site locations. Both the national network sites used to calibrate the models and verification sites are shown.

Figure 8.3 Verification of background annual mean PM_{10} model 2002.

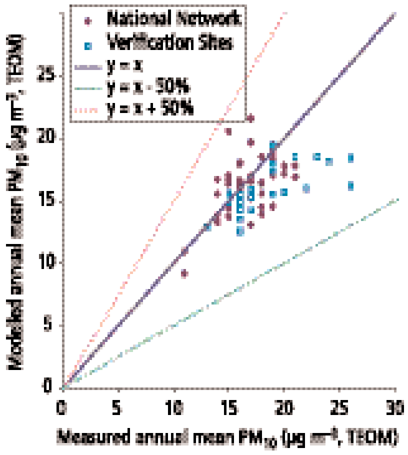
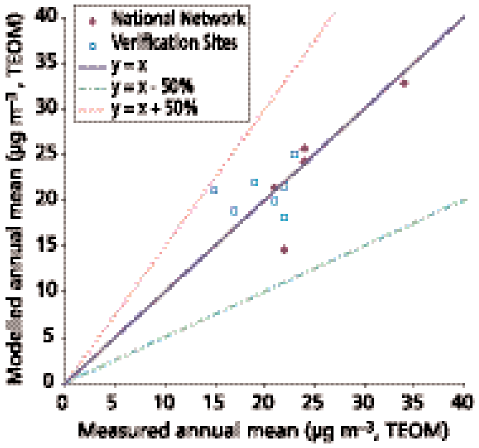
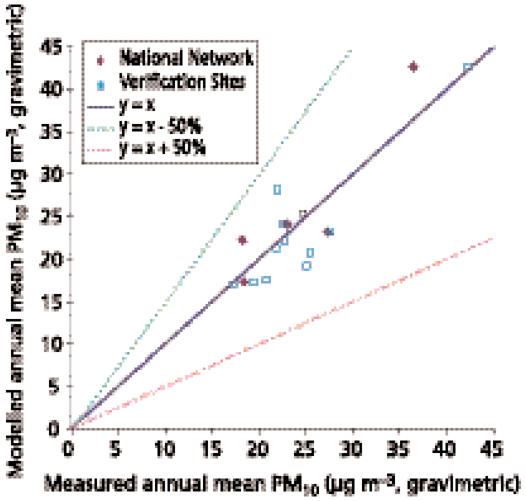


Figure 8.4 Verification of roadside annual mean PM_{10} model 2002.



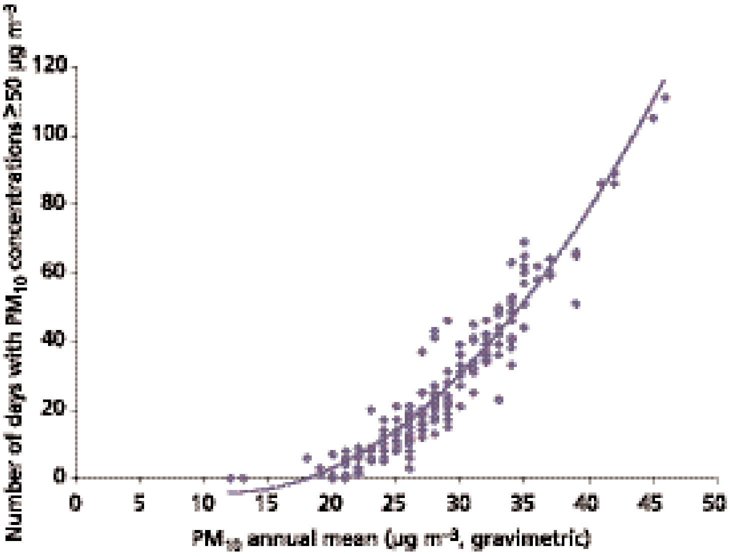
758. The PM_{10} models were calibrated using measurement data from TEOM instruments. Measurements of PM_{10} concentrations using gravimetric instruments (KFG and Partisol instruments) are now available for a number of sites in the UK. These measurements provide an additional independent verification of the model results. A comparison of measured annual mean concentrations in 2002 with estimates calculated by multiplying the modelled PM_{10} concentrations by 1.3 is shown in Figure 8.5.

Figure 8.5 Verification of annual mean TEOM PM_{10} * 1.3 model 2002: comparison with gravimetric measurements.



759. The 24-h mean concentrations are not explicitly modelled for comparison with the 24-h limit values. Instead an annual mean concentration of 31.5 µg m^{-3} (gravimetric) is taken to be equivalent to 35 days with 24-h mean concentrations greater than 50 µg m^{-3} (gravimetric; the Stage I 24-h limit value). This equivalence is derived from an analysis of recent monitoring data (Stedman *et al.*, 2001b) and is reproduced in Figure 8.6. The relationship between the number of days with concentrations >50 µg m^{-3}, gravimetric and annual mean become increasingly uncertain at lower numbers of exceedences.

Figure 8.6 The relationship between the number of days with PM_{10} concentrations ≥50 µg m^{-3} and annual mean concentration (1992–1999).



8.2.1.2.7 $PM_{2.5}$ maps

760. Maps of $PM_{2.5}$ concentrations are calculated using similar models to those applied to calculate the national maps of PM_{10} concentrations. Measurement data for $PM_{2.5}$ are only available for a much smaller number of sites and a generally applicable scaling factor between the different measurement methods is not available and so no equivalent factor to 1.3 for PM_{10} has been used for $PM_{2.5}$ in the Netcen mapping model. A factor of 1.3 has, however, been used in the ADMS-Urban calculations.

761. The NAEI PM_{10} emission inventory was used to calculate the $PM_{2.5}$ maps. The emissions from sectors for which the majority of the emission is expected to be in the size range greater than 2.5 µm were excluded (construction and quarries). Table 8.1 shows the separate scaling factors for TEOM and gravimetric measurements within the national mapping model for secondary PM concentrations and the concentrations of PM not modelled explicitly.

762. There is clearly some inconsistency between the models adopted for PM_{10} and $PM_{2.5}$ because a factor of 1.3 has been applied to the PM_{10} TEOM maps to scale to the PM_{10} gravimetric maps, whereas the different $PM_{2.5}$ maps account for the difference between the monitoring methods by applying different scaling factors for nitrate alone.

Table 8.1 Assumptions within the national modelling of PM_{10} and $PM_{2.5}$.

	Sulphate factor	**Nitrate factor**	**Residual PM ($\mu g\ m^{-3}$)**
PM_{10} TEOM	1.354	1.000	6.75
PM_{10} gravimetric[a]	1.760	1.300	8.80
$PM_{2.5}$ TEOM	1.354	0.333	4.50
$PM_{2.5}$ gravimetric	1.354	1.500	4.50

[a] Assumed to be TEOM * 1.3.

8.2.2 Urban models

763. The urban models described in this section are used to calculate the pollutant concentrations at high resolution in an urban area. The impacts of road sources are fully resolved so that spatial variations in concentration across and near roads can be calculated.

8.2.2.1 ERG PM_{10} model and predictions

764. For predictions of PM_{10} in London, ERG has utilised the measurements available in London to derive a receptor-based PM_{10} model. Regression analyses of NO_X and PM_{10} have been extended to include $PM_{2.5}$, and thus PM_{10} has been divided into four components: $PM_{2.5}$ that is related to NO_X:$PM_{2.5}\ f(NO_X)$; $PM_{2.5}$ that is not related to NO_X:$PM_{2.5} <> f(NO_X)$; coarse particles that are related to NO_X:$PM_C\ f(NO_X)$; and coarse particles that are not related to NO_X:$PM_C <> f(NO_X)$. The NO_X and non-NO_X components can be combined to produce PM_{10} that is related to NO_X:$PM_{10}\ f(NO_X)$ and PM_{10} that is not related to NO_X:$PM_{10}<>f(NO_X)$.

Where: $PM_{10}\ f(NO_X) = PM_{2.5}\ f(NO_X) + PM_C\ f(NO_X)$
$PM_{10}<>f(NO_X) = PM_{2.5}<>f(NO_X)+ PM_C<>f(NO_X)$.

Total PM_{10} can be calculated:
$$PM_{10} = PM_{10}\ f(NO_X) + PM_{2.5}<>f(NO_X)+ PM_C<>f(NO_X).$$

765. In the formulation above it is assumed that the particle fractions that are related to NO_X are related to primary emissions. Fine particles, which cannot be related to NO_X, are assumed to comprise secondary aerosol. Coarse particles that are not related to NO_X might be expected to consist of wind-blown dusts and other natural particles. The particle fraction definitions include a coarse component that can be related to concentrations of NO_X.

766. Annual mean values of NO_X, PM_{10} and $PM_{2.5}$ were calculated, at monthly intervals, at sites with co-located measurements. Annual means were chosen to eliminate the effects of seasonality and a minimum of 75% data capture was required to ensure that the measurement was representative of the year. Relationships between annual mean NO_X, and PM_{10} and between NO_X and $PM_{2.5}$ were established using linear regression. Each monthly analysis used annual mean measurements from all site types, including kerbside, roadside, urban background, suburban and rural locations. A maximum of 22 sites have been used for the

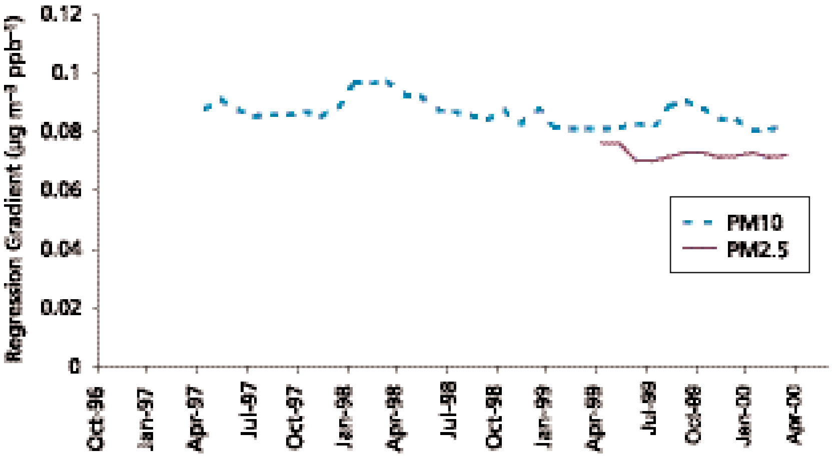
PM_{10} analysis and a maximum of five sites for $PM_{2.5}$. Linear regressions have been derived of the form:

$$PM_{10}\ (\mu g\ m^{-3}) = A\ NO_X\ (ppb) + B\ (\mu g\ m^{-3})$$
$$PM_{2.5}\ (\mu g\ m^{-3}) = C\ NO_X\ (ppb) + D\ (\mu g\ m^{-3})$$

767. The gradient (A and C) enables the calculation of the PM_{10} and $PM_{2.5}$ that is related to NO_X, for example, combustion-related particles. The intercept (B and D) gives the annual mean of the PM_{10} and $PM_{2.5}$ that is not related to NO_X, which would include the secondary aerosol, for example. The time series of gradients and intercepts is shown in Figures 8.7 and 8.8. Only results from annual mean NO_X against PM_{10} regressions with $r^2 > 0.8$ have been used to produce the time series of A and B. Fewer sites were available for $PM_{2.5}$, therefore NO_X against $PM_{2.5}$ regressions with $r^2 > 0.75$ and more than three sites have been used to produce the time series of C and D. The variation in the gradients shown in Figure 8.7 is reasonably consistent over the period of analysis, as indicated by the small standard deviation (σ) in Equation 1. For PM_{10}, for example, over 5 years the gradient has not varied much above or below 0.086. Assuming that the underlying ratios of PM_{10} and $PM_{2.5}$ to NO_X are constant, the mean gradients can be used to derive the overall relationships which are applicable across the range of site types:

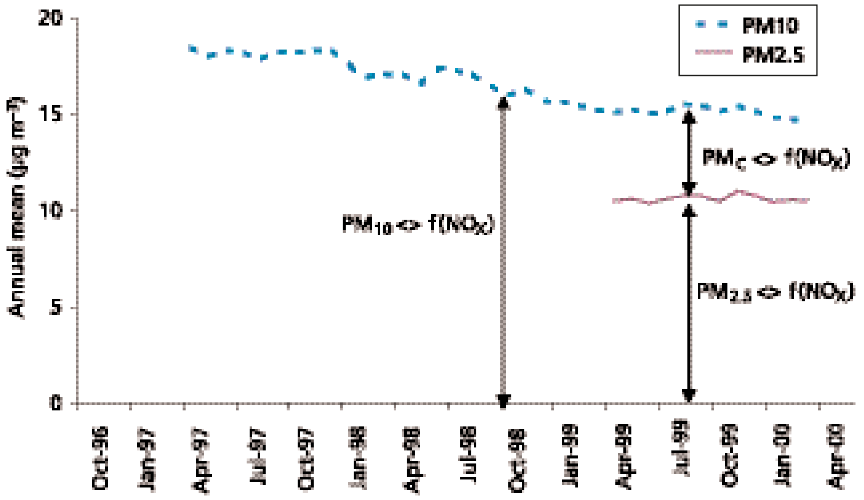
$$PM_{10}f(NO_X) = 0.086\ [NO_X]\ (\sigma = 0.005)\ (\mu g\ m^{-3}\ ppb^{-1}) \qquad (1)$$
$$PM_{2.5}f(NO_X) = 0.072\ [NO_X]\ (\sigma = 0.002)\ (\mu g\ m^{-3}\ ppb^{-1}) \qquad (2)$$

Figure 8.7 Regression gradients for PM_{10} and $PM_{2.5}$.



768. The annual mean concentrations of the two components that are independent of NO_X can be seen in Figure 8.8. From this analysis of annual means it appears that PM_{10}<>$f(NO_X)$ has declined considerably during the 4-year period (unlike the gradients shown in Figure 8.7), from around 18.4 $\mu g\ m^{-3}$ to 15.3 $\mu g\ m^{-3}$. This decline might be due to the differing meteorology in each year. However, it is more likely that the decline is due to reductions in the emissions that result in the formation of secondary aerosol arising from measures being taken on the European scale. The concentration of PM_{10}<>$f(NO_X)$ has remained relatively stable since January 1999, which is reflected in the stability of $PM_{2.5}$<>$f(NO_X)$ and PM_C<>$f(NO_X)$. The annual mean PM_C<>$f(NO_X)$ has remained relatively constant at around 5 $\mu g\ m^{-3}$.

Figure 8.8 Regression intercepts for PM_{10} and $PM_{2.5}$.



769. A time series of daily means for each of the particulate components was calculated by applying the factors derived from the regression in Equations 1 and 2 to the daily mean NO_X, PM_{10} and $PM_{2.5}$ measured at each of the sites with co-located measurements. This approach allowed the calculation of the NO_X-dependent components. The non-NO_X dependent components were then calculated by subtraction.

770. To enable predictions at other locations, several key assumptions were made. The daily mean NO_X independent components across the area were assumed to be independent of site location. This assumption is probably reasonable for the fraction that includes the secondary aerosol because the sources of these components are distant compared to the size of the London region. The fraction most related to wind-blown dusts, for example, will be influenced by very local conditions, but is also assumed to be constant. The daily mean variation in concentrations of NO_X is required to derive $PM_{2.5}$ $f(NO_X)$ and PM_C $f(NO_X)$. A time series of daily mean NO_X concentrations can be derived in several ways, depending on the application. For site-specific assessments, use was made of actual NO_X measurements. For the prediction of PM_{10} at locations where NO_X concentrations are not measured (or for predictions into the future), an alternative approach is required that uses a dispersion model to predict the NO_X concentrations. The fraction of $PM_{2.5}$ related to secondary aerosol is reduced in line with that estimated by APEG (1999), that is, a 30% reduction between 1996 and 2010. Further information on the general approach adopted by ERG to modelling concentrations of primary pollutants is considered in the AQEG NO2 report (AQEG, 2004).

771. The $PM_C<>f(NO_X)$ component is relatively small, around one-third of the overall $PM_{10}<>f(NO_X)$, with an annual mean of ~5 $\mu g\ m^{-3}$. Accurate determination of the daily mean $PM_C<>f(NO_X)$ is, therefore, difficult. The $PM_C<>f(NO_X)$ shows no overall trend during 1996–1999. The $PM_{2.5}<>f(NO_X)$ derived from separate NO_X and $PM_{2.5}$ measurement sites shows very good agreement, confirming that $PM_{2.5}<>f(NO_X)$ is largely invariant over the London region, even on a daily basis.

772. Figures 8.9 and 8.10 show the predictions of PM_{10} at monitoring sites in London for different years.

Figure 8.9 Site-specific annual mean PM_{10} predictions: ERG model compared with measured data.

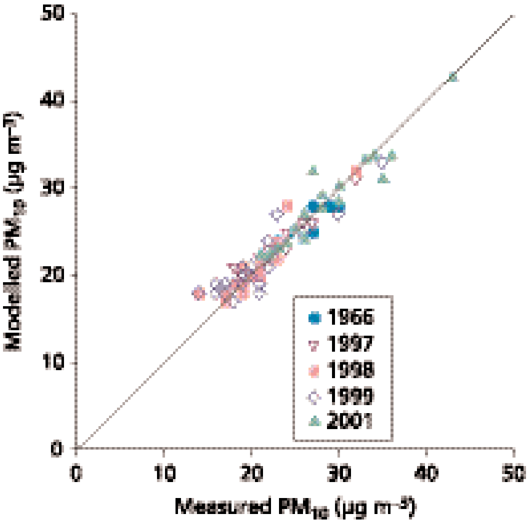
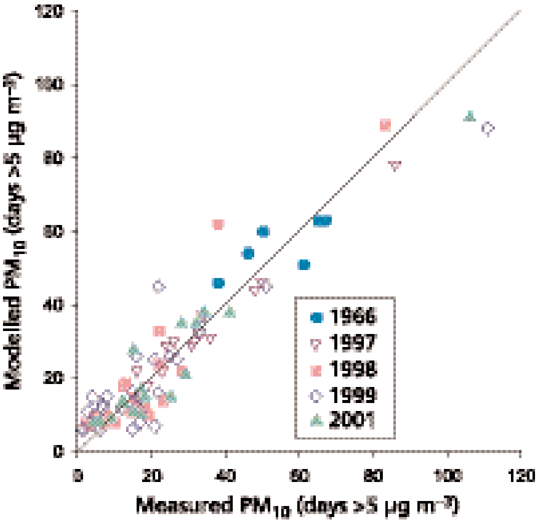


Figure 8.10 Number of days exceeding 50 µg m^{-3} (TEOM * 1.3): ERG model compared with measured data.



8.2.2.2 ADMS-Urban

773. A description and discussion of ADMS-Urban and its application to the calculation of NO_X and NO_2 concentrations in urban areas with specific examples from London has recently been presented in AQEG's report on NO_2 (AQEG, 2004). In this section, only specific aspects of the model relevant to PM calculations will be described together with model validation and verification for PM. Further details may be found in Carruthers *et al*. (2002) and Blair *et al*. (2003). Total concentrations are summed from three components broadly similar to the split used in the Netcen model:

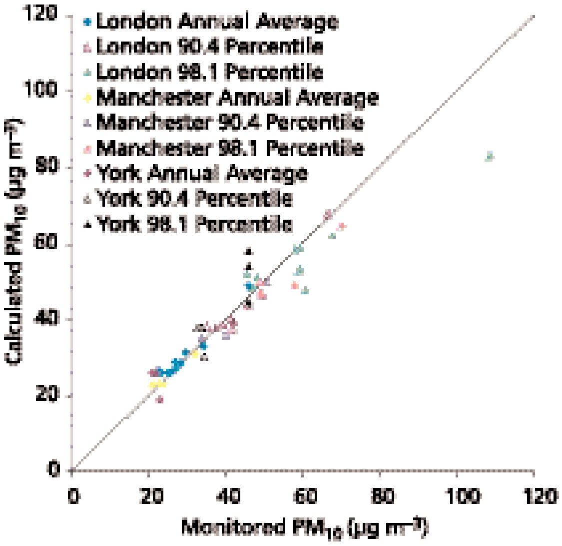
(i) Concentrations of PM arising from the primary emissions specifically accounted for in the emissions inventory for the urban area under consideration. These are calculated from the advection and dispersion algorithms within ADMS-Urban.

(ii) A background component representing secondary PM and PM from other sources (for example, sea salt) transported into the urban area. This component is estimated from measured rural PM concentrations, which – in the case of London – are Harwell and Rochester. Forward projections of the secondary contribution are based on EMEP model calculations; for a base year of 1999 meteorology, the secondary component is reduced by 33% by 2010.

(iii) 'Other' PM representing sources not accounted for in (i) and (ii). These are considered to be coarse PM (for example, non-exhaust traffic emissions, construction dust) arising from within the urban area for which emissions cannot be specified. For the case of PM_{10} this component is taken as a constant 5 µg m^{-3} (gravimetric) across London and is broadly consistent with the observed increment between rural and London background sites. A further coarse component is assumed to be included in the regional background contribution (ii), thus the total coarse contribution for PM_{10} (rural and urban) is 9.9 µg m^{-3} (gravimetric), which is similar to values used in the national model. For $PM_{2.5}$ the 'other' component is assumed to be 30% of that in PM_{10}, consistent with values measured by Harrison *et al*. (2003) and Harrison *et al*. (2004).

774. ADMS-Urban has been used to calculate PM_{10} concentrations in cities both in the UK and elsewhere. Examples of comparisons of model calculations with data from automatic monitoring sites are presented for London, Manchester and York in Figure 8.11: this shows both annual means and percentiles corresponding to 35 exceedences (90.4th percentile) and 7 exceedences (98.1th percentile). The London calculations are for 1999 and utilize the 1999 LAEI for 1999 met data from Heathrow Airport. For Manchester the calculations used are the Greater Manchester Inventory for 2002 and meteorological data from Manchester Ringway Airport for that year. Background data are from Manchester South, Wirral Tranmere, Preston and Ladybower AURN sites. For York the calculations used York City Council's inventory for the year 1999 and meteorological data from Leeds for that year. The background data are from Ladybower and Harwell.

Figure 8.11 PM_{10} Comparison of ADMS-Urban and monitored data.



775. The data for London are also presented in Table 8.2. Considering both the figure 8.11 and the table 8.2, the annual average and daily average concentrations corresponding to 35 exceedences and 7 exceedences are well within the requirements of the EU directive. This is despite the fact that the daily average concentration exceeded 7 times is slightly underestimated, as might be expected because no account is taken of the daily variation of the 'other' component ((iii) above) or of other unspecified intermittent sources. Table 8.3 shows the statistics for $PM_{2.5}$ concentrations calculated at two sites within London for 1999. As with PM_{10}, the comparisons are for TEOM-based measurements with a factor of 1.3 for both the London sites and background sites at Rochester and Harwell. Although there is no established factor for this conversion, the average value derived from three sites where TEOMs and gravimetric monitors are calculated for the period 2000–2002 is 1.3 (Johnson *et al.*, 2004).

Table 8.2 1999 Monitored and calculated PM_{10} concentrations ($\mu g\ m^{-3}$) in London, Manchester and York.

	Annual average		**90.4th percentile of daily average**		**98.1th percentile of daily average**		**Standard deviation**	
	Monitored	**Calculated**	**Monitored**	**Calculated**	**Monitored**	**Calculated**	**Monitored**	**Calculated**
A3	30	31	46	44	59	59	16	13
Camden	34	33	50	47	68	62	18	15
Harringey	28	29	42	41	59	53	15	12
Marylebone Road	46	49	67	68	109	83	42	20
Sutton Roadside	26	26	40	39	48	51	14	12
Bury Roadside	32	31	51	50	70	65	20	18
Fishergate	22	26	34	38	46	54	12	16
Clifton	21	26	33	38	46	58	11	16
Roadside mean	**30**	**31**	**45**	**46**	**63**	**61**	**19**	**15**
Bexley	25	26	42	38	61	48	16	12
Bloomsbury	28	29	42	41	59	54	14	13
Brent	23	26	38	38	46	52	14	12
Eltham	23	27	35	38	47	48	12	12
Hillingdon	27	29	42	41	58	59	17	13
North Kensington	27	27	42	39	59	53	15	12
Bolton	21	23	34	35	49	50	13	13
Salford Eccles	24	23	40	36	58	49	15	13
Stockport Shaw Heath	23	23	34	35	49	47	14	12
Bootham	23	19	35	30	46	45	12	12
Background mean	**24**	**25**	**38**	**37**	**53**	**51**	**14**	**12**
Overall statistic	**27**	**28**	**41**	**41**	**58**	**55**	**16**	**14**

Table 8.3 Monitored and calculated 1999 $PM_{2.5}$ concentrations (µg m^{-3}) in London.

	Annual average		90.4th percentile of daily average		98.1th percentile of daily average		Standard deviation	
	Monitored	**Calculated**	**Monitored**	**Calculated**	**Monitored**	**Calculated**	**Monitored**	**Calculated**
Marylebone Road	29	37	42	52	53	67	10	11
Bloomsbury	19	19	30	30	43	41	8	7

8.2.3 Regional models

8.2.3.1 The EMEP model

776. The EMEP Centre West in Oslo has developed a new Eulerian model to simulate the dispersal, atmospheric chemistry and deposition of pollutants across Europe. This will be used in forthcoming assessments under the CLRTAP of the UNECE and by the EC within the CAFE programme in revision of national emission ceilings to combat effects of acidification, eutrophication, excess tropospheric ozone and fine particulate concentrations. This single unified model will replace the former Lagrangian models used for acidifying pollutants (SO_X, NO_X and NH_X) and ozone during development of the Gothenburg protocol and National Emissions Ceilings Directive. The estimated concentrations of secondary inorganic aerosol concentrations are substantially different from those of the Lagrangian model available at the time of the APEG report.

777. The model has 20 vertical levels using σ (pressure level) coordinates, with a lowest layer of ~92 m, and the top of the domain at 100 hPa. The horizontal grid is a polar stereographic projection, true at 60° north, with grid-cells approximately 50 km x 50 km. Meteorological data with a 3-h resolution are used from PARLAM-PS, a dedicated version of the High Resolution Limited Area Model (HIRLAM) weather prediction model. National emissions are distributed across the EMEP grid cells and distributed vertically according to SNAP sector. The treatment of boundary conditions, atmospheric chemistry and deposition processes are described in detail in an EMEP report (EMEP, 2003a).

778. The EMEP model domain encompasses the whole of Europe with grid cells of the order of 50 km x 50 km. However, comparison here is limited to a smaller region over the UK.

8.2.3.2 The FRAME model

779. Fine Resolution Atmospheric Multi-pollutant Exchange (FRAME) is a Lagrangian model developed by CEH Edinburgh and the University of Edinburgh (Fournier *et al*., 2004). It models SO_X, NO_X and NH_X, calculating sulphur and nitrogen deposition to different ecosystem types, and also SO_4, NO_3 and NH_4 aerosol concentrations.

8.2.3.2.1 Model domain

780. The domain of the model covers the British Isles with a grid resolution of 5 km and grid dimensions of 172 x 244 km. Input gas and aerosol concentrations at the edge of the model domain are calculated using FRAME-EUROPE, a larger scale European simulation which was developed from TERN to run a statistical model over the entirety of Europe with a 150-km scale resolution. FRAME is a Lagrangian model that simulates an air column moving along straight-line trajectories. The atmosphere is divided into 33 separate layers extending from the ground to an altitude of 2500 m. Layer thicknesses vary from 1 m at the surface to 100 m at the top of domain. A year-specific wind rose is used to give the appropriate weighting to directional deposition and concentration for calculation of total deposition and average concentration. Diffusion of gaseous and particulate species in the vertical is calculated using K-theory eddy diffusivity and solved with a Finite Volume Method. The vertical diffusivity Kz has a linearly increasing value up to a specified height Hz and then remains constant (K_{max}) to the top of the boundary layer. During day time Hz is taken as 200 m and K_{max} is a function of the boundary layer depth and the geostrophic wind speed. At night time these values depend on the Pasquill stability class.

8.2.3.2.2 Emissions

781. Emissions of NH_3 are estimated for each 5-km grid square using national data for farm animal numbers (cattle, poultry, pigs and sheep) as well as fertiliser application, crops and non-agricultural emissions (including traffic and contributions from human sources and wild animals). The ammonia emissions inventory is described in Dragosits *et al*. (1998). NH_3 is emitted into the lowest layer. Emissions of SO_2 and NO_X are from the NAEI for the UK. For SO_2 ~80% of 1996 and 1999 emissions from the UK are associated with a small number of strong point source emissions. For NO_X, point source emissions account for ~25% of the total. Point source emissions of SO_2 and NO_X are treated individually with a plume rise model that uses stack height, temperature and exit velocity to calculate an 'effective emissions height'.

8.2.3.2.3 Chemistry

782. The chemical scheme in FRAME is similar to that employed in the EMEP Lagrangian model. The prognostic chemical variables calculated in FRAME are: NH_3, NO, NO_2, HNO_3, PAN, SO_2 and H_2SO_4 as well as NH_4^+, NO_3^- and SO_4^- aerosol. The gas phase reactions for oxidised nitrogen include photolytic dissociation of NO_2, oxidation of NO by ozone, formation of peroxyacetyl nitrate (PAN) and the creation of nitric acid by reaction with the OH free radical. NH_4NO_3 aerosol is formed by the equilibrium reaction between HNO_3 and NH_3. A second category of large nitrate aerosol is present and simulates the deposition of nitric acid on to soil dust or marine aerosol. The formation of H_2SO_4 by gas phase oxidation of SO_2 is represented by a predefined oxidation rate. H_2SO_4 then reacts with NH_3 to form ammonium sulphate aerosol. The aqueous phase reactions considered in the model include the oxidation of S(IV) by O_3, H_2O_2 and the metal catalysed reaction with O_2.

8.2.3.2.4 Wet deposition

783. The model employs a constant drizzle approach using precipitation rates calculated from a climatological map of average annual precipitation for the British Isles. Wet deposition of chemical species is calculated using scavenging coefficients based on those used in the EMEP model. An enhanced washout rate is assumed over hill areas due to the scavenging of cloud droplets by the seeder-feeder effect. The washout rate for the orographic component of rainfall is assumed to be twice that calculated for the non-orographic component.

8.2.3.2.5 Dry deposition

784. Dry deposition of SO_2, NO_2 and NH_3 is calculated individually for five different land categories (arable, forest, moorland, grassland and urban). For NH_3, deposition is calculated individually at each grid square using a canopy resistance model (Fournier *et al*., 2004). The deposition velocity is generated from the sums of the aerodynamic resistance, the laminar boundary layer resistance and the surface resistance. Dry deposition of SO_2 and NO_2 is calculated using maps of deposition velocity derived by the CEH 'big leaf' model (Smith *et al*., 2003), which takes account of surface properties as well as the geographical and altitudinal variation of wind-speed.

8.2.3.2.6 Meteorology

785. The depth of the boundary layer in FRAME is calculated using a mixed boundary layer model with constant potential temperature capped by an inversion layer with a discontinuity in potential temperature. Solar irradiance is calculated as a function of latitude, time of the year and time of the day. At nighttime, a single fixed value is used for the boundary layer depth according to Pasquill stability class and surface windspeed. The wind rose employed in FRAME uses 6-hourly operational radiosonde data from the stations of Stornoway, Hillsborough, Camborne and Valentia spanning a 10-year period (1991–2000) to establish the frequency and harmonic mean wind speed as a function of direction for the British Isles.

8.2.3.3 The NAME model

786. The Met Office's Lagrangian dispersion model, NAME simulates the release of atmospheric pollutants by releasing air parcels into a three-dimensional model atmosphere driven by three-dimensional meteorological data from the Met Office's numerical weather prediction model, the Unified Model (UM). The air parcels are carried passively by the UM wind fields and random walk techniques are used to simulate the local turbulent dispersion. Detailed descriptions of the NAME model can be found in Physick and Maryon (1995) and Ryall and Maryon (1998).

787. In order to calculate the species concentrations required for the chemistry scheme, a three-dimensional grid is constructed over the model domain. The model is driven using EMEP[1] 2001 emissions data on a 50-km grid, hence this resolution grid was also used in the horizontal for the chemistry calculations. Five vertical layers are used (0–100 m, 100–300 m, 300–800 m, 800–5000 m and

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[1] See http://www.emep.int/

5000–20000 m) and the extent of the model domain for the sulphate nitrate calculations presented in this chapter is 10W to 16E and 44S to 60N. The model emits 13 primary species, including seven VOCs, that are then scaled to represent the full VOC emission inventory. Details of the chemistry scheme can be found in Redington *et al*. (2001).

788. Figure 8.12 presents the model and observed daily particulate sulphate concentrations for Bridge Place in London, Strathvaich in Scotland and Yarner Wood, Devon. An evaluation of model performance is included in Table 8.4 below. The comparison between the model and observations for daily particulate sulphate shows that the model generally predicts peaks when these occur but that the quantitative argument is not good. The correlation coefficients are low. Generally, over all three sites the model over-predicts in January, underpredicts in March and overpredicts in December. The overall performance at Yarner Wood is degraded by the large erroneous peak predicted in December but not seen in the observations. Strathvaich and London show a negative bias that could be removed if a background particulate sulphate contribution was included for air masses advected across the North Atlantic Ocean.

Table 8.4 An evaluation of NAME model performance against the observed daily mean particulate sulphate observations at Stratvaich, London and Yarner Wood during 1996.

Site	Correlation	Bias (μg S m^{-3})	NMSE	Percentage within a factor of 2
Strathvaich	0.36	–0.25	3.23	21.3
Yarner Wood	0.32	1.20	2.93	40.1
London	0.35	–1.94	1.30	45.8

789. Focussing attention on 2002, Figure 8.13 gives a comparison of model and observed daily mean particulate sulphate concentrations for Yarner Wood, Devon for the year 2002. The wintertime underprediction is not seen as strongly as in Figure 8.12 but now the summertime levels appear to be overpredicted. However, during all seasons the model predicts accurately the onset and finish of the regional pollution episodes. This gives some confidence in the ability of the model to represent the long-range transport of particulate sulphate and hence its country attribution.

Figure 8.12 Modelled and observed particulate sulphate in µg SO_4 m^{-3} plotted upwards and model particulate nitrate in µg NO_3 m^{-3} plotted downwards for Strathvaich, Yarner Wood and London for 1996.

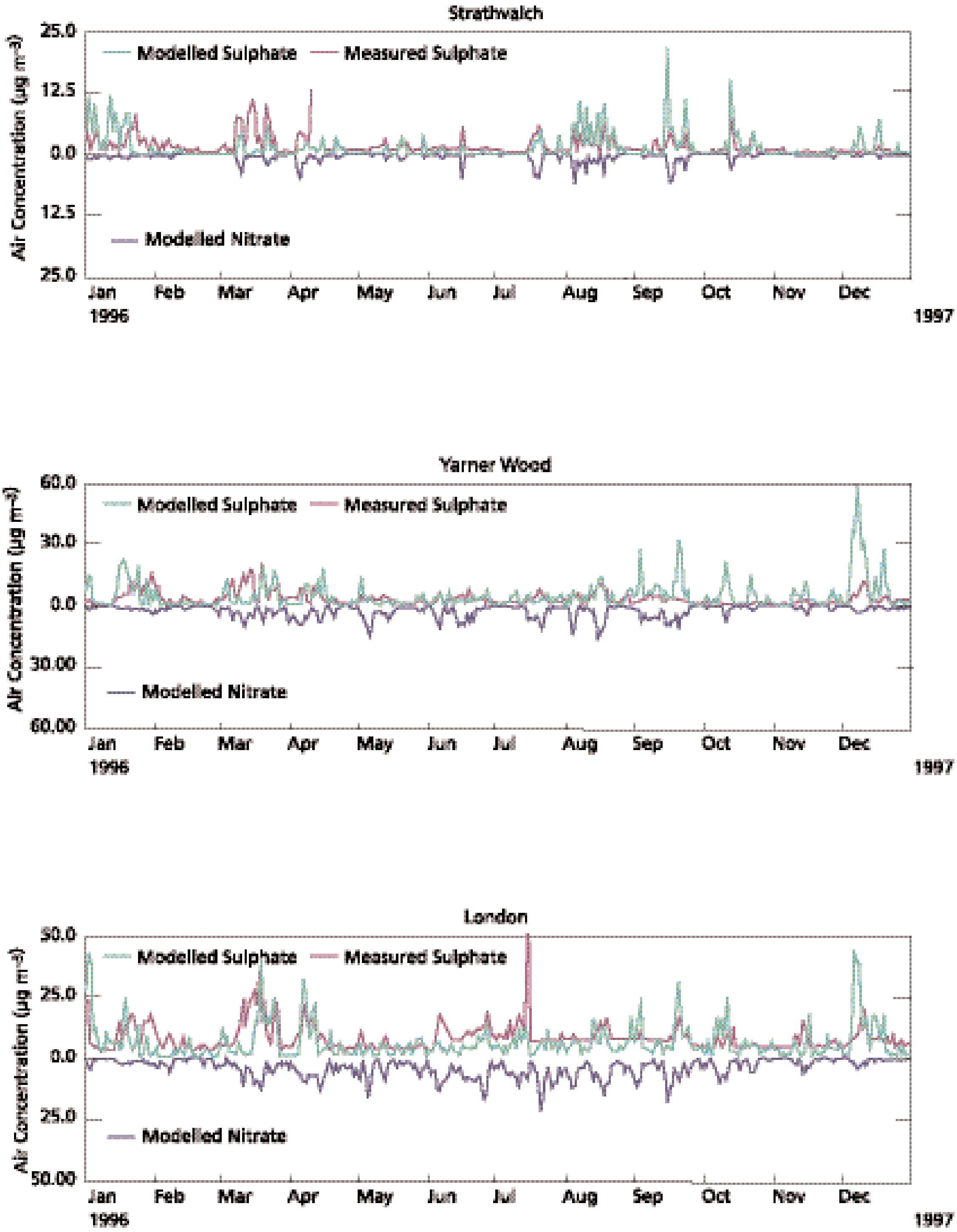
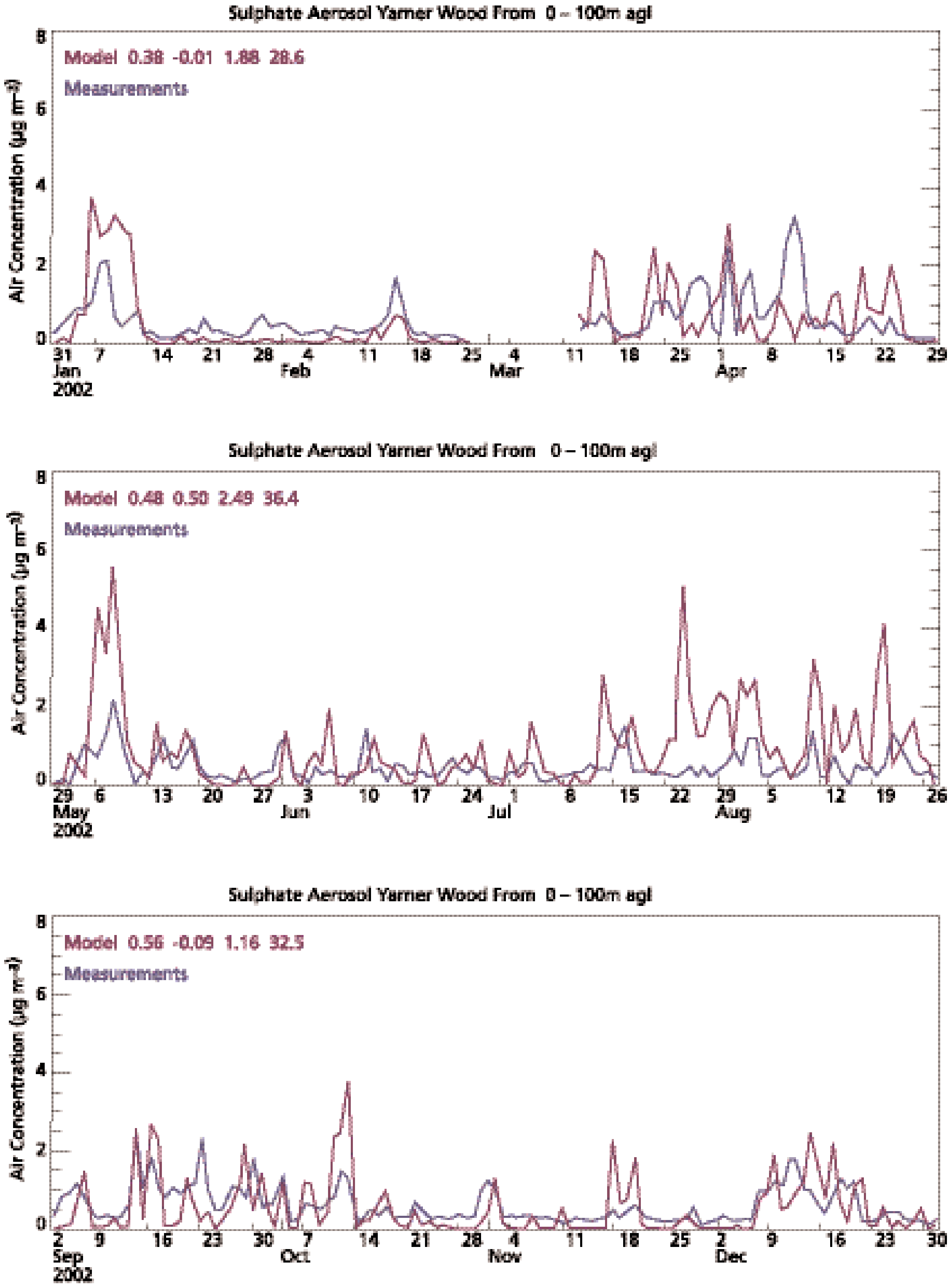


Figure 8.13 A comparison of model and observed daily mean particulate sulphate levels for Yarner Wood during 2002.



8.2.4 Other models

790. The models discussed in the previous section have been used extensively for policy development in the UK. Of course many other models have been developed for similar purposes and for specific applications, for example DMRB, Caline, ADMS-Roads and OSPM for road traffic emissions. Features of some of these models are presented in Table 8.5 and other models are also discussed in the AQEG NO_2 Report (AQEG, 2004).

8.2.4.1 Models for sources of PM due to wind generation, non-exhaust traffic emissions

791. Although many sources of PM deriving from products of combustion can be estimated from the appropriate emission factors and amount of fuel combusted, speed of process and so on, sources of PM due to mechanical processes such as wind generation and non-exhaust traffic emissions are inherently more difficult to quantify depending for instance on wind speed and turbulence, occurrence of recent rain, road material, weight of vehicles. Theoretical models based on a threshold saltation velocity for the different particle sizes have been developed for uptake of sand particles and the generation of equilibrium shapes for sand dunes (Bagnold, 1941). These ideas have also been applied to other types of surfaces and to stockpiles (Nalparis *et al*., 1993); however, there are no practically useful models based on well-founded theoretical ideas for resuspension of sources of particles from construction sites. Models that do exist are almost exclusively empirical and based on the AP-42 emission factors of the USEPA estimated from field studies. However, these are generally not appropriate for UK conditions.

8.2.4.2 Advanced transport models including aerosol dynamics

792. In the most advanced atmospheric transport models describing aerosol development, the aerosol size distribution is represented by a number of modes – for example, ultrafine, fine and coarse – and the amount of material in each mode arising as a result of processes – such as emission, condensation and coagulation – is tracked. The chemical composition of the particles is represented by a number of components, such as sulphates, nitrates, organic carbon, mineral dust, sea salt and particle number. It is not practicable in a transport model to represent in detail the multi-mode interactions between each chemical component. Instead, all the components within each mode are assumed to have the same size distribution and the same chemical composition (internally mixed aerosols). The particles within each mode are transported in a similar manner to the gaseous species within atmospheric models and similarly subject to dry and wet deposition appropriate to particles.

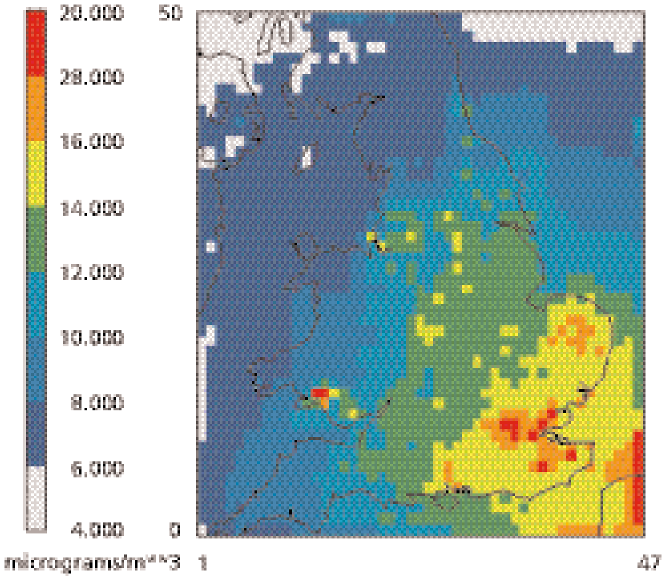
793. Even with simplified assumptions, modelling is exceedingly complex and dependent on having good knowledge of processes, such as gas-particle conversion, speciated emission rates and the aggregation of particles. The main benefit of the comprehensive approach, if it can be demonstrated to perform well, is that the particle composition within different size ranges could be predicted. Hence results could be interpreted in terms of the part of the aerosol mass or number which is thought to have an adverse health effect.

794. The approach has been developed as part of the EMEP programme, in order to assess particle concentrations on a regional scale in Europe (Tsyro, 2002). Secondary organic aerosols and natural dust are not yet included. The aerosol dynamics within the Models-3 modelling system (Byun and Ching, 1999) allows for the interaction of ultrafine particles generated by nucleation and direct emission with aged, fine (accumulation mode) particles. The chemical species include sulphates, nitrates, ammonium, water, manmade and biogenic organic carbon, elemental carbon and other unspecified material of manmade origin. Successful performance of the model clearly depends on a reliable speciated source inventory, which is a challenge in its own right. These models are undergoing continual development. At the present time they cannot be considered sufficiently well tested to be regarded as the basis of a reliable assessment method, not least because speciated aerosol measurements are only just becoming available to test them. Hence in this report simpler empirical approaches have been used to predict trends in particle concentration.

795. It should be recognised that in time these simpler methods may be challenged by the more detailed approaches. An illustration of the potential of comprehensive models is given in the study of Models-3 by Cocks *et al*. (2003), in which Models-3 was run for the whole of 1999 to predict regional acid deposition over England and Wales. The results included hourly concentration fields over the country throughout the year at a spatial resolution of 12 km. Although the purpose of the study was to determine acid deposition not particle concentration, as the comprehensive model is part of an integrated atmospheric modelling system, the 21 components constituting PM_{10} particles were automatically calculated as part of the model run.

796. The mean annual concentration of particulates is shown in Figure 8.14. The resolution means that comparisons can only be made with the few regional background monitoring sites available.

Figure 8.14 Annual mean concentration of total particulates calculated using Models-3.



797. To illustrate the kind of comparison that should be possible using comprehensive models, unscaled measured PM_{10} concentrations are compared with the calculated annual average obtained by summing all components (Figure 8.15).

798. The sites at Harwell, Narbeth and Rochester are rural sites within the national network. The remaining sites are rural sites operated by power generation companies as a check to determine concentrations in the neighbourhood of coal and oil-fired power stations. In principle the calculation includes contributions to the rural ground-level PM from power stations, but – as demonstrated elsewhere in this report – the power station contribution is likely to be a small fraction of the total. A strict comparison between measured and calculated concentrations should not be made, as the TEOM instrument is not thought to measure all components, the more volatile components being lost. In future it may be possible to perform more detailed comparisons component by component. This is illustrated in figures 8.16 and 8.17 for sulphate, using data from acid deposition sites in operation in 1999 and calculations from Models-3. The preliminary agreement looks encouraging. However, more research is needed to turn the comprehensive models into reliable, practical assessment tools.

Figure 8.15 Comparison between measured annual average PM_{10} concentration ($\mu g\ m^{-3}$, TEOM) and calculations from Models-3 (year).

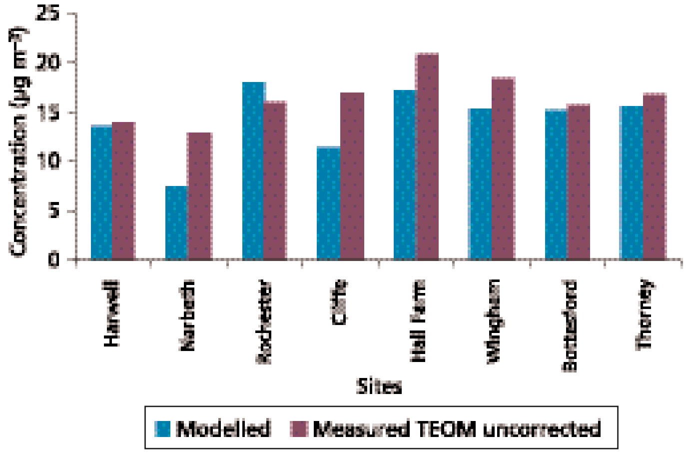
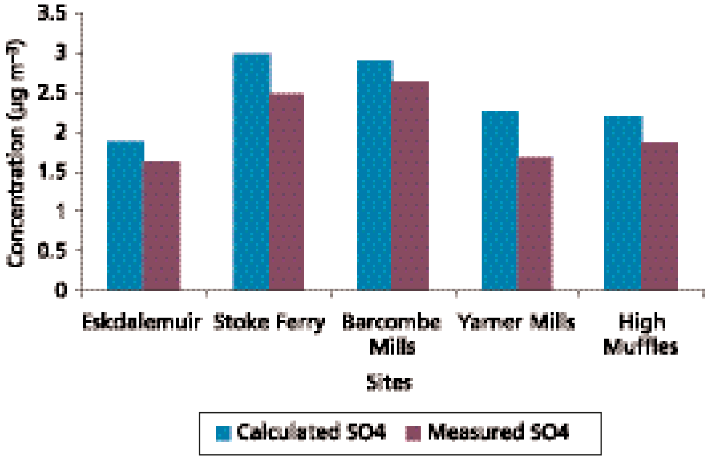


Figure 8.16 Comparison between 1999 measured annual average sulphate concentration and calculations from Models-3 (year).



8.3 Comparison of model features

799. The main features of the key models presented in Section 8.2 and some other models are shown in the Table 8.5. This includes and contrasts some basic details of the models and also the methodologies they employ to take account of the coarse component and so on.

8.3.1 Netcen, ERG and ADMS-Urban models

800. These models have some common themes and major differences, which the model intercomparison exercise has discussed in detail (Carruthers *et al*., 2002). Figure 8.17, which compares annual means calculated by each of the models, suggests that the models show broadly similar performance at monitoring sites. However, scatter plots (Figure 8.18) comparing calculated concentrations on road segments calculated by ERG and ADMS show quite different predictions and also illustrate the different treatment of the background (the ERG model exhibits a clear minimum value).

801. Broadly, in the Netcen and ERG models the emphasis, especially for predictions at monitoring sites, is on the use of monitoring data. The performance of these models, therefore, depends critically on the availability of good data coverage, now the case in London for which the ERG approach was developed, but perhaps not in other parts of the UK. The methods do have the disadvantage that their predictions depend to a large extent on the measurement method employed; in addition there is some ambiguity as to what the PM related to NO_X is and in particular how much non-exhaust traffic emission this includes. In ADMS-Urban the emphasis is on modelling the primary emissions in the area of interest; these calculated concentrations are not dependent on the measurement method. The ADMS method is not able to assimilate data but uses broad comparisons with data to refine the physical parameters within the model (for example, surface roughness or minimum Monin Obukhov length). These parameters themselves are subject to uncertainty in urban areas. With regard to the background, Netcen and ADMS use similar general approach in using rural data (Netcen sulphate and nitrate data, ADMS measured PM data) and both approaches could use background data from other sources or employ regional model outputs. ERG derives the background from its analysis of background within the urban area.

Figure 8.17 Measured PM_{10} annual average concentration compared with predicted values ($\mu g\ m^{-3}$).

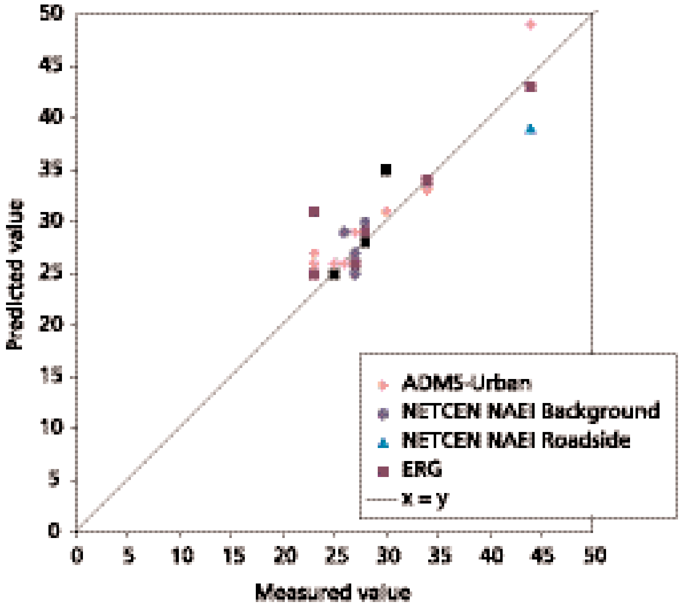


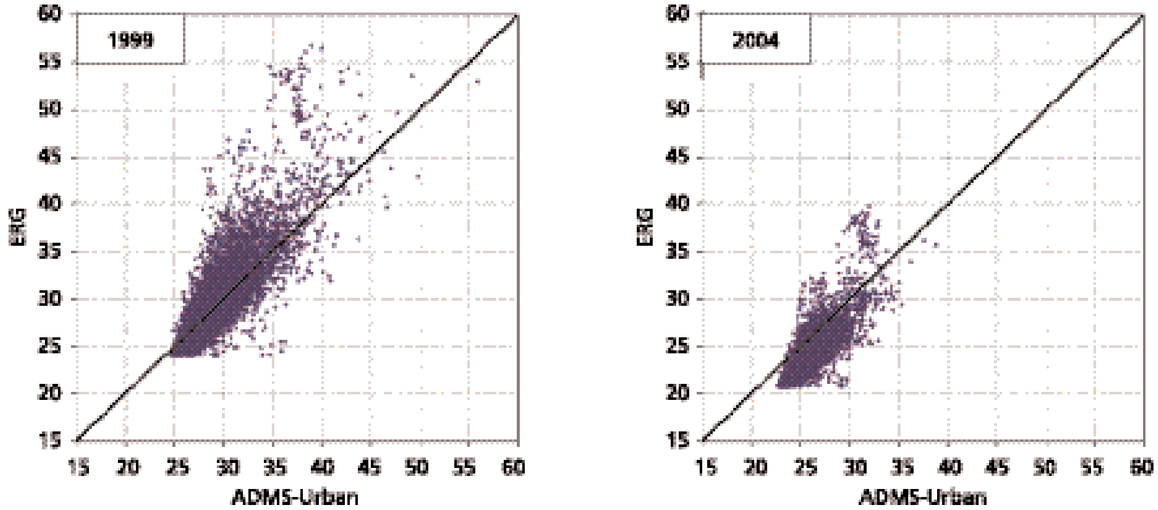
Table 8.5 Description of the main models and modelling techniques used for PM calculations in the UK.

Model	Basic description	Meteorology	Transport and dispersion	Chemistry for particulate formation	Spatial scale	Averaging times	Background, including	Treatment of 'other' projections contributions e.g. non-exhaust traffic emissions
Models used for urban and national assessment against AQ limit values and objectives.								
Netcen Stedman *et al* (2002)	National model for all emission types	Different statistical analyses for different years	Regression analysis based on ADMS 3 calculations and monitoring data for urban background; simple formulation for representative calculation on road segments	None	National to suburban scale representative concentrations for road segments	Annual	Based on measured sulphate and nitrate. Reductions in secondary based on Gottenberg protocol and National Emissions Ceilings Directive	'other' included as 8.8 μg m^{-3} gravimetric. Road contribution implicitly included through roadside only adjustment factor
ADMS-Urban McHugh *et al* (1997)	Models for stationary source emissions and/or road traffic emissions	Hourly sequential data from one site	Quasi Gaussian type model using h/M_{MO} parameterisation of boundary layer. Also includes trajectory model and canyon model based on OSPM approach	Simple sulphate chemistry	Urban down to local, including spatial variation at local street scale	Short term (mins) to annual averages	From rural monitoring sites. Projections based on EMEP calculations (33% reduction 1999-2010 for 1999 base year)	9.9 μg m^{-3} gravimetric for urban background (4.9 μg m^{-3} rural contribution, 5 μg m^{-3} urban increment). Additional roadside contribution only through emissions for PM_C

Model	Basic description	Meteorology	Transport and dispersion	Chemistry for particulate formation	Spatial scale	Averaging times	Background, including	Treatment of 'other' projections contributions e.g. non-exhaust traffic emissions
ERG Carslaw *et al* (2001)	All emission types; set-up for London	Different statistical analyses for different years	Regression analysis based on ADMS 3 calculations and monitoring data	None	Urban down to local scale including spatial variation at local street scale	Annual, daily	Derived as $PM_{2.5}$ not related to NO_X. Reduction of 30% between 1996 and 2010	PM_C derived through regression analysis as 50 µg m^{-3}.
Regional models								
EMEP	Mesoscale model for pollutant dispersion at European scale	Mesoscale model	Eulerian	Inorganic	50 km x 50 km grid cells	Short term/ annual	Modelled	Not considered
FRAME	Regional model for UK	Based on radiosonde data over 10 year period	Straight line trajectory model	Inorganic	5 km x 5 km grid cells	Annual	From FRAME-Europe with 150 km resolution	Not considered
NAME Ryall and Mayor (1998)	Mesoscale model for pollutant dispersion	Mesoscale model	Lagrangian particle model	Inorganic	15 km x 15 km cells	Short term to annual	Models secondary component	Not considered
Road impact models								
DMRB (2003) Highways Agency (2001)	Screening model for road traffic emissions and air quality	Fixed, at $2 m/s^{-1}$ equally distributed from all directions	Incorporates a fixed empirically adjusted Pasquill stability category	None	Local	Annual	Uses specific constant value	Implicates through use of road type adjustment factor

Model	Basic description	Meteorology	Transport and dispersion	Chemistry for particulate formation	Spatial scale	Averaging times	Background, including	Treatment of ‘other’ projections contributions e.g. non-exhaust traffic emissions
CALINE Benson (1979) US-EPA www.EPA.gov /scram001	Road traffic emissions	Hourly sequential data	Gaussian model. Pasquill parameterisation of boundary layer	None	Local	Short term. commercial versions allow short term to annual		
ADMS-Roads	Road traffic emissions model	Hourly sequential data from one site	Quasi Gaussian type model using h/M_{MO} parameterisation of boundary layer. Canyon model based on OSPM approach	None	Local	Short term annual	Specified by user	Within background
OSPM	Street canyon model	Hourly sequential	Combination of plume and box model for concentration within street canyon	None	Canyon	Short term to annual	Specified by user	Within background
CAR	Road traffic emissions		Gaussian model for road network		Street scale	Short term to annual	Specified by user	Within background

Figure 8.18 A comparison of ADMS and ERG of PM_{10} concentrations on road segments in London in 1999 and 2004.



8.3.2 Air pollution models for road traffic

802. An ability to model roadside concentrations is crucial to the management and assessment of roadside PM. Current practical models are generally based on a Gaussian-type dispersion of the pollutants away from the source. At its simplest, the screening model in the Design Manual for Roads and Bridges (DMRB) uses an average dispersion curve to calculate annual mean concentrations downwind of a line source. More detailed models take account of dispersion on a finer time frame, usually of 1-h, which allows for the effect of varying windspeed and direction and atmospheric turbulence, with the annual mean or 24-h means being derived by summing the individual hourly concentrations. Although the models have a broadly similar basis there can still be quite significant differences in detail. For example Figure 8.19 shows hourly average concentration from CALINE and ADMS-Roads for a range of stability conditions and two different wind directions along and perpendicular to a 10-m road. Note that maximum concentration occurs for the wind parallel to the road for CALINE but for perpendicular flow for ADMS-Roads.

803. Recent work using the dispersion model used to derive the DMRB algorithms (Boulter *et al.*, 2003; Figure 8.20) has suggested that dispersion near roads seems to be dependent on road type in ways not generally accounted for in the models. The basic model was overpredicting for rural motorways and underpredicting for non-motorway urban roads. Further investigation showed that the model performance was related to traffic flow and speed, overpredicting more at higher flows and higher speeds. This analysis has led to a calibration relationship based on Annual average daily traffic (AADT) flow (Figure 8.21), which is now applied to the raw DMRB model output. For a given emission rate this results in a fivefold increase in concentration going from very high flows on open motorways to roads with lower flows and lower speeds in towns. Independent work by Netcen (Stedman *et al.*, 2004) has shown a similar behaviour in their empirical model, which relates emission rates (g m^{-1} s^{-1}) to measured roadside concentrations (Figure 8.22). Their work suggests a sevenfold difference between open motorways with high flows and roads with low flows in town centres.

Figure 8.19 Roadside concentrations calculated by ADMS-Roads and CALINE 4; emission rate 0.0171 g km^{-1} s^{-1}. (a) wind perpendicular to road; (b) wind parallel to road. Results shown for three stability conditions: B, unstable; D, neutral; G, very stable.

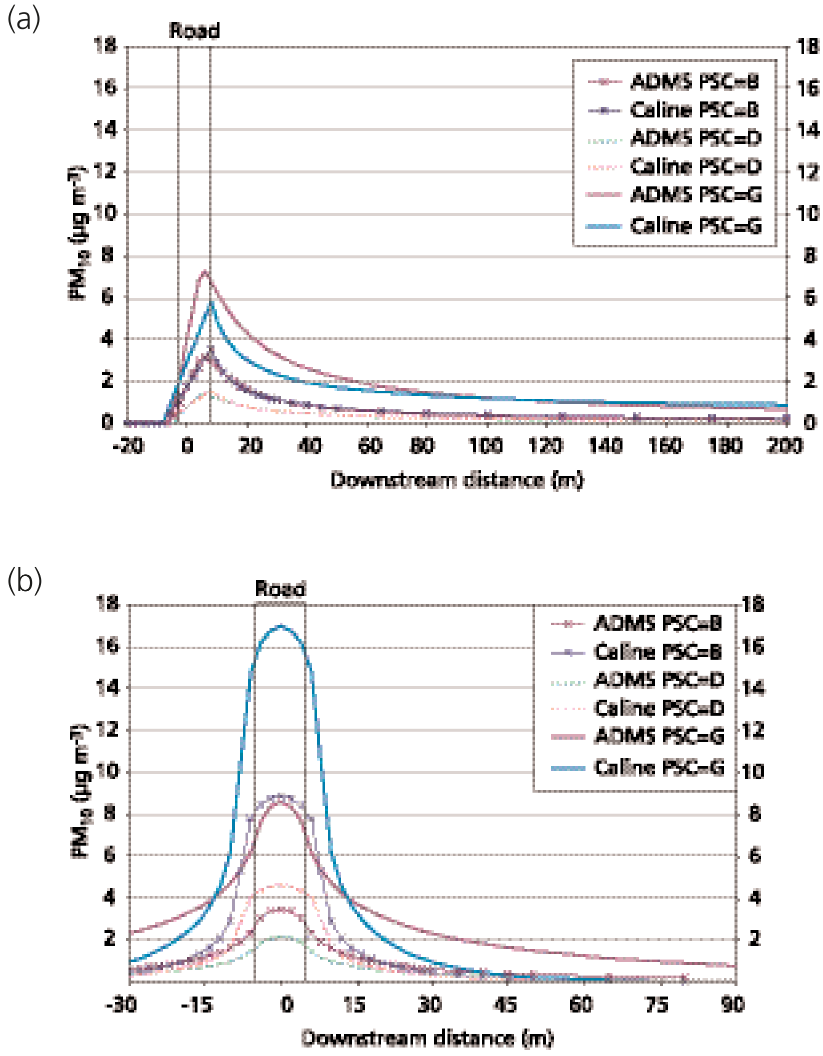
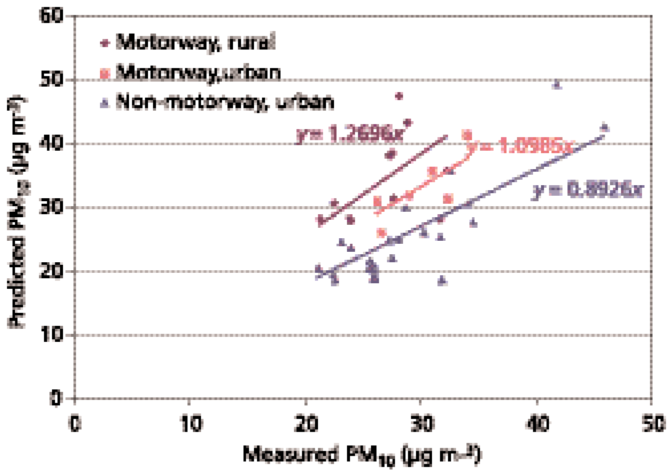


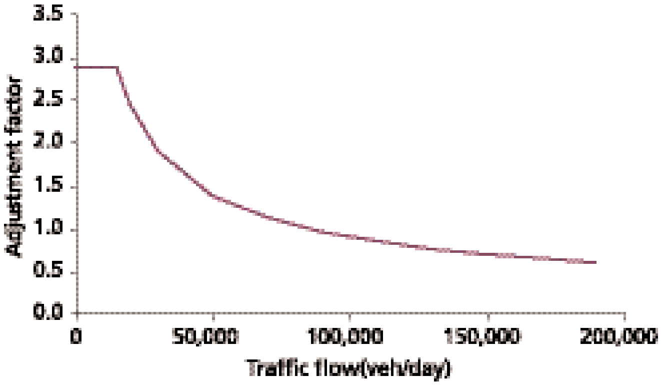
Figure 8.20 Relationship between raw DMRB predictions of annual mean PM_{10} versus measured values as a function of road type (Boulter *et al.*, 2003).



804. Some model verification of more detailed models, where model output is verified with local data from local authority studies for review and assessment purposes, have shown similar patterns (Laxen, 2004). For example CAL3QHC, a variant of the USEPA model CALINE3, appears to perform well for motorways and rural roads (for which it was originally validated), but underpredicts the road contribution significantly, typically by a factor of 4 to 8 in urban settings, where traffic is slower moving and more likely to be congested.

805. Vehicle emissions are sensitive to vehicle speed/vehicle type and so on and may not be well specified; however, there are also many reasons why dispersion is likely to differ significantly in different types of road. These include differing effects of traffic-induced turbulence both mechanically generated by the vehicles and induced by the buoyancy of the exhaust (di Sabatino *et al*., 2003); the impact of surface roughness changes between rural and urban sites (that is, the impact of buildings and other local feature changes), which will change both mean flow and turbulence and hence dispersion; the presence of street canyons; and the impact of the height of vehicle exhaust (for example, exhausts of HGVs are often elevated, whereas bus emissions are close to ground level, and the relative numbers of vehicles of different types will vary with location). In Figure 8.23 ADMS has been used to model the sensitivity of the annual mean concentrations at roadside to some of these different parameters – it is seen that the concentrations vary with all the parameters considered but are very sensitive to the initial vertical mixing height. Thus a specification of how this parameter varies according to vehicle type, vehicle speed and so on is important for improved treatment of sensitivity to road type by dispersion models because currently, in routine calculations, this parameter is held constant in models such as ADMS-Roads and CALINE.

Figure 8.21 Adjustment factors for raw DMRB predictions of annual mean PM_{10} as a function of daily traffic flow (as applied in v1.02 of the DMRB) (Boulter *et al*., 2003).



8.4 Model outputs and comparisons

806 This section includes model output firstly for the national and urban scale model and secondly for the regional models. Included are source apportionment calculations both PM_{10} and $PM_{2.5}$; urban, national and regional maps of concentrations for comparison with the air quality limit values for PM_{10}; maps for $PM_{2.5}$; and regional maps of the inorganic components of the secondary particulates. In addition there are summary tables of areas of exceedence and so on for the national and urban models. Where available, different model outputs are presented for comparison purposes.

Figure 8.22 Adjustment factors to apply to vehicle emissions as a function of daily traffic flow (Stedman *et al.*, 2003).

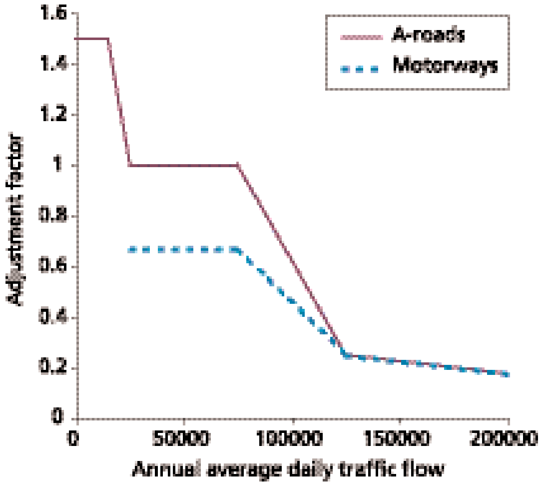
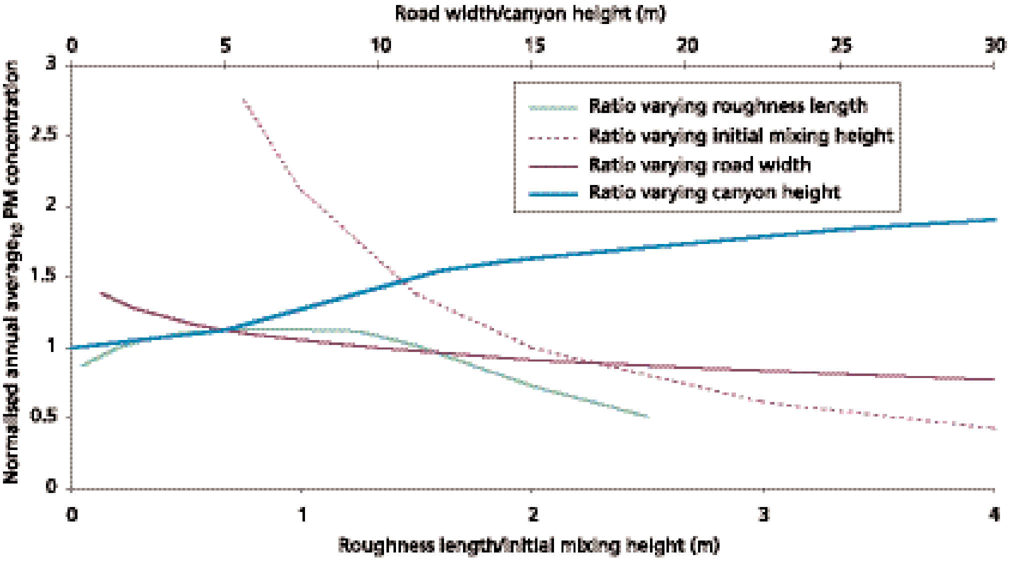


Figure 8.23 Normalised maximum annual average PM_{10} concentration varying with canyon height, road width, roughness length and initial mixing height. (The curves were calculated using ADMS-Roads.)



8.4.1 High resolution models up to national scale

8.4.1.1 Site-specific source apportionment

807 Tables 8.6–8.9 present calculations from the APEG source apportionment model (Section 8.2) corresponding to monitoring site across the UK and additional calculations are presented for ADMS-Urban and the ERG model in London (percentages only). Similar tables are presented for $PM_{2.5}$ in Tables 8.10–8.13. Contour maps of the source apportionment of the different components of the traffic fleet in London are shown in Figure 8.24.

808 Modelled PM_{10} concentrations for the current year show that the traffic contribution dominates at roadside locations, but makes a relatively small contribution at urban background and suburban sites. For all sites, there is also a stationary source contribution and a substantial contribution from the regional

Table 8.6 PM_{10} µg m^{-3} source apportionment: concentration from APEG receptor model for 200 NAEI, 2002 measurement, base year.

	Traffic	Stationary	Sulphate	Nitrate	Residual	Total
Marylebone Road TEOM	14.4	3.5	2.7	1.6	12.1	34.3
Marylebone Road KFG	13.2	5.5	2.7	5.5	9.9	36.8
Marylebone Road PART	16.4	6.7	2.7	7.4	10.3	43.5
Haringey Roadside TEOM	4.4	2.1	2.7	1.2	10.6	21.0
M25 Staines TEOM	8.9	1.9	2.6	3.1	6.5	23.0
Bury Roadside TEOM	6.5	3.6	2.1	2.0	9.8	24.0
Harwell TEOM	0.7	0.9	2.5	1.1	7.6	12.8
Harwell KFG	2.4	2.9	2.5	5.0	5.7	18.5
Harwell PART	1.6	2.0	2.5	4.5	8.7	19.3
Rochester TEOM	—	—	—	—	—	—
London Bloomsbury TEOM	—	—	—	—	—	—
London North Kensington TEOM	1.8	2.8	2.7	2.2	10.0	19.4
London North Kensington PART	2.9	4.6	2.7	5.8	9.1	25.1
London Bexley TEOM	1.7	3.7	2.7	2.8	8.0	19.0
Thurrock TEOM	1.1	4.2	2.7	3.7	9.5	21.1
Thurrock KFG	2.0	6.7	2.7	9.0	9.7	30.1
Belfast Centre TEOM	1.4	4.6	1.4	3.0	7.1	17.5
Belfast Centre KFG	2.3	7.2	1.4	6.0	5.9	22.8
Belfast Centre PART	2.0	6.3	1.4	5.4	9.7	24.9
Glasgow Centre TEOM	1.8	2.7	1.4	2.9	6.7	15.5
Glasgow Centre KFG	3.2	4.7	1.4	6.2	2.9	18.4
Glasgow Centre PART	3.6	5.3	1.4	5.5	7.4	23.2
Port Talbot TEOM	—	—	—	—	—	—
Port Talbot KFG	—	—	—	—	—	—
Port Talbot PART	—	—	—	—	—	—
Birmingham Centre TEOM	1.1	3.1	2.4	0.8	9.2	16.7
Birmingham Centre PART	2.7	7.0	2.4	5.1	7.0	24.2
Birmingham Hodge Hill TEOM	1.1	2.3	2.5	0.5	8.7	15.0
Manchester Piccadilly TEOM	2.1	3.7	2.2	2.8	10.7	21.5
Manchester Piccadilly PART	3.2	5.4	2.2	6.5	12.8	30.1

background, including the secondary PM component. The 'residual' component also makes a very significant contribution to the total. By 2010, the traffic component is estimated to fall to about 50–75% of its current contribution, with the secondary component then dominating at non-roadside sites. However, the residual component is assumed to remain unchanged by 2010, and represents 50% or more of the total PM_{10} concentration at non-roadside locations. For $PM_{2.5}$, a broadly similar pattern emerges for both the current and future years, although the residual component is much lower than that assumed for PM_{10}.

Table 8.7 PM_{10} source apportionment expressed as percentages: comparison for sites in London of Netcen 2002 NAEI and ADMS Urban 1999 LAEI.

	Traffic (%)	Stationary (%)	Secondary (%)	Other (%)
Marylebone Road				
ADMS-Urban	50.1	2.7	27.4	19.8
Netcen TEOM	42.0	10.2	12.5	35.3
Netcen KFG	35.9	14.9	22.3	26.9
Netcen PART	37.7	15.4	23.2	23.7
Bloomsbury				
ADMS-Urban	13.5	4.7	47.5	34.3
Netcen TEOM	—	—	—	—
North Kensington				
ADMS-Urban	7.3	5.3	50.7	36.7
Netcen TEOM	9.2	14.4	25.1	51.3
Netcen PART	11.6	18.3	33.9	36.3

Table 8.8 PM_{10} source apportionment projection to 2010: APEG receptor model 2001 NAEI, 2002 measurement base year.

	Traffic	Stationary	Sulphate	Nitrate	Residual	Total
Marylebone Road TEOM	8.0	3.3	2.4	1.4	12.1	27.1
Marylebone Road KFG	7.3	5.2	2.4	4.6	9.9	29.5
Marylebone Road PART	9.1	6.4	2.4	6.2	10.3	34.4
Haringey Roadside TEOM	2.6	2.0	2.4	1.0	10.6	18.5
M25 Staines TEOM	8.9	1.9	2.6	3.1	6.5	23.0
Bury Roadside TEOM	3.6	3.3	1.9	1.7	9.8	20.4
Harwell TEOM	0.4	0.8	2.2	1.0	7.6	11.9
Harwell KFG	1.4	2.5	2.2	4.2	5.7	16.0
Harwell PART	0.9	1.7	2.2	3.8	8.7	17.3
Rochester TEOM	—	—	—	—	—	—

	Traffic	Stationary	Sulphate	Nitrate	Residual	Total
London Bloomsbury TEOM	—	—	—	—	—	—
London North Kensington TEOM	1.0	2.6	2.4	1.8	10.0	17.8
London North Kensington PART	1.7	4.2	2.4	4.9	9.1	22.3
London Bexley TEOM	1.0	3.5	2.4	2.4	8.0	17.3
Thurrock TEOM	0.6	4.0	2.4	3.1	9.5	19.5
Thurrock KFG	1.2	6.4	2.4	7.6	9.7	27.2
Belfast Centre TEOM	0.9	3.4	1.2	2.5	7.1	15.1
Belfast Centre KFG	1.4	5.2	1.2	5.0	5.9	18.7
Belfast Centre PART	1.2	4.5	1.2	4.6	9.7	21.3
Glasgow Centre TEOM	1.0	2.6	1.2	2.4	6.7	14.1
Glasgow Centre KFG	1.8	4.6	1.2	5.3	2.9	15.7
Glasgow Centre PART	2.0	5.2	1.2	4.6	7.4	20.5
Port Talbot TEOM	—	—	—	—	—	—
Port Talbot KFG	—	—	—	—	—	—
Port Talbot PART	—	—	—	—	—	—
Birmingham Centre TEOM	0.6	3.2	2.1	0.7	9.2	15.9
Birmingham Centre PART	1.5	7.2	2.1	4.3	7.0	22.1
Birmingham Hodge Hill TEOM	0.6	2.2	2.1	0.4	8.7	14.1
Manchester Piccadilly TEOM	1.2	3.4	1.9	2.4	10.7	19.7
Manchester Piccadilly PART	1.8	5.1	1.9	5.5	12.8	27.2

Table 8.9 PM_{10} source apportionment projection to 2010: comparison of Netcen NAEI and ADMS Urban LAEI PM_{10} source apportionment for 2010.

	Traffic (%)	Stationary (%)	Secondary (%)	Other (%)
Marylebone Road				
ADMS-Urban	27.8	3.6	32.9	35.7
Netcen TEOM	29.4	12.1	14.0	44.5
Netcen KFG	24.8	17.7	23.8	33.7
Netcen PART	26.5	18.6	25.0	29.9
Bloomsbury				
ADMS-Urban	5.4	4.9	43.0	45.8
Netcen TEOM	—	—	—	—
North Kensington				
ADMS-Urban	3.4	5.3	43.8	47.6
Netcen TEOM	5.6	14.6	23.6	56.2
Netcen PART	7.6	18.8	32.7	40.8

Table 8.10 $PM_{2.5}$ ($\mu g\ m^{-3}$) source apportionment: APEG receptor model 2001 NAEI, 2002 measurement base year.

	Traffic	Stationary	Sulphate	Nitrate	Residual	Total
Marylebone Road TEOM	9.8	1.7	2.7	1.2	5.6	21.0
Marylebone Road PART	9.2	3.6	2.7	4.2	5.4	25.0
M25 Staines	5.5	1.5	2.6	0.6	2.3	12.5
Harwell TEOM	1.1	1.2	2.5	0.3	5.0	10.0
Harwell PART	0.4	0.4	2.5	5.8	3.9	13.0
Rochester TEOM	0.6	2.1	2.7	2.0	3.5	11.0
Bloomsbury TEOM	1.7	2.6	2.7	1.5	5.5	14.0
London North Kensington PART	3.5	4.2	2.7	4.8	2.9	18.0
Belfast Centre PART	1.6	4.7	1.4	4.8	3.5	16.0
Glasgow Centre PART	3.3	4.3	1.4	0.8	4.2	14.0
Birmingham Centre PART	1.8	4.4	2.4	3.4	4.1	16.0
Birmingham Hodge Hill TEOM	1.0	1.8	2.4	0.9	5.9	12.0
Manchester Piccadilly PART	1.6	2.7	2.2	3.1	6.5	16.0

Table 8.11 $PM_{2.5}$ source apportionment: comparison of Netcen 2002 NAEI and ADMS-Urban 1999 LAEI $PM_{2.5}$ source apportionment.

