A regression-based method for mapping traffic-related air pollution: application and testing in four contrasting urban environments

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Abstract

Accurate, high-resolution maps of traffic-related air pollution are needed both as a basis for assessing exposures as part of epidemiological studies, and to inform urban air-quality policy and traffic management. This paper assesses the use of a GIS-based, regression mapping technique to model spatial patterns of traffic-related air pollution. The model — developed using data from 80 passive sampler sites in Huddersfield, as part of the SAVIAH (Small Area Variations in Air Quality and Health) project — uses data on traffic flows and land cover in the 300-m buffer zone around each site, and altitude of the site, as predictors of NO\textsubscript{2} concentrations. It was tested here by application in four urban areas in the UK: Huddersfield (for the year following that used for initial model development), Sheffield, Northampton, and part of London. In each case, a GIS was built in ArcInfo, integrating relevant data on road traffic, urban land use and topography. Monitoring of NO\textsubscript{2} was undertaken using replicate passive samplers (in London, data were obtained from surveys carried out as part of the London network). In Huddersfield, Sheffield and Northampton, the model was first calibrated by comparing modelled results with monitored NO\textsubscript{2} concentrations at 10 randomly selected sites; the calibrated model was then validated against data from a further 10–28 sites. In London, where data for only 11 sites were available, validation was not undertaken. Results showed that the model performed well in all cases. After local calibration, the model gave estimates of mean annual NO\textsubscript{2} concentrations within a factor of 1.5 of the actual mean (approx. 70–90\%) of the time and within a factor of 2 between 70 and 100\%

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of the time. $r^2$ values between modelled and observed concentrations are in the range of 0.58–0.76. These results are comparable to those achieved by more sophisticated dispersion models. The model also has several advantages over dispersion modelling. It is able, for example, to provide high-resolution maps across a whole urban area without the need to interpolate between receptor points. It also offers substantially reduced costs and processing times compared to formal dispersion modelling. It is concluded that the model might thus be used as a means of mapping long-term air pollution concentrations either in support of local authority air-quality management strategies, or in epidemiological studies. © 2000 Elsevier Science B.V. All rights reserved.

Keywords: Air pollution; Exposure assessment; Road traffic; GIS; Mapping

1. Introduction

Growing concern about the effects of traffic emissions on respiratory health, and growing pressures for policy and management action to reduce air pollution levels, have highlighted the need for improved methods of mapping traffic-related pollution in urban areas both for exposure assessment and policy support.

The acute health effects of short-term exposures to traffic-related pollution have been widely demonstrated (for detailed reviews see Committee on Medical Effects of Air Pollutants, 1995; Committee of the Environmental and Occupational Health Assembly of the American Thoracic Society, 1996). Much less, however, is known about the chronic effects of exposure. A number of studies, mainly in the USA, have reported associations with particulates and/or sulfur oxides (Dockery et al., 1989; Schwartz, 1993; Pope et al., 1995), but replication of these effects in European studies has proved difficult. Several studies have found (often relatively weak) associations between chronic morbidity or mortality and traffic-related pollution, measured either in terms of distance from road or traffic volume on the nearest roads (Wije et al., 1993; Edwards et al., 1994; Welland et al., 1994), or using estimated exposures to NO$_2$ as a marker for traffic-related pollution (NILU, 1991; Oosterlee et al., 1996). On the other hand, a number of studies — including several which have attempted to use more specific measures of exposure, such as modelled or measured NO$_2$ concentrations — have found no detectable effects (Livingstone et al., 1996; Magnus et al., 1998; Wilkinson et al., 1999). The extent to which long-term exposure to relatively low levels of traffic-related air pollution causes or increases susceptibility to respiratory illness thus remains uncertain. Much of this uncertainty relates to the problems of acquiring reliable estimates of exposure to traffic-related pollution, at the individual or small-area level, across large populations and cities. Nevertheless, if such effects do occur, they would have serious public health implications, for though the relative risk may be low, the number of people exposed would be very large, representing a high attributable risk. This would also have significant policy implications, for it would imply the need to reduce background, as well as peak, concentrations of the pollutants concerned.

Maps are needed, equally, to inform management and policy. In the UK, for example, the National Air Quality Strategy (Department of Environment, Transport and the Regions, 1997), which was updated in 1999 (Department of Environment, Transport and the Regions, 1999), introduces air-quality targets to be met by the year 2005 and obliges local authorities to establish Air Quality Management Areas (AQMAs) in zones where these are likely to be exceeded. The Integrated Transport White Paper, published in July 1998, further calls for local traffic management plans aimed at reducing traffic congestion and air pollution, and promises legislation giving greater powers to local authorities to control road traffic in order to improve air quality. Local authorities thus have an urgent need for ways of mapping traffic-related air pollution, to help identify potential AQMAs, to target management action, and to predict and monitor the effects of intervention.
Nevertheless, mapping traffic-related pollution is no trivial task. Levels of road traffic pollution vary substantially, often over distances of metres, and pollution patterns in urban areas are complex (Laxen and Noordally, 1987; Hewitt, 1991). Monitored data on urban air pollution are also sparse. Most urban networks comprise only a few sites, and those which do exist can rarely be taken as representative of the exposures experienced by the population as a whole. Some form of modelling is, therefore, essential if accurate maps of urban air pollution are to be obtained.

