A review of urban dispersion modelling

Prepared for the ADMLC by

SE Belcher, O Coceal, JCR Hunt, DJ Carruthers, AG Robins

Version 2, 5th December 2012

1. Introduction

Dispersion modelling in urban areas is a rapidly developing subject fuelled by the accumulation of data from numerous experimental and computational studies and the consequent increase in understanding that those studies have generated. This review focuses on advances in the understanding of dispersion from localised sources in urban areas, with special emphasis on recent developments since the review of Hunt et al. (2004), also commissioned by ADMLC. For the purpose of this review, an urban area is any region consisting of a group of buildings; the buildings may range from very sparse to very dense, or from low rise to very tall towers. Part of our aim is therefore to document the approaches and tools appropriate for modelling dispersion in these different regimes.

There are different reasons for making quantitative predictions of dispersion from localised releases, and therefore different objectives in developing any calculation procedures and systems. The practical situations in which these calculations are performed affect the methods that are used. Predictions are needed in real time, firstly in a 'direct mode' when an accidental release of a known strength and at a known location has occurred. Secondly, in some situations, predictions in the 'inverse mode' are needed because the exact location of the source is not known even though the effects on people of a toxic release of gas may already be apparent. Thirdly, 'scenario' predictions are needed for evaluating the effects of different types of accidental releases in all the relevant meteorological conditions in order that responses to likely eventualities can be planned in advance, and their relative risks assessed. Whereas the dispersion predictions in the first and second cases have to be performed, at least for initial estimates, in a matter of a few minutes, for the third case hours to days might be acceptable.

For each of the three cases the end goals are similar - to calculate the statistics of the concentration denoted by $\langle C\rangle(x,t;\,\Delta x,\Delta t)$ at a point in space, x, and time t averaged over certain spatial and temporal intervals $\Delta x,\Delta t$. The statistics required are the mean concentration, $\bar{C}$, r.m.s. values of the fluctuations, C', and extreme values $\hat{C}$. In ideal atmospheric conditions in air flow over level terrain with slowly changing meteorological conditions, the form of the mean concentration profiles, $\bar{C}$ and C', due to localised sources are well-defined and repeatable (i.e. the plume profiles). Models for these circumstances have been well validated: see, for example, Carruthers *et al.*, 1994 (ADMS 4); Cimorelli *et al.*, 1998 (AERMOD); Olesen *et al.*, 1992 (OML); Thykier-Nielsen *et al.*, 1999 (RIMPUFF); Scire *et al.*, 1999 (CALPUFF). For some applications it is the time-integrated concentration, or dosage, and its related statistics that are required. In this review we focus largely on dispersion in statistically steady flows, so that there is a straightforward relation between concentration and dosage.

A key question for this review is whether dispersion patterns from local sources have repeatable characteristic forms in urban areas as they do over level terrain. If so, this would enable robust and easily computable formulae to be used for the most common situations. In fact, this methodology has already been applied at the building/street scale in the models for dispersion from steady line sources (of vehicles) in urban street canyons (e.g. Operational Street Pollution Model OSPM Berkowicz *et al.*, 1997; SIRANE Soulhac *et al.*, 1998). It has also been applied to dispersion around hills, over roughness changes and around isolated buildings where plumes/puffs have forms that are simple modifications of the Gaussian profiles (e.g. a double plume structure in the wake of a building). The basic concepts for plumes were developed by Hunt and Mulhearn (1973), Puttock and Hunt (1979), Hunt *et al.* (1979) and Davidson *et al.* (1995). Robins and Apsley (2000) brought many of the results together for single buildings in the Buildings Module of the ADMS model. This model is designed for predicting air quality due to routine releases rather than accidental releases. Both this building module and a modified version of the OSPM street canyon module are used within the framework of the ADMS-Urban model (Carruthers *et al.*, 2000); this uses quasi-Gaussian type models for pollutants once they have been transported out of the street canyon or out of the influence of building wakes. The plume spread parameters are calculated from vertical profiles of mean wind speed and turbulence determined from a uniform roughness representative of the urban area or by the quasi-linear model FLOWSTAR (Carruthers *et al.*, 1988) which calculates the impacts of changes in local terrain elevation and surface roughness on mean wind and turbulence.

This approach has also provided the basis of the accidental release model UDM (Hall *et al.*, 2001) which was originally developed to take account of the interaction of wakes in areas with low building density by modelling appropriately the changed cloud dimensions. However, in close-packed buildings and in street canyons with arbitrary source distributions the mean spatial distributions of concentration are significantly non-Gaussian at least on the building/street scale. The model has been developed for application to high-density urban areas, through suitable empirical adjustment, and by extensive wind tunnel and field validation. In such situations it is inevitably less accurate, nevertheless, comparisons with field data have shown (Brook *et al.*, 2002) that the UDM model describes many of the observed plume features. UDM may also be applied to the large numbers of buildings over the neighbourhood scale (300 to 3,000m approximately) by applying cloud splitting and merging techniques in which groups of overlapping split plumes are merged and formed into new, larger plumes.

For dispersion on larger scales (e.g. whole urban area), matter from the accidental source is dispersed throughout the internal boundary layer or the mixed layer. It is then usual to assume that the dispersion is similar to that over level ground with an appropriately enhanced level of roughness length z_o, e.g. as in the Gaussian plume model typically used for simple terrain. However, as plumes or clouds of matter travel over extended urban scales, the travel time becomes long enough that the air flow and the atmospheric stability can change, so that simple models can be seriously in error.

The emphasis in this review is on identifying the physical processes that control dispersion in urban areas through measurement and detailed simulation, and then the representation of these processes in simplified practical models. In so doing we aim to identify when the different existing methods are appropriate and when new methods are required. This analysis then identifies an important regime of dispersion in urban areas when the lateral dispersion is controlled by transport through the network of streets in the urban area. Our discussion of modelling methods then reviews in some detail these new *street network models* of urban dispersion.

The review is organised as follows. Section 2 briefly reviews the processes affecting dispersion at the city scale. Since this subject is described in detail in the previous reviews of Hunt et al. (2004), Britter and Hanna (2003) and Fernando (2010), the focus here is on two recent developments: (i) the characterisation of drag in terms of building canopy characteristics and its effect on the adjustment of winds within and above the urban canopy, which strongly affect dispersion and (ii) the control of the urban surface energy balance on the stability and turbulence and thence the dispersion characteristics in the urban boundary layer.

Section 3 is an extended review of some of the recent empirical findings on flow and dispersion processes at the street and neighbourhood scales. It represents the collective wisdom generated by numerous field and wind tunnel experiments as well as numerical studies especially over the last decade or so. The importance of such a review for dispersion modelling is that it sets out the key processes and features that dispersion models should represent or reproduce, and identifies key studies and sources of data that can be used to evaluate these models. This section is split into three parts. The first part discusses the flow and dispersion processes associated with a number of urban elements (some of which may be idealised), namely two-dimensional street canyons, intersections and tall buildings. In the second part, the combined effect of many of these processes in a complex, realistic geometry is then illustrated with case studies from the DAPPLE field experiments in London. It is then shown that, despite the complexity of the flow and dispersion in these environments, the decay of maximum concentration with distance follows a very simple empirical $1/r^2$ law, where r is the distance between the source and receptor. This $1/r^2$ law is also reproduced in experiments over a wind tunnel scale model as well as in field studies in other cities. Part three of this section then reviews recent results from large-eddy simulations and direct numerical simulations of urban flow and dispersion. The empirical input of studies using these methods is one of the major developments since the review of Hunt et al. (2004), and plays an increasingly important role alongside experimental studies.

In Section 4 we review the range of approaches for modelling dispersion within and above urban areas. To set the context, we propose a regime diagram delineating the link between the geometrical parameters characterising an urban area and the appropriate modelling approach in each regime. Three broad regimes are identified, corresponding to sparse arrays, street networks and tall building canopies. The methods for modelling sparse arrays are well-known, having been previously discussed in detail by Hunt et al. (2004), and hence only a brief review is given (including a brief description of the UDM model). On the other hand significant new developments have taken place for the street network regime, and considerable space is therefore devoted to explaining both the physical basis and operational implementations of these so-called street network models (e.g. SIRANE and SIRANE-RISK). We also critically examine the capabilities and limitations of these models. An alternative approach in this regime is provided by street canyon models, exemplified by ADMS-Urban. Recent years have also seen the development of quick methods for approximate building resolving simulations (e.g. using parameterised and mass-consistent schemes such as in QUIC and MSS). These models are equally applicable to the previously mentioned regimes and are briefly reviewed. The third regime mentioned above, the tall canopy regime, is particularly relevant for cities consisting of densely packed tall buildings such as central New York or Hong Kong. Modelling capability for this regime is currently lacking, but we propose a porous drag approach similar to that employed for many years in vegetation canopies (reviewed by Finnigan 2000 and Belcher et al 2012) and also developed recently for urban canopies (Belcher et al. 2003, Coceal & Belcher 2004). Section 4 ends by considering the effect on flow and dispersion due to the impact of the boundaries of a city as a whole. These larger-scale effects are frequently neglected but can be significant.

We conclude in Section 5 with a summary of the different modelling approaches, their main limitations, and ideas and recommendations for further progress in urban dispersion modelling.

2. The urban atmospheric boundary layer

The city-scale flow and the urban atmospheric boundary layer set the context for any dispersion within urban areas. Hunt et al (2004) describe in detail a range of processes that control the flow at the city scale, and Britter & Hanna (2003) and Fernando (2010) review mean flow, turbulence and dispersion on the city scale.

Here, therefore, we consider city-scale flow and turbulence only briefly. We focus on the two aspects that are highlighted as needing additional effort in Hunt et al (2004), namely (i) the role of building form and layout on the development of winds in the boundary layer and (ii) the surface energy balance of urban areas which determines the stability of the urban boundary layer, and thence the turbulence statistics that control dispersion. In each of these areas there has been substantial progress since the Hunt et al (2004) review.

In order to see how these processes affect dispersion within the urban canopy, consider the Gaussian plume model (e.g. Pasquill & Smith 1983), which demonstrates how the mean flow and turbulence affect dispersion of a scalar. The concentration of scalar, *C*, depends on mean wind speed, *U*, and cross-wind and vertical turbulence variances, σ_y^2, σ_z^2, which can be written generically as

$$C(x,y,z)=\frac{Q}{2\pi U\sigma_y\sigma_z}f_y\left(\frac{y^2}{\sigma_y^2}\right)f_z\left(\frac{z}{\sigma_z}\right). \qquad (2.1)$$

Here *x* is the along-wind direction, *y* is the cross-wind direction and *z* is vertical, *Q* is the (steady) source of scalar and f_y, f_z are the functions defining the cross wind and vertical shape of the plume. It is clear that (i) as the mean wind speed increases the mean concentrations are reduced, because the plume is stretched more in the windward direction, and (ii) as the turbulence variances increase the concentration is reduced because of enhanced mixing. So how do these properties compare between urban and rural areas?

2.1 Winds in the urban boundary layer

Firstly consider the variation of wind speed within a building canopy with packing density of the buildings. The buildings of the urban area represent roughness elements to the boundary layer above. In most cases the boundary layer is sufficiently deep that the buildings lie below the roughness sublayer of the atmospheric boundary layer (Rotach 1995). The bulk effect of the buildings can then be represented as a roughness length z_0 and displacement height d, with typical values of $z_0 \sim 1m$ and $d \sim H$ the mean height of the buildings. These values are larger than for rural terrain, for example over crops $z_0 \sim 0.1m$ and $d \sim$ crop height.

The values of z_0 and d depend on the form and density of the buildings. Consider a regular array of cuboidal buildings, with a square base of dimensional *L* x *L* and height *H*, with equal spacings of width *W* between them. Such an array can be characterised by the plan area index and the frontal area index (Britter and Hanna 2003) defined respectively by

$$\lambda_p=\frac{\text{plan area}}{\text{total plan area}}=\frac{L^2}{(L+W)^2}$$
$$\lambda_f=\frac{\text{frontal area}}{\text{total plan area}}=\frac{HL}{(L+W)^2} \qquad (2.2)$$

For cubes H = L and $\lambda_f = \lambda_p = \lambda$. Figure 2.1, taken from Coceal & Belcher (2004) shows the variation for cubes of roughness length and displacement height with λ. The normalised roughness z_0/H increases and peaks for $\lambda \sim 0.15$, corresponding to $W/H \sim 1.6$. The normalised roughness then decreases with increasing packing density (decreasing W/H).

Of great interest in this review is the wind within the urban canopy itself, because this wind carries scalar material through advection. The mean winds within the urban canopy, U_c, are proportional to u_*, the friction velocity in the surface layer above (Britter & Hanna 2003). Hence, for fixed synoptic forcing conditions, the characteristic wind speed within a canopy of cubes decreases as the packing density increases from zero because the roughness length increases, reaches a minimum at $\lambda \approx 0.15$ and then increases.

For more complex arrays of buildings with more complex shapes a range of methods have been developed for estimating the roughness length and displacement height, but, as reviewed by Grimmond & Oke (1999), simpler methods, such as Macdonald's (1998) method, perform as well as any.

Secondly, the discussion so far has focussed on a boundary layer adjusted to a large urban area with uniform bulk properties (e.g. roughness length, displacement height and energy balance). The question then arises as to the adjustment of a boundary layer flowing from a rural area into an urban area. One approach to modelling the flow is to represent the urban area by a porous volume with a finite resistance; i.e. as a porous canopy. The porosity varies from zero if the buildings are very closely packed, to nearly unity if the flow is hardly disturbed. This approach has been used extensively to model flow through forest canopies (Finnigan 2000, Belcher et al 2012). Belcher et al (2003) and Coceal & Belcher (2004) develop theory and modelling using this method for urban canopies. A code to compute flow through such canopies is available from CERC called FLOWPOR, which has been applied to wind energy. Los Alamos has also developed an urban model based on a similar approach (M.Brown LA-UR-98-3831; 1998).

The basic features of the flow of a boundary layer adjusting from rural to urban terrain can be understood by modelling the urban area as a porous volume, and is shown schematically in Figure 2.2. The figure, taken from Belcher et al (2012), and adapted from Belcher et al (2003), shows contour plots of the evolution of flow across a canopy edge from the LES of Dupont & Brunet (2008). Beyond the canopy edge at x = 0 is an adjustment region (denoted A in Figure 2.2a), of length x_A (corresponding to 0 < x/H < 10 in Figure 2.2a), where the canopy drag decelerates the wind and as a result there is a mass flux from the top of the canopy, which will have an important effect on scalar dispersion. The extent of the adjustment region depends to some extent on the canopy resistance - i.e. to the packing density of buildings in an urban area (Coceal & Belcher 2004). Further downwind, the flow develops as it does downwind of a roughness change. Swirling motions develop when the buildings in the canopy have varying heights (e.g. Britter and Hunt 1979) and the external flow then penetrates further, so that the average wind speed within the canopy $\boldsymbol{U_c}$ increases.

There is also a large change in turbulence structure inside the canopy where it is greatly weakened relative to the external turbulence, which is very significant for dispersion because pollutants can be spread across the canopy in the time it takes for the weak mean flow to pass through it.

Finally, there is a question as to where to measure a reference wind speed to characterise the boundary layer above an urban surface. If the reference wind speed is taken too low down and too near the urban surface then there is a risk that its value is strongly affected by the surrounding individual buildings. Hence measurements need to be taken higher up, so that the effects of individual buildings are mixed out by the turbulence, and the measured wind speed is representative of the urban boundary layer over the larger-scale urban surface. Barlow et al (2009) show that for London measurements on the BT Tower at a height of 190m above the surface provide a wind speed that characterises the urban boundary layer. More generally perhaps mesoscale meteorological models are needed to estimate the evolution of the boundary layer wind speed and thence the wind speed and turbulence within the urban canopy (e.g. Bohnenstengel et al 2011).

2.2 Turbulence and stability of the urban boundary layer

Roth (2000) provides a critical review of the measurements of turbulence in the surface layer of the urban boundary layer up to the year 2000. He established criteria for accepting the measurements and establishing when they lie within the surface layer of the atmospheric boundary layer. He showed that the mean shear and turbulence in the surface layer are related through the standard Monin-Obukhov stability functions. This suggests that the turbulence, at least within the surface layer of the urban boundary layer follows the same scaling relations as rural boundary layers.

Most of the data reviewed by Roth were collected over relatively short measurement campaigns and so little could be said about the seasonal variation of the urban boundary layer. More recently, Wood et al. (2010) have analysed more than a year's continuous data obtained in London on the BT Tower, at a height of 190m above the surface. Normalized standard deviations of the turbulence were found to follow Monin-Obukhov similarity theory well in both unstable and stable stratifications, with values in neutral conditions of $\sigma_u / u_* = 2.3$, $\sigma_v / u_* = 1.85$ and $\sigma_w / u_* = 1.35$ (for the streamwise, spanwise and vertical components), not dissimilar to those found over rural terrain. In addition, the analysis demonstrated that mixed layer scaling applies well when the boundary layer is convective, and that the vertical velocity variance follows the well-known form of Lenschow et al. (1980) found over simpler surfaces. This further supports the view that urban boundary layers follow similar scalings to those found over simpler surfaces.

Figure 2.3, taken from Wood et al (2010), shows the frequency distribution of the local stability parameter measured on the BT Tower between October 2006 and May 2008. Overall, the boundary layer was predominantly unstable, and at night it was as likely to be unstable as stable. In comparison, measurements taken as the 180m-high tower at Cabauw, a flat rural site in the Netherlands, in 1996 show twice as many occurrences of stable boundary layers compared with unstable boundary layers at night (Verkaik & Holtslag, 2007).

Taken together these studies suggest that the structure of the turbulence within the urban boundary layer is the same as the structure of the turbulence in a boundary layer over simpler surfaces. But there is a marked difference in the prevalence of the different stability regimes. In particular, there is a much greater prevalence of unstable conditions at the London site. We conclude that the particular characteristics of the urban boundary layer are driven by the energy balance of the urban surface.

The urban energy balance has received much study in the last 10 years and there is now a range of methods for modelling it: see Grimmond et al (2010, 2011) for an overview and first results of an inter-comparison study. The urban surface has a high heat capacity (partly because of the urban geometry and hence the large area of urban surface in contact with the atmosphere and partly because of the building materials), so that the urban surface absorbs heat during the day and releases heat after sunset into the night. This has two main effects on the urban boundary layer. Firstly, it reduces the diurnal temperature range, which then explains why, at night, urban areas tend to have higher screen level temperatures than the surrounding rural areas. Secondly, it means that after sunset there can be a positive heat flux into the urban boundary layer, driving an unstable urban boundary layer when the rural boundary layer is stably stratified. Harman & Belcher (2006) show the dominant processes in an idealised modelling study.

Many of the numerical weather prediction models used in national weather centres now include some representation of the urban energy balance, for example the Met Office Unified Model has the MORUSES scheme (Porson et al 2010a,b). Bohnenstengel et al (2010) use simulations with the Met Office Unified Model to show that advection from the rural to the urban surface plays an important role in setting the structure of the urban boundary layer, and particularly the depth of the urban boundary layer.

3. Empirical results and data

The aim of this section is to describe, largely empirically, the important processes that govern the flow through a built-up area and the associated dispersion of pollutants emitted within that area. The focus is on the short range, loosely defined as the area within a few streets and intersections of a release position, perhaps up to a kilometre or two of the source. The key elements of this picture are described individually and in combination as an urban street network.

Much of our current knowledge of urban dispersion processes originates from a combination of scaled wind tunnel and field experiments as well as field studies at actual urban sites. More recently, computational studies using large-eddy simulations and direct numerical simulations have played an increasingly important role in complementing the experimental work. The last part of this section highlights some of the advances made using these approaches.

3.1 Elementary urban units and associated dispersion processes

3.1.1 Two-dimensional street canyons

Flow and dispersion conditions in the classical street canyon, effectively a straight two-dimensional canyon with order one width to height ratio, have been the subject of much wind tunnel and computational research. The external flow drives both a circulation within the canyon and a mean flow along it, together creating helical flow conditions, as illustrated in Figure 3.1 and discussed in detail in Dobre et al. (2005). The circulation is driven by the component perpendicular to the canyon and the majority of the research effort has actually concentrated on wind directions in which this is the only motion. However, the canyon flow becomes much more interesting and effective in dispersing pollutants when there is a component of the external flow parallel to the canyon. Pollutant from a point source within the canyon is rapidly mixed throughout the cross section by the circulation, then carried along the canyon in the downwind direction. At the same time, there is a continuous exchange of pollution between the canyon and the air flow above. In broad terms, this results in an exponential decay of concentration along the canyon (see Section 4.4.2). To the external flow, the point source within the canyon appears as a non-uniform line source. Dispersion behaviour, at least at relatively short range, is clearly far from that of the classical Gaussian plume. In some circumstances, fluctuations in the direction of the external wind can have a remarkable effect on dispersion in the street canyon. This arises when the wind is near normal to the canyon axis, such that the direction fluctuations change the sign of the component parallel to the canyon. In this manner, a modest variation in the wind direction above roof level changes the direction of flow in the canyon through 180^{o}.

As shown in the full-scale measurements of Louka et al (2000), a shear layer develops from the upwind roof, which controls exchange between the canyon and the external flow. This shear layer fluctuates in position and intensity and with these fluctuations the external wind may penetrate further into the canyon, when the shear layer is weak and displaced downward, or the canyon ventilates when the shear layer is displaced upwards. The overall picture is of an unsteady exchange process and its description through, say, an exchange velocity is simply an ensemble-average point of view that may be adequate for predicting average behaviour but not what prevails in a single realisation of the process (Barlow & Belcher 2002; Harman et al 2004; Cai et al 2008).

On average, the shear layer that separates from the upwind building roof is deflected downwards by pressure forces as it passes over the canyon and when the canyon geometry is appropriate it impinges on the downwind building wall and feeds and forms the canyon vortex. Appropriate here implies that the canyon width is no more than about twice the building height, the precise value depending on the characteristics of the upstream buildings and the external boundary layer. The intensity of the circulation is determined by a balance between the driving shear force from above and the friction forces on the canyon surfaces. The intensity therefore diminishes as the canyon becomes deeper and the frictional forces increase relative to the driving force. At the other extreme, when the canyon width exceeds about three building heights, the shear layer reaches to street level and the recirculation region fills only a part of the street canyon, on the upwind side. The external flow now penetrates to the surface over the remainder of the canyon floor, providing well ventilated conditions that contrast markedly with those in the recirculation region (Harman et al 2004).

Pollutant from a point source becomes reasonably well mixed across the canyon after about one cycle of the across-canyon circulation. In this time, the pollutant typically travels a distance along the canyon of order a canyon length scale (the height or width). This is illustrated in Figure 3.2, which shows wind tunnel measurements of contours of dimensionless mean concentration in two cross-sections of a street at the DAPPLE site. The external wind was at 45˚ to the street axis and the source in the centre of the street. Figure 3.2a at 0.8W from the source, where W is the street width, shows a concentrated plume confined to the lower right quadrant of the cross-section, whereas Figure 3.2b at 3.5W shows well-mixed conditions within the street canyon and a steep concentration gradient near roof level. The fetch along the street to obtain well-mixed conditions decreases as the wind direction moves closer to the normal to the canyon axis and, conversely, increases as the direction moves towards the axis. More conventional plume dispersion processes become the key to determining the fetch in the latter case. It is, perhaps, not fruitful to dwell on these issues any further as matters are likely to be very different in reality because of non-uniformity in canyon geometry and, perhaps most importantly, effects of finite canyon length.

Before moving to consider finite length street canyons between intersections, it is worth briefly summarising the changes in the characteristic cross-canyon flow patterns as the geometry moves towards deep (narrow) or shallow (wide) canyons. The simple, single-cell recirculation pattern cannot persist as the canyon depth increases relative to its width beyond about a factor of about two. Stacked, multi-cell circulations may then form, the number depending on the height to width ratio. Flow conditions in the bottom cell become weak relative to the single-sell circulation, leading to poor dispersion conditions and increased concentration levels. The external wind is able to penetrate to street level in wide street canyons, meaning width to height ratios greater than about two. The transition is likely to be one of increasing and intermittent penetration as this ratio increases. The circulation remains on the upwind side of the street but is now bounded by a greater surface area across which exchanges of pollutant take place, a situation likely to increase the rate at which any pollutant within the circulation is lost to the external flow. There is also the possibility that the source may be located in the downwind side of the street cross-section, beyond the circulation region, in which case dispersion behaviour will change accordingly (since the location of a source in relation to varying flow structure can result in very different dispersion behaviour).