	Traffic (%)	Stationary (%)	Secondary (%)	Other (%)
Marylebone Road				
ADMS-Urban	58.2	2.6	31.1	8.1
Netcen TEOM	46.7	8.1	18.6	26.7
Netcen PART	36.7	14.3	27.5	21.5
Bloomsbury				
ADMS-Urban	17.8	5.2	61.2	15.8
Netcen TEOM	12.1	18.6	30.0	39.3
North Kensington				
ADMS-Urban	9.7	5.9	67.0	17.4
Netcen PART	19.3	23.2	41.4	16.0

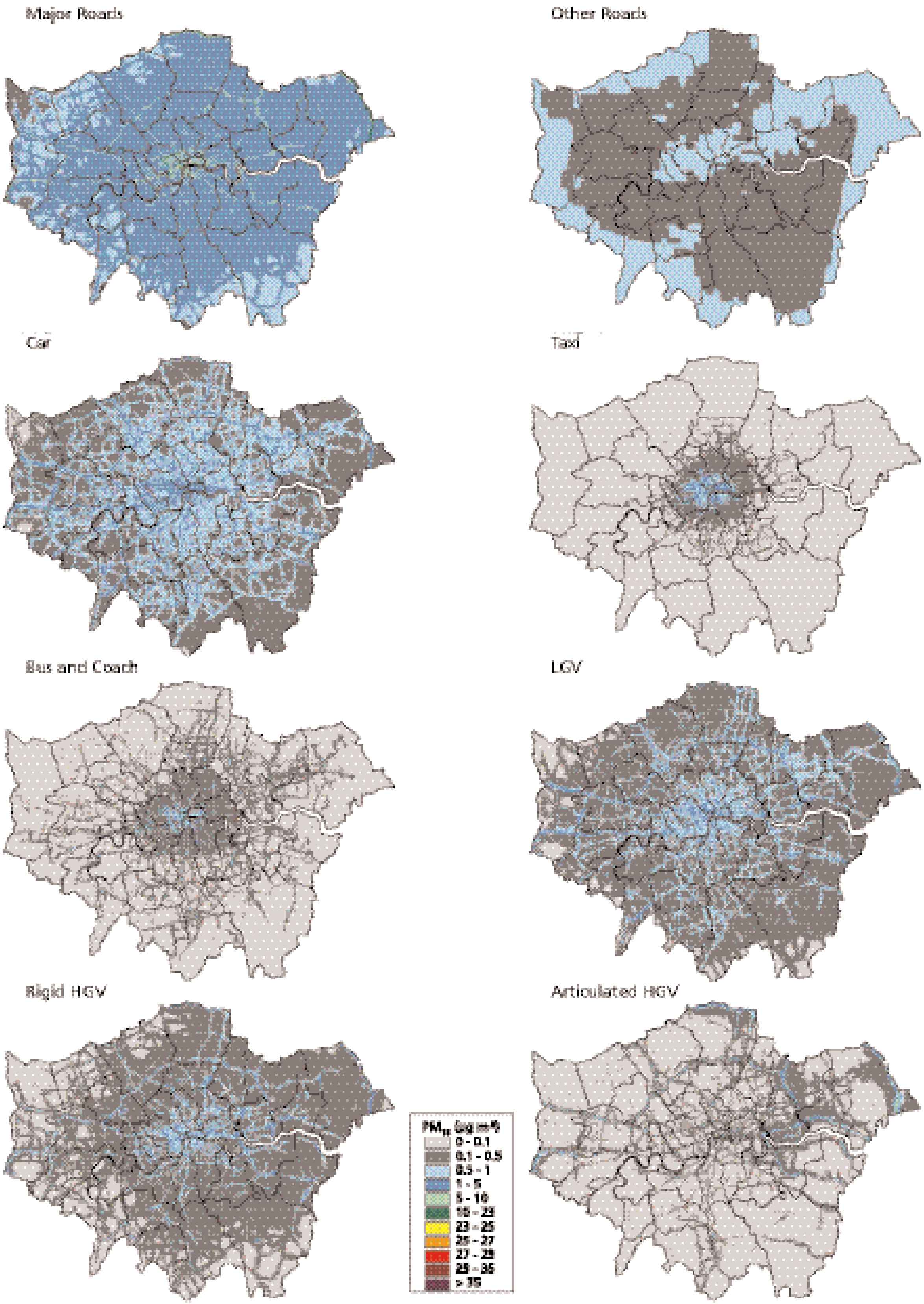
Table 8.12 $PM_{2.5}$ ($\mu g\ m^{-3}$) source apportionment projection to 2010: APEG receptor model 2001 NAEI, 2002 measurement base year projections to 2010.

	Traffic	Stationary	Sulphate	Nitrate	Residual	Total
Marylebone Road TEOM	5.4	1.6	2.4	1.0	5.6	16.0
Marylebone Road PART	5.1	3.3	2.4	3.5	5.4	19.7
M25 Staines	3.2	1.4	2.4	0.5	2.3	9.7
Harwell TEOM	0.6	1.1	2.2	0.2	5.0	9.1
Harwell PART	0.2	0.4	2.2	4.9	3.9	11.6
Rochester TEOM	0.4	2.0	2.4	1.7	3.5	10.0
Bloomsbury TEOM	0.9	2.5	2.4	1.3	5.5	12.6
London North Kensington PART	2.0	3.7	2.4	4.0	2.9	15.0
Belfast Centre PART	1.0	3.3	1.2	4.0	3.5	13.1
Glasgow Centre PART	1.8	4.2	1.2	0.6	4.2	12.2
Birmingham Centre PART	1.0	4.5	2.1	2.8	4.1	14.5
Birmingham Hodge Hill TEOM	0.6	1.8	2.1	0.7	5.9	11.1
Manchester Piccadilly PART	0.9	2.5	1.9	2.6	6.5	14.4

Table 8.13 $PM_{2.5}$ source apportionment projection to 2010: comparison of Netcen NAEI and ADMS Urban LAEI $PM_{2.5}$.

	Traffic (%)	Stationary (%)	Secondary (%)	Other (%)
Marylebone Road				
ADMS-Urban	36.8	3.7	42.8	16.6
Netcen TEOM	33.8	10.0	21.3	35.0
Netcen PART	25.9	16.8	29.9	27.4
Bloomsbury				
ADMS-Urban	7.8	5.5	62.4	24.2
Netcen TEOM	7.1	19.8	29.4	43.7
North Kensington				
ADMS-Urban	4.8	6.3	64.1	24.9
Netcen PART	13.3	24.7	42.7	19.3

Figure 8.24 London UK annual mean PM_{10} concentrations by traffic category, 1999.



809. The detailed source apportionment maps for the different components of the traffic fleet in London (Figure 8.24) illustrate the large contribution of cars, HGVs and LGVs to PM_{10} concentrations across London and the importance of buses and taxis in the central area.

8.4.1.2 Mapped concentrations of PM_{10}

810 National maps of annual mean PM_{10} calculated using the netcen mapping model are presented in Figures 8.25–8.30 for 2002, 2005 and 2010. Figures 8.25, 8.27, 8.29 present backgound concentrations and Figures 8.26, 8.28 and 8.30 the calculated roadside concentrations determined from summing the roadside increment and background for each major road segment. The background maps generally show higher levels over much of the most populated parts of England with generally lower levels in the North and Southwest of England and Scotland, Wales and Northern Ireland. The levels show only modest declines between 2002 and 2010. The roadside concentrations are as anticipated greatest in the main conurbations and more especially in London. The annual average limit value of 40 $\mu g\ m^{-3}$ is broadly achieved; however, there remain widespread exceedences focussed in urban areas of the Stage II indicative limit value of 20 $\mu g\ m^{-3}$ in 2010. ADMS-Urban also predicts widespread exceedences of the 2010 limit value in London (Figures 8.32–8.35); however, ERG predicts that the exceedences will mainly be confined to roadside with only small additional areas of exceedence (Figure 8.38).

811 London maps calculated using ADMS-Urban are also presented for daily averaged concentration exceeded 35 times in 2004 and seven times in 2010 (Figures 8.36 and 8.37). These suggest that some exceedence of the 2004 limit values is likely at roadsides but only at roadsides of major roads in 2010, except in adverse meteorological conditions. The main conclusions of the maps are summarised in Tables 8.14 and 8.15.

812 The maps have been analysed by GIS to provide a summary of data relevant to exceedence of the annual mean limit values in Tables 8.14 and 8.15. The tables show the number of road links, total road length and area and population within the area exceeding limit values for 2004/2005 and 2020 for each of the netcen, ERG and CERC models. The Netcen model results cover all regions of the UK, but results from the ERG and CERC models are for London only. Further discussion of the differences and simulations of the different model predictions for exceedence of both annual mean and daily average limit values is given in Chapter 9.

Figure 8.25 Estimated UK annual mean background PM_{10} concentration, 2002, in µg m^{-3} (gravimentric).

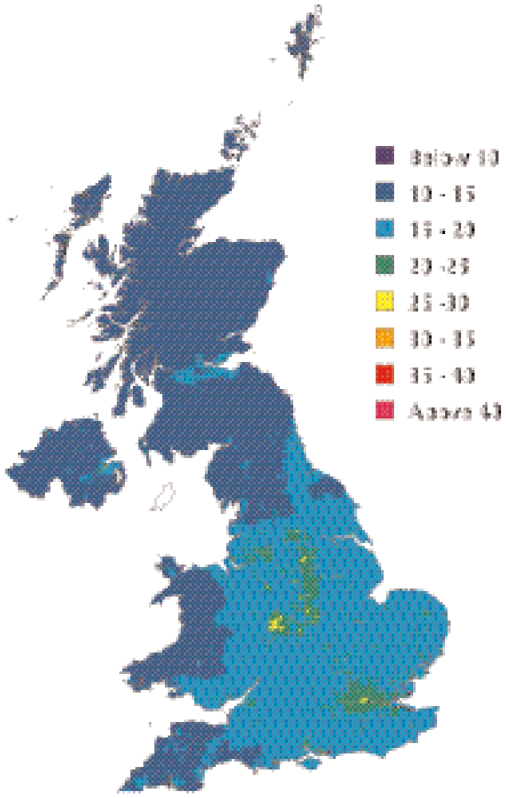


Figure 8.26 Estimated UK annual mean roadside PM_{10} concentrations for major built-up roads, 2002, in µg m^{-3} (gravimetric).

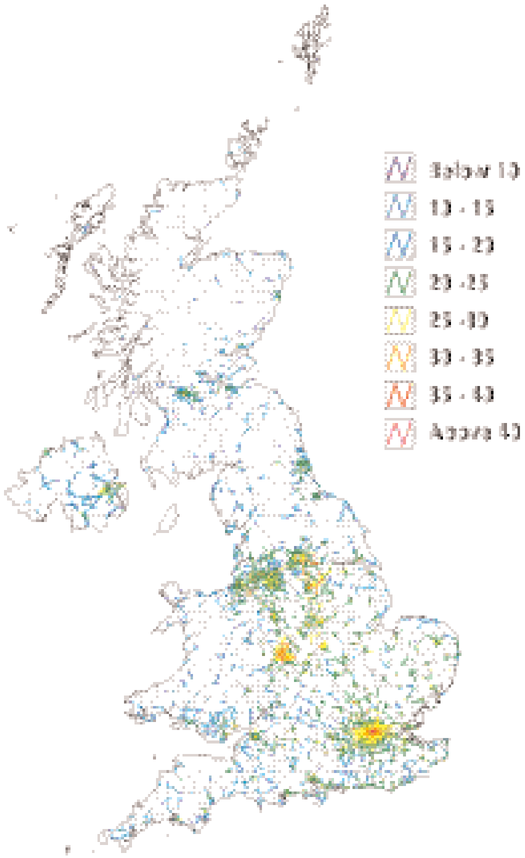


Figure 8.27 Estimated UK annual mean background PM_{10} concentration, 2005, in $\mu g\ m^{-3}$ (gravimentric).

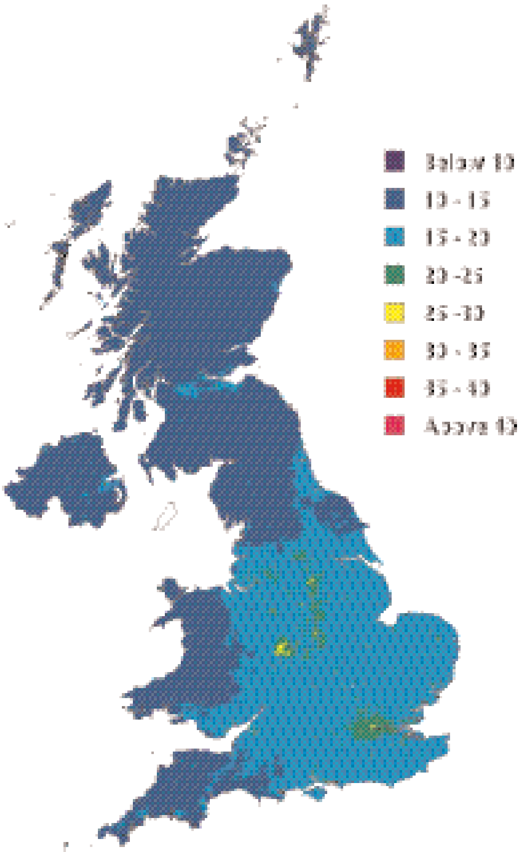


Figure 8.28 Estimated UK annual mean roadside PM_{10} concentrations for major built-up roads, 2005, in $\mu g\ m^{-3}$ (gravimetric).

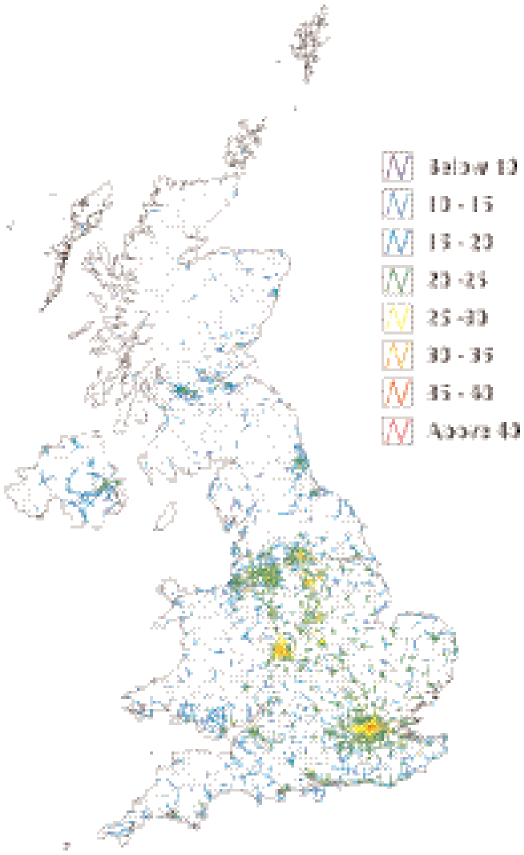


Figure 8.29 Estimated UK annual mean background PM_{10} concentration, 2010 in µg m^{-3} (gravimentric).

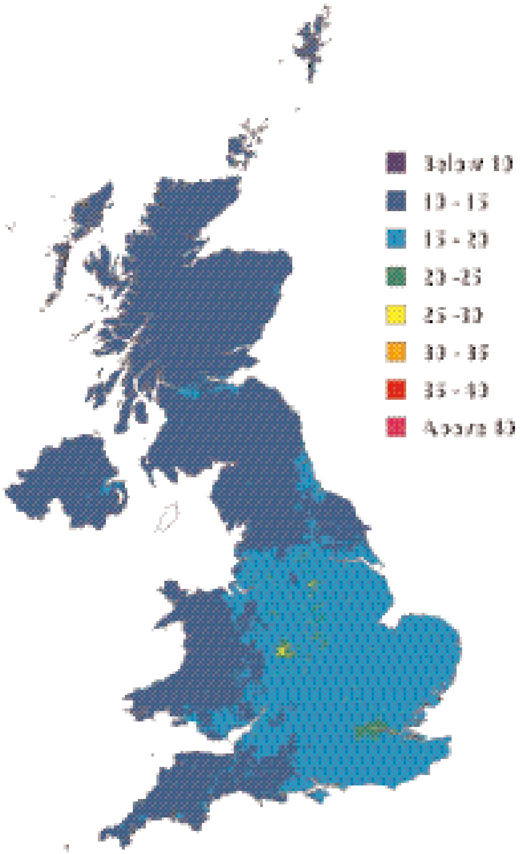


Figure 8.30 Estimated UK annual mean roadside PM_{10} concentrations for major built-up roads, 2010, in µg m^{-3} (gravimetric).

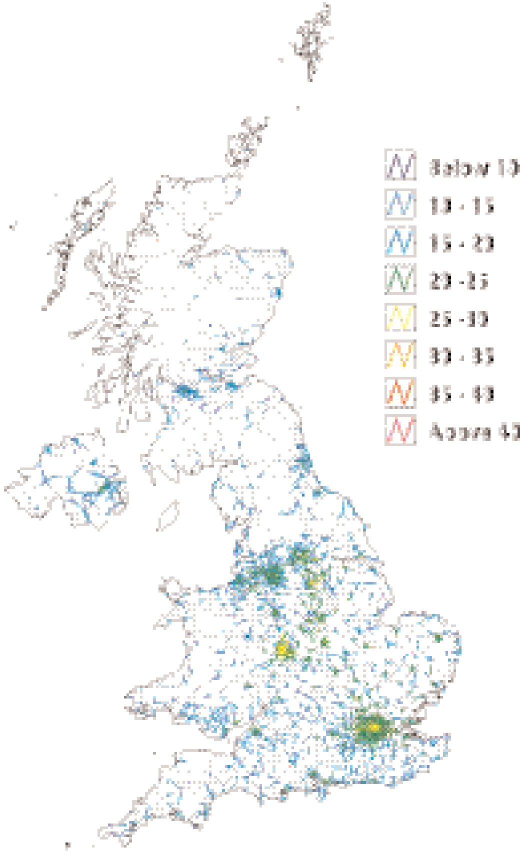


Figure 8.31 Annual average PM_{10} concentrations in London for 1999 calculated by ADMS-Urban (typical meteorology).

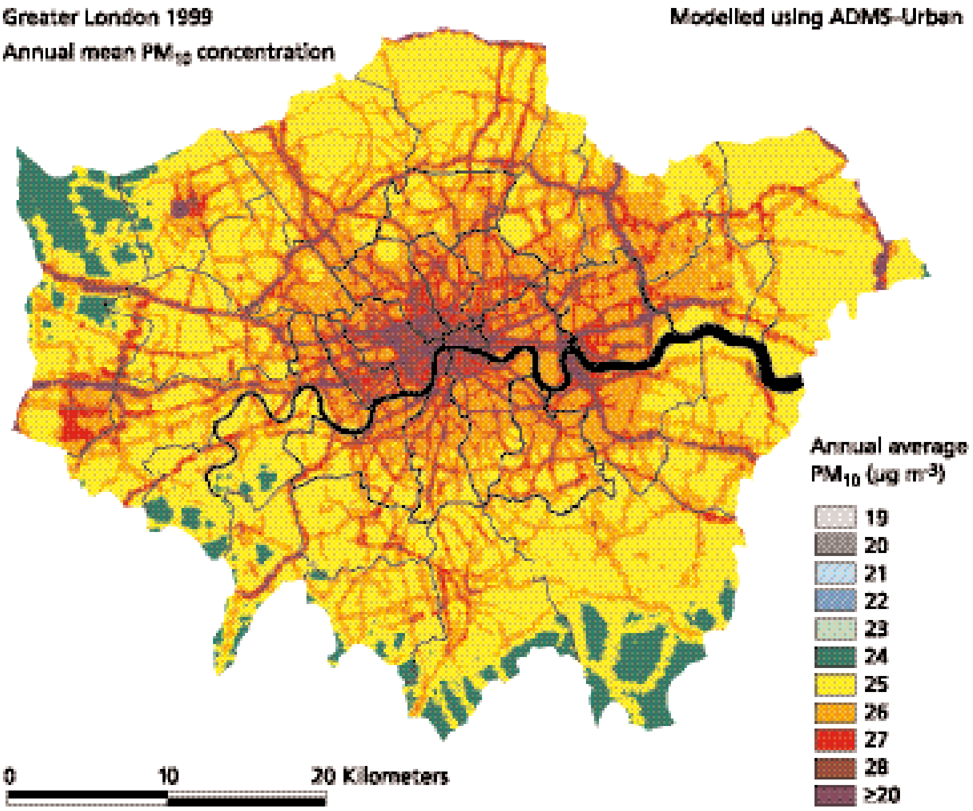


Figure 8.32 Annual average PM_{10} concentrations in London for 2004 calculated by ADMS-Urban (typical meteorology).

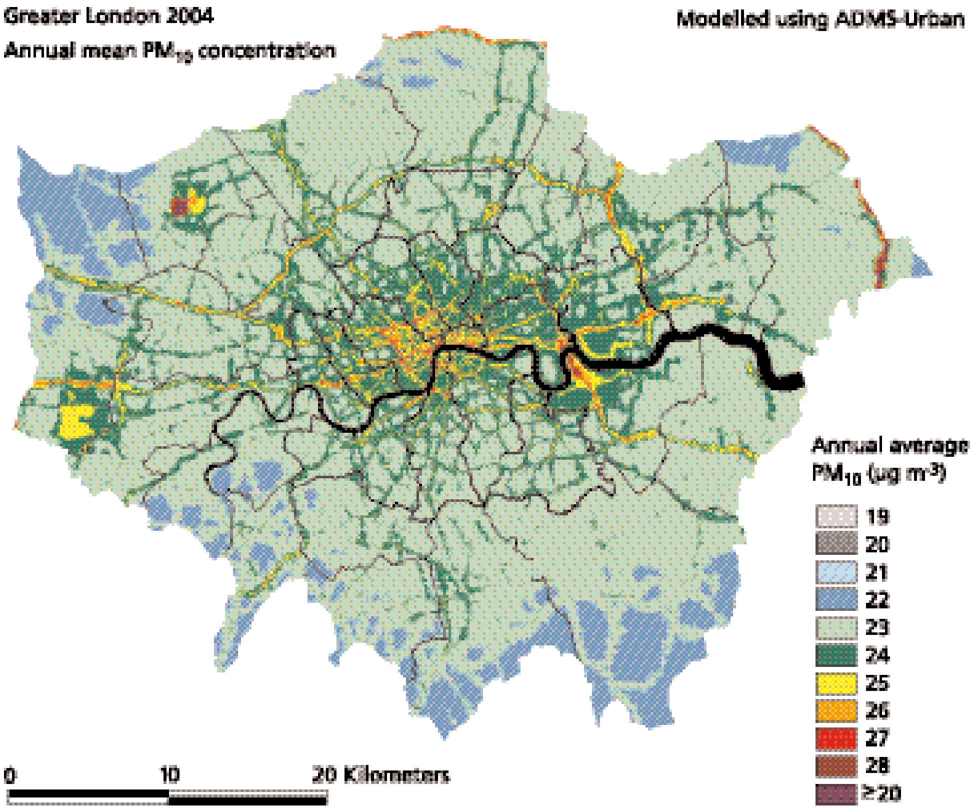


Figure 8.33 Annual average PM_{10} concentrations in London for 2010 calculated by ADMS-Urban (typical meteorology).

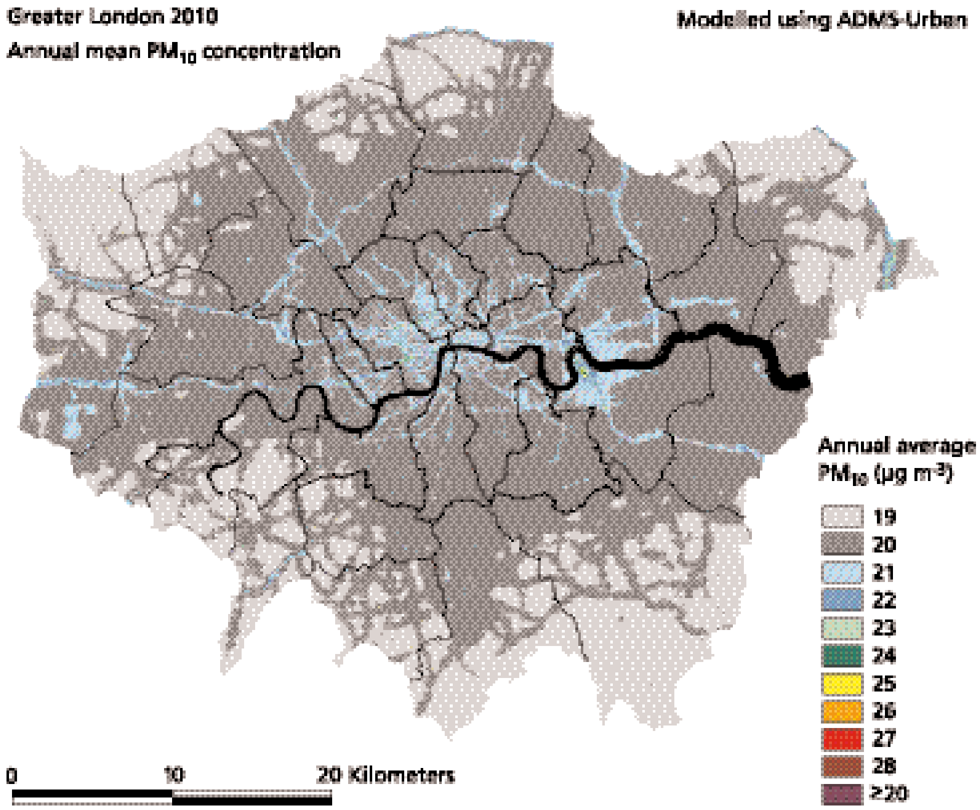


Figure 8.34 Annual average PM_{10} concentrations in London for 2010 calculated by ADMS-Urban (worst case meteorology).

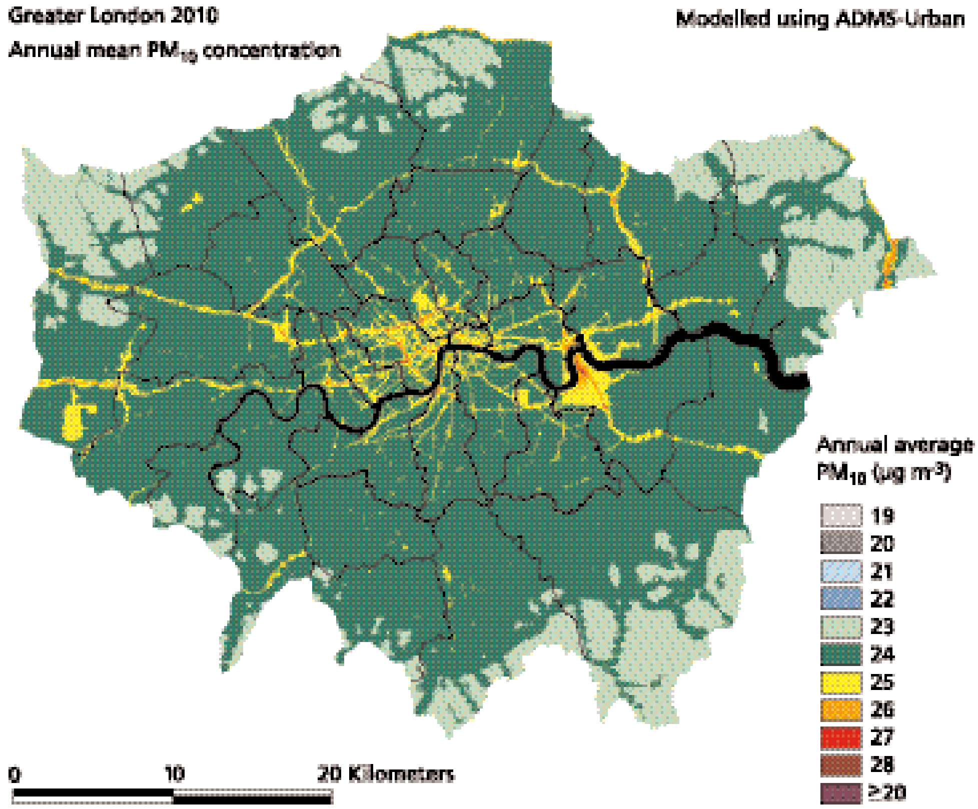


Figure 8.35 The 90th percentile of 24-hourly average PM_{10} 2004 (typical meteorology), corresponding to daily average concentration exceeded 35 times.

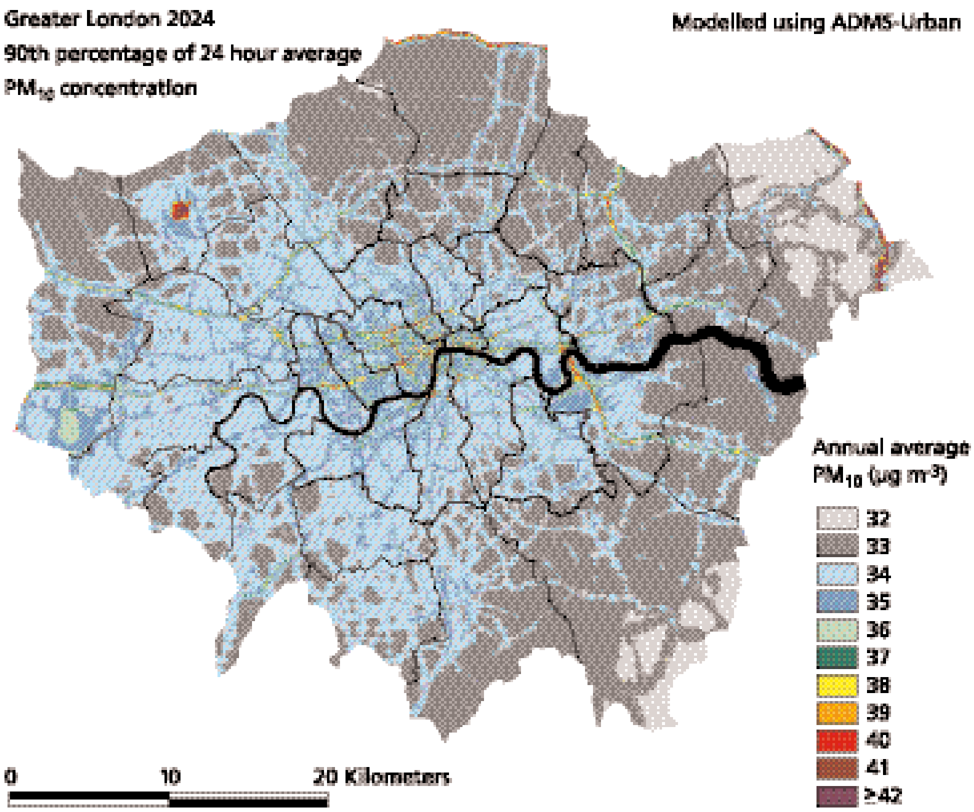


Figure 8.36 The 98th percentile of 24-hourly average PM_{10} 2010 (typical meteorology), corresponding to daily average concentration exceeded 7 times.

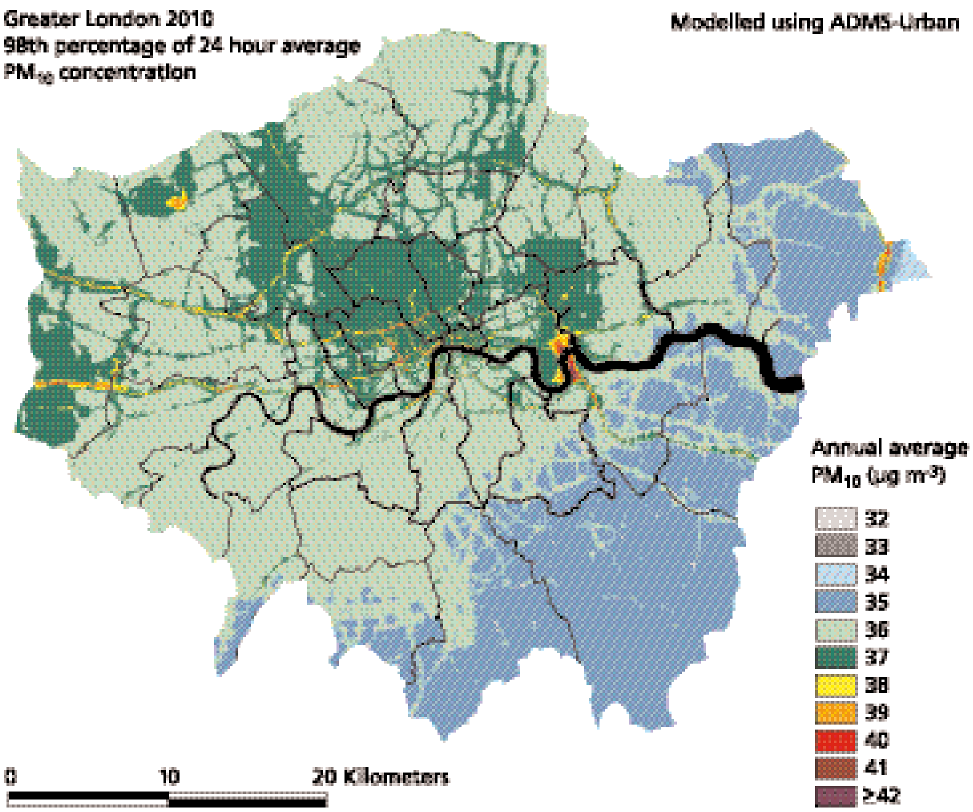
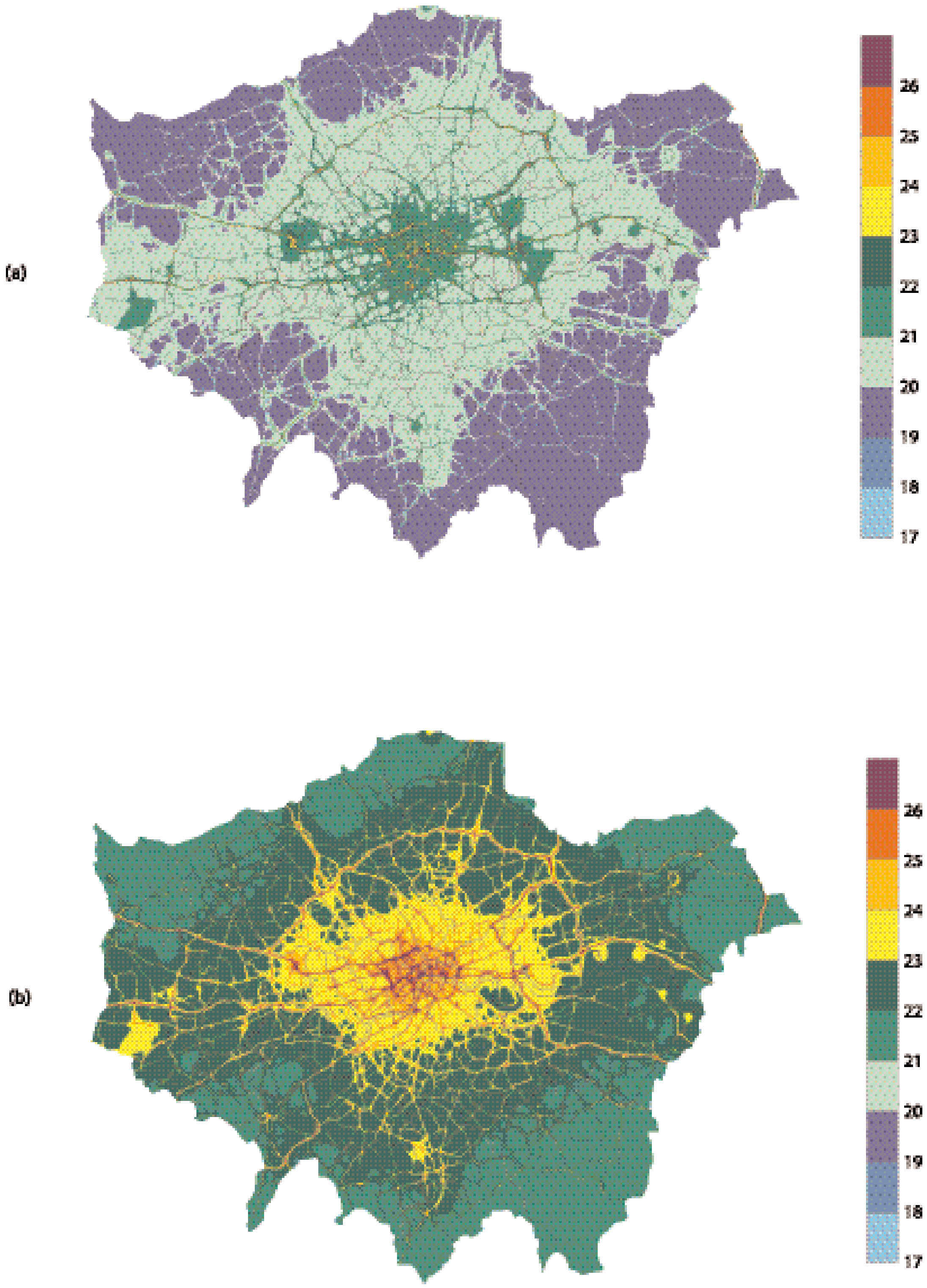
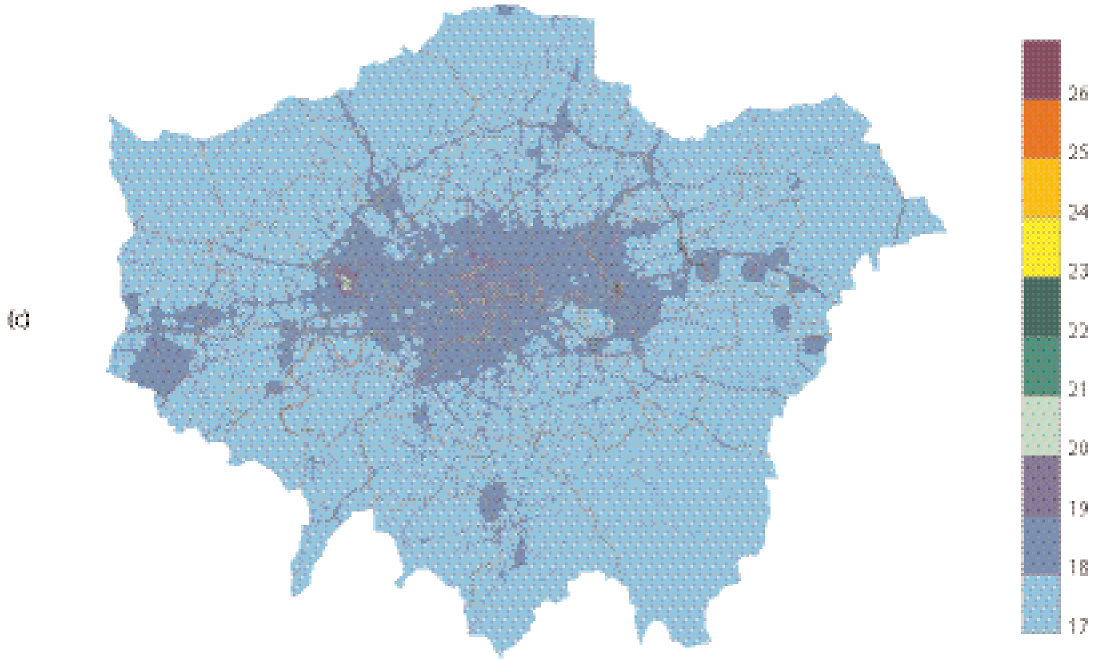


Figure 8.37 London maps predicted using the ERG model (a) Annual mean PM_{10} for 2001; (b) annual mean PM_{10} predicted for 2005 assuming 2001 meteorology; (c) annual mean PM_{10} predicted for 2010 assuming 2001 meteorology (μg m^{-3} (TEOM * 1.3)).





8.4.1.3 Site-specific projections at monitoring sites

813. Figure 8.39 describes the best assessments for annual mean PM_{10} concentrations over the period from 1992–2010 from a combination of observation and site-specific modelling. The observations refer to the average of the annual mean PM_{10} concentrations for 1997–2003 for ten selected background sites: values for earlier years are the average over a subset of these sites in operation in each year. These ten sites comprise one rural site, Rochester and nine urban background sites: London Bloomsbury, Birmingham centre, Cardiff centre, Edinburgh centre, Belfast centre, Liverpool centre, Newcastle centre, Manchester Piccadilly, Bristol centre.

814. The average annual mean PM_{10} concentration declines from 27.6 µg m^{-3} (TEOM) in 1992 to a minimum of 18.5 µg m^{-3} in 2000 before rising again to 20.8 µg m^{-3} in 2003. The upwards trend from 2000 onwards is continued to 2003 but it is likely that 2003 was exceptional as has been indicated in Figure 8.31 by plotting the preliminary result for the first quarter of 2004.

815. Also plotted are the ten-site averages of the annual mean PM_{10} concentrations determined with the site-specific projections model. These model concentrations span the range from 24.1 to 26.5 µg m^{-3} (TEOM) in 1992 and from 15.0 to 17.0 µg m^{-3} in 2010. Each line intersects the observed line at one year during the period 1996 to 2002 (projections from the base years of 2000 and 2003 have not been calculated). The projections reflect the trends in the average concentrations over the ten sites estimated in the site-specific model. Each site has its own source apportionment of current concentrations, which influences the predicted future concentrations, driven by the corresponding emission projections.

Table 8.14 Summary results from Netcen, ADMS-Urban and ERG modelling for PM_{10}. (All calculations outside London and total are for Netcen calculations.)

		Baseline (A)					
		Number of roadlinks with >20 μg m^{-3} (gravimetric) annual mean PM_{10}			Number of roadlinks with >40 μg m^{-3} (gravimetric) annual mean PM_{10}		
	Total assessed	ADMS 1999	Netcen 2002/ 2005	2010	ADMS 1999	Netcen 2002/ 2005	2010
ADMS-Urban London	18195	18195	18195	9042	76	0	0
Netcen London	1926	1923	1922	1883	44	12	0
Rest of England	6267	5321	4825	3130	1	0	0
Scotland	629	197	129	43	0	0	0
Wales	398	135	73	39	0	0	0
Northern Ireland	140	59	37	22	0	0	0
Total	9360	7635	6986	5117	45	12	0

		Road length (km) with >20 μg m^{-3} (gravimetric) annual mean PM_{10}			Road length (km) with >40 μg m^{-3} (gravimetric) annual mean PM_{10}		
	Total assessed	Netcen 2002/ ADMS 1999	2005	2010	Netcen 2002/ ADMS 1999	2005	2010
ERG London	4814	4814	4814	1742	4.2	0	0
ADMS-Urban London	3651	3651	3651	1626	0.6	0	0
Netcen London	1786	1782	1781	1730	24	9	0
Rest of England	10911	7964	6959	4113	0	0	0
Scotland	1348	304	197	59	0	0	0
Wales	982	236	121	55	0	0	0
Northern Ireland	1010	363	244	139	0	0	0
Total	16037	10649	9302	6096	24	9	0

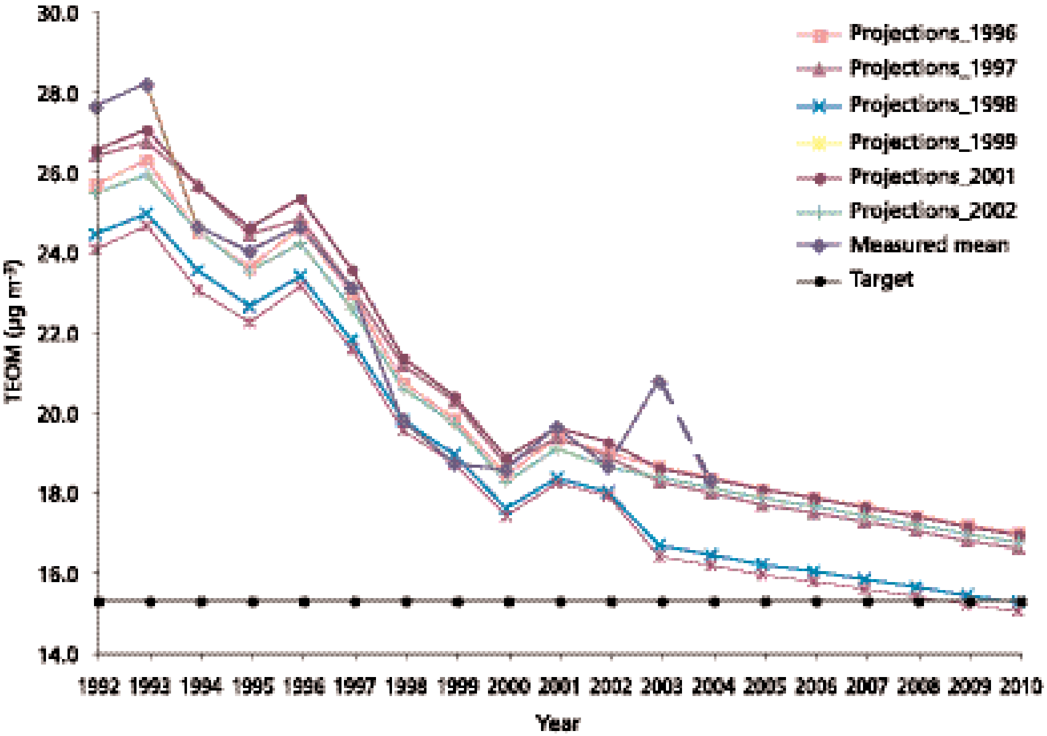
	Total assessed	Area (km^2) with >20 µg m^{-3} gravimetric annual mean PM$_{10}$			Area (km^2) >40 µg m^{-3} gravimetric annual mean PM$_{10}$		
		Netcen 2002/ ADMS 1999	2005	2010	Netcen 2002/ ADMS 1999	2005	2010
ERG London	1858	1858	953	39	3	0	0
ADMS-Urban London	1574	1574	1574	1175	0.4	0	0
Netcen London	1624	1491	1342	778	0	0	0
Rest of England	128765	8990	4853	1841	0	0	0
Scotland	77535	35	18	8	0	0	0
Wales	20745	136	52	18	0	0	0
Northern Ireland	13680	101	18	3	0	0	0
Total	242349	10753	6283	2648	0	0	0

	Total assessed	Population with >20 µg m^{-3} gravimetric annual mean PM$_{10}$			Population with >40 µg m^{-3} gravimetric annual mean PM$_{10}$		
		Netcen 2002/ ADMS 1999	2005	2010	Netcen 2002/ ADMS 1999	2005	2010
Netcen London	7,650,944	7,034,197	6,665,653	4,526,147	0	0	0
Rest of England	38,037,527	15,052,120	9,402,849	4,139,160	0	0	0
Scotland	4,905,019	60,549	27,981	3,362	0	0	0
Wales	2,916,782	174,391	73,786	31,515	0	0	0
Northern Ireland	1,577,855	319,520	58,715	13,326	0	0	0
Total	55,088,127	22,640,776	16,228,984	8,713,511	0	0	0

816. There is good correspondence between the time profiles of the observed and site-specific model annual mean concentrations over the period up to 2003. Both the observations and the predicted concentrations show a steeply declining trend followed by a levelling off and an increase from 2000 to 2001. It is noted that the projections from the site-specific model tend to show monotonic declines over the period from 2004 up to 2010. The model estimates for 2010 come close to, but still exceed, the indicative Stage II limit value of 20 µg m^{-3} (gravimetric), equivalent to 15.3 µg m^{-3} (TEOM), as an annual mean concentration. To have achieved the indicative limit value in 2010, the measured annual mean concentrations would have needed to have shown a decline of –4 % per year over the entire 1997–2010 period, which is close to that observed for the long-running urban background sites over the period 1992–2003.

817. Several factors have been identified that may lead to systematic over or underestimates by 1 to 2 µg m^{-3}. The influence of a perturbation of plus or minus such amounts has been investigated within the Netcen and ERG models, and the effect is indicated in the tables giving a factor of up to 2 on the extent of exceedence relative to the base case. The tables imply good agreement between the Netcen and ERG estimates allowing for such uncertainties.

Figure 8.38 Observed annual mean PM_{10} concentrations over the period 1992–2003 for the background sites for which results are available from the site-specific model and their projections through to 2010.



8.4.1.4 Mapped concentrations for $PM_{2.5}$

818. Two sets of national maps for $PM_{2.5}$ have been calculated, calibrated using TEOM (Figures 8.39 and 8.40) and gravimetric measurements (Figures 8.41 and 8.42). The TEOM maps were calibrated using measurements from four national network sites. The gravimetric maps were calibrated using data from seven national network sites. The maps for $PM_{2.5}$ show similar spatial patterns to PM_{10}; however,

Figure 8.39 Estimated UK annual mean background $PM_{2.5}$ concentration, 2002, in µg m^{-3} (TEOM).

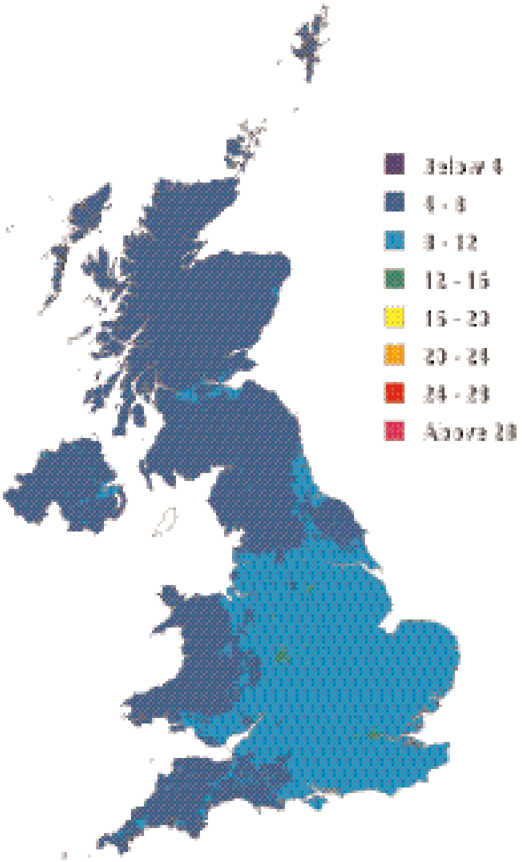


Figure 8.40 Estimated UK annual mean roadside $PM_{2.5}$ concentrations for major built-up roads, 2002, in µg m^{-3} TEOM).

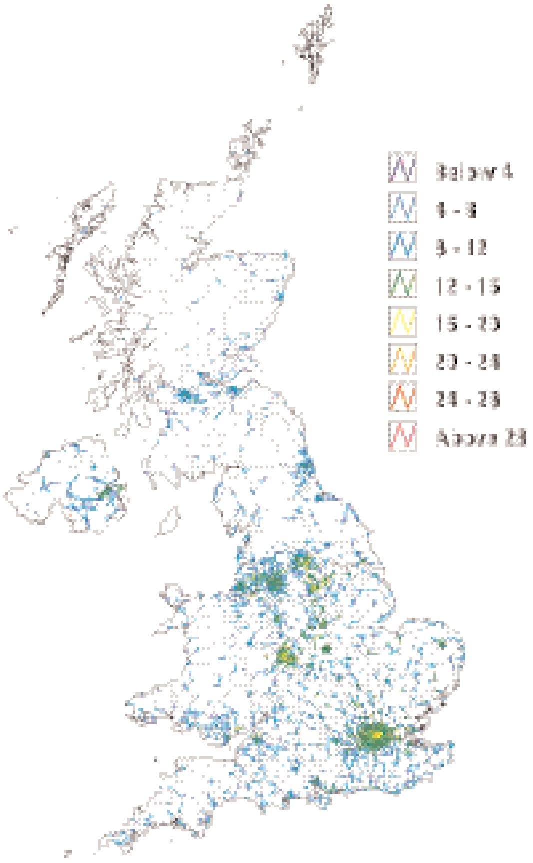


Figure 8.41 Estimated annual mean background $PM_{2.5}$ concentration, 2002, in µg m^{-3} (gravimentric).

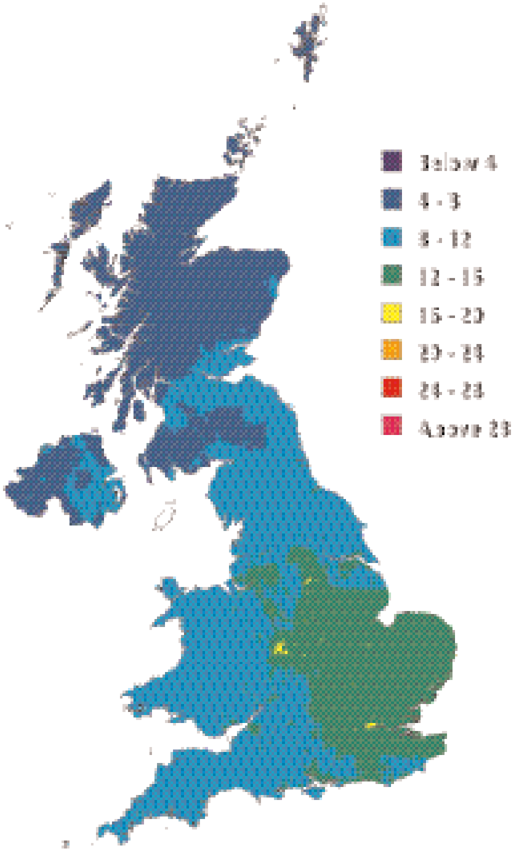


Figure 8.42 Estimated UK annual mean roadside $PM_{2.5}$ concentrations for major built-up roads, 2002, in µg m^{-3} (gravimetric).

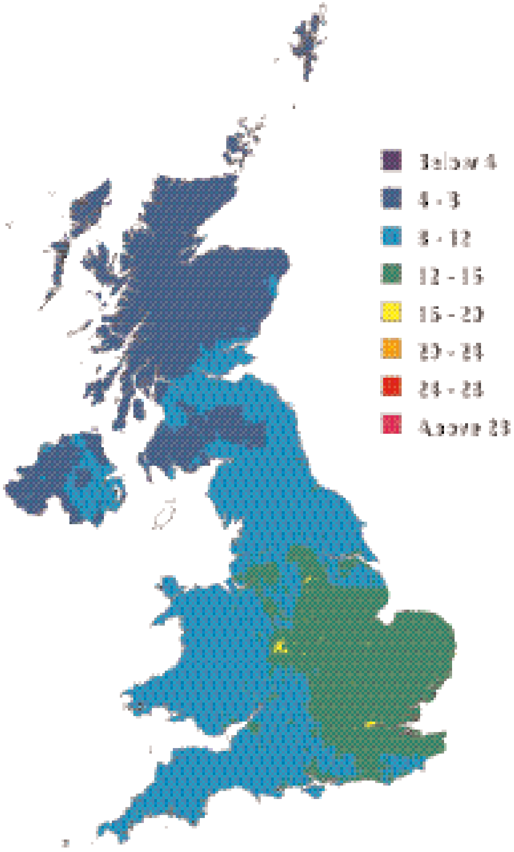


Figure 8.43 Annual average $PM_{2.5}$ mean concentration calculated using ADMS-Urban for (a) 2004 and (b) 2010.

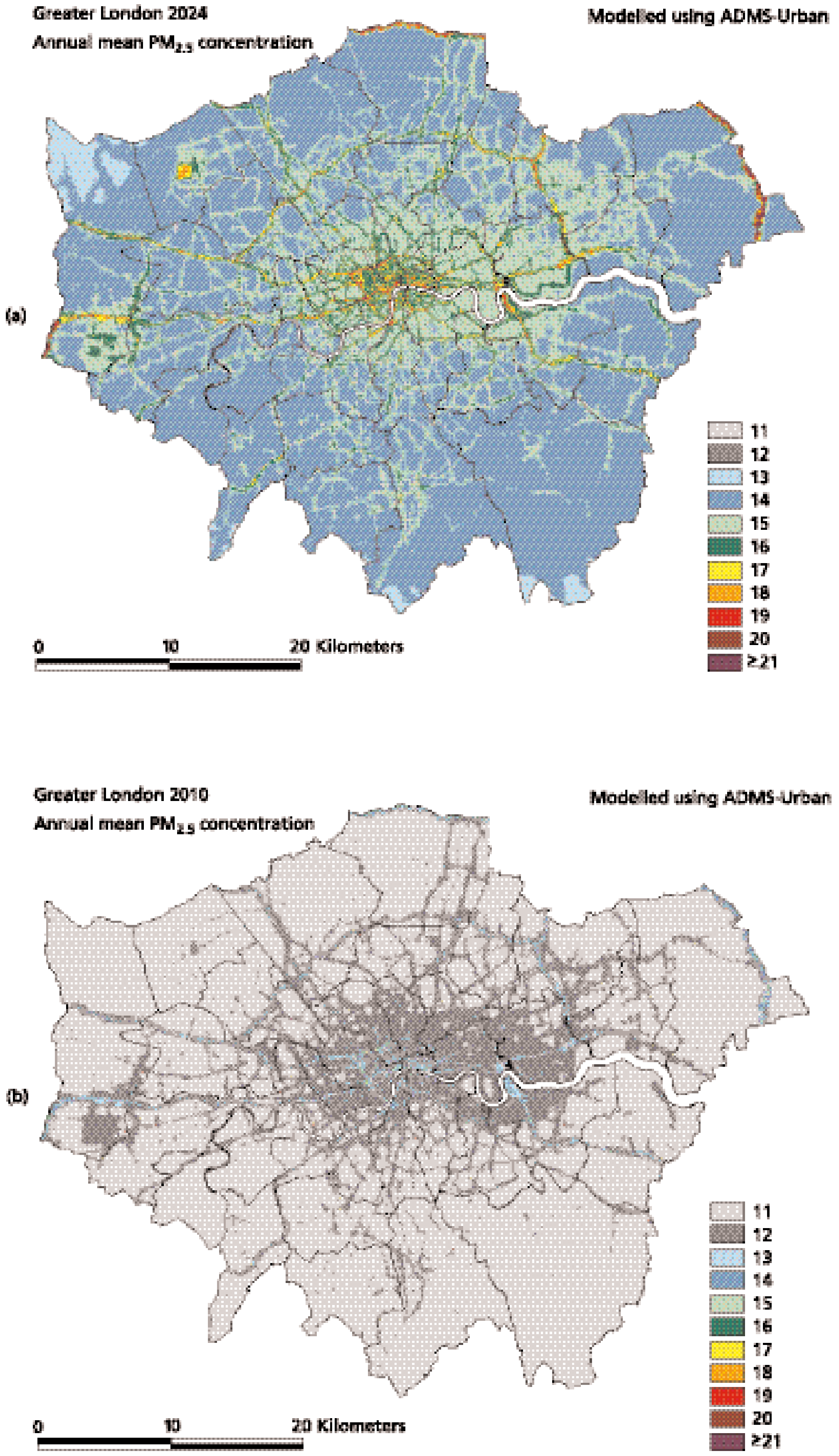


Table 8.15 Summary results from Netcen and ADMS-Urban mapping for $PM_{2.5}$. (All areas outside London and total are for Netcen calculations.)

	Total assessed	**Number of roadlinks with >12 $\mu g\ m^{-3}$ annual mean $PM_{2.5}$**				**Number of roadlinks with >16 $\mu g\ m^{-3}$ annual mean $PM_{2.5}$**				**Number of roadlinks with >20 $\mu g\ m^{-3}$ annual mean $PM_{2.5}$**			
		Netcen 2002/ ADMS 1999		**2010**		**Netcen 2002/ ADMS 1999**		**2010**		**Netcen 2002/ ADMS 1999**		**2010**	
ADMS-Urban London	18195	18195		5047		17303		64		3409	463	0	35
		TEOM	Grav.	TEOM	Grav.	TEOM	Grav.	TEOM	Grav.	TEOM	Grav.	TEOM	Grav.
Netcen London	1926	1821	1926	1035	1923	548	1742	79	629	82	141	1	3
Rest of England	6267	2710	5866	752	4553	257	1665	17	321	10	0	0	0
Scotland	629	53	168	2	15	0	2	0	0	0	0	0	0
Wales	398	38	187	8	45	1	10	1	0	0	0	0	0
Northern Ireland	140	32	50	3	13	1	7	0	0	0	0	0	0
Total	9360	4654	8197	1800	6549	807	3624	97	950	92	604	1	38

	Total assessed	**Number of roadlinks with >12 $\mu g\ m^{-3}$ annual mean $PM_{2.5}$**				**Road length (km) with >16 $\mu g\ m^{-3}$ annual mean $PM_{2.5}$**				**Road length (km) with >20 $\mu g\ m^{-3}$ annual mean $PM_{2.5}$**			
		Netcen 2002/ ADMS 1999		**2010**		**Netcen 2002/ ADMS 1999**		**2010**		**Netcen 2002/ ADMS 1999**		**2010**	
ADMS-Urban London	3651	3651		815		3289		7		433		0	
		TEOM	Grav.	TEOM	Grav.	TEOM	Grav.	TEOM	Grav.	TEOM	Grav.	TEOM	Grav.
Netcen London	1786	1663	1786	719	1782	306	1547	39	358	40	245	1	19
Rest of England	10911	3505	9595	998	6861	424	2223	26	420	13	236	0	2
Scotland	1348	74	264	3	27	0	3	0	0	0	0	0	0
Wales	982	53	262	15	61	2	16	0	0	0	0	0	0
Northern Ireland	1010	208	318	21	65	4	38	0	0	0	0	0	0
Total	16037	5503	12325	1757	8795	736	3828	65	778	53	481	1	21

	Total assessed	Area (km^2) with >12 µg m^{-3} annual mean PM$_{2.5}$				Area (km^2) with >16 µg m^{-3} annual mean PM$_{2.5}$				Area (km^2) with >20 µg m^{-3} annual mean PM$_{2.5}$			
		Netcen 2002/ ADMS 1999		2010		Netcen 2002/ ADMS 1999		2010		Netcen 2002/ ADMS 1999		2010	
ADMS-Urban London	1574	1574		410		1440		1		51		0	
		TEOM	Grav.	TEOM	Grav.	TEOM	Grav.	TEOM	Grav.	TEOM	Grav.	TEOM	Grav.
Netcen London	1624	213	1624	27	1531	0	111	0	9	0	0	0	0
Rest of England	128765	580	54866	239	12660	2	292	1	52	0	0	0	0
Scotland	77535	2	26	0	4	0	0	0	0	0	0	0	0
Wales	20745	5	212	1	53	0	0	0	0	0	0	0	0
Northern Ireland	13680	9	52	0	1	0	0	0	0	0	0	0	0
Total	242349	809	56780	267	14292	2	403	1	61	0	0	0	0

	Total assessed	Population with >12 µg m^{-3} annual mean PM$_{2.5}$				Population with >16 µg m^{-3} annual mean PM$_{2.5}$				Population with >20 µg m^{-3} annual mean PM$_{2.5}$			
		Netcen 2002/ ADMS 1999		2010		Netcen 2002/ ADMS 1999		2010		Netcen 2002/ ADMS1999		2010	
ADMS-Urban London	7,172,407	7,172,407		1,907,378		6,604,814		640		171,045		9	
		TEOM	Grav.	TEOM	Grav.	TEOM	Grav.	TEOM	Grav.	TEOM	Grav.	TEOM	Grav.
Netcen London	7,650,944	1,590,589	7,269,158	114,880	7,120,938	0	935,757	0	25,529	0	0	0	0
Rest of England	38,037,527	1,465,710	28,913,434	645,334	13,640,192	2,900	774,943	944	123,226	0	0	0	0
Scotland	4,905,019	354	33,805	0	1,068	0	0	0	0	0	0	0	0
Wales	2,916,782	8,975	328,012	76	55,387	0	0	0	0	0	0	0	0
Northern Ireland	1,577,855	29,959	157,259	0	4,926	0	0	0	0	0	0	0	0
Total	55,088,127	3,095,587	36,701,668	760,290	20,882,510	2,900	1,710,700	944	148,755	0	0	9	0

the actual concentration calculated depends a great deal on the measurement method, as no scaling factor has been used for the TEOMs. The ADMS-Urban calculations of $PM_{2.5}$ for London (Figure 8.43) are more similar to the partisol-based values from the national models. In Table 8.15 GIS has been used for each of the model runs to calculate the number of road links, roadlength, area and population within the area exceeding the annual means of 12, 16 and 20 $\mu g\ m^{-3}$. It can be seen that by 2010 all models predicted almost no exceedence of 20 $\mu g\ m^{-3}$ and limited exceedence (none in some cases) of 16 $\mu g\ m^{-3}$.

8.4.2 Regional models

8.4.2.1 Comparison of modelled sulphate, nitrate and ammonium concentrations for the UK

819. EMEP and FRAME were both originally developed for sulphur and nitrogen deposition, and work on the secondary particulates has evolved more recently as increased emphasis has been placed on the health effects of fine PM. Both FRAME and EMEP provide concentrations of sulphate, ammonium and aerosol nitrate, although EMEP provides further subdivision into coarse and fine size fractions for the nitrate. Results from a detailed intercomparison study in progress are available from two versions of EMEP, EMEP (1) and EMEP (2), from a current version of FRAME plus the previous data provided for use in UKIAM. The illustrations given below include maps of FRAME with one EMEP version in each case.

820. Figure 8.44 shows a comparison of the estimated sulphate concentrations from FRAME and EMEP. In the EMEP model the variation across the UK is less marked than in FRAME, which indicates rather lower values in the more remote areas and the higher values close to the major sources on the eastern side of the country. Such locally enhanced concentrations are very sensitive to the assumed fraction of sulphur emitted as sulphate.

821. In Figure 8.45 the model results are compared with measurements from the 12 background monitoring sites as scatter plots.

822. Figure 8.46 shows the corresponding results for ammonium. The FRAME model indicates somewhat smaller values than the EMEP model. Figure 8.47 again gives comparison, with the measurements showing good agreement with the EMEP model, but some underestimation with FRAME. In all the maps there is a clear gradient from the southeast to the remote northwest.

823. Figure 8.48 compares model results for maps of aerosol nitrate concentrations, and Figure 8.49 shows the corresponding scatter plots comparing with measurements. (Here the EMEP results correspond to a more recent version of the model in April 2004 with some revised nitrate chemistry.)

824. The aerosol nitrate will depend both on the overall rate of oxidation to nitrate and the partitioning between the aerosol nitrate and the gaseous nitric acid. For the latter, both models differ from measured values over the UK network with the EMEP values being particularly low. There is ongoing work on the nitrate chemistry to resolve these discrepancies. However, for the nitrate aerosol component the models are generally in good agreement both with each other and the measurements.

Figure 8.44 Calculated sulphate, by EMEP and FRAME.

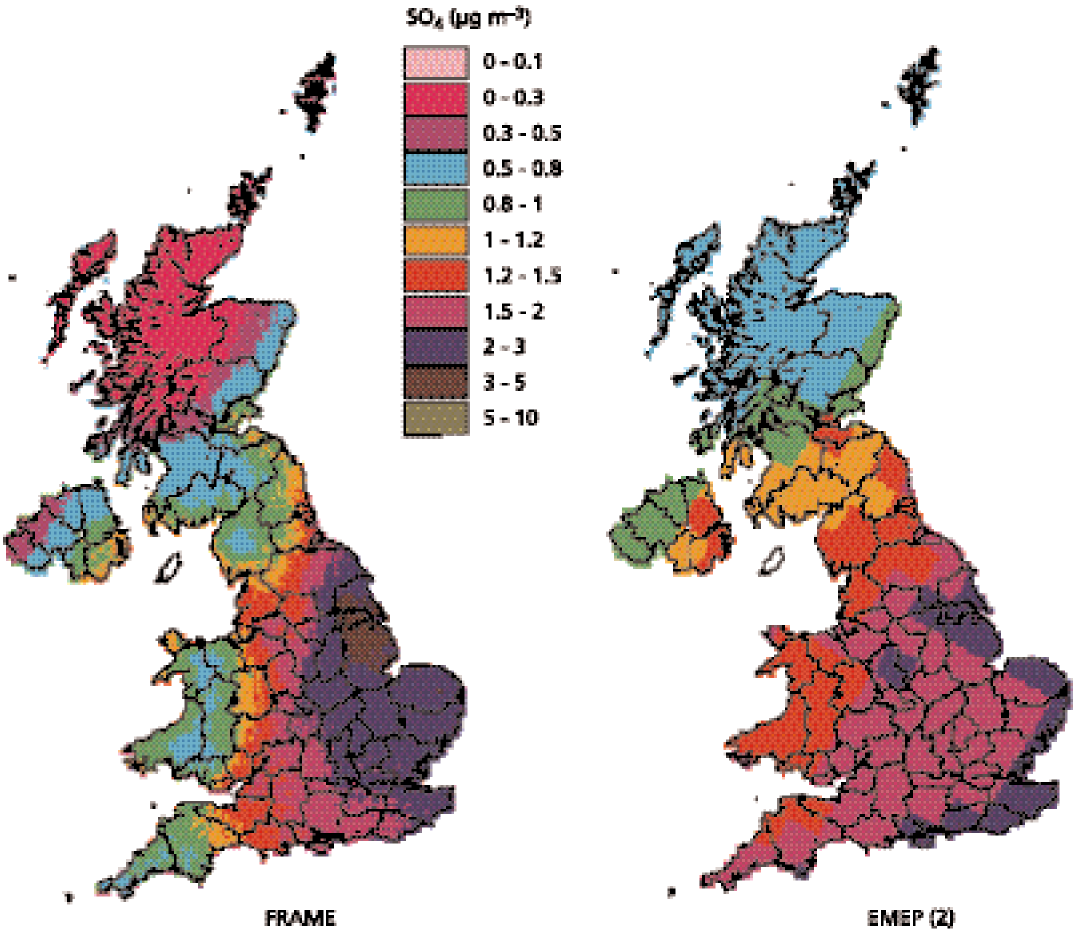
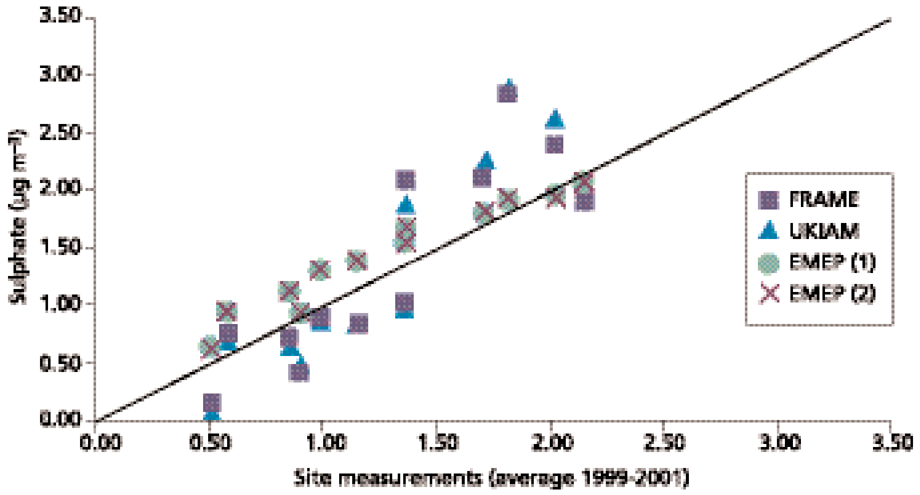


Figure 8.45 Comparison between models and measurements of sulphate.



825. The reason why the secondary inorganic aerosol concentrations are of concern is the health effects attributed to exposure to fine PM. As a measure of risk we have calculated population exposure to the combined total sulphate plus nitrate plus ammonium concentrations. In Figure 8.50 maps of total secondary inorganic aerosols from the FRAME and EMEP model are compared with the estimates from

Figure 8.46 Calculated ammonium aerosol, by EMEP and FRAME.

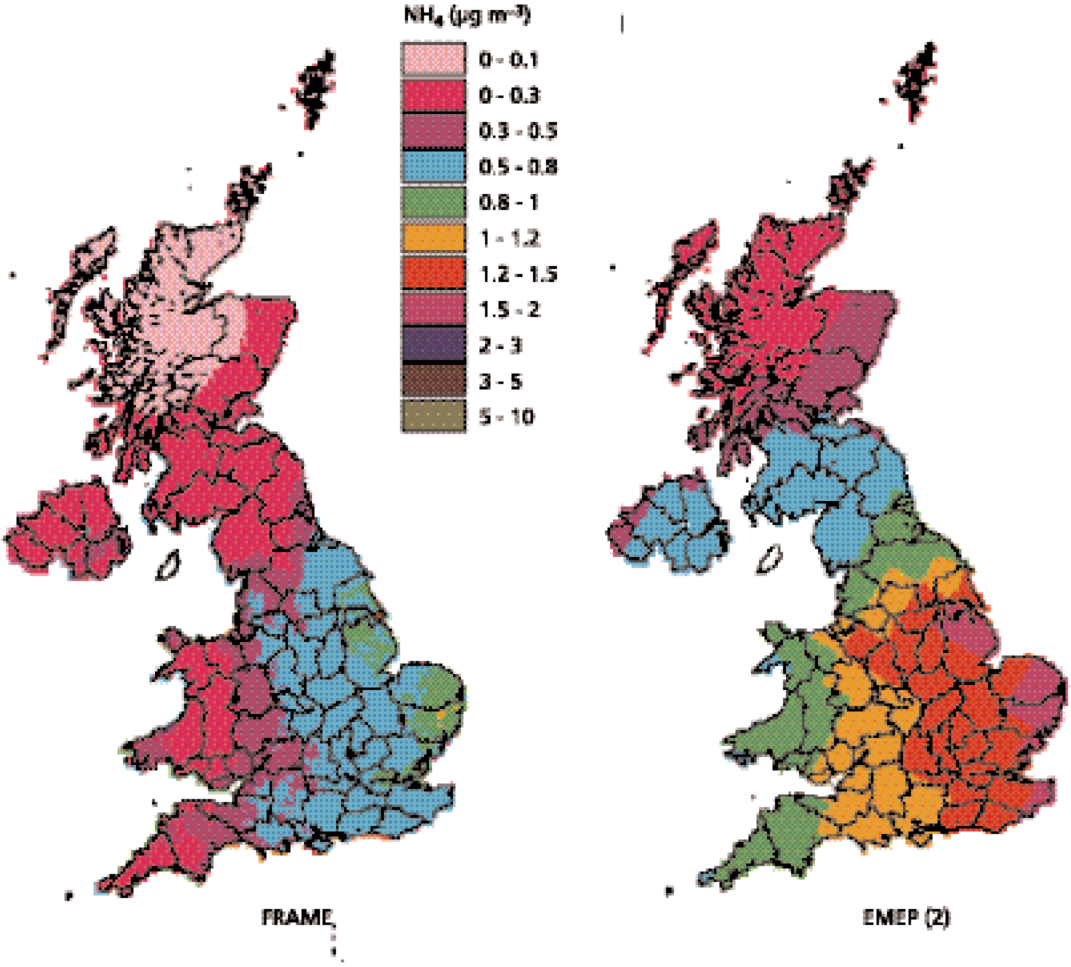
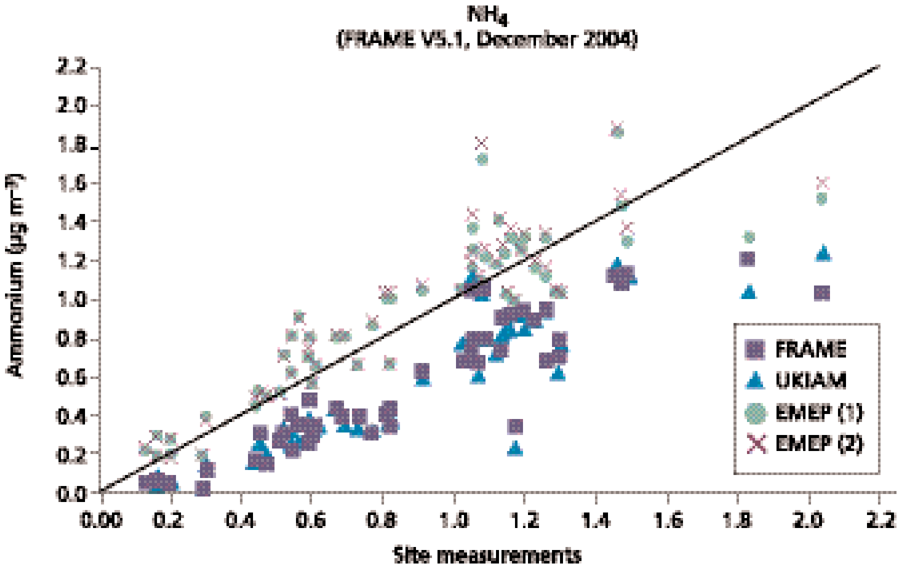


Figure 8.47 Comparison between models and measurements of ammonium concentrations.



the Netcen mapping model. Table 8.16 shows the corresponding estimates of population exposure in person g m^{-3} (derived by summing over the grid squares the population from census data times the annual average concentration in µg m^{-3}).

Figure 8.48 Calculated nitrate aerosol, by EMEP and FRAME.

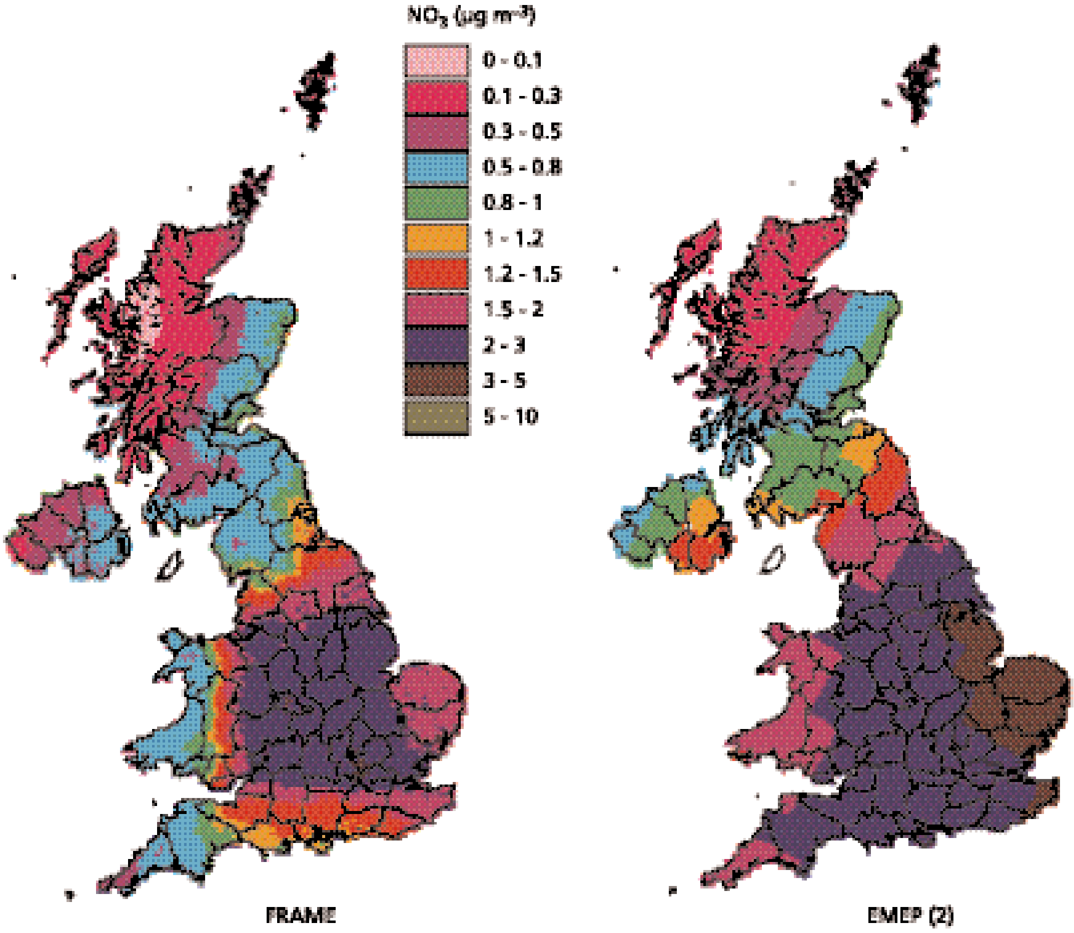


Figure 8.49 Comparison between models and measurements of nitrate aerosol.

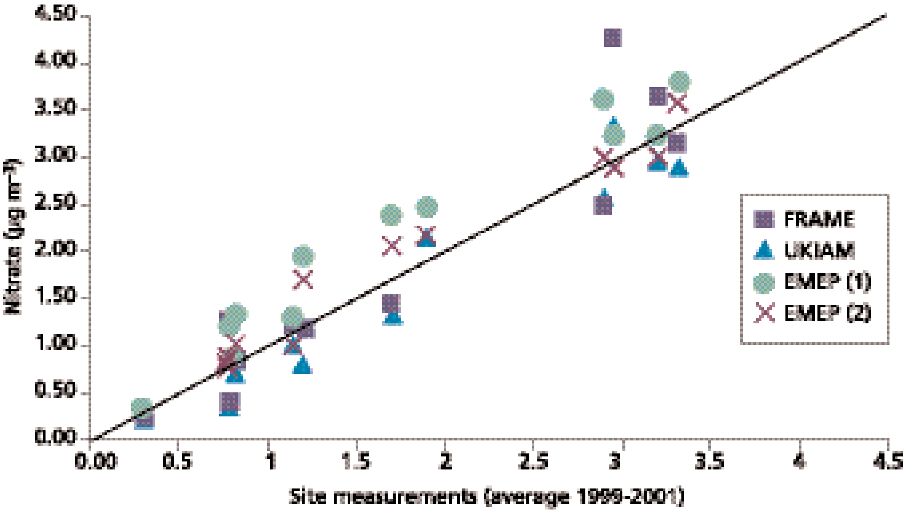


Figure 8.50 Calculated total secondary inorganic aerosol, by EMEP, FRAME and Netcen.

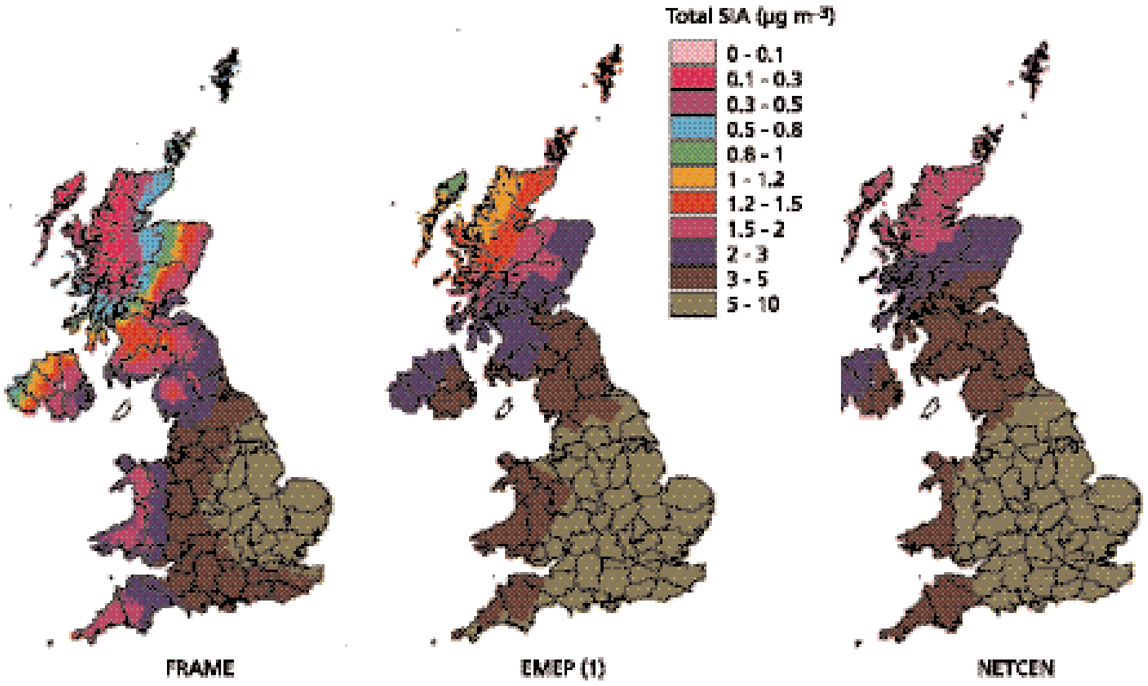


Table 8.18 Total UK population exposure to secondary inorganic aerosol (SIA) (in person µg m^{-3}) calculated from year 2000 annual average concentrations (2002 for Netcen data).