A wide range of line-source dispersion models have been developed in recent years, which might ostensibly be used for this purpose. These include: the CALINE models (Benson, 1992); the CAR model (Eerhens et al., 1993); the ADMS model (CERC, 1999); the Operational Street Pollution Model (OPMS) (Berkowicz et al., 1994); and the AERMOD model (USEPA, 1998). In general, however, the performance of line-source models has not always been good (Henriques and Briggs, 1998), and the data demands and intensive processing requirements of the more sophisticated models mean that they are often difficult to apply for pollution mapping across whole cities at the small-area scale. The costs of many of these models may also make them prohibitive for local authority use. For many applications, therefore, the need is for simpler yet more robust methods of pollution mapping which can provide high-resolution, city-wide maps of traffic-related pollution using existing or readily obtainable data.

Regression mapping offers one such approach. It is based on the principles that: (a) environmental conditions for the variable of interest can be estimated from a small number of readily measurable predictor variables; and (b) that the relationship between the target variable and these predictors can be reliably assessed on the basis of a small sample survey or ‘training’ area. Probably the main use of this approach to date has been in the interpretation of remotely sensed data (Fuller et al., 1998; Li et al., 1998), where regression methods are used to determine the relationship between the measured signature (e.g. reflectance) and land cover or other attributes derived from ground truth surveys. Increasingly, however, regression methods have also been used for a wide range of other applications, including: mapping of landscape quality (Briggs and France, 1980); soil conditions (Knotters et al., 1995); salt contamination (Mattson and Godfrey, 1994); and air pollution (Wagner, 1994). The study reported here builds upon one such application — the use of regression mapping to estimate mean annual concentrations of traffic-related pollution as a basis for examining small area variations in air quality and chronic respiratory health (the SAVIAH study).

2. The SAVIAH study

Details of the SAVIAH study have been reported elsewhere (Briggs et al., 1997; Elliott and Briggs, 1998; Fischer et al., 1998; Lebret et al., 1999). In brief, the study was an EU-funded, multicentre project, aimed at developing and testing methods for assessing the relationship between traffic-related air pollution and health, at the small-area scale. The study took place in four areas: Huddersfield (UK), Amsterdam (NL), Prague (CR) and Poznan (PO). With the exception of Poznan, the main emission source of interest was road traffic, and the main pollutant of interest was NO\textsubscript{x}. A key part of the study was, thus, to devise methods for mapping levels of NO\textsubscript{x} as a marker for traffic-related pollution.

As part of the study, a range of methods were applied and compared, including dispersion modelling [CALINE and CAR — in Huddersfield only (Collins, 1998)], spatial interpolation (contouring, kriging and trend surface analysis) and regression mapping. To help develop and calibrate these methods, data were obtained on mean annual NO\textsubscript{x} concentrations for a dense network of approximately 80 sites, using passive diffusion tubes (van Reeuwijk et al., 1998; Lebret et al., 1999). Performance of the various models was then tested against an independent set of reference data from 8 to 10 sites in each area, which had not been used in the initial modelling. In all areas, results showed that the regression mapping method was the most accurate, with $r^2$ values
across the 8–10 reference sites of 0.8–0.9 and S.E. of 3.7–4.7 μg/m³ (Table 1).

In the original SAVIAH study, three key variables were used as predictors in the regression model: traffic volume in the 300-m buffer zone around each site; land cover in the 300-m buffer zone; and surface altitude at the site. Because of differences in data availability in the various study areas, the way in which these were defined and the exact form of the multiple regression model were allowed to vary. Details are presented by Briggs et al. (1997) and Collins (1998). Subsequent research, reported here, was aimed at assessing the extent to which the model initially developed and applied in Huddersfield could be transferred elsewhere, with local calibration.

This study quickly revealed a difficulty with the original SAVIAH model developed in Huddersfield: namely, that the sine transformation used for the altitude variable, whilst providing a good fit to the specific range of altitudes encountered in the Huddersfield area, tended to produce extreme values at some altitudes outside this range. The original data for the Huddersfield area were, therefore, re-analysed, using the approach described by Briggs et al. (1997). In brief, this involves the following steps.