The two-dimensional street canyon is an abstraction that has only limited value in describing the processes taking place in the finite length canyons found in urban areas. An important aspect of these is the intersections between which they run. The circulation in a canyon appears to be maintained almost right up to the downwind intersection but requires a fetch of order two or three street widths to become established at its upwind, or 'entry' end. The entry region has not been studied in any detail and the fetch to fully developed flow conditions (i.e. attaining the state found in a two-dimensional canyon) is unknown and likely to be greater than the values suggested above for establishing some form of circulation. The measurements of Dobre et al (2005) and Carpentieri et al. (2009) indicate that the basic flow pattern is reached after a fairly short distance.

Another key difference between real and idealised street canyons is the variability in the buildings flanking the canyon. The simple planar shear layer of the two-dimensional case is broken-up by changes in roof height and the introduction of vorticity with different orientation that this creates. This is likely to lead to both increased strength and unsteadiness in the exchange process with the external flow. A consequence will be the existence of regions of greatly enhanced efflux of polluted air or ingress of relatively clean air. Again, a simple mixing velocity applicable to the whole street segment might be reasonable as an ensemble averaged model, with the ensemble extended in space over the extent of the street canyon.

3.1.2 Intersections

Intersections act to redistribute the flow and pollutant fluxes amongst the streets that meet there, some acting as inputs and the others as outputs and any imbalance driving exchanges with the external flow. Most attention has been concentrated on the intersection of two, mutually perpendicular, straight streets and that proves to be a good exemplar. In general, the wind direction is not aligned with either street axis and there are two unequal input flows and two unequal output flows, see Figure 3.3. The differences between the input flows reflect the wind direction and the geometry of the two street canyons; the same is true for the output flows. There appears to be a fundamental difference between these situations and those in which the external flow is aligned with the direction of one of the streets. There is then just one input street and the great majority of the input leaves the intersection in the downwind continuation of that input street. Some exchange with the cross-streets occurs but it is weak in overall terms because there is no component of the external wind to drive a mean flow along these streets. The symmetry implicit in this description is not found in practice, although it has been assumed in many numerical studies. Wind tunnel experiments intended to establish symmetrical conditions showed very clearly that any small perturbation in the geometry, or misalignment of the intersection and the wind, broke the symmetry and established flow conditions closer to those found in the general case, see Figure 3.4 and Robins et al (2002). Variation in the heights of the buildings around an intersection has a similar effect (Scaperdas, 2000).

There is normally a classical helical flow field in the input streets that persists up to the intersection. There the flows or jets (using the word rather imprecisely) from the input streets meet and interact, the details of the interaction depending on their relative strengths. A typical flow pattern in a horizontal plane near street level reveals large, vertically-aligned, vortex-like circulations occupying much of the cross section of the output streets (see Figures 3.5 and 4.7). These are created in the flow separating from the upwind edge of the buildings at the intersection and are similar to the vortices that are found in similar locations downwind of large, isolated buildings. They link with the shear layers at roof level deeper within the street canyons. Thus much of the direct advection of flow from one street to the other is directed around these vortices, producing highly three-dimensional fields at the entrance to the output streets. Nevertheless, observations show that a near-classical helical flow is established in the output streets within a length equivalent to about two or three street widths. Whether or not this is a fully developed helical flow (i.e. typical of a very long street canyon) isn't clear.

The interplay between the jets from the input streets is very unsteady and three dimensional, as has now been firmly established in wind tunnel and field experiments and in LES simulations. Flow simulation and tracer studies show material from each input carried to each output, in proportions that reflect the geometry and orientation relative to the wind, though arriving at different levels. For example, video available on the DAPPLE web-site (www.dapple.org.uk/downloads.html) shows an intersection between two unequal streets, one about twice as wide as the other. The bulk of the flow from the smaller input is deflected into the larger output street, but in doing so the weaker jet separates from the surface and a fraction of the flow continues near roof level into the output street that is a continuation of the narrower input street. Some flow from the larger street is also deflected into this output. The separation is unsteady, as are the heights of the outgoing streams, to the extent that conditions at a fixed position within the intersection might some times be governed by one stream and some times by the other. Variability in the driving wind speed and direction above roof level acts to exacerbate the unsteadiness and bimodal distributions of short-term (e.g. in 10 Hz observations during a 1 hour period, Balogun et al., 2010) horizontal wind direction result.

To illustrate further flow conditions at a real intersection, Figure 3.5 presents wind tunnel measurements of mean velocity at the intersection of Marylebone Road and Gloucester Place at the DAPPLE site. The mean wind well above roof level is at 51° to Marylebone Road, the x-street. Figure 3.5a shows mean velocity vectors in horizontal planes at heights equivalent to 5 and 20m at full scale. The two buildings on the east side of the intersection are taller than 20m and their presence at this height perturbs the otherwise relatively uniform wind field. Conditions at the lower level clearly show the channelling effect of the street canyons and the interaction of the flows into the intersection from the south and west. At this level, all the flow from the south, the weaker inflow, is deflected to the east into Marylebone Road, whilst inflow from the west is divided unequally between the two outflow streams. Large recirculation regions are apparent where the outflows leave the intersection, most clearly on the south side of Marylebone Road. The extension of this recirculation can also be inferred at the higher level.

Figures 3.5b and 3.5c add details of the mean flow in vertical planes, which aid interpretation. Firstly, a street canyon recirculation can be seen to exist in the flow entering the intersection form Marylebone Road. However, the picture mid-intersection is quite different and is dominated by the interaction between the two inflow streams. The weaker flow from the south is displaced upwards and as a consequence some continues to the north along Gloucester Place. Figure 3.5c shows the development of the outflow along Marylebone Road. A simple recirculation pattern has been established at the final measurement plane, which is about one street width from the intersection.

Any imbalance between the incoming and outgoing flows in the street canyons appears as an exchange with the mean flow above roof level. This will carry pollutant out of or into the intersection, depending on the difference between the in and outflows, an exchange by the mean flow that is additional to any exchange by turbulent mass fluxes. Carpentieri and Robins (2010) carried out a volume flux balance and showed that, in this particular example, the vertical exchange by the mean flow is very small. The volume flow along Marylebone Road increased by about 40% across the intersection, with that in Gloucester reducing in compensation so that the net vertical flux was negligible. Thus it might be the case that the vertical mean flow exchanges engendered by the intersection are actually expressed in the developing flow in the initial sections of the outflow streets.

Similar conclusions can be drawn from the somewhat larger body of work with balanced intersections, balanced in that the building heights are uniform and the geometry of the intersecting streets does not change across the intersection. Again, for simple intersections of this sort, almost the whole total of the input flows leaves in the output streets, with very little exchange with the overlying boundary layer. However, the ratio of the flow rates in the output streets does not necessarily match that in the input. There is, therefore, a development fetch in these streets in which the 'equilibrium' canyon flow rates are re-established. This has already been discussed in the context of the re-establishment of the helical circulation but, inevitably, it also involves a mean flux from one of the streets into the external flow and a compensating mean flux into the other street from the external flow. Thus, the transfer of pollutants from the street network into the external flow is likely to become very patchy at and near intersections. The implication is that, in dividing a street network into street segments and intersections, the intersections should extend about a street width into the output streets.

The picture is likely to be very different when the street geometry changes significantly across an intersection, as the 'capacities' of the input and output streets no longer balance. Substantial exchanges with the external flow then result, being positive (i.e. upwards) when the output is of lower capacity than the input and negative in the opposite circumstance. An extreme example is found in the case of the 'T-junction', Belcher (2005). Three characteristic situations can be identified, one with the wind aligned with the 'blocked' street, a second with it aligned with the continuing street (which is similar to the aligned case previously discussed) and a third with it aligned with neither.

In the first of these examples, the external wind is perpendicular to the cross-street and, were it not for the junction, a simple street canyon vortex would develop (see Figure 4.7a). The jet from the incoming flow at the intersection is split and deflected into both branches of the cross-street, with a large fraction also escaping into the external flow. A vertically aligned vortex is set-up in the entrance section of each lateral segment, formed in the shear layer separating from the upwind building corners at the intersection. The flow entering the street, having passed around the corner vortex, interacts with the canyon circulation, forming a helical flow that passes deeper into the street segment. Gradually, the flow along the street decays so that far from the intersection the simple canyon vortex motion dominates. In this way, the balance of the flux from the incoming street that was not immediately lost to the external flow at the intersection is vented over a fetch of the cross-street on either side of the intersection. How long this fetch might be is not known but estimated to be of order four street length scales. In this manner, an urban T-junction creates a region of abnormally large transfer rates into the external flow, in which much of the accumulated pollution from the input street is vented. Of course, the flows into the two branches of the cross-street will not balance in practice but the overall picture remains valid.

The third example has two forms, either two input streets feeding one output or one input feeding two outputs. The first of these is similar to the situation just described in that the input from the side street provides fluxes in excess of those in the developed main street. The excess is vented to the external flow at and downwind (in the sense of the flow in the street canyon) from the intersection. Again, the T-junction becomes a region of large positive transfer rates. For similar reasons, the intersection becomes a region of negative transfer (i.e. inflow) when one input feeds two outputs.

The description so far has dealt with simple intersections between street canyons, not on the more open form found, for example, at roundabouts. Here, the lateral extent of the intersection may become large compared with the height of the surrounding buildings and the external flow can then penetrate to the surface. However, there is still flow through the output canyons that is fed from the intersection. What is likely to change is the transfer to the external flow and the flow speed entering the output canyons, which become greater than in the case of the simple intersection. The initial volume flow along the output streets then exceeds the fully developed values and there is a region downwind from the intersection in which this is lost to the external flow. This topic is briefly returned to when open spaces are discussed in Section 3.1.6.

3.1.3 Street networks

The combination of intersections and the streets between them creates a street canyon network. As is clear from the preceding discussion, dispersion in such a network will, in general, be very unlike that of the traditional Gaussian plume, as Figure 3.6 and DAPPLE (2011) clearly demonstrate. The key features, at least at short range, are the channelling of pollutants along streets, the exchanges between streets at intersections and losses to the overlying boundary layer that appear there as a dispersed and non-uniform source distribution. The combined effect of these features has its greatest impact when the wind is, broadly speaking, misaligned with the street network; as a corollary, the impact is least when the wind is aligned with a set of streets. This comment really applies to rectangular street networks and in many situations the complexity of the street network might imply that all conditions are effectively those of misalignment, regardless of the wind direction. Further aspects of dispersion in and above street networks are discussed as a case study in Sections 3.2.1 and 3.2.2)

The component of the external wind that is parallel to a street axis drives a mean flow along the street and this can lead to very rapid lateral dispersion. The case study discussed in Section 3.2.1 illustrates just such behaviour. Upstream spread is also observed, perhaps over one city block, and results from the basic unsteadiness of the flow field in the street network, coupled to short term variations in the external wind direction. Greater degrees of upwind spread might arise through the action of traffic movement, a subject that will be highlighted in the case study and is also briefly discussed in Section 3.1.7.

The air flow exchanges at intersections remove the direct link between local concentrations and local emissions. As demonstrated by Figure 3.4 (Robins et al., 2002), there are locations where the concentrations experienced are actually largely the result of emissions in one of the other streets; i.e. reflecting the transfer of pollutants from an input street to an orthogonal output street at the intersection of two street canyons. Downstream from the intersection, in the local flow directions, conditions gradually revert to those of a well-mixed canyon. There is however, no notable upstream effect in the input streets. This is another argument for defining the region described as an intersection asymmetrically so as to include some of the initial fetch in the output streets, perhaps a length of order two canyon length scales.

Unsteadiness in flow and dispersion behaviour has already been noted as a local phenomenon. This becomes spatially organised when driven by large-scale features of the turbulence in the overlying boundary layer.

3.1.4 Regular arrays

There is a most extensive literature concerning flow and dispersion in regular arrays of cuboids, frequently cubes, in aligned or staggered configurations (e.g. see Macdonald et al., 1997, Macdonald et al., 1998, Macdonald, 2000 and other work referenced in Section3.3). The reason for this is easy enough to understand but the uniformity and obstacle shapes imply that these might not be good approximations to central urban conditions. Nevertheless, some important findings emerge from these studies that have advanced our understanding of the more general problem. One key feature identified was the role of plume splitting around obstacles in driving lateral spread, a process most apparent in staggered arrays or when the wind direction was at incidence to the street alignment, see Figure 3.6 (DAPPLE, 2011). This has an obvious relation to the intersection flows discussed above; indeed, it might be argued that it is simply another way of viewing that process. Another process that was seen to be central in enhancing lateral spread was the entrainment of material into the recirculation region between obstacles, as this formed a path for pollutant to cross from one ‘street’ to a parallel one, in circumstances where plume splitting was weak (e.g. with the wind aligned with the street axis). The process can then repeat itself as the plume continues to spread laterally, but ceases to be effective if the spacing between obstacle rows becomes too great.

Systematic studies of dispersion in regular arrays have been central to obtaining empirical expressions for the flow conditions within and above the canopy (see Section 4.4). This work is discussed elsewhere in this report and will not be further analysed here. Only a few studies have considered wind directions at incidence to arrays. One interesting feature that emerges is that the average direction of the flow at any level within the canopy generally differs from that of the wind above and this is associated with a side force on the array (Claus et al., 2012). This is really the regular array equivalent of the processes described above for street canyons, where large lateral plume displacement was identified as a key feature of dispersion in urban areas.

3.1.5 Tall buildings

Here, it is worthwhile first recalling what is known about flow and dispersion near isolated buildings and then set it in the context of an urban area. Cuboid shapes, particularly the cube, have been extensively studied in wind tunnel and computational simulations. There are two cases to consider for cuboids, the first when the wind direction is perpendicular (or nearly so) to the front face (Castro & Robins 1977). Pressure is highest towards the top of the front face, where the approach flow stagnates; this is typically at a height of about two-thirds of the building height but dependent on the geometry. Above this level the flow moves upwards, generally to separate at the leading edge of the roof, whereas the flow is directed downwards below this level. The descending flow feeds an upwind recirculation that is swept around the building to form the characteristic horseshoe vortex around and downwind from the building. The flow separates from the edges of the front face to create recirculating regions over the roof and side walls. If the roof is pitched and sufficiently steep the separation may be delayed to the ridge line. For present purposes, it is sufficient to assume the roof to be flat. The separated flow from the leading edge of the roof may reattach to the roof if the geometry permits (i.e.

the roof is long enough, given the turbulent conditions) and then separate again at the trailing edge. In either case, a downwind recirculation region is formed immediately behind the obstacle and extends to a distance of the order of the building height for a cuboid. The length of this recirculation region, often referred to as the near-wake, increases as the lateral extent of the building increases.

The wake downwind of an isolated building is a region of reduced mean flow speed and excess turbulence relative to conditions in the flow well ahead of the building. Wake spread and decay requires flow into the wake that leads to the downward and inward deflection of mean streamlines. Downwash is much more pronounced behind a cuboid at incidence to the free stream due to the secondary flows associated with the roof vortex system. The simplest case to describe is a square, flat roof aligned diagonally to the approaching wind. There are two forward facing walls and on each the oncoming flow is deflected upwards towards roof level, separating at the sharp edge. The shear layers then rapidly bend over in the cross-flow above the roof and are stretched in the mean flow direction. The vortices are particularly 'tight' near the leading corner of the roof and very low pressures are observed on the roof beneath them. The vortex pair, of equal strengths in this special case, trail downstream from the building, generating downwash in the central part of the wake, to the extent that a velocity excess may develop far downwind, rather than the deficit usually expected of a wake flow. In general, one vortex is stronger than the other because the wind is not usually aligned with the roof diagonal. This is also the likely situation in the wake of a slab-like building at incidence to the wind where one vortex dominates, not unlike the trailing vortex from an aircraft wingtip. The whole near-wake may then display a bulk swirling motion so that in some areas pollutant from street level is carried to the roof, whereas elsewhere the reverse occurs.

Material released or entrained into the recirculation region in the near-wake of a cuboid is relatively well mixed throughout that region, to the extent that the near-wake is often modelled as a region of uniform concentration (e.g. Robins and Apsley, 2000). This simple picture must fail as the building height increases relative to the base dimensions. However, the separated flow region remains effective in dispersing material over the height of the building even for quite tall buildings. As an example, Figure 3.7 shows wind tunnel results for a 285mm tall model building with a rectangular base measuring 96x65mm; the height being $H = 3.6\sqrt{WL}$, where W and L are the base dimensions. Vertical profiles of mean concentration were measured at 30 mm from the rear of the building for ground and roof level passive emissions, the latter adjacent to the rear face and the former at the roof centre. Concentration gradients over the height of the building were large but the surface emission was detected at roof level and the roof level emission at the surface for all three wind directions investigated, vertical dispersion being most effective for the diagonal orientation, 34˚. How tall a building needs to be for dispersion in the near-wake over its full height no longer to arise isn't known. It is worth noting that the interaction between the shear in the wind profile and the cross-sectional shape of the building generally results in a mean vertical flow in the near-wake.

The discussion now moves to a large building on one side or the other of a conventional street canyon. The features described above interact strongly with the canyon flow and may completely change its character. Consider a tall, slab-like building on the downwind side of a street canyon with the wind direction close to perpendicular to the canyon. The downward flow on the front face intensifies the canyon vortex ahead of the building and feeds flow into the street canyon system that drives a diverging flow. The low pressures on the rear face of the building have the opposite effect, generating a converging flow through the street network toward the building. Thus ‘clean’ boundary layer air enters the street network upstream and polluted air is ventilated from it downwind. These phenomena have been clearly revealed by wind tunnel and CFD simulations (e.g. Brixley et al., 2009; Heist et al., 2009). Mean streamline deflections around the side of the building also bring air from aloft into the street system, creating the characteristic ‘windy regions’ alongside and just downwind of isolated tall buildings.

That part of a tall building that lies below the displacement height of the flow over the surrounding buildings is sheltered from the wind, something that is explicitly acknowledged in wind loading codes. Whether or not a horseshoe vortex is formed around the base of the exposed fraction is debatable as the downward flow on the front face is largely into the street canyon vortex rather than the classical upwind recirculation. There are, though, circumstances in which a more or less conventional horseshoe vortex can form and trail at the average rooftop level behind the building, creating secondary flows that enhance the overall exchanges between the downwind street canyons and the external boundary layer. The final exchange mechanism of note is that driven by the bulk swirling motion of the whole wake when a slab-like tall building is at incidence to the oncoming wind.

Thus, through a variety of processes, an exceptionally tall building can generate mixing over its full height and severely perturb the flow field in the surrounding street canyons. What occurs in the vicinity of a group of closely spaced high-rise buildings, as found in many central business districts, is a different matter and is discussed in Section 4.6.

3.1.6 Open spaces

Open spaces play many roles in dispersion in urban areas but from the current, rather narrow point of view their virtue lies in allowing the external wind to spread towards the surface, sweeping away polluted air, where that is feasible, and feeding ‘clean’ air into the street canyon network. This is really achieved by enhancing, in an overall sense, the vertical spread of pollutant into the external flow. The recirculation region behind, say, a row of buildings orientated across the wind typically extends to between H and 5H downstream (depending on the ‘porosity’ of the urban form) and an ‘open’ space may therefore be very loosely defined as something that is at least as large in downwind extent as this. The flow into the street canyons downwind of an open space is driven by the wind and generally exceeds the equilibrium flow for the geometry in question. There is therefore an adjustment zone in which the equilibrium conditions are established and the excess mass flow ejected from the street network (see Belcher et al. 2003, Coceal & Belcher 2004 and Section 2.1). This process is clearly seen in flow visualisation studies with a ground level source upwind of an obstacle array. The development length scale is quite short, being of the order of a few building heights. We should distinguish between finite and continuous open spaces, the discussion so far having focussed on the former because the context is the neighbourhood scale. The latter (typically rivers, the sea shore, etc.) are far more effective in removing polluted air from the street network, at least for some wind directions. Of course, perhaps the most significant feature of many open spaces from the air quality viewpoint is the absence of emissions.

3.1.7 Traffic

A brief comment on the role of traffic movement in the dispersion of pollutants in urban areas is worthwhile, even though the emphasis of the review as a whole is with flow and dispersion that are driven and controlled by the external wind. The role of traffic-generated turbulence has long been accepted and is generally assumed to mix emissions from the tailpipe over a depth comparable with the vehicle height. This occurs in the near-wake of the vehicle and is usually considered as defining an effective source condition for dispersion further away (e.g. Di Sabatino et al., 2003; Kastner-Kline et al., 2003). The vehicle wake is a region of excess turbulence levels and becomes the dominant source of turbulence when the external wind is sufficiently light. Vehicle movement also contributes to the dispersion of pollutants along the street, but in the direction of the traffic flow that may well be contrary to the wind driven flow. One mechanism, which appears to be very effective in this respect, is the detrainment of material from the near-wake of large vehicles. There is a characteristic residence time for pollutants in the near-wake that, in still air, is proportional to the vehicle size and inversely proportional to the speed. Vehicle motion converts this into a characteristic length that is proportional to the vehicle size but independent of the vehicle speed. The combined effect of many such processes appears to be capable of transporting emission over considerable distances. The case study in the next section highlights just such events.

3.2 Measurements in complex geometries

3.2.1 Case Study – DAPPLE field experiments

We begin with a discussion of results from the DAPPLE experiments in central London, and use these to illustrate some of the processes that occur in real cities. Wood et al (2009) provide an overview of the measurements and some preliminary results. Here, we present further analysis of the data based on the final reports, DAPPLE (2011). A map of the site is included as Figure 3.8 for reference as it is often more straightforward to use street names to discuss the results.

Days 2 and 3 (7 and 8th June, 2007) of the 2007 DAPPLE field experiments are used as case studies. These are selected as they illustrate many of the processes discussed above and also show clear upwind dispersion. The most significant fact though is that the wind direction was the same, within a few degrees, allowing inter-comparison of results amongst the ensemble of 8 experiments.

Four experiments were carried out on each day; in experiments A, B and C different tracers were released from each of three source points (X, Y and Z) and in experiment D from two source points (Y and Z). The tracer was an inert gas that was emitted at constant rate for 15 minutes and sampled at 18 fixed locations in the study area, the region around the junction between Marylebone Road and Gloucester Place in central London. The source and sampling sites were chosen from a pre-selected list according to the wind direction on the day of the experiment. All were near kerbside in unobstructed locations and each released a separate, distinct tracer species at a height of 0.5 m above street level. An ultrasonic anemometer was located close to each source position, measuring at a height of 1.5 m above street level at X, and at 1 m at Y and Z. All air samples were collected from a height of 1.5 m. Wind conditions were observed at the top of the BT Tower (at a height of 190 m above street level) and on a local roof top (on the WCC Library) in the study area.

The average wind speeds measured at the BT Tower and WCC Library roof were 6.2 and 1.9 ms^{-1} respectively and their ratio, 0.31, is close to the climatic average of 0.28 (Barlow et al 2009). The mean wind directions were -132˚ and -122˚, a veering with height of 10˚, very close to the climatic average of 9˚. The average direction measured at the WCC Library was north-easterly and therefore oblique to the street network. During such reference wind conditions the resulting flows within the street network were likely to have been a combination of channelling and recirculating flow (i.e. a helical canyon vortex).

Figure 3.9 shows the general arrangement of the buildings around the positions of Source Y and Z on Days 2 and 3. One-way traffic flows in Gloucester Place, Melcombe Street, Chagford Street and Dorset Square are indicated by the open blue arrows. Dorset Square is a park area with many mature, tall deciduous trees. The small red arrows indicate the expected direction of the along-street wind in selected streets, assuming that the wind direction above roof level is steady and the building heights uniform, and the green arrows the mean directions inferred from tracer movement, where that was different. A dotted green arrow implies a direction in which tracer dispersed but not necessarily the pathway. The wind direction on both days was similar, being diagonal to the street network, which implies that, all else being equal, a well defined helical vortex flow was set up in all street canyons. This explains the direction shown for the near surface winds in Melcombe Street and Gloucester Place. In the area around Dorset Square the street level wind was likely to have become more closely aligned with the wind above roof level.