	SO_4	NO_3	NH_4	Total SIA
FRAME	95.54	154.88	32.85	283.27
UKIAM	100.06	140.93	42.04	283.04
EMEP (1)	97.28	149.61	63.82	310.71
EMEP (2)	95.52	129.98	64.25	289.74
Netcen	—	—	—	337.12

8.4.2.2 Europe-wide calculations

826. Figure 8.51 presents the annual average boundary layer concentration map for particulate sulphate for 1996. The map shows a widespread distribution over much of Northwest Europe with a maximum over Northern France, Belgium, the Netherlands and Northern Germany. A second maximum is found over Northern Italy, separated from the rest of Europe by the Alps. The 1.5 µg S m^{-3} contour spreads into the UK from Northwest Europe, showing the influence of long-range transport both out of and into the UK.

827. The spatial pattern obtained from the NAME model is in broad agreement with the spatial pattern obtained from the observations within the EMEP network, as described by Hjellbrekke *et al.* (1997). The model and observed maps are similar over the UK and Scandinavia but they are markedly different over the rest of

Europe. The model values are consistently higher than those measured over the region extending from Northern France into Northern Germany. The observed maximum is further east, over Poland and the Czech Republic. The model agrees well with the observations of Ottley and Harrison (1992) over the North Sea.

828. Figure 8.52 presents the corresponding map for particulate nitrate expressed as the sum of both fine ammonium nitrate and coarse nitrate formed by displacement reactions. This shows a maximum over Belgium and The Netherlands and a secondary maximum over Northern Italy. Again the moderately polluted contours spread into the UK.

829. The spatial pattern generated by the NAME model compares well with the map derived from observations (Hayman *et al*. 2001). The model overestimates levels across Scotland, 0.2–1.5 µg NO_3 m^{-3} in the model compared with 0.2–1.0 µg NO_3 m^{-3} in the observations. Similarly in the Southeast of England the model gives 3–5 µg NO_3 m^{-3} in the region where the observations report concentrations of NO_3 of greater than 1.5 µg m^{-3}.

830. By adding together the total mass of ammonium nitrate, ammonium sulphate and coarse particle nitrate – excluding the contribution to the mass from ions such as sodium and calcium – it is possible to map the modelled secondary inorganic particulates as PM_{10} and this is shown in Figure 8.53. The maximum over Northern Italy is now clearly the strongest with the maximum over the low countries a secondary one. The 10 µg m^{-3} PM_{10} contour spreads across Northwest Europe and into the UK. This can be compared with the observed regional background over London of 16.5 µg m^{-3} inferred from the observations in the LAQN. The pattern predicted by NAME for 1996 can also be broadly compared with calculations of the EMEP model (Figure 8.54). This also shows the highest levels of sulphate and nitrate over the low countries and Northern Italy; however, the areas of highest concentration are much less focussed.

Figure 8.51 Annual average particulate sulphate concentrations during 1996 across Northwest Europe, as calculated by NAME.

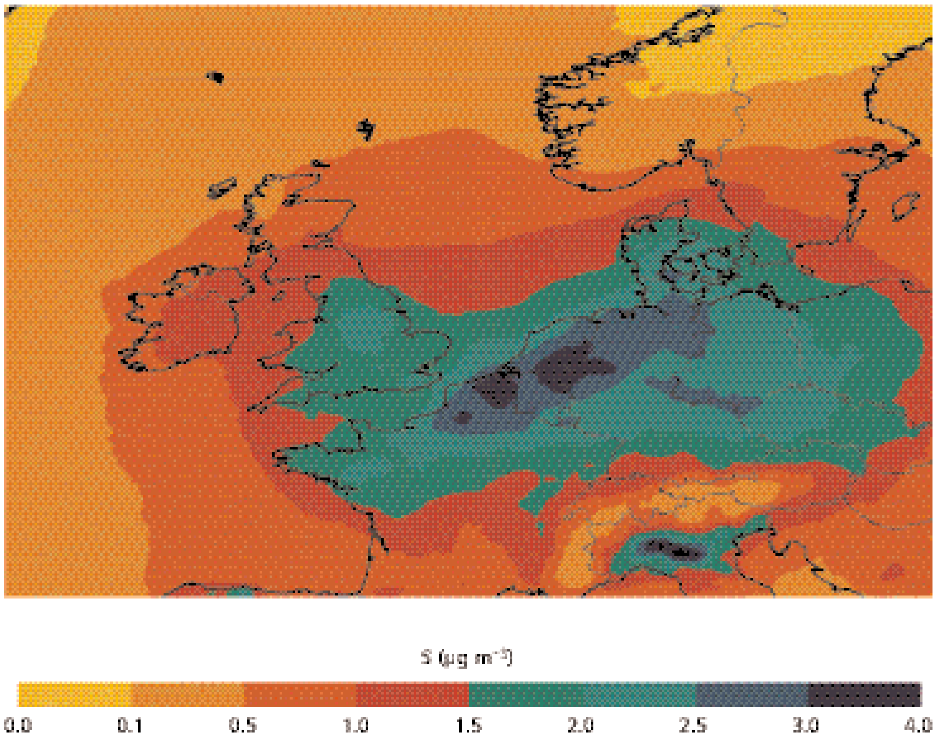


Figure 8.52 Annual average particulate nitrate concentrations during 1996 across Northwest Europe, expressed as the sum of both fine and coarse components.

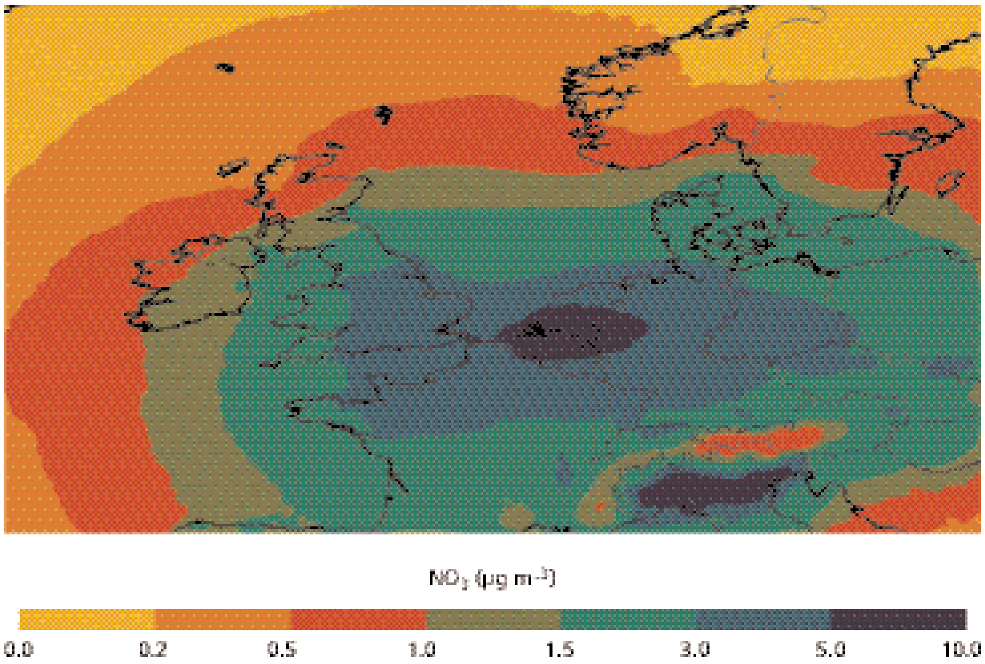


Figure 8.53 Annual average secondary inorganic particulate concentrations plotted as PM$_{10}$ for 1996.

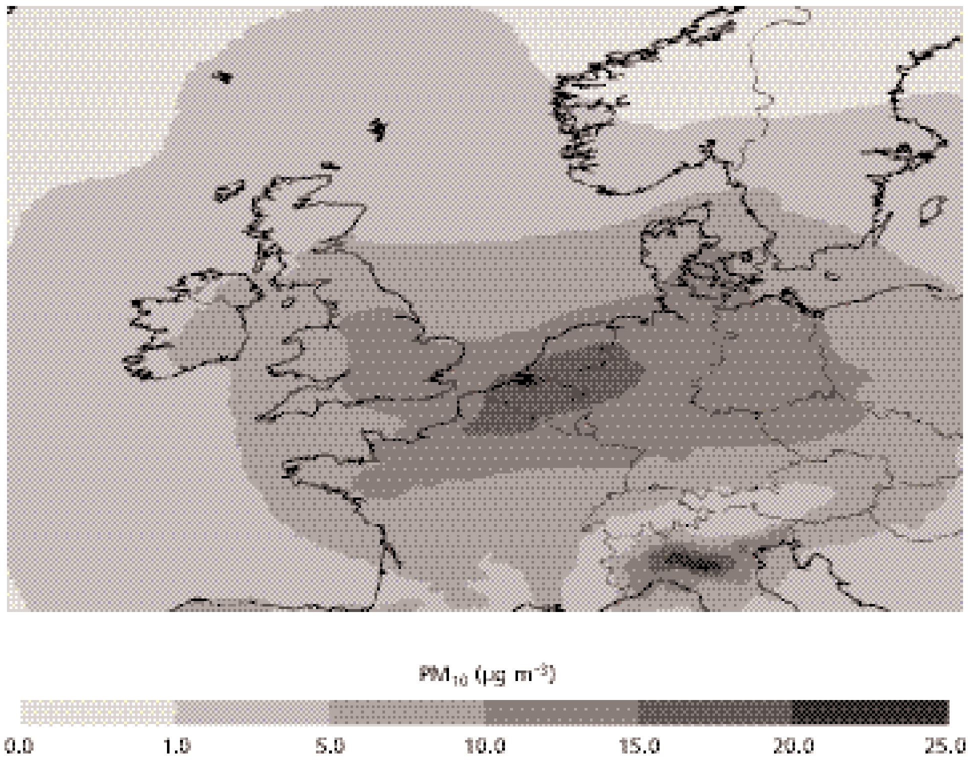
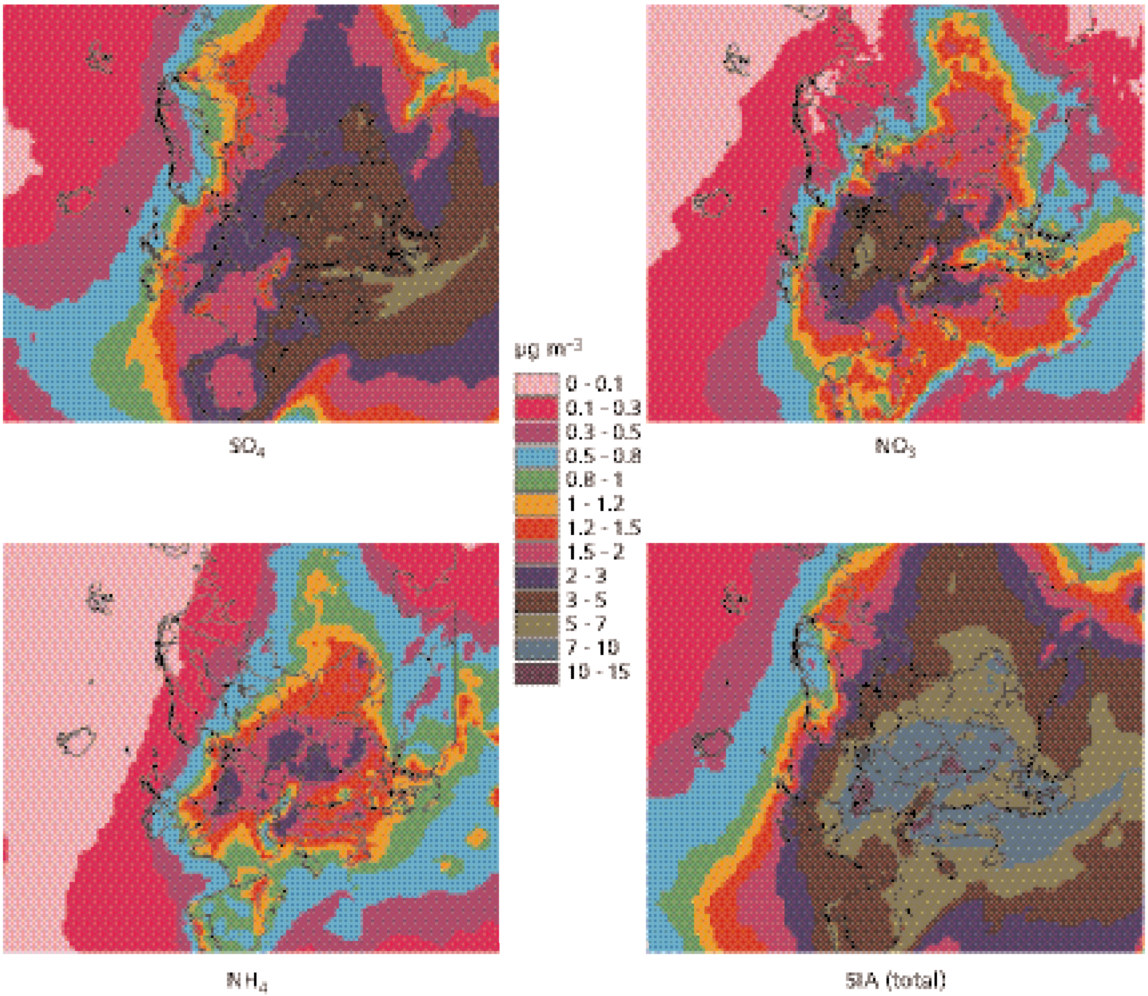


Figure 8.54 Secondary inorganic aerosol concentrations across Europe estimated using EMEP (2), 2003 emissions and 2000 meteorology.



831. Figure 8.55 shows how the NAME model can give an indication of the likely country attribution of the SO_2 emissions that acted as precursors to the model particulate sulphate. Each of the main observed pollution episodes is represented as vertical bars in Figure 8.55. The bar is sectioned according to the country of origin of the SO_2 that had been transformed into particulate sulphate *en route* to Yarner Wood.

832. During the January 2002 episode, the particulate sulphate appears to have originated from the UK, France and the Benelux countries. During the prolonged episode during March 2002, however, the UK appears to have been the dominant source. The May 2002 episode appears to have involved the UK, the Benelux countries and Germany. During the summer period from July to August, the UK appears to have been the dominant source. The episode during September is characteristically different, with contributions from France, Germany, Poland and the Baltic. This is the time when long-range transport brought elevated PM_{10} levels to much of the UK from the forest fires near Moscow. The episodes during December 2002 brought particulate sulphate from the UK and Germany to Yarner Wood.

Figure 8.55 Daily attribution of sulphate aerosol at Yarner Wood for 2002.

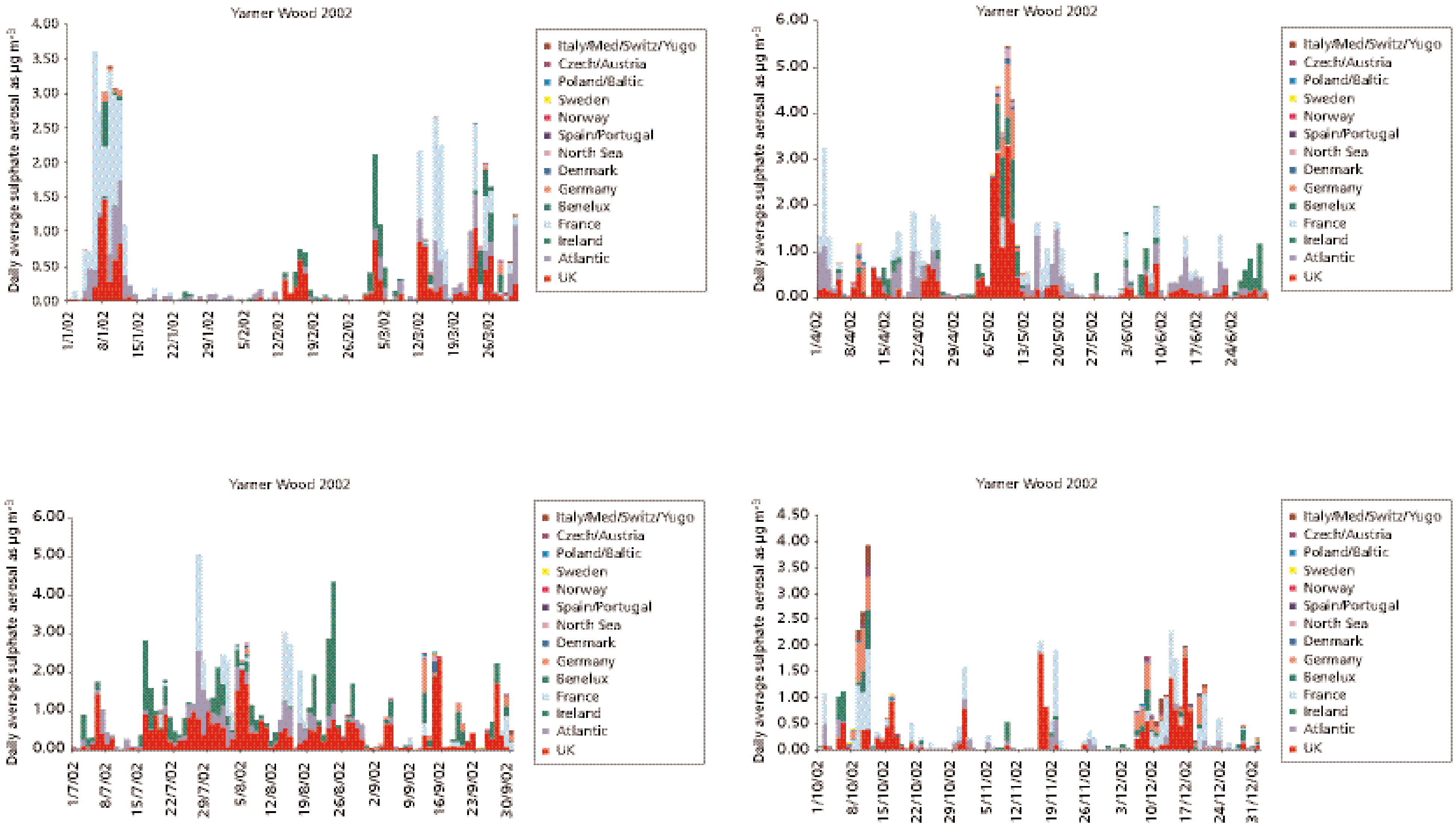
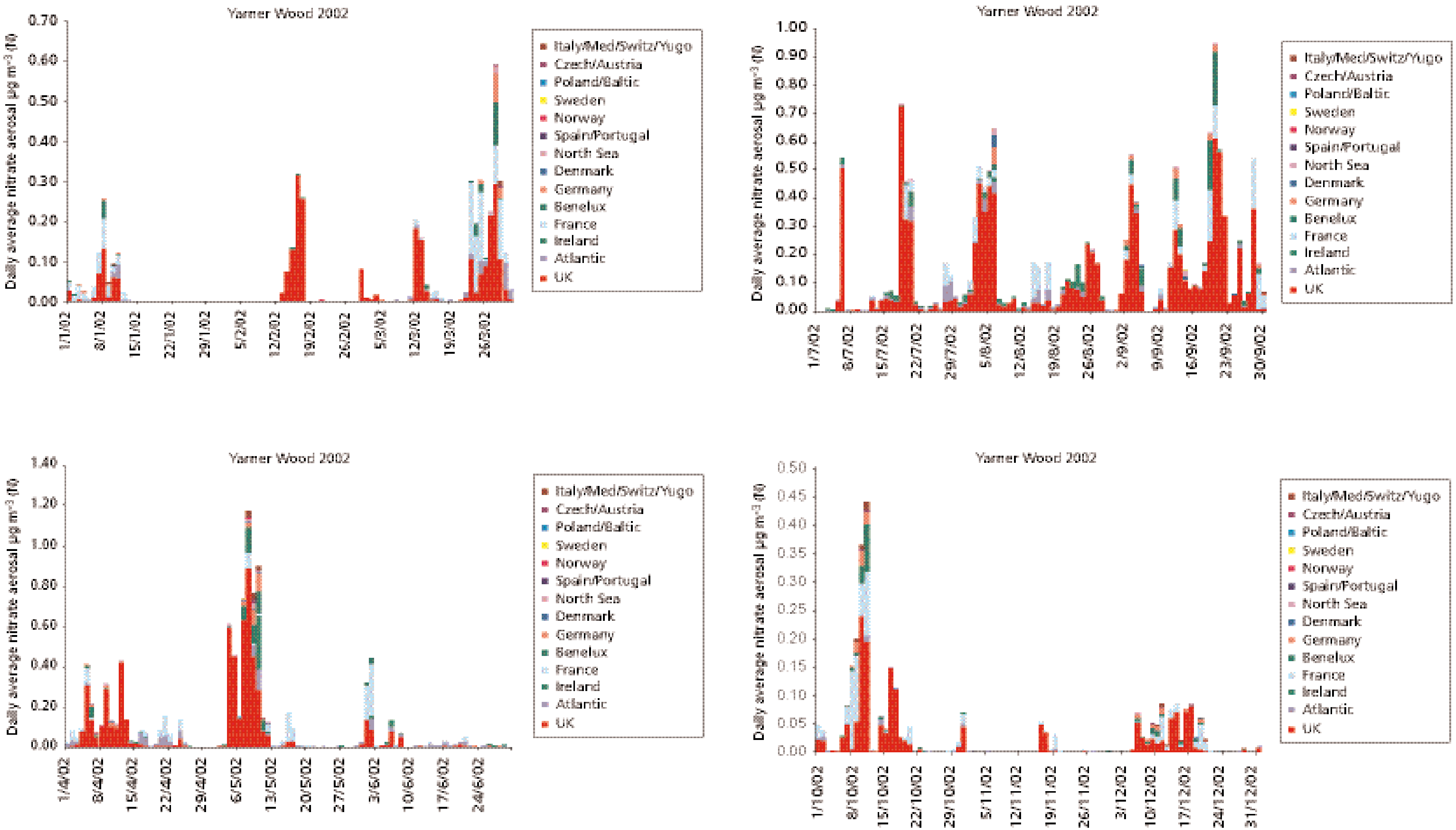


Figure 8.56 Daily attribution of nitrate aerosol at Yarner Wood for 2002.



833. Integrating over the entire year 2002, about 25% of the particulate sulphate modelled at Yarner Wood has been derived from SO_2 sources outside of the UK. Figure 8.56 shows a series of graphs for the attribution of nitrate aerosol at Yarner Wood for the same period. There are no measurement data available for nitrate aerosol at Yarner Wood. It is thought, based on previous work, that the NAME model underpredicts nitrate aerosol, particularly in the winter, and that these results are indeed very low with a maximum of <1.2 μg m^{-3} in May and values very close to zero through much of the winter.

834. Similar calculations to those described for Yarner Wood have been conducted at 30 sites across the UK and Ireland to obtain the percentage contributions to the annual average for each country or group of countries.

835. Tables 8.13 to 8.15 show, respectively, the percentage contribution to modelled sulphate aerosol, nitrate aerosol and sulphur at a subset of these sites. Some countries have been grouped and it should be noted that the model domain does not fully extend into some countries, for example Norway, Sweden and Italy.

836. Looking at the tables, the model is predicting that (as a percentage of the total for that species) the greatest import from outside the UK is by sulphate aerosol. Sites to the south and east of the UK see greatest import from other countries of which the larger part originates from France. Lough Navar and Belfast (Northern Ireland) and Mace Head (Ireland) have large contributions from Ireland, as would be expected. Substantial Irish contributions are also seen at northern and western UK sites, for example, Eskdalemuir and Narberth.

837. The contribution from other European countries falls off markedly with distance from the UK, as would be expected.

838. In contrast to Table 8.16, the percentages shown in Table 8.17 for sulphur dioxide show that UK emissions are much more dominant. France has quite a big effect on the London sites and in the south and east generally, reaching a maximum of 19.4% of sulphur dioxide at Barcombe Mills. The percentage of sulphate aerosol arriving at Barcombe Mills is 38.1%, however, which demonstrates how generation of the secondary aerosol can actually have a more dominant effect than the primary species. It should be noted that the attribution for the secondary species includes aerosol formed during the travel time from the country of origin of the primary species (sulphur dioxide in this case).

839. The percentage contributions for nitrate aerosol in Table 8.18 show that sites are, generally speaking, less influenced by European nitrate aerosol than they are by European sulphate aerosol. For example, 63% of nitrate aerosol at Barcombe Mills is of UK origin compared with 38% of sulphate aerosol.

Table 8.19 Percentage contributions to sulphate aerosol in 2002 at the listed sites.

Site	UK	Atlantic	Ireland	France	Benelux	Germany	Denmark	North Sea	Norway	Sweden	Poland/ Baltic	Czech/ Austria	Italy/ Med/ Switz/ Yugo
Barcombe Mills	38.2	7.6	3.5	38.1	5.1	4.8	0.7	1.0	0.0	0.1	0.1	0.3	0.5
Belfast	37.7	5.8	40.9	6.7	2.4	4.2	0.8	0.8	0.0	0.2	0.2	0.3	0.1
Birmingham	60.3	6.1	6.8	15.3	4.7	4.2	0.9	0.8	0.0	0.1	0.2	0.3	0.2
Bloomsbury	46.7	6.4	3.6	28.8	6.5	5.1	0.9	1.1	0.0	0.1	0.2	0.4	0.4
Edinburgh	65.8	3.5	14.0	7.3	2.5	3.7	1.2	1.2	0.0	0.2	0.3	0.2	0.1
Mace Head	23.4	9.3	43.9	10.1	3.8	6.4	1.0	0.9	0.1	0.2	0.2	0.4	0.2
Narberth	39.9	15.1	19.8	13.9	4.2	4.9	0.6	0.8	0.0	0.1	0.1	0.4	0.3
Stoke Ferry	51.4	4.7	4.3	20.8	8.1	7.0	1.2	1.6	0.0	0.1	0.2	0.4	0.2
Strathvaich	54.7	5.6	17.9	8.6	3.2	4.4	2.5	2.0	0.1	0.3	0.5	0.2	0.1
Yarner Wood	36.0	17.7	12.4	23.3	4.2	4.3	0.5	0.7	0.0	0.1	0.1	0.2	0.5

Table 8.20 Percentage contributions to nitrate aerosol in 2002 at the listed sites.

Site	UK	Atlantic	Ireland	France	Benelux	Germany	Denmark	North Sea	Norway	Sweden	Poland/ Baltic	Czech/ Austria	Italy/ Med/ Switz/ Yugo
Barcombe Mills	63.1	1.1	0.4	22.7	8.2	2.6	0.3	1.2	0.0	0.0	0.0	0.2	0.2
Belfast	66.6	3.5	12.8	6.8	4.4	2.8	0.5	2.1	0.1	0.1	0.0	0.2	0.1
Birmingham	80.2	1.3	0.7	7.5	5.5	2.5	0.6	1.4	0.0	0.0	0.0	0.2	0.2
Bloomsbury	70.3	0.9	0.4	15.1	8.6	2.8	0.3	1.3	0.0	0.0	0.0	0.2	0.2
Edinburgh	81.7	1.1	2.0	4.3	4.1	2.4	0.7	3.2	0.0	0.1	0.0	0.1	0.1
Mace Head	43.3	7.3	31.4	6.6	5.2	4.0	0.3	1.0	0.1	0.2	0.0	0.3	0.3
Narberth	71.5	6.9	5.0	8.0	4.1	2.6	0.5	0.8	0.0	0.0	0.0	0.2	0.3
Stoke Ferry	70.4	0.6	0.4	10.6	10.7	4.0	0.4	2.7	0.0	0.0	0.0	0.2	0.1
Strathvaich	76.3	2.8	3.4	5.1	3.9	2.8	1.0	4.3	0.1	0.2	0.0	0.1	0.1
Yarner Wood	66.4	6.0	3.0	15.5	5.1	2.5	0.3	0.7	0.0	0.0	0.0	0.1	0.4

Table 8.21 Percentage contributions to sulphur dioxide in 2002 at the listed sites.

Site	UK	Atlantic	Ireland	France	Benelux	Germany	Denmark	North Sea	Norway	Sweden	Poland/ Baltic	Czech/ Austria	Italy/ Med/ Switz/ Yugo
Barcombe Mills	72.8	2.0	0.5	19.4	3.4	1.2	0.1	0.4	0.0	0.0	0.0	0.2	0.0
Belfast	64.3	1.9	30.9	1.3	0.4	0.6	0.1	0.3	0.0	0.0	0.1	0.1	0.0
Birmingham	95.6	0.9	0.6	1.5	0.7	0.5	0.0	0.2	0.0	0.0	0.0	0.1	0.0
Bloomsbury	84.2	1.4	0.5	9.3	2.9	1.1	0.1	0.4	0.0	0.0	0.0	0.2	0.0
Edinburgh	98.1	0.2	0.8	0.3	0.2	0.2	0.0	0.2	0.0	0.0	0.0	0.0	0.0
Mace Head	15.5	1.7	79.8	0.9	0.8	0.8	0.1	0.2	0.0	0.0	0.0	0.1	0.0
Narberth	66.5	20.8	5.2	4.5	1.4	1.0	0.1	0.3	0.0	0.0	0.0	0.2	0.0
Stoke Ferry	86.4	1.0	0.7	5.8	3.5	1.5	0.1	0.8	0.0	0.0	0.1	0.2	0.0
Strathvaich	84.1	2.8	5.2	2.4	1.2	1.6	0.6	1.6	0.1	0.1	0.3	0.1	0.0
Yarner Wood	67.5	15.0	2.2	11.7	2.0	1.0	0.1	0.3	0.0	0.0	0.0	0.1	0.0

8.5 Uncertainty in modelling PM

840. This section considers the accuracy and precision with which the models described in Section 8.2 are able to calculate airborne PM concentrations at sampled and unsampled locations, at locations similar and different to those where measurements are made, in the past, present and future. The focus will be on PM_{10} and the model results for PM_{10} that were presented in Section 8.4, but the ability to consider other PM metrics will also be considered. The larger amount and variety of uncertainty in PM_{10} modelling compared with NO_2 merits more detailed consideration of uncertainty in this report than was included for similar models in AQEG's NO_2 report (AQEG, 2004).

8.5.1 Methods for quantification of uncertainty and error

841. Sources of information on model uncertainty can be divided into the following three categories.

- Scientific assessment of the assumptions made in each model, compared with the theory of PM processing that was presented in Chapter 2 and elsewhere.
- Empirical quantification of error:
    - ☐ by comparison with measured data, necessarily in past years (but including retrospective assessment of the model's past ability to make future predictions, for example, model runs completed in 1998 predicting 2002 concentrations that can now be compared with measurements); or
    - ☐ by comparison between models for past or future scenarios.
- Sensitivity analysis of the extent to which varying model inputs causes the output to change (which can include inputs such as user choice of model assumptions and tunable parameters). In cases where a large number of parameters and/or inputs can be varied, a Monte Carlo approach to sensitivity analysis can be adopted.

842. Ideally, scientific assessment can confirm the results of sensitivity analysis, which, in turn, should be able to explain the results of empirical evaluation of model performance. In practice, full sensitivity analysis is impossible for the more complex models because of the number of parameters that can be varied. In theory a full scientific assessment, is possible for any well-documented model, but again, model complexity often renders this impossible to do completely and quantitatively. All three sources of information have, therefore, been used here.

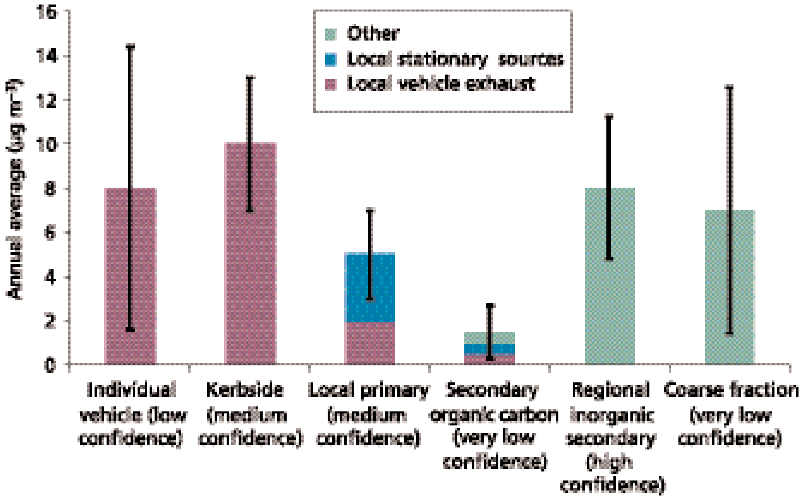
843. In theory, it would be instructive to perform an analysis of the error due to the model itself, separately from the effects of error in emissions data or error in measurements that feed through into the modelling. In practice, it is usually impossible to separate these three main sources of error. The contributions of errors in measurement, emissions and modelling are discussed separately in relation to their impact on our confidence in the overall conclusions of this report in Chapter 9 but here, all three sources of error will be considered together.

8.5.2 Results of uncertainty analysis

8.5.2.1 Annual average past concentrations of PM_{10}

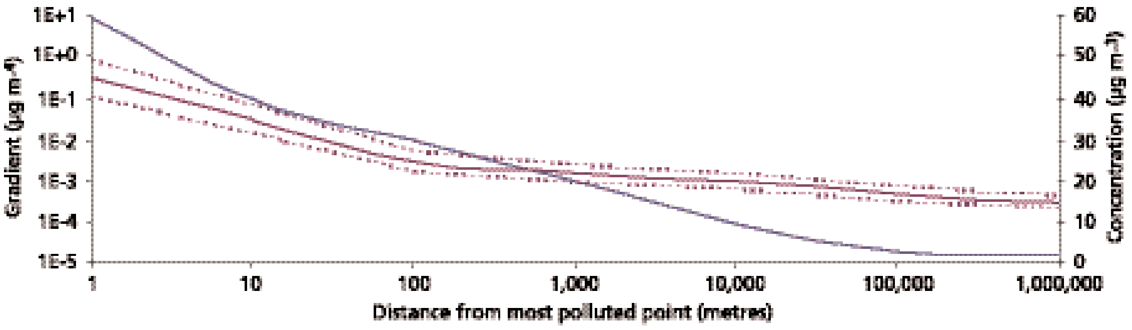
844. Figure 8.57 summarises the results of the uncertainty analysis. The format is taken from IPCC's analysis of causes of climate change. For a detailed discussion of these different types of error and uncertainty, refer to copious explanatory text and footnotes throughout IPCC (2001). The source apportionment of total annual average PM_{10} in Figure 8.56 is based on the general levels and source apportionment reported in Section 8.4. Estimates of the magnitude of the error in the calculated contribution of each component are based on the model evaluation information in Section 8.2. The level of scientific uncertainty described is based on information drawn from Sections 8.2 and 8.3 as well as earlier chapters of this report.

Figure 8.57 Contributions to PM_{10} and uncertainty therein at the most polluted point in the UK, 2001.



845. Figure 8.58 refers to the hypothetical 'most polluted point in the UK'. This is taken to be closer to traffic sources than kerbside, that is, on the carriageway of a heavily-trafficked, canyon-type city centre road, close to individual highly-emitting vehicles. At such a location, the total PM_{10} concentration is the sum of all the bars in the chart. Other locations can be considered by combining the summary data differently. For example, an urban background location is the sum of the four bars to the right of the chart. A rural motorway could similarly be considered as the sum of bars one, two, four, five and a non-urban part of six.

Figure 8.58 Variability in annual average PM_{10} as a function of distance for traffic sources.



846. Bar one of Fig. 8.58 represents the contribution of traffic emissions to the concentration of PM_{10} at a receptor on the carriageway (Gomez-Perales *et al*., 2003). On-street concentrations are highly variable (Arnold *et al*., 2005) and regulatory model paramaterisation dispersion on the carriageway is currently restricted to a parameter describing the gross effect of traffic-induced turbulence designed to correct the kerbside and roadside contributions not the on-street pollution levels. Nevertheless, this contribution is an important part of commuters' daily experience of pollution, but is not accurately determined by models. Being a component of exposure means current air quality management policy does not require it to be included.

847. Bar two represents the contribution of traffic on the nearest road to PM_{10} at a typical kerbside or roadside location. Our ability to quantify this is described as 'medium confidence' since Gaussian Plume and Street Canyon models are believed to contain most of the necessary science to model this, and there is sufficient roadside monitoring to allow empirical models to perform well in this area. The emissions from traffic averaged over the fleet and over a length of road are also relatively well known. The major source of error arises from the discrepancy between roadside models discussed in Section 8.3.1.

848. Bar three represents the local combustion of primary PM_{10} emissions, further away than the nearest road, to urban concentrations. It is described as 'medium confidence' since Gaussian Plume and empirical models have well-characterised reliable performance at this scale. A major contribution to the uncertainty in this component is the lack of consensus on an appropriate emissions factor for domestic and commercial combustion of natural gas that was noted in Chapter 4.

849. Bar four, secondary organic carbon, is highlighted as a component subject to large uncertainty. This is because neither the precursor emissions nor the identification of the important chemical reactions are well known, and the speciated hydrocarbon chemistry with three-dimensional mixing and oxidation reactions is inherently difficult to model. Note that primary particulate organic carbon is not included here, but probably forms a significant part of the roadside and kerbside contributions.

850. Bar five, regional inorganic secondary PM_{10}, is described as 'high confidence' since the basic chemistry and physics are well known, as described in Chapters 2 and 4. Nevertheless, the complexity of models required to quantify this, and debate on the effect of interannual variability of meteorological conditions and photo-oxidant availability, means that the actual error involved in calculating this component is larger than otherwise might be expected. The general level of agreement between models of this component showed in Section 8.4 is encouraging, but there are marked differences in spatial variability and source apportionment that can give rise to a larger error at specific locations. For this component, empirically deriving the concentration from measured speciation or rural sites can be more accurate than indicated in Figure 8.58.

851. Bar six, coarse fraction, is described as 'very low confidence' with large error since its source is not quantified and its spatial distribution from roadside to background and urban to rural is unknown. The size of the bar merely indicates that its contribution could be large, at least at some locations.

852. When the total PM_{10} contribution is computed from the sum of its components, the magnitude of the total error is critically dependent on the extent to which errors in individual components are correlated with each other. Model evaluation exercises reported in Section 8.2 indicate that it is possible to reproduce past measured PM_{10} annual average concentrations at monitoring sites with an accuracy of the order of ±2 $\mu g\ m^{-3}$ or about ±10%. This is a much smaller percentage error than that in individual components. The smallness of the error in total PM_{10} arises partly because some of the larger errors are in components that make a small contribution at the monitoring sites used for model evaluation. But AQEG has concluded that in addition to this, nearly all modellers adopt a process of iteratively comparing their model with measured PM_{10} concentrations and selecting model improvements that tend to reduce differences between measurement and model results. This introduces some empiricism, even into the least empirical models. The resulting tendency is for errors of opposite sign in the modelling of individual components to cancel out to a greater extent than would be the case if they were simply uncorrelated.

8.5.2.2 Future projections

853. Since most models achieve small errors in total PM_{10} at the expense of correct source apportionment, model performance will tend to deteriorate considerably when moving from mapping of past conditions to projection into the future. Where all sources are abated by similar amounts, model performance will be preserved. But where one source is abated much more strongly than others, model performance should be reassessed to take into account explicitly the implications of a factor of two error in the contribution of the most strongly abated component.

8.5.2.3 Other metrics of PM concentration

8.5.2.3.1 90th percentile

854. Model evaluation shows models that calculate daily PM_{10} concentrations before deriving long-term average metrics calculate the 90th percentile concentration with errors no larger than the annual average.

855. Examination of the fit between measured data and the relationships used empirically to convert annual average to 90th percentile in other models shows that an additional error can be introduced at this step, which is the same magnitude as the error in the annual average concentration. The resulting difference in total error in 90th percentile is, however, not large. The difference between ±10% and ±15% is smaller than the difference between one and three standard deviations of model error. Air quality management policy is sufficiently insensitive to the exact probability of exceedence as to demand the more detailed error analysis that would be required to resolve such a small difference.
The convenient rule of thumb that we can calculate the annual average or the 90th percentile both within about ±10% using an empirical or deterministic model therefore still holds.

856. A note of warning, however, needs to be made when modelling the effect of abatement of only one source of PM_{10} instead of all sources together. Analysis of monitoring data discussed by AQEG, but not included in this report, confirms that different components of PM_{10} are often not highly correlated with each other at a given location. The result of this is that the relative change in the 90th percentile will usually (but not always) be smaller than the relative change in the annual average. For example, halving the transboundary secondary component without any local emissions abatement typically might result in a decrease of around one-third in the 90th percentile of total PM_{10}. Models that fail to take account of this can, therefore, systematically over-predict the effects of abatement.

8.5.2.3.2 Higher percentiles

857. Moving towards less than 35 days exceedence gives much more cause for concern over model performance than the change from annual average to 90th percentile. One of advantages of allowing a certain number of days exceedence of a daily air quality standard is that it allows unrestricted emissions from temporary sources such as national and cultural celebrations involving fireworks. This also has an important advantage for quantitative air quality management since emissions from such temporary sources are inherently very difficult to quantify.

858. AQEG, therefore, recommends that decreasing the number of days exceedence allowed should be done only very deliberately and carefully and not simply as a convenient way of tightening an air quality objective. Reducing the number of days exceedence changes the source apportionment of PM_{10} on the days when the relevant exceedences occur and also increases the error involved in modelling the extent and severity of the exceedence. The kind of source that currently can be ignored in air quality management but starts to contribute or even dominate the analysis when the number of exceedences allowed is reduced from 35 per year to seven, is one that might be responsible for 500 $\mu g\ m^{-3}$ of PM_{10} for 90 min per day, two days per week throughout 3 months of the year. It is difficult to assess how widespread sources, such as repeated barbecues or bonfires, are that produce this amount of smoke. Nevertheless, such a source certainly would make a major contribution to exposure and potential health effects of people who are at home at the times when the emissions occur, and so arguably they should be included in air quality management and not merely considered to be a nuisance. Some detailed investigations of intermittent source emissions are, therefore, required to support a reduction in the number of days exceedence allowed.

8.5.2.3.3 $PM_{2.5}$

859. The smaller quantity of monitoring data for $PM_{2.5}$ compared with that for PM_{10}, and the lesser amount of experience modelling this fraction, makes $PM_{2.5}$ somewhat more difficult to model currently than PM_{10}. We note the lack of realism in some of the assumptions in the emissions inventory, especially the lack of fairly obvious changes in particle size with time. But excluding the coarse fraction of PM_{10} from the assessment removes some major areas of scientific uncertainty. AQEG's conclusion overall is therefore that $PM_{2.5}$ can be modelled with accuracy comparable with that of PM_{10}.

8.5.2.3.4 Finer particle size fractions and particle number

860. The availability of measurement data and our understanding of the spatial and temporal variability as well as some of the basic science all decrease on moving to smaller particles than $PM_{2.5}$. Ultrafine particle concentration is more sensitive than $PM_{2.5}$ to the amount of time people spend in close proximity to combustion sources. It would, therefore, be relatively straightforward to consider exposure reduction measures for ultrafine particles, although quantifying these with a high degree of precision is more challenging. Emissions factors for ultrafine particles from some vehicle classes exist, and on-street as well as roadside measurements have been made in a few locations. There is also some debate over the implications of formation of large numbers of ultrafine particles in clean air.

8.5.2.4 Sensitivity of air quality management outcome to model uncertainty

861. Figure 8.58 shows schematically the rate of change of PM_{10} concentration with distance away from a heavily-trafficked city centre location. It is based on a synthesis of the graphs of PM_{10} as a function of distance away from a road, and maps of PM_{10} across London, presented earlier in this chapter.

862. The left side of the graph represents a point on the carriageway of a busy canyon-type road close to the centre of a large city in Southeast England. The right side represents a rural location in the Northwest of Britain. The red line represents a typical annual average PM_{10} concentration at such a location (base year circa 2002) with ±10% considence intervals on a linear scale. The blue line shows the rate of change in concentration with distance on a logarithmic scale. Note the steep gradient, up to 10 μg m^{-3} change in PM_{10} over distances as short as 1 m very close to traffic sources, falling to a rather shallow gradient less than 0.01 μg m^{-3} per m 100 m away.

863. A peculiar phenomenon of PM_{10} is how the gradient decreases much more rapidly with distance than the concentration because of the large contribution of distant sources to the secondary accumulation mode particle mass. Because of this, PM_{10} concentration isopleths enclosing areas between 100 m^2 and 10 km^2 are most sensitive to small errors in concentration – easily a factor of 10 or more for a ±10% concentration error. Even isopleths that are smaller or larger than this can vary in area by a factor of five for the same change in concentration. This should be kept in mind when considering policy implications based on area of exceedence as presented in Chapter 9.

864. Given this sensitivity, and the possibility discussed above that source apportionment errors could be exacerbated in future projections, any policy based on future projection of area of exceedence of an air quality standard is inherently difficult to support quantitatively using modelling.

865. Projected changes in average population exposure can be modelled rather more robustly than the absolute area itself. Numbers of lives saved by air quality management policies can, therefore, perhaps be estimated more easily than absolute numbers of people living at locations where air quality standards are exceeded.

866. Health end-points that are sensitive to short periods of exposure to high concentrations are more strongly influenced by air pollution ‘hotspots’ through which large numbers of people travel per day. In principle, the steeper PM_{10} concentration gradients at such locations make quantification of exposure reduction benefits inherently easier for such short-term effects than is the case for effects of exposure over 24 h or longer, even though the detailed modelling of concentrations very close to traffic is currently an area of some difficulty.

867. If short-term peak exposures to ultrafine particles can be shown to be important, these are subject to even steeper gradients than PM_{10} close to traffic sources, although our understanding of their origin and fate is less strong. On balance, it is, therefore, unclear to what extent assessment of short-term peak exposure to ultrafine particles might be more or less difficult than short-term peak exposure to PM_{10}.

868. Health end-points that can be shown to be caused by exposure to secondary inorganic aerosol can also be quantitatively assessed relatively easily, as large numbers of people are exposed to the same concentration irrespective of the time they spend at more or less polluted locations within an urban area.

869. The regime in which it is most complicated and difficult to draw the boundary of an area of exceedence is the urban background. The combination of high concentration, shallow gradient and contribution of sources at a wide range of distances means that indicators of air quality of pollution impact in these areas need to be selected with care, to avoid making impossible demands on model performance. This is discussed at length in Colvile *et al*. (2002). This is also the environment in which most of our population live, so that moving the boundary of an area of exceedence results in very large numbers of people being excluded or included. Place of residence is not a good indicator of PM_{10} exposure variability for these people, and so greater consideration of the influence of lifestyle and daily movement on their exposure to outdoor sources of air pollution might allow much better use to be made of the ability of current modelling and mapping techniques to identify opportunities to reduce exposure.

8.6 Conclusions and recommendations

870. The chapter has presented details of a range of models of varying complexity for different spatial scales that are used for air quality assessment and policy development in the UK. Although there is significant divergence between the models, especially for future projections, the following general conclusions can be reached regarding the limit values for 2005 and 2010.

871. In 2005 the annual limit value 40 $\mu g\ m^{-3}$ is likely to be achieved throughout the UK. However, the daily limit value of 35 exceedences of 50 $\mu g\ m^{-3}$ may not be achieved near major roads in urban areas in particular in London and, more especially, in meteorological conditions that lead to high background levels transported from mainland Europe.

872. In 2010 the Stage II indicative annual limit value will be exceeded in many urban and roadside areas. Models diverge significantly on the likely exceedence of the indicative daily limit value of seven exceedences of 50 $\mu g\ m^{-3}$.

873. The model descriptions and uncertainty analysis have highlighted significant areas where the reliability of model calculations is limited. These include the coarse fractions – both urban background and roadside, the treatment of secondary organic carbon and the variability of roadside concentration with road characteristics. In contrast, the secondary component is well modelled at least for annual averages.

874. Model development is best focussed on areas of model uncertainty that have significant impacts on calculated concentrations.

875. It is recommended that models are developed for the coarse component both from the urban background, for example, from construction sites and from traffic – both direct emissions and non-exhaust traffic emissions. Any significant development will require improved parameterisation of these emissions. Secondly improved modelling and understanding of the impacts of road characteristics including traffic induced turbulence, vehicle exhaust height, urban topography etc. These will both increase confidence of the road type adjustment factors and improve the reliability of dispersion models adjacent to roads.

876. Further modelling of $PM_{2.5}$ should also be conducted in anticipation of new air quality standards. However, reliable model validation will require appropriate resolution of the difference between the different measurement techniques (for example, stipulation of appropriate conversion factors and so on).

Chapter 9

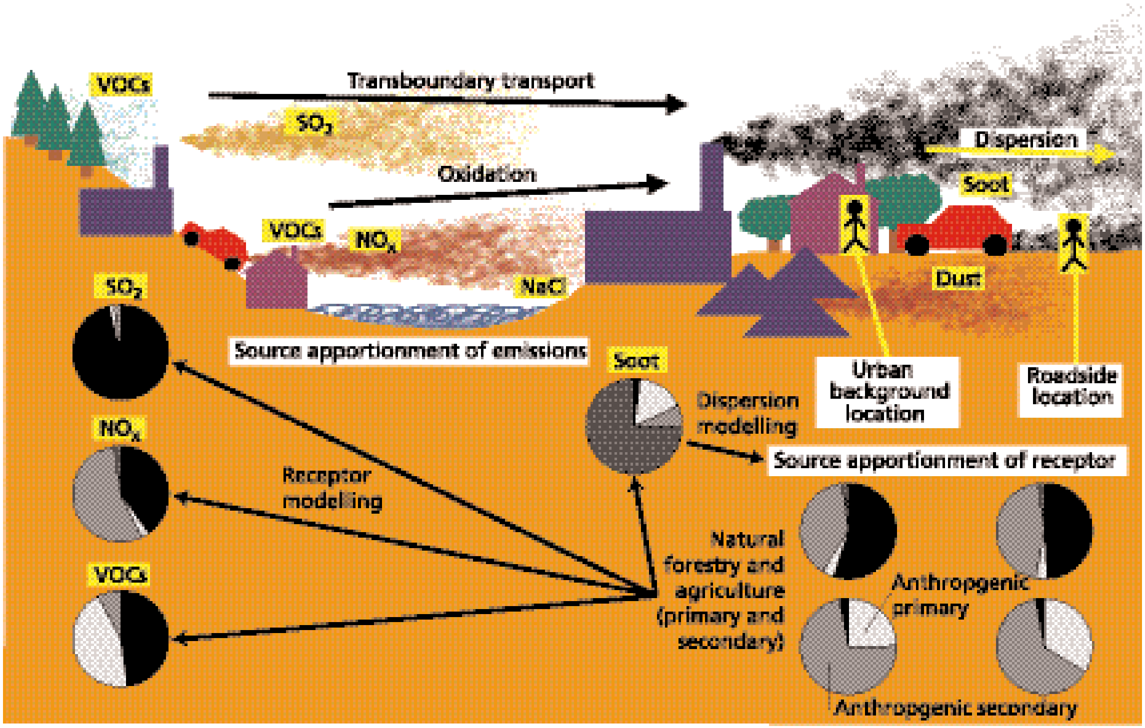
Discussion

9.1 Introduction

877. This chapter provides a direct response to the questions set for AQEG by Defra and the Devolved Administrations (see Chapter 1). It draws extensively upon the emissions, measurement and modelling data described in detail within previous chapters.

878. An important recurring theme throughout Chapters 2–8 of this report is source apportionment, which was introduced in Section 2.5. PM has many different sources, and it is important to identify these correctly. Figure 9.1 illustrates approximate PM_{10} source apportionment by source location, source type and particle type, summarising the main conclusions of Chapters 4, 6 and 8. The sources of PM and precursor gaseous emissions are shown to the left side of the figure. Dispersion, pollutant transport and transformation occurs from left to right across to receptors on the right hand side.

Figure 9.1 Summary of PM_{10} source apportionment and methods of assessment used in this report (see text for detailed explanation).



879. On the far left of Figure 9.1, farthest upwind of the receptors, are the gaseous precursors of secondary PM that were described in Figure 4.7. The colours of the pie slices correspond to the colours of the major source types shown on the diagram: industrial (blue); transport (red); domestic and commercial (purple); and natural (green).

880. At the centre of Figure 9.1 are the sources of primary PM that were described and quantified in detail throughout Chapter 4. These are most significant closer to the receptors than the gaseous secondary PM precursor emissions sources. The pie chart indicates a typical source apportionment of combustion-related emissions for a major UK city. Note that sea salt and fugitive dust are additional local sources of primary PM, partly natural and partly anthropogenic, that are not included in this pie.

881. The right-hand half of Figure 9.1 shows how the emissions, dispersion, pollutant transport and transformation processes add up to produce the current annual average source apportionment of concentration at two types of receptor in a typical UK city, an urban background and a roadside location. The precise contribution of each source depends on the location of the receptor relative to the source and relative to local topography. For local sources of primary PM, changes of a few kilometres in the location of an urban background receptor (Sections 6.2.1 and 6.2.2) or a few metres for a roadside one (Section 6.2.4) can cause the source apportionment to change. For secondary PM, changes of a few hundred kilometres will produce a difference of similar magnitude (Section 6.2.6). The source apportionment shown is typical of a large city in Southeast England; for Scottish, Welsh, Northern Irish and Northern English cities, the secondary contribution is smaller because they are further from mainland Europe. Smaller cities have similar source apportionment, except the contribution of local primary emissions to urban background concentrations tends to be smaller and similar source apportionment can be found in the suburbs of larger cities.

882. The coloured receptor source apportionment pie charts in Figure 9.1 have the same key as the emissions pie charts, indicating typical contributions of industrial, transport, domestic and commercial and natural source types. These include the contribution of these source types to gaseous precursors of secondary PM and fugitive and sea salt primary emissions, in addition to primary combustion-related emissions.

883. The monochrome receptor source apportionment pie charts in Figure 9.1 indicate approximate contributions of anthropogenic primary (including fugitive dusts and road surface emissions), anthropogenic secondary and natural particle types to the total annual average PM_{10} (based on Table 8.6(b)). Note that 'natural' includes sea salt and secondary PM from biogenic emissions of VOCs.

884. The arrows on Figure 9.1 linking the various parts of the diagram indicate processes that occur in the atmosphere as well as modelling and data analysis methods used in this report. The arrows from left to right representing dispersion, pollutant transport and transformation reflect processes that are parameterised in the emissions-based models that were described in Chapter 8 (see paragraphs 748 and 749 and also Sections 2.5.2, 8.2.2.2, 8.2.3 and 8.2.4.2.). The arrows from right to left indicate how source apportionment can be deduced from measurements as discussed in Section 2.5.1 (described in more detail in Sections 8.2.1 and 8.2.2.1.).

9.2 Answers to questions

885. The level of confidence with which AQEG has been able to provide answers to the questions posed in Chapter 1 varies. For PM, difficulties arise for several reasons. Firstly, the level of scientific uncertainty in our understanding of the underlying processes contains some gaps. Secondly, as a consequence of these gaps, some of our quantitative calculations and estimates lack precision. Thirdly, some properties of PM make this an inherently difficult pollutant to assess in accordance with the requirements of current policy. This sometimes causes inconsistency between the results of different models or between models and the analysis of measurements. The recommendations for future research set out in Chapter 10 have been devised with the aim of resolving some of these problems.

9.3 Are the current assessment methods (emissions inventories, measurements and modelling) fit for purpose? How could they be improved?

9.3.1 Answer

886 The methods of assessment are considered fit for purpose in terms of characterising current annual mean PM concentrations at urban background and rural locations and the approximate magnitude and distribution of concentrations at roadside locations. The methods of assessment become progressively more uncertain when used to quantify precise areas of exceedence, future concentrations and exceedences of the daily mean limit values.

9.3.1.1 Emissions inventories

887 Emissions of primary combustion PM, and emissions that form secondary inorganic particles (NO_x, SO_x and NH_3) are well established. Emissions from other sources, mostly particles in the coarse fraction, are either very uncertain (for example, from quarrying and agriculture) or poorly understood (for example, non-exhaust vehicle emissions). The latter adds significantly to the difficulty of predicting roadside concentrations.

888 Emissions from intermittent and natural sources (such as forest fires) are not well defined, but may contribute to exceedences of the daily mean limit value.

889 For all source categories, emissions inventories for future years (for example, 2010) have a greater degree of uncertainty attached to them compared to the current year.

890 The PM_{coarse} component, between 2.5 and 10 µm, makes a large but ill-defined contribution to total PM_{10} concentrations, and further work to better quantify this component is required. In particular, the contribution from non-exhaust road traffic emissions needs to be better quantified. Specific recommendations are provided in Chapter 10.

9.3.1.2 Measurements

891 There is an extensive network of PM_{10} monitoring in the UK that provides invaluable information on current concentrations and trends. The networks are principally founded on the TEOM analyser, which produces continuous hourly data that are valuable in analysing patterns of exposure and source apportionment. However, the elevated temperature of the TEOM inlet causes inevitable losses of semi-volatile species and makes direct comparison with the European reference sampler extremely difficult. To enable comparison of measured concentrations with the UK objectives and EU limit values, TEOM data are currently adjusted by applying offset and scaling factors, which impedes any accurate assessment of compliance with the objectives and limit values, particularly in future years. However, despite these problems, AQEG does not consider the use of gravimetric techniques (such as the reference sampler) to be a viable alternative, as these methods introduce other problems.

892. There are several ways in which the current monitoring networks could be improved: with the addition of more rural PM_{10} sites, additional monitoring of $PM_{2.5}$ concentrations and further monitoring of the chemical composition of PM. Specific recommendations are provided in Chapter 10.

9.3.1.3 *Modelling*

893. The modelling of particulate concentrations is inherently more complex than for other common pollutants. This is because of the need to combine the contributions from different sources, for example, long-range transport of secondary PM, primary contributions from urban sources and very local contributions from individual roads. A substantial ‘residual’ contribution is also included in many models, but is poorly understood. Models perform reasonably well for current years, but the complexity of PM and the manner in which source contributions may change adds to the difficulty of predicting future concentrations.

894. In general terms, the models are better at predicting the annual mean concentration rather than the number of daily exceedences in a given year and are better at predicting the changes associated with a prescribed change in emissions rather than the absolute PM concentration.

895. Further work is required to improve and refine the models and to check their accuracy with respect to the different components and source apportionment. Specific recommendations are provided in Chapter 10.

9.3.2 Rationale

9.3.2.1 *Emissions inventories*

9.3.2.1.1 *Uncertainty of annual emissions inventories*

896. The analysis carried out for the NAEI gives an overall uncertainty for the annual rate of UK emissions of PM_{10} of –20% to +50% at the 95% CI. However, uncertainties in emission estimates for particular sources and in particular locations at particular times, will be considerably higher, for example, in the case of emissions near a roadside or near sources of mechanically generated PM. This partly reflects the level of detail required to understand and quantify localised emissions due to their transient nature (for example, vehicle exhaust emissions) or the amount of local activity (for example, level of construction activity), which may be poorly understood.

897. Emissions of the finer particle size fractions ($PM_{2.5}$ and smaller) are more difficult to quantify and are less certain than estimates for PM_{10} emissions. Although there are increasing data from research on the particle size distribution of emissions from vehicle exhausts, there are very few data for other combustion and non-combustion sources.

9.3.2.1.2 Local and episodic emissions

898. The limit values and objectives place an emphasis on PM concentrations at 'hotspots', such as busy roads, and there is, therefore, a need to characterise emissions at such locations. This is dependent on traffic data, which introduces additional assumptions and uncertainties; it is also difficult to allow for the 'stop-start' flows in congested conditions and at junctions. Coupled with uncertainties regarding the non-exhaust component of vehicle emissions, in particular the resuspension of road dusts by traffic, the quantification of local emissions at critical urban locations causes significant difficulties.

899. Ambient concentrations of PM are also affected by short-term emissions, for example, construction/demolition activities, road works and bonfire night, and exceedences of the daily mean limit values may be directly associated with such events. Emissions inventories are unable to describe the temporal variation in emissions from these source types, as many are unique or random in occurrence and some are influenced by external factors such as meteorology. Many of these processes are not amenable to inclusion in emissions inventories.

9.3.2.1.3 Improvements to the annual emissions inventories

900. An analysis carried out by the NAEI found that the major contributors to uncertainty at a national level are emissions from a number of sectors, including quarrying and construction, domestic combustion of solid fuels and gas, tyre and brake wear, coal-fired power stations, Part B industrial processes and agricultural livestock housing.

901. The NAEI has recently carried out a review of a number of these sources. Emission factors for tyre and brake wear have recently been revised leading to an overall increase in national emissions. Current estimates of emissions from the domestic combustion of gas have been revised downwards, but there remains a lack of robust data. The review also suggested changes to the activity data and to emission factors for the quarrying and construction sectors that would lower the current national PM estimates. However, it remains difficult to accurately quantify emissions from any of these sectors and uncertainties remain high.

9.3.2.1.4 Emissions in future years

902. The prediction of concentrations in future years depends heavily on accurate forecasting of changes in emissions of both primary PM and secondary precursors in the UK and Europe. This requires assumptions to be made as to how individual sectors will evolve, which becomes increasingly uncertain as forecasts are projected further into the future.

903. For road transport, emissions projections are based on the expected implementation of new vehicle emissions standards. They also depend on factors affecting the make-up of the vehicle fleet, particularly the mix of petrol and diesel cars. A study by the NAEI has demonstrated the importance of assumptions made regarding the proportion of diesel cars. The current assumption that 30% of all new cars sold will be diesel by 2010 now appears to be an underestimate because current sales are rapidly increasing towards this level. The impact of increasing diesel car sales to 42% by 2010 would be to increase urban PM exhaust emissions from road transport by 10% above levels previously forecast for 2015.

904. Emission projections for the power generation, industry and domestic sectors are based upon energy and economic growth forecasts. The accuracy of emissions projections will depend upon the implications of energy policies and how they are implemented as well as the uptake rates and effectiveness of new technologies. Where these are linked to legislative requirements, they can be predicted with some certainty, but general improvements are more uncertain.

9.3.2.2 Measurements

905 Because of the complex nature and composition of airborne PM, the method that is selected for the collection and determination of particle mass significantly influences the result. The requirements for PM_{10} monitoring are very much dependant upon the eventual use of the data. For reporting against the limit values, as set out in the First Daughter Directive, it is a requirement that the European reference sampler or equivalent is used. The current UK approach of adjusting measured TEOM concentrations using an internal adjustment offset and a further scaling factor of 1.3 is unlikely to prove satisfactory beyond the immediate term. However, restructuring of the networks based on the reference sampler is not considered to be a viable alternative, as this approach does not provide the required time resolution of data and is also subject to both positive and negative artefacts, depending upon environmental conditions.

906. A programme of work to evaluate new instrumentation for the automatic monitoring of PM_{10} concentrations is currently under way and, depending upon the outcome of this work, the existing networks may need to be modified or replaced. It is noted that if the reference method were changed so that only the non-volatile component of the particle mass were measured, this would greatly improve the consistency and practicality of measurements for regulatory purposes. These measurements of ‘core particulate’ mass could then be supplemented at representative sites by specific monitoring techniques to determine the volatile species.

907. A detailed analysis of the TEOM adjustments and their implications is given in Annex 6. This discusses how removal of the internal offset would lead to the need for a larger ‘default scaling factor’ and provides illustrations of how such changes could alter estimated compliance or non-compliance. Removal of the internal offset would also affect the empirically modelled concentrations. For sensitivity analysis with respect to compliance with the limit values (discussed in Section 9.4), the implied differences are of the order of 1 $\mu g\ m^{-3}$ in the typical examples given.

9.3.2.2.1 Relationships between episode days and annual means

908. The application of a uniform scaling factor to daily TEOM concentrations significantly affects the estimated number of days above the daily mean limit value. In practice the volatile fraction lost will vary from day to day and it is highly unlikely that a single factor could account for this. This also has significant implications for modelling results where the number of days $>50\ \mu g\ m^{-3}$ is estimated from a relationship with the annual mean. Such assumptions critically influence the number of predicted exceedences of the daily mean limit value.

9.3.2.2.2 Spatial representation

909. Roadside concentrations can vary both spatially and temporally within a street. For this reason a statistical picture from fixed long-term sites is important both in establishing the extent of exceedence and trends, and it is recommended that greater attention is given to the pairing of background and roadside sites to improve definition of roadside increments.

910. A particular deficiency in the existing national network is the paucity of rural sites: currently, measurements of PM_{10} are being carried out at only four rural locations. More rural sites would help to clarify the urban enhancement, enabling checks of consistency with current source apportionment and modelling estimates. They would also improve the definition of background concentrations for non-urban AQMAs. Specific recommendations are set out in Chapter 10.

9.3.2.2.3 Measurements of $PM_{2.5}$

911. Monitoring of both $PM_{2.5}$ and PM_{10} is currently carried out at only a small number of sites. Generally, the data indicate a close correlation between the two metrics and provide important information on the PM_{coarse} fraction. If, however, the limit values were revised to include $PM_{2.5}$ this would require either:

- considerable expenditure in more direct measurements of this size range – this would be both costly and difficult to implement at all sites due to space limitations;
- reliance on derived correlations – but there would be only extremely limited historic data from which to establish such relationships; or
- reconfiguration of the existing PM_{10} network towards $PM_{2.5}$ – this would result in the loss of valuable trend data in PM_{10}.

912. Irrespective of whether new limit values are set for $PM_{2.5}$, AQEG recommend more co-located hourly measurements of $PM_{2.5}$ and PM_{10} at selected sites to characterise the relative contributions at roadside, urban background and rural sites. In conjunction with other measurements this would help to clarify the PM_{coarse} fraction and its spatial and temporal variation. Specific recommendations are provided in Chapter 10.

9.3.2.2.4 Chemical composition and source apportionment

913. It is important to develop a better understanding of the different source contributions to PM concentrations, including the primary and secondary components and long-range versus local contributions. The UK Acid Deposition Networks provide data on sodium, ammonium and particulate nitrate (together with gaseous HNO_3) based upon daily or monthly sampling, but these are rural networks that do not indicate whether there is significant enhancement of nitrate in major cities. In addition, both particulate nitrate and sulphate make a substantial contribution to the regional background levels upon which urban and roadside increments are superimposed. There is a requirement to more fully understand the particulate nitrate and sulphate concentrations, particularly during PM episodes, and to obtain measurement data that can be used to validate the regional model assessments and support source apportionment studies.

914. Particulate iron is a valuable indicator of the non-exhaust road traffic emission component of PM_{10} emissions. As discussed in earlier chapters, non-exhaust traffic emissions may become of even greater importance in future years, and a more accurate quantification of this source is needed.

915. AQEG recommends expansion of the networks to include paired urban background/rural monitoring of continuous nitrate and sulphate and paired roadside/urban background monitoring for continuous elemental and organic carbon measurements. It is also recommended that existing networks are modified and extended to include additional daily measurements of iron. Specific recommendations are provided in Chapter 10.

9.3.2.3 *Modelling*

916. The manner in which different models account for the different source contributions is key to understanding their performance and how improvements may be provided in the future.

9.3.2.3.1 *Secondary inorganic component*

917. The Netcen, ADMS and ERG models incorporate a simple treatment of the secondary inorganic component, based on measurements. They are all thus highly empirical and do not address the effect of reducing UK precursor emissions directly, but use scaling factors to project into the future using different assumptions. Recent calculations based on the Eulerian EMEP model for current scenarios within the CAFE programme indicate somewhat greater reductions in secondary inorganic aerosol up to 2010, which suggests that assumptions made in the UK may be on the conservative side.

9.3.2.3.2 *Primary PM*

918. The contribution of primary particles is accounted for either directly from dispersion of the emissions, or empirically from regression analysis against the measurements. Annex 7 (Table A7.1) provides an illustration of source apportionment calculations for both PM_{10} and $PM_{2.5}$ based upon calculations using the ADMS model for selected sites in London.

919. The results show that predicted exceedences at roadside locations are highly dependent on the increment due to road traffic. However, this is variable, depending upon characteristics of the local road and the traffic including street canyon effects, vehicle speeds, traffic densities, street layout and traffic control schemes. Research funded by EPSRC is in progress (DAPPLE[1]) to gain a better understanding of dispersion and concentrations within urban streets. Uncertainties of at least a factor of 2 in the prediction of the roadside increment for annual mean concentrations are likely at some locations.

920. Empirical estimates of roadside increments are an alternative to deterministic modelling, but are hampered by the paucity of paired roadside and background sites.

9.3.2.3.3 *Residual component*

921. The residual component accounts for particulate emissions, largely within the PM_{coarse} fraction, that are not well-characterised or are not included in emissions inventories, such as wind-blown dust and sea salt. This component is poorly understood within models. Some models simply add a constant background (both in space and time) based on measurements. Other models explicitly account for the rural background contribution from rural monitoring data (although these

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[1] DAPPLE (Dispersion of Air Pollution and Penetration into the Local Environment) is a 4-year consortium project. See www.dapple.org.uk for details.

data are very limited) with a constant urban coarse component then added. Despite these uncertainties, the residual component represents a substantial fraction of the whole. There is good reason to believe that a part of this component is associated with the suspension of road dust by traffic, hence contributing a significant increment to roadside emissions, which importantly is not expected to reduce in the future. Different assumptions regarding both the scale and spatial representation of this residual component would have a very significant impact upon estimated exceedences in future years.

9.3.2.3.4 Air pollution episodes and daily limit values and objectives

922. It is also a requirement for models to consider exceedences of the daily mean limit values and objectives, which is considerably more challenging. Such episodes are highly correlated with meteorology and are thus subject to large interannual variation. Even more sophisticated models are challenged in representing the dispersion and chemical and physical characteristics of particle behaviour in such episodic conditions.

923. The Netcen, ADMS and ERG models take a simplified but different approach to assessing the number of daily exceedences. These approaches include the derivation of a surrogate annual mean concentration based on regression analysis with daily mean exceedences or direct modelling of the individual days based on the predicted primary PM component superimposed on the secondary PM derived from rural measurement and a constant 'residual' component. Each approach is subject to additional uncertainty over and above that associated with predictions of the annual mean concentration. For example, the models are unable to directly predict high nitrate episodes (likely to be of increasing importance) or deal with episodes due to more local natural or intermittent sources that are not adequately represented in emission inventories. Validation (see Table 8.2) suggests that the approaches used provide useful predictions of the number of exceedences of 50 µg m^{-3} for comparison with the 2005 limit value (daily mean to be exceeded no more than 35 times), but with increasing underestimates/uncertainty for the higher percentiles required by the indicative Stage II limit value (no more than 7 exceedences). In the latter case the lack of consideration of local intermittent sources, including coarse particle sources, and limitations of model dispersion in extreme conditions becomes more important.

9.3.2.3.5 Links between modelling and measurements

924. The model predictions considered in this report have been 'calibrated' to match the measurements, either by adjustment of parameters (for example, urban boundary layers) or by application of multiplying factors (derived from regression analysis) to particular components of the particulate concentration. Such procedures could lead to an underestimation, or more critically an overestimation of the reduction in PM concentrations associated with a reduction of the corresponding emissions. For example, if the model does not accurately represent traffic-related non-exhaust emissions, calibration of the model against current monitoring data may result in an underprediction of PM concentrations in future years. Uncertainties of the order of 1–2 µg m^{-3} in projected annual mean concentrations might easily be expected to occur.

9.3.2.3.6 *Interannual variation and extrapolation of trends*

925. There are additional difficulties with future predictions of PM concentrations. Interannual variability introduces difficulties for establishing trends and extrapolating into the future; in particular, 2003 was an unusual year, with very high temperatures and episodes of significantly elevated nitrate concentrations during the spring. Concentrations during 2003 were typically higher than those measured during 2002, despite a likely slight decrease in manmade primary emissions and secondary particulate precursor emissions. It is difficult to judge how exceptional this year was or how the potential effects of climate change could affect future pollution episodes and overall concentrations. As a result, the model results presented in this chapter for comparison with limit values relate to more average years of meteorology.

9.3.2.3.7 *Other uncertainties when assessing exceedence of limits*

926. The modelling results reported later in this chapter are concerned with local areas of exceedence along major roads and wider areas of exceedence at urban background locations. As discussed by Colvile *et al*. (2002), a small perturbation in a relatively smooth concentration field of the order of 10% (or 1–2 µg m^{-3}, well within the range of uncertainty) can make a large difference to the concentration isopleths, moving them by possibly several kilometres and causing the appearance and disappearance of 'blob-shaped' areas of exceedence. Similarly, different assumptions about the residual component and its spatial variability could also make a significant difference. As a result, predicted areas of exceedence are highly sensitive to uncertainties and assumptions.

927. The trend towards higher percentiles to tighten legislation on air quality is welcomed as a means of protecting the public from infrequent exposure to higher concentrations, but causes difficulties with respect to the capabilities of current assessment methods that effectively ignore a number of local and intermittent sources. The implications of moving from 35 days exceedence of the daily mean limit value to only 7 days exceedence (as proposed in the indicative Stage II limit values) needs to be carefully considered, bearing in mind the limitations of the available analysis tools. Current modelling technology used in the 2002/2003 analysis conducted for the AQS Addendum considered Stage II limit values as objectives across the UK: Scotland put the Stage II limit values into regulation for the purpose of LAQM.

9.4 Are there sources missing from (a) UK emissions inventories and (b) other European inventories?

9.4.1 Answer

928. The NAEI is a comprehensive database of emissions for the UK. A number of potential sources of PM emissions are not included in the NAEI because there are insufficient data available, but these are not considered to be significant and it is estimated that their inclusion would only increase the UK estimate of PM_{10} emissions by no more than a few percent.

929. In addition, there are also many important natural sources of particles – including sea salt, forest fires and wind-blown dust – that contribute significantly to ambient PM concentrations, but which are not included in the NAEI. These sources are difficult to quantify and are not amenable to inclusion within national inventories. Although estimates of PM emissions from both non-exhaust road traffic emissions and construction activities are included in the NAEI at a national level, it is extremely difficult to derive representative emission rates at a local level. Nationally, these sources make only a small contribution to total PM emissions, but at a local level (close to major roads or construction and demolition sites), they may be extremely important.

930. A detailed analysis of European emissions has recently been carried out by the CAFE Working Group on Particulate Matter. This report has drawn extensively upon the conclusions of the Working Group and no further analysis has been undertaken.

9.4.2 Rationale

931. A recent study has identified a number of potential sources of PM emissions that have been omitted from the NAEI (see Chapter 4). Although their impact at a national level is not considered important, they could be significant at particular times at a local level, contributing to exceedences of the daily mean limit values. In some cases, insufficient data are currently available to allow quantification of these sources, but in other cases emissions estimates will be included in future revisions to the NAEI.

932. Many natural sources of particles are random in origin and cannot be accurately quantified in terms of their magnitude or temporal characteristics, and they are best described through ambient measurements and source apportionment studies. Nonetheless, it is recognised that such sources can make significant contributions to particle concentrations particularly during ‘episode’ conditions, for example, impacts of sea salt aerosol and Saharan dust, which are described in Chapter 6.

933. Emissions associated with non-exhaust road traffic sources are included in the NAEI, but are poorly understood. At a national level this is unlikely to be an important source, but at a local level it may be extremely important, perhaps more so than exhaust emissions. Studies indicate that HGVs are more important in terms of non-exhaust road traffic emissions than smaller vehicles, such as cars, but as the rate of resuspension is governed by many local factors, such as the road surface condition and meteorological conditions, it is currently difficult to derive emission rates that can be included in inventories in the normal manner.

934. Emissions estimates from construction activities are also included in the NAEI, but again are poorly understood. At a local level, emissions may be very significant, particularly if the activities are scheduled over a long time period. However, as emissions are governed by a wide range of local factors, such as the size and nature of the works, the meteorological conditions and the application of abatement measures, it is again extremely difficult to derive emission rates in the normal manner.

9.5 Is the UK likely to achieve, with current abatement measures and technologies (a) the Stage I and indicative Stage II annual and daily mean PM_{10} limit values in the First Air Quality Directive and (b) the Air Quality Strategy objectives for PM_{10}? If not, why not? What levels of PM_{10} are likely to be achieved by current measures and policies?

9.5.1 Answer

935. For the air quality objectives and Stage I limit values to be met by 2004/2005, the daily mean target will be more difficult to attain than the annual mean, and there will almost certainly be some exceedences along major roads, mainly in London. It is difficult to be more precise due to the interannual variability in the measurements and uncertainties in the modelling. There are also areas affected by non-traffic sources, which have been identified by local authorities as part of their duties under LAQM, where additional exceedences may occur.

936. With respect to the indicative Stage II limit values, based on current measures and policies there will be many urban background and roadside locations where these targets will not be achieved. These exceedences are more extreme in London and remain even if the less stringent provisional objectives for 2010 are considered. Although exceedences of the Stage I annual mean limit value are very small in Scotland, a much wider area of exceedence of the 2010 objective is predicted.

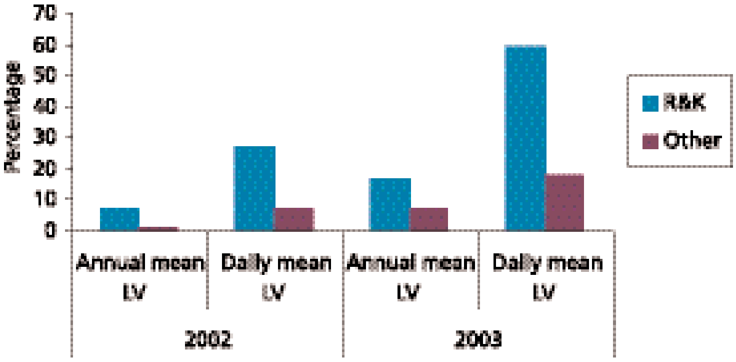
937. There are a number of different sources contributing to the predicted exceedences in both 2004 and 2010, and it is not possible to single out any specific sector. A more detailed analysis of source apportionment is provided in Section 9.7.

9.5.2 Rationale

938. For the Stage I limit values and air quality objectives to be met by 2004/2005, it is expected that there will be little change from current measured concentrations. Figure 9.2 shows a summary of the measured PM_{10} concentrations in 2002 and 2003 compared with the Stage I annual mean and daily mean limit values (a more detailed description of the results is provided in Annex 7, Tables A7.2 and A7.3).

939. Annual mean PM_{10} concentrations in 2002 and 2003 were below 40 µg m^{-3} at the majority of monitoring sites, with exceedences largely confined to roadside and kerbside sites in London. The daily mean limit value was more widely exceeded, particularly at roadside and kerbside sites. There were more exceedences in 2003 than in 2002.

Figure 9.2 Percentage of UK PM_{10} monitoring sites in 2002 and 2003 exceeding the Stage I annual and daily mean limit values. (R&K, roadside and kerbside sites; other, all other site classifications. Data collected using TEOM analysers have been adjusted using a 1.3 scaling factor.)



940. Corresponding estimates of predicted exceedences in 2004/2005 have been derived using the Netcen, ADMS and ERG models. Table 9.1 describes the results for London and the UK, indicating areas of exceedence at both roadside and urban background locations. The results are broadly consistent with the measured results for 2002, with a small number of exceedences of the annual mean limit value and a much greater number of exceedences of the daily mean limit value. As expected, exceedences are predominantly located along major roads. These modelling results are based on average meteorology; in a year with more extreme meteorology such as 2003, the number of exceedences may be significantly higher.

9.5.2.1 AQMAs and exceedences away from roads

941. As part of their duties under the system of LAQM, local authorities are required to carry out regular reviews and assessments of air quality in their areas and to identify locations where one or more of the air quality objectives is unlikely to be met by the relevant due date. Where such an exceedence is identified, the authority must designate an AQMA and prepare an action plan setting out measures in pursuit of meeting the objectives.

942. Of the 433 local authorities in England, Scotland, Wales and Northern Ireland, a total of 57 have designated AQMAs for PM_{10}, largely associated with road traffic emissions. However, AQMAs have also been designated in other areas, associated with fugitive dust sources from industrial processes and materials/waste handling activities, and it is anticipated that additional AQMAs associated with domestic solid fuel combustion will be designated in Northern Ireland. Action plans will be required for each of these areas; further work is underway to integrate as far as possible the local and national assessments.

9.5.2.2 Achievement of Stage II limit values and provisional air quality objectives

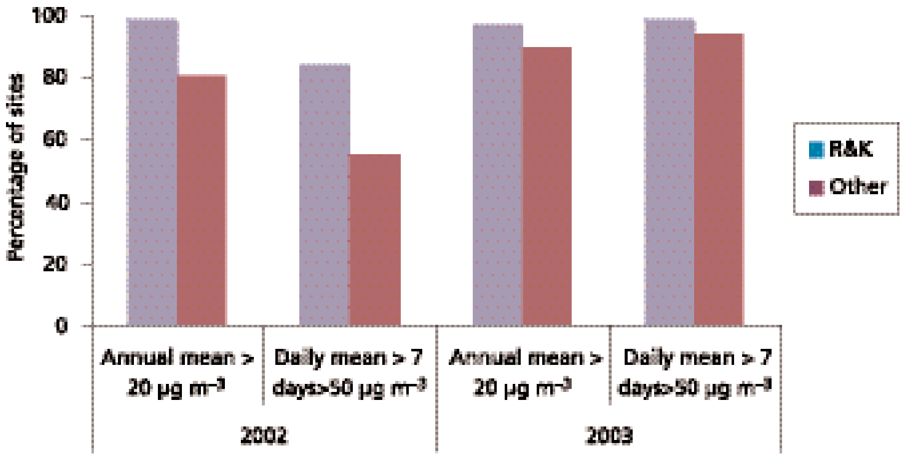
943. Figure 9.3 shows a summary of the measured PM_{10} concentrations in 2002 and 2003 compared with the Stage II annual mean and daily mean limit values (a more detailed description of the results is provided in Tables A7.2 and A7.3 in Annex 7). For both 2002 and 2003, the measured annual mean PM_{10} concentration widely exceeded 20 μg m^{-3} at both roadside and kerbside and background sites. The daily limit value (no more than 7 days >50 μg m^{-3}) is exceeded at a smaller number of sites in 2002, but at a greater number in 2003.

Table 9.1 Modelled exceedences of objectives for 2004/2005.

