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Evaluation of a neighbourhood scale, street network dispersion model through comparison with wind tunnel data

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ABSTRACT

This study compared dispersion calculations using a street network model (SIRANE) with results from wind tunnel experiments in order to examine model performance in simulating short-range pollutant dispersion in urban areas. The comparison was performed using a range of methodologies, from simple graphical comparisons (e.g. scatter plots) to more advanced statistical analyses. A preliminary analysis focussed on the sensitivity of the model to source position, receptor location, wind direction, plume spread parameterisation and site aerodynamic parameters. Sensitivity to wind direction was shown to be by far the most significant. A more systematic approach was then adopted, analysing the behaviour of the model in response to three elements: wind direction, source position and small changes in geometry. These are three very critical aspects of fine scale urban dispersion modelling. The overall model performance, measured using the Chang and Hanna (2004) criteria can be considered as 'good'. Detailed analysis of the results showed that ground level sources were better represented by the model than roof level sources. Performance was generally 'good' for wind directions that were very approximately diagonal to the street axes, while cases with wind directions almost parallel (within 20°) to street axes gave results with larger uncertainties (failing to meet the quality targets). The methodology used in this evaluation exercise, relying on systematic wind tunnel studies on a scaled model of a real neighbourhood, proved very useful for assessing strengths and weaknesses of the SIRANE model, complementing previous validation studies performed with either on-site measurements or wind tunnel measurements over idealised urban geometries.

1. Introduction

Operational dispersion models are commonly used to determine the exposure of the population to pollutants in urban areas and to help develop strategies for its reduction. This requires mapping of pollutant concentrations throughout an area of interest, generally in terms of both time and spatial scales. Atmospheric dispersion models are essential for that purpose because the data provided by monitoring stations cannot provide a sufficiently detailed coverage (Ball et al., 2008; Scaperdas and Colvile, 1999). The temporal and spatial details to which dispersion models must extend depends on their purpose and application. Traditionally (Oke, 1987), the scale of atmospheric motions and related phenomena have been classified according to their horizontal scale into four broad categories: macroscale, mesoscale, local scale and microscale. Britter and Hanna (2003) interpret these as 'regional' scale (up to 100 or 200 km), 'urban' (city) scale (up to 10 or 20 km), 'neighbourhood' scale (up to 1 or 2 km) and 'local' (street) scale (less than 100–200 m). These horizontal scales are strongly linked to corresponding vertical and temporal scales.

Britter and Hanna (2003) argue that the neighbourhood scale is a spatial scale over which some statistical homogeneity can be anticipated and thus general parameterisations of the flow can be attempted. It is also a scale at which detailed computational study is feasible with the required high resolution implied in this form of investigation. Many recent urban modelling development efforts have concentrated on the neighbourhood scale (see, e.g., Belcher, 2005; Hamlyn et al., 2007; Di Sabatino et al., 2008). Dispersion models at this particular scale must be able to deal with the mechanisms that determine pollutant transport within and above the urban canopy (Britter and Hanna, 2003). These are complex and chiefly related to the channelling effect due to advective transport

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along streets, vertical mass exchanges between street canyons and the overlying atmosphere and pollutant transfer at street intersections (Carpentieri et al., 2009; Carpentieri and Robins, 2010). Current 'operational models' describe these processes to some degree or another by means of parametric relations.

Standard methods for predicting urban air pollution, resulting primarily from emissions from traffic, have been reviewed by Vardoulakis et al. (2003). The street canyon model described by Berkowicz (2000) is one of the well-established, standard models for this purpose. Like all operational models, it uses a simplified description of flow and dispersion conditions in street canyons, in this case based on a box model that treats pollutant recirculation and exchange with the external flow, and a plume model that describes the direct advection of pollutant from source to receptors within a street canyon. OSPM (Operational Street Pollution Model), an operational version of the model, has been extensively tested against observations (e.g. Kukkonen et al., 2003). The same approach is used in ADMS-Urban where, again, it has been extensively tested against observations (e.g. CERC, 2001). OSPM and its derivatives can be regarded as mature systems with a performance that is now firmly established. Their strength is in predicting exposure to pollution from traffic flows when the sources and receptors are in the same street canyon. It does not though treat the exchange of pollutants between streets at intersections, other than in a very general sense through the background concentration field.

