GIS-based models for ambient PM exposure and health impact assessment for the UK

John R Stedman, Susannah Grice, Andrew Kent and Sally Cooke

AEA Energy & Environment, The Gemini Building, Fermi Avenue, Harwell Didcot, Oxfordshire, OX11 0QR

John.stedman@aeat.co.uk

Abstract. GIS-based models have been developed to map ambient PM$_{10}$ and PM$_{2.5}$ mass concentrations across the UK. The resulting maps are used for the assessments of air quality required by the EU ambient air quality directives, health impact assessment and the development of UK air quality policy. Maps are presented for 2006 along with projections to 2020. The largest single contribution to the UK population-weighted mean annual mean background concentrations of PM$_{10}$ in 2006 is estimated to be from secondary PM (43%), followed by the contribution from primary PM (24%). Concentrations are predicted to decline by 15% for PM$_{10}$ and 13% for PM$_{2.5}$ over the period from 2006 to 2020. The extent of exceedence of the 24-hour limit value is predicted to decline from 1.9% to 0.1% of urban major roads over the same period. The potential health benefits of reductions in ambient PM are large. A reduction in concentration of 0.93 $\mu$g m$^{-3}$ as a result of a possible package of measures has been estimated within the UK Air Quality Strategy to result in a reduction in life years lost of approximately 2 – 4 million over a period of 100 years.

1. Introduction

There remains considerable concern about the health effects of airborne particulate matter [1]. This has lead to the adoption of objectives and limit values for PM$_{10}$ for the protection of human health at a UK national scale via the Air Quality Strategy [2] and a European scale via the recent Ambient Air Quality Directive [3]. Objectives and exposure reduction targets have also recently been adopted for PM$_{2.5}$ [2,3].

A model of ambient particulate matter concentration across the UK has been developed within a Geographical Information System (GIS). This model is known as the Pollution Climate Mapping (PCM) model. The development of this model has been described by Stedman et al [4], who presented maps of both PM$_{10}$ and PM$_{2.5}$ for 2004. This paper describes the further development of the PCM model to provide maps of concentration for 2006. The resulting maps are used to supplement ambient monitoring data in the compilation of the annual air quality assessments required by the EU ambient air quality directives [3]. Maps of projections of future air quality are also compiled and are used to underpin the development air UK air quality policy. The calculation of maps for a baseline and various different policy scenarios enables the difference between the baseline and different options to be assessed in terms of compliance with air quality objectives and limit values. A key summary statistics derived from the mapping studies is the population-weighted annual mean PM concentration, which can be used to estimate short and long term health impacts. A value can then be assigned to these.
health effects and compared with the costs of different policy options within a cost benefit analysis [2,5].

Maps of background PM concentrations are calculated by summing contributions from a wide range of different sources. The maps therefore provide a useful resource for the assessment of the relative contributions to total PM mass from the different chemical components and sources. Maps for 2006 are presented along with population-weighted mean and exceedence statistics for these maps and projections to 2020. Examples of health impact assessments included in the Air Quality Strategy published in 2007 [2] are also presented.

2. Method

2.1. Introduction

Maps of background PM concentrations are calculated within the PCM model by summing contributions from the following sources within the GIS:

- Secondary inorganic aerosol
- Secondary organic aerosol
- Large point sources of primary particles
- Small point sources of primary particles
- Area sources of primary particles
- Regional primary particles
- Iron and calcium rich dusts
- Sea salt

Maps have been calculated on a 1 km x 1km grid for the whole of the UK land area. The listed components have been found to provide approximate mass closure and no additional residual is required in order to provide good agreement with measured concentrations. A roadside increment model has also been used to calculate estimates of concentrations for 10,027 individual urban major road links.