1. Computation of traffic volume (vehicle kilometres travelled = road length × vehicle numbers) for each 20-m buffer zone around the sampling sites, for 20-m radius buffer zones from 0–20 to 280–300 m. Data on vehicle numbers for each road length were derived from traffic counts conducted by the local authority, supplemented by expert knowledge for uncounted roads (mainly minor roads and small urban streets).
2. Computation of the area of land (ha) by land cover class (industry, commercial land, high-density housing, low-density housing, open land) for the same buffer zones. Data on land cover were derived from interpretation of 1:10 000 aerial photographs.
3. Use of regression analysis to identify the ‘best-fit’ weighted combination of buffer zones for traffic volume.
4. Use of regression analysis to identify the ‘best fit’ weighted combination of buffer zones and land cover classes for the unstandardised regression residual from step 3 above.
5. Entry of both these variables, together with data on altitude (derived from a 10-m resolution digital elevation model), variously transformed, into a regression model with the NO₂ data for the 80 sites, to derive a full predictive model of pollution concentrations.

Standardised to a sampling height of 2 m above ground level, this gave:

\[ C = 38.52 + 0.003705 \times \text{Traf} + 0.232 \times \text{Land} \]
\[ - 5.673 \log_{10}(\text{Alt}) \]

Table 1
Comparison of the performance of pollution mapping methods

<table>
<thead>
<tr>
<th>Site</th>
<th>Statistic</th>
<th>CALINE-3</th>
<th>TIN-contouring</th>
<th>Kriging</th>
<th>Trend surface analysis</th>
<th>Regression mapping</th>
</tr>
</thead>
<tbody>
<tr>
<td>Amsterdam (NO₂)</td>
<td>Adjusted r²</td>
<td>–</td>
<td>0.39 (10)</td>
<td>–</td>
<td>0.48 (10)</td>
<td>0.79 (10)</td>
</tr>
<tr>
<td>(S.E.)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Huddersfield (NO₂)</td>
<td>Adjusted r²</td>
<td>0.63 (8)</td>
<td>0.56 (7)</td>
<td>0.44 (8)</td>
<td>0.27 (8)</td>
<td>0.82 (8)</td>
</tr>
<tr>
<td>(S.E.)</td>
<td></td>
<td>5.25</td>
<td>5.69</td>
<td>6.45</td>
<td>8.04</td>
<td>3.69</td>
</tr>
<tr>
<td>Prague (NO₂)</td>
<td>Adjusted r²</td>
<td>–</td>
<td>0.09 (9)</td>
<td>0.34 (9)</td>
<td>0.37 (9)</td>
<td>0.87 (10)</td>
</tr>
<tr>
<td>(S.E.)</td>
<td></td>
<td>12.47</td>
<td>10.66</td>
<td>10.44</td>
<td>4.67</td>
<td></td>
</tr>
</tbody>
</table>

*Note. Figures in brackets refer to number of sites.
where, $C$ = mean annual NO$_2$ concentration at 2 m above ground level, and Traff = weighted traffic volume factor for the 300-m buffer zone around the site, computed as:

$$\text{Traff} = 15 \times \text{Tvol}_{0-40} + \text{Tvol}_{40-300}$$  \hspace{1cm} (2)

where: Tvol$_{0-40}$ = vehicle kilometres travelled in the 40-m buffer zone around the site (thousand vkt, during an 18-h day); Tvol$_{40-300}$ = vehicle kilometres travelled in the 40–300-m buffer zone around the site (thousand vkt, during an 18-h day); and Land$_{0-300}$ = area of land surface under industrial and high-density residential land in the 0–300-m buffer zone around the site (ha), computed as:

$$\text{Land} = 8 \times \text{HDH}_{0-300} + \text{Ind}_{0-300}$$  \hspace{1cm} (3)

where: HDH$_{0-300}$ = area of high-density housing within the 300-m buffer zone of the site (ha); Ind$_{0-300}$ = area of industrial land within the 300-m buffer zone around the site (ha); and Alt = altitude of the site (metres above ordnance datum).

The adjusted $r^2$ value for the 80 monitoring sites was 0.604 and the S.E. of the estimate was 6.06 $\mu$g/m$^3$. As in the original study (Briggs et al., 1997), this model was then validated locally by comparing predicted concentrations with monitored concentrations at the eight independent sites (not used in the preceding analysis) at which monitoring had been carried out continuously over the 1993–1994 study period. Regression analysis gave $r^2 = 0.674$ ($P < 0.008$), $\beta = 1.08$ and S.E.E. = 4.95 $\mu$g/m$^3$, confirming the satisfactory performance of the revised model in the original study area.

This revised equation forms the basis for the remainder of the study reported here, and is referred to hereafter as ‘the SAVIAH model’.