Statistic	Model	Area	Road length (km)	Base case	Base case -2 µg m^{-3}	Base case + 2 µg m^{-3}
Annual mean >40 µg m^{-3}	NETCEN	UK	16037	9	—	—
	NETCEN	Greater London	1786	9	2	19
	ADMS	Greater London	3651	0	0	0
	ERG	Greater London	4814	5	3	9
Daily limit more than 35 days >50 µg m^{-3}	NETCEN	UK	16037	269	106	589
	NETCEN	Greater London	1786	119	60	236
	ADMS	Greater London	3651	0.48	0	3
	ERG	Greater London	4814	81	71	91
Annual mean >40 µg m^{-3}	NETCEN	UK	242349	0	0	0
	NETCEN	Greater London	1624	0	0	0
	ADMS	Greater London	1574	0	0	0
	ERG	Greater London	1858	0	0	0
Daily limit more than 35 days >50 µg m^{-3}	NETCEN	Greater London	1624	1	0	3
	ADMS	Greater London	1574	0.14	0.03	0.43

NB: Base case ±2 µg m^{-3} indicates the likely range of exceedence.

Figure 9.3 Measured PM_{10} concentrations in 2002 and 2003 compared with the indicative Stage II annual and daily mean limit values, showing the percentage of sites exceeding the proposed limit value in each year. (R&K = Roadside and kerbside sites; Other = all other site classifications. Data collected using TEOM analysers have been adjusted using a 1.3 scaling factor.)



9.5.2.3 *Site-specific projections*

944. An analysis of predicted annual mean PM_{10} concentrations in 2010 based on a combination of observed measurements and site-specific modelling has been described in Chapter 8. These predictions show that although PM_{10}

concentrations closely approach the indicative Stage II annual mean limit value, exceedences still occur. To a large extent, these exceedences are driven by the change from a steeply declining trend in measured PM_{10} concentrations (over the period 1992 to 2000) to a levelling off in recent years.

945. The observed trends are complex, showing considerable year-to-year variation that is determined largely by meteorology and, in particular, by variation in the contribution from secondary particles and other long range transport, such as Saharan dust events. This leads to years with relatively high concentrations (1996, 1997 and 2003) and relatively low concentrations (1998 and 1999). The preliminary monitoring results for the first part of 2004 are in much closer agreement with projections from previous years, suggesting that the upward measured trend to 2003 may be reversed with the long-term trend gently downwards in line with PM_{10} emissions reduction.

946. It is considered likely that the reversal of trends in recent years reflects some influence of changing weather patterns that has masked the influence of decreasing PM emissions. This could be some form of climate drift, in which case it may be a permanent feature of the future PM climate of the UK or it may just reflect year-on-year variability. Regardless, it is considered unlikely that the indicative Stage II limit values will be met by 2010.

947. There are a number of assumptions inherent in the site-specific model itself but it is considered a reasonably reliable tool for estimating future annual mean PM_{10} concentrations, as it has shown good agreement with past trends. There are, however, two important limitations.

- The assumption of a constant residual particle contribution when levels may be increasing.

- The difficulties resolving the apparent differences between roadside and kerbside increments and emission inventories in terms of either locally resuspended coarse material or the underestimation of fine particle emissions due to the emission of semi-volatile organic compounds from diesel fuels or lubricants. The site-specific projections for roadside sites could be systematically optimistic as they do not include an explicitly increasing contribution from resuspended dusts due to increased traffic movements.

9.5.2.4 GIS-based modelling results

948. Exceedences of the indicative Stage II limit values in 2010 have also been derived using the Netcen, ADMS and ERG models. The results are described in Table 9.2 (and in more detail in Chapter 8) and show a reasonable consistency with the measurement and trend data discussed above. There are substantial exceedences of the annual mean limit value at roadsides and within the surrounding urban areas. The daily mean limit value appears less restricting, but this does not allow for episodic exceedences due to other intermittent sources.

Table 9.2 Exceedences of air quality objectives in 2010. (a) Length of road (km) exceeding different objectives in 2010. (b) Area (km^2) exceeding different objectives.

(a) Road length

Statistic	Model	Area	Road length modelled (km^2)	Base case	Base case –2 µg m^{-3}	Base case + 2 µg m^{-3}
Annual mean >20 µg m^{-3}	NETCEN	UK	16037	6096	3135	10040
	NETCEN	Greater London	1786	1730	1300	1781
	ADMS	Greater London	3651	1626	146	3651
	ERG	Greater London	4814	1742	247	4007
Annual mean >23 µg m^{-3} (Greater London)	NETCEN	Greater London	1786	914	—	—
	ERG	Greater London	4814	247	86	866
Annual mean >18 µg m^{-3} (Scotland)	NETCEN	Scotland	1348	211	—	—
Daily limit more than 7 days >50 µg m^{-3}	ADMS	Greater London	3651	0	0	0.4
	ERG	Greater London	4814	263	146	1049
Daily limit more than 10 days >50 µg m^{-3}	ADMS	Greater London	3651	0	0	0
	ERG	Greater London	4814	2.5	1.7	3.9

(b) Area

Statistic	Model	Area	Area modelled (km^2)	Base case	Base case –2 µg m^{-3}	Base case + 2 µg m^{-3}
Annual mean >20 µg m^{-3}	NETCEN	UK	242349	2648	634	12278
	NETCEN	Greater London	1624	778	113	1528
	ADMS	Greater London	1574	1175	21	1574
	ERG	Greater London	1858	39	12	374
Annual mean >23 µg m^{-3} (Greater London)	NETCEN	Greater London	1624	40	—	—
	ERG	Greater London	1858	7	3	21

Statistic	Model	Area	Area modelled (km^2)	Base case	Base case –2 µg m^{-3}	Base case + 2 µg m^{-3}
Annual mean >18 µg m^{-3} (Scotland)	NETCEN	Scotland	77535	42	—	—
Daily limit more than 7 days >50 µg m^{-3}	ADMS ERG	Greater London Greater London	1574	0.0011	0.0003	0.003
Daily limit more than 10 days >50 µg m^{-3}	ADMS ERG	Greater London Greater London	1574	0	0	0

Note: Base case ±2 µg m^{-3} indicates the likely range of predicted exceedence.

949. The small number of predicted exceedences in Scotland increases if the more stringent annual mean objective (18 µg m^{-3}) is assumed. Relaxation of the annual mean objective in London from 20 µg m^{-3} to 23 µg m^{-3} substantially reduces, but does not eliminate, exceedences. The extent of predicted exceedences in London varies considerably when a ±2 µg m^{-3} perturbation is applied, and ranges from very little, or almost all, of London being in exceedence (a more detailed explanation is provided in Annex 8). In practice there is likely to be considerable spatial variability in the residual component, maybe enhancing the importance of roads. Until better information is available it is not possible to resolve this, but it is an important limitation in assessing the precise areas of exceedence in future years.

9.6 Will the UK be able to meet the range of targets for PM$_{2.5}$ as proposed in the draft CAFE Position Paper on Particulate Matter? If not, why not? What levels of PM$_{2.5}$ are likely to be achieved by current measures and policies?

9.6.1 Answer

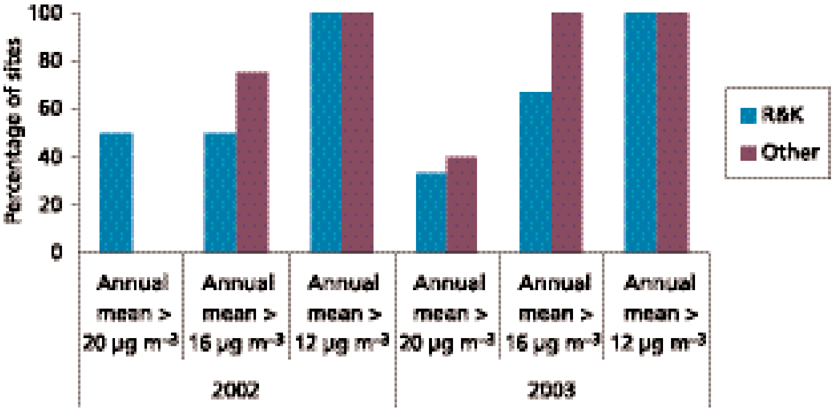
950. The CAFE Position Paper does not set firm targets for PM$_{2.5}$, but suggests that the attainment date would not be before 2010 and recommends a limit value based on annual means within the range 12–20 µg m^{-3}. Assuming a business as usual scenario, it is predicted that exceedences would be restricted to a small number of busy roads, predominantly within London at the upper end of this range (20 µg m^{-3}). At the lower end (12 µg m^{-3}) exceedences would be widespread across the UK, including both roadside and urban background locations.

951. There are a number of different sources contributing to the predicted exceedences in 2010, and it is not possible to single out any specific factor. A more detailed analysis of source apportionment is provided in Section 9.7.

9.6.2 Rationale

952. Figure 9.4 shows a summary of measured $PM_{2.5}$ concentrations in 2002 and 2003 compared with annual mean targets of 12, 16 and 20 µg m^{-3} (a more detailed description of the results is provided in Annex 7, Tables A7.4 and A7.5). There are currently few exceedences of 20 µg m^{-3}, but a considerably greater number if the limit value was reduced to 16 µg m^{-3}. If a 12 µg m^{-3} limit value was assumed, there would be non-compliance at all sites.

Figure 9.4 Measured $PM_{2.5}$ concentrations (gravimetric measurements only) in 2002 and 2003 compared with the CAFE proposed annual and daily mean limit values: percentages of sites exceeding the proposed limit value in each year are shown. (R&K, Roadside and kerbside sites; Other, all other site classifications.)



953. Predicted concentrations of $PM_{2.5}$ in 2010 have also been based on modelling studies, albeit with additional uncertainties compared with PM_{10}. Table 9.3 provides modelling estimates for exceedence of different annual mean limit values for the roads and regional areas considered in the Netcen model, for both 2002 and 2010. Corresponding estimates from the ADMS model for London in 2004 and 2010 are also included for comparison; these appear generally more optimistic for 2010 despite lower assumed reductions in the secondary inorganic component.

954. For 2010, if an annual mean limit value of 20 µg m^{-3} was assumed, then exceedences would be limited to a small number of road links, predominantly within London. As the assumed limit value is decreased, then exceedences occur at roads outside of London and then within wider urban areas. If a limit value of 12 µg m^{-3} was imposed, it is predicted that exceedences would occur along over 70% of roads in the UK and at about 25% of the urban areas.

9.7 What are the practical maximum feasible reductions of PM_{10} and $PM_{2.5}$ concentrations at (a) hotspots and (b) urban background, for example central London locations?

9.7.1 Answer

955. Unless and until mass closure can be achieved between PM components and PM_{10} and $PM_{2.5}$ mass, quantitative answers are compromised by uncertainty in our basic understanding. If the major manmade contributions to PM_{10} and $PM_{2.5}$ levels

Table 9.3 Predicted exceedences of proposed annual mean $PM_{2.5}$ concentrations by (a) road length and (b) area in 2002 and 2010 (Netcen model) and 2004 and 2010 (ADMS model).

	>12 µg m^{-3}			>14 µg m^{-3}			>16 µg m^{-3}			>18 µg m^{-3}			>20 µg m^{-3}		
	2002	**2004**	**2010**	**2002**	**2004**	**2010**	**2002**	**2004**	**2010**	**2002**	**2004**	**2010**	**2002**	**2004**	**2010**
(a) Road lengths (km) Region (km length)															
London (1,786)	1786	—	1782	1781	—	1638	1547	—	358	678	—	187	245	—	19
ADMS (3,651)	—	*3651*	*815*	—	*3247*	*84*	—	*698*	*7*	—	*186*	*0*	—	*67*	*0*
Rest of England (10,911)	9695	—	6861	6273	—	1371	2223	—	420	729	—	49	236	—	2
Scotland (1,348)	264	—	27	46	—	0	3	—	0	0	—	0	0	—	0
Wales (982)	362	—	61	58	—	7	16	—	0	7	—	0	0	—	0
Northern Ireland (1.010)	318	—	65	154	—	1	38	—	0	4	—	0	0	—	0
TOTAL (16,037)	**12325**	**—**	**8795**	**8312**	**—**	**2963**	**3828**	**—**	**778**	**1418**	**—**	**236**	**481**	**—**	**21**
(b) Area (km^2)															
Region (area in km^2)															
London (1,624)	1624	—	1531	1332	—	146	111	—	9	13	—	0	0	—	0
ADMS (1,574)	—	*1574*	*410*	—	*1555*	*10*	—	*139*	*1*	—	*21*	*0*	—	*6*	*0*
Rest of England (128,765)	54866	—	12660	4167	—	489	292	—	52	27	—	1	0	—	0
Scotland (77,535)	26	—	4	2	—	0	0	—	—	0	—	0	0	—	0
Wales (20,745)	212	—	53	12	—	1	0	—	0	0	—	0	0	—	0
Northern Ireland (13,680)	52	—	1	2	—	0	0	—	0	0	—	0	0	—	0
TOTAL (242,349)	**56780**	**—**	**14249**	**5615**	**—**	**636**	**403**	**—**	**61**	**40**	**—**	**1**	**0**	**—**	**0**

could be reduced to insignificant levels by foreseeable emissions controls, this would still leave natural sources and a proportion of manmade sources that are difficult or impossible to control (such as non-exhaust road traffic emissions). We estimate that these might account for up to 10 μg m^{-3} PM_{10} and 5 μg m^{-3} $PM_{2.5}$ in urban background locations. The uncertainties are such that it is difficult to estimate corresponding levels for hotspots close to roads.

956. Although it is not possible to directly answer what reductions are feasible, it is possible to consider what level of reduction would be required. An analysis of the emissions reduction required relative to baseline projected emissions at selected locations is provided in Table 9.4 below.

Table 9.4 Reductions in concentration and emissions required to meet objectives at selected sites.

Location and objective	Reductions required
Marylebone Road 2005 objective of ≤35 days >50μg m^{-3}	31% all primary and secondary; or 40% all primary; or 62% traffic emissions.
Marylebone Road 2010 objective of 23μg m^{-3} annual mean	46% all primary and secondary; or 61% all primary; or 100% all traffic emissions.
Bury roadside 2010 objective of 20μg m^{-3} annual mean	23% all primary and secondary; or 34% all primary; or 84% traffic emissions.
Glasgow kerbside 2010 objective of 18μg m^{-3} annual mean	38% all primary and secondary; or 49% all primary; or 83% traffic emissions.

9.8 Where and what are the main source contributors to current and future concentrations of PM_{10} and $PM_{2.5}$? What are the contributions of different sources to forecast exceedences of the EU limit values and UK objectives?

9.8.1 Answer

957. Modelled PM_{10} concentrations for the current year show that the traffic contribution dominates at roadside locations, but makes a relatively small contribution at urban background and suburban sites. For all sites, there is also a stationary source contribution and a substantial contribution from the regional background, including the secondary PM component. The 'residual' component also makes a very significant contribution to the total. By 2010, the traffic component is estimated to fall to about 50–75% of its current contribution, with the secondary component then dominating at non-roadside sites. However, the residual component is assumed to remain unchanged by 2010, and represents 50% or more of the total PM_{10} concentration at non-roadside locations.

958. For $PM_{2.5}$, a broadly similar pattern emerges for both the current and future years, although the residual component is much lower than that assumed for PM_{10}.

9.8.2 Rationale

959. Figure 9.5 describes calculations from the APEG source apportionment model for two sites in London (the Marylebone Road kerbside site and the London North Kensington urban background site). Comparisons are made between 2002 and 2010, for both PM_{10} and $PM_{2.5}$. The calculations are based on measurements carried out using the Partisol gravimetric sampler in order to take account of the volatile component.

960. At the Marylebone Road site, the traffic contribution strongly dominates PM_{10} concentrations in 2002, but declines by about 50% by 2010, by which time the residual component makes the greatest contribution. A very similar pattern is observed for $PM_{2.5}$.

961. At the London North Kensington background site the traffic contribution is significantly lower, with the secondary and residual components contributing equally to PM_{10} concentrations in 2002 and 2010. The distribution for $PM_{2.5}$ concentrations is very different, with the secondary component making the largest contributions in both 2002 and 2010.

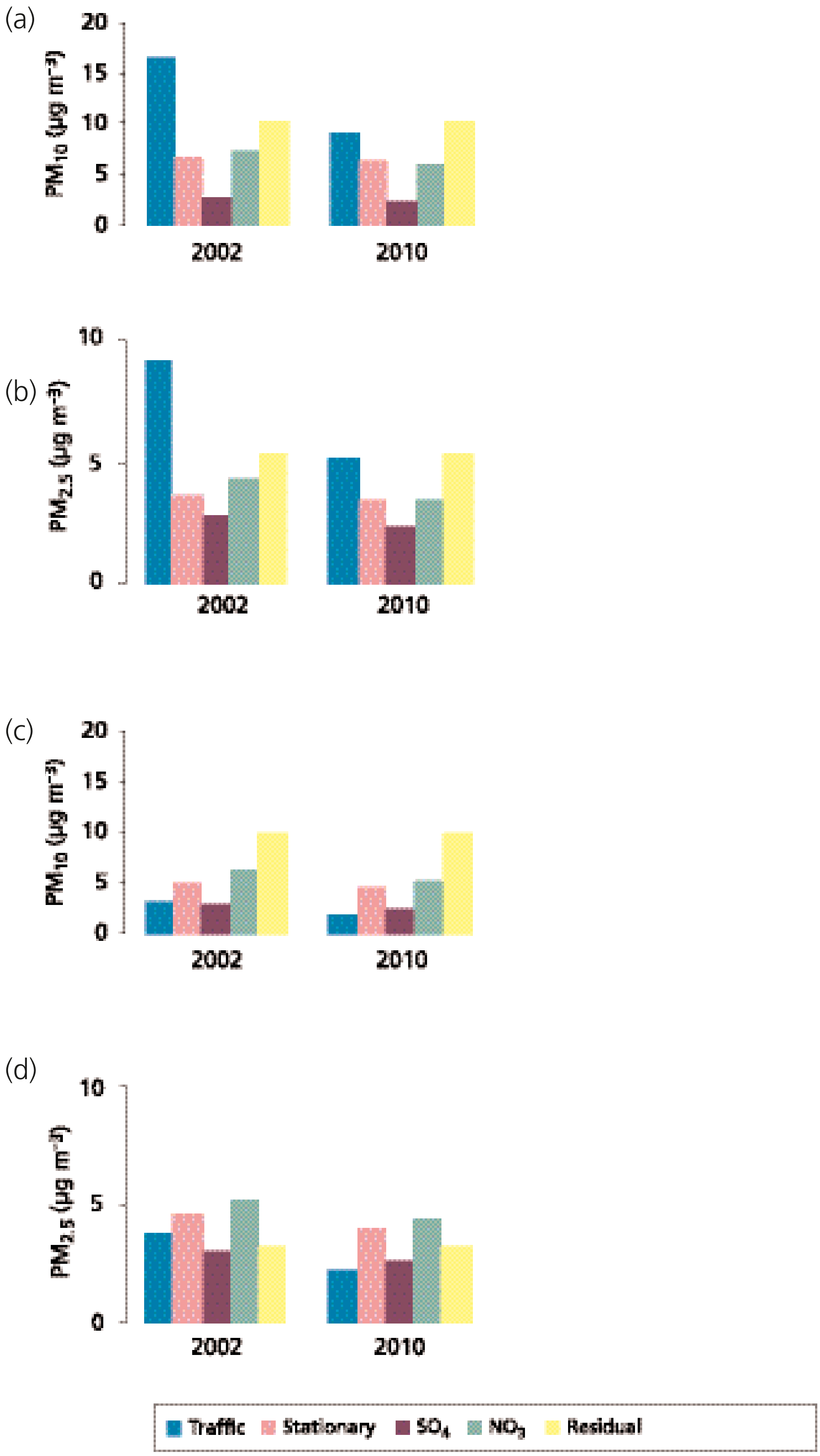
9.9 What are the potential sources of abatement and types of measures to reduce particle concentrations at (a) hotspots, such as near busy roads, (b) at urban background, central London and (c) across the whole country? What role can local/national/EU-wide measures play in meeting targets? These measures should be defined as technical (for example, vehicle standards); non-technical (for example, traffic management systems); and international (for example, controlling European/hemispheric emissions). Are there alternatives to emissions reduction?

9.9.1 Answer

962. There are a wide range of abatement measures beyond 'business as usual' that have the potential to reduce primary, secondary precursor and PM_{coarse} emissions. For the control of primary emissions, these include technical measures (such as the fitting of particulate traps on new diesel vehicles, retrofit of FGD to power stations, a switch from domestic coal to gas combustion and so on) and non-technical measures (such as the introduction of LEZs). Technical measures can be generally expected to be of benefit at the local and regional levels as well as nationally and may even be specifically targeted at hotspots.

963. A significant proportion of the precursor emissions are associated with long-range transport and are hence dependent upon action in Europe and beyond. The results of a European-wide assessment of emissions have been considered by the CAFE Working Group on Particulate Matter, which concluded that a large

Figure 9.5 Source apportionment during 2002 and 2010 based on APEG receptor model for (a) Marylebone Road PM_{10}, (b) Marylebone Road $PM_{2.5}$, (c) London North Kensington PM_{10} and (d) London North Kensington $PM_{2.5}$. All results based on Partisol measurement data.



potential exists for reducing PM precursor emissions in Europe. At a national level, various initiatives to control NO_x emissions (for example, for new diesel vehicles) may also have a significant impact on reducing precursors, but only a small impact on PM concentrations at a local level.

964. Measures for controlling PM_{coarse} emissions are specifically targeted at the local level and could include the sweeping or watering of road surfaces for example.

965. Alternatives to emission reduction include measures to reduce traffic volumes, such as incentives to use public transport or congestion charging schemes.

9.9.2 Rationale

966. The NAEI is currently investigating the emission reduction benefits of a number of transport and non-transport scenarios beyond the 'business as usual' case. Table 9.5 lists some of the measures being assessed, and additional measures considered by the CAFE Working Group on Particulate Matter. The table describes the reduction in emissions from the source being targeted (if it is a technical measure) and, where it has been quantified, the overall percentage reduction in UK PM_{10} emissions. The table also shows where the reductions in emissions will generally occur (for example, in urban areas or at specific points). It should be noted that the measures shown in the table are illustrative only.

967. The NAEI has assessed a number of scenarios relating to the setting of more stringent standards on emissions from road vehicles. The EC is considering tighter Euro V standards for cars and light duty vehicles and Euro VI standards on heavy-duty vehicles, which are likely to be introduced between 2010 and 2013. These might effectively mandate the fitting of particulate traps on all new diesel vehicles, which could lead to a significant reduction in urban exhaust emissions of PM_{10} from traffic in 2015. Other schemes being considered include incentives for using water-diesel emulsion.

968. Non-technical transport measures are also being assessed within the NAEI, for example, increasing the scrappage rate to accelerate the removal of higher emitting pre-Euro I and Euro I cars. Other measures being examined include various road charging schemes and LEZs, which could be targeted to control emissions at local hotspots.

969. The CAFE Position Paper on Particulate Matter considered several non-technical measures for reducing PM emissions from traffic. For example, non-exhaust road traffic emissions could potentially be reduced by vacuum sweeping and water spraying of road surfaces and so on. These measures could be targeted at hotspots and could prove beneficial at controlling PM emissions in and around construction and mineral extraction sites. The CAFE Position Paper also lists a number of traffic management measures, green travel plans and modal transport policies with potential at reducing local emissions, some of which are currently being picked up in local authority action plans to help improve air quality.

970. Among other modes of transport, emissions of PM, NO_x and SO_2 may benefit from the reduction of the sulphur content of gas oil used to power diesel trains and fuel oil used for shipping. Reducing the sulphur content of fuels may itself lead to reductions in PM emissions, but will also enable opportunities for

Table 9.5 Illustrative beyond ‘business as usual’ abatement measures for primary PM emissions.

Measure	Percentage reduction in PM_{10} at source	Effect on PM[a] size fractions	PM components affected	Percentage reduction in UK PM_{10} emissions	Where reductions will mainly occur	Note
Particulate traps on all new diesel vehicles – by 2010 for light duty vehicles, 2013 for heavy duty vehicles	90%	Reductions in fine PM; smaller reductions on ultrafines	Solid carbon, maybe less effective on low volatile organics	45% in 2015 for all urban road transport exhaust emissions; 4.7% in 2015 for all UK emissions	Urban and rural areas	A
Uptake of water-diesel emulsion – all buses and 10% HGVs running on water-diesel emulsion from 2006	25%	Reductions in fine PM; effect on ultrafines unknown	Not known	1.1% in 2010 for all urban road transport exhaust emissions; 0.1% in 2010 for all UK emissions	Mainly urban areas	A
Increase car scrappage rate by 10% from 2006	Not a technology-based measure	Reductions in fine PM	Solid carbon and volatile organic fractions	1.8% in 2010 for all urban road transport exhaust emissions; 0.2% in 2010 for all UK emissions	Urban and rural areas	A
LEZ – restricting access to Euro III+ heavy duty vehicles and 10-year age limit on vans and taxis by 2010	Not a technology-based measure	Reductions in fine PM	Solid carbon and volatile organic fractions	23% reduction in exhaust PM_{10} emissions if scheme applied to London in 2010	London. Scheme could be applied to other conurbations	B
Washing of roads (vacuum sweeping, water flushing)	Not quantified	Reductions in coarse PM	Inorganic material, solid carbon	Not assessed	Urban areas	C

Measure	Percentage reduction in PM_{10} at source	Effect on PM size fractions	PM components affected	Percentage reduction in UK PM_{10} emissions	Where reductions will mainly occur	Note
Local traffic management schemes	Not quantified; depends on scheme	Reductions in coarse and fine PM	Solid carbon and volatile organic fractions and inorganic material	Not assessed; depends on scheme	Urban areas	
Domestic combustion – 100% switch from coal to gas completed by 2010	99.8%	Reductions in coarse and fine PM	Metals, inorganic material, solid carbon	2.5% in 2010	Urban areas	A
Low cost filters on small industry processes – 30% uptake by 2010	99.5%	Reductions in coarse and fine PM	Depends on process, but expect effect on metals, inorganic material, solid carbon	1.1% in 2010	Industrial areas	A
Sinter plant fabric filter on iron and steel plants – 100% by 2010	80%	Reductions in coarse and fine PM	Metals, inorganic material, some carbon	1.8% in 2010	Point sources in industrial areas	A
FGD on coal-fired power stations – all remaining plants fitted by 2010	90%	Reductions in coarse and fine PM	Metals, inorganic material, some carbon	4.4% in 2010	Point sources	A, D
Spraying water at construction sites	20%	Reductions in coarse PM	Minerals	Not assessed	Construction areas	C

A: Based on assessment of illustrative measures by the NAEI. B: From Watkiss *et al*. (2003). C: Taken from CAFE Draft Second Position Paper on Particulate Matter, 2003. D: Business as usual case assumes seven power stations (54% of generating capacity) will already be fitted with FGD by 2010. Beyond business as usual assumes remaining 12 power stations fitted with FGD. These figures are based on projections from the 2001 NAEI using data from the Pollution Inventory on power station emissions in 2001.

implementation of after-treatment technologies like exhaust gas recirculation and selective catalytic reduction (AQEG, 2004).

971. In terms of stationary source emissions, important abatement measures include further retrofitting of FGD and electrostatic precipitators to UK combustion plant and improved filter technologies to smaller industrial processes. Fuel switching from coal to gas would also bring benefits within both the industrial and domestic sectors. In the latter case, measures could be focussed upon urban hotspot regions.

972. The National Emissions Ceilings Directive is currently under review with proposals expected in 2006. Potential 'add-on' measures for additional abatement of SO_2, NO_x and NH_3 have been assembled for each Member State, including fuel switching and energy conservation schemes. An additional contribution to precursor emissions comes from shipping, which now contributes about 10% of the secondary sulphate and nitrate background concentrations across the UK; additional control measures, such as the reduction in the sulphur content of fuels, are also being considered.

973. Controlling precursor emissions leads to fairly uniform reductions of secondary particles over large areas, as opposed to targeted reductions in urban areas. Correspondingly, local action to reduce precursor emissions has little effect on local PM concentrations.

974. A summary of illustrative 'beyond business as usual' measures for the UK reduction of PM precursor emissions is provided in Table 9.4. The table describes the impact upon NO_x or SO_2 reduction at source and the overall impact upon UK emissions. At a European level, the CAFE Position Paper discusses a number of abatement measures, including studies by the UK Interdepartmental Group on Costs and Benefits (IGCB), which investigated the effects of possible measures to further reduce PM_{10} levels in the UK. Although based on an older set of UK emission projections than the current NAEI projections, the IGCB study concluded that overall, transport measures appeared less cost-effective than many potential industrial measures in terms of reducing background PM_{10} concentrations. However, targeted transport measures were more effective at reducing roadside concentrations than industry measures. These conclusions may need to be reassessed on the basis of more recent baseline emission projections for road transport and industry and domestic sources in the UK.

975. The CAFE Position Paper also covers the results of a European-wide assessment of emissions carried out by IIASA and summarises the emission reduction potentials of a number of PM emission abatement options included in the study. In its initial study, IIASA concluded that a large potential exists for reducing PM precursor emissions in Europe, but that results for the reduction potential of primary PM sources and measures are somewhat conflicting. A further assessment of abatement measures to reduce PM is currently underway by IIASA.

976. Alternative measures to emissions reduction include various schemes to reduce traffic volumes, which would be particularly valuable in urban areas. Such measures include incentives to use public transport, park-and-ride schemes, LEZs and designation of car-free areas. Financial measures to restrict private car use include higher taxation, increased parking fees and congestion charging. The latter is of particular interest following its introduction in London during 2003, and a summary of the effects is provided in Annex 9.

Table 9.6 Illustrative beyond business as usual abatement for PM precursor emissions.

Measure	Pollutant	Percentage reduction at source	Percentage reduction in UK emissions	Note
50% reduction in NO_x limits for new diesel vehicles from 2010 (light duty vehicles) and 2013 (heavy duty vehicles) plus tougher durability requirements on petrol vehicles	NO_x	50%	26% in UK road transport emissions in 2015; 7% in all UK emissions in 2015	A
Uptake of hybrid cars in vehicle fleet – reaching 10% of car sales by 2010	NO_x	~70%	0.8% in UK road transport emissions in 2010; 0.2% in all UK emissions in 2010	A
Uptake of water-diesel emulsion – all buses and 10% HGVs running on water-diesel emulsion from 2006	NO_x	14%	1.5% in UK road transport emissions in 2010; 0.5% in all UK emissions in 2010	A
Increase car scrappage rate by 10% from 2006	NO_x	Not a technology-based measure	2% in UK road transport emissions in 2010; 0.6% in all UK emissions in 2010	A
Selective catalytic reduction fitted to iron and steel plants (100%), power stations (100%) and refineries (80%) by 2010	NO_x	60–85%	7% in 2010	A
Domestic combustion – 100% switch from coal to gas completed by 2010	SO_2	100%	0.9% in 2010	A
Selective non-catalytic reduction fitted to cement kilns, Selective catalytic reduction on iron and steel plants – 100% by 2010	NO_x	60%	3.1% in 2010	A
FGD on coal-fired power stations – all remaining plants fitted by 2010	SO_2	90%	32% in 2010	A, B

A: Based on assessment of illustrative measures by the NAEI. B: 'Business as usual' case assumes seven power stations (54% of generating capacity) will already be fitted with FGD by 2010. Beyond business as usual assumes remaining 12 power stations fitted with FGD. These figures are based on projections from the 2001 NAEI using data from the Pollution Inventory on power station emissions in 2001.

9.10 A number of recent studies (that is, the WHO report) have highlighted the health effects of certain components of PM (fines, ultrafines, particle number, metals and elemental/organic carbon). Where further abatement techniques are known, how might they specifically affect reduction of the different PM metrics (for example PM_{10}, $PM_{2.5}$, PM_1 and particle numbers) and chemical components?

9.10.1 Answer

977. Current limit values have focused attention on reducing particle mass and specifically PM_{10}. The relative effectiveness of measures in reducing finer fractions or specific chemical components depend both on the source type and particle characteristic and the type of abatement technique employed. Measures aimed at reducing coarser particles, such as mineral dust from roads, quarrying or construction sites, will have little effect on reducing $PM_{2.5}$ or finer fractions, whereas measures to control primary particle emissions from combustion sources should be very effective in reducing fine particles. Measures that reduce activity levels, such as traffic control measures, reduce all components of the corresponding particles equally. Secondary inorganic particles are mainly in the $PM_{2.5}$ range except for a proportion of the nitrate attached to coarser particles, and measures to reduce precursor emissions are correspondingly effective across all size ranges.

978. There are currently insufficient measurement data to determine the precise impacts of PM_{10} control measures upon ultrafines and chemical components.

9.10.2 Rationale

979. Table 9.5 describes in qualitative terms the likely impact of the different beyond business as usual measures upon the different PM size fractions and chemical components, where these are known. Principal measures targeted at reducing PM emissions from traffic, such as particulate traps and the uptake of water-diesel emulsion, should be effective in reducing fine fraction ($PM_{2.5}$ and PM_1) and ultrafine emissions but this is by no means certain and cannot be currently quantified. In terms of industrial emissions, measurement data on smaller PM size fractions are even less plentiful, but improvements in technology, such as the use of bag filters, are expected to be beneficial in terms of controlling ultrafines.

980. Chapter 6 describes in detail the mechanism of ultrafine particle formation at Marylebone Road. Measures that reduce fine-fraction particle emissions may also reduce the effectiveness of the condensation sink, favouring instead the growth of fresh ultrafines, although there is no evidence to support this theory at present.

9.11 Ultrafines – what have we learned from the measured data, including source apportionment? Are the observed trends real? What fraction of ultrafine particles volatilise?

9.11.1 Answer

981. The ultrafine fraction contributes little to overall mass but dominates the particle number count. From measured data, it is clear that traffic is a major source of ultrafine particles, but there is also widespread formation by photochemical processes in the atmosphere. Time series of measurements in the UK are insufficient at present to comment on long-term trends. Current evidence suggests that volatilisation is not a significant process.

9.11.2 Rationale

982. Measurement programmes for both size-fractionated and total particle number concentrations have shown that road traffic is a major source of ultrafine particles. A substantial proportion of these particles is formed during dilution of the exhaust gas with ambient air, rather than being emitted directly from the exhaust. The formation process is significantly dependent on meteorology, with lower temperatures favouring the formation of high concentrations of particles at the lower end of the size range. Particle number concentrations are significantly higher at Marylebone Road (a kerbside site) than at the urban background sites, which show a reasonable consistency between one city and another.

983. There is clear evidence at the Harwell (rural) site for the formation of new particles, through the process of homogenous nucleation. Such processes are widespread in the atmosphere and are favoured by the low pre-existing particle concentrations at rural sites. While the formation processes are photochemically driven, the precise chemistry of particle formation is currently not well understood.

984. The subsequent fate of ultrafine particles will depend upon processes of dilution, coagulation and deposition, which are not yet fully understood.

9.12 Source apportionment how does the UK source apportionment for PM_{10}, $PM_{2.5}$ and other metrics compare with other modelling in Europe? Is road traffic more important than current models show? How is the coarser fraction between 2.5 and 10 μm accounted for?

9.12.1 Answer

985. The models used within the UK have both common themes and major differences in the manner in which they operate. In general terms, the models account for the PM_{coarse} fraction by the addition of a constant background. There is good reason to believe that part of this component is associated with non-exhaust traffic emissions (resuspension of road dust). Such an assumption in modelling leads to a different distribution of PM_{10} concentrations at roadside sites, with significant implications for where the limit values may be exceeded.

986. In terms of annual mean concentrations for the current year, the models show a broadly similar performance at monitoring sites. In terms of source apportionment, the ADMS model shows a greater traffic contribution at roadside sites and a greater secondary and a lower stationary source contribution at all sites for both PM_{10} and $PM_{2.5}$.

987. An overview of models used within Europe for the prediction of PM concentrations was provided by the CAFE Working Group on Particulate Matter, and no further analysis has been carried out within this report.

9.12.2 Rationale

988. The models that are routinely used in the UK for predictions at the national level are based either upon semi-empirical methods (relying upon monitored data and local dispersion calculations) or based primarily on dispersion modelling. An intercomparison of these models shows a broadly similar performance for the prediction of annual mean PM_{10} concentrations at monitoring sites, but an intercomparison of calculated concentrations on individual road segments in London shows a wide scatter between the ADMS and ERG models. This reflects the different manner in which the models account for individual source components and, in particular, the different treatment of the background. No such intercomparison has been carried out for $PM_{2.5}$.

989. Treatment of the PM_{coarse} fraction is handled in a slightly different manner by each of the UK models, including the derivation of 'roadside specific' adjustment factors or the derivation of a PM_{coarse} component based on regression analysis with NO_x. Although these approaches have a sound base, it is currently not clear how successful they are in taking full account of non-exhaust road traffic emissions. As this is believed to be a significant component of PM_{coarse} emissions, the current models may underestimate the importance of road traffic to predicted PM_{10} concentrations.

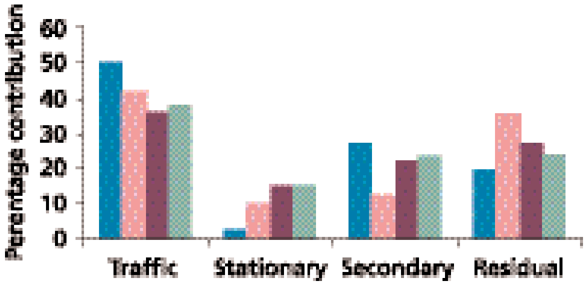
990. Site-specific source apportionment studies have been carried out using the Netcen and ADMS models for London. The detailed results are provided in Chapter 8, with an intercomparison of the model performance summarised in Figure 9.6 for Marylebone Road. Several important conclusions may be drawn.

- The Netcen model shows significantly different results, depending upon which measurement method was used to collect the data. This may be directly attributed to the loss of the volatile species by the TEOM analyser.

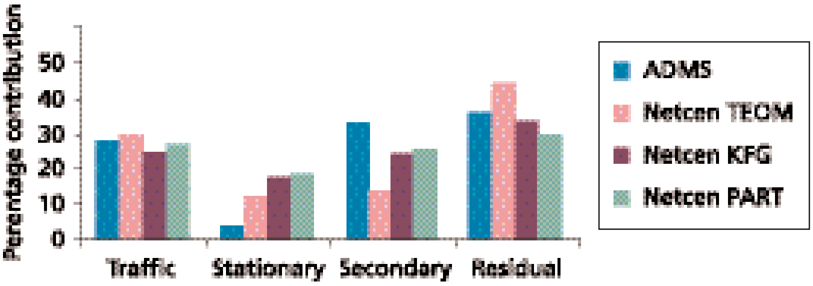
- The ADMS model shows a higher traffic contribution, much greater secondary and much lower stationary source contribution.

Figure 9.6 Comparison of source apportionment at Marylebone Road site based on Netcen (NAEI) and ADMS (LAEI) models for (a) 2002 and (b) 2010. (Netcen model results shown for three measurement methods: TEOM, KFG and Partisol.)

(a) 2002



(b) 2010



9.13 Can we explain the trends in measured PM_{10}, sulphur and black smoke since 1992?

9.13.1 Answer

991. The downward trends in measured PM_{10}, sulphur and black smoke since 1992 are complicated by interannual variations. Across the UK, PM_{10} concentrations declined steadily throughout the 1990s, broadly in line with declining PM_{10} and precursor emissions. This downward trend appears to have flattened off, or even become an increase, in more recent years. The influence upon these trends of the significant regional pollution episodes in 2003, due to meterological conditions, is not yet clear.

992. Particulate sulphate concentrations are also declining, albeit at a lower rate than SO_2 emissions. This may point to increasing oxidising capacity of the atmosphere as NO_x emissions decline.

993. Black smoke concentrations have declined in line with emissions across most of the UK with the exception of London where diesel traffic emissions make a significant contribution.

9.13.2 Rationale

994. There are 12 long-running monitoring sites across the UK that allow an analysis of PM_{10} trends from 1992 onwards. For all of these sites, there are statistically significant downward trends in PM_{10} concentrations up to 2003. Average PM_{10} concentrations are declining at about –4.4% per annum, which is close to the –5% per annum decrease in PM_{10} emissions described within the emissions inventory.

995. This trend in PM_{10} concentrations has not remained constant, and there is evidence to suggest that it may have reversed over the period 2000 to 2003. Although PM_{10} emissions have continued to decline at about –5% per annum over the period 1997 to 2003, this has not been the case for urban PM_{10} concentrations, which have slowed to about –2% per annum, with some sites in London exhibiting trends much smaller than this or close to zero.

996. At some urban sites there is evidence of increasing PM_{10} concentrations over the period 2000 to 2003, although this may have been strongly influenced by large-scale regional pollution episodes during 2003. However, although 2003 stood out as a significantly different year for about one-third of the monitoring sites, this difference was not apparent at other monitoring locations and there is no clear spatial distribution.

997. The UK network sites show a significant downward trend in particulate sulphate concentrations over the period 1990 to 2002, in the range of –4.3 to –9.3% per annum. This is somewhat smaller than the decline in UK SO_2 emissions over the same period, suggesting that an increasing fraction of the emitted SO_2 is being oxidised. A potential mechanism for this may be declining NO_x emissions and hence an increased photochemical oxidation rate for SO_2 to sulphate aerosol.

998. There is a large network of black smoke monitoring sites across the UK. Over the period 1990 to 2002, average concentrations declined at a rate of about –5% per annum. Over the same period, black smoke emissions are estimated to have declined at –6.8% per annum. A more detailed analysis of the trend data on a region-by-region basis demonstrates that significant downward trends have been observed in locations where domestic coal burning has been phased out. Within London, some of the smallest downward trends in the UK have been identified. This is partly due to the much earlier phase-out of domestic coal burning and the increasing contribution from road traffic, in particular diesel vehicles.

9.14 What are the differences between strategies that address hotspots of exceedence and those that aim to reduce population exposure? Should policy evaluation consider impacts on population exposure as well as concentrations at specific locations?

9.14.1 Answer

999. Strategies based solely on limit values tend to focus attention on reducing concentrations in localised areas or 'hotspots' where exceedences are measured or predicted. However, such localised areas are unlikely to be associated with large population exposure. For pollutants such as PM, where there is no evidence of a threshold concentration below which health effects do not occur, a strategy based on reducing exposure to the largest population would seem to offer improved benefits to public health. AQEG recommends such an exposure reduction strategy.

9.14.2 Rationale

1000. Air quality control is currently based upon the use of limit values that are defined in the Air Quality Daughter Directives. Limit values have proved to be an extremely useful mechanism over the past 20 years or so; they provide a simple, uniform measure of progress in improving air quality as well as providing a driver for emissions controls and helping raise public awareness. There are, however, emerging limitations with this approach.

1001. The limit values for human health apply to all locations within a Member State, and this inevitably leads policy makers towards the improvement of conditions in areas of very poor air quality, that is hotspots, where the limit values are exceeded. For PM, these hotspots will include major roads, particularly within urban centres in the majority of European cities. If there are no exceedences of the limit values, then the Member State will not be required to develop any policy measures to further reduce exposure to PM, even though health benefits would be achieved.

1002. However, it is widely recognised that in the case of PM, there is no threshold concentration below which there are no health effects. It may be concluded that any reduction in exposure would be associated with an improvement to health, even if levels are below the limit value. In terms of maximising health benefits to the general population, such an approach is expected to be far more effective. For example, reducing the exposure of 10 million people by 0.1 $\mu g/m^3$ is ten-times more effective than reducing the exposure of 1,000 people (at hotspots) by 10 $\mu g/m^3$.

1003. A system of reducing long-term population-weighted exposure would provide a useful supplement to limit values. This could be based on reducing concentrations measured principally in urban background locations, where people are most exposed.

Chapter 10

Conclusions and recommendations

1004. In summarising current understanding and answering the questions posed by Defra, it has become apparent that further work should be instigated to assist in any future policy analyses on suspended particulate matter. These activities are outlined in this chapter in the form of some concluding statements, followed by recommendations for further monitoring and research activities. Whilst there is no question that basic scientific research in this area would be helpful, this would not necessarily improve understanding of the underpinning processes on a timescale that would be useful to Defra's immediate policy requirements. The activities described below, therefore, largely focus on monitoring because in the short term this offers significant opportunities to improve understanding in national terms without unduly focusing on specific pollution episodes, field campaigns or localized issues. In formulating this chapter, AQEG has adopted a pollutant-by-pollutant or component-by-component approach rather than a process-based approach dealing with sources, emissions, dispersion, transformations and removal processes.

1005. The Air Quality Expert Group took the view that unless mass closure could be achieved between the particulate matter (PM) components and $PM_{2.5}$ and PM_{10} mass, it would be difficult to assess the causes of exceedence and the appropriate remedial measures. Hence, each of the main PM components is discussed in turn, before PM_{10} and $PM_{2.5}$, and before turning to research and monitoring requirements.

10.1 Particulate sulphate

1006. Particulate sulphate makes a substantial contribution to the regional background PM levels upon which urban, roadside and kerbside increments are built. There is no evidence for any urban or traffic enhancement of particulate sulphate levels. Rural particulate sulphate concentrations have declined steadily since measurements began in 1977, although between 2000 and 2003 levels have risen slightly. Particulate sulphate levels have not declined as rapidly as rural SO_2 levels or SO_2 emissions. There has, therefore, been a steady increase in the fraction of airborne sulphur species present as particulate, reflecting an increase in the oxidising capacity of the atmosphere with time.

1007. There is a need to understand the relationship between particulate sulphate trends and SO_2 emissions and to build confidence in the regional model assessments of future particulate sulphate concentrations under the influence of the UNECE Gothenburg Protocol and the EU National Emissions Ceilings Directive.

1008. **It is recommended that continuously recording instruments for particulate sulphate are deployed because we currently have no understanding of the behaviour of particulate sulphate concentrations in PM pollution episodes and their coupling with ammonia and particulate nitrate levels.** Without this understanding, it will be difficult to develop robust strategies for the reduction of the regional background PM levels upon which urban, roadside and kerbside levels are superimposed.

10.2 Particulate nitrate

1009. Particulate nitrate makes a substantial contribution to the regional background PM levels upon which urban, roadside and kerbside levels are superimposed. Because of the marked downwards trends in rural sulphate levels, particulate nitrate now makes up the largest component of regional background levels. There is some evidence for urban enhancement of particulate nitrate levels. Particulate nitrate has never been monitored satisfactorily in terms of spatial coverage, time resolution or by methods that are artefact-free. It is present in the atmosphere as fine mode ammonium nitrate and coarse mode sodium and calcium nitrates. There is inadequate information on the split between these forms on an hourly, daily and seasonal basis. Furthermore, we have no quantitative information on how the different forms of particulate nitrate are detected by the different PM_{10} and $PM_{2.5}$ monitors deployed in the UK. Currently, the monitoring base for particulate nitrate is so unsatisfactory that robust policy formulation for particulate nitrate is difficult.

1010. Continuous monitoring of particulate nitrate is currently carried out at two locations in the UK, but this is wholly unsatisfactory in terms of spatial coverage. In addition, continuous measurements of nitrate in the coarse particle fraction are not undertaken. **It is recommended that additional continuously recording nitrate monitors are deployed to understand the phenomenology of nitrate episodes, to understand their correlation with sulphate and PM episodes, to identify the extent of any urban enhancement in nitrate levels and to validate the models used to estimate future trends in regional particulate nitrate levels though to 2010 in response to the UNECE Gothenburg Protocol and the EU National Emissions Ceilings Directive. Measurements should be carried out in both the PM_{coarse} and $PM_{2.5}$ fractions.**

10.3 Elemental or black carbon

1011. Elemental or black carbon emissions arise mainly from diesel vehicles in urban areas and represent the component of primary PM emissions in which we have the greatest confidence. Elemental carbon is one of the few PM components for which there is a discernible urban increment. The indications are that urban elemental carbon levels have been declining steadily as coal burning has been phased out. Monitoring of elemental carbon on a continuous basis is currently carried out at four sites in the UK, but this is inadequate for policy assessment and development purposes. **It is recommended that the current monitoring regime be expanded and that additional continuous monitors are deployed at urban background, kerbside and roadside sites.**

1012. Elemental carbon emissions are projected to decline further as diesel exhaust emission controls are introduced. **It is recommended that continuous monitoring of elemental carbon is carried out at urban, roadside and kerbside locations to check that the new diesel vehicle emission standards are having the desired effect in the UK. It is recommended that traffic increment studies are repeated with continuously recording elemental carbon monitors to enable a robust test of the emission inventories to be completed.**

10.4 Organic carbon

1013. Monitoring of organic carbon on a continuous basis is currently carried out at four UK sites, but this is inadequate for policy assessment and development purposes and scientific understanding of this important PM component is rudimentary. Particulate organic carbon is important for policy because it is one of the few PM components for which there is evidence of a demonstrable urban enhancement above regional background levels. Particulate organic carbon is present in the urban environment either as primary or secondary material. The main source of primary emitted particulate organic carbon in urban areas is diesel traffic and includes semi-volatile organic compounds derived from diesel fuels and lubricants that attach themselves to pre-existing particles. At rural locations, secondary particulate organic carbon dominates primary particulate organic carbon, particularly during summertime. Photochemical reactions oxidise natural biogenic and manmade organic compounds to low volatility species that can condense on pre-existing particles. Secondary particulate organic carbon makes a substantial contribution to regional PM levels during summertime.

1014. **It is recommended that continuously recording organic carbon monitors are deployed at urban background locations without delay.** Furthermore, these instruments should be employed in studies of urban, roadside and kerbside increments to validate emissions inventory estimates of the emissions of semi-volatile species from diesel fuels and lubricants.

1015. **It is recommended that the contribution from secondary organic carbon to regional PM background levels be determined.** It will be important to separate the contributions from the oxidation of terpenes of natural biogenic origins from that from the oxidation of aromatic compounds of man-made origins. Currently, there is no understanding of how concentrations of secondary organic carbon particulates vary across the UK and during summertime regional pollution episodes. Furthermore, there is no understanding of how these levels might change in the future and whether they will be influenced by the implementation of the UNECE Gothenburg Protocol and the EU National Emission Ceilings Directive.

10.5 PM_{10}

1016. Urban background PM_{10} concentrations have shown a steady decline during the 1990s, but during 2000 to 2003 these downwards trends have slowed up and some sites have shown an increase. The current PM_{10} monitoring network is an important European asset and should be maintained in its current state to follow trends through to 2010. **An increase in rural PM_{10} monitoring is recommended for determining urban increments, for the quantification of regional PM_{10} background levels and for the characterisation of PM_{10} episodes, when co-located with ozone monitors.** Further rural PM_{10} monitoring would also improve the performance of urban dispersion modelling and local authority air quality assessments.

1017. A detailed evaluation exercise to compare the performance of a variety of PM_{10} samplers and analysers commenced in 2004. **Upon completion of this exercise it is recommended that the current default USEPA adjustment factor applied to TEOM PM_{10} data be reviewed and, if appropriate, all data**

within the UK Air Quality Information Archive be rescaled. Of additional concern are the potential losses of semi-volatile nitrate and organic species from measurements in the existing UK network. **It is also recommended that either a continuous method of monitoring PM_{10} that is not subject to losses of semi-volatile species and positive artefacts be identified or that routine measurements be carried out for the 'core particulate' mass, with additional monitoring to quantify the semi-volatile components.**

10.6 $PM_{2.5}$

1018. There is currently an inadequate level of co-located $PM_{2.5}$ and PM_{10} monitoring in urban background and rural sites for policy assessment purposes. **Further rural $PM_{2.5}$ monitoring is recommended to assist in the characterisation of regional $PM_{2.5}$ background levels, the quantification of urban increments and the characterisation of $PM_{2.5}$ episodes.**

1019. A detailed evaluation exercise to compare the performance of a variety of $PM_{2.5}$ samplers and analysers commenced in 2004. **Upon completion of this exercise it is recommended that the current default USEPA adjustment factor applied to TEOM $PM_{2.5}$ data be reviewed and, if appropriate, all data within the UK Air Quality Information Archive be rescaled.** Of additional concern, as with PM_{10}, are the potential losses of semi-volatile nitrate and organic species from measurements in the existing UK network. **It is also recommended that either a continuous method of monitoring $PM_{2.5}$ that is not subject to losses of semi-volatile species and positive artefacts be identified or that routine measurements be carried out for the 'core particulate' mass, with additional monitoring to quantify the semi-volatile components.**

10.7 Ultrafine particles

1020. Although laboratory toxicological studies show ultrafine particles to be more toxic per unit of mass than coarser particles, the proportion of PM_{10} mass comprised by ultrafine particles is very small and the contribution of ultrafine particles to adverse health outcomes is currently not well quantified. Atmospheric measurements of ultrafine particles show clear evidence for two types of sources: formation from road traffic emissions and new particle formation in rural areas through homogeneous nucleation. The magnitude of both sources is critically dependent on pre-existing atmospheric particle loadings and upon meteorological factors. Given the highly variable nature of these formation processes and the uncertainties regarding both the contribution of ultrafine particles and the relative influences of particle size and composition in eliciting adverse health effects, it would not be appropriate at the current time to envisage formulating an air quality standard in relation to the mass or number concentration of ultrafine particles alone.

10.8 Particulate iron

1021. Particulate iron is a valuable tracer for non-exhaust particulate emissions. **It is recommended that daily monitoring of particulate iron is begun with a view to establishing whether the Site-specific Receptor and Forecast Model employed by Defra for its policy assessment could be extended to**

include a daily assessment of non-exhaust traffic emissions using particulate iron as a tracer. Furthermore, daily kerbside and roadside monitoring of particulate iron may dramatically improve our understanding of the impact of non-exhaust traffic emissions on air quality in the vicinity of roads and highways away from industrial sources.

10.9 Natural PM sources

1022. A number of natural PM sources have been identified in this report that make a substantial contribution to hourly and daily PM levels. These include sea salts, wind-driven resuspension of crustal material, Saharan dusts, forest fires and secondary particulate organic carbon from terpene-oxidation. These natural sources have produced elevations to both PM_{10} and $PM_{2.5}$ levels and their presence has been inferred by meteorological and air quality data analysis. No attempt has been made to assess their overall significance on a regional or national basis. Generally speaking, those episodes have been analysed that gave the highest concentrations and smaller episodes have been lost in the PM background.

1023. A more considered analysis of natural PM sources could be made by linking together the Defra PM Air Quality Forecast Model to the rural and suburban PM network observations. Natural PM episodes would appear as periods and regions of systematic model underestimation. Coupling together a forecast model with a real-time network with some form of data assimilation scheme would allow a rigorous determination to be made of the magnitudes of natural PM sources on a regional and national basis.

10.10 Other manmade sources

1024. A number of manmade sources of PM components have been identified in this report for which well-tried emission inventory approaches may not be appropriate because of the lack of statistical data. These include bonfire nights, construction and demolition, open burning of garden waste, agricultural waste burning, wood burning stoves and fires and fugitive emissions from industrial plant, refuse handling and quarrying, some of which are episodic.

1025. Again, a more considered analysis of these man-made sources could be made by linking together the Defra Urban PM Air Quality Forecast Model with urban PM monitoring data. The presence of these other manmade sources would be indicated by systematic model underestimation at particular times and locations. Coupling together a forecast model with a real-time network with some form of data assimilation scheme would allow a rigorous determination to be made of the magnitudes of these other manmade sources of urban PM.

10.11 Ozone and PM air quality

1026. Although it is generally understood that in the summertime there is a strong link between ozone and PM air quality, rural PM monitoring has been inadequate to quantify the link with any accuracy. Furthermore, it is not straightforward to quantify how summertime PM concentrations might respond in the future to the implementation of the UNECE Gothenburg Protocol and the EU National Emissions Ceilings Directive and whether they might decline in line with summertime ozone levels. Although modelling tools are available to address these questions, the rural PM monitoring base has been insufficient to validate model performance for particulate sulphate and nitrate. **It is recommended that continuous monitoring of particulate sulphate is begun at a rural ozone monitoring site (alongside existing particulate nitrate, PM_{10} and $PM_{2.5}$ measurements) with a view to validating photochemical model performance during summertime PM and ozone episodes.**

1027. Measurements of continuous elemental/organic carbon at this rural site should be used to quantify the contribution from secondary organic aerosol to regional background PM levels. Currently, we have little understanding of whether there are episodes in particulate organic carbon and, if they do occur, whether these episodes correlate with ozone and PM episodes. Air mass attribution may help to determine whether the likely precursors to secondary organic aerosols are manmade or natural biogenic sources.

10.12 Projected exceedences of future air quality targets based on the Site-specific Receptor and Projections Model

1028. The Site-specific Receptor and Projections Model has been able to account for the observed trends in PM_{10} levels at 16 urban background and kerbside sites during the period up to 2002 and has been used to assist Defra in the formulation of its strategy on future urban air quality. Its projections to 2010 point to the conclusion that it is unlikely that target air quality in terms of annual mean PM_{10} levels will be achieved on current policies at the selected urban background and kerbside locations. These projections take into account proposed controls to diesel exhaust emissions for cars and heavy goods vehicles and agreed reductions in secondary inorganic particulates through the implementation of the UNECE Gothenburg Protocol and the EU National Emissions Ceilings Directive. For air quality objectives and indicative limit values to be reached in 2010, then PM_{10} levels would need to decline at rates significantly higher than their historic rates at urban background sites and this does not appear plausible.

1029. **In view of the importance of this conclusion concerning the future exceedence of air quality objectives and indicative limit values, it is recommended that the Site-specific Receptor and Projections Model be extended and improved and these conclusions revisited.** Improvements should be made to the treatment of coarse particles and an approach adopted to the inclusion of a manmade coarse particles due to non-exhaust traffic emissions using daily measurements of particulate iron. Specific terms in the model should be included based on continuous measurements of particulate sulphate and particulate nitrate, elemental carbon and organic carbon.

10.13 Projected exceedences of future air quality targets based on dispersion modelling

1030. Dispersion models have been utilised to describe the fine scale distribution of PM concentrations across major conurbations and the increment in PM due to traffic. These models show that PM air quality on a large number of major road links will continue to exceed objectives and indicative limit values beyond 2010 on current policies.

1031. In view of the importance of these conclusions for policy, a number of improvements in these models are appropriate so that their robustness can be checked. Urban background monitoring of elemental and organic carbon would provide a more convincing test of model performance. Furthermore, urban monitoring of indicator species for coarse particulates could also provide a further test of model performance.

10.14 Local air quality management

1032. The UK has developed a sophisticated system of local air quality management. This is well suited to identifying local hotspots that are not necessarily identified through national studies. Air quality management areas have been established by 59 local authorities. These are areas where exceedences of the 2004 objectives for PM_{10} are likely. The majority of these are for traffic sources, but there are also air quality management areas for industrial, commercial and domestic sources of PM_{10}. The action plans being developed by the local authorities will help ensure that concentrations are driven down in these areas, but it is difficult to predict by how much conditions will be improved. As these measures will be focussed on local hotspots they will probably only make a marginal contribution to the wider reduction in PM emissions. The exception to this might be in London and other major conurbations, where many local authorities are working together to develop wider scale plans, for example, the low emission zone initiative being developed in London.

10.15 Future PM policy development

1033. In principle, future PM policy development can involve either/both the management of peak urban PM concentrations at hotspots or management of the regional PM background levels upon which urban, roadside and kerbside increments are built. Both policies have major uncertainties.

1034. Road traffic, especially heavy duty diesel traffic, produces three particular forms of PM pollution: elemental carbon from exhaust emissions, semi-volatile organic carbon from diesel fuels and lubricants and non-exhaust material. **Current emission inventories do not characterise the emissions of these three PM components with sufficient accuracy for urban PM policy development purposes and validation studies are recommended at roadside and kerbside sites.** Without these important inventory validation studies, it will be difficult to say with any certainty what the appropriate balance should be between controls to diesel exhaust emissions, fuels and lubricants and traffic management to meet future target air quality in any given situation.

1035. Management of future regional PM background levels appears to be an attractive policy option for the control of future exceedences of target PM air quality because it operates city-wide and does not require detailed understanding of the spatial distribution of PM exceedences. SO_2 emissions have been falling, both in the UK and in the rest of Europe, significantly faster than ammonia emissions. This has had the result that, with the increased availability of ammonia, there has been possibly an increasing formation of ammonium nitrate during the wintertime. This may be the explanation of why particulate nitrate has taken over from particulate sulphate as the most important PM component in the regional PM background. With the UNECE Gothenburg Protocol and the EU National Emissions Ceilings

Directive, the trend of SO_2 emissions reducing faster than NH_3 emissions will continue. This makes the assessment of future particulate nitrate concentrations and regional PM background levels difficult. Without studies of regional PM pollution episodes with continuous monitors for particulate sulphate, particulate nitrate, PM_{10} and $PM_{2.5}$ at rural sites, it will be difficult to assess future regional PM background levels with any certainty. Further attention should be given to the cost and feasibility of reductions in NH^3 emissions from agriculture in the renegotiation of the UNECE Gothenburg Protocol and the EU National Emissions Ceilings Directive.

10.16 Summary of research recommendations

1036. Research requirements have been identified in the following areas.

1037. Development of a better understanding of non-exhaust traffic emissions , including:

a. measurement of the particle size distributions of brake wear, tyre wear and road dust resuspension;

b. investigation into the distinct chemical signatures of particles from brake wear, tyre wear and road dust resuspension, which may allow them to be identified and quantified; and

c. assessment of factors such as driving mode, vehicle weight and speed and heavy acceleration and braking, which are likely to influence emissions of non-exhaust particles from vehicles.

1038. The provision of robust PM emissions factors, using UK sources, suitable for use in inventories and dispersion models. Specific emissions categories identified as of high priority include the following:

a. emissions of particulate matter from construction, mining and quarrying;

b. primary PM emissions from natural gas combustion;

c. primary PM from coal combustion;

d. PM emissions and abatement from industry;

e. effect of fuel quality on PM emissions from combustion of liquid fuels; and

f. non-road mechanical machinery.

1039. Improvements to and refinement of the current model for source apportionment of PM in the UK including work on:

a. multivariate receptor modelling statistical methods applied to multi-element datasets;

b. the use of molecular source tracers;

c. single particle techniques such as aerosol mass spectrometry; and

d. isotope ratio techniques.

1040. The provision of particulate matter monitoring and characterisation data to support epidemiological studies.

1041. Development of state-of-the art comprehensive aerosol models capable of handling particle size distributions and detailed chemistry for the UK.

1042. Development of models for the coarse component both from the urban background, for example from construction sites and from traffic, both direct and non-exhaust emissions.

1043. Improved modelling and understanding of the impacts of road characteristics, including traffic induced turbulence, vehicle exhaust height and urban topography. These will both increase confidence of the road type adjustment factors and improve the reliability of dispersion models adjacent to roads.

1044. Further modelling of $PM_{2.5}$ should also be conducted in anticipation of new air quality standards. However, reliable model validation will require appropriate resolution of the difference between the different measurement techniques (for example, stipulation of appropriate conversion factors) .

10.17 Summary

1045. The Air Quality Expert Group appreciates the efforts made by Defra in establishing a comprehensive network of continuous PM monitoring. This network has shown that despite a decade of decreasing PM levels during the 1990s, problems still remain and future air quality targets for PM will not be met on current policies. It recognises that it will be difficult to build a robust strategy of future PM based on $PM_{2.5}$ and PM_{10} monitoring alone. **It recommends that in the next phase of PM monitoring, Defra establishes a comprehensive network for individual PM components based as far as possible on continuously recording monitors and co-located with existing monitoring activities.**

1046. Recommendations for how the existing monitoring network should be expanded are set out above and are summarised in Table 10.1.

1047. The Air Quality Expert Group is strongly of the view that unless and until mass closure can be achieved between PM components and PM_{10} and $PM_{2.5}$ mass, quantitative policy assessment will be compromised by uncertainty in basic understanding.

1048. The Air Quality Expert Group requests the opportunity to revisit the question of future PM policy when the continuous monitoring of PM components is in place. **It recommends that consideration is given to a more flexible and holistic approach to urban air quality management and to the control of acid rain, eutrophication and ground-level ozone.** Such approaches might deliver a more cost-effective solution to future PM air quality than that based on PM alone.

Table 10.1 Summary of monitoring recommendations.

Measurement	Recommendation
Continuous $PM_{2.5}$ nitrate	Expand existing network to provide paired urban background/rural data at four locations throughout the UK including London.
Continuous PM_{coarse} nitrate	Expand existing network to include measurements at the four urban background/rural locations identified above.
Continuous elemental and organic carbon	Expand existing network to provide paired roadside/urban background data at four locations throughout the UK including London.
Continuous $PM_{2.5}$ sulphate	Expand existing network to provide paired urban background/rural data at four locations throughout the UK including London.
Daily iron concentrations (PM_{10})	Expand existing network to include all Defra Site-Specific Receptor Model sites.

Abbreviations and glossary

Abbreviations

AC10	The ten Accession Member States that joined the EU in May 2004
ADMS	Atmospheric Dispersion Modelling System
AMS	Aerosol mass spectrometer
APEG	Airborne Particles Expert Group
APHEA	Air Pollution and Health, a European Approach
AQEG	Air Quality Expert Group
AQMA	Air quality management area
AQS	Air Quality Strategy
ATOFMS	Aerosol time-of-flight mass spectrometer
AURN	Automatic Urban and Rural Network (air quality monitoring)
BAM	Met One Beta Attenuation Monitor or BAM 1020
BaP	Benzo(a)pyrene
BAT	Best available techniques
BATNEEC	Best available techniques not entailing excessive cost
BS	Black smoke
BSI	British Standards Institute
BST	British Summer Time
CAFE	Clean Air For Europe
CALINE	California Line Source Model
CAP	Common Agricultural Policy
CCGT	Combined cycle gas turbines
CCS	Congestion Charging Scheme
CEH	Centre for Ecology and Hydrology
CEN	European Committee for Standardisation
CEPMEIP	Coordinated European Programme on Particulate Matter Emission Inventories, Projections and Guidance
CERC	Cambridge Environmental Research Consultants
CI	Confidence interval
CL	Confidence limit
CLRTAP	Convention on long-range transboundary air pollution
CNG	Compressed natural gas
CO	Carbon monoxide
CO_2	Carbon dioxide
COMEAP	Committee on the Medical Effects of Air Pollutants
COPERT III	Computer Programme to Calculate Emissions from Road Transport (version III)
CORINAIR	The air emissions section of CORINE
CORINE	CoOoRdination d'Information Environmentale
CPC	Condensation particle counter
CRT	Continuously regenerating traps
CVS	Constant volume sampler
DA	Devolved Administration
DAPPLE	Dispersion of Air Pollution and Penetration into the Local Environment
Defra	Department for Environment, Food and Rural Affairs
DfT	Department for Transport
DL	Detection limit
DMA	Differential mobility analyser
DMPS	Differential mobility particle sizer
DMRB	Design Manual for Roads and Bridges

DMS	Dimethyl sulphide
DPF	Diesel particulate filter
DTI	Department for Trade and Industry
EA	Environment Agency
EAF	Electric Arc Furnace
EC	European Community
EEA	European Environment Agency
EFTA4	European Fair Trade Agreement countries
ELR	European load response
EMEP	Cooperative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe
EPAQS	Expert Panel on Air Quality Standards
EPEFE	European Programme on Emissions, Fuels and Engine technologies
EPER	European Pollutant Emissions Register
EPSRC	Engineering and Physical Sciences Research Council
ERG	Environmental Research Group, King's College London
ESI	Electricity supply industry
ESP	Electrostatic precipitators
ETC-ACC	European Topic Centre on Air and Climate Change
EU	European Union
EU15	The 15 countries that were members of the European Union before May 2004
FAS	Free acceleration smoke
FDMS	Filter dynamics measurement system
FGD	Flue gas desulphurisation
FRAME	Fine Resolution Atmospheric Multi-pollutant Exchange
GDI	Gasoline direct injection
GIA	Global irradiation amount
GIS	Geographical information system
GLA	Greater London Authority
GMT	Greenwich Mean Time
HARM	Hull Acid Rain Model
HBAPMN	Hertfordshire and Bedfordshire Air Pollution Monitoring Network
HDV	Heavy duty vehicles – road vehicles greater than 3.5 tonnes weight (GVW)
HGV	Heavy goods vehicles – road vehicle greater than 7.5 tonnes (GVW), where GVW is the gross vehicle weight, i.e. the combined weight of the vehicle and the goods
HIRLAM	High Resolution Limited Area Model
HNO_3	Nitric acid
H_2SO_4	Sulphuric acid
HVS	High volume sampler
ICAO	International Civil Aviation Organisation
ICP-MS	Inductively coupled plasma – mass spectrometry
IGCB	Interdepartmental Group on Costs and Benefits
IIASA	International Institute for Applied Systems Analysis
INAA	Instrumental neutron activation analysis
IPC	Integrated pollution control
IPCC	Intergovernmental Panel on Climate Change
IPPC	Integrated Pollution Prevention and Control
JEP	Electricity Supply Industry Joint Supply Programme
KAQN	Kent Air Quality Network
KFG	Kleinfiltergerat (low volume sampler specified as a reference sampler for PM_{10})

kt	Kilotonne
LAEI	London Atmospheric Emissions Inventory
LAPC	Local air pollution control
LAQM	Local air quality management
LAQN	London Air Quality Network
LCPD	Large Combustion Plant Directive
LDV	Light Duty Vehicles – road vehicles less than 3.5 tonnes weight
LEZ	Low emission zone – a specific type of 'clear zone'
LGV	Light goods vehicles – goods vehicles less than 3.5 tonnes in weight
LNG	Liquid natural gas
LPG	Liquefied petroleum gas
LT	London Transport
LV	Limit value
LVS	Low volume sampler
MAQS	Mayor's Air Quality Strategy
MARPOL	Marine Pollution Convention
MODIS	Moderate resolution imaging spectroradiometer
MSW	Municipal solid waste
$mg\ m^{-3}$	Milligrams per cubic metre of air
µm	Micrometres
$\mu g\ m^{-3}$	Micrograms per cubic metre of air
NAEI	National Atmospheric Emissions Inventory
NAME	Numerical Atmospheric Dispersion Modelling Environment
Netcen	National environmental technology centre, part of AEA Technology plc
NH_3	Ammonia
NH_4	Ammonium
Nm	Nanometres
NMMAPS	National Morbidity, Mortality and Air Pollution Study
NMVOC	Non-methane volatile organic compound
NO	Nitrogen monoxide, also termed nitric oxide
NO_2	Nitrogen dioxide
NO_3	Nitrate
NO_X	Nitrogen oxides ($NO + NO_2$)
NPL	National Physical Laboratory
NRTF	National road traffic forecasts
O_2	Oxygen
O_3	Ozone
OEF	Oxford economic forecasting
OH	Hydroxyl radical
OPG	Other petroleum gas
OSPM	Operational Street Pollution Model
PAH	Polycyclic aromatic hydrocarbon
PAN	Peroxyacetyl nitrate
PCB	Polychlorinated biphenyl
PI	Pollution Inventory
PIXE	Particle-induced X-ray emission
PM	Particulate matter
PM_{10}	Airborne particulate matter passing a sampling inlet with a 50% efficiency cut-off at 10 µm aerodynamic diameter and which transmits particles of below this size
$PM_{2.5}$	Airborne particulate matter passing a sampling inlet with a 50% efficiency cut-off at 2.5 µm aerodynamic diameter and which transmits particles of below this size
PM_1	Airborne particulate matter passing a sampling inlet with a 50%

	efficiency cut-off at 1 µm aerodynamic diameter and which transmits particles of below this size
$PM_{0.1}$	Airborne particulate matter passing a sampling inlet with a 50% efficiency cut-off at 0.1 µm aerodynamic diameter and which transmits particles of below this size
PM_{coarse}	Fraction of the measured particle mass concentration determined from PM_{10} minus $PM_{2.5}$.
Ppb	Parts per billion (1,000,000,000)
PPC	Pollution Prevention and Control
Ppm	Parts per million
PSA	Particle surface area
QUARG	Quality of Urban Air Review Group
R&P	Rupprecht & Patashnick Co., Inc.
RGAR	Review Group on Acid Rain
RMA	Reduced major axis
RMS	Root mean square
ROS	Reactive oxygen species
SAQSG	Sussex Air Quality Steering Group
SCC	Sharp-cut cyclone
SEPA	Scottish Environment Protection Agency
SES	Sequential equilibrium system
SI	Spark ignition
SIA	Secondary inorganic aerosols
SJAC	Steam-jet aerosol collector
SMMT	Society of Motor Manufacturers and Traders Limited
SMPS	Scanning mobility particle sizer
SO_2	Sulphur dioxide
SO_4	Sulphate
SOA	Secondary organic aerosol
SSF	Solid smokeless fuel
TCA	Total cloud amount
TEOM	Tapered Element Oscillating Microbalance
TfL	Transport for London
TRAMAQ	DfT funded, Traffic management and air quality research programme. http://www.roads.dft.gov.uk/roadnetwork/tramaq/
TRL	Transport Research Laboratory
TSP	Total suspended particles
TYP	Ten Year Plan
WHO	World Health Organisation
UK	United Kingdom
UKAS	United Kingdom Accreditation Service
UKIAM	United Kingdom Integrated Assessment Model
UKOOA	United Kingdom Offshore Operations Association
ULSD	Ultra-low sulphur diesel
ULSP	Ultra-low sulphur petrol
UM	Unified model
UNECE	United Nations Economic Commission for Europe
USA	United States of America
USEPA	United States Environmental Protection Agency
VOC	Volatile organic compound
WRAC	Wide range aerosol classifier
WS	Wind speed
XRF	X-ray fluorescence

Glossary

Accumulation mode	Particles from around 0.5 to 1 μm diameter, resulting from primary emissions, condensation of secondary sulphates, nitrates and organics from the gas phase and coagulation of smaller particles. Particles can have a long atmospheric lifetime, typically 7–30 days.
Accuracy	A measure of the closeness of the agreement between the result of a measurement and the true value (see also Uncertainty and Precision).
Acute health effect	Short-lasting or short-term in reference to either duration of exposure or effect of exposure to a pollutant.
Aerosol	A mixture of suspended particulate matter and its gaseous suspended medium.
Air quality objective	Policy targets generally expressed as a maximum ambient concentration to be achieved, either without exception or with a permitted number of exceedences within a specified timescale (see also air quality standard).
Air quality standard	The concentration of a pollutant, and associated averaging period, which is without significant effect on human health at a population level.
Ambient air	Outdoor air in the troposphere, excluding workplace air.
Annual mean	The average of the concentrations measured for each pollutant for one year. In the case of the air quality objectives this is for a calendar year.
AQMA	Air quality management area, an area which a local authority has designated for action, based upon predicted exceedences of air quality objectives.
Atmospheric dispersion model	A mathematical, often computer-based method for calculating pollutant concentrations from emissions data and specified meteorological conditions. Models vary from screening models to detailed 'new-generation' types.
AURN	Automatic Urban and Rural Network of air pollution measurement sites, managed by contractors on behalf of Defra and the Devolved Administrations.
Black Smoke	Non-reflective (dark) particulate matter associated with the smoke stain measurement method (BS 1747 pt 2: BSI 1969).
Brownian motion	Constant small movement of suspended particles due to bombardment by surrounding molecules.
Calibration (modelling)	The process of multiplying the output of a model by a fixed correction factor to give, on average, a 1:1 relationship with measured data.
Calibration (monitoring)	The process of reducing the uncertainty of monitoring data by controlled tests on the analyser, normally traceable to internationally accepted measurements standards.
Carcinogenic	Known or believed to cause cancer in humans.
Cardiopulmonary	Pertaining to the heart and lungs.
Cardiovascular	Pertaining to the heart and blood vessel (circulatory) system.
Chronic health effect	Long-lasting or long-term in reference to either duration of exposure or effect of exposure to a pollutant.
Coagulation	Process by which particles collide and coalesce together.
Coarse particle mode	Particles greater than 1 μm diameter, typically generated mechanically rather than through nucleation and condensation processes. Atmospheric lifetimes are much shorter than for the accumulation mode.

Cohort study	Study in which a group or cohort of people are followed over time to see whether they develop a disease in response to exposure to the factor of interest.
Concentration	The amount of a (polluting) substance in a volume (of air), typically expressed as a mass of pollutant per unit volume of air at standard conditions of temperature and pressure (e.g. micrograms per cubic metre, µg m^{-3}) or as the ratio of the number of molecules of the pollutant to the total number of molecules in the volume of air (for example, parts per billion, ppb).
Condensation	A physical process with more vapour molecules arriving at particle's surface than leaving the surface, resulting in a net growth of the particle.
Confounding factor	A condition or variable that is both a risk factor for disease and associated with an exposure of interest. This association between the exposure of interest and the confounder may make it falsely appear that the exposure of interest is associated with the disease.
Correction factor	See scaling factor.
Correlation coefficient	The fraction of the variability in one set of data that is proportional to the value of some other set of data.
Data capture	The percentage of all the possible measurements for a given period that were validly measured.
Elemental carbon	Black, graphitic carbon formed in the high temperature combustion of fossil and contemporary biomass fuels.
Emission	The amount of a (polluting) substance emitted in a certain amount of time, typically expressed as a mass of pollutant per unit time (e.g., grams per second or tonnes per year for a single source). May also be expressed per unit length of a road (e.g., g s^{-1} m^{-1}), or per unit area of an urban area (e.g., t a^{-1} km^{-2}).
Emissions inventory	A quantification and compilation of emission sources by geography and time, usually including data covering one or more years.
Epidemiology	The study of the distribution and determinants of health and disease in populations.
EURO I	Europe-wide vehicle standard that required vehicles manufactured after 1992 to achieve set emissions limits. For petrol cars this was achieved by the fitting of three-way catalysts.
EURO II, III, IV and V	Europe-wide vehicle standards that are progressively stricter for the years 1996, 2000, 2006 and 2008, respectively.
Exceedence	A period of time where the concentration of a pollutant is greater than the appropriate air quality objective.
Endotoxin	Potent inflammatory agents present primarily in the cell walls of Gram-negative bacteria.
Genotoxic	A term used to describe carcinogens that act either directly or after transformation in the body on the genetic material (DNA) of cells.
Heterogeneous nucleation	Process in which newly formed low-volatility substances condense onto existing particles causing the growth of those particles.
Homogeneous nucleation	Process by which newly formed molecules of extremely low vapour pressure condense with one another to form wholly new particles.
In vitro	Taking place in isolation from a living organism.
In vivo	Taking place within a living biological organism.
Mass closure	The concept that sum of the individual chemical components equals the measured mass of particles.

Microgram (µg)	One millionth of a gram.
mg m^{-3}	Milligrams per cubic metre of air. A unit for describing the concentration of air pollutants in the atmosphere as a mass of pollutant per unit volume of air. This unit is one thousand-times larger than the µg m^{-3} unit listed below.
µg m^{-3}	Micrograms per cubic metre of air. A unit for describing the concentration of air pollutants in the atmosphere, as a mass of pollutant per unit volume of air. A concentration of 1 µg m^{-3} means that one cubic metre of air contains one microgram of pollutant.
Micrometre (µm)	One millionth of a metre, also referred to as a micron.
Minor roads	Non A roads or motorways.
Morbidity	Illness.
Mutagenic	Capable of increasing the rate of genetic mutation in living organisms.
Nanometre (nm)	10^{-9} metres.
Nanoparticle	Particle smaller than 50 nm diameter.
Nucleation	Process by which secondary particles are formed: molecules of low volatility condense to form solid or liquid matter (see also heterogeneous nucleation and homogeneous nucleation).
Nucleation mode	Particles smaller than around 50 nm and usually consisting of fresh aerosols created *in situ* from the gas-phase by nucleation. Such particles have a relatively short lifetime in the atmosphere.
Organic carbon	Carbon in the form of organic compounds, either primary from automotive or industrial sources or secondary from the oxidation of VOCs.
Particulate matter	Suspended particulate matter is any non-gaseous material (liquid or solid) that, owing to its small gravitational settling rate, remains suspended in the atmosphere for appreciable time periods.
PM$_{10}$	Airborne particulate matter passing a sampling inlet with a 50% efficiency cut-off at 10 µm aerodynamic diameter and which transmits particles of below this size.
PM$_{2.5}$	Airborne particulate matter passing a sampling inlet with a 50% efficiency cut-off at 2.5 µm aerodynamic diameter and which transmits particles of below this size.
PM$_{coarse}$	Fraction of the measured particle mass concentration determined from PM$_{10}$ minus PM$_{2.5}$.
ppb	Parts per billion. The concentration of a pollutant in air in terms of molar ratio. A concentration of 1 ppb means that for every billion (10^9) molecules in a volume of air, there is one molecule of the specified pollutant present. For practical purposes in ambient air, the molar ratio and volume ratio (the volume occupied by the pollutant gas within a given volume of air) are identical.
ppm	Parts per million. The concentration of a pollutant in air in terms of molar ratio. A concentration of 1 ppm means that for every million (10^6) molecules in a volume of air, there is one molecule of the specified pollutant present. For practical purposes in ambient air, the molar ratio and volume ratio (the volume occupied by the pollutant gas within a given volume of air) are identical.
Precision	A measure of the closeness of the agreement between the results of successive measurements where the true value remains constant (see also Accuracy and Uncertainty).

Primary particles	Particles emitted directly into the environment. This includes particles from both natural sources, such as the entrainment of soils by the wind, and anthropogenic sources, such as particles arising directly from processes such as combustion and quarrying.
Residual component	A component that is commonly incorporated into both semi-empirical and dispersion models to account for primary particulate matter that is not accounted for in the emissions inventory or otherwise included in the calculations. For predictions of PM_{10} concentrations, the residual component will usually be dominated by particles within the PM_{coarse} fraction and will comprise emissions from various sources including sea salt, wind and blown dust and any fraction of non-exhaust road vehicle emissions not otherwise included in the modelling.
Scaling factor	Due to the need to eliminate the effect of changing humidity on the mass measurement of PM, TEOMs must maintain the sample filter at an elevated temperature. This has led to reported differences in concentrations of PM between the TEOM and the European reference sampler that is largely attributed to the loss of volatile species such as ammonium nitrate. As an interim measure, a default 'scaling factor' (also known as correction factor) of 1.3 is currently applied to all nationally reported TEOM PM_{10} data in the UK as recommended by the EC Working Group on Particulate Matter.
Secondary particles	Particles formed in the atmosphere as a result of chemical reactions leading to the formation of substances of low volatility that consequently condense into the solid or liquid phase.
Stage II indicative limit values	The First Air Quality Daughter Directive (1999/30/EC) sets Stage II indicative limit values for both 24 h and annual average PM_{10} to be achieved by 1 January 2010. These Stage II limit values are only indicative and will be reviewed by the EC in light of further information on health and environmental effects, technical feasibility and experience gained in the application of Stage I limit values by Member States. They have no legal standing.
Susceptible group	A group of people who, as a result of genetic predisposition, illness or unusual exposure, are more affected by toxic substances than other people.
TEOM	Tapered element oscillating microbalance. Equipment used for measuring fine particulate matter such as PM_{10}.
Total suspended particles (TSP)	A term describing the mass of airborne particles, usually determined with a high-volume air sampler, which draws air through a filter membrane over a 24-hour period. Includes particles with a wide range of sizes.
True value	The value of a concentration, for example, which is entirely consistent with the definition of the units in which it is given. This is the value that would be obtained by a perfect measurement.
Ultrafine particles	Particles smaller than 100 nm diameter.
Uncertainty	A measure, associated with the result of a measurement, that characterizes the range of values within which the true value is expected to lie. Uncertainty is usually expressed as the range within which the true value is expected to lie with a 95% probability, where standard statistical and other procedures have been used to evaluate this figure. Uncertainty is more clearly defined than the closely related parameter accuracy, and has replaced it on recent European legislation.