2.2. Secondary aerosol

Secondary inorganic aerosol concentrations have been derived by interpolation and scaling of measurements of sulphate, nitrate and ammonium at 28 rural sites [6]. The scaling factors are required to take account of the contribution from bound water, counter ions and to apportion the mass between the fine (PM$_{2.5}$) and coarse (PM$_{10}$ – PM$_{2.5}$) fractions [7]. Secondary organic aerosol (semi-volatile organic compounds formed by the oxidation of non-methane volatile organic compounds) concentrations have been estimated from results from the HARM/ELMO model [8].

2.3. Primary particulate matter

Estimates of the emission of primary particulate matter (PM$_{10}$ or PM$_{2.5}$) are available from the UK National Atmospheric Emission Inventory [9] (NAEI). The contribution to ambient PM of emissions from identified large point sources with emissions of greater than 500 tonnes per year have been modelled explicitly using the air dispersion model ADMS [10]. The contribution from point sources of primary particles with annual emission of less than 500 tonnes have been estimated using a generic small points model [7] derived using ADMS.

The emissions from area sources of primary particles are represented within the NAEI as estimates for 1 km x 1 km grid squares across the UK. The contribution from local area sources (within 15 km) to ambient PM concentrations has been estimated using a dispersion kernel based model derived by running the air dispersion model ADMS for unit emissions. The contribution from more distant sources of primary particles can be described as a regional background of primary PM. This was estimated using the TRACK [11] receptor oriented, Lagrangian statistical model and emissions estimates from the NAEI and EMEP.
2.4. Iron and calcium rich dusts
The contribution to ambient PM concentrations from iron and calcium rich dusts was estimated from a combination of measurements made in Birmingham [12] and surrogate variables (vehicle movements and population, respectively) for the spatial distribution of the emission associated with these dusts. Rural concentrations across the UK were set to the values measured in the Birmingham area, while the additional urban increment (urban background – rural) was set to vary with the surrogate emission variables such that the modelled concentration in Birmingham matched the measurements.

2.5. Sea salt
The contribution from sea salt was derived by interpolation of measurements of chloride at the 28 rural sites [6] and scaling to take account of the sodium counter ion.

2.6. Roadside increment
The concentration of PM at the roadside (within approximately 10 m) can be considered as the sum of the background concentration described above and a roadside increment to take account of the very local emission from the vehicles on the adjacent road. We have used an empirical model [7] to estimate the roadside increment from the road link specific emissions estimates provided by the NAEI.

2.7. Monitoring data
There are a wide variety of methods for measuring PM mass concentrations and the mass obtained is dependent on the instrument used and this is an area of active research in itself [13]. A very brief summary of the comparability of UK measurements with our mapping studies is included here. The reference method for comparison with the EU directive limit values is a gravimetric instrument. Tapered Element Oscillating Microbalance (TEOM) instruments fitted with a Filter Dynamics Measurement System (FDMS) have been shown to be equivalent to the reference method, as have a number of β-attenuation instruments if a suitable scaling factor is applied [14]. The standard TEOM instrument, widely used in the UK, has been shown not to be equivalent even if the commonly applied scaling factor of 1.3 is used. A method utilizing the results from TEOM FDMS instruments to correct standard TEOM data, the Volatile Correction Model (VCM) has, however, been described [15].

Monitoring data from the UK national monitoring networks is available from the UK Air Quality Archive [16].

2.8. Model calibration and verification
Ambient monitoring data is required to calibrate the area source dispersion model component of the PCM model and the roadside increment model, while the models for the remaining components are not calibrated. Measurements of PM$_{10}$ and PM$_{2.5}$ from Partisol 2025 gravimetric instruments have been used to calibrate the models for 2006. There is a known high blank filter artefact with these measurements, thought to be related to filter conditioning and humidity problem with the quartz filters used, although the exact causes are not fully understood. We have therefore subtracted a total correction of 6.59 µg m$^{-3}$ from the reported annual means for 2006. This is made up of two components, 4.09 µg m$^{-3}$ for laboratory blank issues and 2.5 µg m$^{-3}$ for the blank due to the use of quartz filters, as opposed to teflon coated filters. The choice of these corrections has been informed by consideration of PM$_{10}$ and PM$_{2.5}$ monitoring data and PCM models for 2005 and 2006. FDMS measurements are available for 24 national network monitoring sites for 2007 and corrected Partisol 2025 for 2007 are consistent with a provisional model for 2007 calibrated using FDMS data only.