### 3. Methods

#### 3.1. The study areas

The revised SAVIAH model, described above, was applied and tested in four contrasting areas: (a) in the original Huddersfield area, using data for the following sampling year (May 1994–April 1995); (b) in the London boroughs of Hammer-

<table>
<thead>
<tr>
<th>Name</th>
<th>Surface area (km$^2$)</th>
<th>Year of data</th>
<th>No. of sites</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Huddersfield</td>
<td>305</td>
<td>1994–1995</td>
<td>20</td>
<td>An area of valleys and hills; altitude range 80–582 m O.D.; residential, industry (mainly chemicals) and commercial land use, surrounded by farming and moorland.</td>
</tr>
<tr>
<td>Hammersmith and Ealing</td>
<td>100</td>
<td>1993</td>
<td>11</td>
<td>Inner-London boroughs: mainly medium–high density residential areas, with commercial/industrial land and open space; heavily trafficked; altitude range 5–20 m O.D.</td>
</tr>
<tr>
<td>Northampton</td>
<td>8</td>
<td>1997–1998</td>
<td>39</td>
<td>‘Northern corridor’ — area surrounding main commuter route and most heavily trafficked roads of Northampton; primarily residential area with small satellite commercial centre; altitude range 65–130 m O.D.</td>
</tr>
<tr>
<td>Sheffield</td>
<td>140</td>
<td>1997–1998</td>
<td>28</td>
<td>Industrial city on flanks of Pennines; residential, heavy industrial, light industrial and commercial land, with areas of open space; altitude range 30–310 m O.D.</td>
</tr>
<tr>
<td>Data</td>
<td>Huddersfield</td>
<td>Hammersmith and Ealing</td>
<td>Northampton</td>
<td>Sheffield</td>
</tr>
<tr>
<td>----------------------</td>
<td>-------------------------------------</td>
<td>------------------------</td>
<td>-------------</td>
<td>-----------</td>
</tr>
<tr>
<td>Road network</td>
<td>Digitised from 1:10000 aerial photos</td>
<td>Bartholomew 1:5000 road network</td>
<td>Digitised from 1:10000 aerial photographs</td>
<td>OSCAR 1:1250 digital road lines</td>
</tr>
<tr>
<td>Traffic volume</td>
<td>Based on traffic counts from local authorities; flows for small roads interpolated</td>
<td>Based on traffic counts and models by London Research Centre</td>
<td>Based on SATURN traffic model; for major roads only</td>
<td>Based on traffic counts from local authorities; flows for small roads interpolated</td>
</tr>
<tr>
<td>Land cover</td>
<td>Derived from interpretation of 1:10000 aerial photographs</td>
<td>Derived from interpretation of 1:10000 aerial photographs</td>
<td>Derived from interpretation of 1:10000 aerial photographs</td>
<td>Derived from urban development plan maps</td>
</tr>
<tr>
<td>Altitude</td>
<td>Digital terrain model (Institute of Hydrology)</td>
<td>Digital terrain model (Ordnance Survey)</td>
<td>Digital terrain model (Ordnance Survey)</td>
<td>Digital terrain model (Ordnance Survey)</td>
</tr>
<tr>
<td>Monitored NO₂</td>
<td>Purpose-designed surveys</td>
<td>Local authority passive sampler surveys</td>
<td>Purpose-designed surveys</td>
<td>Purpose-designed surveys</td>
</tr>
<tr>
<td>Meteorology</td>
<td>Leeds</td>
<td>Northolt</td>
<td>Wittering</td>
<td>Shefield University and Nottingham</td>
</tr>
</tbody>
</table>
smith and Ealing; (c) in the city of Sheffield; and (d) in a small part of Northampton. Details of these study areas are given in Table 2. The study in Hammersmith and Ealing was undertaken as part of a project to investigate relationships between hospital admissions for respiratory illness and traffic-related air pollution, funded by the British Lung Foundation (Wills, 1998; Wilkinson et al., 1999). The study in Sheffield was undertaken as part of a study, funded by the Medical Research Council and Department of Health, to examine links between self-reported asthma and traffic-related pollution in adolescent children (de Hoogh, 1999). The study in Northampton was carried out as part of a project funded by the Engineering and Physical Sciences Research Council to assess the impacts of traffic management on air pollution and exposure (Briggs et al., 1998).

3.2. Data collection

3.2.1. Spatial data

For each study area, a spatial database was first compiled within a geographical information system (ArcInfo or ArcView). Data comprised: road network; traffic flow; altitude; land cover; monitored NO\textsubscript{2} concentrations; and meteorology (temperature, windspeed, atmospheric pressure). Details of data sources are given in Table 3.

3.2.2. Pollution monitoring

Data on NO\textsubscript{2} concentrations for a set of reference sites in each area were obtained from passive diffusion tube surveys (Palmes et al., 1976). In Huddersfield, Northampton and Sheffield, purpose-designed surveys were conducted by the authors. In Huddersfield, duplicate tubes were exposed at 20 sites for 21 consecutive periods of 2 weeks each, throughout the study period (March–December 1995). Data from one survey (14) were, however, lost due to operational problems, giving a total of 20 usable surveys. In Sheffield, five surveys of 2 weeks each were conducted during the period from July 1997 to March 1998; duplicate tubes were exposed at 28 sites on each occasion. In Northampton, tubes were used in triplicate at 39 sites during six surveys of 2 weeks each, over the period July 1997–June 1998. In Hammersmith and Ealing, data were initially obtained for 17 sites for the year April 1995–March 1996 from the South East Institute of Public Health, which co-ordinates pollution monitoring in London. These comprised monthly mean values for each site. Because of gaps in the records, reliable estimates of mean annual con-