Mean wind speeds recorded at all source positions were low, generally below 1 ms^{-1}. Wind directions observed at Y and Z were very variable, though on average consistent with a helical circulation. Source position Z measured an average wind direction of south-east which was a combination of a channelling flow (from the east) and a recirculation flow (from the south). The flow at source position Y was a combination of a southerly component (channelling) and an easterly component (due to recirculation). No clear pattern was discernable in the horizontal component at X; however, the vertical component was negative as would be expected in a canyon vortex flow.

Figure 3.10 summarises the results for experiment C (cf. Figure 7 in Wood et al. 2009). It shows the dimensionless dosage $D^* = DUH^2/M$ at each receptor site, where D is the measured dosage, U the reference WCC wind speed, H the average building height (22 m) and M the total tracer released. The justification for this form of dimensionless variable is discussed below. The results are colour coded (red refers to Source X, green to Source Y and blue to Source Z) and given both as numerical values and also as bar charts. The receptor site numbers are also shown.

As expected from the flow patterns, concentrations were high in Gloucester Place in all cases. Tracer was also observed upwind, though in the direction of the one-way traffic flow, at Site 15; only occasionally did the wind at Y have a northerly component and it is difficult to explain the dispersion as wind driven. No tracer from either source was found at Sites 13 and 14. Tracer from X dispersed significantly upwind and was detected at Sites 8 and 9 (180 m upwind; three minutes travel time at 1 ms^{-1}), to the east in Marylebone Road, sometimes at relatively high concentrations, but not further east at Site 10 (300 m upwind). In some experiments it was also detected upwind but to the south of Marylebone Road at Site 4 in Gloucester Place, Site 5 in Bicknell Street and north of Marylebone Road at Site 11 in Gloucester Place. Transport to Sites 1 and 2, almost due south of X, can probably be explained in terms of more or less conventional dispersion in a street network with unsteady wind conditions but that is not so for the other cases. It is highly probable that traffic movement played a significant role in the dispersion of the tracer.

Very large lateral spread was observed in streets containing a source. Tracer from Source Y was observed to the south of the source at four sites in Gloucester Place, over a distance in excess of 500m. Of course, the external wind generated a wind component at street level to the south along Gloucester Place. An exchange process like that illustrated in Figure 3.11. probably occurred at every intersection, successively diluting the tracer material as it was carried southwards. There are five intersections of note in Gloucester Place between the source at Y and Site 1 and at each the flux of tracer moving south along Gloucester Place was reduced. Material was also lost from the street canyon sections of the road to the external flow and there was a continuous decay in concentration as a result.

The meteorological conditions on Day 3 were very similar to those on Day 2, the only significant difference being a lower wind speed. The ensemble-averaged dimensionless dosage, $\langle D^* \rangle$ would be the same on the two days if dispersion behaviour remained the same. Ideally, $\langle D^* \rangle$ is the ensemble average from a large number of experiments in near-identical conditions - a large number is needed because the inherent variability in dosage levels is large. Analysis is, however, restricted to ensembles of four and this will inevitably leave considerable uncertainty attached to ensemble-averaged values that may mask relationships between results from the two days. Figure 3.12 is a scatter plot of the ensemble-average dosages from Days 2 and 3. Results have been partitioned by the position of receptors relative to the appropriate source; ‘downwind’ is defined as the ±45° sector downwind of a source, ‘upwind’ the ±90° sector upwind of a source and ‘crosswind’ the remaining 45° sectors on either side of the mean wind direction. Uncertainty bars equal to the standard deviation of the ensemble values relative to the mean are shown for data in the downwind sector. The 1:1 correspondence line is also shown, together with the factor of two boundaries (dotted lines). Overall, there appears to be a tendency for results from Day 2 to be larger than those from Day 3, with some data falling more than a standard deviation below a 1:1 correspondence. However, the standard deviations of dosages relative to their respective ensemble-averages was uniformly about 45% in the downwind sector, so it would be unwise to draw firm conclusions from this analysis. The observation is tantalising though, as the expectation might be for dosages to be lower on Day 2, when the wind speed was low and plume spread might have been enhanced by thermal and traffic effects. The high degree of variability in repeated measurements in nominally similar conditions has significance for dispersion modelling as it defines the lowest level of uncertainty that can be expected of predictions, whether from computation or wind tunnel simulation.

The effects of traffic induced flow and turbulence are expected to become more significant as wind speeds decrease but there is only mixed evidence of greater upwind dispersion on Day 3 than on Day 2 from the dosages recorded at ‘upwind’ sampling sites. Tracer dosages from Source X at Sites 2, 4, 8, and 9 were actually lower on Day 3 than Day 2, their ratio averaging about 0.5. The picture was mixed for Sources Y and Z, with the ratio falling to about 0.3 at Site 8, but averaging about 2 overall. The most dramatic increases occurred at Site 15, north of Y and Z in Gloucester Place, where the ratio averaged about 5. This is perhaps the most straightforward test as the traffic flow is one-way and the geometry straightforward.

The sets of four experiments on Days 1 to 3 can also be used to examine variability in dispersion behaviour as the wind conditions were relatively steady on each day. A simple measure is the ratio of the maximum and minimum dosage in each set of four results at any receptor site (e.g. Cases A to D on Day 2 at Site 8 for Source X). Ratios for the whole data set are plotted in Figure 3.13 as a function of dimensionless downwind distance, $R^* = R/H$; R is the straight-line source-receptor separation and H the mean building height.

The results have been partitioned by the position of the receptor relative to the source; as before, 'downwind' is defined as the ±45˚ sector downwind of the source, 'upwind' the ±90˚ sector upwind of the source and 'crosswind' the remaining 45˚ sectors on either side of the mean wind direction. Data from the downwind sector are labelled 'channelled' when located in streets in which channelled flow occurred. The factor of two line is shown for reference and all data in channelled conditions are bounded by this. Otherwise variability is much greater and exceeds a factor of ten on a number of occasions. A near-source region has been identified, stretching to about *R*/*H* = 5, in which variability is particularly high in all directions, though there is unlikely to be any common reason for the behaviour indicated. Beyond about *R*/*H* = 10, high variability is confined to the downwind sector and decreases with increasing fetch downwind. Variability in this range is likely to result from forms of plume meander but also near-source effects being carried downwind.

3.2.2 Case Study – DAPPLE wind tunnel experiments

Extensive wind tunnel flow and dispersion experiments were a feature of the DAPPLE project and continuing work thereafter. The wind tunnel work used a 1:200 scale model of the area around the DAPPLE field site, covering an area of diameter approximately 750 m. The model buildings were, for the most part, constructed with flat roofs, a simplification introduced to ensure that the blocks could be simulated accurately in CFD simulations. The aim of the wind tunnel work was not to provide a faithful simulation of the field conditions but rather to establish an adequate simulation that could be used to understand processes at work in the field experiments and to investigate the sensitivity of flow and dispersion behaviour in the face of perturbations to a base condition of the block-shaped buildings in a standard neutrally stable atmospheric boundary layer simulation. This standard simulated boundary layer was 1 m deep, with friction velocity $u*/U_{ref} = 0.05$ and roughness length $z_o = 1.0mm$; U_{ref} is the reference speed at the edge of the boundary layer.

The only similarity criterion of note is that the building Reynolds number, $U(H)H/\nu$, should exceed a minimum value of about 11,000 (Snyder, 1981) to ensure Reynolds number independent flow around the building blocks. The 1:200 scale wind tunnel experiments were run with $U_{ref} = 2.5ms^{-1}$, $H = 0.11m$, $U(H) = 1.8ms^{-1}$ and a Reynolds number of 13,000, ensuring that the criterion was met. No other consideration arise as all emissions were passive (i.e. possessing no significant buoyancy or momentum).

Here, we use the results from DAPPLE (2011) and Smethurst (2012) to discuss the nature of dispersion in and above a street network, the behaviour of short duration emissions and, finally, some aspects of concentration fluctuations.

The mean concentration field

Concentrations were measured at a number of heights in the streets downstream from a number of sources for a range of wind directions. From such data, the outline of the dispersing plume could be somewhat loosely defined and dispersion in and above the site model contrasted to that in a similar boundary layer over a rough surface. Figure 3.14a shows such an analysis for an elevated emission (i.e. well above roof level) at 153mm height, equivalent to 30.6m at full scale. The source is in York Street and marked by a red circle, and the wind well above roof level is at 45˚ to the street network, as it is for all the results discussed in this section. The plume outline is drawn at heights of 153mm (source height), 102mm (near roof level) and 10mm (street level). At source height, the plume travels and spreads almost exactly as it does in the equivalent rough wall boundary layer. It spreads to the ground somewhere near the intersection between Gloucester Place and Marylebone Road, at sample point 2 in the diagram. Once in the street network, it rapidly spreads through a 90˚ sector, whether measured at near street or roof level. As will be seen later, there are therefore streets that contain emitted material but have little or no plume material overhead, transport having been through the street network once the plume had 'touched down'. In turn, though, these streets become secondary sources of material above roof level through the usual process of mixing between the street canyon and boundary layer flows.

Figure 3.14b shows equivalent results for a street level emission. As before, the plume initially disperses within a 90˚ sector, behaviour forced by the geometry of the blocks in the region close to the source. Thereafter, the plume is confined to a slightly narrower sector within the street network. The appearance of material overhead is delayed until dispersion has proceeded downwind over a fetch of about 6*H*, where *H* is the mean building height, 110mm. The external plume that then develops from the material lost from the street network in only a little narrower than the plume within the network.

Vertical profiles of concentration at the numbered locations in Figure 3.14 will now be used to illustrate further features of dispersion behaviour. These locations were chosen to be either well within or outside of the elevated plume shown in Figure 3.14a. Figure 3.15a shows conditions at point 1, under the elevated plume in Gloucester Place, for the three emission heights of 10, 102 and 153mm. No material from the most elevated emission is detected at street level and only a little of that released near mean roof level. In contrast, the plume from the street level emission is almost entirely confined to the street canyon. Conditions have developed at point 2 (Figure 3.15b), again under the elevated plume but further downwind in Marylebone Road, so that material from the elevated emissions is now seen at street level and material from the street level emission has spread well above roof level. This general picture of plume development is continued at point 5 (not shown), under the plume but yet further downwind in Baker Street.

Figures 3.15c and d show quite different features, as the locations are located outside of the outline of the elevated plume in Figure 3.14a. Firstly, at point 3 (Figure 3.15c) there is very little overhead material from the 153mm emission (e.g. compare with Figure 3.15b), though concentrations within the street canyon are far from negligible when compared with the other two cases shown. The plume from the roof level emission has become relatively well mixed within the street canyon but much more structure is seen in the plume from the street level source. Why this is cannot be said with any certainty but it is worth noting that point 3 is located at the centre of a major intersection and concentration distributions can become complex as a result of the mixing processes occurring there (see Section 3.1.2). Finally, point 4 reveals an even more extreme picture, with scarcely any overhead material for all three plumes; material has reached this location through the street network. The reason behind this behaviour is to be found in the flow fields set up in the street canyons. The mean flow is to the east in streets like Marylebone Road that run from west to east, and to the north in streets like Gloucester Place that run from south to north. Thus dispersion to the west in Marylebone Road is against a weak mean flow along the street. The consequence is made clear in Figure 3.16a, which shows mean concentration profiles along the street for plumes from the three source heights, compared with profiles from the undisturbed rough wall boundary layer. The plumes are extended to the east by the mean flow along the street and are much more extensive as a result. Figure 3.16b shows a further consequence by adding concentration fluctuations to the mean concentrations for the street level emission. The western edge of the plume is characterised by a very large intensity of concentration fluctuations, c'/C, where c' is the standard deviation of the fluctuations. This is not unlike conditions at the edge of a plume dispersing in an undisturbed boundary layer (e.g. see Fackrell and Robins, 1982). The eastern edge is quite different though, with the intensity remaining at a modest level as the mean concentration decays. This general mode of behaviour was, in fact, seen in all the streets that were studied.

Figure 3.16a shows concentrations in the street network that are less than in the undisturbed flow for the low level emission but larger for the two elevated emissions, most notably for the 153mm source. Closer to the source, along Gloucester Place, the elevated emissions had little significant impact at street level, as Figure 3.14a would suggest, but in this case dispersion through the street network led to much greater concentrations from the low level source than in the undisturbed flow. The maximum concentration occurred at the intersection with York Street (where the source was located) at y ~ -700mm (see Figure 3.14), whereas the elevated plume crossed Gloucester Place at about location 1 in the figure.

Finite duration emissions

All discussion of dispersion has so far implicitly or explicitly assumed a steady, continuous emission. Other issues arise with finite duration emissions and these will be briefly discussed here, though it will again be assumed that the emission rate is steady. The basics structure and development of the concentration field in an undisturbed, rough wall boundary layer has been described by Robins and Fackrell (1998), using the theoretical framework established by Chatwin (1968). A key result delineates conditions under which the dispersing cloud can be described as a puff (i.e. as if from an instantaneous emission) and this arises when the travel time is much larger than the emission duration. Conversely, when the emission time is much greater than the travel time, a central region appears in the concentration time series that has characteristics that are independent of the emission duration and therefore the same (at least, in an ensemble average sense) as conditions in a plume from a continuous source. This general description of the character of the dispersing cloud of material has been found to hold equally well at the DAPPLE site.

Figure 3.17a shows wind tunnel measurements of ensemble averaged concentration time-series at a fixed location in Marylebone Road at a distance of 7H from a source in York Street for a wind direction of 45˚. Emission durations between 0.25 and 4s were used and ensembles of approximately 190 realisations compiled. Inspection of the figure shows that the travel time was of order 3s and, reflecting this, only the 4s emission has a ‘plateau’ region in the time series. The shortest duration emissions, say emission times below 1s, are clearly puff-like in form. Concentration has been made dimensionless in the form applicable to a continuous plume, that is using the emission rate. This implies that the plateau value becomes independent of emission time once the plume limit has been attained. On the other hand, the maximum concentration in a short duration puff scales on the total emission, so dimensionless concentrations as presented here decrease in proportion to the emission duration. Of course, an individual realisation can depart considerably from the ensemble mean as is demonstrated in Figure 3.17b, which contrasts the concentration time trace for a single 4s emission with the average. Note that, in this particular example, the maximum concentration is about 5 times that seen in the ensemble average and that no plateau region can be identified.

Results for pollutant travel times and concentration rise and fall times are presented in Figure 3.18a; the definitions of these scales are explained in Figure 3.18b. Time has been made dimensionless by the reference speed at the edge of the boundary layer and the average building height. The results for the dimensionless travel time, T^*, show that the advection speed was approximately $U_{ref}/8$ over the fetch shown, R/H < 11. Further downwind, the advection speed will increase as the plume deepens and is progressively exposed to the wind profile above roof level. The rise and fall times were typically 1/3 of the travel time, considerably longer than the equivalent results in a rough wall boundary layer reported by Robins and Fackrell (1998). Evidently, material is 'held-up' in the street network and this is reflected both by a low advection speed and relatively extended rise and fall times.

Concentration fluctuations in plumes

Figure 3.17 shows a concentration record from the passage of a single emission and this exhibits many common features of concentration fluctuations in dispersing plumes, namely a very wide range with periods of near-zero concentration and 'peak' values that can be many times larger than the mean. The picture is perhaps more readily appreciated from Figure 3.19, which shows a sample from a continuous, steady emission.

To set the scene for the discussion that follows here and in Section 3.3.2, some features of concentration fluctuations in plumes in undisturbed boundary layers are first summarised.

Ground level source

i) concentration fluctuation levels, measured by the intensity, c'/C, where c' is the standard deviation of the fluctuations, are relatively insensitive to source size.

ii) maximum fluctuation levels occur at a height $z \sim \sigma_Z$, being about 60% of the local ground level, centreline mean concentration.

iii) plume intermittency (I, the probability of plume material being present) decreases from one in the plume centre to zero at the edges.

Elevated source, elevated plume

i) concentration fluctuation levels are very sensitive to the source size relative to the scales of ambient turbulence.

ii) maximum fluctuation levels can become large relative to the local maximum mean concentration when the source size is small.

iii) maximum fluctuation levels occur at a height $z \sim h$, where h is the source height.

iv) plumes are intermittent throughout, becoming more intermittent in the centre as the source size decreases.

Fluctuation intensity profiles

i) laterally, the fluctuation intensity, c'/C, increases from a minimum on the centreline, becoming large at the plume edges, where I tends towards zero (i.e. C goes to zero more rapidly than c').

ii) the vertical profile is more complex because of the effect of the surface but c'/C again becomes large and I tends towards zero at the upper edge.

Figure 3.19 shows samples of the concentration time series from three locations, selected to give some indication of the range of behaviour observed. Figure 3.19a refers to location A in Figure 3.14, at relatively short range but in a small side street where high concentrations arise from the occasional 'puffs' of material that enter the street – one is shown in the figure. Normally this would lead to a high level of intermittency (i.e. frequent periods with no concentration) in the concentration field but a long ‘residence’ time for material in this street maintains low concentration levels for most of the time between the puffs. The probability that emitted material is observed here is $I = 0.79$ and the intensity of concentration fluctuations, $c'/C = 2.1$. Note that the peak shown in the figure is well in excess of three standard deviations above the mean concentration. Fluctuation levels are much less at location B, Figure 3.19b, a street running north from Marylebone Road, where $I = 0.84$, $c'/C = 0.85$. Here, the peaks shown are only slightly in excess of three standard deviations above the mean concentration. Finally, at location C (Figure 3.19c) in Marylebone Road, the fluctuation level is even lower (see also Figure 3.16) and plume material continuously present; $I = 0.99$, $c'/C = 0.24$.

A number of general conclusions can be drawn, contrasting dispersion in the street network with that in an undisturbed boundary layer. Firstly, conditions in many streets can be described as well mixed and in such cases fluctuation levels are, at best, modest, with c'/C being no more than one and often considerably less. Large fluctuation levels seem to arise in two circumstances, either where material is only intermittently carried to the receptor (as in Figure 3.19a) or at the local ‘upwind’ edge of a plume in a street canyon (as in Figure 3.16b). Analysis of the time traces also provides integral time scales, a measure of the time over which concentration levels remain correlated. This again shows that material is held-up in the urban area, particularly at short range, compared with the undisturbed flow.

Time at model and full scale is related through the dimensionless variable, *T**

$$T^* = \frac{U(H)t}{H} \qquad (3.1)$$

where *H* is the mean building height (0.11 m at model scale, 22m at full scale). Conversion between the two scales implies that:

$$t_{fs} = t_m \frac{H_{fs}}{H_m} \frac{U_m}{U_{fs}} \qquad (3.2)$$

where the subscripts ‘m’ and ‘fs’ denote model and full scale, respectively. For example, with the scale ratio of 1:200 and, say, a speed ratio of 1:4 (i.e. model and full scale reference speeds of $2.5 ms^{-1}$ and $10 ms^{-1}$) the time scale ratio becomes 1:50. Thus the 200Hz response of the wind tunnel instrumentation is equivalent to about 4Hz at full scale. This is rather fast when compared with the adult breathing rate of about 1/3Hz and knowledge of fluctuation levels on this time scale might be more appropriate for applications such as the consequences of exposure to toxic material. In the three cases discussed above, c'/C reduces from 2.1 to 2.0 for A, from 0.84 to 0.75 for B and from 0.24 to 0.20 for C. Similar changes were found in dispersion in the undisturbed flow, except at very short range (R/H ≤ 3-5) where the changes were greater. The decrease in c' with sample averaging time proved to be quite well predicted by Taylor (1921) theory and the differences in behaviour at short range were shown to result from much longer integral time scales in the urban case.

A useful conclusion to this section is to summarise results from a few related studies of significance. Concentration fluctuations observed in the Mock Urban Setting Trials (MUST, see Section 3.2.5) at the Dugway Proving Ground in Utah were analysed by

Yee and Biltfoft (2004). Mean dispersion through this regular array rapidly established a Gaussian plume-like structure, as has been observed in many wind tunnel experiments with regular arrays of cuboids. The structure of the concentration fluctuation field was similar to that observed in undisturbed flow conditions, with a minimum fluctuation intensity, c'/C, on the centre-line increasing steadily to very large values at the plume edges. However, the intensities in the central regions of the plumes were much less than in undisturbed flow, being of order 20-40%. Increased plume spread and the reduction in turbulence scales within the canopy resulted in reduced plume meander and more intense mixing, both of which contributed to the reduction in c'/C.

Properties of concentration fluctuations were also included in the data analysis associated with the Joint Urban 2003 trials in Oklahoma City (Klein and Young, 2011, see Section 3.2.5) and the associated wind tunnel work (Klein et al., 2011). Much of the focus of this work was on the form of the probability density function of concentration levels. Concentration fluctuation intensities, c'/C, in the central part of the plumes were typically between 50 and 100%, becoming large at the plume edges. Again, the overall conclusion was that enhanced mixing in the urban setting led to lower c'/C in comparison with dispersion in an undisturbed flow. The wind tunnel simulations of Pavageau and Schatzmann (1999) should also be mentioned. These addressed dispersion from a line source in a two dimensional street canyon and mapped conditions within the canyon. The intensity of concentration fluctuations was high near roof level but well below 100% near street level. Further aspects of concentration fluctuations are discussed in Sections 3.2.3 and 3.3.2.

3.2.3 Decay of concentration with distance measured during DAPPLE

We now turn to more quantitative analysis of the full-scale DAPPLE measurements and comparisons with wind tunnel modelling. The aim is to pull together some of the ideas developed above and to put them into a more quantitative setting for comparison with modelling.

In order to compare the full-scale and wind tunnel data, the measurements are made dimensionless in a way that follows from conservation of an emitted tracer. The mass of tracer released from the source, M (measured in kg), must balance the time-integrated flux of tracer advected through a plane perpendicular to the mean wind direction. Mathematically, this can be written:

$$M = \iint UD\,\mathrm{d}y\,\mathrm{d}z\,, \qquad (3.3)$$

where the dosage of gas accumulated over the sampling time, D (in kg m^{-3} s), is made dimensionless by choosing appropriate scales for the wind speed, U (in m s^{-1}), and lateral and vertical distances, y and z. Since we focus on the neighbourhood scale, we expect the urban geometry to control the dispersion. Furthermore, since the wind tunnel model is a scaled representation of the real geometry, any linear dimension of the urban geometry will suffice. Here we choose the mean building height, H, to make the y and z dimensionless. It is then natural to choose the mean wind speed at mean roof level, U_H, for a scale of wind speed.

If these scales are substituted into (3.3) then the dimensionless dosage is

$$D^* = \frac{DU_H H^2}{M}. \qquad (3.4)$$

Figure 3.20 shows scatter plots of D^* against the straight-line distance from source to receptor made dimensionless on building height, $R^*=R/H$. Figure 3.20a shows results from four days of full-scale experiments characterized by different above-roof wind directions and speeds (Table 1). Figure 3.20b shows results from the wind tunnel when the wind blows from the south west, at an angle of 45^o to Marylebone Road with a wind speed sufficient to ensure turbulence controls mixing. Figure 3.20c shows wind tunnel measurements for a range of wind directions.

Initially gas is carried from the source along the street containing the source as a spreading plume. When the wind blows at an angle to the street axes, the centre of the plume at this stage follows a helical path, because the wind has a component along street and another component driving a re-circulation across street (DePaul & Sheih 1986; Dobre et al 2005). Natural fluctuations in above-roof wind direction change the ratio of the two wind components, leading to strong variability in the path of the plume centreline. At very short distances from the source, the plume is narrow compared to the height and width of the street (Chang & Robins 2008). A receptor then samples either the narrow plume or the uncontaminated air, and so registers strong temporal fluctuations in gas concentration. This renders the details of dispersion very close to the source unpredictable. The data in Figures 3.20a and 3.20b support this view because they show that, within one street length of the source (R/H < 4), the highest measured values of the dosage can be directly downwind of the source, in a broad downwind sector, in a crosswind sector, or even upwind of the source!

At distances greater than about 4-5 building heights (approximately one street length) from the source, the colours in Figures 3.20a and b stratify, indicating that for a given R/H the highest concentrations are for channelled conditions, the next highest for a broader downwind sector from the source, then at crosswind locations and the smallest at upwind locations. The reason is that at greater distances gas is well mixed across street, so that the dosage is less affected by fluctuations in the above roof wind direction.