The PCM maps of PM$_{10}$ and PM$_{2.5}$ for 2006 have been calibrated using the corrected Partisol 2025 monitoring data and the results verified by comparison with TEOM data for London and the South East as corrected using the VCM model. Further indications of model performance have been obtained
by comparison with TEOM instruments multiplied by 1.3 and data suitably scaled data from β-
attenuation instruments.

2.9. Projections of concentration to 2020.
Maps of PM concentration in 2010, 2015 and 2020 have also been calculated for a baseline scenario
incorporating our best estimate of the likely impact of currently agreed national and international
policies. The emissions of primary PM for future years have been provided by the NAEI. Secondary
inorganic aerosol concentrations have been assumed to decline in line with the impact of changes in
precursor emissions on concentration predicted by the EMEP model [17]. The concentrations of
secondary organic aerosol, iron and calcium rich dusts and sea salt have been assumed to remain
unchanged from the values in 2006.

2.10. Population-weighted means
Population-weighted mean concentrations have been calculated by multiplying the 1 km x 1 km
background maps by 1 km x 1 km population statistics from the 2001 census. The values for all of
the grid squares are summed and then divided by the total population to calculate the population-weighted
mean. This is a useful summary statistic, which is related to human health impacts if the dose response
function is assumed to be linear with no threshold.

3. Results

3.1. Mapped PM$_{10}$ and PM$_{2.5}$ concentrations for 2006
Mapped PM$_{10}$ and PM$_{2.5}$ concentrations for 2006 are shown in Figure 1. PM$_{10}$ concentration are higher
than PM$_{2.5}$, the main contributors to the different include coarse sea salt, iron and calcium rich dusts
and nitrate. The south east to north west gradient is driven by the gradient in secondary inorganic
aerosol and the higher concentrations in urban areas and close to major road corridors are due to
emissions of primary PM and iron and calcium rich dusts.

Figure 2 shows verification plots for the background and roadside maps of PM$_{10}$ concentration for
2006. As expected the agreement is good for the national network sites used to calibrate the models.
The agreement with the TEOM data corrected using the VCM model is also good. This provides some
additional support for our choice of correction applied to the Partisol 2025 data prior to model
calibration. Comparison with TEOM data x 1.3 indicates that these measurements are systematically
higher than predicted by the model but that the ranking of the sites is quite reasonable. This suggests
that TEOM x 1.3 is likely to overestimate PM$_{10}$ concentrations at many locations.

3.2. Maps of PM$_{10}$ components for 2006
Figure 3 shows the individual contributions to ambient PM$_{10}$ from primary PM, secondary PM, iron
and calcium rich dusts and sea salt. The population-weighted mean contribution from each of these
sources are listed in Table 1. The largest single contribution is from secondary PM (43%), followed by
the contribution from primary PM (24%).

| Component                     | Concentration |
|-------------------------------|---------------|
| Primary PM                    | 3.91          |
| Secondary PM                  | 7.06          |
| Iron and calcium rich dusts   | 3.22          |
| Sea salt                      | 2.30          |
| Total                         | 16.49         |
Figure 1. Mapped annual mean background PM concentrations in 2006 ($\mu$g m$^{-3}$)

a) Background

b) Roadside

Figure 2. Verification plots for mapped PM$_{10}$ in 2006.
a) Primary PM

b) Secondary PM

c) Iron and calcium rich dusts

d) Sea salt

Figure 3. The contributions to mapped annual mean background PM$_{10}$ concentrations in 2006 from different components (µg m$^{-3}$)
3.3. Projections to 2020