Models that are to handle the dispersion of hazardous material from point-like sources in urban areas must represent not only dispersion within a street canyon and mixing with the external flow but also exchanges at intersections, as contaminated air works its way through a street network (Hunt et al., 2004). The basic street canyon model needs some additional capabilities to enable this. An alternative approach is based on treating urban areas as a canopy, through which the flow is determined by a balance between the driving shear force from above and the drag within the canopy. This is the basis for models such as UDM, Urban Dispersion Model, (Hall et al., 2001; Brook et al., 2003) and the so-called Baseline Urban Dispersion Model (Hanna et al., 2003). In the latter, dispersion is treated through simple modifications to standard plume spread relationships whereas in the former a more complex approach is adopted based on statistical relationships between dispersion and the geometrical characteristics of the urban area, together with explicit treatment of plume partitioning by interactions with large buildings (i.e. large relative to the plume). These models have been extensively tested against field and wind tunnel data and their limitations understood.

Street network models, of which SIRANE (Soulhac et al., 2011) is the prime example, are relatively new and consequently their strengths and weaknesses less well understood. They are box models at heart but formulated with explicit modelling of the exchange of flow and pollutant fluxes at urban intersections and capable of providing a detailed concentration field at the street scale over a whole urban district. SIRANE was developed at the Laboratoire de Mecanique des Fluides et d'Acoustique (LMFA) de l'Ecole Centrale de Lyon. In this work, we are interested in evaluating how this class of 'street network' model performs in a real geometry, and in the sensitivity of its performance to input data and application.

For model evaluation purposes, there are many collections of good quality data from urban dispersion experiments, from field – e.g. Salt Lake City (Allwine et al., 2002); London (Wood et al., 2009) – and wind tunnel work (e.g. the CEDVAL data-base at the University of Hamburg). A first evaluation of the performance against a field dataset collected in a district of Lyon, has been already performed and is reported elsewhere (e.g. Soulhac et al., 2003, 2012). The sensitivity analysis showed that the model outputs were mainly sensitive to errors in two input parameters: the intensity of pollutant sources and the wind velocity. In the present work, the uncertainties related to both inputs are minimised, since the reference dataset refers to wind tunnel experiments, with a fully controlled environment with steady and fully defined boundary conditions. This allows us to focus on the influence of uncertainties arising from other input parameters, which would be difficult to analyse in field experiments. This is the case, for example, for local changes in the geometry of the street network, that are easy to do in small-scale urban models. Furthermore, this allows us also to test some of the parameterisations implemented in the model, such as those related to pollutant dispersion in the wind field above the urban canopy. Tests of the parameterisation for the pollutant transfer within the canopy have already been the object of a previous study (Garbero, 2008) that was performed by reducing as far as possible the irregularities of the urban geometry on a physical domain that was as close as possible to the conceptualisation of the modelled system (see Fig. 1a).

In comparison with the methodology developed by Jakeman et al. (2006) and applied, e.g., by Blocken and Gualtieri (2012) for CFD (Computational Fluid Dynamics) environmental models, the evaluation presented here covers part of the objectives included in the last two steps of the development and evaluation of an environmental model. The results presented here, complemented by those of other validations and analyses (e.g. Soulhac et al., 2003, 2012; Garbero, 2008), provide the essential information needed to understand and explain the limitations of the model performance in realistic urban settings. A further step can now be contemplated, return to and revise the definition of the parametric laws adopted by the model. The new version then has to be tested against experimental data, giving rise to an iterative evaluation process for the improvement of the overall model performances.

For critical examination of model performance we need an exhaustive data-base that can be used to investigate effects of wind direction, source location and receptor location, implying a degree of systematic variation in those parameters. Ideally, the geometry should be relatively easy to define so that it can be simulated in the model without need of any significant simplification. Finally, a sufficiently large set of data is required to ensure statistical confidence in the results. Wind tunnel work undertaken in the Environmental Flow Research Centre (EnFlo) at the University of Surrey as part of the DAPPLE project (Dispersion of Air Pollutants and their Penetration into the Local Environment; Arnold et al., 2004; Wood et al., 2009) provides data that meet these requirements and has therefore been used for the model evaluation. The data and final reports from the project are freely available from www.dapple.org.uk.

The statistical indices proposed by Chang and Hanna (2004) are very widely applied to evaluating the performance of atmospheric dispersion models. Four of the indices are used here, the fractional bias, the normalised mean square error, the correlation coefficient and the fraction of predictions within a factor of two of the observations. Chang and Hanna (2004) define a 'good' model in terms of numerical ranges for these parameters. We adopt that procedure here and use the description 'a good model' in this formal sense. We also use scatter plots of predicted and observed concentrations and decay plots showing how concentrations decay away from a source to shed further light on model performance.

Section 2 provides a brief review of the basis of the SIRANE model, Section 3 the wind tunnel work and Section 4 the model set-up. Two sub-sets of the full DAPPLE data-base are used, the first to examine overall features of model performance (Section 5) and the second to study response to a range of systematic variations in the boundary conditions (Section 6). Section 7 summarises the overall outcome of the study.