Figures 3.20a, 3.20b, and particularly 3.20c, show that further downwind, 5 < R/H < 20, there is a strikingly-sharp upper bound to the normalized dosage. An inverse square variation with normalized distance fits the data. Substituting from (3.4) indicates that the maximum dosage then varies with distance from the source as:

$$D^*_{\max} = A\left(\frac{H}{R}\right)^2, \text{ so that } D_{\max} = \frac{AM}{U_H R^2}, \qquad (3.5)$$

showing that D_{max} is independent of mean building height, H, where A is an empirical constant. Figure 3.20c shows that the wind tunnel data attains the upper bound for the full range of wind directions. Since the building geometry of the DAPPLE site is strongly heterogeneous, details of the dispersion pathways depend strongly on wind direction. The robustness to wind direction therefore provides evidence that (3.5) is of more general validity, and not an artefact of special geometry. Hanna et al (2007) demonstrate this yields a useful upper bound for other measurements.

A value of A = 12 certainly provides an upper bound for both the wind tunnel and full-scale measurements over the whole range of wind speeds and directions. This provides evidence that the normalization (3.4) works. In the full-scale data, the upper bound is attained for channelled flow, when the wind blows material along one of the major thoroughfares. For oblique wind directions, such as the data for 50^o shown as black points in Figure 3.20a, A = 6 provides a tighter upper bound on the full-scale data. Factors such as the natural variability in atmospheric dispersion (such as fluctuations in wind direction) and traffic-induced mixing not included in the wind tunnel might force greater mixing at full scale, yielding the lower maximum concentrations implied by A = 6. Nevertheless agreement to within a factor of two is indicative that the main processes are captured within the wind tunnel model, and that (3.5) is acceptable in dispersion applications.

A similar analysis was applied to concentration fluctuations but in this case only wind tunnel data were available. An example of the outcome is shown in Figure 3.21, adapted from Smethurst (2012) for release heights at 10, 102 and 153mm (see also the discussion in Section 3.2.2). The fast flame ionisation instrumentation used in this work had a frequency response of order 200Hz and a sampling volume of order 1mm in diameter (see Fackrell, 1980, for further details). An upper bound of the form expressed by Equation (3.5) is included in the figure as:

$$c'^{*} = \frac{c'U(H)H}{Q} = A\left(\frac{R}{H}\right)^{-2} \qquad (3.6)$$

where c' is the standard deviation of the concentration fluctuations and $A \sim 9$; data for other wind directions were also consistent with this relationship. A similar result was found to apply to plumes in the undisturbed flow but with $A \sim 20$, demonstrating again much lower fluctuation levels in the urban environment as a consequence of mixing in street canyons. Inter-comparison of the results for the three source heights shows that all conform to the upper bound once the respective plumes have become fully entrained into the street network; mean concentration fields (not shown) displayed the same behaviour. Note that the high intensity of concentration fluctuations, c'/C, at 'upstream' plume edges or where elevated plumes first 'touch-down' occur through a combination of low fluctuation levels, well below the upper bound in the figure, combined with even lower mean concentrations. The concentration field in such circumstances is highly intermittent.

3.2.4 Experimentation; full scale and wind tunnel

The study of the effects of traffic emissions on air quality has been by far the most common activity treating pollutant dispersion in urban areas. The ubiquitous nature of traffic emissions means this is not that revealing for present purposes, where carefully designed tracer and flow field observations are much more useful. The issue then becomes one of obtaining large enough ensembles of observations that can provide useful statistical information over the range of external parameters, such as wind direction, rather than being simply a set of individual realisations. This is, in fact, a very demanding requirement and very few 'scientific' dispersion studies have ever really produced sufficiently large datasets to meet it. The constraint is felt particularly strongly in short-range urban dispersion because of the high degree of inherent natural variability, coupled with confounding factors such as traffic conditions. Of course, the obvious virtue of long duration pollution monitoring, as opposed to experimental campaigns, is that it can meet the constraint; however, monitoring cannot satisfy present demands.

Wind tunnel work does not suffer from this limitation, as experimental conditions can be maintained for as long as is needed, and are repeatable. Output can also be as detailed as required (within reason) and, supported by flow visualisation, become a powerful tool for developing understanding of the physics of the processes investigated. However, wind tunnel work addresses a sub-set of the issues at work in the real problem. This 'weakness' can, of course, be turned to an advantage; for example, the effects of traffic can be eliminated; the mean wind direction in the boundary layer can be held steady and so on. Nevertheless, wind tunnel data cannot provide a stringent test of models that are to be used in the real world, though they do produce very useful tests.

The implication is that neither form of experimental study is complete of itself and that they should therefore be used in conjunction. This is in fact a well-established principle and has been exploited in most really successful studies. The wind tunnel plays two roles in this, to explore the physics and to compile high quality datasets. The field work complements both activities and is best designed also to quantify the variability in the phenomena at play. The downside, is the cost and duration that is implied.

3.2.5 Data

Although a number of urban flow and dispersion experiments have been undertaken few treat circumstances typical of cities in the UK. The sites of major experiments in the USA were Salt Lake City, Oklahoma and New York and in Europe Basel, Birmingham and London - perhaps only the latter two are of generic value to the UK. Indeed, to satisfy that requirement was one of the key reasons for the seven year DAPPLE programme. Reduced scale field experiments of note include the MUST study in the USA and a number of experiments in the UK, conducted at Alcar (by UMIST, as it then was), and also at Cardington. In all cases, the subject was a regular array, a very open array for MUST (and hence not typical of UK urban conditions) and cuboid arrays for Alcar and Cardington. A brief summary of the main studies and the key references is given below. More details on a number of these experiments are provided in Hunt et al. (2004). Wind tunnel work is included only where it was directly associated with the projects listed.

Altcar - UMIST
Key reference: Macdonald et al., 1997; Macdonald, 1997
Associated wind tunnel work: Hall et al., 1996; Macdonald, 1997
Wind tunnel flow visualisation work: Hall et al., 1997
Altcar (near Formby) Lancashire, UK
Meteorological and tracer dispersion studies using 1.13m cubes
100 cubes arranged in arrays with 7 or 8 rows; array densities 6, 16, 44%
cubes combined to form arrays of rectangular obstacles
source at 2H upstream or in first row
source and receptors at mid-cube height

BUBBLE - Basel
Key references: Rotach, 2002; ; Rotach et al., 2004; Rotach et al., 2005
Wind tunnel simulation (1:300 scale): Feddersen et al., 2004
Web-site: http://pages.unibas.ch/geo/mcr/Pbojects/BUBBLE
Basel, Switzerland, year long meteorological study, including intense observation periods
4 tracer experiments in June and July, 2002
light wind, unstable conditions
near roof level sources and sampling
measurements on arcs at 700 and 1000m, with additional samplers to 2.4km
6x30 min samples per experiment

Birmingham
Key reference: Britter et al., 2002
Birmingham, UK, 1999 - 2000
3 dispersion experiments in July 1999, February and August 2000
windy, near-neutral conditions

near surface release and sampling
measurements on arcs between 1 and 4km
1999: 40 min release and 15 min sampling
2000: Feb, 35 min and 3 min; Aug, 20 min and 6 min.

Cardington
Key reference: Davidson et al., 1995
Wind tunnel simulation (1:20 and 1:200 scale): Davidson et al., 1996
Cardington, UK, 1990
15 dispersion and smoke visualisation experiments in October and November, 1990
windy conditions
aligned and staggered array of 39 2.2x2.45x2.3 m tall obstacles
upstream source positions between 1 and 24 building length scales
source and receptors at obstacle mid-height

CEDVAL - University of Hamburg
Key reference: Pavageau et al., 2001
Web-site: www.mi.uni-hamburg.de/Introducti.433.0.html
database of flow and dispersion results from wind tunnel simulations
isolated buildings, 7 cases
regular arrays, 6 cases

DAPPLE - London
Key references: Wood et al., 2009; Arnold et al., 2004; Dobre et al., 2005, DAPPLE, 2011
Wind tunnel simulation (1:200 scale): Robins et al., 2010
Web-site: www.dapple.org.uk
London, UK, urban meteorology and dispersion study, 2002-2009
light to medium wind conditions
51 tracer releases of 15 minute duration
short range dispersion, less than 1km
mainly near surface sources and samplers

Joint Urban 2003 - Oklahoma
Key references: Allwine et al., 2004
Wind tunnel simulation (1:300 scale): Kastner-Kline et al., 2004
Web site: www.noaa.inel.gov/projects/ju03/ju03.htm
Oklahoma City, USA, 2003
meteorological and tracer dispersion campaign, summer 2003
10 intensive study periods; 6 day time, 4 night time
3x30 mins plus 4 puff release per study period
range: near field to 4km

MUST
Key references: Biltoft, 2001; Biltoft et al., 2002
Wind tunnel simulation (1:75 scale): Leitl et al., 2007
Water tank simulation (1:205 scale): Hilderman et al., 2004
Dugway Proving Ground, Utah, USA, 2001
MUST (Mock Urban Setting), sparse array (approximately 10% area coverage)
10x12 array of shipping containers (12.2 m long, 2.4 m wide, and 2.5 m high)
vertical and horizontal concentration profiles and associated meteorology
63 continuous releases providing 16 hours of data
5 multiple puff releases providing 4.75 hours of data

New York, 2004 – 2007

Key references: Allwine and Flaherty, 2006, 2007; Reynolds et al., 2006
Meteorological and tracer campaigns, New York, USA
all wind conditions
Madison Square (MSG05), March 2005
2 intensive study period; 6 tracers emitted, 5 sources per period
sampling area 1x1km (some indoor samplers)
Manhattan (MID05), August, 2005
6 intensive study period; 8 tracer emitted per period, 10 source locations
sampling area 2x2km (some indoor and subway samplers)

Urban 2000 - Salt Lake City
Key reference: Allwine et al., 2002
Web site: www.noaa.inel.gov/projects/urban2000/urban2000.htm
Salt Lake City, Utah, USA, 2000
meteorological and tracer dispersion campaign, October 2000
night time, light to moderate wind speeds
7 intensive observation periods, multiple sources and tracers
total: 18x1 hour, 12x6 hour and 12x18 hour emissions
range: near field to 6km

3.3 Direct numerical simulation and large eddy simulation

In the last decade or so, large-eddy simulations (LES) and direct numerical simulations (DNS) of urban flow and dispersion have progressed to the extent that they can now be viewed as credible sources of data, on par with experimental measurements. In contrast to most other computational methods, the approach of DNS and LES is to directly resolve, rather than model, the turbulence scales of interest. This comes at a high computational price: simulations of relatively simple geometries typically take days or weeks, running on dozens of processors on currently available supercomputers. DNS and LES are therefore impractical for operational dispersion modelling, and are likely to remain so for the foreseeable future. However, they are excellent research tools for the investigation of turbulent flow and dispersion processes, complementing wind tunnel and field experiments. With the additional advantage of providing comprehensive high-resolution spatial and temporal data, they can be used to evaluate simplified models and their assumptions more stringently than is practicable with experimental data.

Most of the work in LES and DNS of urban flows in existing literature has focused on three general types of urban geometry: 2D street canyons, 3D regular arrays of cuboids, and more realistic urban geometries that represent some degree of randomness (Barlow and Coceal, 2009). Examples of studies in each of these categories will be reviewed in turn in the following sections, together with a summary of their validation

3.3.1 Two-dimensional street canyon geometry

A number of LES studies exist for flow and dispersion over 2D 'street canyon' geometries. Notable examples include a series of papers by Liu and collaborators (Liu et al. 2002, 2004; Li et al. 2008, 2009) that systematically examined the influence of street canyon aspect ratio H/W on flow patterns and ventilation (with H/W ranging from 1 to 10) and similar studies by Cai et al. (2008) but for a different range of H/W values from 1/3 to 3/2. Together, these studies give a consistent picture of the main dispersion processes in infinitely long, two-dimensional regular street canyons.

Liu and Barth (2002) simulated perpendicular flow over a canyon with aspect ratio H/W = 1 in a domain with streamwise periodic boundary conditions to simulate an infinite sequence of street canyons. They demonstrated good agreement of predicted mean velocity, turbulence intensities and Reynolds stress profiles with wind-tunnel measurements. Good spatial resolution of the flow field allowed the authors to study scalar transport in the street canyon. A line source of continuous passive scalar was placed at street level in the middle of the canyon to simulate traffic emissions. With the approach flow perpendicular to the street axis, the simulations generated a primary vortex in the street canyon which was isolated from the free stream flow. The detailed concentration pattern was found to be dependent on the source location. The authors concluded that turbulent diffusion rather than mean advection was the dominant mechanism for scalar removal at roof level.

Liu et al. (2004) extended this study to canyons of aspect ratio H/W = 0.5, 1 and 2. They found similar flow patterns for canyons of aspect ratio 0.5 and 1, namely a primary recirculation in the canyon centre and secondary recirculations in the ground-level corners. For the higher aspect ratio, H/W = 2, there were two primary recirculations above each other and two ground-level secondary recirculations. The ground level concentration was found to be greatest at the leeward corner for the street canyons of aspect ratio 0.5 and 1, and at the windward corner for the canyon of aspect ratio 2. This is linked to the scalar following the trajectories of the primary and secondary recirculations.

After extensive validation of the results of calculations for aspect ratios of 1 and 2, Li et al. (2008, 2009) extended these studies to even higher canyon aspect ratios of 3, 5 and 10. They found that respectively three, five and eight vertically-aligned primary recirculations were formed in these canyons, their strength decreasing with decreasing height. It was also found that advection was responsible for pollutant redistribution within the primary recirculations, whereas turbulent transport was responsible for pollutant exchange between the primary recirculations as well as removal from the street canyon.

Cai et al. (2008) performed a well-validated set of LES for street canyons of aspect ratio H/W = 1/3, 1/2, 2/3, 1 and 3/2 using line and area sources in the street canyons. Scalar concentrations, concentration fluctuations and scalar fluxes across the roof level were compared with the wind-tunnel measurements of Barlow et al (2004) and showed promising agreement. Profiles of mean scalar flux at the roof level were found to be indicative of the flow regimes recommended by Oke (1988) – namely isolated building, wake-interference and skimming flow, depending on the value H/W.

3.3.2 Three-dimensional regular arrays

Real urban areas rarely consist of long, two-dimensional street canyons. As discussed in the earlier part of this Section, the three-dimensional nature of the flow and urban geometry, and the presence of street intersections modify the dispersion patterns and processes considerably. Credible LES and DNS of flow and dispersion over 3D urban-type geometry have emerged in the last few years and the recent DNS study of Branford et al. (2011) is especially pertinent to the objectives of this part of the review. It will therefore be described in some detail.

Following the approach described in Coceal et al. (2006, 2007), Branford et al. (2011) performed a series of DNS runs for different wind directions (0°, 30° and 45°) over a regular array of cubical obstacles of height H arranged in an aligned configuration with plan area density of 0.25. This choice of building dimensions and packing density corresponds to the boundary of the 'street network' region in the regime diagram of Figure 4.1, discussed in Section 4.1. Hence, the 'streets' are rather short and, based on the discussion in Section 3.1, end effects would be expected to play a more significant role. The domain size was 16H x 16H in the horizontal and 8H in the vertical – see Figure 3.22. The simulations were conducted under conditions of neutral stability and fully rough turbulent flow. Periodic boundary conditions were imposed in the horizontal directions, effectively simulating the flow over an infinite array of cubes. Dispersion of a passive scalar released from an ensemble of point sources close to the ground (at a height of 0.0625H) within the simulated urban area was investigated with the explicit aims of: (i) establishing the fidelity of DNS for dispersion in this complex geometry through numerical tests and comparisons with available experimental data (ii) analysing the generated data to investigate key dispersion processes in urban areas at street and neighbourhood scales (iii) generating high quality datasets to evaluate simpler models such as street network models and Gaussian plume models discussed in Section 4 below.

Validation of the simulations

The Reynolds number based on the velocity at the top of the domain and the cube height was typically between 4750 and 7000. While this is much less than Reynolds numbers at full scale, it is comparable to typical Reynolds numbers achieved in many wind tunnel experiments. Numerical tests showed that a grid resolution of H/32 was sufficient, producing flow and concentration statistics that agreed with test runs at double the resolution (H/64) to within a few percent (Coceal et al. 2006, Branford et al. 2011). Numerical tests were also performed to demonstrate scalar conservation. For the 0° simulation, when the wind was perpendicular to the face of the buildings, detailed comparisons of mean concentration profiles at different locations within and above the array showed good agreement with data from a water-channel experiment on the same configuration of cubical buildings performed by Hilderman and Chong (2007) – see Figure 3.23. Suitable experimental data for comparison for the other wind directions was not available.

Dispersion processes

Analysis of data from the DNS reproduced a number of qualitative features and processes that had been observed in a number of field and wind-tunnel experiments on urban dispersion (e.g. Davidson et al. 1995, 1996; Macdonald et al. 1997, 1998), as discussed previously in this Section.

First, the 'channelling' of scalar by advection along streets and unobstructed passages through the array was evident. Channelling arises in regions where the mean advection dominates over turbulent dispersion. In a source-free region this implies that $\overline{\vec{u}} \cdot \nabla \overline{c} \approx 0$, so that the mean velocity at a point is normal to the gradient of mean concentration or, equivalently, is parallel to the isoline of concentration at that point. By comparing the mean concentration pattern with the mean flow field this condition can therefore be used as a diagnostic to identify channelling regions.

The amplification of lateral turbulent dispersion by 'topological diffusion' (Davidson et al. 1995, Hunt et al. 2004; Belcher 2005) at the intersections was noted for oblique flow directions, resulting in a much wider plume than for perpendicular flow through the array. In this relatively close-packed regime this is the dominant mechanism for the plume spread, so that the plume width is effectively determined by the building geometry, see Section 4.4 below.

Visualisation of the instantaneous concentration pattern for the 45° simulation revealed the existence of two distinct, smaller plumes originating from the wakes of the two buildings on either side of the source location, which was in an intersection for this simulation. These arise due to the entrainment and re-release of material in the building wakes, giving rise to 'secondary sources' (see Figure 3.24). Whilst such secondary wake sources have been observed in previous experimental studies, such as those of Davidson et al. (1995, 1996), their effect in the present simulations appeared to be more significant since the original source was in the array, so that a large fraction of material is entrained into the building wakes before it is able to diffuse out to any significant degree. The wake sources then cause a rapid detrainment of material out of the canopy top. As a consequence, the secondary sources have a greater effect than the original source on the near-field concentration pattern. Goulart (2012) showed that they give rise to non-Gaussian bimodal lateral concentration profiles in the near field. Similar bimodal concentration profiles were observed in the earlier experiments of Macdonald et al. (1997, 1998).

In addition to these well-documented processes, the DNS brought to light a lesser-known phenomenon. For non-symmetric flow configurations (e.g. the 30° DNS case) the flow turns with height within the canopy, due to a 'side' force (Claus et al. 2012), causing a skewing of the plume with height. This has two consequences: lateral dispersion within the canopy is enhanced, and the average orientation of the plume within the canopy differs from that above it. This effect is absent when the flow is symmetric (e.g. for the 0° and 45° runs on the regular DNS array). Branford et al. (2011) report a more diffuse concentration distribution for their 30° run compared to the 45° run, which they attribute to this mechanism. Any asymmetry in the flow direction or building configuration would induce this effect. We therefore expect such an effect to operate in a real urban geometry. In a sense this is also a channelling effect (assuming horizontal advection dominates in the canopy), with the channelling being along the mean wind vector field (rather than necessarily the canyon axis), whose direction may be different at different heights. The effect of this mechanism is that the plume also changes direction through the depth of the canopy and is therefore spread through a wider angle, in addition to having a different mean orientation relative to that of the plume above the canopy.

Dispersion regions

Goulart (2012) further analysed the DNS data from these simulations and showed that the dispersion through the street network could be characterised broadly in terms of three regions: a near-field region where detrainment out of the canopy top is rapid and horizontal topological dispersion through the array is pronounced, an intermediate region where re-entrainment of material from the above flow into the canopy is significant, and a far-field region where the plume is large compared with the building size and detrainment and re-entrainment fluxes are small and in balance. Goulart (2012) further showed that in this far field region lateral profiles of mean concentration are approximately Gaussian, in accordance with previous experimental findings (e.g. Davidson et al. 1995, 1996; Macdonald et al. 1997, 1998).This can be taken as a definition of the far field.

Concentration fluctuations

One of the advantages of DNS, compared with the RANS (Reynolds Averaged Navier-Stokes) approach more widely used in CFD simulations, is the ability to compute reliably concentration fluctuations as well as mean concentrations. Such information is important because even brief exposure to extremes of concentration of certain substances can be harmful. Empirical information on fluctuating concentrations can usefully complement mean concentrations derived from simple predictive models.For example the SIRANERISK model uses such a semi-empirical approach to predict concentration fluctuations – see Section 4.4; similarly, ADMS has a concentration fluctuation module based on a stochastic model for the motion of particle pairs that requires empirical input – see Thomson (1990, 1992). Branford et al. (2011) compared concentration fluctuations at different distances from the source within their cubical array (at $z = 0.25H$) with those just above (at $z = 1.25H$). They found that there was a large decrease in the fluctuations from the first row to the third row downstream of the source (cf. the discussion around Figure 3.13). Concentration fluctuations and intermittency were found to be much larger above the array than within it – see Figure 3.25. The reduction of concentration fluctuations and low intermittency within the array compared to that above it is reminiscent of the findings of Davidson et al. (1995), who observed a similar reduction compared to the fluctuations in a control plume outside of (next to) their simulated urban array. As discussed by Davidson et al. (1995), several factors contribute to the reduction in fluctuations within the array: the increased plume size due to rapid lateral spread and vertical mixing through the canopy depth, reduced turbulence scales within the array which contribute to mixing rather than meander, and multiple wake sources which tend to smooth out any plume meander.

The characteristics of the time series at different distances from the source as shown in Figure 3.25 reflect the spatial spreading and overall displacement of the plume. Close to the source the whole plume is narrow compared to the buildings and so is easily displaced by the predominant eddies in the flow, which have sizes comparable to the building dimensions. Hence, any particular fixed point around the source (say at a distance not more than 1–4 building heights) can be either within or outside of the plume at any given moment. This explains the large fluctuations and intermittency in the time series at these points. If this picture is true then it also implies that different points in the vicinity of the release can register large concentrations intermittently, irrespective of where they are located around the source. This corresponds with the DAPPLE observations described in Section 3.2.2.

The concentration fluctuations were quantified by plotting the relative concentration fluctuation ratio c_{rms}/C_{mean} as a function of downstream distance from the source. The relative concentration fluctuations decrease monotonically with distance from the source, the rate of decrease being higher outside of the urban array. Within the array the ratio appears to reach an asymptotic value of approximately 0.4–0.5 by the third row downstream – see Figure 3.26. This asymptotic value is similar to the experimental results of Fackrell and Robins (1982) and the large-eddy simulations of Xie et al. (2007) over a rough surface, as well as the scaled field experiments of Davidson et al. (1995) for urban arrays.

The successful quantitative and qualitative validation of the DNS give confidence in the credibility of DNS as a tool for accurate urban dispersion simulation, and therefore in the potential use of the generated DNS data for validating and testing simpler models. An example of such use is discussed in Section 4.4.

3.3.3 Realistic three-dimensional geometry

DNS studies of realistic three-dimensional geometries do not currently exist, but a number of recent examples of the application of LES have emerged in the last decade or so. To establish the credibility of LES as a simulation tool for practical application to real urban areas, Xie & Castro (2009) performed a careful simulation of flow and dispersion over a detailed representation of an area of central London – the 'DAPPLE site'. The modelled area comprised more than 50 buildings centred around the intersection between Marylebone Road and Gloucester Place, a pollution hotspot in central London. The average building height in this area is 22 m. Building shapes were taken to be flat-roofed blocks, as in the wind tunnel work that provided the data for evaluating the LES work. The authors confronted some difficult, but important, numerical issues, including whether both the flow and scalar dispersion could be adequately resolved using a computationally feasible grid resolution and time step, and how best to implement upwind boundary conditions to model a necessarily limited area. They found that a resolution of one meter in space and one second in time was sufficient to give reasonable turbulence and scalar statistics. They also found that it was crucial to have a proper inflow generator; for example using periodic boundary conditions did not yield satisfactory agreement with wind-tunnel data. On the other hand, they also found that the Reynolds number dependency of such flows was very weak so that complex urban flows are significantly easier to compute than equivalent flows over smooth walls.