Figure 4 shows population-weighted mean PM\textsubscript{10} and PM\textsubscript{2.5} concentration projections for our baseline scenario to 2020. Concentrations are expected to decline but not dramatically, by 15% for PM\textsubscript{10} and 13% for PM\textsubscript{2.5} over this period. The decline is significantly less steep than the decline in primary PM emissions or the precursor emissions of secondary inorganic PM. This is because the concentrations of secondary PM are expected to decline more slowly than precursor emissions and there are a number of components, such as iron and calcium rich dusts and sea salt, which are not expected to decline. The modelled PM\textsubscript{10} value in 2006 is close to the provisional FDMS-calibrated concentration for 2007, which provides additional confidence that an appropriate correction has been applied to the 2006 Partisol 2025 measurement data.

![Population-weighted annual mean PM concentrations](image)

Figure 4. UK Population weighted mean annual mean PM\textsubscript{10} and PM\textsubscript{2.5} concentrations 2006 – 2020 (µg m\textsuperscript{-3}).

Table 2 shows the expected change in the extent of exceedence of the 24-hour EU limit value for PM\textsubscript{10} at the roadside in urban areas between 2006 and 2020. Concentrations are typically higher at the roadside than in background locations, so it is roadside locations that are most at risk of exceeding the limit value. The 24-hour limit value came into force in 2005 and specifies that concentrations should not exceed 50 µg m\textsuperscript{-3} more than 35 days per year. We have modelled annual mean concentrations and have taken an annual mean concentration of more than 31.5 µg m\textsuperscript{-3} to be equivalent to an exceedence of the 24-hour limit value. This relationship has been derived from an examination of ambient monitoring data [7].

The UK has been divided into 43 zones for air quality assessment. These are 28 large cities and 15 geographical areas. Eight zones are estimated to have exceeded the limit value in 2006, reducing to just 10 km of road in one zone (London) by 2020. The new air quality directive specifies that the contribution from natural sources can be excluded prior to the air quality assessment. Table 2 shows that subtracting the contribution from sea salt significantly reduces the extent of exceedence and exceedences are then confined to the busiest roads in the largest cities in 2006.
Table 2. Predicted extent of exceedence of the 24-hour PM$_{10}$ limit value at the roadside (values in brackets with sea salt contribution subtracted).

|                  | Total assessed | 2006  | 2010  | 2015  | 2020  |
|------------------|----------------|-------|-------|-------|-------|
| Length of road   | 13,880         | 266(138) | 182(66) | 45(9) | 10(2) |
| Percentage of road length (%) | 100         | 1.9(1.0) | 1.3(0.5) | 0.3(0.1) | 0.1(0.0) |
| Number of zones  | 43             | 8(3)  | 2(2)  | 1(1)  | 1(1)  |

4. Discussion

4.1. Possible uses of the maps within epidemiology studies

The type of maps presented here can be used as an alternative to using individual monitoring site data for epidemiology studies of the health impact of air pollutants [18]. We believe that this method of estimating individual exposure to outdoor air pollution has some advantages over others. Whichever technique is used, there will always be some exposure misclassification. Most commonly, researchers have used single monitoring station data to assign exposure to all individuals living in large areas, which are likely to reflect individual exposure poorly because of the distance from the monitoring station and individual mobility [19]. Other researchers have used population density [20] or estimated traffic exposure [21] in an attempt to reduce exposure misclassification. The disadvantage of this type of approach (based on land use) is that the variables employed to improve upon fixed site monitoring may not always be the best markers for air pollutant emission densities.

We consider that the use of a GIS based PCM model of ambient air quality concentrations has considerable advantages over regression-based models. This is because our model incorporates detailed information from the emission inventory and meteorological data is used to model the dispersion of these emission in that atmosphere. A range of different tools have been used to estimate the contributions from various other PM sources to ambient PM including the use of measurement data for different chemical species as appropriate. An additional advantage of this type of approach is that there is scope to investigate the relative contributions to the total PM mass from the different chemical components and sources.