<table>
<thead>
<tr>
<th>Year</th>
<th>Huddersfield</th>
<th>Hammersmith and Ealing</th>
<th>Northampton</th>
<th>Sheffield</th>
</tr>
</thead>
<tbody>
<tr>
<td>1995</td>
<td>20</td>
<td>&lt;12</td>
<td>6</td>
<td>5</td>
</tr>
<tr>
<td>1996</td>
<td>2</td>
<td>4–5</td>
<td>2</td>
<td>2</td>
</tr>
<tr>
<td>No. of sites</td>
<td>20</td>
<td>17</td>
<td>39</td>
<td>28</td>
</tr>
<tr>
<td>No. complete</td>
<td>20</td>
<td>11</td>
<td>35</td>
<td>28</td>
</tr>
<tr>
<td>Mean (µg/m\textsuperscript{3})</td>
<td>50.0</td>
<td>42.1</td>
<td>27.0</td>
<td>46.4</td>
</tr>
<tr>
<td>S.D. (µg/m\textsuperscript{3})</td>
<td>16.4</td>
<td>10.7</td>
<td>8.8</td>
<td>14.7</td>
</tr>
<tr>
<td>Inter-quartile range (µg/m\textsuperscript{3})</td>
<td>24.0</td>
<td>19.5</td>
<td>14.8</td>
<td>22.2</td>
</tr>
<tr>
<td>Inter-sampler CV (%)\textsuperscript{a}</td>
<td>4.3</td>
<td>n/a</td>
<td>15.9</td>
<td>7.7</td>
</tr>
<tr>
<td>Inter-survey CV (%)\textsuperscript{b}</td>
<td>13.2</td>
<td>n/a</td>
<td>12.4</td>
<td>11.0</td>
</tr>
<tr>
<td>Inter-site CV (%)\textsuperscript{c}</td>
<td>32.8</td>
<td>25.4</td>
<td>32.7</td>
<td>31.8</td>
</tr>
</tbody>
</table>

\textsuperscript{a}Average of coefficients of variation of replicate samplers, across all sites and surveys.
\textsuperscript{b}Coefficient of variation of site means across all surveys.
\textsuperscript{c}Coefficient of variation of annual means across all sites.
centrations could not be derived for every site, and only 11 sites (for which data were available for at least 8 months in each year) were ultimately used for analysis. Summary data, where known, are included in Table 4. In each case, laboratory analysis was carried out ‘blind’, so that pairs were not known to the analyst.

Table 4 shows standard deviations and coefficients of variation (CVs) for samplers, sites and surveys in each study area. In both Huddersfield and Sheffield, the inter-sampler CV is relatively low (4.3 and 7.7%, respectively), but in Northampton it is notably larger (15.9%). In part, this reflects the lower concentrations in Northampton; in addition, however, it appears to be due to somewhat larger measurement errors during laboratory analysis in the Northampton survey. Inter-site CVs range from 25.4% in Hammersmith and Ealing to almost 33% in Huddersfield and Northampton, while inter-survey CVs are similar in all four study areas (11–13.2%).

In order to examine the main sources of variance in the data, a Variance Components analysis was conducted in SPSS. Data were entered into the analysis for replicate tubes at each site (to define within-site variation), for each survey (to define between-survey variation) and for all sites (to define between-site variation). In London, data did not exist for replicate tubes, so within-site variation could not be determined. Results are shown in Table 5. In each case, between-site variation dominates, accounting for between 55% (Northampton) and 78% (Huddersfield) of the total. Between-survey effects make up between 7% (Sheffield) and 22% (Huddersfield) of the total; between sampler effects are minimal (<0.1%) in each case. Substantial interactive effects are also indicated in each study area by the residual term, which accounts for up to one-quarter of the total variance.

Together, these results show that the majority of the variation in air pollution in all four study areas is geographical (i.e. between-site) rather than temporal (i.e. between-survey). Measurement error (between-sampler variation) is small; notwithstanding the recent debate about their reliability (e.g. Heal and Cape, 1997), it is, therefore, evident that the passive samplers show a high degree of reproducibility in these surveys.

In all study areas, missing data occurred for some sites due to loss of, or damage to, passive samplers. In order to obtain ‘best estimates’ of the mean NO₂ concentration at each site, both for each survey (‘survey mean’) and for the whole year (‘annual mean’) a mixed effect model was applied. Analysis was carried out with site and survey effects modelled as fixed factors, and sampler effects as a random factor, using the SPSS (version 8.0) GLM-General Factorial model. The modelled concentrations derived from this procedure were used in all subsequent analyses.