A simulation with an oblique wind direction of –51.4° to Marylebone Road (in a clockwise sense) was compared with wind-tunnel data obtained during the DAPPLE programme (http://www.dapple.org.uk). Mean wind velocity, velocity r.m.s and

Reynolds stress profiles were in good agreement, as were spatial plots of mean wind vectors. Point source dispersion was investigated with five different tracer releases upstream of the main intersection. Mean and fluctuating concentrations along the main streets were compared with the wind-tunnel measurements, and showed reasonable agreement. The mean and r.m.s. concentration patterns were compared for pairs of sources separated across and along a canyon respectively. In both cases the scalar dispersion in the near field (defined therein as within a distance of the order of the local building height from the source) was found to be sensitive to the location of the source, whereas in the far field it was not. This can be viewed as an alternative definition of the far field. In the first case, the position of the source in relation to the canyon circulation was critical; with the source in the middle of the canyon, more of the material was driven to the leeward wall and was more likely to be flushed out of the canyon than if the source was near the windward wall. In the second case, the circulation at the street end led to high concentration upstream when the source was close to the street end, otherwise material was mostly channelled downsream.

In order to investigate the effect of wind direction on the flow and dispersion, Xie (2011) performed a further LES with a wind direction perpendicular to Marylebone Road and the windward faces of most of the buildings. Once again, both the flow and scalar fields were validated using wind-tunnel data. Furthermore, time-varying wind conditions measured on the BT tower at 190m above street level were used as input, leading to significant improvements of the predicted dispersion compared with field measurements. This highlights an important consideration when evaluating models against field data. Not surprisingly, mean statistics and instantaneous concentrations were found to be sensitive to the wind direction. For a regular array it is to be expected that topological dispersion around the buildings would be predominant for oblique flow whereas perpendicular flow would be characterised by chanelling along streets aligned to the flow and recirculations in the streets perpendicular to the flow. However, the more complicated geometry of the DAPPLE site includes non-regular features such as misalignment of buildings at the intersections, and variations in building heights. These irregularities affect the flow and dispersion in a non-trivial way, as discussed in earlier in this section. Tall buildings in the centre of the area were also found to affect the flow disproportionately, with the turbulent kinetic energy and shear stress profiles peaking at the heights of the tallest buildings regardless of the wind direction, confirming the previous LES of Xie et al. (2008).

3.3.4 Recommendations for further LES/DNS work

The provision of comprehensive flow and concentration data at high spatial resolution and the ability to predict fluctuating concentrations make LES and DNS very attractive approaches. As the use of these methods becomes more widespread due to increases in computer power it is becoming feasible to explore a wider range of parameter space including more complex geometries. Much of the literature to date has tended to focus on cube arrays where ‘streets’ are short and end effects are therefore particularly dominant. Such arrangements may not entirely reflect the street network type geometry of most European cities. It would therefore be useful to focus more on geometrical arrangements with longer streets made up of, for example, arrays of relatively flat rectangular blocks similar to the DAPPLE site. Another regime that has not been studied much, and therefore deserves more attention, is that involving groups of tall 3D buildings. Fundamental knowledge of this regime and good datasets are lacking, and would be necessary to inform and test simpler models. Groups of wide buildings provide another interesting avenue of research, as they would shed light on how the flow and dispersion processes evolve in the transition between 2D and 3D geometry. The transition between regular arrays of buildings and the complex geometries of real urban areas is also an area worthy of investigation. It would be of interest to randomise particular aspects of the geometry in isolation to understand their effects in a more controlled way – this could for example include variations of building height for given layouts of buildings, or changes in the arrangement of the buildings.

Finally, all of the above is equally applicable to wind tunnel work. A more fruitful approach is to combine numerical and experimental work so that each supports and complements the other. Typically, wind tunnel measurements can be used to validate LES and DNS which can then be used as reliable sources of data, providing far higher resolution than is possible experimentally. Wind tunnel experiments can also fulfill a significant investigative role. Rapid sensitivity and regime studies (flow visualisation and quantitative) will remain an advantage for some time yet. Where possible, future studies should therefore aim to employ a combination of numerical and physical modelling tools.

4. Modelling dispersion in and above urban areas

In this section we introduce methods for modelling dispersion in urban areas. There are three types of application of such models: (i) direct mode: dispersion from a known source to compute the pattern of concentration; (ii) inverse mode: simulations to determine the location and strength of the source from known concentration measurements (see Rudd et al 2012); (iii) scenario mode: evaluating the effects of different releases from a hazardous or sensitive location in different meteorological situations. Each application requires an atmospheric dispersion model at its heart. When used in direct or inverse mode the dispersion model needs to run quickly, within a few minutes. When used in scenario mode the model can run more slowly. Clearly then there is a need for models that compute rapidly, and these are the models that are the focus of this section. The aim is to calculate the statistics of the concentration, denoted $C(\boldsymbol{x}, t)$. We focus mainly on the mean concentration, although we also mention the root-mean-square of the fluctuations. It would also be desirable to have estimates of the likely peak values of the concentration.

4.1 Controlling parameters and regimes

There is a fundamental question over the relative roles of the turbulence in the atmospheric boundary layer and the complex flow through the buildings in determining dispersion in urban areas. The relative roles of these processes depend on the geometry. Figure 4.1 shows a regime diagram for dispersion in urban areas, motivated by the qualitative discussion in Section 3. The figure is formulated for a regular array of cuboid buildings of base L x L, height H and the gaps between buildings are of width W. The *streets*, defined here to denote the area between the buildings (including road surface, pavements, etc.), are therefore of width W and length L. The axes on the figure are the width of the streets and the height of the buildings, both normalised on the length of the streets. When $W/L < 1$ the width of the streets is less than their length and it makes sense to talk of streets; when $W/L > 1$ the buildings are too widely spaced to speak of streets (in the sense used here). Flow visualisation (e.g. Oke 1987) shows that, for long two-dimensional streets,$W/L \ll 1$, the flow regime is determined by the ratio of H/W. When $H/W \geq 1$, there is a recirculation region in the streets between the buildings. When $H/W < 1/3$ the buildings produce wakes with weak interactions. Hence when $H/W < 1/3$ we have a *sparse array*: the obstacle wakes interact weakly and dispersion beyond a near-source region is as over a rough surface. When $W/L < 1$ so that we can speak of streets, and when $H/W > 1$ and $H/L < 3$ the recirculation region fills the streets between the buildings and mixes nearly uniformly across the width and height of the streets. In this regime a *street canyon* or *street network* approach is appropriate. Finally, when $H/L > 3$ the buildings in the array are much taller than the streets are long and mixing induced at the building tops does not penetrate to the base of the buildings. In this case the array of buildings act as a *tall building canopy*, and methods developed for vegetation canopies are appropriate. Finally the parameters for the DNS of Branford et al (2011) are shown with the filled square. Methods for modelling dispersion in each of the three regimes identified in Figure 4.1 are discussed separately in subsequent sections.

We can estimate the range of regimes in London as an example of a European city by using data developed in Bohnenstengel et al (2011). They analysed the Virtual London dataset (Evans et al 2005) to produce maps of λ_p and λ_f (as defined in Equation 2.2). The London area was divided into a grid of 1km x 1km boxes. Within each grid box the fraction of urban land use, *f*, was determined. Then the frontal and plan areas of the buildings within the 1km x 1km grid boxes were computed. Values of λ_p and λ_f for each grid box were then computed by dividing the plan area and frontal area by the area of urban land use within each grid box (see Figures 2 and 3 in Bohnenstengel et al (2011).

On the rough assumption that the urban geometry is made up of uniform cuboids, the likely range of dispersion regimes encountered in London can then be estimated by using these values of λ_p and λ_f to estimate values for *W/L* and *H/L*, as follows. For cuboidal buildings with base *L* x *L*, height *H* and gaps between buildings, the streets, of width *W*, the parameters λ_p and λ_f can be rearranged to give

$$\frac{H}{L} = \frac{\lambda_f}{\lambda_p}, \qquad \frac{W}{L} = \lambda_p^{-\frac{1}{2}} - 1. \tag{4.1}$$

Hence we can use the values of λ_p and λ_f found by Bohnenstengel et al (2011) to estimate equivalent values of *W/L* and *H/L*. The estimated ranges of values is shown in Figure 4.1 by the blue ellipses, which show, as might have been expected, the suburban surroundings (*f* = 0.1) are a sparse array and the central region (*f* = 0.9) are a street network.

In this section we focus especially on two regimes, sparse arrays and street networks, because these regimes have modelling techniques that are comparatively well developed. In section 5 below we discuss new ideas that could be useful for tall building canopies, which are important for many cities, such as Hong Kong or New York, with very dense arrays of tall buildings.

4.2 Sparse arrays of buildings

In terms of the regime diagram of Figure 4.1, the distinguishing feature of sparse arrays is that there are no clearly defined streets or intersections between the buildings. Flow and dispersion in such arrays can then be treated by considering the superposition of effects from each building; no street canyon modelling is involved.

When the buildings are sufficiently far apart and sufficiently short that $H/W \gtrsim 1/3$ and the wind blows at an angle $90^o - \theta$ to the streets (see Figure 4.2) then the wake of one building does not interact strongly with the downstream buildings. This regime was discussed in detail in Hunt et al (2004) and only a brief overview is given here. First we consider properties of the flow and then dispersion processes.

4.2.1 Superposition of N building wakes with no interaction $(W \gtrsim 2L\, tan\, \theta)$

The buildings are sufficiently far apart that the wakes are parallel to the mean flow direction (see Figure 4.2) and may be calculated separately for each building (denoted by *n*, where *1<n<N*, the building location being at $\boldsymbol{x}_n$). This is the model for the flow used, at least implicitly, in UDM (Hall et al 2001). The mean velocity perturbation due to a single wake is represented by $\Delta \boldsymbol{U}_w$, and total flow field in the sparse array is written as a linear combination of the approach flow $\boldsymbol{U}_o$ and the superposition of the *N* wakes associated with the buildings:

$$\boldsymbol{U} = \boldsymbol{U}_o + \sum_{n=1}^{N} \Delta \boldsymbol{U}_w\, (\boldsymbol{x}\text{-}\boldsymbol{x}_n) \tag{4.2}$$

ADMS represents the effects of buildings in this way, although currently the method

is implemented for only a single building or a group of buildings represented as one effective building. For dispersion calculations it is also necessary to model the relevant turbulence statistics in the wakes, which can also be estimated as a set of linear perturbations. Explicit formulae are given for ADMS in the technical specification for buildings, see:
http://www.cerc.co.uk/environmental-software/assets/data/doc_techspec/CERC_ADMS4_P16_01.pdf

This approach could be extended to allow for weak wake interactions: The effect of upwind buildings is largely to change the approach velocity to downwind structures. The approach wind at the m^{th} building is then estimated as $\boldsymbol{U_o}$ plus wake perturbations[1]. In brief, if the approach flow, $\boldsymbol{U_0}$, is uniform then for an isolated building $\Delta\boldsymbol{U_w} \approx \boldsymbol{U_0}\Delta U$, where ΔU is the change in velocity. In a sparse array of buildings, each building experiences an approach flow $\boldsymbol{U_{local}}$ determined by the upstream wakes. Hence $\Delta\boldsymbol{U}_w \approx \boldsymbol{U_{local}}\,\Delta U$ provided the upstream wakes are wide enough when they impinge upon the downwind building. Thus over many buildings $\Delta\boldsymbol{U}_w$ will decrease exponentially in the downwind direction (as in the estimate of the resistance of obstacles in a canopy in Belcher et al 2003). If the increased turbulence generate in the upstream wakes has a smaller length scale than in the approach flow, then the effect on dispersion is mainly via the mean flow (Davidson *et al.*, 1995).

4.2.2 Flow with wakes enveloping downstream building ($\boldsymbol{W} \lesssim \boldsymbol{L}\,\boldsymbol{tan}\,\boldsymbol{\theta}$)

Consider rows of buildings on either side of the m^{th} street (with a total of M streets). When the gap W between two buildings (on the same side of the street) is small enough and if the buildings are approximately aligned with the wind, i.e. $W < 2L\sin\theta$ where the angle between the wind direction and the line between the buildings is $(90^o - \theta)$, (see Figure 4.3), then the wake of the upwind building tends to envelope the downwind building (Meinders and Hanjalic, 2002), This leads to a weak canyon effect, *i.e.* an increase in the component of the mean flow along the street, denoted by $\Delta\boldsymbol{V}_m$. There is also a reduction in the U component because of the wake deficit, $\Delta\boldsymbol{U}_{w_{n,m}}$, downwind of each of the n buildings in the m^{th} row of buildings. This wake deficit is parallel to $\boldsymbol{U}_o$ so that below the average height of the buildings the U component is reduced and the V component increased, significantly skewing a dispersing plume (Macdonald et al 1998). Adopting a simple linear superposition, the wind speed in the m^{th} street is given by

$$\boldsymbol{U} = \boldsymbol{U}_o \;+\; \underbrace{\textstyle\sum_{m=1} \Delta\boldsymbol{V}_m(\boldsymbol{x} \in D_m)}_{\uparrow \text{ in the } m^{th} \text{ street}} \;+\; \underbrace{\textstyle\sum_{m'}\sum_m \Delta\boldsymbol{U}_{\boldsymbol{w}_{m,m'}}}_{\uparrow \text{ wakes of } n^{th} \text{ buildings in } m^{th} \text{ street}} \qquad (4.3)$$

For typical low rise, suburban streets with uniform height and geometry, the canyon effect in street m is only significant when θ is greater than about 45^o.

4.2.3 Dispersion with no interaction of wakes

For continuous point sources the approach used in ADMS building module, AERMOD prime and UDM is to superpose solutions corresponding to different

[1] Interactions between parallel wakes may have to be considered even when the wakes are several building heights apart – recall that wakes of obstacles in channels or pipes persist longer than wakes in unconfined flows (e.g. Taylor and Whitelaw, 1984).

components of the flow but ensuring that the mass flux of pollutant is conserved.

If the localised source is at $\boldsymbol{x} = \boldsymbol{x}_s$ and $C = C_o(\boldsymbol{x}_s)$, then, as in ADMS, in the wake

$$C = C_o + C_w - \Delta C_o \tag{4.4}$$

Here C_w is the change in concentration caused by the wake, which can be modelled as dispersion from a virtual source, whose strength is determined by the value of C_o at the building surface. For example, C_w is small compared with C_o if the source is at height $\boldsymbol{z}_s$ greater than *2H.* If the source is within the array canopy and is immediately upwind or within a wake of a building, C_w can be much greater than C_o. Downwind of the wake the concentration from the upwind source C_o is reduced ΔC_o by the amount of pollutant that is entrained into the wake. Equation (4.4) is then applied to each building and each source. Details are given in Puttock and Hunt (1979), Robins *et al.* (1997) and Huber (1991). This method is implemented into ADMS for single buildings.

4.2.4 Dispersion with interacting wakes and street models in sparse arrays

The principle of split plumes has been extended to calculate dispersion of plumes from lower or upper level sources that impinge on buildings in streets with significant canyon flows (see Figure 4.4) (Hunt et al 2003).

(a) Elevated source

Consider an elevated source at height $\boldsymbol{z}_s$ greater than the building height H, located at $\boldsymbol{x}_s$ upwind of the m_s^{th} street. The wind is at angle 90^o-θ to the streets (θ is positive), so that as it passes over the m_s^{th} street some of the plume diffuses down into the street. Therefore, as in the previous case of Section 4.2.3, the original plume has concentration depleted to C_o- ΔC_o. Within the street the scalar is transported by the street flow in the *y* direction over a distance of order 3*H* (Fackerrell 1981), as in ADMS BUILD (see also Hunt et al 2003). Then it diffuses upwards again, and a new plume emanates from the street, which acts like a detraining wake, with perturbation concentration C_s. This causes the upper plume above the buildings to be distorted and effectively to have a transverse displacement $\Delta\tilde{y}_{s_m}$ by the mth street and for its width to be increased by $\Delta\sigma_{y_m} \sim H$. For subsequent streets there is an additive displacement Δy_{s_m}, but the increment becomes progressively smaller as the ratio of the plume depth σ_z to the building height H increases.

Since all the pollutant entrained into the street eventually returns to the elevated plume, when averaged over the distance between streets, *w*, the concentration C can be expressed as a Gaussian plume formula above and below the roof level, with the lateral displacements (see Figure 4.4a)

$$\Delta\tilde{y}_{s_m} \sim H, \ \Delta\tilde{y}_{o_m} \sim (\tfrac{H}{\sigma_z})\Delta\tilde{y}_{s_m} \tag{4.5}$$

Note that V_m/U_o varies with wind direction as $1/\tan\theta$. This reduces the displacements $\Delta\tilde{y}_{s_m}$, $\Delta\tilde{y}_{o_m}$ in the street level and upper level plumes.

(b) Source in streets

Suppose now that there is a source of strength Q_s located within the street (see

Figure 4.5). Since the rms turbulence in the street is a significant fraction of the mean wind speed, pollutants can disperse upwind of small localised sources, (NRPB R292 1996; Wood et al 2009). Within the street the centreline of the plume is displaced a distance $\Delta\tilde{y}_{s_m}$ perpendicular to $\boldsymbol{U}_o$; whereas above the street it is displaced $\Delta\tilde{y}_{o_m}$. In other words, above the street the plume diffuses as if from a virtual point source at $\boldsymbol{x}_o$ (Hunt *et al.*, 2004). For subsequent streets $(m > m_s)$, the dispersion is calculated using further the concept of split plumes as defined in Equation (4.5).

Hunt et al (2004) constructed an approximate model for the pollutant concentration near the source in a very long street as a combination of a well mixed model in the canyon, but decaying along the street in the y direction, and Gaussian plume model above the canyon, defined in terms of a virtual source at x_0 upwind of the street in the direction $\boldsymbol{U_0}$, as follows. For a very long street of length L_s where $L_s \gtrsim 5W$, downwind of the source $(y - y_s) \gtrsim H$, so that the street the concentration is well mixed across and within the streets and l_H is the distance the plume disperses along the street before it is advected downwind, Then the concentration within the street is

$$C_s \sim \frac{10Q_s}{Wl_H\boldsymbol{U}_o} \sim \frac{Q_s}{WH\boldsymbol{U}_o} \tag{4.6a}$$

(note that l_H is not sensitive to θ); because of vertical diffusion and downwind advection, the concentration decays exponentially when $(y - y_s) \sim l_H \geq 2H$. If $L_s \lesssim 5W, |\theta| \gtrsim \mathrm{L/10H}$, for $(y - y_s) < L_s$

$$C_s \sim \frac{Q_s}{\boldsymbol{U}_o \sin\theta WH} \tag{4.6b}$$

(i.e. the plume is advected down the whole length of the street)

The concentration above the roofs is calculated using a Gaussian plume model with a virtual source at x_v, y_v where $x_v \cong x_s - (W + L)$ and $y_v = y_s$. The coordinates relative to the virtual source are, in street coordinates, $\Delta y_v = y - y_v$, $\Delta x_v = x - x_v$ and in plume coordinates perpendicular and parallel to the wind direction

$$\begin{aligned} \Delta\tilde{y} &= \Delta y_v \cos\theta - \Delta x_v sin\,\theta \\ \Delta\tilde{x} &= \Delta x_v \sin\theta + \Delta y_v \cos\theta \end{aligned} \tag{4.7}$$

Then the mean concentration above the buildings is

$$C \;\simeq\; \frac{Q}{2\pi U_0 \tilde{\sigma}_z \tilde{\sigma}_y} \exp\left(-\frac{\Delta\tilde{y}^2}{2\sigma_y^2}\right) \exp\left\{-\ \frac{(z-H)^2}{2\tilde{\sigma}_z^2}\right\} \tag{4.8}$$

For networks of street at right angles to each other with directions $\boldsymbol{n}_{s_i}$at angles φ_s and (90-φ_s) to the x-axis, the net displacements of the upper plume are in the direction $\boldsymbol{n}_s sgn(\boldsymbol{U} - \boldsymbol{n}_s)$. This expression is independent of the sign of $\boldsymbol{n}_s$. The deflection of the plume is in the direction of the component of $\boldsymbol{U}$ along to the street. Thus for rectangular network with equal lengths, there is a net positive displacement in the $+y$ direction if $0 < \theta < 45°$ and in the $+x$ direction if $90° > \theta > 45°$.

4.2.5 The Urban Dispersion Model, UDM

UDM was developed to provide a rapid means of estimating dispersion in urban areas, for ranges between about 10 m and 10 km (Hall et al., 2001). It is a puff or cloud dispersion model, in which a series of such releases is used to represent any desired emission profile (i.e. it is not confined to steady emission rates). It can treat multiple emissions from fixed or moving sources. Extensive experimental work was conducted to support model development, which was undertaken on behalf of the UK Ministry of Defence (e.g. Hall et al., 1988; Macdonald et al., 1997). Little has been published in the peer-reviewed open literature that describes the model in any detail.

The dispersion conditions of interest are divided into three sub-regimes, based around the relative dimensions of the dispersing pollutant cloud and the obstacles through which it is moving, and the density of obstacles on the ground. The three regimes are:

1. the open/interacting regime,
2. the urban regime, and
3. the long-range open regime,

as defined below.

The open/interacting regime

This regime is characterised by sparse obstacle densities (area coverage below 5%) and short range dispersion (generally within 1 km of a source), so that the pollutant cloud is small relative to the obstacles and interacts strongly with the obstacles as it travels through them. Important features of dispersion behaviour are the displacement of the cloud around obstacles and the entrainment of material into their near-wakes. The model splits the fraction of the pollutant cloud that is entrained from that which is not. The two fractions are then treated as separate clouds and the processes repeated as the pollutant moves downwind. Obstacles that are small relative to the local cloud scale play no role in this process. Groups of overlapping split clouds are recombined to form larger single clouds where appropriate, and this procedure improves computational times to a significant degree.

The urban regime

This is defined as a region of significant obstacle density (area coverage above 5%) and short range dispersion (again, generally within 1 km of a source). In contrast to the above, the cloud dimensions are larger than the obstacle dimensions, typically a few times larger. Hall et al. (2001) argue that this situation arises very early in the dispersion of an emission, being generally valid once the emitted cloud has travelled around the first line of obstacles that it encounters. Evidence from extensive wind tunnel and field studies is used to justify the use of a Gaussian plume model at this stage. The lateral and vertical dispersion parameters are related empirically to the travel fetch and the geometrical characteristics of the obstacles encountered by the pollutant.

The long-range open regime

This regime covers regions where the dimensions of the dispersing cloud have become large relative to the individual obstacles. Standard aerodynamic properties, the roughness length and atmospheric stability, are now used to characterise dispersion behaviour and conventional Gaussian plume modelling adopted, with dispersion rates again adjusted for the nature of the underlying roughness.

Model evaluation

Results of a number of model performance evaluations have been published in conference proceedings. Brook et al. (2002) and Griffiths et al. (2002) summarise studies based on the wind tunnel data of Macdonald (1997), the MUST field experiments (Biltoft, 2001) and the Urban 2000 study (Allwine, 2002). Standard measures were used to quantify performance, namely the fractional bias (FB), the normalised mean square error (NMSE) and the fractions within a factor of two (FA2) and three (FA3). Tabulated results from Brook et al. are repeated below.

i) Macdonald
Peak concentrations and plume widths were derived by Gaussian fits to both the wind tunnel and predicted dispersion results.

	Plume width			Peak concentration		
Area density	FA2	FB	NMSE	FA2	FB	NMSE
unobstructed	0.92	0.37	0.24	0.40	-0.45	0.55
6.25%	0.92	-0.11	0.15	0.88	0.42	0.89
16%	0.74	0.45	0.48	0.71	0.45	1.19
44%	1.00	0.18	0.04	0.86	-0.05	0.15

ii) MUST
Again, Gaussian fits were made to both the field and model results but only for experiments in which the dispersing plume was catered by the observations to an acceptable level.

	Plume width				Peak concentration			
	FA2	FA3	FB	NMSE	FA2	FA3	FB	NMSE
15 min	0.79	0.90	-0.04	0.37	0.50	0.60	0.02	1.85
1 min	0.93	0.96	-0.04	0.13	0.75	0.82	0.17	1.54

iii) Urban 2000
Comparisons were confined to the measurement arcs at 2 and 4 km from the emission.