4.2. Health impact assessment

The type of maps presented here have been extensively used to support the development of UK air quality policy through the review and revision of the Air Quality Strategy [2]. We have adopted the methods for health impact quantification recommended by the Interdepartmental Group on Costs and Benefits (IGCB) [5]. Table 3 shows examples of the estimated health benefits of a package of measures for 2020 illustrated in the Air Quality Strategy [2]. The package of measures included early uptake of Euro 5/6 measures to reduce emissions from cars and Euro VI measures for heavy duty vehicles, increased uptake of low emission vehicles and measures to reduce emission from international shipping. The base year for this analysis was 2003. The life table calculations carried out for the Air Quality Strategy [2] used changes in concentration for 100 years to assess the change in life expectancy due to the long-term effect of PM. The change in annual mean PM$_{10}$ concentration in 2020 has been assumed to apply for 100 years. A coefficient of 6.0% per 10 $\mu g$ m$^{-3}$ change in PM has been used in these calculations. The results of the health impact assessment for the package of measures are shown in Table 3. The long-term effects (life years saved) dominate the health impact assessment. The range for this statistic represents the impact of different assumptions concerning the
lag between the change in concentration and the health impact (0 – 40 years) [5]. These health benefits have been calculated from a reduction in UK population-weighted mean annual mean PM$_{10}$ concentration of 0.93 μg m$^{-3}$. The value of the total health benefits of this measure have been calculated by the IGCB to be £886 – 2,039 million (annual net present value). This shows the large potential value of reductions in PM concentrations.

Table 3. Results of the health impact assessment for PM10 for the package of measures ‘R’ in 2020 [5]

| Life years saved 2010-2109 | Respiratory hospital admissions (2020 per year) | Cardiovascular hospital admissions (2020 per year) |
|---------------------------|-----------------------------------------------|-----------------------------------------------|
| 2,202,000 – 3,805,000     | 325                                           | 326                                           |

Acknowledgements
This work was funded by the UK Department for Environment, Food and Rural Affairs, Welsh Assembly Government, the Scottish Executive and the Department of the Environment in Northern Ireland under contract CPEA 15.

References
[1] Pope, C. A. and Dockery, D. W. (2006) Health Effects of Fine Particulate Air Pollution: Lines that Connect. J. Air & Waste Mange. Assoc. 56: 709-742
[2] Defra et al (2007) Department for Environment, Food and Rural Affairs, The Scottish Executive, Welsh Assembly Government and The Department of the Environment for Northern Ireland. The Air Quality Strategy for England, Scotland, Wales and Northern Ireland: Volume 1 and 2. (Cmdn 7169). The Stationery Office 2007 http://www.defra.gov.uk/environment/airquality/strategy/index.htm
[3] Directive on ambient air quality and cleaner air for Europe (2008) http://register.consilium.europa.eu/pdf/en/07/st03/st03696.en07.pdf
[4] Stedman, J. R., Kent, A. J., Grice, S., Bush, T. J. and Derwent, R. G. (2007). A consistent method for modelling PM$_{10}$ and PM$_{2.5}$ concentrations across the United Kingdom in 2004 for air quality assessment. Atmospheric Environment, 41, 161-172.
[5] DEFRA et al (2007). Department for Environment, Food and Rural Affairs, The Scottish Executive, Welsh Assembly Government and The Department of the Environment for Northern Ireland. An Economic Analysis to Inform the Air Quality Strategy: Updated Third Report of the Interdepartmental Group on Costs and Benefits. http://www.defra.gov.uk/environment/airquality/strategy/index.htm
[6] Personal communication, Sim Tang, CEH (2004). CEH Edinburgh, Centre for Ecology and Hydrology, Natural Environment Research Council. http://www.nbu.ac.uk/cara/networks.htm
[7] Kent, A J, Grice S, Stedman, J R, Cooke, S, Bush, T J, Vincent K J, Abbott J, (2008). UK air quality modelling for annual reporting 2006 on ambient air quality assessment under Council Directives 96/62/EC, 1999/30/EC and 2000/69/EC. AEA Energy & Environment. Report AEAT/ENV/R/2502.
[8] Whyatt, J.D., Metcalfe, S.E., Nicholson, J., Derwent, R.G., Page T. and Stedman, J. (2007) Regional scale modelling of particulate matter in the UK: source attribution and an assessment of uncertainties. Atmospheric Environment, 41, 3315-3327.
[9] Dore, C. J., Watterson, J. D., Murrells, T. P., Passant, N. R. et al., (2007). UK Emissions of Air Pollutants 1970 to 2005. National Atmospheric Emissions Inventory, AEA Technology, Report AEAT/ENV/R/2526. ISBN: 0-9554823-3-X http://www.airquality.co.uk/archive/reports/cat07/0801140937_2005_Report_FINAL.pdf
[10] CERC (2005) Cambridge Environmental Research Consultants
[11] Lee, D. S., Kingdon, R. D., Jenkin, M. E., and Garland, J. A. (2000) Modelling the atmospheric oxidised and reduced nitrogen budgets for the UK with a Lagrangian multi-layer long-range transport model. *Environmental Modelling and Assessment*, 5, 83-104.