### 3.3. Model application

The SAVIAH model developed in Huddersfield is calibrated to conditions in that area. In applying the model to other urban areas, or other periods of time, it is evidently necessary to recalibrate the model to take account of differences in meteorology, vehicle fleet mix, local topography and other factors which influence the relationship between model predictions and observed concen-

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**Table 5** Variance components in the monitored data

<table>
<thead>
<tr>
<th>Site</th>
<th>Total variance</th>
<th>Between-site (%)</th>
<th>Between-survey (%)</th>
<th>Between-sampler (%)</th>
<th>Residual (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Huddersfield</td>
<td>242866</td>
<td>78.4</td>
<td>11.8</td>
<td>&lt;0.1</td>
<td>9.1</td>
</tr>
<tr>
<td>Hammersmith and Ealing</td>
<td>21427</td>
<td>60.1</td>
<td>15.2</td>
<td>–</td>
<td>24.8</td>
</tr>
<tr>
<td>Northampton</td>
<td>94083</td>
<td>54.7</td>
<td>21.8</td>
<td>&lt;0.1</td>
<td>25.4</td>
</tr>
<tr>
<td>Sheffield</td>
<td>73040</td>
<td>76.5</td>
<td>7.0</td>
<td>0.1</td>
<td>16.1</td>
</tr>
</tbody>
</table>

*No estimate of between-sampler variation possible due to lack of data on duplicates.*
trations. In the original SAVIAH study, the model was built using a total of 80 sample sites. If the method is to be readily applicable to other areas, it clearly needs to be recalibrated using a smaller number of sites. In most urban areas, no more than one or two fixed monitoring sites are usually available for NO$_{2}$; in the UK, however, many towns run a network of 10 or more monitoring sites using passive samplers either as part of the national diffusion tube survey (Campbell et al., 1994), or independently. Typically, these sites are stratified on the basis of the DETR classification to represent three types of pollution environment: kerbside sites (within 1–5 m of major roads); intermediate sites (20–30 m from major roads); and background sites (more than 50 m from major roads). These sites thus provide a potentially useful basis for calibrating the SAVIAH model in these towns. Significantly, in the original SAVIAH study, 8–10 sites were used to validate the model in each of the study areas, and these gave consistently strong correlations. It might be expected that local calibration of the model could thus be achieved with a similar number of sites.

In Hammersmith and Ealing, only 11 usable sites were available; for other areas, however, data for between 20 and 39 sites were available. This gave the opportunity further to validate the locally calibrated model against an independent set of data. Thus, in all areas except Hammersmith and Ealing, a two-stage analysis was carried out:

1. local calibration of the SAVIAH model against a randomly selected set of 10 sites; and
2. validation of the locally adjusted model against the remaining sites.

In each case, computation of the SAVIAH model was carried out in the GIS. The traffic factor was computed by buffering around each site at a radius of 40 and 300 m, then intersection with the road traffic coverage. Computation of the land factor was carried out by intersecting the 300-m buffer with the land cover coverage. Altitude was determined by dropping the co-ordinates of the monitoring sites onto a coverage from the digital terrain model. The variables so extracted were then weighted, according to the SAVIAH model, and aggregated to provide a measure of mean annual NO$_{2}$.

In order to select a representative subset of sites for calibration, sites were initially stratified in terms of their DETR classification and land use type. Sites were then drawn randomly from each stratum, on a proportional basis.

Predicted concentrations were compared with observed values (derived from the mixed effect model) for the 10 calibration sites in each area, both graphically and using regression analysis. The regression coefficients (constant and slope of the regression line) thus derived gave a calibration which allowed the model to be adjusted to local conditions. Computed values for the remaining sites were adjusted using these coefficients, to give locally calibrated estimates of mean annual concentrations. In order to validate this locally adjusted model, the adjusted estimates were then compared with the observed concentrations for these sites using regression analysis.

**Table 6**

<table>
<thead>
<tr>
<th>Location</th>
<th>$n$</th>
<th>$r^2$</th>
<th>$P$</th>
<th>Constant</th>
<th>Slope</th>
<th>S.E.E. ($\mu g/m^3$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Huddersfield</td>
<td>10</td>
<td>0.51</td>
<td>0.020</td>
<td>$-14.29$</td>
<td>2.08</td>
<td>10.98</td>
</tr>
<tr>
<td>Hammersmith and Ealing</td>
<td>11</td>
<td>0.76</td>
<td>0.001</td>
<td>$-0.30$</td>
<td>0.97</td>
<td>5.58</td>
</tr>
<tr>
<td>Northampton</td>
<td>10</td>
<td>0.60</td>
<td>0.008</td>
<td>$-10.6$</td>
<td>0.96</td>
<td>6.86</td>
</tr>
<tr>
<td>Sheffield</td>
<td>10</td>
<td>0.61</td>
<td>0.000</td>
<td>$-32.2$</td>
<td>1.93</td>
<td>10.31</td>
</tr>
</tbody>
</table>
Fig. 1. Calibration of SAVIAH model in the four study areas.
4. Results

4.1. Calibration

Results of the calibration analysis are shown in Table 6 and Fig. 1. As these indicate, local calibration of the SAVIAH model generates strong and statistically significant regression models in all four study areas. $r^2$ values range from 0.51 ($P = 0.02$) in Huddersfield to 0.76 ($P = 0.001$) in Hammersmith and Ealing. As is to be expected, however, both the regression coefficients (intercept and slope) and standard errors of the estimates vary substantially between the study areas, broadly reflecting differences in the general level of pollution (see Table 4). In both Huddersfield and Sheffield, the SAVIAH model thus tends to under-estimate NO concentrations; in Northampton it tends to over-estimate concentrations.