	Plume width			Peak concentration			
	FA2	FB	NMSE	FA2	FA3	FB	NMSE
5 min, 2 km	1.00	0.07	0.08	0.55	0.89	0.65	0.83
5 min, 4 km	0.96	0.27	0.12	0.41	0.77	0.58	0.80
1 hr, 2km	1.00	0.33	0.16	0.61	0.94	0.21	0.47
1 hr, 4 km	0.96	0.47	0.31	0.59	0.76	0.27	0.52

Taken together, the results show no particular overall bias for plume spread and a bias to modest under-prediction (FB > 0) for concentration. The overall conclusion was that UDM performed well against these different classes of experiment that covered short and medium range field studies and wind tunnel simulations.

iv) Hanna et al, 2004

Hanna et al. (2004) compared the performance of UDM with a number of other models, including the version of UDM embedded in the US HPAC code, using data from the Urban 2000 and Salt Lake City experiments. UDM was seen to under-predict (FB = 0.41) whilst HPAC over-predicted (FB = -0.23). The general consensus was that UDM under-predicted arc-maximum concentrations, with performance better at short range than at long range. Performance varied with the concentration or dose measure targeted and the choice of meteorological data used to drive predictions. This was particularly apparent when the Urban 2000 experiments were considered because of the range of averaging times available and the large number of meteorological stations.

Discussion

UDM has many other features in addition to the basics described above but these are not discussed in the open literature and therefore this description is confined to the fundamental features of the model, how it represents dispersion in urban areas. It is important to note that UDM has been widely adopted for military use in both the UK and USA and much more widely evaluated than in the examples summarised above.

The two near-field regimes (i.e. within 1 km of an emission) of UDM do not include dispersion behaviour that is typical of street canyons and the interchanges between them at intersections. The second regime also assumes Gaussian behaviour once a dispersing plume has interacted with the first line of buildings downwind. The DAPPLE dispersion experiments in central London have shown that street canyon-like behaviour and interchanges at intersections are fundamental to dispersion behaviour in heavily built-up urban centres (of the kind typical of the UK) over ranges comparable with the extent of the first two UDM regimes. Further, within this range, dispersion patterns are far from Gaussian in form, which is one reason why the street network model is successful in these circumstances. UDM would appear not to have the capacity to represent the form of dispersion seen in the DAPPLE tracer experiments and these would therefore form a challenging further test of the model.

4.3 Street canyon models

As discussed in relation to the regime diagram in Section 4.1, when buildings are close enough together streets and street networks are formed. One approach to modelling dispersion in streets in urban areas is the street canyon model (e.g. Hertel et al ,1990). In this case it is assumed that the streets are sufficiently long that the pollutant concentration in a particular street are dominated by emissions in that street (there is no network). However the approach allows for both a recirculating region and a region where pollutant is advected downstream and therefore predicts gradients in concentration both across and vertically within the street. In order to model dispersion in urban areas the approach has been combined with a network of line sources unaffected by the buildings in ADMS-Urban (Carruthers et al 2000).

4.3.1 ADMS-Urban

ADMS-Urban is an operational air dispersion modelling tool, which has been developed to provide predictions at high spatial resolution of pollution concentrations for all sizes of study area from in-road and near road domains to citywide domains. It is the most widely used advanced dispersion model for urban and regional air quality worldwide. In the UK ADMS-Urban has been used by over 80 local authorities in conducting their Review and Assessment of air quality under the Local Air Quality Management program. Applications in the UK have included modelling Greater London, Greater Manchester and potential developments of Heathrow Airport. Around the world applications include use in Beijing, China for planning the large-scale development for the 2008 Olympic Games, in Budapest, Hungary for decision-making and air quality forecasting, in most areas of France including Marseille and Strasbourg for air quality assessment and in California, USA to model traffic sources.

ADMS-Urban is a development of the Atmospheric Dispersion Modelling System (ADMS) (CERC, 2010) for industrial sources. The main features of ADMS-Urban are:

- Modelling of the full range of source types encountered in urban areas including point, line, area, volume, road, aircraft and grid sources.

- An advanced Gaussian, “new generation” dispersion module using boundary layer depth and Monin-Obukhov length to characterise the boundary layer, and a skewed-Gaussian vertical concentration profile under convective meteorological conditions.
- Modelling of street canyons and vehicle–induced turbulence using a formulation based on the Danish Operational Street Pollution Model, OSPM (Hertel et al, 1990).
- Inclusion of a range of atmospheric NOx chemistry reaction schemes to suit different types of modelling study. The standard scheme is the seven reaction Generic Reaction Set (Venkatram et al, 1994) plus conversion of nitric oxide (NO) to nitrogen dioxide (NO_2) using molecular oxygen when concentrations of NO are very high and transformation of SO_2 to sulphate particles, which are added to the PM_{10} concentration. An alternative scheme is the carbon Bond Mechanism CB-IV scheme that has 95 reactions and uses input of 10 species of VOC.
- Inclusion of a large-scale Lagrangian-style trajectory model that calculates the change in background ozone across large urban areas. The local model is nested within the trajectory model.

Representation of the urban boundary layer

A meteorological pre-processor (The Met Office, 2010) calculates boundary layer parameters from a variety of input data, typically hourly averaged data including date and time, wind speed and direction, near surface temperature and cloud cover. The boundary layer depth is calculated taking into account the growth during the previous hours in the day and is assumed constant in the urban area. It can be different from the value at the meteorological data site, which may be a rural site, if different surface roughness, albedo and Priestley-Taylor parameters are specified.

ADMS-Urban uses boundary layer similarity profiles to parameterise the variation of turbulence with height within the boundary layer.

Modelling of non-canyon streets

Road sources are represented as line sources of finite length with no plume rise. The height of the line sources is equal to the actual height of the road above the 0m datum plus h_0, the initial mixing height, which is set to 1m by default but can be modified by the user. The vertical plume spread parameter, σ_{z_road}, is increased compared with that of a non-road line source, σ_z:

$$\sigma^2_{z\_road} = \sigma^2_z + h^2_0 \qquad (4.9)$$

To model the effect of vehicles on the lateral turbulence, an extra component, $\sigma_{y_{vehicle}}$, is included in the lateral plume spread parameter σ_y. The formulation of this extra component is as follows:

$$\sigma_{y_{vehicle}} = \sigma_{v_{vehicle}} t \left\{ 1 + \left(\frac{t}{t_d} \right)^2 \right\}^{-1/2} \qquad (4.10)$$

where

$$\sigma_{v_{vehicle}} = b\left(\frac{\sum_{i=1}^{n_v} N_i U_i A_i}{W}\right)^{1/2} \tag{4.11}$$

and the turbulence decay time, t_d, is given by

$$t_d = \left(\frac{W}{\tau}\right) \Big/ \sigma_{v_{vehicle}} \tag{4.12}$$

In the above definitions,

t = time to travel from source to this point(s)
b = constant (0.3) [from OSPM street canyon model]
τ = constant (0.3) [from OSPM street canyon model]
n_v = number of vehicle categories
N_i = number of vehicles per second for that category
U_i = speed of vehicles for that category (m/s)
A_i = effective area covered by vehicles in that category (m^2)
W = road width (m)

This extra component is not included in the lateral spread when modelling street canyons.

Modelling of street canyons

The canyon model is used for calculating the concentration at points which lie within roads lined with buildings with heights greater than 0.5m. Concentrations inside the road are a weighted sum of the non-canyon and canyon concentrations, tending to the non-canyon results in the limit as the canyon height is reduced to zero or as the ratio of canyon height to road width decreases to zero. Concentrations at points outside the canyon are identical with those that would be obtained if the road were not a canyon.

The component of the wind blowing perpendicular to the axis of the street generates a vortex in the recirculation region that may occupy part or all of the width of the canyon. In the recirculation region the canyon concentration has a component due to the recirculation, determined from a balance of inflow and outflow of pollutant. There is also a direct component to the canyon concentration, due to dispersion from the emissions within the recirculation region. At street level in the recirculation region, the wind direction is opposite to that at roof level so the direct concentration will be greatest at the upwind edge of the canyon. Outside the recirculation region there is a direct contribution to the canyon concentration due to dispersion from the emissions outside the recirculation region. The wind direction is that of the roof level wind and so the direct concentration will be greatest at the downwind edge of the canyon.

Vehicle induced turbulence is calculated in a similar manner to the non-canyon case, but in the canyon case it is an additional component of the vertical turbulent velocity σ_w, not the lateral turbulent velocity:

$$\sigma_{w_{vehicle}} = b\left(\frac{\sum_{i=1}^{n_v} N_i U_i A_i}{W}\right)^{1/2}$$

(4.13)

The canyon model ignores end effects such as junctions. It assumes a straight length of road with constant width, lined continuously on both sides by flat-roofed buildings of height of constant height. The traffic emissions are assumed to occupy the whole canyon width, so no account taken of the any pavement that may be present. There is no variation of canyon concentration with height within the canyon.

Complex Effects

In ADMS-Urban the effect of terrain and spatially varying surface roughness, its effect on mean wind and turbulence and hence on dispersion is modelled using the FLOWSTAR (Carruthers et al, 1988) model developed by CERC. The changes to flow and turbulence affect dispersion from all the source types except grid sources.

4.3.2 Validation of ADMS-Urban

ADMS-Urban is a development of the Atmospheric Dispersion Modelling System (ADMS), which is used throughout the UK by industry and the Environment Agency to model emissions from industrial sources. ADMS has been subject to extensive validation, both of individual components (e.g. point source, building effects and meteorological pre-processor) and of its overall performance. It has been part of many international inter-comparison and validation studies, many of which were carried out and reported to the series of Harmonisation Conferences (www.harmo.org) and some of which were reported to the AWMA (Carruthers et al, 2001; Carruthers et al, 2009; Carruthers et al, 2011).

ADMS-Urban has been extensively tested and validated against monitoring data for large urban areas in the UK, including Central London (Carruthers et al, 2003) and Birmingham, for which a large scale project was carried out on behalf of the DETR (now DEFRA). The Project for the Sustainable Development of Heathrow commenced with a Model Inter-comparison and comparison with monitored data of which ADMS-Airport (a version of ADMS-Urban) was a part (DfT, 2007). The main scenario report also included a comparison with monitored data (McHugh et al, 2007). In these complex urban modelling situations ADMS-Urban performs well, usually achieving, on average, a model accuracy of within 10% of annual average monitored values. Components of ADMS-Urban such as the street canyon model and chemistry scheme have been validated outside the model as part of their development.

Three tracer experiments under NERC's UK Urban Regeneration and the Environment (URGENT) Programme were conducted in Birmingham in 1999. These experiments and comparisons between the measurements and predictions of ADMS are described in Hunt et al 2003. The model generally showed good agreement with the measurements.

Validation of ADMS-Urban in a wide range of urban environments in cities across the world have shown its generally good performance. Where performance is less good relevant issues are usually specification of emissions, background concentrations and sometimes the formulation of the OSPM street canyon model (e.g. emissions are spread across the width of the canyon which may be wider than the road). The lack of channelling of pollutant from one street to the next does not appear to be a major factor for traffic emissions.

4.3.3 Summary

The dispersion model ADMS-Urban assumes that material dispersed from sources is advected in the direction of the mean wind unless a canyon model is invoked for a specific street canyon in which case an empirically based scheme allows for gradients in concentration in both vertical and transverse direction within the canyon. It is instructive to contrast this methodology with that of the operational street network model SIRANE to be described in the next section: SIRANE is based on the basic assumption that pollutant is well mixed within streets canyons and it is channelled from street to street by buildings. A comparison of the two modelling approaches and other models in Paris showed generally good performance of both models for modelling traffic pollution in street canyons (LCQSA 2009).

ADMS-Urban has the following current limitations, most of which it shares in common with SIRANE; these may of course be addressed in future versions of the model:

- Model runs using ADMS-Urban typically assumes a uniform wind field, however the model may take account of spatially varying terrain and surface roughness through the use of the FLOWSTAR model (Carruthers et al 1988).
- The model neglects the presence of the roughness sublayer above roof level, assuming Monin-Obukhov similarity down to roof level.
- The ADMS-Urban canyon model based on OSPM neglects the effect of the canyon on the flow when calculating the advection of pollutant into the canyon from neighbouring sources; i.e. each canyon only affects the flow and dispersion of material released in that canyon. It may be possible to improve this approach by using some aspects of the street network methodology.
- Street canyons are treated as infinitely long 2D canyons in estimating the advection velocity through them. Finite street length and width effects are not accounted for.
- Exceptionally tall buildings and enclosed spaces in buildings such as courtyards are not represented in the model.
- As with SIRANE as well as most other regulatory dispersion models, there are limitations in the treatment of light winds.

4.4 Street network models

4.4.1 Formulation of street network model

Figure 3.6 shows the DAPPLE measurement site viewed from the East; overlaid in pink is a visualization using smoke, obtained in a wind tunnel dispersion experiment. The experiment is lit to emphasise the dispersion within the streets. This image shows vividly how, within the neighbourhood scale, gas is carried along streets and branches at intersections, whilst being slowly vented into the boundary layer above the roofs where it is further dispersed.

If the buildings are close together, the streets between them relatively narrow, then it is natural to think of the flow as being within the streets between buildings (rather than as being the flow around buildings). As discussed in Section 3.1, if the above roof flow is at angle θ to the street axis, then the component of above-roof wind across the street, $U\cos\theta$, tends to force a recirculation across the street, whereas the component of the above-roof wind along the street, $U\sin\theta$, generates a mean flow along the street (Figure 4.6). Fluid paths are then helical and tend to mix scalar across the street as it is advected along the street. It is then possible to model the scalar as if it were well mixed within the street.

Clearly this approximation makes sense only if

1. The streets are narrow enough that there is a recirculation across the street, which requires $W/H < 3$.
2. The buildings are short enough that the recirculation mixes across the whole depth of the street.
3. The streets are long enough that the air makes more than one circuit around the recirculation before being advected the length of the street.

These are the conditions that delineate the area of parameter space where the street network approach is appropriate that is shown in Figure 4.1.

As discussed in Section 3.1.2, at intersections between streets the flow is more complex: Figure 4.7 shows schematics for T-junctions and for a four-way junction. In either case there is a flux of scalar into the intersection along the incoming streets and then a flux of material out of the intersection at the outgoing streets. Scalar thus passes through the branching street network, and thus is spread in the horizontal.

This is the basis of a family of models of dispersion in urban areas called *street network models*, because they emphasise the advection of scalar through the network of streets (rather than thinking of the flow and dispersion around groups of buildings). Street network models are a relatively new way of modelling dispersion, and so we describe them in some detail.

The common element in street network models is that they are based formally on taking a budget of scalar over the volume of each street and the volume of each intersection. These discrete volumes are then connected together as a network of boxes. There is a family of network models because there are different parameterisations of the processes to represent mixing and transport between the boxes. In some implementations (e.g. Soulhac, 2000, Hamlyn et al 2007), the concentration within the street is assumed well mixed and so constant. In another formulation (Belcher et al 2012) a simplified model for the variation of concentration along the street is then constructed, balancing advection along the street and detrainment into the boundary layer aloft (see below). At intersections the fluxes due to advection along streets into the intersection are balanced with detrainment into the boundary layer aloft and advection along streets out of the intersection. Different implementations of the network approach differ in detail in how they represent this flux balance. For example, Soulhac (2000) develops simplified models for flows through simple intersection geometries and then models the out-going fluxes of scalar as additive fractions of the in-going fluxes. In contrast, Belcher et al (2012) assume that the concentration within the intersection is well mixed so that advective fluxes out of the intersection are the product of this well mixed concentration, the advection speed and the cross-sectional area of the street. To date there has not been a systematic comparison of the different methods.

Mathematical formulation

Network models are based around the idea of separating the urban area into boxes, where, within the urban canopy itself, a box can be either the volume of a street (Figure 4.6), a street segment or the volume of an intersection (Figure 4.8). (The treatment of the above canopy air is discussed below.) We then aim to model the concentration averaged over each box. This is justified because the concentration within individual streets or intersections can be regarded as approximately well mixed (the case of long streets, when this assumption is no longer true, is considered later). Each box can be referenced using a pair of indices (i,j). Hence define the spatially averaged concentration of the (i,j)th box to be

$$\langle c \rangle_{i,j} = \frac{1}{V_{ij}} \int_{V_{ij}} c \, d^3x, \qquad (4.14)$$

where c is the concentration, and V_{ij} is the volume of the (i,j)th box. On taking the spatial average of the scalar conservation equation we obtain

$$\frac{d\langle c \rangle_{i,j}}{dt} + \frac{1}{V_{ij}} \int_{\partial V_{ij}} c\vec{u} \cdot d\vec{S} = \langle q \rangle_{i,j}, \qquad (4.15)$$

where $\vec{u}$ is the air flow velocity vector, ∂V_{ij} is the area of the facets bounding the volume V_{ij} of the (i,j)th box through which air flows and $\langle q \rangle_{ij}$ is the source within the (i,j)th box normalised on the box volume. Finally, on taking an ensemble average we obtain the budget equation for the ensemble-mean spatially-averaged, denoted here *mean*, concentration through the network of boxes:

$$\frac{dC_{i,j}}{dt} + \frac{1}{V_{ij}} \sum_{k=1}^{K} \Phi_{i,j}^{k} = Q_{i,j}. \qquad (4.16)$$

Here the mean concentration within the (i,j)th box is $C_{ij} = \langle \bar{c} \rangle_{ij}$, where the mean source is $Q_{ij} = \langle \bar{q} \rangle_{ij}$, and the Φ_{ij}^{k} is the flux of scalar through the *kth* facet of the (i,j)th box, which has 2 parts, an advective flux, F_{ij}^{k}, and a turbulent flux, f_{ij}^{k}:

$$\Phi_{i,j}^{k} = F_{i,j}^{k} + f_{i,j}^{k} = -\int_{\partial V_{i,j}} \left(\bar{c}\bar{\vec{u}} + \overline{c'\vec{u}'} \right) \cdot d\vec{S}. \qquad (4.17)$$

If the spatial and ensemble average is taken of the mass continuity equation then we obtain

$$\int_{\partial V_{ij}} \vec{u} \cdot d\vec{S} = 0 \qquad (4.18)$$

Whilst Equations (4.16) and (4.17), express conservation of scalar, they cannot be used as a prognostic model until the fluxes are parameterised.

Parameterisation of the scalar fluxes

The street network model becomes closed on parameterisation of the scalar fluxes. As shown in Equation (4.17) there are two types of flux.

Firstly consider the advective flux through the facet ∂V_{ij}^{k} into or out of the (i,j)th box. Since the concentration in the box upwind of ∂V_{ij}^{k} is approximately equal to the volume average concentration in the upwind box, say $C_{\alpha\beta}$, then

$$F_{i,j}^{k} = -\int_{\partial V_{ij}^{k}} \bar{c}\bar{\vec{u}} \cdot d\vec{S} \approx -C_{\alpha\beta} \int_{\partial V_{ij}^{k}} \bar{\vec{u}} \cdot d\vec{S} \equiv -C_{\alpha\beta} . U_{ij}^{k} \partial V_{ij}^{k} \qquad (4.19)$$

Thence the velocity, U_{ij}^k, used to advect the scalar is the fluid velocity averaged across the facet, and so the velocity decouples from the concentration. The last factor is geometric.

Secondly, consider the turbulent flux through the facet ∂V_{ij}^k into or out of the (i,j)th box. We model turbulent exchange so that it tends to equalise the concentrations within the boxes joined by the facet ∂V_{ij}^k, and hence is proportional to the difference in concentration between the boxes, $\left(C_{ij} - C_{\alpha\beta}\right)$ at a rate characterised by an exchange velocity, E_{ij}^k, so that

$$f_{i,j}^k = -\int_{\partial V_{ij}^k} \overline{c'\vec{u}'} \cdot d\vec{S} \equiv -\left(C_{ij} - C_{\alpha\beta}\right). E_{ij}^k \partial V_{ij}^k \qquad (4.20)$$

This is a finite-volume equivalent of a gradient diffusion approximation.

In the simplest implementation of the street network model, the horizontal advective fluxes within the urban canopy are assumed to be larger than the turbulent fluxes. These parameterisations are illustrated more explicitly for a simplified geometry next.

4.4.2 Flux budgets within a street network

The conservation of scalar mass, expressed in (4.16) and (4.17), together with the parameterisations of the advective and turbulent fluxes, given in (4.19) and (4.20), give a model for the time-mean spatially-averaged concentration, C_{ij}, in each of the streets and intersections. Once the geometry, the advection and exchange velocities, and the sources are specified, this model can be used to evaluate the C_{ij}.

This procedure is illustrated here for dispersion through a regular, aligned, and extensive, array of cubical buildings from a single steady source at an intersection. This simplified geometry illustrates many of the important processes. These conditions lead to a number of simplifications:

1. We compute the steady state concentration produced by steady sources, so that dC/dt in (4.16) can be ignored. The incorporation of unsteady sources presents no fundamental problem and is done in SIRANERISK (Section 4.4.5 below), although quantities then need to be interpreted as ensemble mean concentrations (turbulent fluctuations are of course not computed).
2. The source term appears only within the (0,0) intersection, in other intersections Equation (4.16) reduces to a balance between advective fluxes in the horizontal and a turbulent exchange in the vertical between the canopy and the boundary layer above.
3. Since the geometry is regular and the buildings cubical, the geometric term $\partial V_{ij}^k / V_{ij}$ is independent of *i* and *j*. The streets are of width W_x and W_y in the *x* and *y* directions respectively, and the buildings are of height *H*.
4. Since the array of cubes is extensive, the velocity field within the cube array is regular in the sense that the velocity field is identical in the identical locations within each repeating unit of the array. Hence the advection and exchange velocities are the same in each intersection. This means that the U_{ij}^k and E_{ij}^k are independent of *i* and *j*. Hence the

advection velocities are U_i and U_s, and the exchange velocities are E_i and E_s, in the intersections and streets respectively. Since the velocities U_i and U_s could be different, we need to accommodate vertical advection even in this simple geometry.

To our knowledge three different network models have been described in the literature (Soulhac 2000, Hamlyn et al 2007, Belcher et al 2012). They differ in how they represent the budgets of the different elements of the street network. In order to describe these differences consider separately the streets and the intersections.

Treatment of streets

Consider first the flux of mass within the street, which is given by

$$HWU_i = HWU_s + LWW_s \qquad (4.21)$$

where U_i is the advection speed out of the intersection into the entrance of the street, U_s is the advection speed out of the end of the street, and W_s is the average vertical velocity out of the top of the street. The length and width of the street are *L* and *W* respectively.

Consider now the flux of concentration. SIRANE (Soulhac, 2000; Soulhac et al. 2011) treats the streets as well mixed, as do Hamlyn et al (2007). The concentration within a street is governed by the balance between the flux in and the flux out (Equation 4.16), which is:

$$\Phi_i + \Phi_s + \Phi_v = 0 \qquad (4.22)$$

Here $\Phi_i = U_i C_i$ is the flux into the street coming from the upstream intersection (C_i is the concentration coming into the street out of the intersection), and $\Phi_s = -HWU_s C_s$ is the flux out of the end of the street, where C_s is the average concentration in the street. The vertical flux out of the street is driven by both turbulent transport and advection, and is given by $\Phi_v = -WL\{E_s(C_s - D_s) + W_s C_s\}$, where D_s is the concentration in the above the street and E_s and W_s respectively characterise turbulent detrainment and vertical advection out of the top of the street. Thus

$$C_s = \frac{HU_i}{HU_s + L(E_s + W_s)} C_i + \frac{LE_s}{HU_s + L(E_s + W_s)} D_s \equiv rC_i + eD_s \qquad (4.23)$$

which defines *r* and *e*. The last result shows that W_s can be absorbed into an effective detrainment velocity E_s.

Belcher et al. (2012) describe a refinement for the case when the streets in the network are longer. It is then important to take into account the detrainment of material along their length. The well-mixed assumption within the street would then be a poor approximation, as the concentration can decrease considerably along the street's length. In this case the variable concentration along the street can be modelled by dividing it into shorter segments and considering the flux from one segment to another. We note that this method could also be used for streets with varying cross sections. Figure 4.10 shows a long street divided up into segments of length ds. The flux balance through the middle segment can be written as $\Phi_1 = \Phi_2 + \Phi_3$, where Φ_1 is the flux gained from the previous segment, Φ_2 is the flux lost to the next segment and Φ_3 is the flux lost by detrainment into the air above. Denoting the concentration along the street as a continuous function *C(s)* of the distance *s* along the street and the external concentration just above the street as a corresponding function *D(s)*, this flux balance condition can be written as:

$$d(WHU_sC) = -E_sW(C - D)ds \qquad (4.24)$$

where *W* and *H* are the width and height of the street, U_s is the advection velocity along the street and E_s is the detrainment velocity out of the street top. When the street is uniform in width and height, this gives

$$\frac{dC(s)}{ds} = -\frac{[C(s) - D(s)]}{L_d} \qquad (4.25)$$

whereupon exchange of a passive scalar between the street and the air above occurs over the detrainment length scale given by $L_d = HU_s / E_s$.