[12] Yin, J. and Harrison, R. M. (2008). Paragmatic mass closure study for PM$_{1.0}$, PM$_{2.5}$ and PM$_{10}$ at roadsdie, urban background and rural sites. *Atmospheric Environment*, 42, 980-988.

[13] Air Quality Expert Group (AQEG, 2005). Particulate Matter in the United Kingdom. [http://www.defra.gov.uk/environment/airquality/aqeg/particulate-matter/index.htm](http://www.defra.gov.uk/environment/airquality/aqeg/particulate-matter/index.htm)

[14] Harrison, D., Maggs, R and Booker, J. (2006). UK Equivalence Programme for Monitoring of Particulate Matter. BV/AQ/AD202209/DH/2396 [http://www.airquality.co.uk/archive/reports/cat05/0606130952_UKPMEquivalence.pdf](http://www.airquality.co.uk/archive/reports/cat05/0606130952_UKPMEquivalence.pdf)

[15] Green, D., Baker, T. and Fuller, G. (2007). King’s College London Volatile Correction Model for PM$_{10}$. [KCLERG\MT\FDMS\EQ](http://www.airquality.co.uk/archive/reports/cat13/0711261345_KCL_Volatile_Correction_Model_for_PM10.pdf)

[16] UK Air Quality Archive (2008). [http://www.airquality.co.uk](http://www.airquality.co.uk).

[17] EMEP (2008) [http://www.emep.int/mscw/mscw_publications.html#2007](http://www.emep.int/mscw/mscw_publications.html#2007)

[18] Chronic exposure to outdoor air pollution and markers of systemic inflammation. Forbes, L. J. L., Patel, M. D., Rudnicka, A. R., Cook, D. G., Bush, T. Stedman, J. R. Whincup, P. H., Strachan, D. P., Anderson, H. R. Submitted to Epidemiology.

[19] Pope CA, III, Burnett RT, Thun MJ, Calle EE, Krewski D, Ito K et al (2002). Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. *JAMA* 287(9):1132-1141

[20] Hoek G, Bruneckreef B, Goldbohm S, Fischer P, van den Brandt PA (2002). Association between mortality and indicators of traffic-related air pollution in the Netherlands: a cohort study. *Lancet* 360(9341):1203-1209.

[21] Jerrett M, Burnett RT, Ma R, Pope CA, III, Krewski D, Newbold KB et al (2005). Spatial analysis of air pollution and mortality in Los Angeles. *Epidemiology* 16(6):727-736.