4.2. Validation

Validation of the adjusted models was only possible in three areas — Northampton, Huddersfield and Sheffield — where sufficient additional monitoring sites were available. Results are given in Fig. 2 and Table 7. As these show, the adjusted models performed consistently in all three areas, with $r^2$ values somewhat better than those in the calibration study (0.58, 0.76 and 0.73, respectively) and with intercept values close to zero and slope coefficients between 0.9 and 1.2. S.E.E. values were also reduced slightly compared to those in the calibration study (5.20, 9.78 and 7.73 μg/m$^3$, respectively).

Fig. 3 shows distributions of the residuals from the validation studies, expressed as a percentage of the monitored concentration. In Huddersfield, 70% of the modelled estimates are within a factor of 1.5 of the monitored concentrations (i.e. 0.67 observed $< $ predicted $< $ 1.5 observed), and the same percentage within a factor of 2 (i.e. 0.5 observed $< $ predicted $< $ 2 observed). In Northampton, the equivalent statistics are 90 and 96%; in Sheffield they are 94 and 100%, respectively. The strong negative skew in the Huddersfield data indicate that the calibrated model is still tending to over-estimate concentrations in the study area to some extent. In the other two areas, the residuals are more symmetrically distributed.

5. Discussion and conclusions

The results of applying and testing the SAVIAH regression model in the four study areas show that the original model provides a reliable relative measure of mean annual NO concentrations across a wide range of urban environments, but may substantially under- or over-estimate actual concentrations. The model may be successfully calibrated at the local level using only a small number (approx. 10) of passive sampler sites, monitored for only a few (approx. 5–7) 2-week periods in any year. The quality of the local calibration inevitably depends, however, on the distribution of these sites monitoring sites. It is important that they are located to reflect the range of actual values in the study area, including both background and kerbside sites. With local calibration in this manner, the model can provide estimates of mean annual NO$_2$ concentrations within a factor of 1.5 of the actual mean (approx. 70–90%) of the time and within a factor of 2 between 70 and 100% of the time. $r^2$ values between modelled and observed mean annual concentrations are in the range of 0.58–0.76. It may also be noted that the model performs to a more or less similar level when applied for indi-
Fig. 2. Validation of the locally calibrated SAVIAH model in three study areas.
Fig. 3. Residuals from the locally calibrated model, expressed as a percentage of the monitored concentration.
individual monitoring periods (i.e. 2-week passive sampler campaigns). Detailed results are not presented here, but in Sheffield, for example, $r^2$ values of 0.52–0.68 (with 100% within a factor of 2) were reported for five individual 2-week surveys of 28 sites each (de Hoogh, 1999). In Huddersfield, $r^2$ values of 0.73–0.78 were found between modelled and monitored concentrations at eight sites for three 2-week surveys, with 100% within a factor of 2 (Smallbone 1998).

The accuracy of the SAVIAH method can usefully be compared with that from more sophisticated dispersion models. Results from validation studies of dispersion models need to be interpreted with caution, for such studies are often undertaken in relatively simple, purpose-designed situations, which do not necessarily reliably simulate the range of pollution levels, emission source conditions, and dispersion pathways found across an urban environment. Most also validate the model against short-term (e.g. hourly) concentrations rather than longer-term, mean pollution levels.