Table A1.1 A summary of the regulation of major industrial sources of particles in England and Wales.

Industry sector	IPPC	LA IPPC	LAPPC
Energy	Combustion plants burning any fuel with >50 MW net thermal input Combustion plants burning waste-derived fuels with >3 MW net thermal input Coke ovens Crude oil refineries	—	Combustion plants burning any fuel with 20–50 MW net thermal input Combustion plants burning waste-derived fuels with <3 MW net thermal input
Metals	Integrated steel works Electric arc steel production Secondary production of non-ferrous metals. Melting of >4 t/day lead or >20 t/day of other non-ferrous metals	Ferrous foundries casting more than 20 t/day of metal	Processes melting or refining iron, steel, or ferrous alloy in cupolas, reverberatory, rotary or induction furnaces Melting of <4 t/day lead or <20 t/day of other non-ferrousmetals
Minerals	Cement clinker production Lime manufacture Fletton brick manufacture	Non-fletton brick manufacture Production of flat and container glass Grinding of metallurgical slag	Quarrying Coal handling processes Mobile crushing and screening processes Cement and concrete batching
Chemicals	Production of inorganic chemicals Production of chemical fertilizers	—	—
Waste	Incineration of municipal solid waste	—	—
Other industries	Production of paper and pulp Food and drink Intensive agriculture	Production of chipboard, fibreboard and oriented strand board	Processing of timber by sawing, drilling and so on Respraying of road vehicles

Key: IPPC, integrated pollution prevention and control; LA IPPC, local authority IPPC; LAPPC, local air pollution prevention and control.

Annex 2

Emission limit values for different vehicle types in Europe

Table A2.1 Emission limit values for different vehicle types in Europe since 1990. (Dates of implementation refer to new registrations.)

	Directive		Date of implementation		Test cycle	Units	NO$_x$	HC	NO$_x$ + HC	PM
Petrol cars	91/441/EEC	Euro I	1/7/1992		ECE15 + EUDC	g km^{-1}	—	—	0.97	—
	94/12/EC	Euro II	1/1/1997		ECE15 + EUDC	g km^{-1}	—	—	0.5	—
	98/69/EC	Euro III	1/1/2001		ECE15 + EUDC	g km^{-1}	0.15	0.20	—	—
	98/69/EC	Euro IV	1/1/2006		ECE15 + EUDC	g km^{-1}	0.08	0.10	—	—
Diesel cars	91/441/EEC	Euro I	1/7/1992		ECE15 + EUDC	g km^{-1}	—	—	—	0.14
	94/12/EC	Euro II	1/1/1997	IDI	ECE15 + EUDC	g km^{-1}	—	—	0.7	0.08
				DI	ECE15 + EUDC	—	—	—	0.9	0.1
	98/69/EC	Euro III	1/1/2001		ECE15 + EUDC	g km^{-1}	0.5	—	0.56	0.05
	98/69/EC	Euro IV	1/1/2006		ECE15 + EUDC	g km^{-1}	0.25	—	0.3	0.025
Petrol vans	93/59/EEC	Euro I	1/10/1994	<1305 kg	ECE15 + EUDC	g km^{-1}	—	—	—	—
			1/10/1994	1305-1760 kg		—	—	—	—	—
			1/10/1994	>1760 kg			—	—	—	—
	96/69/EEC	Euro II	1/1/1998	<1305 kg	ECE15 + EUDC	g km^{-1}	—	—	0.5	—
			1/1/1998	1305-1760 kg			—	—	0.65	—
			1/1/1998	>1760 kg			—	—	0.8	—
	98/69/EC	Euro III	1/1/2001	<1305 kg	ECE15 + EUDC	g km^{-1}	0.1 —5	0.2	—	—
			1/1/2002	13055-1760 kg			0.18	0.25	—	—
			1/1/2002	>1760 kg			0.21	0.29	—	—
	98/69/EC	Euro IV	1/1/2006	<1305 kg	ECE15 + EUDC	g km^{-1}	0.08	0.1	—	—
			1/1/2006	13055-1760 kg			0.1	0.13	—	—
			1/1/2006	>1760 kg			0.11	0.16	—	—

	Directive		Date of implementation		Test cycle	Units	NO$_x$	HC	NO$_x$ + HC	PM
Diesel vans	93/59/EEC	Euro I	1/10/1994	<1305 kg	ECE15 + EUDC	g km^{-1}	—	—	0.97	0.14
			1/10/1994	1305 1760 kg			—	—	1.4	0.19
			1/10/1994	>1760 kg			—	—	1.7	0.25
	96/69/EEC	Euro II	1/1/1998	<1305 kg	ECE15 + EUDC	g km^{-1}	—	—	0.60	0.1
			1/1/1998	1305—1760 kg			—	—	1.1	0.15
			1/1/1998	>1760 kg			—	—	1.3	0.2
	98/69/EC	Euro III	1/1/2001	<1305 kg	ECE15 + EUDC	g km^{-1}	0.5	—	0.56	0.05
			1/1/2002	1305—1760 kg			0.65	—	0.72	0.07
			1/1/2002	>1760 kg			0.78	—	0.86	0.1
	98/69/EC	Euro IV	1/1/2006	<1305 kg	ECE15 + EUDC	g km^{-1}	0.25	—	0.3	0.025
			1/1/2006	1305 1760 kg			0.33	—	0.39	0.04
			1/1/2006	>1760 kg			0.39	—	0.46	0.06
Heavy duty vehicles	91/542/EEC	Euro I	1/10/1993		ECE R49	g kW^{-1} h^{-1}	8	1.1	—	0.36
	91/542/EEC	Euro II	1/10/1996		ECE R49	g kW^{-1} h^{-1}	7	1.1	—	0.15
	99/96/EC	Euro III	1/10/2001		ESC + ELR	g kW^{-1} h^{-1}	5	0.66	—	0.1
					ETC	g kW^{-1} h^{-1}	5	0.78	—	0.16
	99/96/EC	Euro IV	1/10/2006		ESC + ELR	g kW^{-1} h^{-1}	3.5	0.46	—	0.02
					ETC	g kW^{-1} h^{-1}	3.5	0.55	—	0.03
	99/96/EC	Euro IV (Pt 2)	1/10/2008		ESC + ELR	g kW^{-1} h^{-1}	2	0.46	—	0.02
					ETC	g kW^{-1} h^{-1}	2	0.55	—	0.03

Annex 3

The regulatory test method, in-service emissions testing, the measurement of exhaust particles and particles research

A3.1 Limitations of the regulatory test method

1. Ntziachristos *et al*. (2004a) have noted that there are clear advantages to the CVS procedure for certification tests. Dilution of the exhaust reduces the risk of water condensation in the sampling and transport lines; stabilisation of the exhaust reduces pressure and temperature fluctuations, thus simplifying the sampling procedure; and flow rate is precisely controlled and measured. However, a method based on the use of a CVS and the measurement of filter mass alone – whether for comparison against a standard or for any other purpose – has several limitations. Some of these limitations are outlined below.

- Reliability of measurements for low-emitting vehicles. A significant problem with the filter technique is the very low mass emissions associated with modern vehicles.

- Lack of control over sampling conditions. Several exhaust sampling parameters are left uncontrolled in the CVS.

- The particle size distribution and other metrics are not determined. A standard based solely on total particulate mass is probably not ideal in terms of minimising the risks of exhaust emissions to health.

- The single mass value does not permit a detailed understanding of factors influencing particle emissions.

2. Consequently, various alternative methods have been explored for sampling and characterising exhaust particles, and a number of metrics have been proposed in addition to filter mass. Possible metrics for particle characterisation include the size distribution (number or mass-weighted), number concentration, surface area and information on the chemical character of particles (Ntziachristos and Samaras, 2003). However, there are a number of issues relating to both the nature and measurement of particles that still need to be resolved.

A3.2 In-service emissions testing

3. For legislative purposes, the Type Approval testing for particulate emissions is only applicable to diesel engines. At the Type Approval stage, particulate emissions have conventionally been measured gravimetrically. However, EU Directive 99/96/EC introduced the European Load Response (ELR) test, applicable to Euro III and subsequent engines. This engine test consists of a sequence of load steps at constant engine speeds and was introduced for the purpose of smoke opacity

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[1] Full details of the FAS test method and associated equipment are available at http://www.gea.co.uk/docs/130.doc.

measurement. It is important to note that when these vehicles enter service, yearly MOT testing is undertaken using only a measure of exhaust opacity, through the use of an opacity meter[1]. The actual test procedure is referred to as the Free Acceleration Smoke (FAS) test and involves a sequence of up to six accelerations, leaving a gap of at least 10 s between each full depression of the throttle, from engine idle to engine cut-off (governed) speed. However, some subjectivity is also applied to this test procedure, as the results can be affected by the way in which the throttle is depressed and released. Furthermore, it is essential to ensure that the engine and any turbocharger fitted should be at idle prior to the start of each FAS cycle. The calculated smoke value is the arithmetic mean of the last three measurements, on the basis that no measurement is less than 75% of this mean smoke level. In the event that this criterion is not met, the test is returned as void, forcing a repeat or ultimately a test failure.

A3.3 Issues relating to the measurement of exhaust particles

4 Kittelson *et al*. (2002) have noted that the particle size distribution in diesel exhaust is influenced by many factors. These include engine condition, fuel and lubricating oil composition, exhaust system configuration, the sampling system used and the sampling conditions. Engine and sampling system stabilisation as well as previous engine operating history play important roles in determining the size distribution. Furthermore, ambient weather conditions, such as temperature, also affect on-road size distributions. All of these factors make precise definition and simulation of the on-road operating and dilution conditions very difficult. Measurement of the nucleation mode is the most difficult because it is dominated by particles that are formed by the condensation of lower volatility organic precursors as the exhaust dilutes and cools; this process is extremely dependent upon dilution conditions and these nuclei and their precursors may be strongly influenced by losses and sampling artefacts. Ntziachristos *et al*. (2004a) noted that various results could be reported depending upon the sampling parameters used in a particular study.

5 Once the exhaust gas has left the exhaust pipe, several phenomena may occur in the dilution tunnel and sampling system, and these are responsible for changes in a range of properties associated with the PM. They may lead to errors in the particle mass measurement, but more significantly to the characterisation of particle number emissions and size distribution (Burtscher and Majewski, 2004). They include the following.

- Condensation/nucleation: this process occurs in the exhaust system and sampling lines and can be influenced by the way in which the exhaust is diluted and cooled.

- Particle coagulation: the coagulation rate depends on the square of the number concentration. It affects both particle number and size distribution.

- Particle sample losses: although sampling systems are designed to limit these losses, they may occur through several processes including inertial impaction, diffusion and thermophoresis[2] and electrostatic deposition. The loss of

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[2] Thermophoresis is the induced movement of PM by thermal gradient effects. Transparent particles tend to move towards heat sources while opaque particles move away from heat sources.

particles in the smallest size fractions, due to their high diffusion coefficients, is disproportionately high.

- Deposition/re-entrainment: particles may be deposited and subsequently re-entrained in the exhaust system.
- Particle generation from sampling system: particles may be also generated from within the sample lines used to convey the gas from the exhaust pipe to the dilution tunnel.

A3.4 Particle research programmes

6. A novel PM sampling system was designed and constructed in the EU Fifth Framework PARTICULATES project. To address the concerns relating to particle sampling conditions and to allow preferential conditioning of aerosol samples, Ntziachristos *et al*. (2004a) designed a partial-flow sampling system that drew exhaust directly from the tailpipe. The intention was not to develop a system for use in certification tests, but rather to achieve the repeatable formation of the nucleation mode in the laboratory using specific sampling parameters (dilution ratio, residence time, temperature and humidity). This would allow the identification of cases (engine operating modes, fuels and so on) where there is a potential for nucleation mode formation. The sampling system was also designed to measure a broad range of particle properties over transient tests, including size, number concentration and surface area. Additionally, the physical separation of solid soot particles and volatile particles was deemed necessary.

7. The GRPE-PMP programme was initiated in January 2001 and conducted under the auspices of the UNECE WP29/GRPE group. The countries that participated actively in the PMP project were France, UK, Germany, Sweden, The Netherlands and Switzerland. The aim of the programme was to provide recommendations for a new or additional PM measurement system to be used for EU Type Approval testing as well as advice on the development of future emission standards for both light and heavy duty vehicles. The objectives of the programme were to identify the best metrics for future particle measurements, determine which instruments and methods could utilize those metrics, investigate a test procedure for the measurement of particles during Type Approval tests and recommend a suitable test system or systems. Within the programme, a simultaneous comparison was made between 20 particle characterisation instruments.

8. The programme concentrated on the development of a methodology to measure solid (insoluble) particles in the size range of 20–300 nm for three reasons:

(i) The high probability of alveolar deposition, fast penetration into the vascular system and long residence time.

(ii) The availability of filters as highly efficient and technically proven tools to clean the exhaust gas of diesel to an extent that is new in air pollutant abatement.

(iii) Future diesels with PM emissions below the detection limit of the EPA 2007 method that will still emit millions of such particles <100 nm cm^{-3}.

9. It was argued that volatile particles will gradually cease to be an emission problem due to the use of sulphur-free fuels, low-sulphur lubricants and oxidation catalysts. When considering number concentrations, it was envisaged that the nucleation mode should be prevented to ensure consistent and repeatable measurements.

10. The main recommendation from the PMP programme was that two systems should be considered for future regulatory use: a modified 2007-PM (a gravimetric filter-based mass measurement system) and a CVS + thermodiluter + CPC (a number-based measurement system).

Annex 4

Summary of PM_{10} and $PM_{2.5}$ monitoring methods in Europe

1. An analysis of PM_{10} and $PM_{2.5}$ monitoring methods used in the various networks across Europe has been recently completed (CAFE Working Group, 2003). A summary is provided in Table A4.1. ß-absorption analysers, followed by TEOM analysers are the most widely used instruments

Table A4.1 PM_{10} and $PM_{2.5}$ measurement methods currently used in European countries (source: CAFE Working Group on Particulate Matter, 2004).

Country	PM_{10} (24 h)		$PM_{2.5}$ (24 h)	
	Method	Number of measuring points	Method	Number of measuring points
Austria	ß-absorption TEOM Gravimetry	31 12 12	Gravimetry	1
Belgium	ß-absorption TEOM	17 21	TEOM	10
Bulgaria	ß-absorption	8	—	—
Czech Republic	ß-absorption TEOM Gravimetry	55 2 10	—	—
Denmark	ß-absorption TEOM Gravimetry	10 2 10	TEOM	2
Estonia	ß-absorption	3	—	—
Finland	ß-absorption TEOM Gravimetry	13 28 13	ß-absorption	4
France	ß-absorption TEOM	72 253	TEOM	43
Germany	ß-absorption TEOM Gravimetry	254 16 169	ß-absorption TEOM Gravimetry	2 2 19
Greece	ß-absorption TEOM Gravimetry	14 3 1	Gravimetry	1

Country	PM_{10} (24 h)		$PM_{2.5}$ (24 h)	
	Method	**Number of measuring points**	**Method**	**Number of measuring points**
Hungary	ß-absorption Gravimetry	22 2	Gravimetry	2
Iceland	ß-absorption	1	—	—
Italy	ß-absorption Gravimetry	22 4	—	—
Latvia	ß-absorption	1	—	—
Norway	ß-absorption TEOM Gravimetry	10 10 2	TEOM Gravimetry	3 2
Poland	ß-absorption Gravimetry	9 18	—	—
Portugal	ß-absorption	11	—	—
Slovakia	TEOM	23	TEOM	6
Slovenia	Gravimetry	3	—	—
Spain	ß-absorption TEOM Gravimetry Optical	194 61 75 11	TEOM Gravimetry Optical	2 11 10
Sweden	TEOM Gravimetry	18 8	TEOM Gravimetry	10 8
Switzerland	ß-absorption TEOM Gravimetry	10 10 35	Gravimetry	6
Netherlands	ß-absorption	19	TEOM	5
United Kingdom	ß-absorption TEOM Gravimetry	1 69 6	TEOM	4

Annex 5

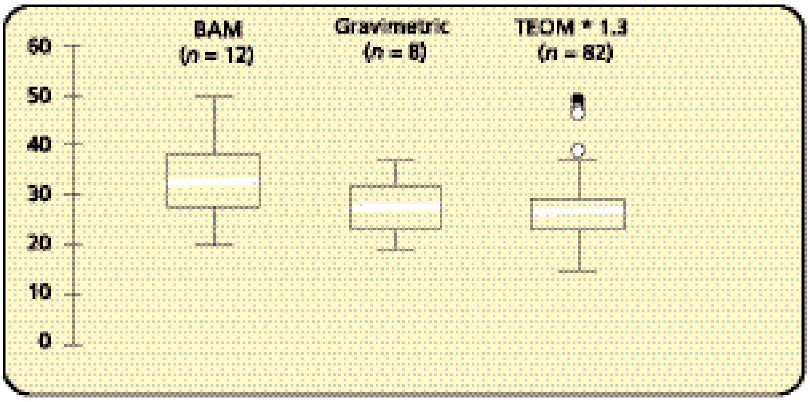
Performance of unheated ß-attenuation analysers

1. ß-attenuation analysers are widely used in the UK by local authorities as well as at one AURN site. They are predominately unheated Met One Beta Attenuation Monitors (BAM 1020). As these samplers are unheated, they will tend to exhibit the same losses of semi-volatile material as a sampler operating at ambient conditions, and the advice has been to treat the results as gravimetric equivalent. There is mounting evidence that these monitors overestimate gravimetric concentrations, which is believed to be a result of water on the filters. (There is now a version of the BAM available that has a 'Smart Heater', which controls the temperature to no more than +5°C above ambient, so as to maintain humidity at 20–45% and minimise problems due to water.) This Annex assembles and assesses the evidence for the performance of unheated ß-attenuation monitors.

2. There have been no systematic long-term co-location studies of ß-attenuation monitors and gravimetric samplers in the UK. A 4-month study was carried out in 1998 by KCL-ERG that used a ß-attenuation monitor alongside a Partisol gravimetric sampler: it showed a mean PM_{10} concentration that was19% higher with the ß-attenuation monitor. The remaining evidence is derived from indirect comparisons.

3. Local authorities throughout the UK have been monitoring PM_{10} at roadside locations as part of their LAQM duties. The information from these studies has been collated and is summarised in Figure A5.1. Assuming that the types of monitor employed are independently distributed in relation to distance from the road and traffic flow, that is, that ß-attenuation monitors are as likely to be used at the kerbside of a very busy road as a TEOM or a gravimetric sampler, then the results suggest that, on average, ß-attenuation monitors give rise to annual mean

Figure A5.1 Box and whisker summary of mean PM_{10} concentrations (μg m^{-3}) measured by local authorities throughout the UK at roadside sites using three different instruments. (Most studies were conducted for 12-month periods. The central line is the mean for each dataset, while the box shows the interquartile range.)



concentrations that are 18% higher than gravimetric values and 22% higher than TEOM * 1.3 values. These results must be treated with some caution, as the assumption may not be valid. Thus, these results alone are insufficient to reach a firm conclusion, but they will add weight to the findings of other indirect studies, as set out below.

4. There is now a substantial monitoring network of PM_{10} monitors in and around London that includes both TEOM and ß-attenuation monitors. The results for 'background' sites in 2003 are shown in Table A5.1. The TEOM * 1.3 annual mean results have been used in Chapter 6 to plot a concentration surface across London (Figure 6.49). Figure A5.2 shows a similar plot including the ß-attenuation monitor results. It is clear that at least three of the four ß-attenuation monitor sites stand out from the pattern created by TEOM * 1.3 data. If the TEOM * 1.3 concentrations at each of these ß-attenuation monitor sites are estimated from Figure 6.49, then this shows that the ß-attenuation monitors are giving rise to annual mean concentrations that are on average 27% higher than the TEOM * 1.3 values.

5. Further evidence of the performance of ß-attenuation monitors derives from the national model developed by Netcen (Section 8.2.1.2). This model has been compared with 2002 measured annual mean PM_{10} values from national monitoring sites. The results are shown separately for roadside and background sites in Figures A5.3 and A5.4. The TEOM * 1.3 values are predicted without obvious bias (except perhaps for a slight underprediction at roadside sites), but on average ß-attenuation monitor values are underpredicted by around 15%. This would suggest that ß-attenuation monitor values are about 18% higher than TEOM * 1.3 values.

Figure A5.2 Interpolated annual mean PM_{10} background concentrations across London in 2003 using ß-attenuation monitor and TEOM * 1.3 data. (May be compared with Figure 6.49, which is TEOM * 1.3 data only.)

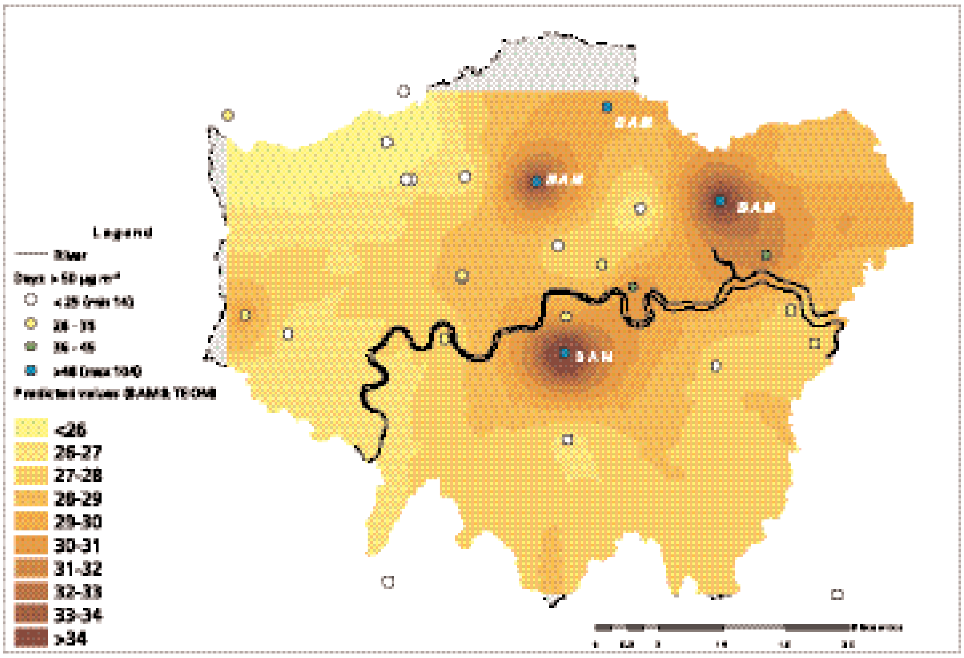


Table A5.1 PM$_{10}$ monitoring at 'background' sites in London in 2003.

Site name	Site type	Network	Site code	Method	Days >50 µg m^{-3}	Data capture (%)	Annual mean TEOM * 1.3 (µg m^{-3})
London Bexley	Suburban	AURN	GB0608A	TEOM	33	87	27
London Brent	Urban centre	AURN	GB0616A	TEOM	24	86	27
London Eltham	Suburban	AURN	GB0586A	TEOM	24	89	27
London Hillingdon	Suburban	AURN	GB0642A	TEOM	28	80	30
London N. Kensington	Background	AURN	GB0620A	TEOM	28	88	29
Thurrock	Background	AURN	GB0645A	TEOM	38	88	31
Hertsmere background (Borehamwood)	Background	HBAPMN	HM1	TEOM	20	88	25
Three Rivers background	Background	HBAPMN	TR1	TEOM	27	78	23
Sevenoaks 2 – Greatness	Background	KAQMN	ZV1	TEOM	14	91	23
Barking & Dagenham 2 – Scrattons Farm	Suburban	LAQN	BG2	TEOM	43	89	32
Barnet 2 – Finchley	Background	LAQN	BN2	TEOM	23	90	26
Bexley 2 – Belvedere	Suburban	LAQN	BX2	TEOM	30	88	27
Brent 1 – Kingsbury (AURN)	Background	LAQN	BT1	TEOM	25	86	26
Croydon 3 – Thornton Heath	Suburban	LAQN	CR3	TEOM	17	86	26
Enfield 3 – Salisbury School Ponders End	Background	LAQN	EN3	**BAM**	41	82	**29**
Greenwich 4 – Eltham	Background	LAQN	GR4	TEOM	24	91	27
Haringey 2 – Priory Park	Suburban	LAQN	HG2	**BAM**	65	87	**35**
Harrow 1 – Stanmore Background	Background	LAQN	HR1	TEOM	16	85	23
Hounslow 2 – Cranford	Suburban	LAQN	HS2	TEOM	20	88	26
Islington 1 – Upper Street	Background	LAQN	IS1	TEOM	25	91	27

Site name	Site type	Network	Site code	Method	Days >50 µg m^{-3}	Data capture (%)	Annual mean TEOM * 1.3 (µg m^{-3})
Kens and Chelsea 1 – North Kensington	Background	LAQN	KC1	TEOM	29	90	29
Lambeth 3 – Loughborough Junction	Background	LAQN	LB3	**BAM**	69	89	**38**
Mole Valley 2 – Lower Ashstead	Suburban	LAQN	MV2	TEOM	14	91	25
Redbridge 1 – Perth Terrace	Background	LAQN	RB1	**BAM**	65	86	**36**
Richmond 2 – Barnes Wetlands	Suburban	LAQN	RI2	TEOM	33	90	27
Southwark 1 – Elephant and Castle	Background	LAQN	SK1	TEOM	31	91	30
Tower Hamlets 1 – Poplar	Background	LAQN	TH1	TEOM	42	88	31
Tower Hamlets 3 – Bethnal Green	Background	LAQN	TH3	TEOM	28	85	27
Waltham Forest 1 – Dawlish Road	Background	LAQN	WL1	TEOM	16	69	25

Key: BAM, ß-attenuation monitor.

Figure A5.3 Comparison of national modelled and monitored PM_{10} in 2002 at roadside sites.

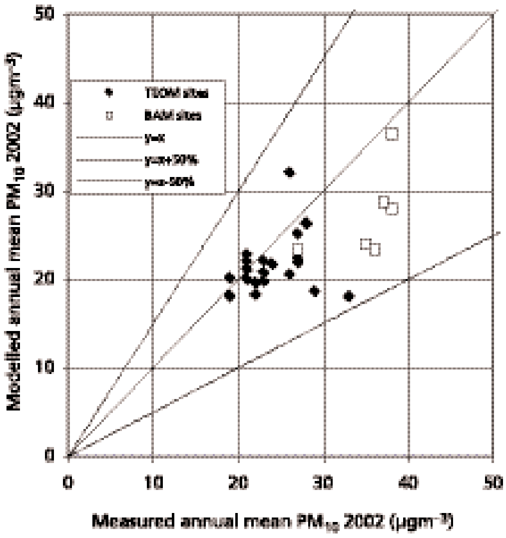
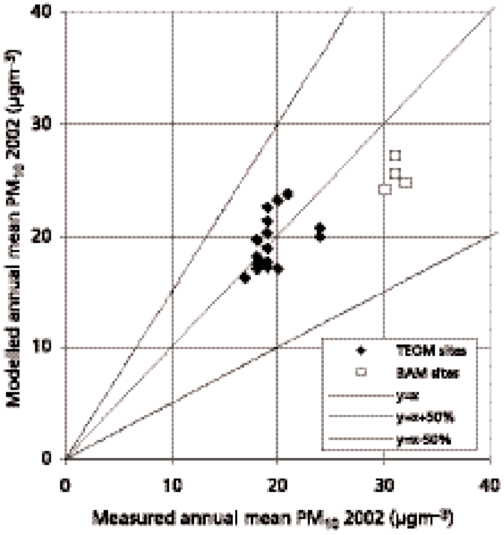


Figure A5.4 Comparison of national modelled and monitored PM_{10} in 2002 at background sites.



6. Finally, the results have been examined to see whether there is a different relationship between the annual mean values for unheated ß-attenuation monitors and the number of days above 50 µg m^{-3} (Figures A5.5 and A5.6). The results show no strong differences between the three types of sampler. TEOM * 1.3 values are broadly similar to the patterns for gravimetric samplers and ß-attenuation monitors at both roadside and background sites, although there is some evidence that TEOM * 1.3 values lie somewhat below those for gravimetric samplers and ß-attenuation monitors, especially at locations with lower annual mean concentrations.

Figure A5.5 Relationship between number of exceedences of 50 µg m^{-3} and annual mean PM_{10} concentrations at background sites for three different instruments. (The consultation draft of this report included an incorrect graph, which showed BAM monitors to have a different relationship. This graph shows the corrected data.)

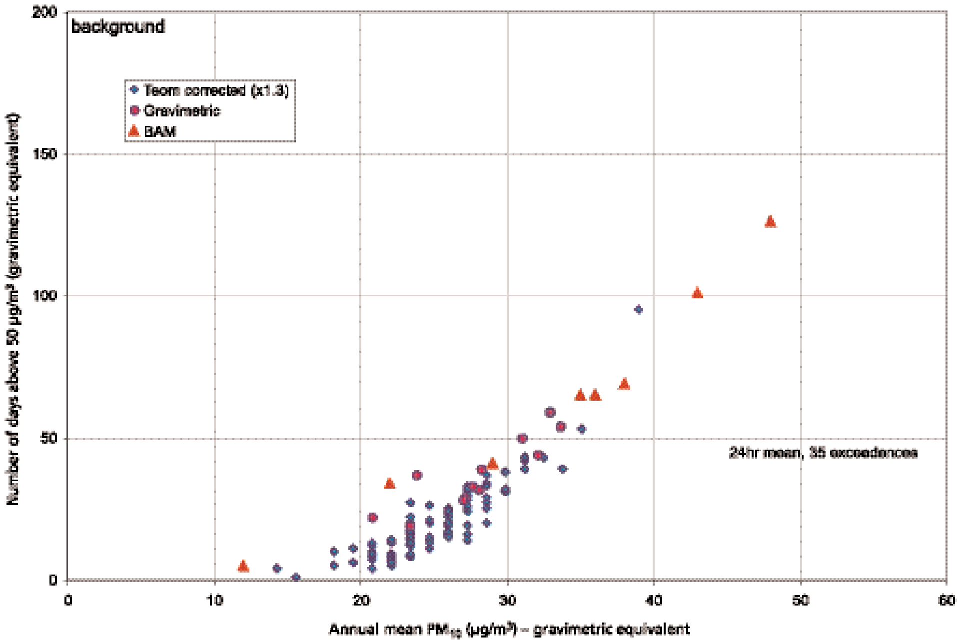
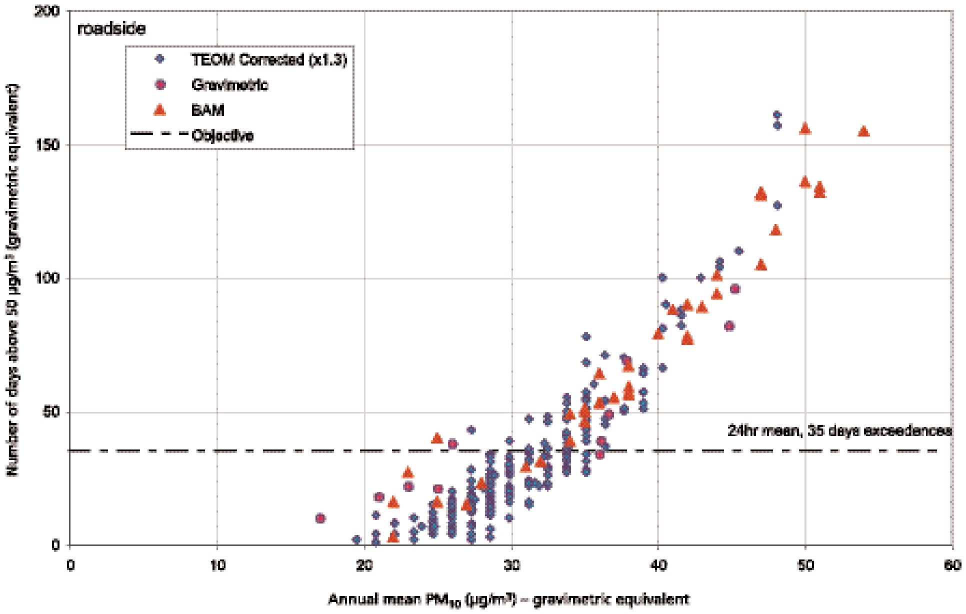


Figure A5.6 Relationship between number of exceedences of 50 µg m^{-3} and annual mean PM_{10} concentrations at roadside sites for three different instruments. (The consultation draft of the report included an incorrect graph, which showed BAM monitors to have a different relationship. This graph shows the corrected data.)



7. In conclusion, there is mounting evidence from a number of sources to show that unheated BAM 1020 analysers overestimate gravimetric PM_{10} concentrations. The annual means derived from ß-attenuation monitors appear to be around 20% higher than gravimetric or TEOM * 1.3 values. Improved information will derive from collocation studies that are being initiated by Defra and Devolved Administrations, which will include the unheated BAM 1020 sampler and the BAM 1020 with the Smart Heater.

Annex 6

Potential implications of the internal TEOM offset

1. The TEOM offset described in Chapter 5 is applied to all UK measurements of both PM_{10} and $PM_{2.5}$. The full implications of the offset to UK ambient measurements is currently not known and requires further investigation. A summary of the potential effects is given in the following paragraphs.

A6.1 Comparison of TEOM with gravimetric measurements

2. Currently in the UK, a scaling factor of 1.3 is applied to TEOM measurements in an attempt to provide an estimate of the equivalent gravimetric concentration, as described in Section 5.3.2. In the absence of the internal TEOM offset, the estimation of this scaling factor would change. An initial comparison between the TEOM and gravimetric (Partisol) instruments at the London Kensington site was undertaken to explore how the relationship between the two instruments would change.

3. A regression analysis of TEOM measurements against the 'Gravimetric' (Partisol) measurements over the period January 2002 to October 2003 at the Kensington site yielded a relationship 'gravimetric' (Partisol) = 1.48 * TEOM –3.7 (r^2 = 0.78). Forcing the regression line through 0 gave a slope of 1.325 (that is, close to 1.3). With the TEOM offset removed, the relationship was 'gravimetric' (Partisol) = 1.51*TEOM 'raw' + 0.7 (r^2 = 0.78), that is the intercept was closer to 0 in the case where the offset was removed. Forcing the regression line through 0 gave a slope of 1.55.

4. Table A3.1 summarises the data analysis in terms of the number of days above 50 µg m^{-3}. The TEOM 'raw' * 1.55 has more days with mean PM_{10} concentration above 50 µg m^{-3} because of the larger scaling factor and absence of an offset. The impact on annual and daily mean EU limit value is different. This analysis suggests that removing the TEOM offset enables the derivation of a more accurate TEOM to 'gravimetric' scaling factor at this site. It should be stressed, however, that although an improvement was observed in the accuracy of the scaling factor at this site by removing the internal offset, further more detailed analysis is required to confirm that a similar improvement would be achieved at other locations.

Table A3.1 Comparison between the TEOM and Partisol instruments at the London Kensington site (January 2002 to October 2003).

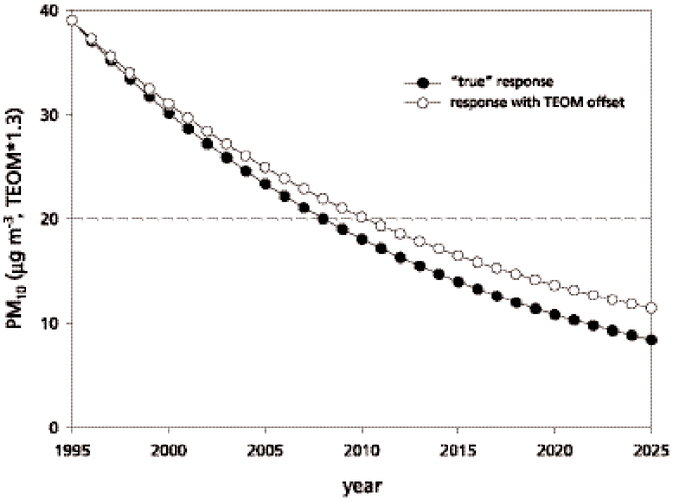
	Partisol	TEOM	TEOM 'raw'	TEOM * 1.325	TEOM 'raw' * 1.551
Mean concentration (µg m^{-3})	27	21	17	28	27
Number of days above 50 µg m^{-3}	44	8	5	35	39

A6.2 Effect on PM_{10} and $PM_{2.5}$ trends

5. The TEOM offset also affects the observed trends in PM_{10} and $PM_{2.5}$. A TEOM instrument would record 3.0 µg m^{-3} even if particle concentrations were 0. The actual effects are complex and would depend on how the data were processed, for example the application of factors other than 1.3 to initially attempt to give the TEOM some sort of equivalence with the gravimetric method. Figure A3.1 shows the hypothetical effect on the trend in PM_{10} at a monitoring site. Two cases have been considered: with and without and offset. An initial concentration of 39 µg m^{-3} (TEOM * 1.3) is assumed in 1995, which is thought to be typical of an inner London roadside location. For the TEOM without the offset a larger factor than 1.3 is required to give the same equivalence with the 'gravimetric' method (in this case a factor of 1.49 would be required based on annual mean concentration) for the base year in 1995. It has then been assumed that ambient PM_{10} concentrations are reduced by 5% year on year, that is, not a linear decrease. For the case with the offset it has been assumed that the 5% annual reduction in PM_{10} only applies to the fraction of PM_{10} that does not include the offset.

6. By 2005 the concentrations would have reduced to 23.4 and 24.9 µg m^{-3} for the without/with offset respectively. By 2010 the corresponding concentrations are 18.1 and 20.2 µg m^{-3}. Clearly, there is a significant difference between the two modes of operation on measured future ambient concentrations and it is important to note that with the offset on, the measured concentration would be above the limit value for this example.

Figure A3.1 Potential effect on the trend in PM_{10} concentrations on TEOM measurements for TEOMs with and without the internal offset scaling.



7. Figure A3.1 shows, therefore, that the TEOM with the offset responds more slowly to decreases in atmospheric concentration than the same instrument with the offset removed. The implication of this example is that the trends in measured PM_{10} and $PM_{2.5}$ considered in Chapter 7 could give a pessimistic impression of the actual reductions in airborne particulate, both PM_{10} and $PM_{2.5}$.

A6.3 Effect on $PM_{2.5}$ to PM_{10} ratios

8. The ratio of $PM_{2.5}$ to PM_{10} is important since this is the basis by which CAFE proposed a range of limit values for $PM_{2.5}$.
The effect on the $PM_{2.5}$ to PM_{10} ratio will depend on the ambient concentrations of each particle fraction and will therefore be site dependent. Table A3.2 shows the effect of the offset at two locations in London. At the Bexley suburban site the ratio decreases from 0.58 to 0.50 if the offset is removed. At the Marylebone Road kerbside sites, where concentrations are much higher than Bexley, the ratio declines less (from 0.65 to 0.61).

9. The constant offset of 3.0 $\mu g\ m^{-3}$ for both PM_{10} and $PM_{2.5}$ TEOM measurements is relatively more important for $PM_{2.5}$ than PM_{10}. This is because ambient concentrations of $PM_{2.5}$ are less than PM_{10} and, therefore, the offset accounts for a larger proportion of $PM_{2.5}$ measurements.

10. Further work is required to calculate the effect on the $PM_{2.5}$ to PM_{10} ratio at UK and European sites. Following this work it would also be necessary to consider the implications (if any) for setting an ambient air pollution standard for $PM_{2.5}$, as is currently recommended by the CAFE Working Group (CAFE, 2004). CAFE has recommended a $PM_{2.5}$ standard derived from a mean $PM_{2.5}$ to PM_{10} ratio of 0.6.

11. The removal of the offset factor would also affect the fraction of particles assumed to be coarse, although not the absolute concentration. Using the results in Table A3.2 as an example, the coarse particle fraction would increase at Bexley Belvedere from being 42% of the total PM_{10} to 50%. Likewise, at Marylebone Road the coarse fraction would increase from 35% to 39%.

Table A3.2 Effect on $PM_{2.5}$:PM_{10} ratios of the TEOM offset (2002).

	PM_{10}	**$PM_{2.5}$**	**$PM_{2.5}$:PM_{10} ratio**	**$PM_{2.5}$:PM_{10} ratio (offset removed)**
Bexley Belvedere (suburban)	11	19	0.58	0.50
Marylebone Road (kerbside)	22	34	0.65	0.61

A6.4 Effect on modelled concentrations

12. The effect of the TEOM offset on the modelled concentrations shown in this report is both complex and dependent on how the models were initially constructed. In the case of the national empirical modelling described in Chapter 8, an initial assessment has been made of the consequences of the TEOM offset on predicted concentrations at the London Kensington and Marylebone Road sites. Again, it should be stressed that the analysis is preliminary and aims only to illustrate the principal potential effects of the offset factor.

Table A3.3 Potential effect of the TEOM offset factor on the national site-specific PM_{10} modelling.

	Year	Primary	Secondary	Other	Total (TEOM)	TEOM factor	Total (gravimetric equivalent)
With offset	2002	4.6	4.9	10	19.5	1.3	25.4
	2010	3.6	4.2	10	17.8	1.3	23.1
No offset	2002	4.6	4.9	7	16.5	1.5	24.8
	2010	3.6	4.2	7	14.8	1.5	22.2

13. Table A3.3 illustrates the potential effects on PM10 predictions using the national empirical modelling on predicted concentrations at the London Kensington site. For the purposes of this illustration it has been assumed that a factor of 1.5 is applicable to convert the TEOM to gravimetric equivalent in the case where the TEOM offset is not used. The overall effect in the model is to reduce by 3.0 µg m–3 the PM10 that is considered in the 'other' category for the case without the offset. The table also shows that the reduction in PM10 concentration predicted between 2002 and 2010 is different in each case. With the TEOM offset the reduction is 2.2 µg m–3 and without the offset it is 2.6 µg m–3. A similar analysis at Marylebone Road also shows that the projected decrease in PM10 between 2002–2010 with the offset removed is larger (10.7 versus 9.2 µg m–3). These results, therefore, suggest that current national modelling of PM10 could be pessimistic in terms of the future reductions in PM10 concentration.

14. The offset factor would manifest itself in a different way for the ERG modelling. In the regression analyses used to derive the different particle fractions, the TEOM offset would be considered as part of the secondary aerosol fraction. It is, therefore, likely that the ERG results are optimistic with regard to future concentrations since the constant TEOM offset of 3.0 µg m^{-3} would also have been adjusted downwards into the future in response to declining concentrations of secondary aerosol. Further work is being undertaken to determine the implications of the offset in the ERG model.

15. A consideration of the TEOM offset used for $PM_{2.5}$ in Canada has resulted in the agreement that as from 1 January 2002, all measurements should be made without the offset applied (Environment Canada, 2003). This decision was reached because it was thought that the factor was not appropriate for $PM_{2.5}$ measurements since it was originally derived using PM_{10} measurements.

16. Because the TEOM offset takes the form of a linear relationship in the form $y = mx + c$, it is possible to retrospectively remove it from all UK (and European) datasets for PM_{10} and $PM_{2.5}$.

Source apportionment calculations for PM and status of roadside monitoring sites

Table A7.1 Source apportionment calculations by CERC for three London sites for PM_{10} in (a) 1999 and (b) 2010 and $PM_{2.5}$ in (c)1999 and (d) 2010.

(a) 1999 PM_{10} concentration (gravimetric equivalent)		**Traffic**	**Stationary**	**Secondary background**	**Other**	**Total**
Marylebone Road	Concentration (μg m^{-3})	25.0	1.4	13.7	9.9	49.8
	Percentage (%)	50.1	2.7	27.4	19.8	—
Bloomsbury	Concentration (μg m^{-3})	3.9	1.4	13.7	9.9	28.9
	Percentage (%)	13.5	4.7	47.5	34.3	—
North Kensington	Concentration (μg m^{-3})	2.0	1.4	13.7	9.9	27.0
	Percentage (%)	7.3	5.3	50.7	36.7	—

(b) 2010 PM_{10} concentration (gravimetric equivalent)		**Traffic**	**Stationary**	**Secondary background**	**Other**	**Total**
Marylebone Road	Concentration (μg m^{-3})	7.7	1.0	9.1	9.9	27.7
	Percentage (%)	27.8	3.6	32.9	35.7	—
Bloomsbury	Concentration (μg m^{-3})	1.1	1.0	9.1	9.9	21.2
	Percentage (%)	5.4	4.9	43.0	46.8	—
North Kensington	Concentration (μg m^{-3})	0.7	1.1	9.1	9.9	20.8
	Percentage (%)	3.4	5.3	43.8	47.6	—

(c) 1999 $PM_{2.5}$ concentration (gravimetric equivalent)		Traffic	Stationary	Secondary background	Other	Total
Marylebone Road	Concentration (μg m^{-3})	21.45	0.96	11.47	2.97	36.85
	Percentage (%)	58.2	2.6	31.1	8.1	—
Bloomsbury	Concentration (μg m^{-3})	3.33	0.98	11.47	2.97	18.75
	Percentage (%)	17.8	5.2	61.2	15.8	—
North Kensington	Concentration (μg m^{-3})	1.66	1.01	11.47	2.97	17.11
	Percentage (%)	9.7	5.9	67.0	17.4	—

(d) 2010 $PM_{2.5}$ concentration (gravimetric equivalent) μg/m^3		Traffic	Stationary	Secondary background	Other	Total
Marylebone Road	Concentration (μg m^{-3})	6.58	0.67	7.65	2.97	17.87
	Percentage (%)	36.8	3.7	42.8	16.6	—
Bloomsbury	Concentration (μg m^{-3})	0.96	0.68	7.65	2.97	12.26
	Percentage (%)	7.8	5.5	62.4	24.2	—
North Kensington	Concentration (μg m^{-3})	0.57	0.75	7.65	2.97	11.94
	Percentage (%)	4.8	6.3	64.1	24.9	—

Note: The attribution to 'traffic' and 'stationary' is from modelling of emissions covered in the London inventory, including brakes and tyre wear as well as exhaust emissions in traffic, whereas 'other' includes additional possible traffic-related components such as resuspension of road dust within the 'residual' fraction.

Table A7.2 Status of PM$_{10}$ monitoring sites in 2002: TEOM concentrations * 1.3 are shown. (R, roadside; K, kerbside; Other, all other classifications. Percentages are shown in brackets.)

Country	Site type	Total number of sites	No. sites annual mean >40 $\mu g\ m^{-3}$	No. sites annual mean >20 $\mu g\ m^{-3}$	No. sites annual mean >23 $\mu g\ m^{-3}$ (London)	No. sites annual mean >18 $\mu g\ m^{-3}$ (Scotland)	No. sites >35 days >50 $\mu g\ m^{-3}$ daily mean	No. sites >7 days >50 $\mu g\ m^{-3}$ daily mean	No. sites >10 days >50 $\mu g\ m^{-3}$ daily mean (London)
Scotland	R and K	2	0	2 (100)	—	2 (100)	0	2 (100)	—
	Other	2	0	1 (50)	—	2 (100)	0	2 (100)	—
Wales	R and K	1	0	1 (100)	—	—	0	1 (100)	—
	Other	5	0	3 (60)	—	—	0	2 (40)	—
Northern Ireland	R and K	1	0	1 (100)	—	—	1 (100)	1 (100)	—
	Other	7	0	5 (71)	—	—	0	6 (86)	
London	R and K	33	3 (9)	33 (100)	33 (100)	—	13 (39)	30 (91)	33 (100)
	Other	28	0	28 (100)	26 (93)	—	4 (14)	22 (79)	28 (100)
Rest of England	R and K	30	2 (7)	29 (97)	—	—	4 (13)	22 (73)	—
	Other	66	1 (2)	50 (76)	—	—	4 (6)	27 (41)	—
UK	R and K	67	5 (7)	66 (99)	—	—	18 (27)	56 (84)	—
	Other	108	1 (1)	87 (81)	—	—	8 (7)	59 (55)	—

Table A7.3 Status of PM$_{10}$ monitoring sites in 2003: TEOM concentrations * 1.3 are shown. (R, roadside; K, kerbside; Other, all other classifications. Percentages are shown in brackets.)

Country	Site type	Total number of sites	No. sites annual mean >40 $\mu g\ m^{-3}$	No. sites annual mean >20 $\mu g\ m^{-3}$	No. sites annual mean >23 $\mu g\ m^{-3}$ (London)	No. sites annual mean >18 $\mu g\ m^{-3}$ (Scotland)	No. sites >35 days >50 $\mu g\ m^{-3}$ daily mean	No. sites >7 days >50 $\mu g\ m^{-3}$ daily mean	No. sites >10 days >50 $\mu g\ m^{-3}$ daily mean (London)
Scotland	R and K	3	0	2 (67)	—	2 (67)	1 (35)	3 (100)	—
	Other	4	0	3 (75)	—	4 (100)	0	4 (100)	—
Wales	R and K	1	0	0	—	—	1 (100)	1 (100)	—
	Other	6	0	1 (17)	—	—	3 (50)	5 (83)	—
Northern Ireland	R and K	2	0	2 (100)	—	—	1 (50)	2 (100)	—
	Other	9	1 (11)	8 (89)	—	—	3 (33)	8 (89)	—
London	R and K	32	8 (25)	32 (100)	32 (100)	—	23 (72)	32 (100)	32 (100)
	Other	28	0	28 (100)	28 (100)	—	7 (25)	28 (100)	28 (100)
Rest of England	R and K	38	5 (13)	38 (100)	—	—	19 (50)	37 (97)	—
	Other	64	1 (2)	60 (94)	—	—	7 (11)	59 (92)	—
UK	R and K	76	13 (17)	74 (97)	—	—	45 (59)	75 (99)	—
	Other	111	2 (2)	100 (90)	—	—	20 (18)	104 (94)	—

Table A7.4 Status of $PM_{2.5}$ monitoring sites in 2002 (gravimetric measurements only).

Country	Site type	Total number of sites	Annual mean >20 $\mu g\ m^{-3}$	Number of sites Annual mean >16 $\mu g\ m^{-3}$	Annual mean >12 $\mu g\ m^{-3}$	Percentage of sites		
						Annual mean >20 $\mu g\ m^{-3}$	Annual mean >16 $\mu g\ m^{-3}$	Annual mean >12 $\mu g\ m^{-3}$
Wales		—	—	—	—	—	—	—
Northern Ireland	Other	1	0	1	1	0%	100%	100%
London	R and K	1	1	1	1	100%	100%	100%
	Other	1	0	1	1	0%	100%	100%
Rest of England	R and K	1	0	0	1	0%	0%	100%
	Other	2	0	1	2	0%	50%	100%
UK	R and K	2	1	1	2	50%	50%	100%
	Other	4	0	3	4	0%	75%	100%

Table A7.5 Status of $PM_{2.5}$ monitoring sites in 2003 (gravimetric measurements only).

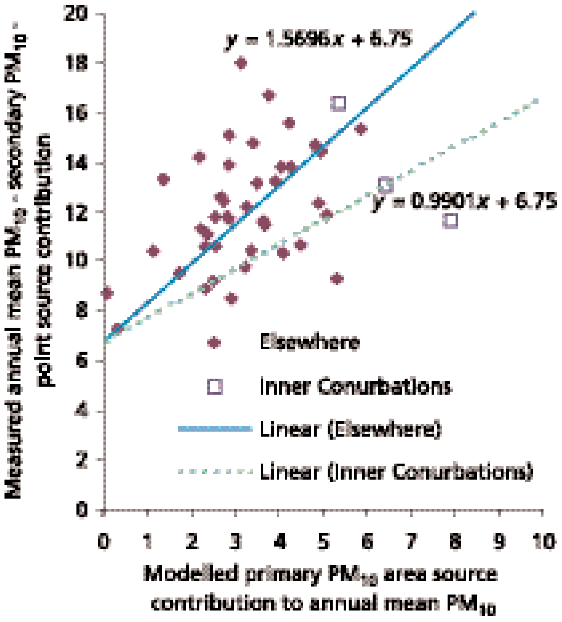
Country	Site type	Total number of sites	Annual mean >20 $\mu g\ m^{-3}$	Number of sites Annual mean >16 $\mu g\ m^{-3}$	Annual mean >12 $\mu g\ m^{-3}$	Percentage of sites		
						Annual mean >20 $\mu g\ m^{-3}$	Annual mean >16 $\mu g\ m^{-3}$	Annual mean >12 $\mu g\ m^{-3}$
Wales		1	0	1	1	0%	100%	100%
Northern Ireland	Other	1	0	1	1	0%	100%	100%
London	R and K	1	1	1	1	100%	100%	100%
	Other	1	1	1	1	100%	100%	100%
Rest of England	R and K	1	0	0	1	0%	0%	100%
	Other	3	1	3	3	33%	100%	100%
UK	R and K	3	1	2	3	33%	67%	100%
	Other	5	2	5	5	40%	100%	100%

Sensitivity analysis of the Netcen and ERG models

1. A sensitivity analysis of the road lengths and areas of London that are predicted to exceed an annual mean PM_{10} concentration of 20 μg m^{-3} with perturbations of ±2 μg m^{-3} indicates some large variations in both the Netcen and ERG models, in some cases varying from a small fraction of exceedence to almost complete exceedence across London. This is explained by the statistical distributions of the estimated road lengths against the critical ranges of concentration and daily exceedence.

2. The distribution of roadside concentrations across the UK network in Figure A8.1 tends to be broader than that for London due to more variation in both background concentrations and road contributions. The annual limit of 20 μg m^{-3} lies close to the centre of the distribution and small variations of ±2 μg m^{-3} make a large difference to the proportion of road links to the right or left in the distribution.

Figure A8.1 Frequency distribution of road links in UK against annual average concentration: Netcen model for 2010.



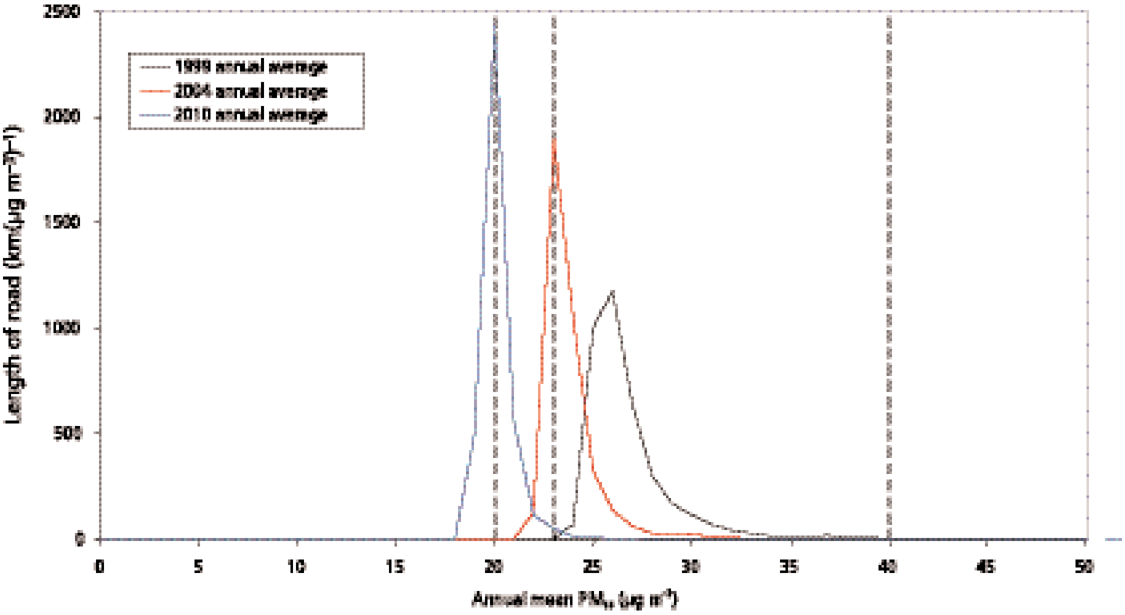
3. Comparing Figures A8.2(a) and A8.2(b), and bearing in mind the differences in years, the picture is very consistent, showing a similar improvement with time. Although the CERC distribution may appear to give slightly higher mean values than the ERG model, it covers less of the more minor roads in the network. Again the results illustrate the problem of a large change in exceedence for a small percentage change in the total concentration, because the distribution of roadside increments is superimposed on a large and fairly uniform urban background.

4. With respect to the areas of exceedence, the small perturbation of ±2 µg m^{-3} on rather flat contours of urban background concentration can lead to very little or almost all of London exceeded. In practice there is likely to be a considerable spatial variability in the coarse component, maybe enhancing the importance of roads and making the contours of concentration far more hilly, with peaks above and dips below the annual limit. Until better information is available it is not possible to resolve this, but it is an important limitation in assessing exceedence.

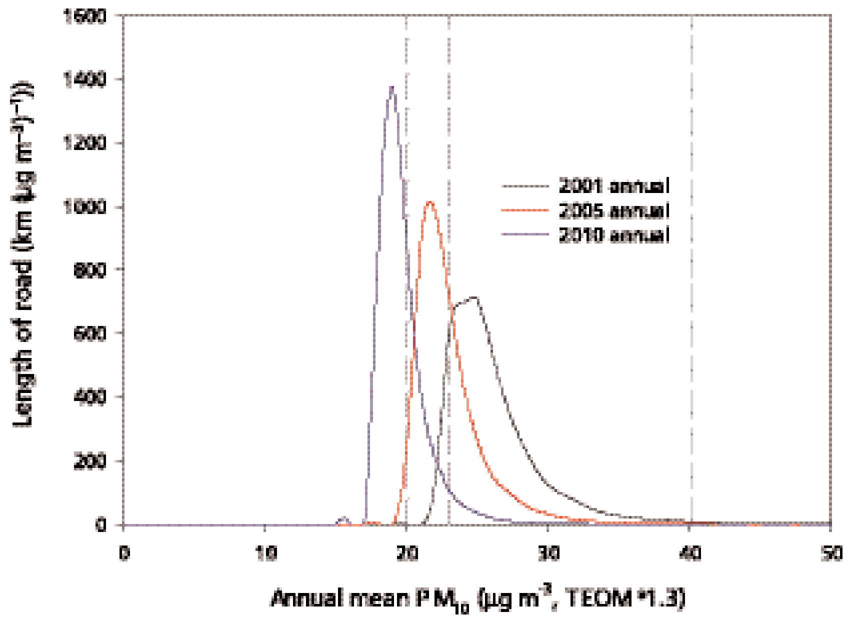
5. Figure A5.2(c) relates to the frequency of occurrence of days above 50 µg m^{-3} and compliance with daily limit values. The problems of accurately representing the frequency of extreme episodic events are discussed in Chapter 9.

Figure A8.2 Statistical distribution of results for London from ERG and CERC. (a) CERC results using ADMS, (b) frequency distribution of annual mean PM_{10} concentration for all roads in the LAEI using the ERG model and (c) frequency distribution of daily mean PM_{10} concentration >50 µg m^{-3} for all roads in the LAEI using the ERG model.

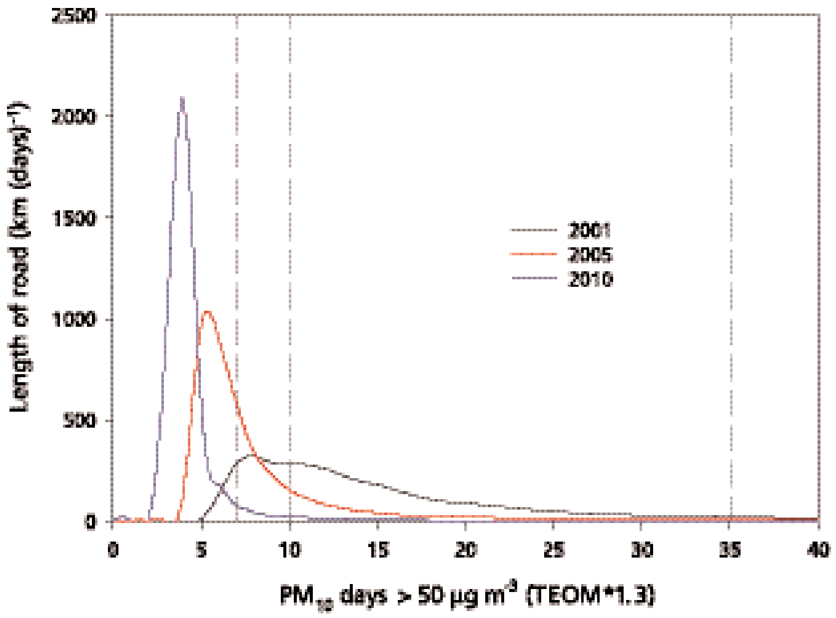
(a)



(b)


Length of road (km (µg m⁻³)⁻¹))
1600
1400
1200
1000
800
600
400
200
0
0
10
20
30
40
50
Annual mean PM10 (µg m⁻³, TEOM*1.3)
2001 annual
2005 annual
2010 annual

(c)


Length of road (km (days)⁻¹)
2500
2000
1500
1000
500
0
0
5
10
15
20
25
30
35
40
PM10 days > 50 µg m⁻³ (TEOM*1.3)
2001
2005
2010

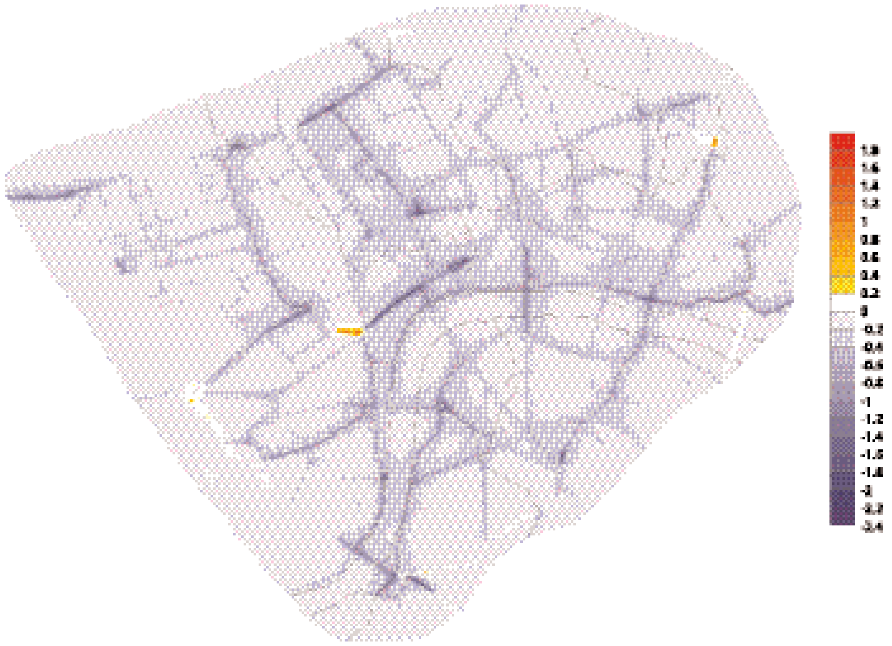
Annex 9

The estimated effect of the London Congestion Charging Scheme on PM_{10} concentrations

1. Chapter 4 summarised the estimated emissions changes of the London Congestion Charging Scheme (CCS), which started in February 2003. Overall it was estimated that road transport PM_{10} emissions within the charging zone decreased by 11.9%, based on 2002 vehicle age profiles through the introduction of the CCS. The 11.9% reduction only includes the reductions brought about by the CCS and not the effects as the result of the ongoing penetration of newer vehicle technologies that have reduced emissions further throughout 2003.

2. It was also stressed in Chapter 4 that the benefits brought by the reductions in road traffic only accounted for exhaust emissions of PM_{10}. Based on recent estimates of non-exhaust PM_{10} emissions considered in Chapters 4 and 6, it is likely that the estimated reductions in total PM_{10} emissions have been underestimated. Two principal non-exhaust PM_{10} sources (tyre and brake wear and resuspension) might be expected to reduce because the CCS has brought about a reduction in the vehicle flows of both light and heavy vehicles. Further work is therefore required to estimate the additional benefits of decreases in the non-exhaust PM_{10} emissions from vehicles. Figure A6.1 shows the predicted change in annual mean PM_{10} concentration brought about by the introduction of the CCS (Barratt *et al*., 2004). Typically, the change in annual mean PM_{10} is small across the zone but greater reductions are predicted close to roadside locations where PM_{10} concentrations are highest and there is most risk of exceedences of air quality limits.

Figure A9.1 Estimated change in annual mean PM_{10} concentration ($\mu g\ m^{-3}$) brought about by the CCS based on 2002 vehicle stock assuming 2001 meteorology using the ERG model.



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Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
1999	Aberdeen	URBAN BACKG	AURN	NJ944073	394416	807408					GB0729A	Teom	PM10	15	14	30	59	20-Sep-99	3	27		Scotland
2000	Aberdeen	URBAN BACKG	AURN	NJ944073	394416	807408					GB0729A	Teom	PM10	15	24	30	51	24-Sep-00	3	94		Scotland
2001	Aberdeen	URBAN BACKG	AURN	NJ944073	394416	807408					GB0729A	Teom	PM10	12	20	28	46	31-Jan-01	2	97		Scotland
2002	Aberdeen	URBAN BACKG	AURN	NJ944073	394416	807408					GB0729A	Teom	PM10	14	25	34	49	11-Oct-02	3	72		Scotland
2003	Aberdeen	URBAN BACKG	AURN	NJ944073	394416	807408					GB0729A	Teom	PM10	17	29	48	69	27-Feb-03	14	98		Scotland
1992	Belfast Centre	URBAN CENTR	AURN	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Teom	PM10	27	43	95	248	20-Dec-92	44	79		N Ireland
1993	Belfast Centre	URBAN CENTR	AURN	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Teom	PM10	32	54	79	120	31-Jan-93	86	96		N Ireland
1994	Belfast Centre	URBAN CENTR	AURN	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Teom	PM10	26	40	66	191	21-Dec-94	38	95		N Ireland
1995	Belfast Centre	URBAN CENTR	AURN	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Teom	PM10	24	38	59	146	27-Dec-95	35	95		N Ireland
1996	Belfast Centre	URBAN CENTR	AURN	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Teom	PM10	24	41	62	112	3-Feb-96	44	95		N Ireland
1997	Belfast Centre	URBAN CENTR	AURN	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Teom	PM10	25	40	60	84	8-Nov-97	41	96		N Ireland
1998	Belfast Centre	URBAN CENTR	AURN	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Teom	PM10	21	32	50	67	31-Jan-98	20	94		N Ireland
1999	Belfast Centre	URBAN CENTR	AURN	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Teom	PM10	20	30	46	65	10-Jan-99	15	97		N Ireland
2000	Belfast Centre	URBAN CENTR	AURN	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Teom	PM10	20	29	39	53	25-Jan-00	8	81		N Ireland
2001	Belfast Centre	URBAN CENTR	AURN	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Teom	PM10	19	29	44	83	12-Dec-01	15	81		N Ireland
2002	Belfast Centre	URBAN CENTR	AURN	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Teom	PM10	18	28	40	64	17-Dec-02	8	98		N Ireland
2003	Belfast Centre	URBAN CENTR	AURN	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Teom	PM10	19	36	47	70	4-Sep-03	26	97		N Ireland
2001	Belfast Centre	URBAN CENTR	defra research	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Partisol 20	PM10	35	55	94	136	30-Jun-01	53	77		N Ireland
2002	Belfast Centre	URBAN CENTR	defra research	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Partisol 20	PM10	25	41	66	96	29-Mar-02	12	83		N Ireland
2003	Belfast Centre	URBAN CENTR	defra research	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Partisol 20	PM10	28	53	80	108	28-Mar-03	39	91		N Ireland
2001	Belfast Centre	URBAN CENTR	defra research	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Partisol 20	PM2.5	14	22	45	77	12-Dec-01		79		N Ireland
2002	Belfast Centre	URBAN CENTR	defra research	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Partisol 20	PM2.5	16	27	49	76	29-Mar-02		88		N Ireland
2003	Belfast Centre	URBAN CENTR	defra research	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	Partisol 20	PM2.5	17	35	60	87	28-Mar-03		92		N Ireland
2001	Belfast Centre	URBAN CENTR	defra research	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	KFG	PM10	22	37	60	88	11-Dec-01	15	67		N Ireland
2002	Belfast Centre	URBAN CENTR	defra research	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	KFG	PM10	22	39	72	89	18-Dec-02	15	86		N Ireland
2003	Belfast Centre	URBAN CENTR	defra research	J339744	UK: 14630	UK: 529900 (IRE: 374432)					GB0567A	KFG	PM10	24	50	80	96	18-Mar-03	37	92		N Ireland
1996	Belfast Clara St	SUBURBAN	AURN	J360734	148400	528800					GB0696A	Bam	PM10	24	*	*	91	27-Dec-96	*	72		N Ireland
1997	Belfast Clara St	SUBURBAN	AURN	J360734	148400	528800					GB0696A	Bam	PM10	31	**	**	82	21-Jan-97	**	25		N Ireland
1998	Belfast Clara St	SUBURBAN	AURN	J360734	148400	528800					GB0696A	Bam	PM10	27	35	56	71	23-Sep-98	10	53		N Ireland
1999	Belfast Clara St	SUBURBAN	AURN	J360734	148400	528800					GB0696A	Bam	PM10	22	38	54	71	1-Apr-99	13	95		N Ireland
2000	Belfast Clara St	SUBURBAN	AURN	J360734	148400	528800					GB0696A	Bam	PM10	16	30	42	69	29-Dec-00	4	94		N Ireland
2001	Belfast Clara St	SUBURBAN	AURN	J360734	148400	528800					GB0696A	Bam	PM10	19	36	58	128	19-Jan-01	14	92		N Ireland
2002	Belfast Clara St	SUBURBAN	AURN	J360734	148400	528800					GB0696A	Bam	PM10	17	33	50	99	29-Mar-02	8	94		N Ireland
2003	Belfast Clara St	SUBURBAN	AURN	J360734	148400	528800					GB0696A	Bam	PM10	22	49	82	121	28-Mar-03	34	95		N Ireland
1992	Birmingham Centre	URBAN CENTR	AURN	SP064868	406342	286862					GB0569A	Teom	PM10	26	41	68	131	23-Dec-92	46	75		England
1993	Birmingham Centre	URBAN CENTR	AURN	SP064868	406342	286862					GB0569A	Teom	PM10	26	42	64	102	30-Apr-93	43	84		England
1994	Birmingham Centre	URBAN CENTR	AURN	SP064868	406342	286862					GB0569A	Teom	PM10	23	39	55	113	22-Dec-94	39	93		England
1995	Birmingham Centre	URBAN CENTR	AURN	SP064868	406342	286862					GB0569A	Teom	PM10	23	38	58	85	13-Mar-95	33	98		England
1996	Birmingham Centre	URBAN CENTR	AURN	SP064868	406342	286862					GB0569A	Teom	PM10	25	41	66	102	19-Mar-96	41	95		England
1997	Birmingham Centre	URBAN CENTR	AURN	SP064868	406342	286862					GB0569A	Teom	PM10	22	35	44	82	31-Oct-97	22	95		England
1998	Birmingham Centre	URBAN CENTR	AURN	SP064868	406342	286862					GB0569A	Teom	PM10	19	31	50	65	30-Jan-98	13	95		England
1999	Birmingham Centre	URBAN CENTR	AURN	SP064868	406342	286862					GB0569A	Teom	PM10	18	30	39	49	7-Aug-99	10	95		England
2000	Birmingham Centre	URBAN CENTR	AURN	SP064868	406342	286862					GB0569A	Teom	PM10	17	28	39	55	10-May-00	10	98		England
2001	Birmingham Centre	URBAN CENTR	AURN	SP064868	406342	286862					GB0569A	Teom	PM10	17	27	39	54	18-Jan-01	10	98		England
2002	Birmingham Centre	URBAN CENTR	AURN	SP064868	406342	286862					GB0569A	Teom	PM10	17	26	36	54	5-Apr-02	6	97		England
2003	Birmingham Centre	URBAN CENTR	AURN	SP064868	406342	286862					GB0569A	Teom	PM10	19	30	48	62	1-Oct-03	15	86		England
2000	Birmingham Centre	URBAN CENTR	defra research	SP064868	406342	286862					GB0569A	Partisol 20	PM10	21	18	31	44	21 Oct 2000	0	19		England
2001	Birmingham Centre	URBAN CENTR	defra research	SP064868	406342	286862					GB0569A	Partisol 20	PM10	24	42	67	97	18 Jan 2001	21	98		England
2002	Birmingham Centre	URBAN CENTR	defra research	SP064868	406342	286862					GB0569A	Partisol 20	PM10	24	35	59	85	05 Apr 2002	16	64		England
2003	Birmingham Centre	URBAN CENTR	defra research	SP064868	406342	286862					GB0569A	Partisol 20	PM10	31	59	83	109	29 Mar 2003	50	91		England
2000	Birmingham Centre	URBAN CENTR	defra research	SP064868	406342	286862					GB0569A	Partisol 20	PM2.5	14	12	26	49	23 Dec 2000		20		England
2001	Birmingham Centre	URBAN CENTR	defra research	SP064868	406342	286862					GB0569A	Partisol 20	PM2.5	18	35	58	87	19 Jan 2001		91		England
2002	Birmingham Centre	URBAN CENTR	defra research	SP064868	406342	286862					GB0569A	Partisol 20	PM2.5	16	27	48	63	29 Mar 2002		81		England
2003	Birmingham Centre	URBAN CENTR	defra research	SP064868	406342	286862					GB0569A	Partisol 20	PM2.5	20	40	65	94	29 Mar 2003		91		England
1994	Birmingham East	URBAN BACKG	AURN	SP116889	411536	288870					GB0595A	Teom	PM10	21	34	51	106	22-Dec-94	25	97		England
1995	Birmingham East	URBAN BACKG	AURN	SP116889	411536	288870					GB0595A	Teom	PM10	22	36	57	113	4-Nov-95	24	95		England
1996	Birmingham East	URBAN BACKG	AURN	SP116889	411536	288870					GB0595A	Teom	PM10	23	39	63	89	19-Mar-96	36	95		England
1997	Birmingham East	URBAN BACKG	AURN	SP116889	411536	288870					GB0595A	Teom	PM10	20	33	48	88	31-Oct-97	21	96		England
1998	Birmingham East	URBAN BACKG	AURN	SP116889	411536	288870					GB0595A	Teom	PM10	15	25	37	57	30-Jan-98	7	98		England
1999	Birmingham East	URBAN BACKG	AURN	SP116889	411536	288870					GB0595A	Teom	PM10	17	27	37	48	13-Oct-99	7	98		England
2000	Birmingham East	URBAN BACKG	AURN	SP116889	411536	288870					GB0595A	Teom	PM10	16	26	36	54	10-May-00	4	97		England

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2001	Birmingham East	URBAN BACKG	AURN	SP116889	411536	288870					GB0595A	Teom	PM10	17	26	39	52	15-Nov-01	9	98		England
2002	Birmingham East	URBAN BACKG	AURN	SP116889	411536	288870					GB0595A	Teom	PM10	16	24	36	52	18-Sep-02	4	97		England
2003	Birmingham East	URBAN BACKG	AURN	SP116889	411536	288870					GB0595A	Teom	PM10	17	17	27	36	5-Nov-03	0	19		England
2000	Blackpool	URBAN BACKG	AURN	SD324333	332289	433173					GB0727A	Teom	PM10	17	21	25	36	29-Dec-00	0	39		England
2001	Blackpool	URBAN BACKG	AURN	SD324333	332289	433173					GB0727A	Teom	PM10	18	27	40	55	11-Dec-01	8	97		England
2002	Blackpool	URBAN BACKG	AURN	SD324333	332289	433173					GB0727A	Teom	PM10	18	26	36	48	12-Sep-02	3	98		England
2003	Blackpool	URBAN BACKG	AURN	SD324333	332289	433173					GB0727A	Teom	PM10	20	33	44	52	20-Apr-03	15	96		England
1997	Bolton	URBAN BACKG	AURN	SD710085	371000	408500					GB0654A	Teom	PM10	20	32	44	64	11-Mar-97	17	83		England
1998	Bolton	URBAN BACKG	AURN	SD710085	371000	408500					GB0654A	Teom	PM10	17	27	39	56	14-May-98	8	96		England
1999	Bolton	URBAN BACKG	AURN	SD710085	371000	408500					GB0654A	Teom	PM10	16	25	32	43	11-Feb-99	1	94		England
2000	Bolton	URBAN BACKG	AURN	SD710085	371000	408500					GB0654A	Teom	PM10	16	24	32	49	4-Nov-00	3	97		England
2001	Bolton	URBAN BACKG	AURN	SD710085	371000	408500					GB0654A	Teom	PM10	16	25	36	60	11-Dec-01	5	98		England
2002	Bolton	URBAN BACKG	AURN	SD710085	371000	408500					GB0654A	Teom	PM10	16	25	33	56	12-Sep-02	4	99		England
2003	Bolton	URBAN BACKG	AURN	SD710085	371000	408500					GB0654A	Teom	PM10	18	30	42	55	20-Apr-03	15	98		England
2001	Bournemouth	URBAN BACKG	AURN	SZ123933	412300	93344					GB0741A	Partisol	PM10	22	29	39	84	3-Nov-01	1	43		England
2002	Bournemouth	URBAN BACKG	AURN	SZ123933	412300	93344					GB0741A	Partisol	PM10	25	39	55	97	5-Apr-02	10	95		England
2003	Bournemouth	URBAN BACKG	AURN	SZ123933	412300	93344					GB0741A	Partisol	PM10	27	46	73	135	29-Mar-03	28	93		England
1998	Bradford Centre	URBAN CENTR	AURN	SE166331	416663	433053					GB0689A	Teom	PM10	24	39	56	80	7-Dec-98	37	97		England
1999	Bradford Centre	URBAN CENTR	AURN	SE166331	416663	433053					GB0689A	Teom	PM10	20	31	42	69	2-Apr-99	13	98		England
2000	Bradford Centre	URBAN CENTR	AURN	SE166331	416663	433053					GB0689A	Teom	PM10	17	27	37	47	22-Mar-00	5	97		England
2001	Bradford Centre	URBAN CENTR	AURN	SE166331	416663	433053					GB0689A	Teom	PM10	21	34	43	72	14-Feb-01	16	94		England
2002	Bradford Centre	URBAN CENTR	AURN	SE166331	416663	433053					GB0689A	Teom	PM10	18	28	39	45	18-Dec-02	8	98		England
2003	Bradford Centre	URBAN CENTR	AURN	SE166331	416663	433053					GB0689A	Teom	PM10	21	35	54	63	28-Mar-03	30	97		England
2003	Brighton Roadside PM1	ROADSIDE	AURN	TQ313043	531302	104331	A23			10,000/day	N/A	Partisol	PM10	36	50	88	151	29-Mar-03	34	75		England
1993	Bristol Centre	URBAN CENTR	AURN	ST594732	359404	173344					GB0585A	Teom	PM10	27	47	64	81	3-Nov-93	59	87		England
1994	Bristol Centre	URBAN CENTR	AURN	ST594732	359404	173344					GB0585A	Teom	PM10	24	40	59	83	20-Oct-94	41	95		England
1995	Bristol Centre	URBAN CENTR	AURN	ST594732	359404	173344					GB0585A	Teom	PM10	24	41	57	64	4-Nov-95	40	97		England
1996	Bristol Centre	URBAN CENTR	AURN	ST594732	359404	173344					GB0585A	Teom	PM10	25	42	61	94	16-Mar-96	43	97		England
1997	Bristol Centre	URBAN CENTR	AURN	ST594732	359404	173344					GB0585A	Teom	PM10	24	38	56	71	14-Jan-97	32	96		England
1998	Bristol Centre	URBAN CENTR	AURN	ST594732	359404	173344					GB0585A	Teom	PM10	21	34	49	59	14-May-98	23	92		England
1999	Bristol Centre	URBAN CENTR	AURN	ST594732	359404	173344					GB0585A	Teom	PM10	20	32	43	51	3-Sep-99	11	95		England
2000	Bristol Centre	URBAN CENTR	AURN	ST594732	359404	173344					GB0585A	Teom	PM10	20	32	43	72	26-Jan-00	10	93		England
2001	Bristol Centre	URBAN CENTR	AURN	ST594732	359404	173344					GB0585A	Teom	PM10	18	31	46	63	20-Jan-01	16	95		England
2002	Bristol Centre	URBAN CENTR	AURN	ST594732	359404	173344					GB0585A	Teom	PM10	20	31	39	57	5-Apr-02	7	96		England
2003	Bristol Centre	URBAN CENTR	AURN	ST594732	359404	173344					GB0585A	Teom	PM10	22	40	50	61	19-Dec-03	37	93		England
1997	Bury Roadside	ROADSIDE	AURN	SD809048	380922	404772	M62	30m	3	200,000/da	GB0652A	Teom	PM10	30	47	62	74	11-Mar-97	66	86		England
1998	Bury Roadside	ROADSIDE	AURN	SD809048	380922	404772	M62	30m	3	200,000/da	GB0652A	Teom	PM10	26	41	54	73	28-Jan-98	47	95		England
1999	Bury Roadside	ROADSIDE	AURN	SD809048	380922	404772	M62	30m	3	200,000/da	GB0652A	Teom	PM10	24	36	49	71	11-Feb-99	31	96		England
2000	Bury Roadside	ROADSIDE	AURN	SD809048	380922	404772	M62	30m	3	200,000/da	GB0652A	Teom	PM10	23	33	43	57	25-Jan-00	10	92		England
2001	Bury Roadside	ROADSIDE	AURN	SD809048	380922	404772	M62	30m	3	200,000/da	GB0652A	Teom	PM10	25	39	53	78	11-Dec-01	36	94		England
2002	Bury Roadside	ROADSIDE	AURN	SD809048	380922	404772	M62	30m	3	200,000/da	GB0652A	Teom	PM10	24	35	45	61	12-Sep-02	22	96		England
2003	Bury Roadside	ROADSIDE	AURN	SD809048	380922	404772	M62	30m	3	200,000/da	GB0652A	Teom	PM10	27	43	58	75	30-Nov-03	54	97		England
1996	Camden Kerbside	KERBSIDE	AURN	TQ267843	526700	184300	A41	1m	3	>50,000/da	GB0636A	Teom	PM10	32	41	59	74	14-Nov-96	45	49		London
1997	Camden Kerbside	KERBSIDE	AURN	TQ267843	526700	184300	A41	1m	3	>50,000/da	GB0636A	Teom	PM10	32	48	74	98	15-Jan-97	88	99		London
1998	Camden Kerbside	KERBSIDE	AURN	TQ267843	526700	184300	A41	1m	3	>50,000/da	GB0636A	Teom	PM10	25	36	45	51	25-Sep-98	22	82		London
1999	Camden Kerbside	KERBSIDE	AURN	TQ267843	526700	184300	A41	1m	3	>50,000/da	GB0636A	Teom	PM10	26	38	51	61	15-Mar-99	33	94		London
2000	Camden Kerbside	KERBSIDE	AURN	TQ267843	526700	184300	A41	1m	3	>50,000/da	GB0636A	Teom	PM10	26	38	47	63	23-Mar-00	29	98		London
2001	Camden Kerbside	KERBSIDE	AURN	TQ267843	526700	184300	A41	1m	3	>50,000/da	GB0636A	Teom	PM10	26	37	50	74	4-Nov-01	28	99		London
2002	Camden Kerbside	KERBSIDE	AURN	TQ267843	526700	184300	A41	1m	3	>50,000/da	GB0636A	Teom	PM10	24	35	44	49	10-May-02	15	99		London
2003	Camden Kerbside	KERBSIDE	AURN	TQ267843	526700	184300	A41	1m	3	>50,000/da	GB0636A	Teom	PM10	27	42	56	68	8-Aug-03	47	99		London
2001	Canterbury	URBAN BACKG	AURN	TR162573	616200	157342					GB0737A	Teom	PM10	18	28	38	56	17-Jan-01	7	98		England
2002	Canterbury	URBAN BACKG	AURN	TR162573	616200	157342					GB0737A	Teom	PM10	19	27	39	55	10-May-02	9	99		England
2003	Canterbury	URBAN BACKG	AURN	TR162573	616200	157342					GB0737A	Teom	PM10	20	32	46	55	11-Aug-03	17	99		England
1992	Cardiff Centre	URBAN CENTR	AURN	ST184765	318412	176530					GB0580A	Teom	PM10	26	32	52	85	31-Dec-92	18	48		Wales
1993	Cardiff Centre	URBAN CENTR	AURN	ST184765	318412	176530					GB0580A	Teom	PM10	31	42	61	89	30-Apr-93	47	56		Wales
1994	Cardiff Centre	URBAN CENTR	AURN	ST184765	318412	176530					GB0580A	Teom	PM10	34	56	76	96	18-Aug-94	105	96		Wales
1995	Cardiff Centre	URBAN CENTR	AURN	ST184765	318412	176530					GB0580A	Teom	PM10	25	40	54	82	15-Jun-95	39	97		Wales
1996	Cardiff Centre	URBAN CENTR	AURN	ST184765	318412	176530					GB0580A	Teom	PM10	25	39	61	91	16-Mar-96	36	95		Wales
1997	Cardiff Centre	URBAN CENTR	AURN	ST184765	318412	176530					GB0580A	Teom	PM10	26	39	55	70	27-Jan-97	40	96		Wales
1998	Cardiff Centre	URBAN CENTR	AURN	ST184765	318412	176530					GB0580A	Teom	PM10	22	33	49	66	13-May-98	19	98		Wales
1999	Cardiff Centre	URBAN CENTR	AURN	ST184765	318412	176530					GB0580A	Teom	PM10	21	31	41	101	26-Aug-99	13	98		Wales