In the near-field region where $C(s) >> D(s)$, this integrates to give an exponential decrease of the concentration with distance along the street:

$$C(s) = C(0)\exp(-s / L_d) \qquad (4.26)$$

The average concentration over the whole street is then

$$\bar{C}_s = C(0)\left[\frac{L_d / L_s}{1 - \exp(-L_s / L_d)}\right] \qquad (4.27)$$

Taking this modification into account, the analysis can then proceed as before.

The intersections

Conservation of mass within the intersection gives

$$\sum_{incoming\ streets} HWU_s = \sum_{outgoing\ streets} HWU_i + A_iW_i \qquad (4.28)$$

where A_i is the plan area of the intersection.

For a regular 4-street intersection as depicted in Figure 4.8, this can be written as

$$HW_yU_s + HW_xV_s = HW_yU_i + HW_xV_i + W_xW_yW_i \qquad (4.29)$$

where now the *U*'s and *V*'s characterise advection in the *x* and *y* directions, W_x and W_y are the width of the streets and *H* is the height of the buildings. Finally, W_i is the mean vertical velocity across the top of the intersection.

Consider now the flux of concentration. The fluxes (shown in Figure 4.8) sum to zero, so that

$$\Phi_{i,j}^1 + \Phi_{i,j}^2 + \Phi_{i,j}^3 + \Phi_{i,j}^4 + \Phi_{i,j}^5 + Q_{i,j} = 0 \qquad (4.30)$$

The fluxes at the intersections have also been parameterised differently in different network models. Perhaps the simplest approximation, used in Hamlyn et al (2007) and Belcher et al. (2012) is to assume that material becomes well mixed in the intersection. Hence, air with a material concentration equal to the concentration at the end of the incoming streets is advected into the intersection, it then becomes well mixed in the intersection, and well-mixed concentration is both detrained into the boundary layer above and advected along the out-going streets.

Note a small change in notation: the intersections are indexed using *i* and *j*, and the streets between them such that $i+\frac{1}{2}$ is the street between the *i*th and $(i+1)$th intersections. The four fluxes at an intersection, Φ_{ij}^{1} to Φ_{ij}^{4}, which are sketched in Figure 4.8 are written as advective fluxes and the vertical flux, Φ_{ij}^{5}, is a combination of advection and turbulent exchange:

$$\begin{gathered}
\Phi_{i,j}^{1} = HW_yU_sC_{i-\frac{1}{2},j} = HW_yU_s\left\{rC_{i-1,j} + eD_{i-\frac{1}{2},j}\right\} \\
\Phi_{i,j}^{2} = HW_xV_sC_{i,j-\frac{1}{2}} = HW_xV_s\left\{sC_{i,j-1} + fD_{i,j-\frac{1}{2}}\right\} \\
\Phi_{i,j}^{3} = -HW_yU_sC_{i,j} \qquad \Phi_{i,j}^{4} = -HW_xV_sC_{i,j} \\
\Phi_{i,j}^{5} = -W_xW_y\{E_i(C_{i,j} - D_{i,j}) + W_iC_{i,j}\}
\end{gathered} \tag{4.31}$$

where *r*, *s*, *e* and *f* are defined in accordance with (4.23). On substituting these flux relations into the flux balance at the (i,j)th intersection, Equation (4.16), and re-arranging, we obtain an expression for the concentration at that intersection, namely

$$C_{i,j} = \alpha\{pC_{i-1,j} + (1-p)C_{i,j-1}\} + \beta D_{i,j} + \gamma D_{i-1/2,j} + \delta D_{i,j-1/2} + S_{i,j}. \tag{4.32}$$

The concentration within the street network, $C_{i,j}$, is directly coupled to the concentration in the upwind intersections, $C_{i-1,j}$ and $C_{i,j-1}$, and to the concentration above, $D_{i,j}$, through the vertical exchange flux, Φ_{ij}^{5}. In Equation (4.32) we have

$$\alpha = \frac{rHW_yU_s + sHW_xV_s}{HW_yU_i + HW_xV_i + W_xW_y(E_i + W_i)} \tag{4.33}$$

is the fraction of scalar transmitted through a street and intersection, so that $(1-\alpha)$ is the fraction of scalar detrained into the boundary layer above as it passes through a street and intersection set;

$$p = \frac{rHW_yU_s}{rHW_yU_s + sHW_xV_s} \tag{4.34}$$

is the fraction of mass that is advected out of the intersection along the *x*-direction, so that $(1-p)$ is the fraction that is advected along the *y*-direction;

$$S_{i,j} = \frac{Q_{i,j}}{HW_yU_i + HW_xV_i + W_xW_y(E_i + W_i)} \tag{4.35}$$

is the source strength normalised on total exchange of mass;

$$\beta = \frac{W_xW_yE_i}{HW_yU_i + HW_xV_i + W_xW_y(E_i + W_i)} \tag{4.36}$$

is the turbulent flux of mass exchanged with the boundary layer above as a fraction of the total exchange of mass; and

$$\gamma = \frac{eHW_y U_s}{HW_y U_i + HW_x V_i + W_x W_y (E_i + W_i)}, \qquad \delta = \frac{fHW_x V_s}{HW_y U_i + HW_x V_i + W_x W_y (E_i + W_i)} \qquad (4.37)$$

are products of the turbulent mass flux detrained out of an incoming street as a fraction of the total mass flux out of that street, and the advective mass flux from that street into the intersection as a fraction of the total mass flux out of the intersection.

SIRANE (Soulhac 2000) models the flux balance at intersections differently. Rather than assuming that the intersection causes the concentration to become well mixed, the flux in the out-going streets is assumed to be a linear combination of the fluxes in the in-coming streets. The processes at an intersection are modelled by computing the partition of mass fluxes along each street assuming the flow through the intersection is two-dimensional. The flux $\Phi_{i,j}^3$ leaving the intersection is then computed by taking into account the contribution of all the incoming streets connected at the intersection. In addition a vertical flux from above the intersection into the out-going streets is computed in the same ratio as the horizontal fluxes. This can be expressed in our notation as

$$\begin{aligned} \Phi_{i,j}^3 &= P_{1\to 3}(\varphi_0)\Phi_{i,j}^1 + P_{2\to 3}(\varphi_0)\Phi_{i,j}^2 + P_{vert\to 3}\mathrm{D}_{i,j} \\ \Phi_{i,j}^4 &= P_{1\to 4}(\varphi_0)\Phi_{i,j}^1 + P_{2\to 4}(\varphi_0)\Phi_{i,j}^2 + P_{vert\to 4}\mathrm{D}_{i,j} \end{aligned} \qquad (4.38)$$

Here the $P_{m\to n}(\varphi_0)$ are average exchange coefficients between the *m* and *n* streets, which are calculated using a two-dimensional potential flow model, and the values depend on the mean wind direction φ_0. It is further assumed that turbulent mixing within the intersection is driven by fluctuations in φ_0. Also $P_{vert\to n}$ is the mass flux entering the intersection through vertical mean flow advection and flowing out along the n^{th} street. This specification is more complex than the simple well-mixed approximation used above.

Analytical solution within the neighbourhood region near to the source

Near to the source, the majority of the scalar is within the street network and so in this region the concentration of scalar in the boundary layer above the buildings is much smaller than the concentration within the building canopy, $C_{i,j} \gg D_{i,j}$. With this condition the $C_{i,j}$ is governed by

$$C_{i,j} = \alpha\{pC_{i-1,j} + (1-p)C_{i,j-1}\} + S_{i,j} \qquad (4.39)$$

When there is a single source, *S*, at (i,j) = (0,0), the solution to this equation is

$$C_{i,j} = \alpha^{i+j}\binom{i+j}{j}p^{i+j}(1-p)^j S \qquad (4.40)$$

Since *n = (i + j)* is the number of intersections the scalar has encountered since leaving the source, the factor α^{i+j} reflects the loss of material as the scalar passes through the street network. The factor $\binom{i+j}{j}p^{i+j}(1-p)^j S$ is a binomial distribution, which represents the lateral spread of the scalar through the street network. See Figure 4.9.

This solution demonstrates the basic processes at work in the street network:

- Lateral spreading as the scalar passes through intersections, so that the plume width depends on the geometry of the street network
- Dilution of the scalar within the urban canopy by detrainment into the boundary layer above

It is remarkable that the concentration is then determined by three parameters: the dimensionless source strength, the transmission parameter α and the direction parameter p.

Gaussian far-field limit

As the plume spreads through more intersections the binomial distribution which governs the horizontal distribution tends towards a Gaussian (central limit theorem). Provided that the transmission parameter is high (so that little mass is lost at each intersection) then this limit is reached before too much material is detrained into the boundary layer above (and the simplified model here remains valid). The concentration then becomes

$$C_{i,j} = \frac{\alpha^{i+j}}{\left(2\pi\sigma^2\right)^{\frac{1}{2}}} \exp\left(-\frac{\left(j-\mu\right)^2}{2\sigma^2}\right) S \,. \qquad (4.41)$$

The parameters of the Gaussian are given by

$$\mu = np, \qquad \sigma^2 = np\left(1-p\right), \qquad (4.42)$$

and we note that because of the α^{i+j} pre-factor in the solution these are not necessarily the centreline and width of the plume. Nevertheless this analysis provides theoretical explanation for measurements that show that the plume is close to Gaussian in a regular array of cubes (e.g. Davidson et al. 1995, 1996; Macdonald et al. 1997, 1998).

4.4.3 Treatment of above-canopy dispersion

The analytical solution derived in the previous sections for the in-canopy concentration is a good approximation in the near field, where the above-canopy concentration can be neglected. Further from the source, a fraction of material from above the canopy is entrained into the canopy, as encapsulated in the D_{ij} term in Equation (4.32). To take this into account the concentration above the canopy needs to be modelled explicitly. Simple analytical solutions for this coupled problem are not known, but the problem can be solved numerically. This is treated differently in the different network model formulations.

In the formulation of Belcher et al. (2012) the resulting model proceeds by representing the above-canopy region as a network of boxes similar to the in-canopy network. The advective fluxes through the facets of these boxes are then parameterised, as in the canopy, according to Equation (4.18). The turbulent fluxes are parameterised using the gradient diffusion assumption:

$$\overline{c'u_i'} = -K_i \nabla_i \bar{c} \qquad (4.43)$$

where K_i are eddy diffusivity coefficients. The values of K_i may differ for different components.

Here the horizontal fluxes include both advective and turbulent components (in order to capture arbitrary wind directions), in contrast to the situation within the canopy, where the horizontal turbulent fluxes can be neglected. This is supported by explicit calculation of the fluxes from the DNS reported in Section 3 (Goulart, 2012).

The time-evolving Equation (4.16) is now generalised by considering the evolution of $C_{ijk} = \langle \overline{c} \rangle_{ijk}$, where the third index represents the vertical level of boxes (with the lowest value of k representing the canopy layer). The resulting system of first order differential equations is easily discretised and integrated for an arbitrary distribution of sources in the network. In general the sources can be unsteady. For a steady, continuous release in one or more cells, the equations can be integrated to a steady state.

A similar numerical procedure was used by Hamlyn et al. (2007), with the difference that they modelled the above–canopy concentration as a superposition of Gaussian plume profiles resulting from effective sources at roof level due to the flux of material out of each cell at each timestep (similar to what is done in the SIRANE model, reviewed below). The paper itself gives little detail on the representation of the above-roof plumes. Hamlyn et al. (2007) demonstrated good agreement of their model predictions with experimental data over regular arrays of cubes from the water channel experiments of Hilderman and Chong (2007). However, they identified shortcomings in their representation of re-entrainment, which improved the predicted centreline concentration near to the source but was over-estimated further downstream.

4.4.4 Application of the numerical model to a regular geometry

The direct numerical simulations of flow and dispersion over a regular array of cubes performed by Branford et al. (2011) and described in Section 3.3 offers the possibility for evaluating the approach and assumptions of the network model as well as its predictive performance – the input parameters of the model can be calculated explicitly using the comprehensive DNS data, and detailed comparisons made with true spatial averages of the concentrations. Implicit here is the assumption that the array configuration simulated in the DNS lies within the domain of application of the network approach. Based on the regime diagram of Section 4.1 this may be just about true. Bearing in mind that this case study may not represent the 'best case scenario' (for example the 'streets' in the array are somewhat short), comparisons of the extended network model just described (which we will call the extended network model – E-NW) with the DNS data showed encouraging agreement, as reported by Goulart (2012). This comparison is briefly reviewed here.

Calculation of model input parameters using DNS data

To apply the network model to the DNS data, the relevant advection and turbulent velocities need to be computed as input parameters to the model. Under the assumptions discussed previously, the advection velocity through a facet can be approximated as the area-averaged velocity over the facet:

$$U = \frac{1}{\partial V} \int_{\partial V} \overline{\vec{u}} \cdot d\vec{S} \qquad (4.44)$$

The detrainment velocities E_i and E_s can be directly computed by dividing the flux out of the canopy by the roof area and the difference in concentration between the canopy and the air above:

$$E = \frac{f}{\partial V\left(C_{can} - C_{ext}\right)} \tag{4.45}$$

The horizontal eddy diffusivity coefficients K_x and K_y for the above-canopy treatment were chosen based on Pasquill (1962) and the vertical eddy diffusivity K_z was found as a best fit to the data. Constant values of K_x, K_y and K_z were used for 7 levels of boxes above the canopy.

The input parameters of the model and their values are then (in non-dimensional units):

Geometry: {$H = W = L = 1$}
Advection velocities in the canopy: {$U_i = V_i = 1.13$, $U_s = V_s = 1.18$}
Detrainment velocities: {$E_i = 0.5$, $E_s = 0.3$}
Advection velocities above the canopy: {$U_a = 3.54$, $V_a = 3.32$}
Eddy diffusivities above the canopy: {$K_x = 0.5$, $K_y = 0.5$, $K_z = 0.3$}

We note that, in the canopy, the advection velocities in the streets and intersections are unequal. Mass balance (Equations (4.21) and (4.29)) then implies that there is a non-zero vertical advection velocity $W_s = -0.05$ in the streets and $W_i = 0.1$ in the intersections. In other words there is a net mean flow into the streets and a net mean flow out of the intersections. Compared to the respective turbulent detrainment velocities, E_i and E_s, this is a relatively small contribution (around 20%) and, as discussed above, can be absorbed into the values of E_i and E_s.

By symmetry, $p = 0.5$. Using the equations given previously, we deduce that the detrainment parameter $\alpha = 0.65$, so that approximately 35% of material is lost by detrainment in passing through one street and one intersection. Since this value is less than 50% it seems reasonable to use the network approach for this geometry. In following this procedure we are giving the network model its best chance in a comparison with the simulations, which therefore provides insight into the success of the network approach.

Evaluation of the network model with the DNS data

The concentration data from the 45° DNS run was time-averaged, then volume-averaged for each 'street' and 'intersection' within the array to give an output similar to that of the network model. Profiles of the resulting averaged concentrations along the plume centreline and lateral profiles at two different downstream distances ($2\sqrt{2}H$ and $6\sqrt{2}H$) are shown in Figure 4.11, compared with the E-NW network model output.

The concentrations are normalised by the value in the cell in which the source is located, denoted by $C_{0,0}$. Also shown are corresponding profiles in the layer just above the array. The source location is at $x' = 4H$, $y' = 4H$, where x' and y' are measured along the street directions. Note that the centreline concentration just above the canopy rises to a maximum in the first intersection after the source before decreasing slowly with fetch, whereas the concentration within the canopy decreases much more rapidly. By the third intersection downstream of the source ($x' = 10H$) the concentration within and above the canopy are nearly equal. The fetch required to achieve this equilibrium of the concentration plume is a function of the detrainment parameter α. The larger the value of α (i.e. the weaker the detrainment out of the canopy), the longer it takes for the concentration above the canopy to equal that within and hence for the detrainment flux out of the canopy to equal the re-entrainment flux back into it.

Despite the simplicity of the network model, it is able to capture the magnitude of the mean concentration at different distances from the source fairly well, both within and above the canopy. The plume width is predicted reasonably well within the canopy, and better predicted above the canopy. Comparisons were also made with a version of the network model that excluded the above-canopy dispersion, but the concentration was under-predicted further downstream (not shown). Further analysis of the DNS data showed a large vertical flux of material out of the initial streets and intersections, and a downward contribution to the vertical flux a little further downstream (Goulart 2012). This indicates that re-entrainment of material from the above-canopy region becomes significant after a relatively short distance downstream for this geometry, which the full model including above-canopy dispersion is able to capture.

The largest discrepancy between the model and the DNS data is found in the cell just above the source location, where the model over-predicts the concentration. Similarly, in the near field at a distance of $2\sqrt{2}H$ from the source (see middle plots in Figure 4.11) the model over-predicts in the canopy and under-predicts above the canopy on either side of the profile peak. This is associated with the effect of the secondary wake sources in the near field as discussed in Section 3.3.2. The fact that the ‘buildings’ are cubes and the ‘streets’ are short in this case probably accentuates the effect of the secondary sources. In a geometry more typical of a street network, with longer streets and hence weaker end effects, one might therefore expect the network model to work better in the near field.

Whilst this comparison is made for the simplified geometry of a regular array of cubes, the good agreement shows the promise of the network approach. In more realistic geometries the success of the approach rests on our ability to estimate the parameters.

4.4.5 Operational models: SIRANE and SIRANERISK

The preceding sections have laid out the theoretical basis for the street network approach to urban dispersion and demonstrated that it shows promise for practical application. However, to be of operational value, suitable parameterisations must be specified for the relevant input parameters in a way that can be applied to realistic urban areas. SIRANE is the first and to date the only network model to accomplish this.

SIRANE: General approach

SIRANE is an operational urban air pollution model based on the street network approach (Soulhac 2000, Soulhac et al. 2011). It is used in several European cities for operational air quality modelling. SIRANE is designed to simulate the pollutant dispersion from point and line sources at the neighbourhood scale, ranging from hundreds of metres to a few kilometres.

In SIRANE the urban canopy region is represented as a street network, in which each street is treated as a box, and each intersection as a node without any dimensions. The region above the canopy is treated separately, with the flow being described by Monin-Obukhov similarity theory and the dispersion modelled using a Gaussian plume model (see below). In representing urban geometry, SIRANE currently neglects larger-scale topographical variations (L > 100 m) and represents the effect of small scale details (L < 1 m) as a uniform wall roughness. The building scale (L ~ 10 m) is explicitly represented in a simplified way. This is done by approximating the real urban geometry by cavities of rectangular cross section, of length L, width W and height H and wall roughness $z_{0,wall}$. If H/W < 1/3, the box is treated as an open space, which corresponds to what we characterise as a sparse array in Figure 4.1. If H/W > 1/3 it is modelled as a street canyon. SIRANE incorporates a numerical tool that automatically maps the street network and computes the relevant street dimensions from GIS data. Another numerical tool establishes a correspondence between the street network and the traffic network determined from traffic data sets in order to estimate source emissions.

The specific parameterisations of the flow and dispersion within and above the street network are described next. SIRANE also incorporates chemical reactions and physical parameterisations related to particle deposition (see Soulhac et al. 2011), but these aspects are not reviewed here.

Treatment of flow and dispersion within the urban canopy

As discussed before, the concentration within a street is governed by the flux balance in it, which may be written as:

$$Q_S + Q_I = HWU_{street}C_{street} + u_d WL\left(C_{ext} - C_{street}\right) \qquad (4.46)$$

where Q_s is the flux due to emissions in the street, Q_I is the flux coming from the upstream intersection, C_{street} is the average concentration in the street and C_{ext} is the concentration in the external flow above the street. The velocities U_{street} and u_d respectively characterise mean advection of pollutants along the street and turbulent detrainment out of the top of the street. So far this is merely re-expressing the flux balance condition given in Equation 4.16 with the assumptions of steady state and that the dispersion along the street is driven by advection and the vertical exchange by turbulent transport, both of which are assumed to be characterised by a velocity. There is no mean vertical exchange from a street in the SIRANE model. But the SIRANE model goes further by prescribing rules for computing these velocities, together with the upwind flux Q_I into the canyon from the previous intersection.

Soulhac et al. (2008) showed that, with the assumption of a long 2D street canyon, U_{street} can be written as the projection of the friction velocity along the street direction θ multiplied by a function of the canyon aspect ratio H/W and the ratio of the canyon wall roughness to canyon width $z_{0,wall}/W$:

$$U_{street} = u_* \cos(\theta) f\left(H / W, z_{o,wall} / W\right) \qquad (4.47)$$

Dobre et al. (2005) use measurements to demonstrate that this form works reasonably well within about a street width of an intersection. The function $f\left(H / W, z_{o,wall} / W\right)$ is complicated, but decreases monotonically with both H/W and $z_{0,wall}/W$, so that the advection velocity U_{street} in the street decreases as the street is made narrower and rougher (Soulhac 2000).

The transfer velocity u_d is taken to be proportional to the standard deviation of vertical velocity at roof level σ_w and estimated as (Soulhac 2000):

$$u_d = \frac{\sigma_w}{\sqrt{2\pi}} \tag{4.48}$$

As discussed before, to estimate the flux coming from the upstream intersection Q_I, the exchange processes at an intersection are modelled by computing the partition of mass fluxes along each street assuming the flow through the intersection is two-dimensional. The flux $Q_{I,j}$ leaving the intersection from a street j is then given by taking into account the contribution of all streets connected at the intersection. This can be expressed as:

$$Q_{I,j} = \sum_i \hat{P}_{i,j}(\varphi_0)\, C_{street,i} + P_{ext \to j}\, C_{I,ext} \tag{4.49}$$

where $C_{street,i}$ is the concentration in street i, $C_{I,ext}$ is the external concentration at the intersection I.

$P_{ext \to j}$ is the mass flux entering the intersection vertically that goes into the street j. It is computed by assuming that the total vertical flux P_{vert} into or out of the intersection is partitioned among the outgoing streets in proportion to the flow rate in each of those streets. P_{vert} is itself calculated, by applying mass conservation, as the imbalance between the horizontal mass fluxes entering and leaving the intersection from the streets. Note that these horizontal mass fluxes are computed independently using the velocity U_{street} in Equation 4.47, and therefore they do not necessarily balance.

$\hat{P}_{i,j}(\varphi_0)$ is an average exchange coefficient between streets i and j that depends on the mean wind direction φ_0. It is assumed that the turbulent mixing within an intersection depends mainly on fluctuations in the wind direction and that these are small, so that they can be modelled by a Gaussian distribution.

Flow and dispersion above the urban canopy

The flow above the canopy is modelled using Monin-Obukhov theory. This gives familiar expressions for the mean wind and temperature profiles. The required aerodynamic parameters for roughness length and displacement height are estimated in terms of the urban geometry using empirical methods in the literature (e.g. Macdonald et al. 1998) – see Section 2.

The dispersion above the canopy is treated by representing the flux of material out of the canopy as a series of point sources at roof level. The concentration distribution from each of these point sources is then modelled by a Gaussian plume model, incorporating plume reflections at the top of the boundary layer and at roof level using the image source technique. The standard deviations σ_y and σ_z in the Gaussian plume equation that characterise the plume spread with fetch depend on the atmospheric stability. In SIRANE they are parameterised using two different methods. The first is based on the Pasquill-Gifford stability classes (Briggs 1973, Pasquill and Smith 1983). The second is a continuous parameterisation based on Monin-Obukhov similarity (Weil 1985, Venkatram 1992, CERC 2001).

SIRANERISK

SIRANERISK is the unsteady version of the SIRANE model (Cierco et al. 2010, Lamaison et al. 2011a) designed to simulate the temporal evolution of concentration due to localised releases in a street network. The model has been formulated to address the needs of emergency response following accidental or deliberate releases of toxic material. It is able to predict the travel time of pollutant from sources to specified receptors, the mean and standard deviation of concentration, and the plume spread as a function of time.