van Pul et al. (1996) compared three models — the GH (Gifford and Hanna, 1973) model, a box model, and a combination of the two — in terms of their ability to predict annual average concentrations of NO$_2$ at seven stations in six cities. Coefficients of determination ($r^2$) were 0.02 for the Box model, 0.45 for the GH model, and 0.44 for the Box–GH model. The percentage of predictions within a factor of 2 of the observed daily average was 0% for the Box model, and 83% for the GH and Box–GH models. Inclusion of the regional background concentration in the model improved the correlations with measured concentrations — $r^2$ values rose to 0.21, 0.62 and 0.62 respectively — but the predictions within a factor of 2 remained unaltered. Rao et al. (1989) compared the performance of seven line-source models (GM, AIRPOL-4, HIWAY, CALINE-2, DANARD, MROAD2, and ROADS) against 594 hourly observations of a tracer gas (SF$_6$). Coefficients of determination ranged from 0.14 (DANARD) to 0.83 (GM); the percentage of predictions within a factor of 2 ranged from 27% (ROADS) to 87% (GM). Yamartino and Wiegand (1986) assessed the performance of three models against half-hourly measurements at two sites in Cologne. Predictions within a factor of 2 were not quoted, but $r^2$ values ranged from 0.44 (for the MAPS and STREET models in Bonner Strasse) to 0.62 (the CPB model). Benson (1992) quotes $r^2$ values of 0.26–0.76 (approx. 85% of predictions within a factor of 2) for measurements of the tracer gas, SF$_6$. Namdeo and Colls (1996) reported an $r^2$ value of 0.68 for correlations between predicted CO and observed concentrations for 104 hourly measurements in Leicester, UK, using the SBLINE model. Predictions within a factor of 2 were not given, but can be estimated as approximately 68% from the data presented. In a study for one site in Dublin, Reynolds and Broderick (1999) compared modelled concentrations from CALINE-4 with monitored concentrations for a range of pollutants. $r^2$ values were 0.80 for CO, 0.86 for NO$_x$, and 0.81 for PM$_{10}$; factor-of-2 values were 71% for all three pollutants.

In recent years, a range of so-called ‘new generation’ models have also been developed, which incorporate improved parameterisation of boundary layer conditions. Two of these — ADMS (CERC, 1999) and AERMOD (USEPA, 1998) — are being widely adopted for regulatory purposes. Relatively few field validations of these models have yet been published and many of those which have been reported refer to non-line emission sources. Hanna et al. (1999), for example, compared the performance of both models against hourly data from five tracer experiments in the USA, representing point, area or volume emission sources. AERMOD was found to under-predict tracer concentrations by 20% on average, while AERMOD under-predicted by 40%. Factor-of-two values were 53% for ADMS and 46% for AERMOD. McHugh et al. (1999) report factor-of-two values of 46–67% for ADMS from three tracer experiments (including two also used in the Hanna et al., 1999 study), compared to 29–76% for AERMOD. Carruthers et al. (1999) present results from using ADMS with two different chemistry sub-models for NO$_2$ and NO$_x$ monitored at 12 sites in London. For the annual mean, all modelled concentrations were within a factor of 2 of monitored concentrations, though
the model tended to over-estimate or under-estimate depending on the chemistry model used. In Sheffield, in one of the studies reported in this paper, de Hoogh (1999) assessed the performance of ADMS (both with and without the hill option) against monitored NO\textsubscript{2} data for the 28 monitoring sites. For the annual mean, and using linear regression, \(r^2\) was 0.42 with the hill option, and 0.53 without. The relationship was, however, strongly curvilinear, and was better represented by an S-curve (\(r^2 = 0.78\) and 0.79, respectively). In both cases, ADMS under-estimated monitored concentrations and less than 10\% of modelled concentrations were within a factor of 2.

These comparisons suggest that the accuracy of the SAVIAH model presented here is broadly equivalent to — and to some extent better than — that of available dispersion models in simulating long-term pollutant concentrations (i.e. over periods from 2 weeks to 1 year or more). Unlike most dispersion models, however, the technique cannot realistically be applied to shorter-term (e.g. hourly) averaging periods. While it offers a useful technique for modelling potential compliance against long-term air pollution standards, or to assess chronic exposures, it does not offer a reliable means of assessing short-term concentrations, either in relation to air pollution standards or as part of acute epidemiological studies.

Various other factors also need to be considered in evaluating the SAVIAH methodology, including its adaptability to other pollutants, cost (both of purchase or development and operation) and ease of use. To date, the model has been developed and calibrated only for one pollutant (NO\textsubscript{2}); most dispersion models, in contrast, can be applied to a range of pollutants (e.g. CO, NO\textsubscript{x}, PM\textsubscript{10}). There is, however, no reason why the SAVIAH approach should not be used, with equal success, for these other traffic-related pollutants so long as data can be obtained from a suitably dense network of training sites. The regression method also has the advantage of being cheap to apply and easy to use compared to many formal dispersion models. It can easily be programmed in most proprietary GIS and is relatively simple in its data demands. Many district councils in the UK already maintain diffusion tube networks comprising at least 10 sites, either as part of the national monitoring network or for local state of environment reporting and policy purposes. Relevant input data (e.g. on traffic flows, land cover and altitude) are also readily available in most cases, through routine monitoring activities and local GIS. For most local authorities, therefore, it should be possible to apply the SAVIAH regression model without the need for additional monitoring or data acquisition. As such, it can provide a useful screening method to help target attention at key areas, and to design more detailed monitoring and modelling strategies. Where detailed maps of NO\textsubscript{2} are needed for epidemiological studies of chronic health effects, it should similarly be feasible to apply the method using existing, readily available data, supplemented if appropriate by purposely designed passive sampler surveys for further validation of the results. With the growing need for accurate, high-resolution data on air pollution, the SAVIAH model is thus a potentially valuable tool.

References


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