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2000	Cardiff Centre	URBAN CENTR	AURN	ST184765	318412	176530					GB0580A	Teom	PM10	19	30	43	58	9-May-00	11	98		Wales
2001	Cardiff Centre	URBAN CENTR	AURN	ST184765	318412	176530					GB0580A	Teom	PM10	20	29	42	57	29-Mar-01	12	98		Wales
2002	Cardiff Centre	URBAN CENTR	AURN	ST184765	318412	176530					GB0580A	Teom	PM10	21	30	43	55	23-Jan-02	14	97		Wales
2003	Cardiff Centre	URBAN CENTR	AURN	ST184765	318412	176530					GB0580A	Teom	PM10	27	43	55	125	21-Jan-03	53	90		Wales
2001	Coventry Memorial Park	URBAN CENTR	AURN	SP326796	432600	279600					GB0739A	Teom	PM10	12	18	25	32	26-Jun-01	0	81		England
2002	Coventry Memorial Park	URBAN CENTR	AURN	SP326796	432600	279600					GB0739A	Teom	PM10	16	25	37	44	5-Apr-02	6	98		England
2003	Coventry Memorial Park	URBAN CENTR	AURN	SP326796	432600	279600					GB0739A	Teom	PM10	14	24	38	54	20-Apr-03	5	82		England
2001	Cwmbran	URBAN BACKG	AURN	ST305955	330510	195436					GB0744A	Teom	PM10	14	18	24	30	19-Nov-01	0	44		Wales
2002	Cwmbran	URBAN BACKG	AURN	ST305955	330510	195436					GB0744A	Teom	PM10	14	21	28	41	5-Apr-02	1	97		Wales
2003	Cwmbran	URBAN BACKG	AURN	ST305955	330510	195436					GB0744A	Teom	PM10	16	28	36	48	20-Apr-03	4	99		Wales
1997	Derry	URBAN BACKG	AURN	C429166	UK: 60200	UK: 5810 (IRE: 417956)					GB0673A	Teom	PM10	21	29	47	70	7-Nov-97	12	60		N Ireland
1998	Derry	URBAN BACKG	AURN	C429166	UK: 60200	UK: 5810 (IRE: 417956)					GB0673A	Teom	PM10	20	33	48	120	30-Jan-98	18	96		N Ireland
1999	Derry	URBAN BACKG	AURN	C429166	UK: 60200	UK: 5810 (IRE: 417956)					GB0673A	Teom	PM10	20	30	42	86	10-Jan-99	11	96		N Ireland
2000	Derry	URBAN BACKG	AURN	C429166	UK: 60200	UK: 5810 (IRE: 417956)					GB0673A	Teom	PM10	15	23	31	64	29-Dec-00	5	96		N Ireland
2001	Derry	URBAN BACKG	AURN	C429166	UK: 60200	UK: 5810 (IRE: 417956)					GB0673A	Teom	PM10	18	28	48	100	18-Jan-01	14	97		N Ireland
2002	Derry	URBAN BACKG	AURN	C429166	UK: 60200	UK: 5810 (IRE: 417956)					GB0673A	Teom	PM10	17	26	46	61	12-Sep-02	9	96		N Ireland
2003	Derry	URBAN BACKG	AURN	C429166	UK: 60200	UK: 5810 (IRE: 417956)					GB0673A	Teom	PM10	18	31	46	69	18-Mar-03	20	97		N Ireland
2001	Dumfries	ROADSIDE	AURN	NX970763	297012	576278	Buccleuch	5m			GB0740A	Partisol	PM10	18	21	30	48	12-Dec-01	0	33		Scotland
2002	Dumfries	ROADSIDE	AURN	NX970763	297012	576278	Buccleuch	5m			GB0740A	Partisol	PM10	21	33	60	82	30-Mar-02	18	94		Scotland
2003	Dumfries	ROADSIDE	AURN	NX970763	297012	576278	Buccleuch	5m			GB0740A	Partisol	PM10	23	45	68	100	26-Feb-03	22	93		Scotland
1992	Edinburgh Centre	URBAN CENTR	AURN	NT254738	325428	673862					GB0581A	Teom	PM10	23	19	35	72	22-Dec-92	6	17		Scotland
1993	Edinburgh Centre	URBAN CENTR	AURN	NT254738	325428	673862					GB0581A	Teom	PM10	23	33	43	66	19-Nov-93	14	82		Scotland
1994	Edinburgh Centre	URBAN CENTR	AURN	NT254738	325428	673862					GB0581A	Teom	PM10	20	28	40	62	10-Oct-94	9	89		Scotland
1995	Edinburgh Centre	URBAN CENTR	AURN	NT254738	325428	673862					GB0581A	Teom	PM10	20	31	52	67	13-Nov-95	19	92		Scotland
1996	Edinburgh Centre	URBAN CENTR	AURN	NT254738	325428	673862					GB0581A	Teom	PM10	19	30	47	65	15-Mar-96	13	97		Scotland
1997	Edinburgh Centre	URBAN CENTR	AURN	NT254738	325428	673862					GB0581A	Teom	PM10	18	28	41	51	12-Aug-97	8	94		Scotland
1998	Edinburgh Centre	URBAN CENTR	AURN	NT254738	325428	673862					GB0581A	Teom	PM10	15	23	32	40	27-Sep-98	1	98		Scotland
1999	Edinburgh Centre	URBAN CENTR	AURN	NT254738	325428	673862					GB0581A	Teom	PM10	15	21	28	62	2-Apr-99	3	92		Scotland
2000	Edinburgh Centre	URBAN CENTR	AURN	NT254738	325428	673862					GB0581A	Teom	PM10	18	26	36	71	14-Mar-00	5	96		Scotland
2001	Edinburgh Centre	URBAN CENTR	AURN	NT254738	325428	673862					GB0581A	Teom	PM10	20	28	43	79	12-Dec-01	10	97		Scotland
2002	Edinburgh Centre	URBAN CENTR	AURN	NT254738	325428	673862					GB0581A	Teom	PM10	21	29	50	62	4-Apr-02	14	82		Scotland
2003	Edinburgh Centre	URBAN CENTR	AURN	NT254738	325428	673862					GB0581A	Teom	PM10	19	24	36	63	21-Apr-03	5	44		Scotland
1996	Glasgow Centre	URBAN CENTR	AURN	NS589651	258902	665028					GB0641A	Teom	PM10	22	27	39	68	5-Dec-96	7	42		Scotland
1997	Glasgow Centre	URBAN CENTR	AURN	NS589651	258902	665028					GB0641A	Teom	PM10	21	33	49	85	28-Sep-97	18	97		Scotland
1998	Glasgow Centre	URBAN CENTR	AURN	NS589651	258902	665028					GB0641A	Teom	PM10	20	31	41	50	17-Nov-98	11	98		Scotland
1999	Glasgow Centre	URBAN CENTR	AURN	NS589651	258902	665028					GB0641A	Teom	PM10	18	27	41	67	2-Apr-99	9	98		Scotland
2000	Glasgow Centre	URBAN CENTR	AURN	NS589651	258902	665028					GB0641A	Teom	PM10	22	36	57	85	28-Mar-00	25	97		Scotland
2001	Glasgow Centre	URBAN CENTR	AURN	NS589651	258902	665028					GB0641A	Teom	PM10	17	28	43	95	12-Dec-01	11	99		Scotland
2002	Glasgow Centre	URBAN CENTR	AURN	NS589651	258902	665028					GB0641A	Teom	PM10	15	24	40	88	19-Dec-02	8	98		Scotland
2003	Glasgow Centre	URBAN CENTR	AURN	NS589651	258902	665028					GB0641A	Teom	PM10	16	28	45	52	16-Apr-03	13	96		Scotland
2000	Glasgow Centre	URBAN CENTR	defra research	NS589651	258902	665028					GB0641A	Partisol 20	PM10	20	13	26	52	06 Nov 2000	1	12		Scotland
2001	Glasgow Centre	URBAN CENTR	defra research	NS589651	258902	665028					GB0641A	Partisol 20	PM10	22	31	55	118	11 Dec 2001	8	61		Scotland
2002	Glasgow Centre	URBAN CENTR	defra research	NS589651	258902	665028					GB0641A	Partisol 20	PM10	23	28	63	143	20 Dec 2002	11	56		Scotland
2003	Glasgow Centre	URBAN CENTR	defra research	NS589651	258902	665028					GB0641A	Partisol 20	PM10	28	48	71	94	23 Mar 2003	33	81		Scotland
2000	Glasgow Centre	URBAN CENTR	defra research	NS589651	258902	665028					GB0641A	Partisol 20	PM2.5	10	8	13	22	16 Nov 2000		13		Scotland
2001	Glasgow Centre	URBAN CENTR	defra research	NS589651	258902	665028					GB0641A	Partisol 20	PM2.5	14	19	43	112	12 Dec 2001		52		Scotland
2002	Glasgow Centre	URBAN CENTR	defra research	NS589651	258902	665028					GB0641A	Partisol 20	PM2.5	14	15	39	109	20 Dec 2002		41		Scotland
2003	Glasgow Centre	URBAN CENTR	defra research	NS589651	258902	665028					GB0641A	Partisol 20	PM2.5	18	24	50	77	23 Mar 2003		44		Scotland
1999	Glasgow Centre	URBAN CENTR	defra research	NS589651	258902	665028					GB0641A	KFG	PM10	19	0	18	33	20 Dec 1999	0	5		Scotland
2000	Glasgow Centre	URBAN CENTR	defra research	NS589651	258902	665028					GB0641A	KFG	PM10	24	37	61	73	22 Oct 2000	18	63		Scotland
2001	Glasgow Centre	URBAN CENTR	defra research	NS589651	258902	665028					GB0641A	KFG	PM10	20	33	58	72	14 Jan 2001	12	61		Scotland
2002	Glasgow Centre	URBAN CENTR	defra research	NS589651	258902	665028					GB0641A	KFG	PM10	18	28	58	74	29 Mar 2002	12	71		Scotland
2003	Glasgow Centre	URBAN CENTR	defra research	NS589651	258902	665028					GB0641A	KFG	PM10	24	41	71	93	26 Feb 2003	22	63		Scotland
1997	Glasgow Kerbside	KERBSIDE	AURN	NS587652	258708	665200	Hope St.	25,000/day	3		GB0657A	Teom	PM10	31	46	62	93	3-Dec-97	76	72		Scotland
1998	Glasgow Kerbside	KERBSIDE	AURN	NS587652	258708	665200	Hope St.	25,000/day	3		GB0657A	Teom	PM10	27	51	72	113	15-May-98	68	97		Scotland
1999	Glasgow Kerbside	KERBSIDE	AURN	NS587652	258708	665200	Hope St.	25,000/day	3		GB0657A	Teom	PM10	21	41	54	81	2-Apr-99	43	97		Scotland
2000	Glasgow Kerbside	KERBSIDE	AURN	NS587652	258708	665200	Hope St.	25,000/day	3		GB0657A	Teom	PM10	21	34	43	58	24-Jan-00	21	98		Scotland
2001	Glasgow Kerbside	KERBSIDE	AURN	NS587652	258708	665200	Hope St.	25,000/day	3		GB0657A	Teom	PM10	24	39	71	113	12-Dec-01	35	98		Scotland
2002	Glasgow Kerbside	KERBSIDE	AURN	NS587652	258708	665200	Hope St.	25,000/day	3		GB0657A	Teom	PM10	23	38	50	101	3-Jan-02	29	97		Scotland
2003	Glasgow Kerbside	KERBSIDE	AURN	NS587652	258708	665200	Hope St.	25,000/day	3		GB0657A	Teom	PM10	24	43	66	87	25-Mar-03	47	93		Scotland
2001	Grangemouth	URBAN INDUST	AURN	NS938810	293840	681032					GB0735A	Teom	PM10	15	23	35	47	12-Dec-01	4	75		Scotland

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2002	Grangemouth	URBAN INDUST	AURN	NS938810	293840	681032					GB0735A	Teom	PM10	13	16	28	42	5-Apr-02	3	57		Scotland
2003	Grangemouth	URBAN INDUST	AURN	NS938810	293840	681032					GB0735A	Teom	PM10	15	25	41	56	18-Mar-03	11	98		Scotland
1996	Haringey Roadside	ROADSIDE	AURN	TQ339906	533900	190600	Tottenham		3	39,000/day	GB0637A	Teom	PM10	26	36	54	65	20-Aug-96	26	59		London
1997	Haringey Roadside	ROADSIDE	AURN	TQ339906	533900	190600	Tottenham		3	39,000/day	GB0637A	Teom	PM10	26	43	62	90	11-Jan-97	48	97		London
1998	Haringey Roadside	ROADSIDE	AURN	TQ339906	533900	190600	Tottenham		3	39,000/day	GB0637A	Teom	PM10	22	35	48	57	23-Nov-98	22	93		London
1999	Haringey Roadside	ROADSIDE	AURN	TQ339906	533900	190600	Tottenham		3	39,000/day	GB0637A	Teom	PM10	22	32	46	56	1-Apr-99	17	97		London
2000	Haringey Roadside	ROADSIDE	AURN	TQ339906	533900	190600	Tottenham		3	39,000/day	GB0637A	Teom	PM10	20	29	41	57	22-Mar-00	12	94		London
2001	Haringey Roadside	ROADSIDE	AURN	TQ339906	533900	190600	Tottenham		3	39,000/day	GB0637A	Teom	PM10	21	32	44	66	3-Nov-01	14	99		London
2002	Haringey Roadside	ROADSIDE	AURN	TQ339906	533900	190600	Tottenham		3	39,000/day	GB0637A	Teom	PM10	21	30	47	56	5-Nov-02	15	99		London
2003	Haringey Roadside	ROADSIDE	AURN	TQ339906	533900	190600	Tottenham		3	39,000/day	GB0637A	Teom	PM10	22	38	50	57	22-Feb-03	34	98		London
1998	Harwell	RURAL	AURN	SU474863	447400	186300					GB0036R	Teom	PM10	14	22	29	62	1-Sep-98	2	71		En gland
1999	Harwell	RURAL	AURN	SU474863	447400	186300					GB0036R	Teom	PM10	14	22	29	36	2-Sep-99	0	98		En gland
2000	Harwell	RURAL	AURN	SU474863	447400	186300					GB0036R	Teom	PM10	14	22	31	35	26-Jul-00	0	97		En gland
2001	Harwell	RURAL	AURN	SU474863	447400	186300					GB0036R	Teom	PM10	14	21	32	44	17-Jan-01	1	97		En gland
2002	Harwell	RURAL	AURN	SU474863	447400	186300					GB0036R	Teom	PM10	13	19	30	82	4-Sep-02	2	99		En gland
2003	Harwell	RURAL	AURN	SU474863	447400	186300					GB0036R	Teom	PM10	16	27	37	52	8-Aug-03	7	98		En gland
1998	Harwell	RURAL	AURN	SU474863	447400	186300					GB0036R	Teom	PM2.5	10	16	22	33	14-May-98		70		England
1999	Harwell	RURAL	AURN	SU474863	447400	186300					GB0036R	Teom	PM2.5	10	16	23	28	11-Sep-99		97		England
2000	Harwell	RURAL	AURN	SU474863	447400	186300					GB0036R	Teom	PM2.5	10	16	24	29	28-Jun-00		98		England
2001	Harwell	RURAL	AURN	SU474863	447400	186300					GB0036R	Teom	PM2.5	11	17	25	37	17-Jan-01		97		England
2002	Harwell	RURAL	AURN	SU474863	447400	186300					GB0036R	Teom	PM2.5	10	16	22	29	12-Dec-02		98		England
2003	Harwell	RURAL	AURN	SU474863	447400	186300					GB0036R	Teom	PM2.5	12	22	30	43	8-Aug-03		99		England
2000	Harwell	RURAL	defra research	SU474863	447400	186300					GB0036R	Partisol 20	PM10	15	17	24	43	23 Dec 2000	0	24		England
2001	Harwell	RURAL	defra research	SU474863	447400	186300					GB0036R	Partisol 20	PM10	19	29	45	68	19 Jan 2001	7	65		England
2002	Harwell	RURAL	defra research	SU474863	447400	186300					GB0036R	Partisol 20	PM10	19	32	50	78	05 Apr 2002	7	93		England
2003	Harwell	RURAL	defra research	SU474863	447400	186300					GB0036R	Partisol 20	PM10	23	42	62	111	29 Mar 2003	19	82		England
2000	Harwell	RURAL	defra research	SU474863	447400	186300					GB0036R	Partisol 20	PM2.5	10	9	19	41	23 Dec 2000		28		England
2001	Harwell	RURAL	defra research	SU474863	447400	186300					GB0036R	Partisol 20	PM2.5	13	23	41	67	19 Jan 2001		73		England
2002	Harwell	RURAL	defra research	SU474863	447400	186300					GB0036R	Partisol 20	PM2.5	13	24	45	67	05 Apr 2002		93		England
2003	Harwell	RURAL	defra research	SU474863	447400	186300					GB0036R	Partisol 20	PM2.5	17	38	55	97	29 Mar 2003		89		England
1999	Harwell	RURAL	defra research	SU474863	447400	186300					GB0036R	KFG	PM10	15	18	31	44	10 Sep 1999	0	39		En gland
2000	Harwell	RURAL	defra research	SU474863	447400	186300					GB0036R	KFG	PM10	17	30	53	78	12 May 2000	9	77		En gland
2001	Harwell	RURAL	defra research	SU474863	447400	186300					GB0036R	KFG	PM10	17	27	44	78	07 Mar 2001	6	76		En gland
2002	Harwell	RURAL	defra research	SU474863	447400	186300					GB0036R	KFG	PM10	18	31	55	77	05 Apr 2002	8	83		En gland
2003	Harwell	RURAL	defra research	SU474863	447400	186300					GB0036R	KFG	PM10	21	42	62	105	29 Mar 2003	22	87		En gland
1994	Hull Centre	URBAN CENTR	AURN	TA097288	509684	428880					GB0596A	Teom	PM10	26	40	56	85	11-Oct-94	38	90		En gland
1995	Hull Centre	URBAN CENTR	AURN	TA097288	509684	428880					GB0596A	Teom	PM10	24	39	54	72	11-Dec-95	35	98		En gland
1996	Hull Centre	URBAN CENTR	AURN	TA097288	509684	428880					GB0596A	Teom	PM10	25	41	69	108	21-Mar-96	42	96		En gland
1997	Hull Centre	URBAN CENTR	AURN	TA097288	509684	428880					GB0596A	Teom	PM10	24	38	56	169	22-Oct-97	33	94		En gland
1998	Hull Centre	URBAN CENTR	AURN	TA097288	509684	428880					GB0596A	Teom	PM10	22	33	45	61	17-Nov-98	17	94		En gland
1999	Hull Centre	URBAN CENTR	AURN	TA097288	509684	428880					GB0596A	Teom	PM10	22	31	46	66	2-Apr-99	16	98		En gland
2000	Hull Centre	URBAN CENTR	AURN	TA097288	509684	428880					GB0596A	Teom	PM10	20	30	38	54	22-Mar-00	6	95		En gland
2001	Hull Centre	URBAN CENTR	AURN	TA097288	509684	428880					GB0596A	Teom	PM10	22	34	46	54	15-Feb-01	20	98		En gland
2002	Hull Centre	URBAN CENTR	AURN	TA097288	509684	428880					GB0596A	Teom	PM10	26	**	27	46	10-Jan-02	1	4		En gland
2002	Hull Freetown	URBAN CENTR	AURN	TA094293	509400	429300					GB0776A	Teom	PM10	23	16	33	47	12-Dec-02	7	14		En gland
2003	Hull Freetown	URBAN CENTR	AURN	TA094293	509400	429300					GB0776A	Teom	PM10	22	34	52	62	15-Apr-03	25	89		En gland
2001	Inverness	ROADSIDE	AURN	NH657457	265720	845680	Telford St.	4m			GB0742A	Partisol	PM10	12	11	17	34	29-Sep-01	0	21		Scotland
2002	Inverness	ROADSIDE	AURN	NH657457	265720	845680	Telford St.	4m			GB0742A	Partisol	PM10	17	25	36	51	31-Dec-02	1	66		Scotland
2003	Inverness	ROADSIDE	AURN	NH657457	265720	845680	Telford St.	4m			GB0742A	Partisol	PM10	17	31	54	68	27-Feb-03	10	93		Scotland
1996	Leamington Spa	URBAN BACKG	AURN	SP319657	432060	265906					GB0643A	Teom	PM10	20	25	36	51	20-Aug-96	4	43		En gland
1997	Leamington Spa	URBAN BACKG	AURN	SP319657	432060	265906					GB0643A	Teom	PM10	19	29	44	50	1-Nov-97	12	87		En gland
1998	Leamington Spa	URBAN BACKG	AURN	SP319657	432060	265906					GB0643A	Teom	PM10	18	28	43	61	30-Jan-98	12	98		En gland
1999	Leamington Spa	URBAN BACKG	AURN	SP319657	432060	265906					GB0643A	Teom	PM10	17	25	33	48	1-Apr-99	2	98		En gland
2000	Leamington Spa	URBAN BACKG	AURN	SP319657	432060	265906					GB0643A	Teom	PM10	16	24	32	57	8-May-00	4	98		En gland
2001	Leamington Spa	URBAN BACKG	AURN	SP319657	432060	265906					GB0643A	Teom	PM10	16	24	34	43	18-Jan-01	3	99		En gland
2002	Leamington Spa	URBAN BACKG	AURN	SP319657	432060	265906					GB0643A	Teom	PM10	16	24	38	55	19-Aug-02	5	98		En gland
2003	Leamington Spa	URBAN BACKG	AURN	SP319657	432060	265906					GB0643A	Teom	PM10	21	30	49	93	24-Mar-03	19	86		En gland
1993	Leeds Centre	URBAN CENTR	AURN	SE299343	429976	434268					GB0584A	Teom	PM10	27	44	64	96	4-Nov-93	58	77		En gland
1994	Leeds Centre	URBAN CENTR	AURN	SE299343	429976	434268					GB0584A	Teom	PM10	26	45	64	114	11-Oct-94	51	92		En gland
1995	Leeds Centre	URBAN CENTR	AURN	SE299343	429976	434268					GB0584A	Teom	PM10	25	42	64	232	5-Nov-95	46	98		En gland
1996	Leeds Centre	URBAN CENTR	AURN	SE299343	429976	434268					GB0584A	Teom	PM10	27	45	71	83	21-Mar-96	64	98		En gland

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
1997	Leeds Centre	URBAN CENTR	AURN	SE299343	429976	434268					GB0584A	Teom	PM10	28	47	86	151	15-Apr-97	59	86		En gland
1998	Leeds Centre	URBAN CENTR	AURN	SE299343	429976	434268					GB0584A	Teom	PM10	22	37	49	74	17-Nov-98	22	97		En gland
1999	Leeds Centre	URBAN CENTR	AURN	SE299343	429976	434268					GB0584A	Teom	PM10	20	30	46	62	6-Sep-99	19	97		En gland
2000	Leeds Centre	URBAN CENTR	AURN	SE299343	429976	434268					GB0584A	Teom	PM10	18	28	38	42	22-Feb-00	3	97		En gland
2001	Leeds Centre	URBAN CENTR	AURN	SE299343	429976	434268					GB0584A	Teom	PM10	19	30	41	74	14-Feb-01	8	95		En gland
2002	Leeds Centre	URBAN CENTR	AURN	SE299343	429976	434268					GB0584A	Teom	PM10	19	31	44	74	7-Jan-02	13	98		En gland
2003	Leeds Centre	URBAN CENTR	AURN	SE299343	429976	434268					GB0584A	Teom	PM10	21	37	51	60	21-Feb-03	28	97		En gland
1994	Leicester Centre	URBAN CENTR	AURN	SK590050	458768	304820					GB0597A	Teom	PM10	21	35	49	65	22-Dec-94	27	93		En gland
1995	Leicester Centre	URBAN CENTR	AURN	SK590050	458768	304820					GB0597A	Teom	PM10	20	33	46	64	11-Dec-95	16	97		En gland
1996	Leicester Centre	URBAN CENTR	AURN	SK590050	458768	304820					GB0597A	Teom	PM10	22	35	58	84	19-Mar-96	22	96		En gland
1997	Leicester Centre	URBAN CENTR	AURN	SK590050	458768	304820					GB0597A	Teom	PM10	20	33	45	58	31-Oct-97	19	97		En gland
1998	Leicester Centre	URBAN CENTR	AURN	SK590050	458768	304820					GB0597A	Teom	PM10	18	29	42	74	30-Jan-98	13	95		En gland
1999	Leicester Centre	URBAN CENTR	AURN	SK590050	458768	304820					GB0597A	Teom	PM10	16	25	34	42	31-Aug-99	4	96		En gland
2000	Leicester Centre	URBAN CENTR	AURN	SK590050	458768	304820					GB0597A	Teom	PM10	13	20	28	37	26-Jan-00	0	97		En gland
2001	Leicester Centre	URBAN CENTR	AURN	SK590050	458768	304820					GB0597A	Teom	PM10	17	26	34	47	18-Jan-01	3	97		En gland
2002	Leicester Centre	URBAN CENTR	AURN	SK590050	458768	304820					GB0597A	Teom	PM10	15	21	34	46	4-Apr-02	3	79		En gland
2003	Leicester Centre	URBAN CENTR	AURN	SK590050	458768	304820					GB0597A	Teom	PM10	19	31	47	97	10-Nov-03	15	89		En gland
1993	Liverpool Centre	URBAN CENTR	AURN	SJ349908	334934	390628					GB0594A	Teom	PM10	29	42	74	163	23-Nov-93	47	54		En gland
1994	Liverpool Centre	URBAN CENTR	AURN	SJ349908	334934	390628					GB0594A	Teom	PM10	25	39	64	84	12-Oct-94	36	84		En gland
1995	Liverpool Centre	URBAN CENTR	AURN	SJ349908	334934	390628					GB0594A	Teom	PM10	27	48	59	78	11-Dec-95	61	94		En gland
1996	Liverpool Centre	URBAN CENTR	AURN	SJ349908	334934	390628					GB0594A	Teom	PM10	25	42	67	94	19-Mar-96	48	97		En gland
1997	Liverpool Centre	URBAN CENTR	AURN	SJ349908	334934	390628					GB0594A	Teom	PM10	25	39	50	79	29-Jan-97	33	96		En gland
1998	Liverpool Centre	URBAN CENTR	AURN	SJ349908	334934	390628					GB0594A	Teom	PM10	22	35	49	80	29-Jan-98	21	98		En gland
1999	Liverpool Centre	URBAN CENTR	AURN	SJ349908	334934	390628					GB0594A	Teom	PM10	20	30	42	50	19-May-99	10	97		En gland
2000	Liverpool Centre	URBAN CENTR	AURN	SJ349908	334934	390628					GB0594A	Teom	PM10	18	28	38	47	9-May-00	5	91		En gland
2001	Liverpool Centre	URBAN CENTR	AURN	SJ349908	334934	390628					GB0594A	Teom	PM10	20	31	44	75	12-Dec-01	14	98		En gland
2002	Liverpool Centre	URBAN CENTR	AURN	SJ349908	334934	390628					GB0594A	Teom	PM10	18	25	36	59	5-Apr-02	4	69		En gland
2003	Liverpool Speke	URBAN BACKG	AURN	SJ438835	343800	383500					GB0777A	Teom	PM10	19	27	40	55	9-Aug-03	8	57		En gland
1997	London A3 Roadside	ROADSIDE	AURN	TQ193653	519000	165223	A3	2.5m	3		GB0659A	Teom	PM10	29	38	56	87	2-Nov-97	34	50		London
1998	London A3 Roadside	ROADSIDE	AURN	TQ193653	519000	165223	A3	2.5m	3		GB0659A	Teom	PM10	24	39	54	88	13-May-98	35	97		London
1999	London A3 Roadside	ROADSIDE	AURN	TQ193653	519000	165223	A3	2.5m	3		GB0659A	Teom	PM10	23	35	45	54	12-Feb-99	20	98		London
2000	London A3 Roadside	ROADSIDE	AURN	TQ193653	519000	165223	A3	2.5m	3		GB0659A	Teom	PM10	20	31	42	50	26-Jan-00	15	98		London
2001	London A3 Roadside	ROADSIDE	AURN	TQ193653	519000	165223	A3	2.5m	3		GB0659A	Teom	PM10	21	32	48	79	18-Jan-01	15	98		London
2002	London A3 Roadside	ROADSIDE	AURN	TQ193653	519000	165223	A3	2.5m	3		GB0659A	Teom	PM10	18	27	32	43	5-Apr-02	2	97		London
2003	London A3 Roadside	ROADSIDE	AURN	TQ193653	519000	165223	A3	2.5m	3		GB0659A	Teom	PM10	26	40	56	65	26-Mar-03	42	97		London
1994	London Bexley	SUBURBAN	AURN	TQ518763	551852	176396					GB0608A	Teom	PM10	25	40	50	70	14-Oct-94	36	59		London
1995	London Bexley	SUBURBAN	AURN	TQ518763	551852	176396					GB0608A	Teom	PM10	24	40	64	83	5-Nov-95	39	95		London
1996	London Bexley	SUBURBAN	AURN	TQ518763	551852	176396					GB0608A	Teom	PM10	24	43	68	90	18-Mar-96	44	94		London
1997	London Bexley	SUBURBAN	AURN	TQ518763	551852	176396					GB0608A	Teom	PM10	23	38	57	92	2-Nov-97	31	93		London
1998	London Bexley	SUBURBAN	AURN	TQ518763	551852	176396					GB0608A	Teom	PM10	19	32	43	56	16-Mar-98	15	97		London
1999	London Bexley	SUBURBAN	AURN	TQ518763	551852	176396					GB0608A	Teom	PM10	19	32	47	59	1-Aug-99	17	98		London
2000	London Bexley	SUBURBAN	AURN	TQ518763	551852	176396					GB0608A	Teom	PM10	18	30	41	69	4-Nov-00	10	97		London
2001	London Bexley	SUBURBAN	AURN	TQ518763	551852	176396					GB0608A	Teom	PM10	19	29	47	141	3-Nov-01	17	96		London
2002	London Bexley	SUBURBAN	AURN	TQ518763	551852	176396					GB0608A	Teom	PM10	19	29	45	59	10-May-02	11	98		London
2003	London Bexley	SUBURBAN	AURN	TQ518763	551852	176396					GB0608A	Teom	PM10	21	37	52	61	26-Mar-03	33	97		London
1992	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM10	30	44	68	94	10-Apr-92	50	78		London
1993	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM10	29	49	75	100	9-Jun-93	64	97		London
1994	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM10	27	44	57	93	14-Oct-94	53	98		London
1995	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM10	28	46	64	77	3-May-95	56	93		London
1996	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM10	30	50	70	103	16-Jan-96	64	92		London
1997	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM10	27	40	60	85	30-Oct-97	43	96		London
1998	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM10	23	32	44	61	16-Mar-98	20	94		London
1999	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM10	22	32	45	60	1-Apr-99	20	96		London
2000	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM10	21	33	41	44	9-May-00	11	97		London
2001	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM10	22	33	45	60	13-Oct-01	17	98		London
2002	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM10	25	27	51	67	4-Apr-02	21	36		London
2003	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM10	23	31	46	62	8-Aug-03	14	58		London
1998	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM2.5	16	21	31	44	13-May-98		73		London
1999	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM2.5	15	22	32	41	11-Sep-99		84		London
2000	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM2.5	14	23	29	34	9-May-00		94		London
2001	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM2.5	13	20	28	46	15-Feb-01		98		London

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2002	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM2.5	14	21	31	39	4-Apr-02		87		London
2003	London Bloomsbury	URBAN CENTR	AURN	TQ302820	530134	181986					GB0566A	Teom	PM2.5	14	25	34	43	22-Feb-03		97		London
1996	London Brent	URBAN CENTR	AURN	TQ200840	520090	189252					GB0616A	Teom	PM10	20	29	41	51	25-Mar-96	13	73		London
1997	London Brent	URBAN CENTR	AURN	TQ200840	520090	189252					GB0616A	Teom	PM10	22	35	56	82	30-Oct-97	24	95		London
1998	London Brent	URBAN CENTR	AURN	TQ200840	520090	189252					GB0616A	Teom	PM10	18	29	40	53	16-Mar-98	8	96		London
1999	London Brent	URBAN CENTR	AURN	TQ200840	520090	189252					GB0616A	Teom	PM10	18	28	35	49	11-Sep-99	6	98		London
2000	London Brent	URBAN CENTR	AURN	TQ200840	520090	189252					GB0616A	Teom	PM10	18	27	36	53	23-Mar-00	6	98		London
2001	London Brent	URBAN CENTR	AURN	TQ200840	520090	189252					GB0616A	Teom	PM10	18	28	43	74	4-Nov-01	9	99		London
2002	London Brent	URBAN CENTR	AURN	TQ200840	520090	189252					GB0616A	Teom	PM10	19	30	45	100	4-Nov-02	11	98		London
2003	London Brent	URBAN CENTR	AURN	TQ200840	520090	189252					GB0616A	Teom	PM10	20	34	48	62	22-Feb-03	24	96		London
1996	London Eltham	SUBURBAN	AURN	TQ440747	544000	174700					GB0586A	Teom	PM10	21	31	44	52	7-Apr-96	15	66		London
1997	London Eltham	SUBURBAN	AURN	TQ440747	544000	174700					GB0586A	Teom	PM10	21	33	51	81	2-Nov-97	24	91		London
1998	London Eltham	SUBURBAN	AURN	TQ440747	544000	174700					GB0586A	Teom	PM10	18	26	38	54	16-Mar-98	5	86		London
1999	London Eltham	SUBURBAN	AURN	TQ440747	544000	174700					GB0586A	Teom	PM10	17	27	36	51	1-Apr-99	4	98		London
2000	London Eltham	SUBURBAN	AURN	TQ440747	544000	174700					GB0586A	Teom	PM10	16	23	33	47	4-Nov-00	2	93		London
2001	London Eltham	SUBURBAN	AURN	TQ440747	544000	174700					GB0586A	Teom	PM10	18	27	41	81	3-Nov-01	11	97		London
2002	London Eltham	SUBURBAN	AURN	TQ440747	544000	174700					GB0586A	Teom	PM10	18	27	34	47	12-Dec-02	5	95		London
2003	London Eltham	SUBURBAN	AURN	TQ440747	544000	174700					GB0586A	Teom	PM10	21	37	55	72	5-Dec-03	25	99		London
1996	London Hillingdon	SUBURBAN	AURN	TQ069786	506939	178621					GB0642A	Teom	PM10	24	30	43	66	20-Aug-96	15	41		London
1997	London Hillingdon	SUBURBAN	AURN	TQ069786	506939	178621					GB0642A	Teom	PM10	25	41	59	88	30-Oct-97	50	98		London
1998	London Hillingdon	SUBURBAN	AURN	TQ069786	506939	178621					GB0642A	Teom	PM10	20	31	46	60	13-Feb-98	13	93		London
1999	London Hillingdon	SUBURBAN	AURN	TQ069786	506939	178621					GB0642A	Teom	PM10	21	31	43	54	12-Mar-99	11	98		London
2000	London Hillingdon	SUBURBAN	AURN	TQ069786	506939	178621					GB0642A	Teom	PM10	20	31	40	54	23-Mar-00	12	98		London
2001	London Hillingdon	SUBURBAN	AURN	TQ069786	506939	178621					GB0642A	Teom	PM10	20	31	44	60	3-Nov-01	12	97		London
2002	London Hillingdon	SUBURBAN	AURN	TQ069786	506939	178621					GB0642A	Teom	PM10	19	29	38	48	5-Apr-02	7	98		London
2003	London Hillingdon	SUBURBAN	AURN	TQ069786	506939	178621					GB0642A	Teom	PM10	23	38	54	59	10-Aug-03	32	89		London
1997	London Marylebone Ro	KERBSIDE	AURN	TQ281820	528100	182000	Marylebone	1m	3	50,000/day	GB0682A	Teom	PM10	39	47	68	97	30-Oct-97	70	45		London
1998	London Marylebone Ro	KERBSIDE	AURN	TQ281820	528100	182000	Marylebone	1m	3	50,000/day	GB0682A	Teom	PM10	32	45	57	67	13-Feb-98	82	99		London
1999	London Marylebone Ro	KERBSIDE	AURN	TQ281820	528100	182000	Marylebone	1m	3	50,000/day	GB0682A	Teom	PM10	35	50	83	139	13-Aug-99	110	95		London
2000	London Marylebone Ro	KERBSIDE	AURN	TQ281820	528100	182000	Marylebone	1m	3	50,000/day	GB0682A	Teom	PM10	37	51	67	87	23-Mar-00	157	99		London
2001	London Marylebone Ro	KERBSIDE	AURN	TQ281820	528100	182000	Marylebone	1m	3	50,000/day	GB0682A	Teom	PM10	34	47	68	89	19-Jan-01	104	89		London
2002	London Marylebone Ro	KERBSIDE	AURN	TQ281820	528100	182000	Marylebone	1m	3	50,000/day	GB0682A	Teom	PM10	34	48	59	76	15-Aug-02	106	98		London
2003	London Marylebone Ro	KERBSIDE	AURN	TQ281820	528100	182000	Marylebone	1m	3	50,000/day	GB0682A	Teom	PM10	37	55	67	77	22-Feb-03	161	99		London
1998	London Marylebone Ro	KERBSIDE	AURN	TQ281820	528100	182000	Marylebone	1m		50,000/day	GB0682A	Teom	PM2.5	20	26	36	45	13-May-98		75		London
1999	London Marylebone Ro	KERBSIDE	AURN	TQ281820	528100	182000	Marylebone	1m		50,000/day	GB0682A	Teom	PM2.5	22	32	41	51	22-Jan-99		90		London
2000	London Marylebone Ro	KERBSIDE	AURN	TQ281820	528100	182000	Marylebone	1m		50,000/day	GB0682A	Teom	PM2.5	26	36	46	58	23-Mar-00		99		London
2001	London Marylebone Ro	KERBSIDE	AURN	TQ281820	528100	182000	Marylebone	1m		50,000/day	GB0682A	Teom	PM2.5	25	37	53	68	3-Nov-01		94		London
2002	London Marylebone Ro	KERBSIDE	AURN	TQ281820	528100	182000	Marylebone	1m		50,000/day	GB0682A	Teom	PM2.5	22	30	37	42	12-Dec-02		96		London
2003	London Marylebone Ro	KERBSIDE	AURN	TQ281820	528100	182000	Marylebone	1m		50,000/day	GB0682A	Teom	PM2.5	19	30	39	48	22-Feb-03		93		London
2000	London Marylebone Ro	KERBSIDE	defra research	TQ281820	528100	182000	Marylebone Road			50,000/day	GB0682A	Partisol 20	PM10	50	33	58	126	26 Dec 2000	18	11		London
2001	London Marylebone Ro	KERBSIDE	defra research	TQ281820	528100	182000	Marylebone Road			50,000/day	GB0682A	Partisol 20	PM10	44	56	78	115	15 Feb 2001	50	45		London
2002	London Marylebone Ro	KERBSIDE	defra research	TQ281820	528100	182000	Marylebone Road			50,000/day	GB0682A	Partisol 20	PM10	44	62	81	128	23 Jan 2002	65	67		London
2003	London Marylebone Ro	KERBSIDE	defra research	TQ281820	528100	182000	Marylebone Road			50,000/day	GB0682A	Partisol 20	PM10	45	69	96	134	29 Mar 2003	96	86		London
2000	London Marylebone Ro	KERBSIDE	defra research	TQ281820	528100	182000	Marylebone Road			50,000/day	GB0682A	Partisol 20	PM2.5	33	34	45	77	15 Nov 2000		22		London
2001	London Marylebone Ro	KERBSIDE	defra research	TQ281820	528100	182000	Marylebone Road			50,000/day	GB0682A	Partisol 20	PM2.5	26	37	51	74	04 Nov 2001		80		London
2002	London Marylebone Ro	KERBSIDE	defra research	TQ281820	528100	182000	Marylebone Road			50,000/day	GB0682A	Partisol 20	PM2.5	25	38	51	69	04 Apr 2002		78		London
2003	London Marylebone Ro	KERBSIDE	defra research	TQ281820	528100	182000	Marylebone Road			50,000/day	GB0682A	Partisol 20	PM2.5	30	45	67	106	29 Mar 2003		80		London
1999	London Marylebone Ro	KERBSIDE	defra research	TQ281820	528100	182000	Marylebone Road			50,000/day	GB0682A	KFG	PM10	40	51	76	91	02 Sep 1999	38	44		London
2000	London Marylebone Ro	KERBSIDE	defra research	TQ281820	528100	182000	Marylebone Road			50,000/day	GB0682A	KFG	PM10	46	64	81	98	15 May 2000	102	72		London
2001	London Marylebone Ro	KERBSIDE	defra research	TQ281820	528100	182000	Marylebone Road			50,000/day	GB0682A	KFG	PM10	38	54	74	98	18 Jan 2001	51	70		London
2002	London Marylebone Ro	KERBSIDE	defra research	TQ281820	528100	182000	Marylebone Road			50,000/day	GB0682A	KFG	PM10	37	54	71	91	04 Apr 2002	49	90		London
2003	London Marylebone Ro	KERBSIDE	defra research	TQ281820	528100	182000	Marylebone Road			50,000/day	GB0682A	KFG	PM10	38	61	80	116	29 Mar 2003	69	94		London
1996	London N. Kensington	URBAN BACKG	AURN	TQ240817	524000	181700					GB0620A	Teom	PM10	23	34	46	66	14-Nov-96	21	75		London
1997	London N. Kensington	URBAN BACKG	AURN	TQ240817	524000	181700					GB0620A	Teom	PM10	24	38	64	89	30-Oct-97	34	98		London
1998	London N. Kensington	URBAN BACKG	AURN	TQ240817	524000	181700					GB0620A	Teom	PM10	20	32	46	55	16-Mar-98	16	98		London
1999	London N. Kensington	URBAN BACKG	AURN	TQ240817	524000	181700					GB0620A	Teom	PM10	21	32	45	59	11-Sep-99	16	99		London
2000	London N. Kensington	URBAN BACKG	AURN	TQ240817	524000	181700					GB0620A	Teom	PM10	20	31	42	58	23-Mar-00	11	96		London
2001	London N. Kensington	URBAN BACKG	AURN	TQ240817	524000	181700					GB0620A	Teom	PM10	20	30	48	75	4-Nov-01	14	96		London
2002	London N. Kensington	URBAN BACKG	AURN	TQ240817	524000	181700					GB0620A	Teom	PM10	19	29	40	49	5-Apr-02	8	99		London

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2003	London N. Kensington	URBAN BACKG	AURN	TQ240817	524000	181700					GB0620A	Teom	PM10	22	37	52	59	8-Aug-03	29	98		London
2000	London N. Kensington	URBAN BACKG	defra research	TQ240817	524000	181700					GB0620A	Partisol 20	PM10	22	No Data	20	34	29 Sep 2000	0	5		London
2001	London N. Kensington	URBAN BACKG	defra research	TQ240817	524000	181700					GB0620A	Partisol 20	PM10	24	36	50	88	04 Nov 2001	7	78		London
2002	London N. Kensington	URBAN BACKG	defra research	TQ240817	524000	181700					GB0620A	Partisol 20	PM10	25	40	58	78	05 Apr 2002	14	87		London
2003	London N. Kensington	URBAN BACKG	defra research	TQ240817	524000	181700					GB0620A	Partisol 20	PM10	28	48	71	119	29 Mar 2003	32	88		London
2000	London N. Kensington	URBAN BACKG	defra research	TQ240817	524000	181700					GB0620A	Partisol 20	PM2.5	15	12	24	49	15 Nov 2000		19		London
2001	London N. Kensington	URBAN BACKG	defra research	TQ240817	524000	181700					GB0620A	Partisol 20	PM2.5	18	29	49	91	19 Jan 2001		88		London
2002	London N. Kensington	URBAN BACKG	defra research	TQ240817	524000	181700					GB0620A	Partisol 20	PM2.5	18	33	51	62	04 Apr 2002		91		London
2003	London N. Kensington	URBAN BACKG	defra research	TQ240817	524000	181700					GB0620A	Partisol 20	PM2.5	21	39	58	104	29 Mar 2003		87		London
2003	London Westminster	URBAN BACKG	AURN	TQ298790	529800	179000					GB0743A	Partisol	PM10	38	43	81	134	29-Mar-03	28	41		London
1996	Lough Navar	REMOTE	AURN	HO65545	19200	521200					GB0006R	Teom	PM10	10	10	18	21	2-Dec-96	0	23		N Ireland
1997	Lough Navar	REMOTE	AURN	HO65545	19200	521200					GB0006R	Teom	PM10	10	16	23	29	26-Sep-97	0	96		N Ireland
1998	Lough Navar	REMOTE	AURN	HO65545	19200	521200					GB0006R	Teom	PM10	10	15	26	34	25-Sep-98	0	98		N Ireland
1999	Lough Navar	REMOTE	AURN	HO65545	19200	521200					GB0006R	Teom	PM10	9	14	22	29	6-May-99	0	96		N Ireland
2000	Lough Navar	REMOTE	AURN	HO65545	19200	521200					GB0006R	Teom	PM10	10	15	23	27	29-Jun-00	0	99		N Ireland
2001	Lough Navar	REMOTE	AURN	HO65545	19200	521200					GB0006R	Teom	PM10	10	14	20	32	12-May-01	0	96		N Ireland
2002	Lough Navar	REMOTE	AURN	HO65545	19200	521200					GB0006R	Teom	PM10	11	17	25	45	12-Sep-02	2	96		N Ireland
2003	Lough Navar	REMOTE	AURN	HO65545	19200	521200					GB0006R	Teom	PM10	12	21	32	42	21-Apr-03	1	99		N Ireland
1995	Manchester Piccadilly	URBAN CENTR	AURN	SJ843983	384334	398326					GB0613A	Teom	PM10	27	**	19	91	28-Dec-95	2	4		England
1996	Manchester Piccadilly	URBAN CENTR	AURN	SJ843983	384334	398326					GB0613A	Teom	PM10	26	44	65	96	19-Mar-96	46	98		England
1997	Manchester Piccadilly	URBAN CENTR	AURN	SJ843983	384334	398326					GB0613A	Teom	PM10	24	37	52	66	11-Aug-97	32	94		England
1998	Manchester Piccadilly	URBAN CENTR	AURN	SJ843983	384334	398326					GB0613A	Teom	PM10	21	33	53	66	28-Jan-98	19	97		England
1999	Manchester Piccadilly	URBAN CENTR	AURN	SJ843983	384334	398326					GB0613A	Teom	PM10	20	32	40	57	11-Jan-99	11	98		England
2000	Manchester Piccadilly	URBAN CENTR	AURN	SJ843983	384334	398326					GB0613A	Teom	PM10	21	34	48	65	25-Aug-00	22	98		England
2001	Manchester Piccadilly	URBAN CENTR	AURN	SJ843983	384334	398326					GB0613A	Teom	PM10	30	52	88	142	19-Nov-01	69	97		England
2002	Manchester Piccadilly	URBAN CENTR	AURN	SJ843983	384334	398326					GB0613A	Teom	PM10	21	33	42	66	5-Apr-02	12	95		England
2003	Manchester Piccadilly	URBAN CENTR	AURN	SJ843983	384334	398326					GB0613A	Teom	PM10	22	37	50	58	17-Apr-03	33	98		England
2002	Manchester Piccadilly	URBAN CENTR	defra research	SJ843983	384334	398326					GB0613A	Partisol 20	PM10	30	45	70	100	05 Apr 2002	24	71		England
2003	Manchester Piccadilly	URBAN CENTR	defra research	SJ843983	384334	398326					GB0613A	Partisol 20	PM10	34	59	85	109	28 Mar 2003	54	96		England
2002	Manchester Piccadilly	URBAN CENTR	defra research	SJ843983	384334	398326					GB0613A	Partisol 20	PM2.5	16	24	39	62	04 Apr 2002		70		England
2003	Manchester Piccadilly	URBAN CENTR	defra research	SJ843983	384334	398326					GB0613A	Partisol 20	PM2.5	17	31	43	62	23 Nov 2003		98		England
1995	Middlesbrough	URBAN INDUST	AURN	NZ505194	450480	519632					GB0583A	Teom	PM10	22	38	59	80	31-Jul-95	34	69		England
1996	Middlesbrough	URBAN INDUST	AURN	NZ505194	450480	519632					GB0583A	Teom	PM10	21	39	65	102	21-Mar-96	33	97		England
1997	Middlesbrough	URBAN INDUST	AURN	NZ505194	450480	519632					GB0583A	Teom	PM10	19	33	54	149	5-Nov-97	21	95		England
1998	Middlesbrough	URBAN INDUST	AURN	NZ505194	450480	519632					GB0583A	Teom	PM10	16	26	39	56	15-May-98	8	96		England
1999	Middlesbrough	URBAN INDUST	AURN	NZ505194	450480	519632					GB0583A	Teom	PM10	17	29	58	98	3-Apr-99	20	97		England
2000	Middlesbrough	URBAN INDUST	AURN	NZ505194	450480	519632					GB0583A	Teom	PM10	16	25	38	50	9-May-00	5	96		England
2001	Middlesbrough	URBAN INDUST	AURN	NZ505194	450480	519632					GB0583A	Teom	PM10	16	26	43	60	22-Oct-01	9	98		England
2002	Middlesbrough	URBAN INDUST	AURN	NZ505194	450480	519632					GB0583A	Teom	PM10	17	25	40	65	4-Apr-02	9	79		England
2003	Middlesbrough	URBAN INDUST	AURN	NZ505194	450480	519632					GB0583A	Teom	PM10	21	38	59	113	14-Jul-03	32	86		England
1997	Narberth	RURAL	AURN	SN146127	214600	212700					GB0043R	Teom	PM10	14	22	27	39	11-Mar-97	0	76		Wales
1998	Narberth	RURAL	AURN	SN146127	214600	212700					GB0043R	Teom	PM10	14	21	36	52	16-Mar-98	6	89		Wales
1999	Narberth	RURAL	AURN	SN146127	214600	212700					GB0043R	Teom	PM10	13	20	28	32	4-Sep-99	0	81		Wales
2000	Narberth	RURAL	AURN	SN146127	214600	212700					GB0043R	Teom	PM10	13	20	29	48	9-May-00	3	81		Wales
2001	Narberth	RURAL	AURN	SN146127	214600	212700					GB0043R	Teom	PM10	11	17	23	33	25-May-01	0	93		Wales
2002	Narberth	RURAL	AURN	SN146127	214600	212700					GB0043R	Teom	PM10	11	17	25	40	5-Apr-02	1	91		Wales
2003	Narberth	RURAL	AURN	SN146127	214600	212700					GB0043R	Teom	PM10	14	26	44	67	26-Mar-03	10	88		Wales
1992	Newcastle Centre	URBAN CENTR	AURN	NZ251649	425016	564940					GB0568A	Teom	PM10	28	41	65	72	29-Jun-92	42	66		England
1993	Newcastle Centre	URBAN CENTR	AURN	NZ251649	425016	564940					GB0568A	Teom	PM10	29	42	63	79	9-Jun-93	54	63		England
1994	Newcastle Centre	URBAN CENTR	AURN	NZ251649	425016	564940					GB0568A	Teom	PM10	26	43	60	77	13-Oct-94	50	95		England
1995	Newcastle Centre	URBAN CENTR	AURN	NZ251649	425016	564940					GB0568A	Teom	PM10	22	36	55	75	5-Nov-95	27	96		England
1996	Newcastle Centre	URBAN CENTR	AURN	NZ251649	425016	564940					GB0568A	Teom	PM10	24	42	67	94	21-Mar-96	42	98		England
1997	Newcastle Centre	URBAN CENTR	AURN	NZ251649	425016	564940					GB0568A	Teom	PM10	21	34	43	53	11-Mar-97	20	97		England
1998	Newcastle Centre	URBAN CENTR	AURN	NZ251649	425016	564940					GB0568A	Teom	PM10	18	29	37	51	27-Sep-98	5	96		England
1999	Newcastle Centre	URBAN CENTR	AURN	NZ251649	425016	564940					GB0568A	Teom	PM10	16	24	36	63	2-Apr-99	6	98		England
2000	Newcastle Centre	URBAN CENTR	AURN	NZ251649	425016	564940					GB0568A	Teom	PM10	13	21	25	34	14-May-00	0	98		England
2001	Newcastle Centre	URBAN CENTR	AURN	NZ251649	425016	564940					GB0568A	Teom	PM10	13	21	28	40	22-Oct-01	1	98		England
2002	Newcastle Centre	URBAN CENTR	AURN	NZ251649	425016	564940					GB0568A	Teom	PM10	14	22	30	39	11-Apr-02	1	98		England
2003	Newcastle Centre	URBAN CENTR	AURN	NZ251649	425016	564940					GB0568A	Teom	PM10	16	28	40	53	15-Apr-03	8	96		England
2001	Northampton	URBAN BACKG	AURN	SP761645	476111	264524					GB0738A	Teom	PM10	16	24	31	41	19-Nov-01	1	95		England
2002	Northampton	URBAN BACKG	AURN	SP761645	476111	264524					GB0738A	Teom	PM10	15	23	34	40	5-Apr-02	3	99		England

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2003	Northampton	URBAN BACKG	AURN	SP761645	476111	264524					GB0738A	Teom	PM10	17	28	40	58	9-Aug-03	9	99		England
2001	Northampton PM10	URBAN BACKG	AURN	SP761645	476111	264524					GB788A	Partisol	PM10	18	20	28	41	14-May-01	0	23		England
2002	Northampton PM10	URBAN BACKG	AURN	SP761645	476111	264524					GB788A	Partisol	PM10	22	33	47	70	4-Apr-02	6	70		England
2003	Northampton PM10	URBAN BACKG	AURN	SP761645	476111	264524					GB788A	Partisol	PM10	27	33	69	102	29-Mar-03	20	42		England
1997	Norwich Centre	URBAN CENTR	AURN	TG230089	623000	308900					GB0684A	Teom	PM10	22	27	41	67	30-Oct-97	9	43		England
1998	Norwich Centre	URBAN CENTR	AURN	TG230089	623000	308900					GB0684A	Teom	PM10	20	29	38	44	28-Jan-98	6	95		England
1999	Norwich Centre	URBAN CENTR	AURN	TG230089	623000	308900					GB0684A	Teom	PM10	18	26	36	50	11-Sep-99	6	88		England
2000	Norwich Centre	URBAN CENTR	AURN	TG230089	623000	308900					GB0684A	Teom	PM10	17	25	34	46	22-Mar-00	1	97		England
2001	Norwich Centre	URBAN CENTR	AURN	TG230089	623000	308900					GB0684A	Teom	PM10	15	23	35	39	17-Jan-01	0	96		England
2002	Norwich Centre	URBAN CENTR	AURN	TG230089	623000	308900					GB0684A	Teom	PM10	16	25	34	38	4-Apr-02	0	97		England
2003	Norwich Centre	URBAN CENTR	AURN	TG230089	623000	308900					GB0684A	Teom	PM10	18	30	41	57	20-Apr-03	9	97		England
1996	Nottingham Centre	URBAN CENTR	AURN	SK574400	457452	340103					GB0646A	Teom	PM10	24	28	40	54	10-Nov-96	8	31		England
1997	Nottingham Centre	URBAN CENTR	AURN	SK574400	457452	340103					GB0646A	Teom	PM10	23	37	51	77	11-Mar-97	29	97		England
1998	Nottingham Centre	URBAN CENTR	AURN	SK574400	457452	340103					GB0646A	Teom	PM10	21	32	46	86	2-Sep-98	21	98		England
1999	Nottingham Centre	URBAN CENTR	AURN	SK574400	457452	340103					GB0646A	Teom	PM10	19	29	40	46	12-Mar-99	9	97		England
2000	Nottingham Centre	URBAN CENTR	AURN	SK574400	457452	340103					GB0646A	Teom	PM10	18	27	37	87	9-May-00	5	99		England
2001	Nottingham Centre	URBAN CENTR	AURN	SK574400	457452	340103					GB0646A	Teom	PM10	20	29	44	84	17-Jan-01	14	98		England
2002	Nottingham Centre	URBAN CENTR	AURN	SK574400	457452	340103					GB0646A	Teom	PM10	19	27	35	45	12-Sep-02	4	89		England
2003	Nottingham Centre	URBAN CENTR	AURN	SK574400	457452	340103					GB0646A	Teom	PM10	20	34	46	57	20-Apr-03	23	91		England
1997	Plymouth Centre	URBAN CENTR	AURN	SX477546	247742	54600					GB0687A	Teom	PM10	21	21	34	39	3-Nov-97	2	22		England
1998	Plymouth Centre	URBAN CENTR	AURN	SX477546	247742	54600					GB0687A	Teom	PM10	17	29	37	62	16-Mar-98	6	98		England
1999	Plymouth Centre	URBAN CENTR	AURN	SX477546	247742	54600					GB0687A	Teom	PM10	16	25	35	38	11-Mar-99	0	97		England
2000	Plymouth Centre	URBAN CENTR	AURN	SX477546	247742	54600					GB0687A	Teom	PM10	16	25	36	86	3-Mar-00	2	95		England
2001	Plymouth Centre	URBAN CENTR	AURN	SX477546	247742	54600					GB0687A	Teom	PM10	15	25	34	45	19-Jan-01	3	97		England
2002	Plymouth Centre	URBAN CENTR	AURN	SX477546	247742	54600					GB0687A	Teom	PM10	15	23	30	49	1-Feb-02	2	98		England
2003	Plymouth Centre	URBAN CENTR	AURN	SX477546	247742	54600					GB0687A	Teom	PM10	17	29	36	59	9-Aug-03	6	98		England
1997	Port Talbot	URBAN BACKG	AURN	SS780882	278036	188249					GB0651A	Teom	PM10	27	44	61	88	8-Jul-97	59	88		Wales
1998	Port Talbot	URBAN BACKG	AURN	SS780882	278036	188249					GB0651A	Teom	PM10	27	47	71	108	15-May-98	59	98		Wales
1999	Port Talbot	URBAN BACKG	AURN	SS780882	278036	188249					GB0651A	Teom	PM10	26	47	66	81	22-May-99	62	97		Wales
2000	Port Talbot	URBAN BACKG	AURN	SS780882	278036	188249					GB0651A	Teom	PM10	26	45	63	92	2-Mar-00	56	96		Wales
2001	Port Talbot	URBAN BACKG	AURN	SS780882	278036	188249					GB0651A	Teom	PM10	23	40	61	88	12-Jul-01	39	97		Wales
2002	Port Talbot	URBAN BACKG	AURN	SS780882	278036	188249					GB0651A	Teom	PM10	21	35	45	77	9-Feb-02	24	98		Wales
2003	Port Talbot	URBAN BACKG	AURN	SS780882	278036	188249					GB0651A	Teom	PM10	24	40	60	70	11-Aug-03	43	98		Wales
2000	Port Talbot	URBAN BACKG	defra research	SS780882	278036	188249					GB0651A	Partisol 20	PM10	25	17	45	71	14 Dec 2000	4	17		Wales
2001	Port Talbot	URBAN BACKG	defra research	SS780882	278036	188249					GB0651A	Partisol 20	PM10	28	44	63	80	06 Apr 2001	21	63		Wales
2002	Port Talbot	URBAN BACKG	defra research	SS780882	278036	188249					GB0651A	Partisol 20	PM10	26	39	60	84	10 Feb 2002	15	73		Wales
2003	Port Talbot	URBAN BACKG	defra research	SS780882	278036	188249					GB0651A	Partisol 20	PM10	32	55	73	110	29 Mar 2003	44	92		Wales
2000	Port Talbot	URBAN BACKG	defra research	SS780882	278036	188249					GB0651A	Partisol 20	PM2.5	11	11	16	29	23 Dec 2000		23		Wales
2001	Port Talbot	URBAN BACKG	defra research	SS780882	278036	188249					GB0651A	Partisol 20	PM2.5	13	21	32	67	11 May 2001		62		Wales
2002	Port Talbot	URBAN BACKG	defra research	SS780882	278036	188249					GB0651A	Partisol 20	PM2.5	13	17	36	70	05 Apr 2002		66		Wales
2003	Port Talbot	URBAN BACKG	defra research	SS780882	278036	188249					GB0651A	Partisol 20	PM2.5	18	35	51	90	29 Mar 2003		91		Wales
1999	Port Talbot	URBAN BACKG	defra research	SS780882	278036	188249					GB0651A	KFG	PM10	24	21	46	62	03 Sep 1999	5	21		Wales
2000	Port Talbot	URBAN BACKG	defra research	SS780882	278036	188249					GB0651A	KFG	PM10	27	44	73	94	06 May 2000	25	63		Wales
2001	Port Talbot	URBAN BACKG	defra research	SS780882	278036	188249					GB0651A	KFG	PM10	25	45	60	84	10 May 2001	20	79		Wales
2002	Port Talbot	URBAN BACKG	defra research	SS780882	278036	188249					GB0651A	KFG	PM10	26	38	76	137	19 Mar 2002	14	73		Wales
2003	Port Talbot	URBAN BACKG	defra research	SS780882	278036	188249					GB0651A	KFG	PM10	27	45	68	118	29 Mar 2003	29	96		Wales
2001	Portsmouth	URBAN BACKG	AURN	SU656036	465686	103607					GB0733A	Teom	PM10	18	26	39	61	25-Jul-01	8	97		England
2002	Portsmouth	URBAN BACKG	AURN	SU656036	465686	103607					GB0733A	Teom	PM10	18	25	34	46	5-Apr-02	2	97		England
2003	Portsmouth	URBAN BACKG	AURN	SU656036	465686	103607					GB0733A	Teom	PM10	18	29	40	53	9-Aug-03	8	97		England
2000	Preston	URBAN BACKG	AURN	SD552301	355200	430100					GB0731A	Teom	PM10	15	20	26	102	5-Nov-00	2	56		England
2001	Preston	URBAN BACKG	AURN	SD552301	355200	430100					GB0731A	Teom	PM10	15	23	32	43	18-Jan-01	4	96		England
2002	Preston	URBAN BACKG	AURN	SD552301	355200	430100					GB0731A	Teom	PM10	15	23	32	76	5-Mar-02	3	98		England
2003	Preston	URBAN BACKG	AURN	SD552301	355200	430100					GB0731A	Teom	PM10	16	28	41	66	19-Mar-03	12	97		England
1997	Reading	URBAN BACKG	AURN	SU727733	472694	173244					GB0683A	Teom	PM10	21	24	37	77	31-Oct-97	7	30		England
1998	Reading	URBAN BACKG	AURN	SU727733	472694	173244					GB0683A	Teom	PM10	18	28	39	51	16-Mar-98	8	98		England
1999	Reading	URBAN BACKG	AURN	SU727733	472694	173244					GB0683A	Teom	PM10	16	25	33	39	12-Mar-99	1	91		England
2000	Reading	URBAN BACKG	AURN	SU727733	472694	173244					GB0683A	Teom	PM10	17	29	39	113	25-Aug-00	8	95		England
2001	Reading	URBAN BACKG	AURN	SU727733	472694	173244					GB0683A	Teom	PM10	17	28	39	70	29-Aug-01	8	96		England
2002	Reading	URBAN BACKG	AURN	SU727733	472694	173244					GB0683A	Teom	PM10	14	23	31	46	12-Dec-02	3	98		England
2003	Reading	URBAN BACKG	AURN	SU727733	472694	173244					GB0683A	Teom	PM10	12	**	13	22	12-Jan-03	0	9		England
2003	Reading New Town	URBAN BACKG	AURN	SU735732	473500	173200						Teom	PM10	14	**	**	36	6-Nov-03	0	7		England

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintorin g method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrecte d)	36th highest day PM (ug -3, uncorrecte d)	8th highest day PM (ug m-3, uncorrecte d)	maximum day PM (ug m-3, uncorrecte d)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
1997	Redcar	SUBURBAN	AURN	NZ599246	459975	524563					GB0679A	Teom	PM10	23	31	43	59	12-Aug-97	17	50		England
1998	Redcar	SUBURBAN	AURN	NZ599246	459975	524563					GB0679A	Teom	PM10	21	33	47	73	20-Mar-98	17	98		England
1999	Redcar	SUBURBAN	AURN	NZ599246	459975	524563					GB0679A	Teom	PM10	19	30	39	74	2-Apr-99	8	98		England
2000	Redcar	SUBURBAN	AURN	NZ599246	459975	524563					GB0679A	Teom	PM10	16	25	34	50	18-Jan-00	3	96		England
2001	Redcar	SUBURBAN	AURN	NZ599246	459975	524563					GB0679A	Teom	PM10	17	27	34	52	11-Dec-01	5	98		England
2002	Redcar	SUBURBAN	AURN	NZ599246	459975	524563					GB0679A	Teom	PM10	17	27	41	48	1-Oct-02	9	98		England
2003	Redcar	SUBURBAN	AURN	NZ599246	459975	524563					GB0679A	Teom	PM10	21	33	52	83	16-Apr-03	19	98		England
1996	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM10	24	37	59	78	18-Mar-96	32	69		England
1997	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM10	20	31	44	82	11-Jan-97	18	91		England
1998	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM10	17	25	34	54	16-Mar-98	3	80		England
1999	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM10	16	24	34	50	10-Sep-99	3	94		England
2000	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM10	17	26	35	46	19-Jun-00	2	88		England
2001	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM10	16	24	36	51	4-Nov-01	6	91		England
2002	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM10	18	24	38	49	10-May-02	6	61		England
2003	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM10	19	28	46	66	20-Aug-03	15	76		England
1998	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM2.5	11	16	24	30	11-Aug-98		64		England
1999	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM2.5	11	19	26	35	1-Apr-99		96		England
2000	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM2.5	11	18	24	34	23-Mar-00		88		England
2001	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM2.5	11	18	24	46	4-Nov-01		91		England
2002	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM2.5	11	19	27	39	10-May-02		97		England
2003	Rochester	RURAL	AURN	TQ831762	583100	176200					GB0617A	Teom	PM2.5	12	24	33	40	22-Feb-03		98		England
1997	Salford Eccles	URBAN INDUST	AURN	SJ779987	377932	398743					GB0660A	Teom	PM10	22	33	45	57	31-Oct-97	15	70		England
1998	Salford Eccles	URBAN INDUST	AURN	SJ779987	377932	398743					GB0660A	Teom	PM10	20	31	47	79	29-Jan-98	11	92		England
1999	Salford Eccles	URBAN INDUST	AURN	SJ779987	377932	398743					GB0660A	Teom	PM10	18	28	38	49	31-Jan-99	5	93		England
2000	Salford Eccles	URBAN INDUST	AURN	SJ779987	377932	398743					GB0660A	Teom	PM10	17	27	36	52	22-Mar-00	6	98		England
2001	Salford Eccles	URBAN INDUST	AURN	SJ779987	377932	398743					GB0660A	Teom	PM10	19	30	44	58	18-Jan-01	11	97		England
2002	Salford Eccles	URBAN INDUST	AURN	SJ779987	377932	398743					GB0660A	Teom	PM10	16	24	37	49	12-Sep-02	4	96		England
2003	Salford Eccles	URBAN INDUST	AURN	SJ779987	377932	398743					GB0660A	Teom	PM10	18	32	44	55	19-Mar-03	17	96		England
1997	Scunthorpe	URBAN INDUST	AURN	SE905107	490592	410689					GB0690A	Teom	PM10	26	**	21	67	17-Dec-97	4	5		England
1998	Scunthorpe	URBAN INDUST	AURN	SE905107	490592	410689					GB0690A	Teom	PM10	21	40	54	60	6-Oct-98	41	100		England
1999	Scunthorpe	URBAN INDUST	AURN	SE905107	490592	410689					GB0690A	Teom	PM10	23	42	64	124	2-Aug-99	45	99		England
2000	Scunthorpe	URBAN INDUST	AURN	SE905107	490592	410689					GB0690A	Teom	PM10	21	37	54	69	17-Jun-00	31	90		England
2001	Scunthorpe	URBAN INDUST	AURN	SE905107	490592	410689					GB0690A	Teom	PM10	24	45	64	112	17-Jul-01	49	93		England
2002	Scunthorpe	URBAN INDUST	AURN	SE905107	490592	410689					GB0690A	Teom	PM10	25	44	64	85	17-Apr-02	47	84		England
2003	Scunthorpe	URBAN INDUST	AURN	SE905107	490592	410689					GB0690A	Teom	PM10	30	62	82	117	17-Apr-03	95	99		England
1995	Sheffield Centre	URBAN CENTR	AURN	SK352869	435134	386885					GB0615A	Teom	PM10	27	**	19	46	29-Dec-95	2	3		England
1996	Sheffield Centre	URBAN CENTR	AURN	SK352869	435134	386885					GB0615A	Teom	PM10	28	49	78	102	19-Mar-96	61	98		England
1997	Sheffield Centre	URBAN CENTR	AURN	SK352869	435134	386885					GB0615A	Teom	PM10	25	39	53	82	11-Mar-97	37	89		England
1998	Sheffield Centre	URBAN CENTR	AURN	SK352869	435134	386885					GB0615A	Teom	PM10	23	37	54	82	7-Aug-98	29	85		England
1999	Sheffield Centre	URBAN CENTR	AURN	SK352869	435134	386885					GB0615A	Teom	PM10	20	32	40	54	1-Apr-99	9	98		England
2000	Sheffield Centre	URBAN CENTR	AURN	SK352869	435134	386885					GB0615A	Teom	PM10	19	30	38	61	9-May-00	3	97		England
2001	Sheffield Centre	URBAN CENTR	AURN	SK352869	435134	386885					GB0615A	Teom	PM10	18	29	41	58	17-Jan-01	11	96		England
2002	Sheffield Centre	URBAN CENTR	AURN	SK352869	435134	386885					GB0615A	Teom	PM10	19	30	47	55	5-Apr-02	12	94		England
2003	Sheffield Centre	URBAN CENTR	AURN	SK352869	435134	386885					GB0615A	Teom	PM10	21	38	55	84	27-Feb-03	31	98		England
1994	Southampton Centre	URBAN CENTR	AURN	SU440120	442575	112260					GB0598A	Teom	PM10	23	35	48	63	11-Oct-94	24	88		England
1995	Southampton Centre	URBAN CENTR	AURN	SU440120	442575	112260					GB0598A	Teom	PM10	22	31	50	64	4-Nov-95	17	73		England
1996	Southampton Centre	URBAN CENTR	AURN	SU440120	442575	112260					GB0598A	Teom	PM10	23	39	55	82	16-Mar-96	35	98		England
1997	Southampton Centre	URBAN CENTR	AURN	SU440120	442575	112260					GB0598A	Teom	PM10	24	37	58	80	14-Jan-97	22	93		England
1998	Southampton Centre	URBAN CENTR	AURN	SU440120	442575	112260					GB0598A	Teom	PM10	21	33	43	62	13-Feb-98	14	98		England
1999	Southampton Centre	URBAN CENTR	AURN	SU440120	442575	112260					GB0598A	Teom	PM10	20	30	40	46	12-Feb-99	8	91		England
2000	Southampton Centre	URBAN CENTR	AURN	SU440120	442575	112260					GB0598A	Teom	PM10	18	28	37	53	6-Feb-00	6	97		England
2001	Southampton Centre	URBAN CENTR	AURN	SU440120	442575	112260					GB0598A	Teom	PM10	20	30	44	79	3-Nov-01	12	98		England
2002	Southampton Centre	URBAN CENTR	AURN	SU440120	442575	112260					GB0598A	Teom	PM10	20	28	39	55	5-Apr-02	8	88		England
2003	Southampton Centre	URBAN CENTR	AURN	SU440120	442575	112260					GB0598A	Teom	PM10	21	33	43	58	26-Mar-03	14	91		England
2000	Southend-on-Sea	URBAN BACKG	AURN	TQ858862	585566	186130					GB0728A	Teom	PM10	13	16	23	31	4-Nov-00	0	43		England
2001	Southend-on-Sea	URBAN BACKG	AURN	TQ858862	585566	186130					GB0728A	Teom	PM10	15	23	36	73	3-Nov-01	4	98		England
2002	Southend-on-Sea	URBAN BACKG	AURN	TQ858862	585566	186130					GB0728A	Teom	PM10	15	25	35	85	17-Dec-02	4	97		England
2003	Southend-on-Sea	URBAN BACKG	AURN	TQ858862	585566	186130					GB0728A	Teom	PM10	16	28	42	54	20-Apr-03	13	84		England
1996	Stockport	URBAN BACKG	AURN	SJ895908	389581	390357					GB0648A	Teom	PM10	24	**	29	70	6-Dec-96	3	9		England
1997	Stockport	URBAN BACKG	AURN	SJ895908	389581	390357					GB0648A	Teom	PM10	21	33	47	78	15-Jan-97	16	84		England
1998	Stockport	URBAN BACKG	AURN	SJ895908	389581	390357					GB0648A	Teom	PM10	20	31	44	86	30-Jan-98	16	99		England

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
1999	Stockport	URBAN BACKG	AURN	SJ895908	389581	390357					GB0648A	Teom	PM10	19	29	39	55	11-Jan-99	7	98		England
2000	Stockport	URBAN BACKG	AURN	SJ895908	389581	390357					GB0648A	Teom	PM10	17	26	33	50	9-May-00	3	97		England
2001	Stockport	URBAN BACKG	AURN	SJ895908	389581	390357					GB0648A	Teom	PM10	18	26	38	71	11-Dec-01	6	99		England
2002	Stockport	URBAN BACKG	AURN	SJ895908	389581	390357					GB0648A	Teom	PM10	15	21	32	50	5-Apr-02	3	71		England
2002	Stockport Shaw Heath	URBAN BACKG	AURN	SJ893896	389386	389604					GB0775A	Teom	PM10	14	14	20	32	12-Dec-02	0	22		England
2003	Stockport Shaw Heath	URBAN BACKG	AURN	SJ893896	389386	389604					GB0775A	Teom	PM10	15	25	37	49	18-Apr-03	6	99		England
2001	Stockton-on-Tees Yarm	ROADSIDE	AURN	NZ419128	441908	512886	no details	3m		no details	GB0734A	Teom	PM10	23	33	44	64	24-May-01	14	58		England
2002	Stockton-on-Tees Yarm	ROADSIDE	AURN	NZ419128	441908	512886	no details	3m		no details	GB0734A	Teom	PM10	22	33	48	59	4-Apr-02	18	97		England
2003	Stockton-on-Tees Yarm	ROADSIDE	AURN	NZ419128	441908	512886	no details	3m		no details	GB0734A	Teom	PM10	25	43	60	73	26-Feb-03	46	96		England
1997	Stoke-on-Trent Centre	URBAN CENTR	AURN	SJ883468	388348	347894					GB0658A	Teom	PM10	22	29	41	82	31-Oct-97	15	48		England
1998	Stoke-on-Trent Centre	URBAN CENTR	AURN	SJ883468	388348	347894					GB0658A	Teom	PM10	20	32	49	79	14-May-98	17	98		England
1999	Stoke-on-Trent Centre	URBAN CENTR	AURN	SJ883468	388348	347894					GB0658A	Teom	PM10	17	28	38	56	17-Mar-99	6	98		England
2000	Stoke-on-Trent Centre	URBAN CENTR	AURN	SJ883468	388348	347894					GB0658A	Teom	PM10	16	25	36	48	25-Jan-00	5	98		England
2001	Stoke-on-Trent Centre	URBAN CENTR	AURN	SJ883468	388348	347894					GB0658A	Teom	PM10	17	27	39	63	19-Jan-01	7	98		England
2002	Stoke-on-Trent Centre	URBAN CENTR	AURN	SJ883468	388348	347894					GB0658A	Teom	PM10	15	25	35	62	4-Sep-02	5	97		England
2003	Stoke-on-Trent Centre	URBAN CENTR	AURN	SJ883468	388348	347894					GB0658A	Teom	PM10	11	21	34	45	20-Apr-03	4	98		England
1996	Sutton Roadside	ROADSIDE	AURN	TQ256646	526600	164600	St. Nichola	3.5m	2.5	20,000/day	GB0623A	Teom	PM10	26	38	57	250	25-Jul-96	34	75		London
1997	Sutton Roadside	ROADSIDE	AURN	TQ256646	526600	164600	St. Nichola	3.5m	2.5	20,000/day	GB0623A	Teom	PM10	24	38	57	102	2-Nov-97	34	99		London
1998	Sutton Roadside	ROADSIDE	AURN	TQ256646	526600	164600	St. Nichola	3.5m	2.5	20,000/day	GB0623A	Teom	PM10	21	33	43	59	13-May-98	12	98		London
1999	Sutton Roadside	ROADSIDE	AURN	TQ256646	526600	164600	St. Nichola	3.5m	2.5	20,000/day	GB0623A	Teom	PM10	20	31	37	46	11-Sep-99	4	98		London
2000	Sutton Roadside	ROADSIDE	AURN	TQ256646	526600	164600	St. Nichola	3.5m	2.5	20,000/day	GB0623A	Teom	PM10	19	28	39	49	9-May-00	9	92		London
2001	Sutton Roadside	ROADSIDE	AURN	TQ256646	526600	164600	St. Nichola	3.5m	2.5	20,000/day	GB0623A	Teom	PM10	20	31	43	67	18-Jan-01	12	86		London
2002	Sutton Roadside	ROADSIDE	AURN	TQ256646	526600	164600	St. Nichola	3.5m	2.5	20,000/day	GB0623A	Teom	PM10	20	22	34	54	5-Apr-02	4	33		London
1994	Swansea	URBAN CENTR	AURN	SS655931	265566	193158					GB0609A	Teom	PM10	22	**	23	46	1-Dec-94	1	6		Wales
1995	Swansea	URBAN CENTR	AURN	SS655931	265566	193158					GB0609A	Teom	PM10	27	44	61	69	25-Jul-95	49	96		Wales
1996	Swansea	URBAN CENTR	AURN	SS655931	265566	193158					GB0609A	Teom	PM10	23	37	53	83	15-Mar-96	33	97		Wales
1997	Swansea	URBAN CENTR	AURN	SS655931	265566	193158					GB0609A	Teom	PM10	24	37	44	56	15-Jan-97	22	98		Wales
1998	Swansea	URBAN CENTR	AURN	SS655931	265566	193158					GB0609A	Teom	PM10	21	32	45	79	15-May-98	18	97		Wales
1999	Swansea	URBAN CENTR	AURN	SS655931	265566	193158					GB0609A	Teom	PM10	17	25	32	42	3-Sep-99	1	96		Wales
2000	Swansea	URBAN CENTR	AURN	SS655931	265566	193158					GB0609A	Teom	PM10	20	29	43	63	9-May-00	12	97		Wales
2001	Swansea	URBAN CENTR	AURN	SS655931	265566	193158					GB0609A	Teom	PM10	20	29	38	54	19-Jan-01	7	95		Wales
2002	Swansea	URBAN CENTR	AURN	SS655931	265566	193158					GB0609A	Teom	PM10	19	29	36	53	11-Apr-02	5	98		Wales
2003	Swansea	URBAN CENTR	AURN	SS655931	265566	193158					GB0609A	Teom	PM10	19	31	41	52	20-Apr-03	11	98		Wales
1996	Thurrock	URBAN BACKG	AURN	TQ611779	561018	177894					GB0645A	Teom	PM10	23	24	36	64	5-Dec-96	5	26		England
1997	Thurrock	URBAN BACKG	AURN	TQ611779	561018	177894					GB0645A	Teom	PM10	22	36	56	95	11-Jan-97	27	93		England
1998	Thurrock	URBAN BACKG	AURN	TQ611779	561018	177894					GB0645A	Teom	PM10	19	32	43	59	16-Mar-98	16	92		England
1999	Thurrock	URBAN BACKG	AURN	TQ611779	561018	177894					GB0645A	Teom	PM10	19	30	43	80	29-May-99	13	95		England
2000	Thurrock	URBAN BACKG	AURN	TQ611779	561018	177894					GB0645A	Teom	PM10	18	28	42	52	22-Mar-00	10	94		England
2001	Thurrock	URBAN BACKG	AURN	TQ611779	561018	177894					GB0645A	Teom	PM10	19	27	45	90	3-Nov-01	12	76		England
2002	Thurrock	URBAN BACKG	AURN	TQ611779	561018	177894					GB0645A	Teom	PM10	21	32	47	88	17-Dec-02	16	82		England
2003	Thurrock	URBAN BACKG	AURN	TQ611779	561018	177894					GB0645A	Teom	PM10	23	41	58	104	9-Apr-03	38	98		England
1999	Thurrock	URBAN BACKG	defra research	TQ611779	561018	177894					GB0645A	KFG	PM10	24	32	56	72	10 Sep 1999	9	50		England
2000	Thurrock	URBAN BACKG	defra research	TQ611779	561018	177894					GB0645A	KFG	PM10	26	42	63	78	13 May 2000	19	74		England
2001	Thurrock	URBAN BACKG	defra research	TQ611779	561018	177894					GB0645A	KFG	PM10	29	48	68	92	20 Jan 2001	33	91		England
2002	Thurrock	URBAN BACKG	defra research	TQ611779	561018	177894					GB0645A	KFG	PM10	30	44	59	97	06 Dec 2002	24	70		England
2003	Thurrock	URBAN BACKG	defra research	TQ611779	561018	177894					GB0645A	KFG	PM10	33	61	89	128	29 Mar 2003	59	99		England
2001	Wigan Leigh	URBAN BACKG	AURN	SJ662999	366468	399842					GB0736A	Teom	PM10	19	29	40	55	17-Jan-01	8	97		England
2002	Wigan Leigh	URBAN BACKG	AURN	SJ662999	366468	399842					GB0736A	Teom	PM10	18	26	37	52	12-Sep-02	7	99		England
2003	Wigan Leigh	URBAN BACKG	AURN	SJ662999	366468	399842					GB0736A	Teom	PM10	20	35	48	58	20-Apr-03	25	98		England
2000	Wirral Tranmere	URBAN BACKG	AURN	SJ320867	332096	386644					GB0730A	Teom	PM10	17	22	42	111	16-Aug-00	9	61		England
2001	Wirral Tranmere	URBAN BACKG	AURN	SJ320867	332096	386644					GB0730A	Teom	PM10	16	23	35	59	18-Jan-01	5	98		England
2002	Wirral Tranmere	URBAN BACKG	AURN	SJ320867	332096	386644					GB0730A	Teom	PM10	16	25	35	53	5-Apr-02	4	97		England
2003	Wirral Tranmere	URBAN BACKG	AURN	SJ320867	332096	386644					GB0730A	Teom	PM10	16	29	41	54	20-Apr-03	10	97		England
1995	Wolverhampton Centre	URBAN CENTR	AURN	SO914989	391368	298942					GB0614A	Teom	PM10	24	**	20	42	19-Dec-95	1	4		England
1996	Wolverhampton Centre	URBAN CENTR	AURN	SO914989	391368	298942					GB0614A	Teom	PM10	26	42	71	110	19-Mar-96	41	71		England
1997	Wolverhampton Centre	URBAN CENTR	AURN	SO914989	391368	298942					GB0614A	Teom	PM10	22	37	51	84	11-Mar-97	30	99		England
1998	Wolverhampton Centre	URBAN CENTR	AURN	SO914989	391368	298942					GB0614A	Teom	PM10	18	32	54	70	14-May-98	17	98		England
1999	Wolverhampton Centre	URBAN CENTR	AURN	SO914989	391368	298942					GB0614A	Teom	PM10	17	29	39	52	1-Apr-99	8	98		England
2000	Wolverhampton Centre	URBAN CENTR	AURN	SO914989	391368	298942					GB0614A	Teom	PM10	18	28	41	73	10-May-00	11	98		England
2001	Wolverhampton Centre	URBAN CENTR	AURN	SO914989	391368	298942					GB0614A	Teom	PM10	17	29	40	68	18-Jan-01	10	96		England
2002	Wolverhampton Centre	URBAN CENTR	AURN	SO914989	391368	298942					GB0614A	Teom	PM10	17	29	40	55	5-Apr-02	9	98		England

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2003	Wolverhampton Centre	URBAN CENTR	AURN	SO914989	391368	298942					GB0614A	Teom	PM10	19	34	47	57	20-Apr-03	21	98		England
2002	Wrexham	ROADSIDE	AURN	SJ329499	332862	349904	Victoria Ro	5m			GB0755A	Partisol	PM10	25	42	65	84	5-Apr-02	21	78		Wales
2003	Wrexham	ROADSIDE	AURN	SJ329499	332862	349904	Victoria Ro	5m			GB0755A	Partisol	PM10	26	51	79	108	28-Mar-03	38	87		Wales
1999	Barking & Dagenham 2	suburban	LAQN	548030 18	548030	183363	-999	-999	-999	-999	BG2	TEOM	PM10	16	15	23	43	20-Dec-99	1	20		London
2000	Barking & Dagenham 2	suburban	LAQN	548030 18	548030	183363	-999	-999	-999	-999	BG2	TEOM	PM10	19	31	43	53	22-Mar-00	17	99		London
2001	Barking & Dagenham 2	suburban	LAQN	548030 18	548030	183363	-999	-999	-999	-999	BG2	TEOM	PM10	20	30	49	71	4-Nov-01	15	72		London
2002	Barking & Dagenham 2	suburban	LAQN	548030 18	548030	183363	-999	-999	-999	-999	BG2	TEOM	PM10	22	32	46	55	1-Oct-02	21	70		London
2003	Barking & Dagenham 2	suburban	LAQN	548030 18	548030	183363	-999	-999	-999	-999	BG2	TEOM	PM10	25	40	52	60	20-Mar-03	43	89		London
1998	Barnet 1 - Tally Ho Corr	kerbside	LAQN	526350 19	526350	192166	NULL	NULL	NULL	NULL	BN1	TEOM	PM10	19	7	15	36	21-Dec-98	0	3		London
1999	Barnet 1 - Tally Ho Corr	kerbside	LAQN	526350 19	526350	192166	NULL	NULL	NULL	NULL	BN1	TEOM	PM10	22	33	44	53	11-Sep-99	17	95		London
2000	Barnet 1 - Tally Ho Corr	kerbside	LAQN	526350 19	526350	192166	NULL	NULL	NULL	NULL	BN1	TEOM	PM10	20	28	38	56	4-Nov-00	6	90		London
2001	Barnet 1 - Tally Ho Corr	kerbside	LAQN	526350 19	526350	192166	NULL	NULL	NULL	NULL	BN1	TEOM	PM10	25	35	48	74	4-Nov-01	20	73		London
2002	Barnet 1 - Tally Ho Corr	kerbside	LAQN	526350 19	526350	192166	NULL	NULL	NULL	NULL	BN1	TEOM	PM10	29	41	57	66	1-Jan-02	50	84		London
2003	Barnet 1 - Tally Ho Corr	kerbside	LAQN	526350 19	526350	192166	NULL	NULL	NULL	NULL	BN1	TEOM	PM10	26	40	51	56	22-Feb-03	40	86		London
2000	Barnet 2 - Finchley	background	LAQN	524328 18	524328	189599	-999	-999	-999	-999	BN2	TEOM	PM10	16	16	22	28	11-Sep-00	0	20		London
2001	Barnet 2 - Finchley	background	LAQN	524328 18	524328	189599	-999	-999	-999	-999	BN2	TEOM	PM10	18	26	34	70	4-Nov-01	6	81		London
2002	Barnet 2 - Finchley	background	LAQN	524328 18	524328	189599	-999	-999	-999	-999	BN2	TEOM	PM10	19	28	41	47	4-Nov-02	8	100		London
2003	Barnet 2 - Finchley	background	LAQN	524328 18	524328	189599	-999	-999	-999	-999	BN2	TEOM	PM10	20	33	48	54	10-Aug-03	23	90		London
2000	Barnet 3 - Strawberry V	background	LAQN	526681 19	526681	190701	-999	-999	-999	-999	BN3	TEOM	PM10	18	20	28	46	4-Nov-00	2	38		London
2001	Barnet 3 - Strawberry V	background	LAQN	526681 19	526681	190701	-999	-999	-999	-999	BN3	TEOM	PM10	20	30	43	60	4-Nov-01	12	99		London
2002	Barnet 3 - Strawberry V	background	LAQN	526681 19	526681	190701	-999	-999	-999	-999	BN3	TEOM	PM10	22	23	41	85	21-Feb-02	10	35		London
1998	Bexley 2 - Belvedere	suburban	LAQN	550000 17	550000	179070	-999	-999	-999	-999	BX2	TEOM	PM10	19	31	42	55	16-Mar-98	19	92		London
1999	Bexley 2 - Belvedere	suburban	LAQN	550000 17	550000	179070	-999	-999	-999	-999	BX2	TEOM	PM10	18	32	48	63	1-Apr-99	18	95		London
2000	Bexley 2 - Belvedere	suburban	LAQN	550000 17	550000	179070	-999	-999	-999	-999	BX2	TEOM	PM10	18	29	41	59	4-Nov-00	11	98		London
2001	Bexley 2 - Belvedere	suburban	LAQN	550000 17	550000	179070	-999	-999	-999	-999	BX2	TEOM	PM10	19	31	53	74	3-Nov-01	24	99		London
2002	Bexley 2 - Belvedere	suburban	LAQN	550000 17	550000	179070	-999	-999	-999	-999	BX2	TEOM	PM10	19	30	47	54	3-Apr-02	16	82		London
2003	Bexley 2 - Belvedere	suburban	LAQN	550000 17	550000	179070	-999	-999	-999	-999	BX2	TEOM	PM10	21	36	54	60	15-Apr-03	30	88		London
1998	Bexley 2 - Belvedere	suburban	LAQN	550000 17	550000	179070	-999	-999	-999	-999	BX2	TEOM	PM2.5	14	24	34	39	29-Mar-98	***	89		London
1999	Bexley 2 - Belvedere	suburban	LAQN	550000 17	550000	179070	-999	-999	-999	-999	BX2	TEOM	PM2.5	12	22	32	46	1-Apr-99	***	93		London
2000	Bexley 2 - Belvedere	suburban	LAQN	550000 17	550000	179070	-999	-999	-999	-999	BX2	TEOM	PM2.5	12	19	28	42	23-Mar-00	***	75		London
2001	Bexley 2 - Belvedere	suburban	LAQN	550000 17	550000	179070	-999	-999	-999	-999	BX2	TEOM	PM2.5	13	16	25	64	4-Nov-01	***	45		London
2002	Bexley 2 - Belvedere	suburban	LAQN	550000 17	550000	179070	-999	-999	-999	-999	BX2	TEOM	PM2.5	11	18	26	38	10-May-02	***	97		London
2003	Bexley 2 - Belvedere	suburban	LAQN	550000 17	550000	179070	-999	-999	-999	-999	BX2	TEOM	PM2.5	13	26	37	45	22-Feb-03	***	90		London
1998	Bexley 3 - Thamesmead	suburban	LAQN	547340 18	547340	181210	-999	-999	-999	-999	BX3	TEOM	PM10	20	31	42	54	16-Mar-98	13	84		London
1999	Bexley 3 - Thamesmead	suburban	LAQN	547340 18	547340	181210	-999	-999	-999	-999	BX3	TEOM	PM10	18	28	40	50	1-Apr-99	13	78		London
2000	Bexley 3 - Thamesmead	suburban	LAQN	547340 18	547340	181210	-999	-999	-999	-999	BX3	TEOM	PM10	18	28	41	57	22-Mar-00	11	94		London
2001	Bexley 3 - Thamesmead	suburban	LAQN	547340 18	547340	181210	-999	-999	-999	-999	BX3	TEOM	PM10	19	29	44	72	3-Nov-01	14	96		London
2002	Bexley 3 - Thamesmead	suburban	LAQN	547340 18	547340	181210	-999	-999	-999	-999	BX3	TEOM	PM10	20	31	43	54	10-May-02	15	98		London
2003	Bexley 3 - Thamesmead	suburban	LAQN	547340 18	547340	181210	-999	-999	-999	-999	BX3	TEOM	PM10	16	12	22	34	19-Feb-03	0	13		London
1998	Bexley 3 - Thamesmead	suburban	LAQN	547340 18	547340	181210	-999	-999	-999	-999	BX3	TEOM	PM2.5	13	22	33	35	13-May-98	***	87		London
1999	Bexley 3 - Thamesmead	suburban	LAQN	547340 18	547340	181210	-999	-999	-999	-999	BX3	TEOM	PM2.5	13	20	31	38	1-Apr-99	***	96		London
2000	Bexley 3 - Thamesmead	suburban	LAQN	547340 18	547340	181210	-999	-999	-999	-999	BX3	TEOM	PM2.5	12	19	29	34	25-Jan-00	***	81		London
2001	Bexley 3 - Thamesmead	suburban	LAQN	547340 18	547340	181210	-999	-999	-999	-999	BX3	TEOM	PM2.5	11	18	33	196	4-Nov-01	***	98		London
2002	Bexley 3 - Thamesmead	suburban	LAQN	547340 18	547340	181210	-999	-999	-999	-999	BX3	TEOM	PM2.5	13	21	30	44	10-May-02	***	99		London
2003	Bexley 3 - Thamesmead	suburban	LAQN	547340 18	547340	181210	-999	-999	-999	-999	BX3	TEOM	PM2.5	13	25	35	43	22-Feb-03	***	91		London
1999	Bexley 4 - Erith	roadside	LAQN	552230 17	552230	177280	NULL	NULL	NULL	NULL	BX4	TEOM	PM10	36	68	99	140	17-May-99	81	62		London
2000	Bexley 4 - Erith	roadside	LAQN	552230 17	552230	177280	NULL	NULL	NULL	NULL	BX4	TEOM	PM10	27	56	83	121	22-Mar-00	78	98		London
2001	Bexley 4 - Erith	roadside	LAQN	552230 17	552230	177280	NULL	NULL	NULL	NULL	BX4	TEOM	PM10	31	58	82	132	3-Nov-01	100	98		London
2002	Bexley 4 - Erith	roadside	LAQN	552230 17	552230	177280	NULL	NULL	NULL	NULL	BX4	TEOM	PM10	33	64	87	150	6-Apr-02	100	95		London
2003	Bexley 4 - Erith	roadside	LAQN	552230 17	552230	177280	NULL	NULL	NULL	NULL	BX4	TEOM	PM10	37	72	90	109	15-Jul-03	127	90		London
1996	Brent 1 - Kingsbury (AU	background	LAQN	519600 18	519600	189300	-999	-999	-999	-999	BT1	TEOM	PM10	20	29	41	57	19-Aug-96	15	70		London
1997	Brent 1 - Kingsbury (AU	background	LAQN	519600 18	519600	189300	-999	-999	-999	-999	BT1	TEOM	PM10	22	35	56	82	30-Oct-97	26	95		London
1998	Brent 1 - Kingsbury (AU	background	LAQN	519600 18	519600	189300	-999	-999	-999	-999	BT1	TEOM	PM10	18	29	40	52	16-Mar-98	8	96		London
1999	Brent 1 - Kingsbury (AU	background	LAQN	519600 18	519600	189300	-999	-999	-999	-999	BT1	TEOM	PM10	18	29	35	49	11-Sep-99	6	98		London
2000	Brent 1 - Kingsbury (AU	background	LAQN	519600 18	519600	189300	-999	-999	-999	-999	BT1	TEOM	PM10	17	27	37	52	23-Mar-00	7	99		London
2001	Brent 1 - Kingsbury (AU	background	LAQN	519600 18	519600	189300	-999	-999	-999	-999	BT1	TEOM	PM10	18	28	43	74	4-Nov-01	9	99		London
2002	Brent 1 - Kingsbury (AU	background	LAQN	519600 18	519600	189300	-999	-999	-999	-999	BT1	TEOM	PM10	19	30	45	99	4-Nov-02	11	98		London
2003	Brent 1 - Kingsbury (AU	background	LAQN	519600 18	519600	189300	-999	-999	-999	-999	BT1	TEOM	PM10	20	34	48	62	22-Feb-03	25	86		London
2001	Brent 2 - Ikea Car Park	roadside	LAQN	520712 18	520712	185164	NULL	NULL	NULL	NULL	BT2	TEOM	PM10	29	39	48	70	3-Nov-01	38	50		London
2002	Brent 2 - Ikea Car Park	roadside	LAQN	520712 18	520712	185164	NULL	NULL	NULL	NULL	BT2	TEOM	PM10	29	40	49	60	1-Jan-02	38	66		London
2001	Brent 3 - Harlesden	background	LAQN	521520 18	521520	183475	-999	-999	-999	-999	BT3	TEOM	PM10	15	13	21	31	3-Nov-01	0	16		London

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2002	Brent 3 - Harlesden	background	LAQN	521520 18	521520	183475	-999	-999	-999	-999	BT3	TEOM	PM10	24	34	47	62	4-Apr-02	21	88		London
2003	Brent 3 - Harlesden	background	LAQN	521520 18	521520	183475	-999	-999	-999	-999	BT3	TEOM	PM10	26	40	57	67	8-Aug-03	39	78		London
2003	Brent 4 - Ikea	roadside	LAQN	520712 18	520712	185164	NULL	NULL	NULL	NULL	BT4	TEOM	PM10	33	39	54	63	27-Nov-03	37	33		London
1998	Bromley 7 - Central	roadside	LAQN	540400 16	540400	169400	NULL	7	3.5	NULL	BY7	BAM	PM10	32	32	64	97	24-Sep-98	10	26		London
1999	Bromley 7 - Central	roadside	LAQN	540400 16	540400	169400	NULL	7	3.5	NULL	BY7	BAM	PM10	38	58	78	141	3-Sep-99	57	90		London
2000	Bromley 7 - Central	roadside	LAQN	540400 16	540400	169400	NULL	7	3.5	NULL	BY7	BAM	PM10	25	41	60	84	23-Mar-00	16	86		London
2001	Bromley 7 - Central	roadside	LAQN	540400 16	540400	169400	NULL	7	3.5	NULL	BY7	BAM	PM10	28	44	78	109	17-Jan-01	23	88		London
2002	Bromley 7 - Central	roadside	LAQN	540400 16	540400	169400	NULL	7	3.5	NULL	BY7	BAM	PM10	27	45	58	88	30-Jul-02	15	92		London
2003	Bromley 7 - Central	roadside	LAQN	540400 16	540400	169400	NULL	7	3.5	NULL	BY7	BAM	PM10	28	34	59	89	8-Aug-03	15	44		London
1998	Bromley 7 - Central	roadside	LAQN	540400 16	540400	169400	NULL	7	3.5	NULL	BY7	BAM	PM2.5	26	26	52	72	24-Sep-98	***	24		London
1999	Bromley 7 - Central	roadside	LAQN	540400 16	540400	169400	NULL	7	3.5	NULL	BY7	BAM	PM2.5	30	39	58	995	3-Sep-99	***	86		London
2000	Bromley 7 - Central	roadside	LAQN	540400 16	540400	169400	NULL	7	3.5	NULL	BY7	BAM	PM2.5	19	33	53	72	23-Mar-00	***	87		London
2001	Bromley 7 - Central	roadside	LAQN	540400 16	540400	169400	NULL	7	3.5	NULL	BY7	BAM	PM2.5	20	35	58	90	17-Jan-01	***	67		London
2002	Bromley 7 - Central	roadside	LAQN	540400 16	540400	169400	NULL	7	3.5	NULL	BY7	BAM	PM2.5	-5	-5	-5	-4	26-Jan-02	***	0		London
1996	Camden 1 - Swiss Cotta	kerbside	LAQN	526650 18	526650	184460	NULL	NULL	NULL	NULL	CD1	TEOM	PM10	31	43	58	73	14-Nov-96	54	64		London
1997	Camden 1 - Swiss Cotta	kerbside	LAQN	526650 18	526650	184460	NULL	NULL	NULL	NULL	CD1	TEOM	PM10	32	48	73	97	15-Jan-97	86	99		London
1998	Camden 1 - Swiss Cotta	kerbside	LAQN	526650 18	526650	184460	NULL	NULL	NULL	NULL	CD1	TEOM	PM10	25	36	45	51	25-Sep-98	23	82		London
1999	Camden 1 - Swiss Cotta	kerbside	LAQN	526650 18	526650	184460	NULL	NULL	NULL	NULL	CD1	TEOM	PM10	26	38	51	61	15-Mar-99	33	95		London
2000	Camden 1 - Swiss Cotta	kerbside	LAQN	526650 18	526650	184460	NULL	NULL	NULL	NULL	CD1	TEOM	PM10	26	38	47	62	23-Mar-00	31	98		London
2001	Camden 1 - Swiss Cotta	kerbside	LAQN	526650 18	526650	184460	NULL	NULL	NULL	NULL	CD1	TEOM	PM10	25	37	50	73	4-Nov-01	29	99		London
2002	Camden 1 - Swiss Cotta	kerbside	LAQN	526650 18	526650	184460	NULL	NULL	NULL	NULL	CD1	TEOM	PM10	24	35	44	49	10-May-02	16	99		London
2003	Camden 1 - Swiss Cotta	kerbside	LAQN	526650 18	526650	184460	NULL	NULL	NULL	NULL	CD1	TEOM	PM10	27	40	56	67	8-Aug-03	43	91		London
2000	Camden 3 - Shaftesbury	roadside	LAQN	530038 18	530038	181142	NULL	NULL	NULL	NULL	CD3	TEOM	PM10	25	29	45	61	29-Jun-00	14	38		London
2001	Camden 3 - Shaftesbury	roadside	LAQN	530038 18	530038	181142	NULL	NULL	NULL	NULL	CD3	TEOM	PM10	27	40	54	69	17-Jan-01	41	98		London
2002	Camden 3 - Shaftesbury	roadside	LAQN	530038 18	530038	181142	NULL	NULL	NULL	NULL	CD3	TEOM	PM10	26	38	49	57	30-Jul-02	32	98		London
2003	Camden 3 - Shaftesbury	roadside	LAQN	530038 18	530038	181142	NULL	NULL	NULL	NULL	CD3	TEOM	PM10	31	46	66	84	8-Aug-03	66	88		London
1997	Croydon 3 - Thornton H	suburban	LAQN	532320 16	532320	168950	-999	-999	-999	-999	CR3	TEOM	PM10	22	30	45	84	2-Nov-97	15	48		London
1998	Croydon 3 - Thornton H	suburban	LAQN	532320 16	532320	168950	-999	-999	-999	-999	CR3	TEOM	PM10	19	31	42	59	13-May-98	11	95		London
1999	Croydon 3 - Thornton H	suburban	LAQN	532320 16	532320	168950	-999	-999	-999	-999	CR3	TEOM	PM10	19	30	39	45	11-Sep-99	8	95		London
2000	Croydon 3 - Thornton H	suburban	LAQN	532320 16	532320	168950	-999	-999	-999	-999	CR3	TEOM	PM10	17	26	37	114	25-Aug-00	5	97		London
2001	Croydon 3 - Thornton H	suburban	LAQN	532320 16	532320	168950	-999	-999	-999	-999	CR3	TEOM	PM10	19	29	45	76	3-Nov-01	12	96		London
2002	Croydon 3 - Thornton H	suburban	LAQN	532320 16	532320	168950	-999	-999	-999	-999	CR3	TEOM	PM10	18	27	36	54	4-Apr-02	5	96		London
2003	Croydon 3 - Thornton H	suburban	LAQN	532320 16	532320	168950	-999	-999	-999	-999	CR3	TEOM	PM10	20	33	42	61	26-Mar-03	17	86		London
1999	Croydon 4 - George Str	roadside	LAQN	532720 16	532720	165680	NULL	NULL	NULL	NULL	CR4	TEOM	PM10	19	17	26	35	20-Dec-99	0	16		London
2000	Croydon 4 - George Str	roadside	LAQN	532720 16	532720	165680	NULL	NULL	NULL	NULL	CR4	TEOM	PM10	23	39	50	61	17-Jul-00	39	97		London
2001	Croydon 4 - George Str	roadside	LAQN	532720 16	532720	165680	NULL	NULL	NULL	NULL	CR4	TEOM	PM10	23	35	50	83	3-Nov-01	26	90		London
2002	Croydon 4 - George Str	roadside	LAQN	532720 16	532720	165680	NULL	NULL	NULL	NULL	CR4	TEOM	PM10	23	34	45	58	5-Apr-02	18	98		London
2003	Croydon 4 - George Str	roadside	LAQN	532720 16	532720	165680	NULL	NULL	NULL	NULL	CR4	TEOM	PM10	24	37	53	61	8-Aug-03	32	88		London
1999	Crystal Palace 1 - C Pal	roadside	LAQN	533900 17	533900	171190	NULL	NULL	NULL	NULL	CY1	TEOM	PM10	22	23	32	38	12-Oct-99	0	24		London
2000	Crystal Palace 1 - C Pal	roadside	LAQN	533900 17	533900	171190	NULL	NULL	NULL	NULL	CY1	TEOM	PM10	23	33	43	175	16-Mar-00	18	97		London
2001	Crystal Palace 1 - C Pal	roadside	LAQN	533900 17	533900	171190	NULL	NULL	NULL	NULL	CY1	TEOM	PM10	24	36	49	73	3-Nov-01	24	99		London
2002	Crystal Palace 1 - C Pal	roadside	LAQN	533900 17	533900	171190	NULL	NULL	NULL	NULL	CY1	TEOM	PM10	22	31	38	61	10-May-02	6	99		London
2003	Crystal Palace 1 - C Pal	roadside	LAQN	533900 17	533900	171190	NULL	NULL	NULL	NULL	CY1	TEOM	PM10	21	30	42	50	26-Mar-03	18	90		London
1998	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM10	23	33	51	62	14-May-98	22	79		London
1999	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM10	23	37	49	64	1-Apr-99	26	97		London
2000	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM10	22	31	44	64	23-Mar-00	20	69		London
2001	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM10	23	38	53	80	3-Nov-01	30	94		London
2002	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM10	23	34	49	52	11-Oct-02	20	97		London
2003	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM10	27	45	60	70	22-Feb-03	57	91		London
1996	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM2.5	20	22	34	52	5-Dec-96	***	28		London
1997	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM2.5	21	35	60	90	2-Nov-97	***	97		London
1998	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM2.5	17	28	41	45	14-May-98	***	95		London
1999	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM2.5	16	26	38	45	1-Apr-99	***	98		London
2000	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM2.5	16	27	34	46	23-Mar-00	***	99		London
2001	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM2.5	17	27	41	66	3-Nov-01	***	99		London
2002	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM2.5	17	26	35	41	1-Jan-02	***	97		London
2003	Ealing 2 - Acton Town H	roadside	LAQN	520300 18	520300	180050	NULL	5	2	NULL	EA2	TEOM	PM2.5	17	15	30	45	22-Feb-03	***	20		London
1998	Ealing 3 - A40 East Act	roadside	LAQN	NULL	NULL	NULL	NULL	NULL	NULL	NULL	EA3	TEOM	PM10	23	20	35	52	18-Nov-98	4	16		London
1998	Ealing 4 - Mobile Ham	roadside	LAQN	512900 18	512900	180350	NULL	NULL	NULL	NULL	EA4	TEOM	PM10	16	6	18	36	7-Dec-98	0	8		London
1999	Ealing 4 - Mobile Ham	roadside	LAQN	512900 18	512900	180350	NULL	NULL	NULL	NULL	EA4	TEOM	PM10	16	13	24	34	22-Jan-99	0	16		London
1999	Ealing 5 - Mobile South	roadside	LAQN	513100 18	513100	180350	NULL	NULL	NULL	NULL	EA5	TEOM	PM10	21	31	42	53	1-Apr-99	13	76		London

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2000	Ealing 5 - Mobile Southa	roadside	LAQN	513100 18	513100	180350	NULL	NULL	NULL	NULL	EA5	TEOM	PM10	21	31	41	58	3-Nov-00	16	92		London
2001	Ealing 5 - Mobile Southa	roadside	LAQN	513100 18	513100	180350	NULL	NULL	NULL	NULL	EA5	TEOM	PM10	20	24	40	52	19-Jan-01	9	45		London
1998	Enfield 2 - Church Stree	roadside	LAQN	532630 19	532630	196590	NULL	NULL	NULL	NULL	EN2	BAM	PM10	32	47	84	104	14-May-98	31	77		London
1999	Enfield 2 - Church Stree	roadside	LAQN	532630 19	532630	196590	NULL	NULL	NULL	NULL	EN2	BAM	PM10	47	73	104	149	11-Sep-99	132	95		London
2000	Enfield 2 - Church Stree	roadside	LAQN	532630 19	532630	196590	NULL	NULL	NULL	NULL	EN2	BAM	PM10	42	68	104	158	23-Mar-00	77	93		London
2001	Enfield 2 - Church Stree	roadside	LAQN	532630 19	532630	196590	NULL	NULL	NULL	NULL	EN2	BAM	PM10	36	54	81	103	15-Feb-01	53	79		London
2002	Enfield 2 - Church Stree	roadside	LAQN	532630 19	532630	196590	NULL	NULL	NULL	NULL	EN2	BAM	PM10	35	54	72	88	4-Apr-02	50	90		London
2003	Enfield 2 - Church Stree	roadside	LAQN	532630 19	532630	196590	NULL	NULL	NULL	NULL	EN2	BAM	PM10	41	71	96	127	29-Mar-03	88	82		London
1998	Enfield 3 - Salisbury Sc	background	LAQN	535440 19	535440	195000	-999	-999	-999	-999	EN3	BAM	PM10	29	21	37	70	7-Dec-98	4	13		London
1999	Enfield 3 - Salisbury Sc	background	LAQN	535440 19	535440	195000	-999	-999	-999	-999	EN3	BAM	PM10	27	45	65	88	11-Sep-99	25	90		London
2000	Enfield 3 - Salisbury Sc	background	LAQN	535440 19	535440	195000	-999	-999	-999	-999	EN3	BAM	PM10	30	49	69	104	23-Mar-00	32	85		London
2001	Enfield 3 - Salisbury Sc	background	LAQN	535440 19	535440	195000	-999	-999	-999	-999	EN3	BAM	PM10	30	48	74	87	1-Aug-01	34	87		London
2002	Enfield 3 - Salisbury Sc	background	LAQN	535440 19	535440	195000	-999	-999	-999	-999	EN3	BAM	PM10	30	50	69	91	4-Apr-02	32	93		London
2003	Enfield 3 - Salisbury Sc	background	LAQN	535440 19	535440	195000	-999	-999	-999	-999	EN3	BAM	PM10	29	52	73	94	29-Mar-03	41	82		London
2000	Enfield 4 - Derby Road,	roadside	LAQN	535056 19	535056	192442	NULL	NULL	NULL	NULL	EN4	BAM	PM10	38	55	85	133	23-Mar-00	46	63		London
2001	Enfield 4 - Derby Road,	roadside	LAQN	535056 19	535056	192442	NULL	NULL	NULL	NULL	EN4	BAM	PM10	42	59	84	130	3-Nov-01	66	70		London
2002	Enfield 4 - Derby Road,	roadside	LAQN	535056 19	535056	192442	NULL	NULL	NULL	NULL	EN4	BAM	PM10	43	66	84	108	4-Apr-02	89	92		London
2003	Enfield 4 - Derby Road,	roadside	LAQN	535056 19	535056	192442	NULL	NULL	NULL	NULL	EN4	BAM	PM10	51	84	111	147	29-Mar-03	134	88		London
1994	Greenwich 4 - Eltham	background	LAQN	543982 17	543982	174676	-999	-999	-999	-999	GR4	TEOM	PM10	21	35	47	93	18-Feb-94	24	83		London
1995	Greenwich 4 - Eltham	background	LAQN	543982 17	543982	174676	-999	-999	-999	-999	GR4	TEOM	PM10	23	39	54	85	1-Nov-95	38	89		London
1996	Greenwich 4 - Eltham	background	LAQN	543982 17	543982	174676	-999	-999	-999	-999	GR4	TEOM	PM10	23	39	64	86	19-Mar-96	39	89		London
1997	Greenwich 4 - Eltham	background	LAQN	543982 17	543982	174676	-999	-999	-999	-999	GR4	TEOM	PM10	21	33	51	80	2-Nov-97	24	91		London
1998	Greenwich 4 - Eltham	background	LAQN	543982 17	543982	174676	-999	-999	-999	-999	GR4	TEOM	PM10	18	26	38	53	16-Mar-98	6	86		London
1999	Greenwich 4 - Eltham	background	LAQN	543982 17	543982	174676	-999	-999	-999	-999	GR4	TEOM	PM10	17	27	36	50	1-Apr-99	5	98		London
2000	Greenwich 4 - Eltham	background	LAQN	543982 17	543982	174676	-999	-999	-999	-999	GR4	TEOM	PM10	15	23	32	46	4-Nov-00	2	93		London
2001	Greenwich 4 - Eltham	background	LAQN	543982 17	543982	174676	-999	-999	-999	-999	GR4	TEOM	PM10	18	27	41	80	3-Nov-01	11	97		London
2002	Greenwich 4 - Eltham	background	LAQN	543982 17	543982	174676	-999	-999	-999	-999	GR4	TEOM	PM10	18	27	35	53	5-Apr-02	6	95		London
2003	Greenwich 4 - Eltham	background	LAQN	543982 17	543982	174676	-999	-999	-999	-999	GR4	TEOM	PM10	21	36	53	60	26-Mar-03	24	91		London
1996	Greenwich 5 - Trafalgar	roadside	LAQN	538960 17	538960	177960	NULL	NULL	NULL	NULL	GR5	TEOM	PM10	22	14	14	37	28-Nov-96	0	1		London
1997	Greenwich 5 - Trafalgar	roadside	LAQN	538960 17	538960	177960	NULL	NULL	NULL	NULL	GR5	TEOM	PM10	25	38	59	83	30-Oct-97	33	90		London
1998	Greenwich 5 - Trafalgar	roadside	LAQN	538960 17	538960	177960	NULL	NULL	NULL	NULL	GR5	TEOM	PM10	22	34	45	62	14-May-98	25	82		London
1999	Greenwich 5 - Trafalgar	roadside	LAQN	538960 17	538960	177960	NULL	NULL	NULL	NULL	GR5	TEOM	PM10	20	27	40	60	4-May-99	9	45		London
2000	Greenwich 5 - Trafalgar	roadside	LAQN	538960 17	538960	177960	NULL	NULL	NULL	NULL	GR5	TEOM	PM10	20	33	42	64	23-Mar-00	12	97		London
2001	Greenwich 5 - Trafalgar	roadside	LAQN	538960 17	538960	177960	NULL	NULL	NULL	NULL	GR5	TEOM	PM10	21	33	51	75	3-Nov-01	18	98		London
2002	Greenwich 5 - Trafalgar	roadside	LAQN	538960 17	538960	177960	NULL	NULL	NULL	NULL	GR5	TEOM	PM10	21	31	41	54	5-Apr-02	13	95		London
2003	Greenwich 5 - Trafalgar	roadside	LAQN	538960 17	538960	177960	NULL	NULL	NULL	NULL	GR5	TEOM	PM10	23	38	56	63	22-Feb-03	33	90		London
2002	Greenwich 7 - Blackhea	roadside	LAQN	538145 17	538145	176706	NULL	NULL	NULL	NULL	GR7	TEOM	PM10	27	40	57	124	28-Aug-02	43	81		London
2003	Greenwich 7 - Blackhea	roadside	LAQN	538145 17	538145	176706	NULL	NULL	NULL	NULL	GR7	TEOM	PM10	27	44	60	68	26-Mar-03	55	84		London
2000	Greenwich Bexley 6 - A	roadside	LAQN	544980 17	544980	175080	NULL	2	3	NULL	GB6	TEOM	PM10	18	17	27	44	4-Nov-00	2	20		London
2001	Greenwich Bexley 6 - A	roadside	LAQN	544980 17	544980	175080	NULL	2	3	NULL	GB6	TEOM	PM10	23	37	55	71	3-Nov-01	32	96		London
2002	Greenwich Bexley 6 - A	roadside	LAQN	544980 17	544980	175080	NULL	2	3	NULL	GB6	TEOM	PM10	21	32	48	58	4-Apr-02	19	99		London
2003	Greenwich Bexley 6 - A	roadside	LAQN	544980 17	544980	175080	NULL	2	3	NULL	GB6	TEOM	PM10	25	46	65	75	25-Feb-03	48	86		London
2001	Greenwich Bexley 6 - A	roadside	LAQN	544980 17	544980	175080	NULL	2	3	NULL	GB6	TEOM	PM2.5	15	23	32	65	3-Nov-01	***	83		London
2002	Greenwich Bexley 6 - A	roadside	LAQN	544980 17	544980	175080	NULL	2	3	NULL	GB6	TEOM	PM2.5	14	23	32	45	10-May-02	***	100		London
2003	Greenwich Bexley 6 - A	roadside	LAQN	544980 17	544980	175080	NULL	2	3	NULL	GB6	TEOM	PM2.5	16	29	40	49	11-Aug-03	***	90		London
1997	Hackney 4 - Clapton	background	LAQN	534910 18	534910	186220	-999	-999	-999	-999	HK4	TEOM	PM2.5	16	25	32	60	30-Oct-97	***	62		London
1998	Hackney 4 - Clapton	background	LAQN	534910 18	534910	186220	-999	-999	-999	-999	HK4	TEOM	PM2.5	15	26	36	44	23-Nov-98	***	98		London
1999	Hackney 4 - Clapton	background	LAQN	534910 18	534910	186220	-999	-999	-999	-999	HK4	TEOM	PM2.5	17	25	36	43	11-Sep-99	***	95		London
2000	Hackney 4 - Clapton	background	LAQN	534910 18	534910	186220	-999	-999	-999	-999	HK4	TEOM	PM2.5	16	26	36	50	15-Nov-00	***	98		London
2001	Hackney 4 - Clapton	background	LAQN	534910 18	534910	186220	-999	-999	-999	-999	HK4	TEOM	PM2.5	17	26	37	58	4-Nov-01	***	92		London
2002	Hackney 4 - Clapton	background	LAQN	534910 18	534910	186220	-999	-999	-999	-999	HK4	TEOM	PM2.5	17	26	38	52	27-Sep-02	***	85		London
2003	Hackney 4 - Clapton	background	LAQN	534910 18	534910	186220	-999	-999	-999	-999	HK4	TEOM	PM2.5	18	30	45	483	9-Oct-03	***	65		London
2000	Hackney 6 - Old Street	roadside	LAQN	532900 18	532900	182600	NULL	NULL	NULL	NULL	HK6	TEOM	PM10	25	15	23	40	9-Jun-00	1	4		London
2002	Hackney 6 - Old Street	roadside	LAQN	532900 18	532900	182600	NULL	NULL	NULL	NULL	HK6	TEOM	PM10	31	15	36	65	25-Nov-02	7	8		London
2003	Hackney 6 - Old Street	roadside	LAQN	532900 18	532900	182600	NULL	NULL	NULL	NULL	HK6	TEOM	PM10	28	43	59	70	7-Apr-03	46	74		London
1994	Haringey 1 - Town Hall	roadside	LAQN	533890 19	533890	190638	NULL	NULL	NULL	NULL	HG1	TEOM	PM10	24	7	26	51	15-Dec-94	3	4		London
1995	Haringey 1 - Town Hall	roadside	LAQN	533890 19	533890	190638	NULL	NULL	NULL	NULL	HG1	TEOM	PM10	26	43	61	120	5-Nov-95	55	92		London
1996	Haringey 1 - Town Hall	roadside	LAQN	533890 19	533890	190638	NULL	NULL	NULL	NULL	HG1	TEOM	PM10	29	49	68	88	18-Mar-96	70	94		London
1997	Haringey 1 - Town Hall	roadside	LAQN	533890 19	533890	190638	NULL	NULL	NULL	NULL	HG1	TEOM	PM10	26	42	62	89	11-Jan-97	50	97		London
1998	Haringey 1 - Town Hall	roadside	LAQN	533890 19	533890	190638	NULL	NULL	NULL	NULL	HG1	TEOM	PM10	22	36	48	56	23-Nov-98	24	93		London
1999	Haringey 1 - Town Hall	roadside	LAQN	533890 19	533890	190638	NULL	NULL	NULL	NULL	HG1	TEOM	PM10	22	32	46	55	1-Apr-99	17	97		London

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2000	Haringey 1 - Town Hall	roadside	LAQN	533890 19	533890	190638	NULL	NULL	NULL	NULL	HG1	TEOM	PM10	20	29	41	55	22-Mar-00	12	94		London
2001	Haringey 1 - Town Hall	roadside	LAQN	533890 19	533890	190638	NULL	NULL	NULL	NULL	HG1	TEOM	PM10	21	32	44	65	3-Nov-01	14	99		London
2002	Haringey 1 - Town Hall	roadside	LAQN	533890 19	533890	190638	NULL	NULL	NULL	NULL	HG1	TEOM	PM10	21	30	47	56	5-Nov-02	15	99		London
2003	Haringey 1 - Town Hall	roadside	LAQN	533890 19	533890	190638	NULL	NULL	NULL	NULL	HG1	TEOM	PM10	22	36	50	56	22-Feb-03	31	90		London
1999	Haringey 2 - Priory Park	suburban	LAQN	529850 18	529850	189150	-999	-999	-999	-999	HG2	BAM	PM10	29	42	63	141	11-Sep-99	22	51		London
2000	Haringey 2 - Priory Park	suburban	LAQN	529850 18	529850	189150	-999	-999	-999	-999	HG2	BAM	PM10	27	47	66	110	23-Mar-00	30	94		London
2001	Haringey 2 - Priory Park	suburban	LAQN	529850 18	529850	189150	-999	-999	-999	-999	HG2	BAM	PM10	30	46	78	181	3-Jul-01	28	91		London
2002	Haringey 2 - Priory Park	suburban	LAQN	529850 18	529850	189150	-999	-999	-999	-999	HG2	BAM	PM10	31	48	65	93	30-Jul-02	30	88		London
2003	Haringey 2 - Priory Park	suburban	LAQN	529850 18	529850	189150	-999	-999	-999	-999	HG2	BAM	PM10	35	61	83	110	29-Mar-03	65	87		London
1999	Haringey 3 - Bounds Gr	roadside	LAQN	530050 19	530050	191350	NULL	NULL	NULL	NULL	HG3	BAM	PM10	36	51	75	127	11-Sep-99	38	59		London
2000	Haringey 3 - Bounds Gr	roadside	LAQN	530050 19	530050	191350	NULL	NULL	NULL	NULL	HG3	BAM	PM10	35	57	76	128	23-Mar-00	51	96		London
2001	Haringey 3 - Bounds Gr	roadside	LAQN	530050 19	530050	191350	NULL	NULL	NULL	NULL	HG3	BAM	PM10	37	29	66	88	18-Jan-01	13	18		London
1999	Harrow 1 - Stanmore Ba	background	LAQN	518018 19	518018	192250	-999	-999	-999	-999	HR1	TEOM	PM10	16	22	33	47	11-Sep-99	3	56		London
2000	Harrow 1 - Stanmore Ba	background	LAQN	518018 19	518018	192250	-999	-999	-999	-999	HR1	TEOM	PM10	16	25	31	50	4-Nov-00	3	93		London
2001	Harrow 1 - Stanmore Ba	background	LAQN	518018 19	518018	192250	-999	-999	-999	-999	HR1	TEOM	PM10	16	25	34	45	13-Oct-01	6	99		London
2002	Harrow 1 - Stanmore Ba	background	LAQN	518018 19	518018	192250	-999	-999	-999	-999	HR1	TEOM	PM10	18	26	40	104	12-Sep-02	8	99		London
2003	Harrow 1 - Stanmore Ba	background	LAQN	518018 19	518018	192250	-999	-999	-999	-999	HR1	TEOM	PM10	18	30	44	55	11-Aug-03	16	85		London
2003	Harrow 2 - North Harrow	roadside	LAQN	513508 18	513508	189000	NULL	NULL	NULL	NULL	HR2	TEOM	PM10	23	29	45	57	10-Aug-03	14	45		London
1998	Havering 2 - Harold Hill	suburban	LAQN	554200 19	554200	192100	-999	-999	-999	-999	HV2	BAM	PM10	28	43	70	89	23-Sep-98	26	69		London
1999	Havering 2 - Harold Hill	suburban	LAQN	554200 19	554200	192100	-999	-999	-999	-999	HV2	BAM	PM10	38	59	82	110	1-Apr-99	60	79		London
2000	Havering 2 - Harold Hill	suburban	LAQN	554200 19	554200	192100	-999	-999	-999	-999	HV2	BAM	PM10	40	58	81	168	4-Nov-00	55	78		London
1998	Havering 3 - Romford	roadside	LAQN	551250 18	551250	188250	NULL	NULL	NULL	NULL	HV3	TEOM	PM10	16	8	9	26	30-Dec-98	0	1		London
1999	Havering 3 - Romford	roadside	LAQN	551250 18	551250	188250	NULL	NULL	NULL	NULL	HV3	TEOM	PM10	22	34	47	53	12-Mar-99	22	98		London
2000	Havering 3 - Romford	roadside	LAQN	551250 18	551250	188250	NULL	NULL	NULL	NULL	HV3	TEOM	PM10	22	38	50	59	23-Mar-00	31	96		London
2001	Havering 3 - Romford	roadside	LAQN	551250 18	551250	188250	NULL	NULL	NULL	NULL	HV3	TEOM	PM10	20	30	46	112	3-Nov-01	13	97		London
2002	Havering 3 - Romford	roadside	LAQN	551250 18	551250	188250	NULL	NULL	NULL	NULL	HV3	TEOM	PM10	19	28	38	48	1-Jan-02	7	97		London
2003	Havering 3 - Romford	roadside	LAQN	551250 18	551250	188250	NULL	NULL	NULL	NULL	HV3	TEOM	PM10	21	36	47	55	21-Feb-03	24	77		London
1999	Hillingdon 1 - South Rui	roadside	LAQN	510770 18	510770	184960	NULL	NULL	NULL	NULL	HI1	TEOM	PM10	18	18	31	51	20-Dec-99	2	27		London
2000	Hillingdon 1 - South Rui	roadside	LAQN	510770 18	510770	184960	NULL	NULL	NULL	NULL	HI1	TEOM	PM10	21	32	43	57	23-Mar-00	16	93		London
2001	Hillingdon 1 - South Rui	roadside	LAQN	510770 18	510770	184960	NULL	NULL	NULL	NULL	HI1	TEOM	PM10	22	33	47	74	18-Jan-01	17	93		London
2002	Hillingdon 1 - South Rui	roadside	LAQN	510770 18	510770	184960	NULL	NULL	NULL	NULL	HI1	TEOM	PM10	22	32	46	80	5-Nov-02	16	96		London
2003	Hillingdon 1 - South Rui	roadside	LAQN	510770 18	510770	184960	NULL	NULL	NULL	NULL	HI1	TEOM	PM10	24	39	55	60	10-Aug-03	36	86		London
2002	Hillingdon 2 - Hillingdon	roadside	LAQN	506991 18	506991	181951	NULL	NULL	NULL	NULL	HI2	TEOM	PM10	28	21	24	47	25-Sep-02	2	2		London
2003	Hillingdon 2 - Hillingdon	roadside	LAQN	506991 18	506991	181951	NULL	NULL	NULL	NULL	HI2	TEOM	PM10	25	32	48	82	17-Jun-03	16	52		London
1999	Hounslow 2 - Cranford	suburban	LAQN	510300 17	510300	177200	-999	-999	-999	-999	HS2	TEOM	PM10	17	29	37	44	11-Sep-99	5	93		London
2000	Hounslow 2 - Cranford	suburban	LAQN	510300 17	510300	177200	-999	-999	-999	-999	HS2	TEOM	PM10	17	28	36	42	23-Mar-00	4	98		London
2001	Hounslow 2 - Cranford	suburban	LAQN	510300 17	510300	177200	-999	-999	-999	-999	HS2	TEOM	PM10	18	28	41	53	17-Jan-01	12	96		London
2002	Hounslow 2 - Cranford	suburban	LAQN	510300 17	510300	177200	-999	-999	-999	-999	HS2	TEOM	PM10	18	26	38	66	4-Nov-02	7	92		London
2003	Hounslow 2 - Cranford	suburban	LAQN	510300 17	510300	177200	-999	-999	-999	-999	HS2	TEOM	PM10	20	33	47	59	22-Feb-03	20	88		London
1999	Hounslow 3 - Brentford	roadside	LAQN	517500 17	517500	178100	NULL	NULL	NULL	NULL	HS3	TEOM	PM10	26	36	47	52	25-Aug-99	21	73		London
2000	Hounslow 3 - Brentford	roadside	LAQN	517500 17	517500	178100	NULL	NULL	NULL	NULL	HS3	TEOM	PM10	26	40	50	63	4-Nov-00	41	100		London
2001	Hounslow 3 - Brentford	roadside	LAQN	517500 17	517500	178100	NULL	NULL	NULL	NULL	HS3	TEOM	PM10	28	41	60	93	3-Nov-01	47	98		London
2002	Hounslow 3 - Brentford	roadside	LAQN	517500 17	517500	178100	NULL	NULL	NULL	NULL	HS3	TEOM	PM10	26	37	47	61	26-Sep-02	27	86		London
1999	Hounslow 4 - Chiswick	roadside	LAQN	521070 17	521070	178480	NULL	NULL	NULL	NULL	HS4	TEOM	PM10	23	25	45	54	29-Oct-99	9	28		London
2000	Hounslow 4 - Chiswick	roadside	LAQN	521070 17	521070	178480	NULL	NULL	NULL	NULL	HS4	TEOM	PM10	23	33	44	63	23-Mar-00	19	78		London
2001	Hounslow 4 - Chiswick	roadside	LAQN	521070 17	521070	178480	NULL	NULL	NULL	NULL	HS4	TEOM	PM10	25	37	56	96	3-Nov-01	28	93		London
2002	Hounslow 4 - Chiswick	roadside	LAQN	521070 17	521070	178480	NULL	NULL	NULL	NULL	HS4	TEOM	PM10	25	34	46	58	5-Apr-02	21	71		London
2003	Hounslow 4 - Chiswick	roadside	LAQN	521070 17	521070	178480	NULL	NULL	NULL	NULL	HS4	TEOM	PM10	28	43	59	65	8-Aug-03	47	67		London
2003	Hounslow 5 - Brentford	roadside	LAQN	517425 57	517425	178074	NULL	NULL	NULL	NULL	HS5	TEOM	PM10	28	34	52	71	8-Aug-03	22	47		London
1999	H'smith and Fulham 1 -	roadside	LAQN	523420 17	523420	178590	NULL	NULL	NULL	NULL	HF1	TEOM	PM10	28	32	46	59	29-Oct-99	16	35		London
2000	H'smith and Fulham 1 -	roadside	LAQN	523420 17	523420	178590	NULL	NULL	NULL	NULL	HF1	TEOM	PM10	26	38	48	62	23-Mar-00	33	89		London
2001	H'smith and Fulham 1 -	roadside	LAQN	523420 17	523420	178590	NULL	NULL	NULL	NULL	HF1	TEOM	PM10	27	40	53	85	3-Nov-01	39	92		London
2002	H'smith and Fulham 1 -	roadside	LAQN	523420 17	523420	178590	NULL	NULL	NULL	NULL	HF1	TEOM	PM10	27	37	48	53	20-Apr-02	28	83		London
2003	H'smith and Fulham 1 -	roadside	LAQN	523420 17	523420	178590	NULL	NULL	NULL	NULL	HF1	TEOM	PM10	29	41	56	67	9-Aug-03	51	78		London
2003	H'smith and Fulham 2 -	background	LAQN	523625 17	523625	179010	-999	-999	-999	-999	HF2	TEOM	PM10	27	19	19	29	14-Oct-03	0	0		London
1999	Islington 1 - Upper Stree	background	LAQN	531635 18	531635	184130	-999	-999	-999	-999	IS1	TEOM	PM10	19	19	33	53	1-Apr-99	3	28		London
2000	Islington 1 - Upper Stree	background	LAQN	531635 18	531635	184130	-999	-999	-999	-999	IS1	TEOM	PM10	20	31	39	57	22-Mar-00	8	99		London
2001	Islington 1 - Upper Stree	background	LAQN	531635 18	531635	184130	-999	-999	-999	-999	IS1	TEOM	PM10	19	30	44	65	4-Nov-01	12	99		London
2002	Islington 1 - Upper Stree	background	LAQN	531635 18	531635	184130	-999	-999	-999	-999	IS1	TEOM	PM10	21	30	44	128	9-Oct-02	15	99		London
2003	Islington 1 - Upper Stree	background	LAQN	531635 18	531635	184130	-999	-999	-999	-999	IS1	TEOM	PM10	21	35	49	55	8-Aug-03	25	91		London
2000	Islington 2 - Holloway R	roadside	LAQN	530583 18	530583	185925	NULL	NULL	NULL	NULL	IS2	TEOM	PM10	25	30	36	49	4-Nov-00	3	45		London

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2001	Islington 2 - Holloway R	roadside	LAQN	530583 18	530583	185925	NULL	NULL	NULL	NULL	IS2	TEOM	PM10	26	37	52	65	4-Nov-01	32	96		London
2002	Islington 2 - Holloway R	roadside	LAQN	530583 18	530583	185925	NULL	NULL	NULL	NULL	IS2	TEOM	PM10	27	36	47	61	1-Oct-02	27	84		London
2003	Islington 2 - Holloway R	roadside	LAQN	530583 18	530583	185925	NULL	NULL	NULL	NULL	IS2	TEOM	PM10	30	43	58	63	6-Aug-03	53	77		London
1995	Kens and Chelsea 1 - N	background	LAQN	524040 18	524040	181740	-999	-999	-999	-999	KC1	TEOM	PM10	26	40	57	69	3-May-95	39	69		London
1996	Kens and Chelsea 1 - N	background	LAQN	524040 18	524040	181740	-999	-999	-999	-999	KC1	TEOM	PM10	25	44	68	102	19-Mar-96	49	98		London
1997	Kens and Chelsea 1 - N	background	LAQN	524040 18	524040	181740	-999	-999	-999	-999	KC1	TEOM	PM10	24	38	64	89	30-Oct-97	35	98		London
1998	Kens and Chelsea 1 - N	background	LAQN	524040 18	524040	181740	-999	-999	-999	-999	KC1	TEOM	PM10	20	33	46	54	16-Mar-98	16	98		London
1999	Kens and Chelsea 1 - N	background	LAQN	524040 18	524040	181740	-999	-999	-999	-999	KC1	TEOM	PM10	20	32	45	58	11-Sep-99	16	99		London
2000	Kens and Chelsea 1 - N	background	LAQN	524040 18	524040	181740	-999	-999	-999	-999	KC1	TEOM	PM10	19	31	41	57	23-Mar-00	11	96		London
2001	Kens and Chelsea 1 - N	background	LAQN	524040 18	524040	181740	-999	-999	-999	-999	KC1	TEOM	PM10	20	30	48	75	4-Nov-01	16	96		London
2002	Kens and Chelsea 1 - N	background	LAQN	524040 18	524040	181740	-999	-999	-999	-999	KC1	TEOM	PM10	19	29	40	48	5-Apr-02	8	98		London
2003	Kens and Chelsea 1 - N	background	LAQN	524040 18	524040	181740	-999	-999	-999	-999	KC1	TEOM	PM10	22	36	52	59	8-Aug-03	29	90		London
1998	Kens and Chelsea 2 - C	roadside	LAQN	526510 17	526510	178970	NULL	NULL	NULL	NULL	KC2	TEOM	PM10	28	36	48	60	24-Sep-98	28	60		London
1999	Kens and Chelsea 2 - C	roadside	LAQN	526510 17	526510	178970	NULL	NULL	NULL	NULL	KC2	TEOM	PM10	30	42	55	162	16-Jun-99	51	99		London
2000	Kens and Chelsea 2 - C	roadside	LAQN	526510 17	526510	178970	NULL	NULL	NULL	NULL	KC2	TEOM	PM10	27	37	50	63	13-Mar-00	31	98		London
2001	Kens and Chelsea 2 - C	roadside	LAQN	526510 17	526510	178970	NULL	NULL	NULL	NULL	KC2	TEOM	PM10	27	38	50	67	3-Nov-01	34	99		London
2002	Kens and Chelsea 2 - C	roadside	LAQN	526510 17	526510	178970	NULL	NULL	NULL	NULL	KC2	TEOM	PM10	28	39	49	60	5-Apr-02	37	95		London
2003	Kens and Chelsea 2 - C	roadside	LAQN	526510 17	526510	178970	NULL	NULL	NULL	NULL	KC2	TEOM	PM10	30	44	58	66	22-Feb-03	57	81		London
1996	Kingston 2 - Town Cent	roadside	LAQN	518200 16	518200	169750	NULL	NULL	NULL	NULL	KT2	TEOM	PM10	26	35	49	64	14-Nov-96	20	67		London
1997	Kingston 2 - Town Cent	roadside	LAQN	518200 16	518200	169750	NULL	NULL	NULL	NULL	KT2	TEOM	PM10	27	42	63	81	30-Oct-97	47	99		London
1998	Kingston 2 - Town Cent	roadside	LAQN	518200 16	518200	169750	NULL	NULL	NULL	NULL	KT2	TEOM	PM10	23	37	49	79	14-May-98	29	96		London
1999	Kingston 2 - Town Cent	roadside	LAQN	518200 16	518200	169750	NULL	NULL	NULL	NULL	KT2	TEOM	PM10	22	35	40	48	31-Jul-99	16	99		London
2000	Kingston 2 - Town Cent	roadside	LAQN	518200 16	518200	169750	NULL	NULL	NULL	NULL	KT2	TEOM	PM10	21	28	40	44	25-Jan-00	8	47		London
2000	Lambeth 1 - Christchur	roadside	LAQN	530658 17	530658	173339	NULL	NULL	NULL	NULL	LB1	BAM	PM10	33	33	46	60	21-Dec-00	6	24		London
2001	Lambeth 1 - Christchur	roadside	LAQN	530658 17	530658	173339	NULL	NULL	NULL	NULL	LB1	BAM	PM10	37	56	86	110	19-Jan-01	55	95		London
2002	Lambeth 1 - Christchur	roadside	LAQN	530658 17	530658	173339	NULL	NULL	NULL	NULL	LB1	BAM	PM10	38	55	77	99	30-Jul-02	59	96		London
2003	Lambeth 1 - Christchur	roadside	LAQN	530658 17	530658	173339	NULL	NULL	NULL	NULL	LB1	BAM	PM10	48	82	109	154	29-Mar-03	118	86		London
2001	Lambeth 2 - Vauxhall C	roadside	LAQN	530275 17	530275	177887	NULL	NULL	NULL	NULL	LB2	BAM	PM10	36	15	22	139	19-Dec-01	2	3		London
2002	Lambeth 2 - Vauxhall C	roadside	LAQN	530275 17	530275	177887	NULL	NULL	NULL	NULL	LB2	BAM	PM10	38	61	84	96	4-Apr-02	67	95		London
2003	Lambeth 2 - Vauxhall C	roadside	LAQN	530275 17	530275	177887	NULL	NULL	NULL	NULL	LB2	BAM	PM10	47	65	106	140	17-Mar-03	52	44		London
2001	Lambeth 3 - Loughboro	background	LAQN	532057 17	532057	175690	-999	-999	-999	-999	LB3	BAM	PM10	30	11	19	126	19-Dec-01	2	3		London
2002	Lambeth 3 - Loughboro	background	LAQN	532057 17	532057	175690	-999	-999	-999	-999	LB3	BAM	PM10	31	50	68	92	4-Apr-02	36	97		London
2003	Lambeth 3 - Loughboro	background	LAQN	532057 17	532057	175690	-999	-999	-999	-999	LB3	BAM	PM10	38	63	90	149	29-Mar-03	69	89		London
2002	Lewisham 2 - New Cros	roadside	LAQN	536246 17	536246	176922	NULL	NULL	NULL	NULL	LW2	TEOM	PM10	23	32	46	60	10-May-02	17	55		London
2003	Lewisham 2 - New Cros	roadside	LAQN	536246 17	536246	176922	NULL	NULL	NULL	NULL	LW2	TEOM	PM10	29	46	65	89	27-Nov-03	48	69		London
1997	Mole Valley 2 - Lower A	suburban	LAQN	518250 15	518250	157750	-999	-999	-999	-999	MV2	TEOM	PM10	19	26	38	70	2-Nov-97	7	67		London
1998	Mole Valley 2 - Lower A	suburban	LAQN	518250 15	518250	157750	-999	-999	-999	-999	MV2	TEOM	PM10	17	27	39	54	16-Mar-98	8	98		London
1999	Mole Valley 2 - Lower A	suburban	LAQN	518250 15	518250	157750	-999	-999	-999	-999	MV2	TEOM	PM10	17	26	31	44	11-Sep-99	1	99		London
2000	Mole Valley 2 - Lower A	suburban	LAQN	518250 15	518250	157750	-999	-999	-999	-999	MV2	TEOM	PM10	16	24	34	45	25-Jan-00	3	91		London
2001	Mole Valley 2 - Lower A	suburban	LAQN	518250 15	518250	157750	-999	-999	-999	-999	MV2	TEOM	PM10	17	26	36	61	17-Jan-01	5	99		London
2002	Mole Valley 2 - Lower A	suburban	LAQN	518250 15	518250	157750	-999	-999	-999	-999	MV2	TEOM	PM10	17	26	36	48	5-Apr-02	4	97		London
2003	Mole Valley 2 - Lower A	suburban	LAQN	518250 15	518250	157750	-999	-999	-999	-999	MV2	TEOM	PM10	19	30	45	56	20-Apr-03	14	91		London
2001	Mole Valley 3 - Dorking	background	LAQN	517250 14	517250	149750	-999	-999	-999	-999	MV3	TEOM	PM10	18	19	27	42	5-Nov-01	2	20		London
2002	Mole Valley 3 - Dorking	background	LAQN	517250 14	517250	149750	-999	-999	-999	-999	MV3	TEOM	PM10	17	25	35	50	5-Apr-02	3	99		London
2003	Mole Valley 3 - Dorking	background	LAQN	517250 14	517250	149750	-999	-999	-999	-999	MV3	TEOM	PM10	20	32	47	56	29-Mar-03	22	87		London
1999	Redbridge 1 - Perth Ter	background	LAQN	544380 18	544380	187648	-999	-999	-999	-999	RB1	BAM	PM10	16	8	23	63	20-Dec-99	1	7		London
2000	Redbridge 1 - Perth Ter	background	LAQN	544380 18	544380	187648	-999	-999	-999	-999	RB1	BAM	PM10	26	43	63	101	23-Mar-00	21	94		London
2001	Redbridge 1 - Perth Ter	background	LAQN	544380 18	544380	187648	-999	-999	-999	-999	RB1	BAM	PM10	31	49	71	114	3-Nov-01	35	97		London
2002	Redbridge 1 - Perth Ter	background	LAQN	544380 18	544380	187648	-999	-999	-999	-999	RB1	BAM	PM10	32	51	67	90	4-Apr-02	37	95		London
2003	Redbridge 1 - Perth Ter	background	LAQN	544380 18	544380	187648	-999	-999	-999	-999	RB1	BAM	PM10	36	59	80	125	29-Mar-03	65	86		London
1999	Redbridge 3 - Fullwell C	kerbside	LAQN	544561 19	544561	190401	NULL	NULL	NULL	NULL	RB3	BAM	PM10	25	14	31	69	20-Dec-99	2	11		London
2000	Redbridge 3 - Fullwell C	kerbside	LAQN	544561 19	544561	190401	NULL	NULL	NULL	NULL	RB3	BAM	PM10	34	55	81	110	23-Mar-00	49	97		London
2001	Redbridge 3 - Fullwell C	kerbside	LAQN	544561 19	544561	190401	NULL	NULL	NULL	NULL	RB3	BAM	PM10	35	55	79	93	19-Jan-01	46	96		London
2002	Redbridge 3 - Fullwell C	kerbside	LAQN	544561 19	544561	190401	NULL	NULL	NULL	NULL	RB3	BAM	PM10	36	55	72	92	10-May-02	64	97		London
2003	Redbridge 3 - Fullwell C	kerbside	LAQN	544561 19	544561	190401	NULL	NULL	NULL	NULL	RB3	BAM	PM10	42	63	85	102	22-Feb-03	83	71		London
1999	Redbridge 4 - Gardner	roadside	LAQN	540825 18	540825	188369	NULL	NULL	NULL	NULL	RB4	BAM	PM10	25	15	27	75	20-Dec-99	2	10		London
2000	Redbridge 4 - Gardner	roadside	LAQN	540825 18	540825	188369	NULL	NULL	NULL	NULL	RB4	BAM	PM10	34	51	75	113	23-Mar-00	39	93		London
2001	Redbridge 4 - Gardner	roadside	LAQN	540825 18	540825	188369	NULL	NULL	NULL	NULL	RB4	BAM	PM10	36	54	95	176	26-Aug-01	53	97		London
2002	Redbridge 4 - Gardner	roadside	LAQN	540825 18	540825	188369	NULL	NULL	NULL	NULL	RB4	BAM	PM10	37	55	76	104	30-Jul-02	55	97		London
2003	Redbridge 4 - Gardner	roadside	LAQN	540825 18	540825	188369	NULL	NULL	NULL	NULL	RB4	BAM	PM10	42	71	94	132	29-Mar-03	90	88		London
2003	Redbridge 5 - A406 Wo	roadside	LAQN	539750 19	539750	190250	NULL	NULL	NULL	NULL	RB5	NULL	PM10	38	20	34	60	28-Nov-03	4	4		London

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2000	Reigate and Banstead 1	suburban	LAQN	528206 14	528206	142333	-999	-999	-999	-999	RG1	TEOM	PM10	16	19	25	35	4-Nov-00	0	45		London
2001	Reigate and Banstead 1	suburban	LAQN	528206 14	528206	142333	-999	-999	-999	-999	RG1	TEOM	PM10	18	27	36	56	17-Jan-01	6	99		London
2002	Reigate and Banstead 1	suburban	LAQN	528206 14	528206	142333	-999	-999	-999	-999	RG1	TEOM	PM10	18	26	36	49	5-Apr-02	6	99		London
2003	Reigate and Banstead 1	suburban	LAQN	528206 14	528206	142333	-999	-999	-999	-999	RG1	TEOM	PM10	20	31	46	58	29-Mar-03	16	90		London
2000	Richmond 1 - Castlenau	roadside	LAQN	522473 17	522473	177174	NULL	NULL	NULL	NULL	RI1	TEOM	PM10	18	23	33	43	28-Jun-00	3	53		London
2001	Richmond 1 - Castlenau	roadside	LAQN	522473 17	522473	177174	NULL	NULL	NULL	NULL	RI1	TEOM	PM10	20	30	46	65	3-Nov-01	16	93		London
2002	Richmond 1 - Castlenau	roadside	LAQN	522473 17	522473	177174	NULL	NULL	NULL	NULL	RI1	TEOM	PM10	19	27	37	49	5-Apr-02	5	92		London
2003	Richmond 1 - Castlenau	roadside	LAQN	522473 17	522473	177174	NULL	NULL	NULL	NULL	RI1	TEOM	PM10	22	34	52	66	8-Aug-03	25	87		London
2002	Richmond 11 - Kew	kerbside	LAQN	518094 17	518094	175267	NULL	NULL	NULL	NULL	RIC	TEOM	PM10	23	16	28	41	12-Dec-02	1	11		London
2003	Richmond 11 - Kew	kerbside	LAQN	518094 17	518094	175267	NULL	NULL	NULL	NULL	RIC	TEOM	PM10	28	34	51	205	7-May-03	27	36		London
2003	Richmond 13 - Upper T	roadside	LAQN	518094 17	517250	170250	NULL	NULL	NULL	NULL	RIE	TEOM	PM10	27	32	48	60	11-Aug-03	18	39		London
2001	Richmond 2 - Barnes W	suburban	LAQN	522629 17	522629	176836	-999	-999	-999	-999	RI2	TEOM	PM10	22	18	35	59	14-May-01	4	16		London
2002	Richmond 2 - Barnes W	suburban	LAQN	522629 17	522629	176836	-999	-999	-999	-999	RI2	TEOM	PM10	15	16	28	40	12-Dec-02	1	30		London
2003	Richmond 2 - Barnes W	suburban	LAQN	522629 17	522629	176836	-999	-999	-999	-999	RI2	TEOM	PM10	21	38	55	66	16-Apr-03	33	90		London
2001	Richmond 5 - St. Marga	kerbside	LAQN	NULL	NULL	NULL	NULL	NULL	NULL	NULL	RI5	TEOM	PM10	22	25	36	51	14-May-01	3	34		London
2002	Richmond 7 - Richmond	suburban	LAQN	520690 17	520690	173230	-999	-999	-999	-999	RI7	TEOM	PM10	15	16	24	40	30-Jul-02	1	25		London
2002	Richmond 9 - George S	kerbside	LAQN	517847 17	517847	174890	NULL	NULL	NULL	NULL	RIA	TEOM	PM10	22	19	28	37	11-Oct-02	0	13		London
2002	Southwark 1 - Elephant	background	LAQN	532245 17	532245	178565	-999	-999	-999	-999	SK1	TEOM	PM10	20	18	28	48	12-Dec-02	1	17		London
2003	Southwark 1 - Elephant	background	LAQN	532245 17	532245	178565	-999	-999	-999	-999	SK1	TEOM	PM10	23	37	54	62	15-Apr-03	31	91		London
2001	Southwark 2 - Old Kent	roadside	LAQN	534400 17	534400	177800	NULL	NULL	NULL	NULL	SK2	TEOM	PM10	24	29	40	72	3-Nov-01	8	39		London
2002	Southwark 2 - Old Kent	roadside	LAQN	534400 17	534400	177800	NULL	NULL	NULL	NULL	SK2	TEOM	PM10	26	36	45	66	1-Oct-02	27	71		London
2003	Southwark 2 - Old Kent	roadside	LAQN	534400 17	534400	177800	NULL	NULL	NULL	NULL	SK2	TEOM	PM10	31	46	66	76	21-Feb-03	50	61		London
1995	Sutton 1 - Town Centre	roadside	LAQN	525680 16	525680	164420	NULL	NULL	NULL	NULL	ST1	TEOM	PM10	24	33	46	107	5-Nov-95	29	51		London
1996	Sutton 1 - Town Centre	roadside	LAQN	525680 16	525680	164420	NULL	NULL	NULL	NULL	ST1	TEOM	PM10	27	46	69	96	15-Mar-96	51	92		London
1997	Sutton 1 - Town Centre	roadside	LAQN	525680 16	525680	164420	NULL	NULL	NULL	NULL	ST1	TEOM	PM10	24	38	57	102	2-Nov-97	34	99		London
1998	Sutton 1 - Town Centre	roadside	LAQN	525680 16	525680	164420	NULL	NULL	NULL	NULL	ST1	TEOM	PM10	21	33	42	58	13-May-98	13	98		London
1999	Sutton 1 - Town Centre	roadside	LAQN	525680 16	525680	164420	NULL	NULL	NULL	NULL	ST1	TEOM	PM10	19	30	37	45	11-Sep-99	4	98		London
2000	Sutton 1 - Town Centre	roadside	LAQN	525680 16	525680	164420	NULL	NULL	NULL	NULL	ST1	TEOM	PM10	19	28	39	48	9-May-00	10	92		London
2001	Sutton 1 - Town Centre	roadside	LAQN	525680 16	525680	164420	NULL	NULL	NULL	NULL	ST1	TEOM	PM10	20	31	43	84	17-Jan-01	13	86		London
2002	Sutton 1 - Town Centre	roadside	LAQN	525680 16	525680	164420	NULL	NULL	NULL	NULL	ST1	TEOM	PM10	20	22	34	53	5-Apr-02	4	33		London
2002	Sutton 4 - Wallington	kerbside	LAQN	528926 16	528926	163830	NULL	2	3	NULL	ST4	TEOM	PM10	24	31	39	51	30-Jul-02	8	43		London
2003	Sutton 4 - Wallington	kerbside	LAQN	528926 16	528926	163830	NULL	2	3	NULL	ST4	TEOM	PM10	26	39	55	64	26-Mar-03	36	91		London
2003	Thurrock 3 - Stanford-L	roadside	LAQN	569306 18	569306	182737	NULL	NULL	NULL	NULL	TK3	TEOM	PM10	22	23	35	44	18-Sep-03	6	24		London
1994	Tower Hamlets 1 - Popl	background	LAQN	537510 18	537510	180870	-999	-999	-999	-999	TH1	TEOM	PM10	26	44	56	82	19-Feb-94	54	93		London
1995	Tower Hamlets 1 - Popl	background	LAQN	537510 18	537510	180870	-999	-999	-999	-999	TH1	TEOM	PM10	26	44	66	82	5-May-95	53	94		London
1996	Tower Hamlets 1 - Popl	background	LAQN	537510 18	537510	180870	-999	-999	-999	-999	TH1	TEOM	PM10	27	47	73	92	18-Mar-96	62	98		London
1997	Tower Hamlets 1 - Popl	background	LAQN	537510 18	537510	180870	-999	-999	-999	-999	TH1	TEOM	PM10	25	39	59	88	30-Oct-97	36	88		London
1998	Tower Hamlets 1 - Popl	background	LAQN	537510 18	537510	180870	-999	-999	-999	-999	TH1	TEOM	PM10	21	34	45	55	14-May-98	23	99		London
1999	Tower Hamlets 1 - Popl	background	LAQN	537510 18	537510	180870	-999	-999	-999	-999	TH1	TEOM	PM10	21	31	48	58	1-Apr-99	22	98		London
2000	Tower Hamlets 1 - Popl	background	LAQN	537510 18	537510	180870	-999	-999	-999	-999	TH1	TEOM	PM10	18	29	37	56	22-Mar-00	5	98		London
2001	Tower Hamlets 1 - Popl	background	LAQN	537510 18	537510	180870	-999	-999	-999	-999	TH1	TEOM	PM10	19	29	45	58	4-Nov-01	15	98		London
2002	Tower Hamlets 1 - Popl	background	LAQN	537510 18	537510	180870	-999	-999	-999	-999	TH1	TEOM	PM10	19	29	43	50	5-Apr-02	11	96		London
2003	Tower Hamlets 1 - Popl	background	LAQN	537510 18	537510	180870	-999	-999	-999	-999	TH1	TEOM	PM10	24	40	61	129	15-Apr-03	42	88		London
1999	Tower Hamlets 3 - Beth	background	LAQN	535100 18	535100	182650	-999	-999	-999	-999	TH3	TEOM	PM10	17	15	20	41	20-Dec-99	1	14		London
2000	Tower Hamlets 3 - Beth	background	LAQN	535100 18	535100	182650	-999	-999	-999	-999	TH3	TEOM	PM10	20	30	42	59	22-Mar-00	15	95		London
2001	Tower Hamlets 3 - Beth	background	LAQN	535100 18	535100	182650	-999	-999	-999	-999	TH3	TEOM	PM10	21	32	48	61	4-Nov-01	19	94		London
2002	Tower Hamlets 3 - Beth	background	LAQN	535100 18	535100	182650	-999	-999	-999	-999	TH3	TEOM	PM10	20	30	44	68	3-Apr-02	14	90		London
2003	Tower Hamlets 3 - Beth	background	LAQN	535100 18	535100	182650	-999	-999	-999	-999	TH3	TEOM	PM10	21	35	50	55	10-Aug-03	28	85		London
1998	Waltham Forest 1 - Daw	background	LAQN	538075 18	538075	187030	-999	-999	-999	-999	WL1	TEOM	PM10	19	23	36	47	23-Nov-98	5	47		London
1999	Waltham Forest 1 - Daw	background	LAQN	538075 18	538075	187030	-999	-999	-999	-999	WL1	TEOM	PM10	19	29	40	51	1-Apr-99	12	90		London
2000	Waltham Forest 1 - Daw	background	LAQN	538075 18	538075	187030	-999	-999	-999	-999	WL1	TEOM	PM10	19	28	38	53	6-May-00	7	91		London
2001	Waltham Forest 1 - Daw	background	LAQN	538075 18	538075	187030	-999	-999	-999	-999	WL1	TEOM	PM10	18	26	38	68	4-Nov-01	7	76		London
2002	Waltham Forest 1 - Daw	background	LAQN	538075 18	538075	187030	-999	-999	-999	-999	WL1	TEOM	PM10	19	28	42	62	4-Nov-02	13	98		London
2003	Waltham Forest 1 - Daw	background	LAQN	538075 18	538075	187030	-999	-999	-999	-999	WL1	TEOM	PM10	19	30	43	51	20-Apr-03	16	69		London
1998	Waltham Forest 2 - Mo	roadside	LAQN		0	0	NULL	NULL	NULL	NULL	WL2	TEOM	PM10	22	27	41	49	7-Dec-98	10	42		London
1999	Waltham Forest 2 - Mo	roadside	LAQN		0	0	NULL	NULL	NULL	NULL	WL2	TEOM	PM10	21	30	43	74	2-Jun-99	13	79		London
2000	Waltham Forest 2 - Mo	roadside	LAQN		0	0	NULL	NULL	NULL	NULL	WL2	TEOM	PM10	20	29	42	60	22-Mar-00	12	70		London
2001	Waltham Forest 2 - Mo	roadside	LAQN		0	0	NULL	NULL	NULL	NULL	WL2	TEOM	PM10	20	28	42	51	15-Feb-01	9	60		London
2003	Waltham Forest 3 - Chi	roadside	LAQN	536750 19	536750	193750	NULL	NULL	NULL	NULL	WL3	TEOM	PM10	22	25	44	82	17-Sep-03	12	36		London
1998	Wandsworth 4 - High St	roadside	LAQN	525760 17	525760	174570	NULL	7	3	NULL	WA4	TEOM	PM10	19	28	44	57	24-Sep-98	12	81		London
1999	Wandsworth 4 - High St	roadside	LAQN	525760 17	525760	174570	NULL	7	3	NULL	WA4	TEOM	PM10	20	33	45	52	31-Jul-99	17	99		London

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2000	Wandsworth 4 - High St	roadside	LAQN	525760 17	525760	174570	NULL	7	3	NULL	WA4	TEOM	PM10	21	33	43	149	9-Sep-00	19	100		London
2001	Wandsworth 4 - High St	roadside	LAQN	525760 17	525760	174570	NULL	7	3	NULL	WA4	TEOM	PM10	22	34	49	73	3-Nov-01	28	99		London
2002	Wandsworth 4 - High St	roadside	LAQN	525760 17	525760	174570	NULL	7	3	NULL	WA4	TEOM	PM10	21	33	45	58	5-Apr-02	17	97		London
2003	Wandsworth 4 - High St	roadside	LAQN	525760 17	525760	174570	NULL	7	3	NULL	WA4	TEOM	PM10	25	42	59	72	22-Feb-03	46	89		London
2002	Ashford Roadside	roadside	KAQMN	601178 14	601178	143000	NULL	NULL	NULL	NULL	ZA2	TEOM	PM10	22	33	43	67	10-May-02	19	91		England
2003	Ashford Roadside	roadside	KAQMN	601178 14	601178	143000	NULL	NULL	NULL	NULL	ZA2	TEOM	PM10	23	36	49	60	9-Aug-03	28	91		England
1999	Canterbury Backgrnd -	background	KAQMN	616200 15	616200	157300	-999	-999	-999	-999	ZY1	TEOM	PM10	13	14	21	36	2-Aug-99	0	25		England
2000	Canterbury Backgrnd -	background	KAQMN	616200 15	616200	157300	-999	-999	-999	-999	ZY1	TEOM	PM10	18	9	19	32	24-Dec-00	0	4		England
2001	Canterbury Backgrnd -	background	KAQMN	616200 15	616200	157300	-999	-999	-999	-999	ZY1	TEOM	PM10	18	28	38	56	17-Jan-01	7	97		England
2002	Canterbury Backgrnd -	background	KAQMN	616200 15	616200	157300	-999	-999	-999	-999	ZY1	TEOM	PM10	18	27	39	54	10-May-02	9	99		England
2003	Canterbury Backgrnd -	background	KAQMN	616200 15	616200	157300	-999	-999	-999	-999	ZY1	TEOM	PM10	20	32	46	54	11-Aug-03	19	91		England
1997	Chatham Roadside - A2	roadside	KAQMN	577487 16	577487	166947	NULL	NULL	NULL	NULL	ZC1	TEOM	PM10	22	34	48	60	10-Mar-97	23	94		England
1998	Chatham Roadside - A2	roadside	KAQMN	577487 16	577487	166947	NULL	NULL	NULL	NULL	ZC1	TEOM	PM10	21	30	41	49	13-May-98	15	86		England
1999	Chatham Roadside - A2	roadside	KAQMN	577487 16	577487	166947	NULL	NULL	NULL	NULL	ZC1	TEOM	PM10	19	29	39	50	1-Apr-99	8	90		England
2000	Chatham Roadside - A2	roadside	KAQMN	577487 16	577487	166947	NULL	NULL	NULL	NULL	ZC1	TEOM	PM10	19	23	35	67	15-Jun-00	3	56		England
2001	Chatham Roadside - A2	roadside	KAQMN	577487 16	577487	166947	NULL	NULL	NULL	NULL	ZC1	TEOM	PM10	20	30	46	56	4-Nov-01	14	80		England
2002	Chatham Roadside - A2	roadside	KAQMN	577487 16	577487	166947	NULL	NULL	NULL	NULL	ZC1	TEOM	PM10	20	27	38	48	4-Apr-02	6	63		England
2003	Chatham Roadside - A2	roadside	KAQMN	577487 16	577487	166947	NULL	NULL	NULL	NULL	ZC1	TEOM	PM10	23	34	52	64	25-Feb-03	27	85		England
1999	Dartford Roadside - St	kerbside	KAQMN	558525 17	558525	174709	NULL	NULL	NULL	NULL	ZR1	BAM	PM10	40	51	102	301	11-Sep-99	37	45		England
2000	Dartford Roadside - St	kerbside	KAQMN	558525 17	558525	174709	NULL	NULL	NULL	NULL	ZR1	BAM	PM10	47	71	99	170	23-Mar-00	131	95		England
2001	Dartford Roadside - St	kerbside	KAQMN	558525 17	558525	174709	NULL	NULL	NULL	NULL	ZR1	BAM	PM10	50	82	117	136	17-Jan-01	136	91		England
2002	Dartford Roadside - St	kerbside	KAQMN	558525 17	558525	174709	NULL	NULL	NULL	NULL	ZR1	BAM	PM10	54	80	115	140	11-Nov-02	155	86		England
2003	Dartford Roadside - St	kerbside	KAQMN	558525 17	558525	174709	NULL	NULL	NULL	NULL	ZR1	BAM	PM10	51	89	122	175	26-Mar-03	132	90		England
2002	Dartford Roadside 2 - T	roadside	KAQMN	554117 17	554117	173852	NULL	NULL	NULL	NULL	ZR2	BAM	PM10	38	31	55	76	16-Nov-02	13	14		England
2003	Dartford Roadside 2 - T	roadside	KAQMN	554117 17	554117	173852	NULL	NULL	NULL	NULL	ZR2	BAM	PM10	44	73	106	158	29-Mar-03	94	87		England
2002	Dartford Roadside 3 - B	roadside	KAQMN	558622 17	558622	172752	NULL	NULL	NULL	NULL	ZR3	BAM	PM10	30	19	47	68	30-Dec-02	4	12		England
2003	Dartford Roadside 3 - B	roadside	KAQMN	558622 17	558622	172752	NULL	NULL	NULL	NULL	ZR3	BAM	PM10	40	69	96	147	29-Mar-03	79	89		England
2000	Dover Roadside - Town	roadside	KAQMN	632277 14	632277	141513	NULL	NULL	NULL	NULL	ZD2	TEOM	PM10	25	15	19	45	21-Dec-00	2	3		England
2001	Dover Roadside - Town	roadside	KAQMN	632277 14	632277	141513	NULL	NULL	NULL	NULL	ZD2	TEOM	PM10	27	41	56	67	16-Jan-01	44	91		England
2002	Dover Roadside - Town	roadside	KAQMN	632277 14	632277	141513	NULL	NULL	NULL	NULL	ZD2	TEOM	PM10	28	41	56	65	3-Apr-02	45	99		England
2003	Dover Roadside - Town	roadside	KAQMN	632277 14	632277	141513	NULL	NULL	NULL	NULL	ZD2	TEOM	PM10	31	50	65	75	26-Feb-03	81	89		England
1997	Folkestone Suburban -	suburban	KAQMN	621365 13	621365	136720	-999	-999	-999	-999	ZF1	TEOM	PM10	20	22	31	48	29-Sep-97	3	32		England
1998	Folkestone Suburban -	suburban	KAQMN	621365 13	621365	136720	-999	-999	-999	-999	ZF1	TEOM	PM10	20	28	41	124	27-Feb-98	10	74		England
1999	Folkestone Suburban -	suburban	KAQMN	621365 13	621365	136720	-999	-999	-999	-999	ZF1	TEOM	PM10	21	31	44	60	10-Sep-99	15	95		England
2000	Folkestone Suburban -	suburban	KAQMN	621365 13	621365	136720	-999	-999	-999	-999	ZF1	TEOM	PM10	18	28	37	64	22-Mar-00	5	99		England
2001	Folkestone Suburban -	suburban	KAQMN	621365 13	621365	136720	-999	-999	-999	-999	ZF1	TEOM	PM10	17	26	37	46	24-Aug-01	5	94		England
2002	Folkestone Suburban -	suburban	KAQMN	621365 13	621365	136720	-999	-999	-999	-999	ZF1	TEOM	PM10	18	26	38	52	10-May-02	6	96		England
2003	Folkestone Suburban -	suburban	KAQMN	621365 13	621365	136720	-999	-999	-999	-999	ZF1	TEOM	PM10	19	31	45	61	9-Aug-03	13	91		England
1999	Gravesham Ind Bgd - N	background	KAQMN	562143 17	562143	174387	-999	-999	-999	-999	ZG3	BAM	PM10	49	88	133	189	17-May-99	129	94		England
2000	Gravesham Ind Bgd - N	background	KAQMN	562143 17	562143	174387	-999	-999	-999	-999	ZG3	BAM	PM10	49	84	136	175	11-May-00	115	96		England
2001	Gravesham Ind Bgd - N	background	KAQMN	562143 17	562143	174387	-999	-999	-999	-999	ZG3	BAM	PM10	42	95	130	204	13-Dec-01	100	96		England
2002	Gravesham Ind Bgd - N	background	KAQMN	562143 17	562143	174387	-999	-999	-999	-999	ZG3	BAM	PM10	43	86	138	179	5-Apr-02	108	94		England
2003	Gravesham Ind Bgd - N	background	KAQMN	562143 17	562143	174387	-999	-999	-999	-999	ZG3	BAM	PM10	48	93	122	165	26-Mar-03	126	86		England
1999	Gravesham Roadside -	roadside	KAQMN	562613 17	562613	172075	NULL	NULL	NULL	NULL	ZG2	BAM	PM10	50	74	97	152	11-Sep-99	156	89		England
2000	Gravesham Roadside -	roadside	KAQMN	562613 17	562613	172075	NULL	NULL	NULL	NULL	ZG2	BAM	PM10	47	74	103	144	23-Mar-00	105	98		England
2001	Gravesham Roadside -	roadside	KAQMN	562613 17	562613	172075	NULL	NULL	NULL	NULL	ZG2	BAM	PM10	38	57	78	104	18-Jan-01	56	95		England
2002	Gravesham Roadside -	roadside	KAQMN	562613 17	562613	172075	NULL	NULL	NULL	NULL	ZG2	BAM	PM10	42	61	81	122	4-Apr-02	78	84		England
2003	Gravesham Roadside -	roadside	KAQMN	562613 17	562613	172075	NULL	NULL	NULL	NULL	ZG2	BAM	PM10	44	70	103	146	29-Mar-03	101	86		England
1997	Luton (Kent) Backgroun	background	KAQMN	577257 16	577257	166660	-999	-999	-999	-999	ZL1	TEOM	PM10	18	30	46	63	11-Jan-97	16	95		England
1998	Luton (Kent) Backgroun	background	KAQMN	577257 16	577257	166660	-999	-999	-999	-999	ZL1	TEOM	PM10	14	24	33	46	16-Mar-98	2	96		England
1999	Luton (Kent) Backgroun	background	KAQMN	577257 16	577257	166660	-999	-999	-999	-999	ZL1	TEOM	PM10	14	23	33	38	11-Sep-99	1	96		England
2000	Luton (Kent) Backgroun	background	KAQMN	577257 16	577257	166660	-999	-999	-999	-999	ZL1	TEOM	PM10	16	23	33	50	4-May-00	3	58		England
2001	Luton (Kent) Backgroun	background	KAQMN	577257 16	577257	166660	-999	-999	-999	-999	ZL1	TEOM	PM10	16	22	39	86	4-Apr-01	8	60		England
2002	Luton (Kent) Backgroun	background	KAQMN	577257 16	577257	166660	-999	-999	-999	-999	ZL1	TEOM	PM10	16	25	34	51	10-May-02	3	79		England
2003	Luton (Kent) Backgroun	background	KAQMN	577257 16	577257	166660	-999	-999	-999	-999	ZL1	TEOM	PM10	19	28	45	57	20-Apr-03	15	61		England
1999	Maidstone Roadside - F	kerbside	KAQMN	575740 15	575740	155615	NULL	NULL	NULL	NULL	ZM2	TEOM	PM10	24	31	42	58	6-Aug-99	14	55		England
2000	Maidstone Roadside - F	kerbside	KAQMN	575740 15	575740	155615	NULL	NULL	NULL	NULL	ZM2	TEOM	PM10	23	37	45	59	4-Nov-00	26	98		England
2001	Maidstone Roadside - F	kerbside	KAQMN	575740 15	575740	155615	NULL	NULL	NULL	NULL	ZM2	TEOM	PM10	24	37	53	70	3-Nov-01	24	98		England
2002	Maidstone Roadside - F	kerbside	KAQMN	575740 15	575740	155615	NULL	NULL	NULL	NULL	ZM2	TEOM	PM10	23	34	44	59	10-May-02	18	99		England
2003	Maidstone Roadside - F	kerbside	KAQMN	575740 15	575740	155615	NULL	NULL	NULL	NULL	ZM2	TEOM	PM10	26	39	56	63	26-Mar-03	38	91		England
1999	Maidstone Rural - Detlin	rural	KAQMN	580075 15	580075	159700	-999	-999	-999	-999	ZM3	TEOM	PM10	15	21	30	43	11-Sep-99	1	57		England

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintorin g method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2000	Maidstone Rural - Detlin	rural	KAQMN	580075 15	580075	159700	-999	-999	-999	-999	ZM3	TEOM	PM10	15	23	32	39	23-Mar-00	4	99		England
2001	Maidstone Rural - Detlin	rural	KAQMN	580075 15	580075	159700	-999	-999	-999	-999	ZM3	TEOM	PM10	15	23	35	43	13-Oct-01	3	99		England
2002	Maidstone Rural - Detlin	rural	KAQMN	580075 15	580075	159700	-999	-999	-999	-999	ZM3	TEOM	PM10	17	26	41	161	2-Aug-02	11	99		England
2003	Maidstone Rural - Detlin	rural	KAQMN	580075 15	580075	159700	-999	-999	-999	-999	ZM3	TEOM	PM10	18	28	49	115	15-Jul-03	18	87		England
1998	Sevenoaks 2 - Greatnes	background	KAQMN	553600 15	553600	156800	-999	-999	-999	-999	ZV1	TEOM	PM10	19	29	41	59	13-May-98	11	90		England
1999	Sevenoaks 2 - Greatnes	background	KAQMN	553600 15	553600	156800	-999	-999	-999	-999	ZV1	TEOM	PM10	17	27	35	46	11-Sep-99	3	84		England
2000	Sevenoaks 2 - Greatnes	background	KAQMN	553600 15	553600	156800	-999	-999	-999	-999	ZV1	TEOM	PM10	17	27	35	1056	12-Sep-00	3	91		England
2001	Sevenoaks 2 - Greatnes	background	KAQMN	553600 15	553600	156800	-999	-999	-999	-999	ZV1	TEOM	PM10	17	25	36	58	17-Jan-01	4	98		England
2002	Sevenoaks 2 - Greatnes	background	KAQMN	553600 15	553600	156800	-999	-999	-999	-999	ZV1	TEOM	PM10	17	25	35	58	4-Apr-02	6	98		England
2003	Sevenoaks 2 - Greatnes	background	KAQMN	553600 15	553600	156800	-999	-999	-999	-999	ZV1	TEOM	PM10	18	29	44	59	29-Mar-03	14	91		England
2003	Swale Background	background	KAQMN	591792 17	591792	174876	-999	-999	-999	-999	ZW2	TEOM	PM10	21	21	34	46	10-Oct-03	5	22		England
2003	Swale Roadside	roadside	KAQMN	600201 16	600201	160895	NULL	NULL	NULL	NULL	ZW1	TEOM	PM10	19	18	32	38	10-Sep-03	0	24		England
2001	Thanet Background	background	KAQMN	635435 16	635435	169832	-999	-999	-999	-999	ZH2	APM	PM10	9	14	20	28	5-Mar-01	0	82		England
2002	Thanet Background	background	KAQMN	635435 16	635435	169832	-999	-999	-999	-999	ZH2	APM	PM10	8	13	18	38	10-May-02	1	88		England
2003	Thanet Background	background	KAQMN	635435 16	635435	169832	-999	-999	-999	-999	ZH2	APM	PM10	12	20	33	55	29-Mar-03	5	77		England
2003	Thanet Roadside - Ram	roadside	KAQMN	NULL	638250	638250	NULL	NULL	NULL	NULL	ZH4	BAM	PM10	44	73	111	145	26-Mar-03	81	68		England
2001	Broxbourne Roadside	roadside	HBAPMN	536113 20	536113	200083	NULL	NULL	NULL	NULL	BB1	TEOM	PM10	21	23	32	74	3-Nov-01	3	29		England
2002	Broxbourne Roadside	roadside	HBAPMN	536113 20	536113	200083	NULL	NULL	NULL	NULL	BB1	TEOM	PM10	22	30	42	59	5-Nov-02	15	87		England
2003	Broxbourne Roadside	roadside	HBAPMN	536113 20	536113	200083	NULL	NULL	NULL	NULL	BB1	TEOM	PM10	23	36	50	57	10-Aug-03	29	91		England
1997	Dacorum Background (	background	HBAPMN	505300 20	505300	207500	-999	-999	-999	-999	DC1	TEOM	PM10	19	29	41	52	30-Oct-97	8	67		England
1998	Dacorum Background (	background	HBAPMN	505300 20	505300	207500	-999	-999	-999	-999	DC1	TEOM	PM10	18	30	43	55	13-Feb-98	11	97		England
1999	Dacorum Background (	background	HBAPMN	505300 20	505300	207500	-999	-999	-999	-999	DC1	TEOM	PM10	16	26	33	45	11-Sep-99	2	95		England
2000	Dacorum Background (	background	HBAPMN	505300 20	505300	207500	-999	-999	-999	-999	DC1	TEOM	PM10	16	26	34	45	23-Mar-00	3	96		England
2001	Dacorum Background (	background	HBAPMN	505300 20	505300	207500	-999	-999	-999	-999	DC1	TEOM	PM10	16	24	34	41	13-Oct-01	4	96		England
2002	Dacorum Background (	background	HBAPMN	505300 20	505300	207500	-999	-999	-999	-999	DC1	TEOM	PM10	16	24	38	89	25-Sep-02	6	96		England
2003	Dacorum Background (	background	HBAPMN	505300 20	505300	207500	-999	-999	-999	-999	DC1	TEOM	PM10	19	20	39	53	29-Mar-03	9	25		England
1998	E. Herts Background (S	background	HBAPMN	549600 21	549600	215700	-999	-999	-999	-999	EH2	TEOM	PM10	17	20	32	37	11-Aug-98	0	48		England
1999	E. Herts Background (S	background	HBAPMN	549600 21	549600	215700	-999	-999	-999	-999	EH2	TEOM	PM10	16	25	37	48	11-Sep-99	6	98		England
2000	E. Herts Background (S	background	HBAPMN	549600 21	549600	215700	-999	-999	-999	-999	EH2	TEOM	PM10	16	25	31	52	4-Nov-00	2	99		England
2001	E. Herts Background (S	background	HBAPMN	549600 21	549600	215700	-999	-999	-999	-999	EH2	TEOM	PM10	16	24	34	50	3-Nov-01	2	97		England
2002	E. Herts Background (S	background	HBAPMN	549600 21	549600	215700	-999	-999	-999	-999	EH2	TEOM	PM10	16	25	35	43	10-May-02	4	93		England
2003	E. Herts Background (S	background	HBAPMN	549600 21	549600	215700	-999	-999	-999	-999	EH2	TEOM	PM10	19	30	46	61	5-Jun-03	17	55		England
2001	E. Herts Roadside (Saw	roadside	HBAPMN	548250 21	548250	215400	NULL	NULL	NULL	NULL	EH3	TEOM	PM10	19	18	34	53	3-Nov-01	5	23		England
2002	E. Herts Roadside (Saw	roadside	HBAPMN	548250 21	548250	215400	NULL	NULL	NULL	NULL	EH3	TEOM	PM10	21	30	37	51	3-Dec-02	7	84		England
2003	E. Herts Roadside (Saw	roadside	HBAPMN	548250 21	548250	215400	NULL	NULL	NULL	NULL	EH3	TEOM	PM10	22	33	47	52	8-Aug-03	20	83		England
1999	Hertsmere Background	background	HBAPMN	519400 19	519400	196300	-999	-999	-999	-999	HM1	TEOM	PM10	17	25	35	48	11-Sep-99	2	95		England
2000	Hertsmere Background	background	HBAPMN	519400 19	519400	196300	-999	-999	-999	-999	HM1	TEOM	PM10	16	25	33	47	23-Mar-00	2	99		England
2001	Hertsmere Background	background	HBAPMN	519400 19	519400	196300	-999	-999	-999	-999	HM1	TEOM	PM10	16	26	35	45	13-Oct-01	4	99		England
2002	Hertsmere Background	background	HBAPMN	519400 19	519400	196300	-999	-999	-999	-999	HM1	TEOM	PM10	17	25	38	59	5-Nov-02	7	99		England
2003	Hertsmere Background	background	HBAPMN	519400 19	519400	196300	-999	-999	-999	-999	HM1	TEOM	PM10	19	34	45	54	20-Apr-03	20	88		England
2003	Luton Airport	background	HBAPMN		511850	221340	-999	-999	-999	-999	LA1	BAM	PM10	38	27	47	65	6-Nov-03	9	10		England
1999	Luton Background	background	HBAPMN	506500 22	506500	222700	-999	-999	-999	-999	LN1	TEOM	PM10	19	25	34	42	13-Oct-99	2	66		England
2000	Luton Background	background	HBAPMN	506500 22	506500	222700	-999	-999	-999	-999	LN1	TEOM	PM10	19	27	34	42	13-Mar-00	2	91		England
2001	Luton Background	background	HBAPMN	506500 22	506500	222700	-999	-999	-999	-999	LN1	TEOM	PM10	19	29	38	48	13-Oct-01	6	95		England
2002	Luton Background	background	HBAPMN	506500 22	506500	222700	-999	-999	-999	-999	LN1	TEOM	PM10	19	27	37	46	4-Apr-02	5	95		England
2003	Luton Background	background	HBAPMN	506500 22	506500	222700	-999	-999	-999	-999	LN1	TEOM	PM10	21	33	47	54	20-Apr-03	26	86		England
2002	N. Herts Background 3 (	background	HBAPMN	535250 24	535250	240750	-999	-999	-999	-999	NH3	TEOM	PM10	16	19	27	35	15-Aug-02	0	43		England
2003	N. Herts Background 3 (	background	HBAPMN	535250 24	535250	240750	-999	-999	-999	-999	NH3	TEOM	PM10	17	20	35	83	21-Mar-03	5	36		England
2003	N. Herts Background 4 (	background	HBAPMN	515400 22	515400	222100	-999	-999	-999	-999	NH4	TEOM	PM10	16	21	38	58	10-Aug-03	7	53		England
1999	N. Herts Roadside 1 (B	roadside	HBAPMN	524500 23	524500	233900	NULL	NULL	NULL	NULL	NH1	TEOM	PM10	22	31	42	52	1-Apr-99	12	83		England
2000	N. Herts Roadside 1 (B	roadside	HBAPMN	524500 23	524500	233900	NULL	NULL	NULL	NULL	NH1	TEOM	PM10	21	22	32	46	22-Mar-00	3	24		England
2000	N. Herts Roadside 2 (B	roadside	HBAPMN	523800 23	523800	233500	NULL	NULL	NULL	NULL	NH2	TEOM	PM10	19	24	30	37	19-Jun-00	0	55		England
2001	N. Herts Roadside 2 (B	roadside	HBAPMN	523800 23	523800	233500	NULL	NULL	NULL	NULL	NH2	TEOM	PM10	20	29	37	55	18-Jan-01	5	83		England
2002	N. Herts Roadside 2 (B	roadside	HBAPMN	523800 23	523800	233500	NULL	NULL	NULL	NULL	NH2	TEOM	PM10	22	23	35	47	10-May-02	3	28		England
2000	South Beds Background	background	HBAPMN	501987 22	501987	221730	-999	-999	-999	-999	SB1	TEOM	PM10	17	16	27	38	7-Dec-00	1	18		England
2001	South Beds Background	background	HBAPMN	501987 22	501987	221730	-999	-999	-999	-999	SB1	TEOM	PM10	19	30	40	54	3-Feb-01	10	83		England
2002	South Beds Background	background	HBAPMN	501987 22	501987	221730	-999	-999	-999	-999	SB1	TEOM	PM10	19	30	42	52	10-May-02	17	96		England
2003	South Beds Background	background	HBAPMN	501987 22	501987	221730	-999	-999	-999	-999	SB1	TEOM	PM10	21	34	54	59	8-Aug-03	29	86		England
1996	St. Albans Background	background	HBAPMN	516070 20	516070	207040	-999	-999	-999	-999	SA1	TEOM	PM10	23	27	41	54	20-Aug-96	13	33		England
1997	St. Albans Background	background	HBAPMN	516070 20	516070	207040	-999	-999	-999	-999	SA1	TEOM	PM10	23	35	52	81	30-Oct-97	24	87		England
1998	St. Albans Background	background	HBAPMN	516070 20	516070	207040	-999	-999	-999	-999	SA1	TEOM	PM10	18	26	35	48	23-Nov-98	4	93		England

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
1999	St. Albans Background	background	HBAPMN	516070 20	516070	207040	-999	-999	-999	-999	SA1	TEOM	PM10	19	28	39	52	11-Sep-99	9	93		England
2000	St. Albans Background	background	HBAPMN	516070 20	516070	207040	-999	-999	-999	-999	SA1	TEOM	PM10	20	30	39	53	22-Mar-00	8	96		England
2001	St. Albans Background	background	HBAPMN	516070 20	516070	207040	-999	-999	-999	-999	SA1	TEOM	PM10	19	30	40	56	13-Oct-01	10	96		England
2002	St. Albans Background	background	HBAPMN	516070 20	516070	207040	-999	-999	-999	-999	SA1	TEOM	PM10	20	28	41	52	4-Apr-02	14	80		England
2003	St. Albans Background	background	HBAPMN	516070 20	516070	207040	-999	-999	-999	-999	SA1	TEOM	PM10	22	35	48	64	10-Aug-03	27	80		England
2002	Stevenage Roadside	roadside	HBAPMN	523620 22	523620	223957	NULL	NULL	NULL	NULL	SE1	TEOM	PM10	19	26	39	47	10-May-02	8	85		England
2003	Stevenage Roadside	roadside	HBAPMN	523620 22	523620	223957	NULL	NULL	NULL	NULL	SE1	TEOM	PM10	22	35	48	55	29-Mar-03	24	86		England
1999	Three Rivers Backgrour	background	HBAPMN	505500 19	505500	194400	-999	-999	-999	-999	TR1	TEOM	PM10	17	20	35	43	11-Sep-99	4	35		England
2000	Three Rivers Backgrour	background	HBAPMN	505500 19	505500	194400	-999	-999	-999	-999	TR1	TEOM	PM10	16	27	35	51	23-Mar-00	2	97		England
2001	Three Rivers Backgrour	background	HBAPMN	505500 19	505500	194400	-999	-999	-999	-999	TR1	TEOM	PM10	17	29	39	53	5-Mar-01	9	96		England
2002	Three Rivers Backgrour	background	HBAPMN	505500 19	505500	194400	-999	-999	-999	-999	TR1	TEOM	PM10	17	27	43	213	22-Nov-02	13	93		England
2003	Three Rivers Backgrour	background	HBAPMN	505500 19	505500	194400	-999	-999	-999	-999	TR1	TEOM	PM10	18	32	49	58	22-Feb-03	27	78		England
1998	Watford Roadside	roadside	HBAPMN	510400 19	510400	196800	NULL	NULL	NULL	NULL	WF1	TEOM	PM10	20	8	24	40	7-Dec-98	1	7		England
1999	Watford Roadside	roadside	HBAPMN	510400 19	510400	196800	NULL	NULL	NULL	NULL	WF1	TEOM	PM10	20	29	38	48	1-Apr-99	7	84		England
2000	Watford Roadside	roadside	HBAPMN	510400 19	510400	196800	NULL	NULL	NULL	NULL	WF1	TEOM	PM10	20	31	37	52	23-Mar-00	6	99		England
2001	Watford Roadside	roadside	HBAPMN	510400 19	510400	196800	NULL	NULL	NULL	NULL	WF1	TEOM	PM10	20	31	42	53	13-Oct-01	10	93		England
2002	Watford Roadside	roadside	HBAPMN	510400 19	510400	196800	NULL	NULL	NULL	NULL	WF1	TEOM	PM10	19	29	38	60	5-Nov-02	6	98		England
2003	Watford Roadside	roadside	HBAPMN	510400 19	510400	196800	NULL	NULL	NULL	NULL	WF1	TEOM	PM10	22	34	49	59	10-Aug-03	24	88		England
2001	Chichester Roadside	roadside	SAQSG	485887 10	485887	103802	NULL	NULL	NULL	NULL	CI1	TEOM	PM10	21	27	35	40	13-Oct-01	2	50		England
2002	Chichester Roadside	roadside	SAQSG	485887 10	485887	103802	NULL	NULL	NULL	NULL	CI1	TEOM	PM10	22	30	36	48	10-May-02	3	97		England
2003	Chichester Roadside	roadside	SAQSG	485887 10	485887	103802	NULL	NULL	NULL	NULL	CI1	TEOM	PM10	23	32	50	62	11-Aug-03	16	88		England
2001	Crawley Background	background	SAQSG	528325 13	528325	135927	-999	-999	-999	-999	CA1	TEOM	PM10	17	25	35	49	17-Jan-01	5	99		England
2002	Crawley Background	background	SAQSG	528325 13	528325	135927	-999	-999	-999	-999	CA1	TEOM	PM10	16	24	32	45	5-Apr-02	3	99		England
2003	Crawley Background	background	SAQSG	528325 13	528325	135927	-999	-999	-999	-999	CA1	TEOM	PM10	18	30	43	57	9-Aug-03	12	88		England
2000	Eastbourne Background	background	SAQSG	561282 09	561282	98407	-999	-999	-999	-999	EB1	TEOM	PM10	19	25	32	40	4-Dec-00	2	58		England
2001	Eastbourne Background	background	SAQSG	561282 09	561282	98407	-999	-999	-999	-999	EB1	TEOM	PM10	19	30	38	43	20-Jan-01	5	99		England
2002	Eastbourne Background	background	SAQSG	561282 09	561282	98407	-999	-999	-999	-999	EB1	TEOM	PM10	20	31	40	56	2-Feb-02	10	100		England
2003	Eastbourne Background	background	SAQSG	561282 09	561282	98407	-999	-999	-999	-999	EB1	TEOM	PM10	22	33	45	64	11-Aug-03	20	91		England
2001	Hastings Roadside	roadside	SAQSG	577138 10	577138	108455	NULL	NULL	NULL	NULL	HT1	TEOM	PM10	26	34	46	54	30-Oct-01	25	50		England
2002	Hastings Roadside	roadside	SAQSG	577138 10	577138	108455	NULL	NULL	NULL	NULL	HT1	TEOM	PM10	28	42	50	60	2-Feb-02	54	81		England
2003	Hastings Roadside	roadside	SAQSG	577138 10	577138	108455	NULL	NULL	NULL	NULL	HT1	TEOM	PM10	30	46	62	81	9-Aug-03	64	87		England
2002	Horsham Background	background	SAQSG	517688 13	517688	133497	-999	-999	-999	-999	HO1	TEOM	PM10	15	22	27	34	12-Dec-02	0	82		England
2003	Horsham Background	background	SAQSG	517688 13	517688	133497	-999	-999	-999	-999	HO1	TEOM	PM10	19	29	45	52	29-Mar-03	11	84		England
2000	Lewes Roadside	roadside	SAQSG	540100 10	540100	101200	NULL	NULL	NULL	NULL	LS1	TEOM	PM10	23	32	41	57	13-Dec-00	14	58		England
2001	Lewes Roadside	roadside	SAQSG	540100 10	540100	101200	NULL	NULL	NULL	NULL	LS1	TEOM	PM10	23	34	45	52	1-Oct-01	19	100		England
2002	Lewes Roadside	roadside	SAQSG	540100 10	540100	101200	NULL	NULL	NULL	NULL	LS1	TEOM	PM10	25	37	44	56	27-Oct-02	28	100		England
2003	Lewes Roadside	roadside	SAQSG	540100 10	540100	101200	NULL	NULL	NULL	NULL	LS1	TEOM	PM10	24	36	46	66	11-Aug-03	23	89		England
2002	Sussex Mobile (location	roadside	SAQSG	491550 10	491550	101600	NULL	NULL	NULL	NULL	SX2	TEOM	PM10	17	19	25	38	30-Sep-02	0	28		England
2002	Sussex Mobile (location	roadside	SAQSG		0	0	NULL	NULL	NULL	NULL	SX3	TEOM	PM10	26	9	31	58	12-Nov-02	3	6		England
2003	Sussex Mobile (location	roadside	SAQSG		0	0	NULL	NULL	NULL	NULL	SX3	TEOM	PM10	20	9	30	48	20-Jan-03	4	7		England
2003	Sussex Mobile A259 (lo	roadside	SAQSG	522072 10	522072	105136	NULL	NULL	NULL	NULL	SX4	TEOM	PM10	29	20	38	49	6-Nov-03	7	11		England
2000	Abingdon	URBAN BACKG	Cal Club	SU 49910	449910	197100						Teom	PM10	16	25	32	37	26-Jan-00	0	94		England
2001	Abingdon	URBAN BACKG	Cal Club	SU 49910	449910	197100						Teom	PM10	17	24	34	41	17-Jan-01	2	87		England
2002	Abingdon	URBAN BACKG	Cal Club	SU 49910	449910	197100						Teom	PM10	15	23	31	42	12-Dec-02	3	91		England
2003	Abingdon	URBAN BACKG	Cal Club	SU 49910	449910	197100						Teom	PM10	17	27	37	50	8-Aug-03	5	89	01/12/2003	England
2003	Ards	URBAN BACKG	Cal Club		349000	374200						Teom	PM10	19	31	43	54	18-Mar-03	11	72	01/09/2003	N Ireland
2001	Birmingham Airport	AIRPORT	Cal Club		417960	283999						Teom	PM10	17	26	40	47	18-Jan-01	8	99		England
2002	Birmingham Airport	AIRPORT	Cal Club		417960	283999						Teom	PM10	16	22	29	41	12-Sep-02	2	58		England
2003	Birmingham Airport	AIRPORT	Cal Club		417960	283999						Teom	PM10	18	31	45	58	20-Apr-03	16	97		England
2002	Bracknell Bagshot Road	ROADSIDE	Cal Club		487297	167802	A322	1m				Teom	PM10	21	29	36	54	10-May-02	4	84		England
2003	Bracknell Bagshot Road	ROADSIDE	Cal Club		487297	167802	A322	1m				Teom	PM10	22	29	43	55	29-Mar-03	12	51	01/01/2003	England
2002	Bracknell Foxhill	URBAN BACKG	Cal Club		486897	168028						Teom	PM10	16	24	32	50	5-Apr-02	4	96		England
2003	Bracknell Foxhill	URBAN BACKG	Cal Club		486897	168028						Teom	PM10	18	26	39	53	29-Mar-03	8	53	01/01/2003	England
2003	Bury Prestwich	ROADSIDE	Cal Club		381600	403200	A56	1m	2.5	30,000/day		Teom	PM10	26	38	53	67	19-Mar-03	35	75	01/12/2003	England
2003	Bury Radcliffe	ROADSIDE	Cal Club		378100	398700	A665	2m	2.5	21,000/day		Teom	PM10	22	38	54	60	27-Mar-03	33	87	01/12/2003	England
1998	Cambridge Gonville Pla	ROADSIDE	Cal Club		545508	257828						Teom	PM10	19	23	33	41	1-Sep-98	1	33		England
1999	Cambridge Gonville Pla	ROADSIDE	Cal Club		545508	257828						Teom	PM10	21	32	48	73	4-Aug-99	14	99		England
2000	Cambridge Gonville Pla	ROADSIDE	Cal Club		545508	257828						Teom	PM10	19	28	36	53	29-Sep-00	6	95		England
2001	Cambridge Gonville Pla	ROADSIDE	Cal Club		545508	257828						Teom	PM10	16	25	31	41	13-Oct-01	1	95		England
2002	Cambridge Gonville Pla	ROADSIDE	Cal Club		545508	257828						Teom	PM10	17	25	35	41	4-Apr-02	4	95		England
2003	Cambridge Gonville Pla	ROADSIDE	Cal Club		545508	257828						Teom	PM10	18	28	43	55	20-Apr-03	10	97	01/11/2003	England

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2002	Cambridge Newmarket	ROADSIDE	Cal Club		546312	258896						Teom	PM10	19	25	36	61	28-Jun-02	1	58		England
2003	Cambridge Newmarket	ROADSIDE	Cal Club		546312	258896						Teom	PM10	21	34	58	76	29-Mar-03	13	83	01/11/2003	England
1999	Cambridge Parker Stree	ROADSIDE	Cal Club		545366	258391						Teom	PM10	24	34	45	64	7-Nov-99	16	95		England
2000	Cambridge Parker Stree	ROADSIDE	Cal Club		545366	258391						Teom	PM10	21	28	35	49	22-Mar-00	2	92		England
2001	Cambridge Parker Stree	ROADSIDE	Cal Club		545366	258391						Teom	PM10	21	31	38	57	17-Jan-01	7	98		England
2002	Cambridge Parker Stree	ROADSIDE	Cal Club		545366	258391						Teom	PM10	25	35	46	61	28-Jun-02	26	98		England
2003	Cambridge Parker Stree	ROADSIDE	Cal Club		545366	258391						Teom	PM10	26	38	56	66	6-Aug-03	33	99	01/11/2003	England
1999	Cambridge Silver Stree	ROADSIDE	Cal Club	TL 44758 5	544758	258391						Teom	PM10	20	29	41	54	11-Sep-99	8	95		England
2000	Cambridge Silver Stree	ROADSIDE	Cal Club	TL 44758 5	544758	258391						Teom	PM10	18	26	35	45	22-Mar-00	2	93		England
2001	Cambridge Silver Stree	ROADSIDE	Cal Club	TL 44758 5	544758	258391						Teom	PM10	21	31	39	59	18-Jan-01	8	92		England
2002	Cambridge Silver Stree	ROADSIDE	Cal Club	TL 44758 5	544758	258391						Teom	PM10	21	30	41	49	12-Dec-02	12	100		England
2003	Cambridge Silver Stree	ROADSIDE	Cal Club	TL 44758 5	544758	258391						Teom	PM10	22	34	50	70	26-Feb-03	26	92	01/11/2003	England
2002	Carrickfergus Rosebroo	URBAN BACKG	Cal Club		341130	387999						Teom	PM10	18	23	38	53	12-Sep-02	6	48		N Ireland
2003	Carrickfergus Rosebroo	URBAN BACKG	Cal Club		341130	387999						Teom	PM10	19	32	47	56	14-Feb-03	13	72		N Ireland
2002	Castlereagh Espie Way	URBAN BACKG	Cal Club		149800	528200						Teom	PM10	15	17	25	56	12-Sep-02	1	32	01/09/2002	N Ireland
2003	Castlereagh Espie Way	URBAN BACKG	Cal Club		149800	528200						Teom	PM10	14	22	33	41	22-Feb-03	4	70	01/09/2002	N Ireland
2002	Castlereagh Lough Viev	ROADSIDE	Cal Club	J364711	336400	371100						Teom	PM10	24	19	30	219	23-Oct-02	4	31	01/09/2002	N Ireland
2003	Castlereagh Lough Viev	ROADSIDE	Cal Club	J364711	336400	371100						Teom	PM10	17	28	40	47	27-Mar-03	8	84	01/09/2002	N Ireland
2000	Craigavon	URBAN BACKG	Cal Club	J08215840	308210	358400						Teom	PM10	18	27	36	118	26-Oct-00	5	72		N Ireland
2002	Craigavon Lord Lurgan	URBAN BACKG	Cal Club	J07965926	307960	359260						Teom	PM10	11	18	25	60	12-Sep-02	2	74		N Ireland
2002	Derry Brandywell	URBAN BACKG	Cal Club		242800	417100						Teom	PM10	19	28	50	81	6-Dec-02	18	73		N Ireland
2003	Derry Brandywell	URBAN BACKG	Cal Club		242800	417100						Teom	PM10	22	39	55	87	5-Jan-03	34	99	01/09/2003	N Ireland
2001	Guildford Beckingham R	ROADSIDE	Cal Club		498178	150713						Bam	PM10	22	31	41	53	3-Nov-01	3	81		England
2002	Guildford Gyratory	ROADSIDE	Cal Club									Bam	PM10	26	33	43	65	10-May-02	1	54		England
2003	Guildford Gyratory	ROADSIDE	Cal Club									Bam	PM10	31	47	74	118	29-Mar-03	29	81		England
2001	King's Lynn Ed Benefer	ROADSIDE	Cal Club									Teom	PM10	16	24	41	167	21-Sep-01	11	87		England
2000	King's Lynn Leziate	ROADSIDE	Cal Club									Teom	PM10	12	15	26	59	11-Sep-00	3	33		England
2000	King's Lynn North Lynn	INDUSTRIAL BA	Cal Club									Teom	PM10	19	23	31	34	14-Nov-00	0	39		England
2002	King's Lynn Railway Ro	ROADSIDE	Cal Club									Teom	PM10	15	17	27	37	12-Dec-02	0	32		England
2003	King's Lynn Railway Ro	ROADSIDE	Cal Club									Teom	PM10	16	28	38	52	26-Mar-03	4	92	01/10/2003	England
2001	King's Lynn South Quay	INDUSTRIAL BA	Cal Club									Teom	PM10	26	36	107	267	10-May-01	33	71		England
2002	King's Lynn South Quay	INDUSTRIAL BA	Cal Club									Teom	PM10	27	41	150	320	10-Jul-02	42	86		England
2003	King's Lynn South Quay	INDUSTRIAL BA	Cal Club									Teom	PM10	28	42	114	223	10-Jun-03	44	65	01/10/2003	England
1999	King's Lynn Southgates	URBAN BACKG	Cal Club									Teom	PM10	15	23	30	41	11-Sep-99	1	67		England
2000	King's Lynn Southgates	URBAN BACKG	Cal Club									Teom	PM10	15	22	31	44	22-Mar-00	1	64		England
2003	Larne	ROADSIDE	Cal Club		413200	017500						Bam	PM10	22	36	65	189	11-Jul-03	19	74	01/09/2003	N Ireland
2000	Liverpool Islington	ROADSIDE	Cal Club		335310	390915						Teom	PM10	19	27	35	50	9-May-00	3	73		England
2001	Liverpool Islington	ROADSIDE	Cal Club		335310	390915	A565	3m				Teom	PM10	22	25	40	58	18-Jan-01	8	40		England
2002	Liverpool Islington	ROADSIDE	Cal Club		335310	390915						Teom	PM10	15	22	28	43	12-Sep-02	2	89		England
2003	Liverpool Islington	ROADSIDE	Cal Club		335310	390915						Teom	PM10	17	21	35	45	9-Aug-03	2	47	01/10/2003	England
2000	Liverpool Speke (LA)	URBAN BACKG	Cal Club	SJ438835	343800	383500						Teom	PM10	16	20	30	47	10-May-00	2	51		England
2000	Liverpool Vauxhall	ROADSIDE	Cal Club		334088	393477	A565	3m				Teom	PM10	22	31	42	63	9-May-00	11	79		England
2001	Liverpool Vauxhall	ROADSIDE	Cal Club									Teom	PM10	23	31	41	52	9-Aug-01	13	48		England
2002	Liverpool Vauxhall	ROADSIDE	Cal Club									Teom	PM10	22	33	41	62	11-Apr-02	14	86		England
2000	Liverpool Victoria Str	ROADSIDE	Cal Club				Victoria Str	0.1m				Teom	PM10	23	33	47	61	1-Jul-00	22	83		England
2001	Liverpool Victoria Str	ROADSIDE	Cal Club									Teom	PM10	23	30	42	62	18-Jan-01	13	63		England
2002	Liverpool Victoria Str	ROADSIDE	Cal Club									Teom	PM10	22	32	41	61	29-Jul-02	12	82		England
1998	Newham Cam Road	ROADSIDE	Cal Club	TQ 38661	538661	183969						Teom	PM10	29	32	44	56	18-Nov-98	18	30		London
1999	Newham Cam Road	ROADSIDE	Cal Club	TQ 38661	538661	183969						Teom	PM10	26	40	51	57	17-Mar-99	40	100		London
2000	Newham Cam Road	ROADSIDE	Cal Club	TQ 38661	538661	183969						Teom	PM10	21	31	42	52	22-Mar-00	13	99		London
2001	Newham Cam Road	ROADSIDE	Cal Club	TQ 38661	538661	183969						Teom	PM10	23	34	48	65	4-Nov-01	21	100		London
2002	Newham Cam Road	ROADSIDE	Cal Club	TQ 38661	538661	183969						Teom	PM10	23	34	52	66	3-Apr-02	19	97		London
2003	Newham Cam Road	ROADSIDE	Cal Club	TQ 38661	538661	183969						Teom	PM10	27	42	60	71	15-Apr-03	46	96	01/11/2003	London
1998	Newham Tant Avenue	URBAN BACKG	Cal Club	TQ 39702	539702	181455						Teom	PM10	23	27	42	50	23-Nov-98	8	54		London
1999	Newham Tant Avenue	URBAN BACKG	Cal Club	TQ 39702	539702	181455						Teom	PM10	23	34	47	55	1-Apr-99	24	97		London
2000	Newham Tant Avenue	URBAN BACKG	Cal Club	TQ 39702	539702	181455						Teom	PM10	23	34	43	61	22-Mar-00	20	94		London
2001	Newham Tant Avenue	URBAN BACKG	Cal Club	TQ 39702	539702	181455						Teom	PM10	22	33	50	72	4-Nov-01	21	96		London
2002	Newham Tant Avenue	URBAN BACKG	Cal Club	TQ 39702	539702	181455						Teom	PM10	24	36	50	72	5-Sep-02	24	92		London
2003	Newham Wren Close	ROADSIDE	Cal Club									Teom	PM10	24	37	56	65	26-Mar-03	28	74	01/11/2003	London
2001	Newry Monaghan Row	URBAN BACKG	Cal Club		307900	326800						Teom	PM10	15	21	34	52	12-Dec-01	4	70		N Ireland
2002	Newry Monaghan Row	URBAN BACKG	Cal Club		307900	326800						Teom	PM10	16	24	41	60	17-Dec-02	8	95		N Ireland

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2003	Newry Monaghan Row	URBAN BACKG	Cal Club		307900	326800						Teom	PM10	18	33	46	61	28-Mar-03	22	95	01/09/2003	N Ireland
2001	Newry Trevor Hill	ROADSIDE	Cal Club		308700	326700						Teom	PM10	26	36	49	66	12-Dec-01	26	60		N Ireland
2002	Newry Trevor Hill	ROADSIDE	Cal Club		308700	326700						Teom	PM10	26	40	53	68	12-Sep-02	38	99		N Ireland
2003	Newry Trevor Hill	ROADSIDE	Cal Club		308700	326700						Teom	PM10	28	48	62	70	27-Mar-03	71	97	01/09/2003	N Ireland
2003	North Down Bangor	URBAN BACKG	Cal Club									Teom	PM10	21	29	44	66	27-Mar-03	12	58	25/02/2003	N Ireland
2003	North Down Holywood	ROADSIDE	Cal Club									Teom	PM10	21	31	39	52	17-Apr-03	10	71	01/03/2003	N Ireland
2003	North Lincs Killingholme	INDUSTRIAL B	Cal Club									Teom	PM10	21	32	47	63	15-Apr-03	16	81	01/09/2003	England
2002	Norwich Airport	AIRPORT	Cal Club									Teom	PM10	13	17	30	36	3-Apr-02	0	56		England
2002	Norwich Airport	AIRPORT	Cal Club									Teom	PM2.5	10	12	23	27	2-Apr-02		43.9		England
2000	Norwich Bracondale	URBAN BACKG	Cal Club									Teom	PM10	14	17	23	28	21-Dec-00	0	38		England
2000	Norwich Bracondale	URBAN BACKG	Cal Club									Teom	PM2.5	10	13	18	22	5-Nov-00		38.1		England
2002	Norwich Golding Place	ROADSIDE	Cal Club									Teom	PM10	16	20	32	42	12-Dec-02	1	32		England
2003	Norwich Golding Place	ROADSIDE	Cal Club									Teom	PM10	17	30	42	59	20-Apr-03	11	69		England
2002	Norwich Golding Place	ROADSIDE	Cal Club									Teom	PM2.5	11	13	20	32	12-Dec-02		32.4		England
2003	Norwich Golding Place	ROADSIDE	Cal Club									Teom	PM2.5	14	23	30	38	21-Feb-03		68.7		England
2001	Norwich Rose Lane	ROADSIDE	Cal Club									Teom	PM10	20	24	33	46	13-Oct-01	2	44		England
2001	Norwich Rose Lane	ROADSIDE	Cal Club									Teom	PM2.5	16	20	27	38	13-Oct-01		43.8		England
1999	Norwich St. Stephens	URBAN BACKG	Cal Club									Teom	PM10	19	28	39	82	13-Oct-99	8	80		England
2000	Norwich St. Stephens	URBAN BACKG	Cal Club									Teom	PM10	19	26	35	73	22-Mar-00	4	57		England
1999	Norwich St. Stephens	URBAN BACKG	Cal Club									Teom	PM2.5	12	18	27	35	11-Sep-99		79.5		England
2000	Norwich St. Stephens	URBAN BACKG	Cal Club									Teom	PM2.5	15	20	28	55	22-Mar-00		56.4		England
1999	Oldham West Endhouse	URBAN BACKG	Cal Club		391943	405472						Teom	PM10	20	28	38	49	4-May-99	4	76		England
2000	Oldham West Endhouse	URBAN BACKG	Cal Club		391943	405472						Teom	PM10	18	27	38	57	9-May-00	5	94		England
2001	Oldham West Endhouse	URBAN BACKG	Cal Club		391943	405472						Teom	PM10	19	29	38	61	2-Aug-01	6	87		England
2002	Oldham West Endhouse	URBAN BACKG	Cal Club		391943	405472						Teom	PM10	16	24	33	53	5-Apr-02	4	91		England
2003	Oldham West Endhouse	URBAN BACKG	Cal Club		391943	405472						Teom	PM10	17	30	38	47	20-Apr-03	7	98	01/06/2003	England
1998	Oxford Cornmarket	URBAN BACKG	Cal Club									Teom	PM10	28	41	51	67	14-May-98	48	96		England
1999	Oxford Cornmarket	URBAN BACKG	Cal Club									Teom	PM10	23	34	46	54	12-Mar-99	18	92		England
1997	Oxford East	URBAN BACKG	Cal Club		453000	205800						Teom	PM10	22	30	55	81	30-Oct-97	16	45		England
1998	Oxford East	URBAN BACKG	Cal Club		453000	205800						Teom	PM10	21	36	49	63	7-May-98	27	95		England
1999	Oxford East	URBAN BACKG	Cal Club		453000	205800						Teom	PM10	19	32	49	62	6-Sep-99	17	96		England
2000	Oxford East	URBAN BACKG	Cal Club		453000	205800						Teom	PM10	18	30	53	75	16-May-00	16	99		England
2001	Oxford East	URBAN BACKG	Cal Club		453000	205800						Teom	PM10	19	30	43	67	24-Jul-01	16	98		England
2002	Oxford East	URBAN BACKG	Cal Club		453000	205800						Teom	PM10	17	26	35	46	5-Apr-02	1	97		England
1997	Oxford St Ebbes	URBAN BACKG	Cal Club		451200	205400						Teom	PM10	19	23	35	59	30-Oct-97	4	40		England
1998	Oxford St Ebbes	URBAN BACKG	Cal Club		451200	205400						Teom	PM10	18	30	37	80	29-Jan-98	4	86		England
1999	Oxford St Ebbes	URBAN BACKG	Cal Club		451200	205400						Teom	PM10	15	22	31	42	12-Mar-99	4	88		England
2000	Oxford St Ebbes	URBAN BACKG	Cal Club		451200	205400						Teom	PM10	14	23	31	34	7-Mar-00	0	92		England
2001	Oxford St Ebbes	URBAN BACKG	Cal Club		451200	205400						Teom	PM10	16	23	33	41	17-Jan-01	2	81		England
2002	Oxford St Ebbes	URBAN BACKG	Cal Club		451200	205400						Teom	PM10	16	21	32	42	5-Apr-02	2	53		England
2003	Oxford St Ebbes	URBAN BACKG	Cal Club		451200	205400						Teom	PM10	17	28	40	52	8-Aug-03	8	86		England
1997	Ribble Valley Chatburn	URBAN INDUST	Cal Club	SD 77950	377950	444350						Teom	PM10	19	29	40	55	4-Dec-97	8	74		England
1998	Ribble Valley Chatburn	URBAN INDUST	Cal Club	SD 77950	377950	444350						Teom	PM10	18	28	39	50	16-Mar-98	8	97		England
1999	Ribble Valley Chatburn	URBAN INDUST	Cal Club	SD 77950	377950	444350						Teom	PM10	17	25	32	45	2-Apr-99	2	91		England
2000	Ribble Valley Chatburn	URBAN INDUST	Cal Club	SD 77950	377950	444350						Teom	PM10	16	24	37	52	26-Jan-00	3	93		England
2001	Ribble Valley Chatburn	URBAN INDUST	Cal Club	SD 77950	377950	444350						Teom	PM10	16	23	31	57	17-Dec-01	1	91		England
2002	Ribble Valley Chatburn	URBAN INDUST	Cal Club	SD 77950	377950	444350						Teom	PM10	16	23	32	40	5-Apr-02	1	56		England
2003	Ribble Valley Chatburn	URBAN INDUST	Cal Club	SD 77950	377950	444350						Teom	PM10	16	27	40	47	6-Dec-03	9	96	01/10/2003	England
2000	Ribble Valley Lillands	URBAN INDUST	Cal Club	SD 73650	373650	443650						Teom	PM10	14	21	26	35	23-Mar-00	0	79		England
2001	Ribble Valley Lillands	URBAN INDUST	Cal Club	SD 73650	373650	443650						Teom	PM10	13	23	31	40	26-Jun-01	3	97		England
2002	Ribble Valley Lillands	URBAN INDUST	Cal Club	SD 73650	373650	443650						Teom	PM10	13	20	30	47	12-Sep-02	1	99		England
2003	Ribble Valley Lillands	URBAN INDUST	Cal Club	SD 73650	373650	443650						Teom	PM10	15	28	41	49	15-Apr-03	11	98	01/10/2003	England
2000	Salford M60	ROADSIDE	Cal Club	SD 74813	374813	400853	M60	17				Teom	PM10	22	35	45	55	23-Mar-00	17	98		England
2001	Salford M60	ROADSIDE	Cal Club	SD 74813	374813	400853	M60	17				Teom	PM10	19	33	48	65	4-Mar-01	15	86		England
2002	Salford M60	ROADSIDE	Cal Club	SD 74813	374813	400853	M60	17				Teom	PM10	22	35	44	67	12-Sep-02	22	100		England
2003	Salford M60	ROADSIDE	Cal Club	SD 74813	374813	400853	M60	17				Teom	PM10	26	44	59	76	20-Feb-03	53	99		England
2001	Slough Colnbrook	URBAN BACKG	Cal Club	TQ 03546	503546	176824						Teom	PM10	18	27	39	55	18-Jan-01	8	92		England
2002	Slough Colnbrook	URBAN BACKG	Cal Club	TQ 03546	503546	176824						Teom	PM10	19	29	40	51	5-Apr-02	8	99		England
2003	Slough Colnbrook	URBAN BACKG	Cal Club	TQ 03546	503546	176824						Teom	PM10	21	35	49	56	22-Feb-03	26	98	01/12/2003	England
2001	Slough Town Centre A4	ROADSIDE	Cal Club	SU 96599	496599	180156						Teom	PM10	20	29	44	47	5-Mar-01	11	97		England
2002	Slough Town Centre A4	ROADSIDE	Cal Club	SU 96599	496599	180156						Teom	PM10	19	26	38	78	4-Nov-02	6	89		England

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2001	South Bucks Gerrards C	ROADSIDE	Cal Club									Teom	PM10	18	26	34	46	13-Oct-01	3	59		England
2002	South Bucks Gerrards C	ROADSIDE	Cal Club									Teom	PM10	18	26	36	44	1-Jan-02	5	93		England
2003	South Bucks Gerrards C	ROADSIDE	Cal Club									Teom	PM10	20	33	50	73	23-Dec-03	20	86	01/10/2003	England
2001	South Cambs Bar Hill	ROADSIDE	Cal Club									Bam	PM10	22	31	58	99	5-Jul-01	16	75		England
2002	South Cambs Bar Hill	ROADSIDE	Cal Club									Bam	PM10	23	37	52	119	14-Jul-02	27	97		England
2003	South Cambs Bar Hill	ROADSIDE	Cal Club									Bam	PM10	25	40	65	105	29-Mar-03	40	92		England
2001	South Cambs Bar Hill	ROADSIDE	Cal Club									Bam	PM2.5	14	21	35	58	5-Jul-01		77.1		England
2002	South Cambs Bar Hill	ROADSIDE	Cal Club									Bam	PM2.5	12	21	34	43	9-Jan-02		97.3		England
2003	South Cambs Bar Hill	ROADSIDE	Cal Club									Bam	PM2.5	14	25	43	83	29-Mar-03		92.7		England
2001	Stockport Bredbury	URBAN BACKG	Cal Club	SJ 93200 9	393200	391800						Teom	PM10	17	26	35	72	11-Dec-01	5	91		England
2002	Stockport Bredbury	URBAN BACKG	Cal Club	SJ 93200 9	393200	391800						Teom	PM10	16	23	32	46	5-Apr-02	3	92		England
2003	Stockport Bredbury	URBAN BACKG	Cal Club	SJ 93200 9	393200	391800						Teom	PM10	18	30	44	72	18-Apr-03	13	94	01/12/2003	England
2000	Stockport Cheadle	URBAN BACKG	Cal Club	SJ 86660 8	386660	387190						Teom	PM10	16	26	38	71	24-Aug-00	5	98		England
2001	Stockport Cheadle	URBAN BACKG	Cal Club	SJ 86660 8	386660	387190						Teom	PM10	15	22	31	35	11-May-01	0	66		England
2002	Stockport Cheadle	URBAN BACKG	Cal Club	SJ 86660 8	386660	387190						Teom	PM10	16	23	35	57	5-Apr-02	4	72	01/12/2002	England
2000	Stockport Marple	URBAN BACKG	Cal Club	SJ 95420 8	395420	388170						Teom	PM10	15	24	33	51	9-May-00	3	99		England
2001	Stockport Marple	URBAN BACKG	Cal Club	SJ 95420 8	395420	388170						Teom	PM10	15	22	31	58	11-Dec-01	3	87		England
2002	Stockport Marple	URBAN BACKG	Cal Club	SJ 95420 8	395420	388170						Teom	PM10	15	22	31	48	12-Sep-02	3	95	01/12/2002	England
2002	Strabane Springhill Pa	URBAN BACKG	Cal Club		234500	397500						Bam	PM10	38	57	107	147	18-Dec-02	48	67		N Ireland
2003	Strabane Springhill Pa	URBAN BACKG	Cal Club		234500	397500						Bam	PM10	43	78	120	164	5-Jan-03	101	97	01/09/2003	N Ireland
1999	Tameside Two Trees S	URBAN BACKG	Cal Club	SJ 93440 9	393440	394377						Teom	PM10	20	29	40	56	19-May-99	12	81		England
2000	Tameside Two Trees S	URBAN BACKG	Cal Club	SJ 93440 9	393440	394377						Teom	PM10	18	27	41	60	9-May-00	12	79		England
2001	Tameside Two Trees S	URBAN BACKG	Cal Club	SJ 93440 9	393440	394377						Teom	PM10	18	28	37	80	11-Dec-01	5	92		England
2002	Tameside Two Trees S	URBAN BACKG	Cal Club	SJ 93440 9	393440	394377						Teom	PM10	17	25	35	59	12-Sep-02	5	85		England
2003	Tameside Two Trees S	URBAN BACKG	Cal Club	SJ 93440 9	393440	394377						Teom	PM10	16	28	39	61	20-Apr-03	7	86	01/12/2003	England
1999	Trafford	URBAN BACKG	Cal Club	SJ 78768 9	378768	394646						Teom	PM10	18	29	37	49	11-Jan-99	6	90		England
2000	Trafford	URBAN BACKG	Cal Club	SJ 78768 9	378768	394646						Teom	PM10	17	26	36	52	9-May-00	6	99		England
2001	Trafford	URBAN BACKG	Cal Club	SJ 78768 9	378768	394646						Teom	PM10	18	30	40	52	18-Jan-01	8	100		England
2002	Trafford	URBAN BACKG	Cal Club	SJ 78768 9	378768	394646						Teom	PM10	19	23	35	59	5-Apr-02	7	51		England
2003	Trafford	URBAN BACKG	Cal Club	SJ 78768 9	378768	394646						Teom	PM10	20	34	49	66	18-Apr-03	25	99	01/12/2003	England
2001	V Glamorgan Font-y-Ga	URBAN BACKG	Cal Club									Bam	PM10	21	28	44	63	18-Jan-01	12	53		Wales
1999	Wigan	URBAN BACKG	Cal Club	SD 57962 0	357962	406104						Teom	PM10	21	34	44	71	11-Feb-99	17	96		England
2000	Wigan	URBAN BACKG	Cal Club	SD 57962 0	357962	406104						Teom	PM10	21	32	44	64	9-May-00	16	83		England
2001	Wigan	URBAN BACKG	Cal Club	SD 57962 0	357962	406104						Teom	PM10	23	37	53	88	12-Dec-01	30	81		England
2002	Wigan	URBAN BACKG	Cal Club	SD 57962 0	357962	406104						Teom	PM10	20	27	43	89	5-Apr-02	11	51	01/12/2002	England
1999	Wigan Leigh (Cal Club)	URBAN BACKG	Cal Club		357790	406158						Teom	PM10	18	29	38	48	31-Jan-99	6	92		England
2000	Wigan Leigh (Cal Club)	URBAN BACKG	Cal Club		357790	406158						Teom	PM10	17	26	36	50	30-Dec-00	4	99		England
2000	Winnersh	URBAN BACKG	Cal Club	SU 78050 7	478050	170950						Teom	PM10	15	19	31	37	6-May-00	0	54		England
2001	Winnersh	URBAN BACKG	Cal Club	SU 78050 7	478050	170950						Teom	PM10	18	27	40	63	23-Aug-01	9	94		England
2002	Winnersh	URBAN BACKG	Cal Club	SU 78050 7	478050	170950						Teom	PM10	26	40	61	80	26-Aug-02	37	83		England
2003	Winnersh	URBAN BACKG	Cal Club	SU 78050 7	478050	170950						Teom	PM10	27	32	61	81	30-Mar-03	22	33	01/04/2003	England
2000	Wokingham Council Off	URBAN BACKG	Cal Club									Teom	PM10	11	14	23	28	21-Jul-00	0	42		England
2001	Wokingham Council Off	URBAN BACKG	Cal Club									Teom	PM10	18	26	41	54	17-Jan-01	9	53		England
2002	Wokingham Earley		Cal Club									Teom	PM10	16	22	32	54	1-Jan-02	3	71		England
2003	Wokingham Woodward Clo		Cal Club		478589	170492						Teom	PM10	21	24	42	48	29-Mar-03	8	31	01/04/2003	England
1999	York Bootham	URBAN BACKG	Cal Club	SE 60150 5	460150	452850						Teom	PM10	15	23	36	61	2-Apr-99	6	83		England
2000	York Bootham	URBAN BACKG	Cal Club	SE 60150 5	460150	452850						Teom	PM10	15	23	29	37	22-Mar-00	0	89		England
2001	York Bootham	URBAN BACKG	Cal Club	SE 60150 5	460150	452850						Teom	PM10	16	23	32	42	5-Mar-01	3	86		England
1999	York Clifton Moor	ROADSIDE	Cal Club	SE 57700 5	457700	454800						Teom	PM10	20	28	45	66	2-Apr-99	13	83		England
2000	York Clifton Moor	ROADSIDE	Cal Club	SE 57700 5	457700	454800						Teom	PM10	19	26	34	42	22-Feb-00	3	74		England
1999	York Fishergate	ROADSIDE	Cal Club	SE 60750 5	460750	451150						Teom	PM10	20	29	42	64	2-Apr-99	12	85		England
2000	York Fishergate	ROADSIDE	Cal Club	SE 60750 5	460750	451150						Teom	PM10	19	27	36	53	22-Mar-00	4	94		England
2001	York Fishergate	ROADSIDE	Cal Club	SE 60750 5	460750	451150						Teom	PM10	22	33	43	64	15-Feb-01	13	93	01/10/2001	England
2001	York Rawcliffe	ROADSIDE	Cal Club	SE 57700 5	457700	454800						Teom	PM10	21	32	45	58	31-May-01	13	94		England
2003	Victoria	Kerbside	TfL/TRL									Partisol	PM10	36.9	49.8	71.1	89.7	16-Apr-03	34	60.8		London
2003	Victoria	Kerbside	TfL/TRL									Partisol	PM2.5	23.0	33.2	47.3	64.0	16-Apr-03	N/A	60.8		London
2002	Victoria	Kerbside										Partisol	PM10	N/A	N/A	N/A	N/A	N/A	N/A	N/A		London
2002	Victoria	Kerbside										Partisol	PM2.5	N/A	N/A	N/A	N/A	N/A	N/A	N/A		London
2001	Victoria	Kerbside	HA/TRL									Partisol	PM10	33.9	44.2	68.9	114.7	15-Jun-01	24	64.7		London
2001	Victoria	Kerbside	HA/TRL									Partisol	PM2.5	23.2	32.6	53.5	111.8	15-Jun-01	N/A	64.7		London
2000	Victoria	Kerbside	HA/TRL									Partisol	PM10	36.1	51.0	67.5	122.9	23-Mar-00	39	92.6		London

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2000	Victoria	Kerbside	HA/TRL									Partisol	PM2.5	24.3	36.3	52.9	98.2	23-Mar-00	N/A	91.3		London
1999	Victoria	Kerbside	HA/TRL									Partisol	PM10	38.3	46.2	64.3	109.7	2-Apr-99	27	53.6		London
1998	Victoria	Kerbside	HA/TRL									Partisol	PM10	44.8	66.3	89.6	116.6	14-May-98	82	75.4		London
2003	M60	Roadside	HA/TRL									TEOM	PM10	27.4	45.0	53.8	64.0	22-Aug-03	60	79.6		England
2002	M60	Roadside	HA/TRL									TEOM	PM10	19.0	27.7	40.2	52.4	12-Sep-02	10	99.1		England
2001	M60	Roadside	HA/TRL									TEOM	PM10	18.4	26.4	39.0	63.6	11-Dec-01	9	68.1		England
2000	M60	Roadside	HA/TRL									TEOM	PM10	16.5	25.3	36.7	51.7	9-May-00	4	72.4		England
2003	M4	Roadside	HA/TRL									TEOM	PM10	25.0	41.2	53.8	63.4	21-Feb-03	47	93.1		England
2002	M4	Roadside	HA/TRL									TEOM	PM10	24.8	37.2	53.5	83.3	27-Jun-02	33	92.9		England
2001	M4	Roadside	HA/TRL									TEOM	PM10	24.3	34.5	46.7	55.7	25-Jun-01	23	82.0		England
2000	M4	Roadside	HA/TRL									TEOM	PM10	18.4	28.6	37.3	50.8	28-Jun-00	7	95.5		England
1999	M4	Roadside	HA/TRL									TEOM	PM10	21.3	33.5	49.1	60.0	3-Sep-99	25	63.2		England
1998	M4	Roadside	HA/TRL									TEOM	PM10	24.8	39.1	54.0	71.4	25-Feb-98	38	92.6		England
1997	M4	Roadside	HA/TRL									TEOM	PM10	26.1	39.0	56.3	96.2	30-Oct-97	37	93.5		England
1996	M4	Roadside	HA/TRL									TEOM	PM10	27.8	43.1	63.3	317.0	31-Dec-96	53	83.2		England
2003	M25	Roadside	HA/TRL									TEOM	PM10	21.6	35.7	48.8	57.7	10-Aug-03	26	84.8		London
2003	M25	Roadside	HA/TRL									TEOM	PM2.5	15.4	25.3	37.9	44.7	15-Apr-03	N/A	73.4		London
2002	M25	Roadside	HA/TRL									TEOM	PM10	23.0	36.6	47.9	69.5	5-Apr-02	30	82.6		London
2002	M25	Roadside	HA/TRL									TEOM	PM2.5	12.5	21.7	28.4	38.2	12-Dec-02	N/A	96.8		London
2001	M25	Roadside	HA/TRL									TEOM	PM10	21.0	36.5	47.1	61.0	18-Jan-01	28	91.1		London
2001	M25	Roadside	HA/TRL									TEOM	PM2.5	13.6	23.2	26.3	59.5	24-Feb-01	N/A	90.7		London
2000	M25	Roadside	HA/TRL									TEOM	PM10	21.2	33.8	44.0	58.2	6-May-00	17	94.4		London
2000	M25	Roadside	HA/TRL									TEOM	PM2.5	14.4	24.1	31.3	39.8	23-Mar-00	N/A	99.9		London
1999	M25	Roadside	HA/TRL									TEOM	PM10	22.2	35.9	46.3	54.3	12-Mar-99	26	82.4		London
1999	M25	Roadside	HA/TRL									TEOM	PM2.5	15.4	21.7	31.6	42.0	11-Sep-99	N/A	57.1		London
1998	M25	Roadside	HA/TRL									TEOM	PM10	21.6	33.6	48.3	65.2	23-Nov-98	21	83.4		London
1997	M25	Roadside	HA/TRL									TEOM	PM10	27.0	42.3	57.8	79.1	11-Jan-97	54	92.4		London
1996	M25	Roadside	HA/TRL									TEOM	PM10	31.2	54.6	83.7	116.9	15-Mar-96	90	95.7		London
2003	Cheltenham	Roadside	HA/TRL									TEOM	PM10	27.1	41.7	55.8	65.3	23-Jan-03	42	79.4		England
2002	Cheltenham	Roadside	HA/TRL									TEOM	PM10	19.1	28.2	38.1	46.0	15-Feb-02	7	85.2		England
2001	Cheltenham	Roadside	HA/TRL									TEOM	PM10	24.6	35.5	50.1	65.2	19-Nov-01	22	81.0		England
2000	Cheltenham	Roadside	HA/TRL									TEOM	PM10	21.1	32.4	41.8	53.2	8-May-00	13	95.2		England
1994	Bottesford	Rural	JEP/Powergen	SK797376	479700	337600						TEOM	PM10							0		England
1995	Bottesford	Rural	JEP/Powergen	SK797376	479700	337600						TEOM	PM10	18.11	29.12	43.79	51.84	9-Jun-95	10	81		England
1996	Bottesford	Rural	JEP/Powergen	SK797376	479700	337600						TEOM	PM10	20.68	36.88	70.35	79.12	29-Sep-96	31	93		England
1997	Bottesford	Rural	JEP/Powergen	SK797376	479700	337600						TEOM	PM10	17.4	29.37	40.49	49.63	22-Jul-97	10	99		England
1998	Bottesford	Rural	JEP/Powergen	SK797376	479700	337600						TEOM	PM10	15.95	25.44	40.62	60.74	18-Jan-98	9	99		England
1999	Bottesford	Rural	JEP/Powergen	SK797376	479700	337600						TEOM	PM10	14.99	23.44	33.11	47.76	7-Mar-99	1	100		England
1997	Thorney	Rural	JEP/Powergen	SK858731	485800	373100						TEOM	PM10							0		England
1998	Thorney	Rural	JEP/Powergen	SK858731	485800	373100						TEOM	PM10	16.42	23.37	36.58	53.69	3-Nov-98	5	74		England
1999	Thorney	Rural	JEP/Powergen	SK858731	485800	373100						TEOM	PM10	15.54	21.19	31.13	39.19	21-Feb-99	2	76		England
1994	Weston On Trent	Rural	JEP/Powergen	SK408278	440800	327800						TEOM	PM10							0		England
1995	Weston On Trent	Rural	JEP/Powergen	SK408278	440800	327800						TEOM	PM10	21.98	34.24	56.72	77.67	1-Mar-95	25	88		England
1996	Weston On Trent	Rural	JEP/Powergen	SK408278	440800	327800						TEOM	PM10	22.77	41.37	66.7	88.15	3-Dec-96	41	87		England
1997	Weston On Trent	Rural	JEP/Powergen	SK408278	440800	327800						TEOM	PM10	18.95	30.97	41.44	54.72	1-Oct-97	12	83		England
1998	Weston On Trent	Rural	JEP/Powergen	SK408278	440800	327800						TEOM	PM10	19.04	20.03	32.38	71.65	3-Mar-98	3	25		England
2000	Bottesford	Rural	JEP/Powergen	SK797376	479700	337600						TEOM	PM10	15.01	23.68	30.81	62.57	3-Apr-00	1	100		England
2001	Bottesford	Rural	JEP/Powergen	SK797376	479700	337600						TEOM	PM10	14.5	22.47	30.68	47.27	22-Jan-01	2	88		England
2002	Bottesford	Rural	JEP/Powergen	SK797376	479700	337600						TEOM	PM10	15.2	22.79	34.65	49.36	29-Dec-02	6	98		England
2000	Thorney	Rural	JEP/Powergen	SK858731	485800	373100						TEOM	PM10	16.28	25.03	34.73	47.23	12-Dec-00	4	90		England
2001	Thorney	Rural	JEP/Powergen	SK858731	485800	373100						TEOM	PM10	15.94	24.45	32.24	70.71	25-Dec-01	4	96		England
2002	Thorney	Rural	JEP/Powergen	SK858731	485800	373100						TEOM	PM10	15.92	26.04	32.4	40.08	28-Apr-02	2	99		England
2003	Harlington	AIRPORT	BAA									Teom	PM10	24	32	55	336	29-Sep-03	23	62		London
1995	Heathrow LHR2	AIRPORT	BAA	TQ084767	508400	176700						Teom	PM10	27	45	62	76	25-Apr-95	56	94		London
1996	Heathrow LHR2	AIRPORT	BAA	TQ084767	508400	176700						Teom	PM10	28	45	72	99	15-Mar-96	53	93		London
1997	Heathrow LHR2	AIRPORT	BAA	TQ084767	508400	176700						Teom	PM10	28	45	62	114	30-Oct-97	47	79		London
1998	Heathrow LHR2	AIRPORT	BAA	TQ084767	508400	176700						Teom	PM10	23	36	48	58	29-Jan-98	29	77		London
1999	Heathrow LHR2	AIRPORT	BAA	TQ084767	508400	176700						Teom	PM10	23	37	44	53	11-Sep-99	28	94		London
2000	Heathrow LHR2	AIRPORT	BAA	TQ084767	508400	176700						Teom	PM10	21	33	45	51	28-Jun-00	17	97		London
2001	Heathrow LHR2	AIRPORT	BAA	TQ084767	508400	176700						Teom	PM10	23	36	48	69	17-Jan-01	21	96		London
2002	Heathrow LHR2	AIRPORT	BAA	TQ084767	508400	176700						Teom	PM10	21	32	44	60	5-Apr-02	15	97		London

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2003	Heathrow LHR2	AIRPORT	BAA	TQ084767	508400	176700						Teom	PM10	24	40	58	70	10-Aug-03	39	96		London
1994	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM10	22	28	47	89	22-Dec-94	13	40		England
1995	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM10	24	37	61	119	4-Nov-95	31	96		England
1996	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM10	22	39	59	74	10-Nov-96	35	100		England
1997	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM10	21	34	46	72	31-Oct-97	20	92		England
1998	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM10	17	26	41	68	30-Jan-98	9	94		England
1999	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM10	16	25	35	42	3-Sep-99	2	97		England
2000	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM10	16	26	37	59	10-May-00	4	98		England
2001	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM10	17	26	40	57	15-Nov-01	11	99		England
2002	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM10	15	23	32	41	5-Apr-02	1	100		England
2003	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM10	17	29	41	52	20-Apr-03	13	99		England
1994	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM2.5	15	19	33	75	22-Dec-94		39.9		England
1995	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM2.5	14	22	32	93	4-Nov-95		95		England
1995	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM2.5	14	22	32	83	4-Nov-95		94		England
1996	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM2.5	15	27	44	59	10-Nov-96		98.9		England
1997	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM2.5	15	24	38	62	31-Oct-97		92.5		England
1998	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM2.5	12	20	33	58	30-Jan-98		97.2		England
1999	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM2.5	11	18	25	34	3-Sep-99		85		England
2000	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM2.5	11	19	26	48	10-May-00		97.5		England
2001	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM2.5	13	22	32	52	15-Nov-01		97.1		England
2002	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM2.5	12	20	29	34	5-Apr-02		99.8		England
2003	Birmingham Hodge Hill	background	Birmingham CC		393500	282500						Teom	PM2.5	14	24	34	40	22-Feb-03		99.1		England
2000	Birmingham Roadside 1	ROADSIDE	Birmingham CC									Teom	PM10	21	32	43	64	19-Mar-00	16	76		England
2001	Birmingham Roadside 1	ROADSIDE	Birmingham CC									Teom	PM10	21	27	43	61	18-Jan-01	11	57		England
2000	Birmingham Roadside 2	ROADSIDE	Birmingham CC		403952	289681	Soho Rd	4.5				Teom	PM10	22	32	44	55	10-May-00	17	75		England
2001	Birmingham Roadside 2	ROADSIDE	Birmingham CC		403952	289681	Soho Rd	4.5				Teom	PM10	23	32	52	106	14-Nov-01	18	87		England
2002	Birmingham Roadside 2	ROADSIDE	Birmingham CC		403952	289681	Soho Rd	4.5				Teom	PM10	19	29	36	52	4-Nov-02	5	98		England
2003	Birmingham Roadside 2	ROADSIDE	Birmingham CC		403952	289681	Soho Rd	4.5				Teom	PM10	21	33	44	58	9-Aug-03	13	91		England
2002	Birmingham Roadside 3	ROADSIDE	Birmingham CC		409043	284260	Stratford S	2.8				Teom	PM10	21	33	41	60	5-Apr-02	12	73		England
2003	Birmingham Roadside 3	ROADSIDE	Birmingham CC		409043	284260	Stratford S	2.8				Teom	PM10	25	41	64	142	26-Feb-03	48	94		England
2001	Birmingham Roadside 4	ROADSIDE	Birmingham CC		406840	285520	Bristol St	7				Teom	PM10	22	28	35	52	19-Nov-01	5	51		England
2002	Birmingham Roadside 4	ROADSIDE	Birmingham CC		406840	285520	Bristol St	7				Teom	PM10	19	28	39	46	29-Mar-02	8	99		England
2003	Birmingham Roadside 4	ROADSIDE	Birmingham CC		406840	285520	Bristol St	7				Teom	PM10	23	38	47	62	9-Aug-03	33	97		England
1996	Birmingham West	URBAN BACK/S	Birmingham CC									Teom	PM10	27	39	53	110	10-Nov-96	36	57		England
1997	Birmingham West	URBAN BACK/S	Birmingham CC									Teom	PM10	24	40	53	106	31-Oct-97	40	98		England
1998	Birmingham West	URBAN BACK/S	Birmingham CC									Teom	PM10	22	38	53	69	20-Feb-98	29	99		England
1999	Birmingham West	URBAN BACK/S	Birmingham CC									Teom	PM10	18	27	37	50	3-Sep-99	5	75		England
1997	Birmingham West	URBAN BACK/S	Birmingham CC									Teom	PM2.5	16	24	35	87	31-Oct-97		63.6		England
1998	Birmingham West	URBAN BACK/S	Birmingham CC									Teom	PM2.5	16	29	45	56	30-Jan-98		98.1		England
1999	Birmingham West	URBAN BACK/S	Birmingham CC									Teom	PM2.5	14	21	30	37	19-May-99		74.6		England
1994	Hall Farm	Rural	JEP	TQ589848	558900	184800						TEOM	PM10	24.1	33.8	42.7	53.7	25-Jul-94	19	52		
1995	Hall Farm	Rural	JEP	TQ589848	558900	184800						TEOM	PM10	22.9	34.2	54.8	67.1	6-May-95	24	65		
1996	Hall Farm	Rural	JEP	TQ589848	558900	184800						TEOM	PM10	22.9	39	59	75.9	18-Mar-96	40	99		
1997	Hall Farm	Rural	JEP	TQ589848	558900	184800						TEOM	PM10	21.6	33.4	47	84.3	11-Jan-97	25	90		
1998	Hall Farm	Rural	JEP	TQ589848	558900	184800						TEOM	PM10	21.1	31.9	42.5	64.8	11-Aug-98	16	99		
1999	Hall Farm	Rural	JEP	TQ589848	558900	184800						TEOM	PM10	21.6	33.2	47.3	64.1	4-Feb-99	22	78		
1999	Hall Farm	Rural	JEP	TQ589848	558900	184800						TEOM	PM2.5	14.9	0	17.1	29.2	14-Oct-99	0	5		
2000	Hall Farm	Rural	JEP	TQ589848	558900	184800						TEOM	PM10	18.3	27.4	39	54.7	4-Nov-00	9	100		
2000	Hall Farm	Rural	JEP	TQ589848	558900	184800						TEOM	PM2.5	12.5	18.3	27.4	42	4-Nov-00	1	71		
2001	Hall Farm	Rural	JEP	TQ589848	558900	184800						TEOM	PM10	17.7	26.8	37.8	74.1	4-Nov-01	6	98		
2001	Hall Farm	Rural	JEP	TQ589848	558900	184800						TEOM	PM2.5	12.3	19.1	28.5	67.1	4-Nov-01	4	92		
2002	Hall Farm	Rural	RWE Innogy	TQ589848	558900	184800						TEOM	PM10	19.1	28.8	36.6	51	10-May-02	7	94		
2002	Hall Farm	Rural	RWE Innogy	TQ589848	558900	184800						TEOM	PM2.5	12.2	19.9	28.1	31.3	10-May-02	0	97		
1996	Wingham	Rural	JEP	TR243554	624300	155300						TEOM	PM10	16.4	13.5	19.5	44.9	7-Jun-96	1	14		
1997	Wingham	Rural	JEP	TR243554	624300	155300						TEOM	PM10	19.4	25.6	42.6	53.4	12-Aug-97	12	58		
1998	Wingham	Rural	JEP	TR243554	624300	155300						TEOM	PM10	17.1	27.9	35.4	52.3	16-Mar-98	4	98		
1998	Wingham	Rural	JEP	TR243554	624300	155300						TEOM	PM2.5	12.2	20.3	27.4	33.7	29-Mar-98	0	89		
1999	Wingham	Rural	JEP	TR243554	624300	155300						TEOM	PM10	17.7	26	37.8	58.7	10-Sep-99	7	94		
1999	Wingham	Rural	JEP	TR243554	624300	155300						TEOM	PM2.5	11.2	18.4	24.8	39.3	10-Sep-99	1	100		
2000	Wingham	Rural	JEP	TR243554	624300	155300						TEOM	PM10	18.7	19.7	31	56	23-Mar-00	1	28		
2000	Wingham	Rural	JEP	TR243554	624300	155300						TEOM	PM2.5	13.2	14.2	21	43.4	23-Mar-00	1	30		

Year	Site Name	Site Type	Network (AURN, LAQN,Cal Club etc)	Grid Reference	East (m)	North (m)	Road Number (roadside/ kerbside sites)	Distance from kerb (m) (roadside/ kerbside sites)	Sampling Height (m) (roadside/ kerbside sites)	Observed annual average daily traffic flow (roadside/ kerbside sites)	Site Code	Monintoring method (TEOM, Partisol, KFG, BAM etc)	Metric (PM10, PM2.5, PM1, etc)	Annual Mean PM (ug m-3, uncorrected)	36th highest day PM (ug -3, uncorrected)	8th highest day PM (ug m-3, uncorrected)	maximum day PM (ug m-3, uncorrected)	date of max day	Number of days above 50 ug m-3, corrected (ie TEOM *1.3)	data capture (%)	provisional from	Site Location
2003	Rosehurst Farm	Rural	RWE Innogy	SU549926	454900	192600						TEOM	PM10	17.7	23.6	35.6	52.4	8-Aug-03	6	57		
2003	Rosehurst Farm	Rural	RWE Innogy	SU549926	454900	192600						TEOM	PM2.5	11.4	16	26.6	40.6	10-Aug-03	2	56		
1994	Cliffe	Rural	JEP	SE659337	465900	433700						TEOM	PM10	21.9	29.2	46.8	68.8	6-Nov-94	16	50		
1995	Cliffe	Rural	JEP	SE659337	465900	433700						TEOM	PM10	21	35.1	51.6	74.3	10-Dec-95	27	85		
1994	Cliffe	Rural	JEP	SE659337	465900	433700						TEOM	PM10	21.9	29.2	46.8	68.8	6-Nov-94	16	50		
1996	Cliffe	Rural	JEP	SE659337	465900	433700						TEOM	PM10	21.9	36	60.1	89.6	21-Mar-96	31	97		
1997	Cliffe	Rural	JEP	SE659337	465900	433700						TEOM	PM10	20.9	33.8	45.5	57.4	11-Jan-97	21	90		
1998	Cliffe	Rural	JEP	SE659337	465900	433700						TEOM	PM10	17.1	26.8	35.2	40.7	27-Feb-98	3	90		
1998	Cliffe	Rural	JEP	SE659337	465900	433700						TEOM	PM2.5	12.7	21.3	32	52.6	18-Nov-98	1	100		
1999	Cliffe	Rural	JEP	SE659337	465900	433700						TEOM	PM10	16.9	24.8	38.7	55.3	2-Apr-99	8	100		
1999	Cliffe	Rural	JEP	SE659337	465900	433700						TEOM	PM2.5	10.8	16.3	25.7	41.9	2-Apr-99	1	100		
2000	Cliffe	Rural	JEP	SE659337	465900	433700						TEOM	PM10	17.3	18.7	26.3	44.3	21-Mar-00	2	25		
2000	Cliffe	Rural	JEP	SE659337	465900	433700						TEOM	PM2.5	10.9	10.8	17.4	28.4	22-Mar-00	0	25		
2002	Downes Ground Farm	Rural	Aire Valley	SE704249	470400	424900						GRAV	PM10	38.8	43	71	95	14-Dec-02	22	29		
2003	Downes Ground Farm	Rural	Aire Valley	SE704249	470400	424900						GRAV	PM10	27.7	53.1	80.7	114	29/03/03	43	98		