In SIRANERISK the dispersion within the canopy is modelled using a street network approach just as in SIRANE, but the above-canopy dispersion is modelled by the advection and dispersion of Gaussian puffs. The growth of the standard deviations of the concentration distribution is based on Monin-Obukhov similarity theory. Version 2.0 of SIRANERISK also takes into account the puff deformation due to the mean shear on the evolution of the longitudinal standard deviation. Examples of the application of SIRANERISK are given in the next section.

4.4.6 Application of SIRANE to real geometries

Validation of SIRANE: application to air quality measurementsin Lyon

SIRANE has been validated by applying it to a case study in Lyon, France (Soulhac et al. 2012). Field data were collected during a 15-day field campaign in Lyon in July 2001. The validation took account of building geometry, meteorology, emission rates and background pollution. Comparisons were made of the temporal evolution at the locations of three monitoring stations, and of the spatial distribution over an extended area using 60 passive diffusion tubes.

The computation of the street network was done from detailed GIS building information. Only traffic sources of pollutant were considered (there were no heavy industry in the selected area). Emissions were estimated by merging data from direct traffic counts with traffic simulations (using the traffic model DAVISUM) and using empirical relations between vehicle speed and emissions according to the COPERT methodology. Meteorological data was obtained at a neighbouring rural site and corrected for the urban effects of increased roughness and displacement height (which were estimated using the morphological method of Macdonald et al. 1998) using SIRANE's meteorological pre-processor. Background pollution concentrations were accounted for by using data from monitoring stations outside of the studied area.

The validation therefore applied not uniquely to SIRANE, but to the 'modelling chain' of SIRANE coupled with the traffic model (DAVISUM) and a methodology for estimating the pollutant emissions (COPERT). The background concentrations were found to represent a significant fraction of the local pollution levels. The spatial distribution of NO_2 and benzene (15 day averages) were simulated quite well by SIRANE, as were the temporal evolution of NO_x, NO_2 and O_3. But NO was not well simulated. This could be due to the chemical modelling in SIRANE. The overall model performance was characterised as 'good' as assessed by the Chang and Hanna (2004) criteria based on four statistical indices: fractional bias (FB), normalised mean square error (NMSE), the correlation coefficient (R) and the fraction of predictions within a factor of two of the observations (FAC2)

Evaluation of SIRANE with wind-tunnel data on the DAPPLE geometry

In order to perform more detailed and controlled evaluations of SIRANE's performance than is possible in the field, comparisons were made with wind tunnel experiments performed at the University of Surrey (Carpentieri et al., 2012). The controlled setup in the wind tunnel with precisely determined flow and emission parameters enabled a more direct evaluation of SIRANE without the additional uncertainties of traffic, meteorology or chemistry modelling.

As described in Section 3.2, the wind tunnel experiments were done as part of the DAPPLE project (http://www.dapple.org.uk) on a 1:200 scale model of the DAPPLE site in central London. Comparisons were made between the wind tunnel experiments and SIRANE runs for nine wind directions between 110° and 31.4° with fixed source and receptor locations. Correlation between the experimental and numerical results was better when the wind direction was nearly diagonal to the streets; there was a poorer correlation for wind directions nearly parallel (within 20°) to the streets. The wind tunnel tests showed that changes in the source and receptor locations within given streets result in significant scatter in the measured concentration. SIRANE cannot reproduce this dependence, as it is based on a box model in which the source or receptor location within any particular box is not represented. The model performance was also tested in response to changes in the street network geometry by blocking selected streets and removing the corresponding links in SIRANE. Overall model performance was generally 'good', as measured using the Chang and Hanna (2004) criteria. However, the model showed a general tendency to under-estimate the concentration close to the source and over-estimate further from the source. Concentrations due to ground level sources were better predicted than roof level sources.

Example of application of SIRANERISK to accidental releases: Lyon and Paris

Lamaison et al. (2011a, 2011b) reported validation studies and examples of application of SIRANERISK-2.0 to areas of Paris and Lyon. The model was compared to a Lagrangian dispersion model and to wind tunnel experiments for unsteady releases over rough ground and idealised urban-like setups consisting of regular blocks, giving good agreement overall for both pollutant cloud arrival and transit times (Lamaison et al. 2011a).

In an application study (Lamaison et al. 2011b), a static instantaneous release of phosphine gas was simulated in front of a busy railway station in Lyon with a daily passenger traffic of 140,000 – the city of Lyon itself has 480000 inhabitants. The area modelled by the street network comprised 3200 streets and 1900 intersections. A simulated release of 10 kg of phosphine gas was made in the first time step, and the model was run thereafter for 600 time steps of 5 s, a total simulated duration of 50 minutes. The computation time was about 2 minutes on a standard desktop computer. The wind speed was constant but the wind direction varied through a 30° angle during the calculation. The model simulations showed that far downstream from the source, material reaches a street first by re-entrainment from above the roof, due to material detrained into the air aloft from upstream being swept downstream faster than material can travel through the canopy. The maximum concentration then occurs when material arrives from upstream streets through the canopy. Not surprisingly, highest concentrations were recorded near the source location, but non-zero concentrations persisted for much longer further from the source – for example persisting for 40 minutes at the city Hall of Lyon 2.1 km away even though the release was instantaneous.

Lamaison et al. (2011b) describes a second case study, in which a moving source was simulated through an area of East Paris. The domain considered was a 4km by 4km square and consisted of 3000 buildings. Pollutant release was simulated from a moving vehicle for 5 minutes. The wind speed and direction were both kept constant.

The SIRANERISK network was configured with 2100 streets and 1300 intersections. The time step was 5 s and total physical simulation time 50 minutes (600 time steps) and took 6 minutes to run. The simulation showed that even after the source emission had stopped the concentration in the streets decreased very slowly – it took 20 minutes to empty the first half of downstream streets after emission had stopped.

4.4.7 Summary: Successes and challenges of the street network approach

The validation studies reported in the literature demonstrate the good performance of the street network approach for urban geometries where the basic assumptions are valid, including its implementation in the operational SIRANE model, for predicting mean concentration distributions over neighbourhood scales. Validation studies of the unsteady SIRANERISK model also show the promise of the network-puff method for emergency response applications, capturing the time evolution of the dispersion pattern from localised unsteady releases. However, further evaluation of the model would be desirable, especially in application to more realistic urban geometries. For example, a wind tunnel evaluation study of SIRANERISK using the DAPPLE scale model (as was done for SIRANE) would be useful.

Despite these successes and promises, street network models do have their limitations. Because they are box models, they are subject to the following fundamental limitations:

- The source or receptor location within a box is not represented and therefore has no effect according to such models. However, wind-tunnel results of Carpentieri et al. (2012) show that different source and receptor locations within given streets result in significant scatter in the measured concentrations. This limitation could be partially overcome by dividing longer streets into shorter segments, or by modelling the variation of concentration along a street explicitly, for example using an exponential decay law (see the above section on the treatment of long streets within the network model).
- The concentration within each box is assumed to be well mixed, and concentration variations within the box are not resolved. This could be an issue with long streets, as described earlier. It could be overcome by incorporating a street-canyon model into the model.
- Plume turning with height within the canopy (see Section 3.3) cannot be simulated because the model assumes a well-mixed vertical distribution.
- The well-mixed assumption would also break down when the height of the buildings is large compared to the street width, corresponding to the ‘tall canopy’ regime described in Figure 4.1, see Section 4.6 for further discussion of this regime.

In addition, the operational SIRANE/SIRANERISK models have the following current limitations, some of which will be addressed in future versions of the model:

- SIRANE assumes a uniform wind field over the horizontal extent of the urban area; this limits the size of the domain of application of the model. It could be remedied by incorporating a model accounting for neighbourhood-scale and city-scale variation of the wind. For example, the street network dispersion model could be coupled with an urban canopy representation of the effects of the buildings on the spatially averaged flow (for example using the methods of Coceal & Belcher, 2004).
- SIRANE neglects the presence of the roughness sublayer above roof level, assuming Monin-Obukhov similarity down to roof level.

- The approach fails when the density of buildings is low, as when streets are partially bordered by buildings or there are large empty squares. This is because there is greater interaction between the canopy and the above flow, which is not built into the network approach. Also, the well-mixed approximation no longer holds for the space between buildings.
- Exceptionally tall buildings within an area of shorter buildings are not treated correctly. Further research is needed to understand and model the effects of tall buildings on flow and dispersion.
- The flow in intersections is treated as planar in the model, based on the results of RANS simulations (Soulhac 2000). However, it is now known from wind-tunnel experiments (Carpentieri and Robins 2009) as well as DNS (Goulart 2012) that the flow in intersections in highly three-dimensional, even for regular arrays of cubes. This could substantially modify the flux balance at the intersections.
- Streets are treated as infinitely long 2D canyons in estimating the advection velocity through them. Finite street length effects are not accounted for. Similarly, the variation of wind speed across the street width is not considered. Canyon porosity could be included to account for lateral dispersion through real canyons.
- Enclosed spaces in buildings such as courtyards are not represented in the models.
- SIRANE performs less well for roof level sources or when the wind is almost parallel to the street.
- Light winds are not treated, although this is an issue with most regulatory dispersion models.

4.5 Building-resolving models

Street network models are formulated in terms of the averaged flow and concentration within streets and intersections. Another approach aims to resolve the flow and concentration around each building. In this class of models, the general approach is to first compute the velocity field around the buildings, which is then used in a plume dispersion model. Models differ in how they accomplish these two tasks. The wind field can be computed quickly using parameterised mass-consistent methods (as in QUIC-URB and Micro SWIFT, described below) or fast CFD (QUIC-CFD). More recently, methodologies have been developed for pre-computing and storing detailed velocity fields at high resolution using full CFD (with RANS or LES) for a range of parameters (e.g. wind directions and stabilities). Two examples of such approaches are also briefly reviewed here (Flow Air 3D and FAST3D-CT). In most of these models dispersion is handled using a stochastic Lagrangian particle model (e.g. QUIC-PLUME, Micro SPRAY and SLAM, see below).

Lagrangian particle models model dispersion by computing the motion of a large number of particles in an unsteady wind field composed of a mean wind velocity and a turbulent wind velocity component. With this decomposition, the instantaneous position $\vec{x}$ of a particle may be written as:

$$\vec{x} = \vec{x}_p + \vec{U}\Delta t + \tfrac{1}{2}\left(\overrightarrow{u'}_p + \overrightarrow{u'}\right)\Delta t, \qquad (4.50)$$

where $\vec{U}$ is the mean velocity, $\overrightarrow{u'}$ is the fluctuating velocity, Δt is the time step and the superscript *p* denotes the previous time step. The fluctuating velocity may be written as

$$\overrightarrow{u'} = \overrightarrow{u'}_p + d\vec{u}, \qquad (4.51)$$

where the equations for $d\vec{u}$ are generally complicated but simplify considerably if the mean vertical velocity is zero and mean horizontal winds are uniform. A further simplification occurs by considering turbulent diffusion only in the lateral and vertical dimensions, in which case dv and dw reduce to a form involving three terms: a *fading memory* term, a *drift correction* term and a *stochastic acceleration* term (Rodean, 1996).

4.5.1 QUIC – A building resolving parameterized model

The Quick Urban & Industrial Complex (QUIC) model is a fast response dispersion modelling system for predicting chemical, biological and radiological (CBR) releases in urban areas. It is comprised essentially of a 3D wind field model called QUIC-URB and a Lagrangian dispersion model called QUIC-Plume. QUIC also includes a graphical user interface called QUIC-GUI to aid with setting up and running problems and visualising results. There are further tools for evaluating mean pressure fields around buildings (QUIC-PRESSURE), estimating concentrations inside of buildings (QUIC-INDOOR) and mapping population exposures (QUIC-POP).

QUIC was designed with emergency response applications in mind and can therefore run in seconds or minutes on a laptop. More recently, an alternative wind flow computational fluid dynamics solver based on the RANS methodology was devised (QUIC-CFD), although it is significantly slower than QUIC-URB. QUIC includes a user-friendly graphical user interface, detailed documentation and is available, by permission, for research purposes.
(see http://www.lanl.gov/projects/quic/).

QUIC-URB

QUIC-URB computes flow fields around buildings using empirical algorithms and mass conservation, based on the original work of Rockle (1990) on low rise buildings. Later improvements include modifications to account for isolated buildings, wide and tall buildings, and algorithms for dealing with overlapping flow zones of neighbouring buildings as described in Brown et al. (2009).

In QUIC-URB, empirical parameterisations of the flow field are based on assuming different regions around buildings, such as upwind rotor, downwind cavity and wake, street canyon vortex and rooftop vortex. An initial flow field (u_0, v_0, w_0) is prescribed based on the incident flow and the various flow regions associated with the given building geometries, using empirical relationships in terms of the building height, width and length and the spacing between buildings. Mass consistency is then imposed to obtain the final flow field (u, v, w). The adequacy of the final mass-consistent flow field is dependent upon a good quality initial estimate of the flow field.

QUIC-CFD

The QUIC-CFD solver (Gowardhan et al. 2011) is an alternative flow model based on a Reynolds-Averaged Navier-Stokes (RANS) approach using a zero-equation (algebraic) mixing-length based turbulence model. The use of a zero-equation model results in a reduction in computational time of 2-8 times relative to more traditional RANS models, but it requires algebraic specification of length scales throughout the solution domain. Although still significantly slower than QUIC-URB, QUIC-CFD produces more realistic wind fields. Gowardhan and Brown (2012) compared the performance of the two models using Joint Urban 2003 data (see below).

QUIC-PLUME

QUIC-PLUME is a Lagrangian random-walk dispersion model that uses the mean field computed by the QUIC wind solver together with turbulence modelled using the Langevin random-walk equations. It includes more terms than the usual three-term random-walk equation for boundary layer flow problems, to take into account the inhomogeneity of the flow around buildings. Normal and shear stresses and turbulent dissipation are computed, based on similarity theory, gradient transport and a non-local mixing formulation. QUIC-PLUME can simulate the dispersion of gases and aerosols, including gravitational settling, deposition and decay. The model has been tested for both idealised and realistic case studies.

Validation studies using QUIC

Several validation studies for QUIC exist. We report briefly on two: the first one examined the performance of QUIC-PLUME against a wind-tunnel test case; the second one looked at the performance of different wind models (QUIC-URB and QUIC-CFD) on predicted dispersion patterns using Joint Urban 2003 data.

Williams et al. (2004) describe the results of a simulation of a wind-tunnel experiment with a release made a short distance downwind of a tall building. The release was made 3 m above ground and 6 m behind the back wall of a building of dimensions 12m by 12m and height 36 m. Measured concentrations on the back wall of the building and for a plane along the building axis were compared with the simulations. The results showed that the model could reproduce the measured concentrations satisfactorily if a non-local mixing parameterisation was included. This was done by representing horizontal mixing induced by vertically oriented eddies on the sides of the building and vertically-induced mixing by horizontally oriented eddies. A purely local representation of mixing did not give satisfactory results, overestimating the concentration at the back side of the building by an order of magnitude due to excessive trapping of particles and compact predicted concentration fields.

A comparative study of the performance of QUIC-URB and QUIC-CFD against Joint Urban 2003 data was recently reported by Gowardhan and Brown (2012). The Joint Urban 2003 field experiment was performed in Oklahoma City and consisted of a large number of tracer releases at three different locations in the central business district (CBD) and a network of concentration sensors and meteorological instruments. The modelling domain was an area of 1.2km by 1.2km and covered most of the CBD. The simulation domain had 2 million grid cells. The same grid and inflow conditions were used in simulations employing QUIC-URB and QUIC-CFD. The wind field from these models were then used to drive the QUIC-PLUME dispersion model. The simulations were performed on a single-processor PC and the wind field was generated in 1 minute using QUIC-URB and in 20 minutes using QUIC-CFD. Not surprisingly, the wind field produced by QUIC-CFD compared better with the field data and captured more of the flow features observed in urban areas. The concentration fields in the two simulations were fairly similar further than a block or two from the source. However, close to the source the plume direction and dispersion pattern were better predicted by QUIC-CFD.

4.5.2 MSS – Micro Swift Spray, a fast CFD Lagrangian dispersion model

Micro Swift Spray, MSS, (Moussafir et al., 2004) is another fast building-resolving dispersion modelling system similar in many ways to QUIC. It consists of two modules: Micro SWIFT and Micro SPRAY.

The methodology of Micro SWIFT can be summarised in the following steps:

- A first guess of the mean flow is computed using interpolation of available meteorological data.
- This initial guess is then modified to take into account the presence of buildings by implementing analytical corrections based on prescribing different zones around the buildings.
- Conditions of mass consistency and impermeability on solid surfaces are then applied to compute the final flow field in an iterative manner.
- Turbulent diffusion coefficients and turbulent kinetic energy dissipation rate are then derived from the flow field and used as input into a Lagrangian particle dispersion model, Micro SPRAY.

Micro SPRAY is a Lagrangian dispersion model that simulates the motion of a large number of particles in a velocity field consisting of the mean field derived by Micro SPRAY plus a stochastic velocity component modelled according to the method of Thomson (1987). The coefficients of the model are computed using inputs of the turbulent dispersion coefficients and dissipation rate given by Micro SWIFT.

PMSS – the parallel version of MSS

A parallel version of MSS (PMSS) also exists (Oldrini et al., 2011), and a number of application and validation studies have been published (Tiranelli et al. 2007, Moussafir et al. 2010, Duchenne et al. 2011). PMSS consists of two components: PNSWIFT and PSPRAY, which are the parallelized versions of Micro SWIFT and Micro SPRAY respectively. Moussafir et al. (2010) point out that, although the CPU cost of MSS is typically two orders of magnitude lower than that of full-blown CFD, it is still not negligible compared to a Gaussian plume or puff model. The difference is particularly felt for long-term impact simulations. Hence, the parallelization of MSS potentially opens the way to making the use of 3D Lagrangian modelling more widespread. Moussafir et al. (2010) report on examples of two such long-term impact simulations in industrial areas. One example involved the health impact evaluation due to the dispersion of several volatile organic compounds (VOCs) from a car factory in Asia. The plant was surrounded by an urban development including several residential tall towers. The need to include this urban development as well as to resolve the plant sufficiently necessitated a nested grid approach. MSS was used in an inner grid of area 2km x 2km with a resolution of 10m. On the outer grid, of area 6km x 9km and at 20m resolution, clusters of particles were modelled as puffs as they emerged from the inner domain. The CPU cost of the simulation was about a week of wall clock time on a 10-processor machine per year simulated.

Simulation using PMSS over the whole of Paris

Duchenne et al. (2011) describe the application of PMSS to a simulation domain covering the whole of the city of Paris, represented in a rectangular 12km x 10.5 km domain. A horizontal resolution of 3m was used for PNSWIFT, giving a horizontal grid consisting of 4001 x 3501 nodes. The domain height was 1km, resolved with a stretched logarithmic grid over 27 levels. The total number of nodes in PNSWIFT was thus nearly 380 million. For PSPRAY the same horizontal grid was used but a uniform vertical grid with 20 nodes was used up to a height of 300m. The flow in PNSWIFT was driven based on forecasts provided for a real meteorological situation. Twelve time frames with a 1 hour resolution were computed with PNSWIFT. Two hypothetical release scenarios, both near the ground, were simulated with PSPRAY:

(i) a 20-minute release of a gaseous chemical substance in place de l'Etoile in Paris (ii) a 2-minute release of a radioactive aerosol from Meynadier Street in the North East of Paris. Concentration and deposition fields were recorded every minute over a two-hour period. Calculations were done for various number of cores. It was found that PNSWIFT scaled well up to a few hundred cores. The time to compute twelve 3D flow field time frames decreased from one day using 10 cores to less than half an hour using 701 cores. For PSPRAY, the computing time decreased steadily from about 14 hours using 49 cores to just under 2 hours using 201 cores, There was no significant speedup beyond 201 cores. This study demonstrates the feasibility of operational forecasting at the scale of entire cities using these methods.

4.5.3 Methods using velocity fields pre-computed using CFD

Full building-resolving CFD provides the most accurate method to model dispersion in urban areas. But although of practical value for planning and other non-emergency applications, it is too expensive for operational emergency response applications. This is where the simpler parameterised schemes in QUIC and SWIFT have an advantage, albeit at the expense of reduced accuracy. RANS-based CFD solutions typically take a few hours to run on common desktop computers whereas fully-converged LES solutions take days even on multi-processor supercomputers. The need for both accuracy and speed has prompted the development of methodologies for pre-computing and storing velocity fields for simulated flows over specific urban areas. The stored velocity fields can then be used for modelling dispersion for different meteorological conditions and source locations.

The obvious attraction of this approach is that velocity fields can be computed accurately at high spatial resolution. But the approach is subject to a number of limitations, such as (i) the parameter space that must be spanned to cover every realistic scenario is impossibly large (ii) the need to validate the flow field computations themselves (especially if RANS methods are used) (iii) the computing time to construct such databases and the volume of data that must be stored are considerable (iv) hence, the method is not scalable to very large urban areas. Nevertheless, the method has been successfully applied in studies over limited domains. Two examples of this approach are outlined below.

Flow Air 3D – SLAM

In this approach (Vendel et al., 2010) it is assumed that the wind field depends only on the geometrical parameters characterising the urban area (which are fixed for a given urban area) and on three meteorological parameters: the friction velocity u_*, the wind direction φ and the Monin-Obukhov length L_{MO}. Dimensional analysis then shows that the velocity field may be written in the form:

$$u = u_* f_u \left(1/L_{MO}, \varphi\right) \tag{4.52}$$

where f_u is a normalised velocity field and the reciprocal of the Monin-Obhukhov length is used to avoid a singularity in the neutral case. Similar scaling relationships can be derived for the turbulent kinetic energy and dissipation rate:

$$k = u_*^2 f_k \left(1/L_{MO}, \varphi\right) \tag{4.53}$$

$$\varepsilon = \frac{u_*^3}{\kappa z} f_\varepsilon \left(L_{MO}, \varphi\right) \tag{4.54}$$

The 3D fields f_u, f_k and f_ε are then computed using CFD with a k-ε RANS turbulence model (Vendel et al., 2010).

The Flow Air 3D methodology consists of pre-computing and storing the 3D wind field database for a number of different sets of parameter values of φ and $1/L_{MO}$. In an operational application, meteorological observations are used to derive values for u_*, φ and $1/L_{MO}$. A bi-linear interpolation method is then applied to the wind field in the database for these parameter values. Using the interpolated wind field a Lagrangian dispersion model (SLAM) is then used to compute the concentration field for given source characteristics. The main source of error in this methodology (apart from the use of a RANS closure) is the interpolation error. This error may be reduced by using a larger number of discrete values of φ and $1/L_{MO}$ in the database production. Vendel at al. (2010) show that using 18 wind directions (every 20°) and 7 values of $1/L_{MO}$ (a total of 126 CFD simulations) gives an interpolation error of a few percent, which is deemed reasonable for an operational model.

The Safety Lagrangian Atmospheric Model (SLAM) is a stochastic dispersion model based on computing the motion of individual particles in a velocity field composed of the interpolated mean velocity field obtained from the database plus a fluctuating velocity field determined by a stochastic differential equation (Thomson, 1987). The latter includes coefficients that depend on the standard deviation of velocity fluctuations, σ_u, and the Lagrangian time, T_L. These are obtained in terms of the turbulent kinetic energy and turbulent dissipation rate from the velocity fields pre-computed using the CFD. For example, if using the $k-\varepsilon$ turbulence model, σ_u, and $T_L$are given by

$$\sigma_u = \sqrt{\frac{2}{3}k}\,, \qquad T_L = \frac{2\sigma_u^2}{C_0\varepsilon}, \tag{4.55}$$

where the coefficient $C_{0=}4$.

Vendel et al. (2011) report validation studies comparing results obtained using the Flow Air 3D – SLAM method with wind tunnel and full CFD (performed with a k-ε RANS model using FLUENT) data for an industrial site. A domain size of 1km x 1km x 0.3 km was simulated with 1.4 million elements, with a grid size of 1m near buildings and 10cm at the source. A release of 1000 particles per second was simulated until the concentration field reached steady state. SLAM agreed well with concentrations computed using FLUENT and wind tunnel results, at a computational cost 40 times lower than that of the FLUENT simulations (the SLAM calculation took 6 min CPU time on a workstation). It was able to reproduce the decay of maximum ground concentration with distance, as well as the increase in the plume transverse standard deviation.

FAST3D-CT – CT-Analyst

Another model employing the use of pre-computed flow fields stored in a database was reported recently by Schatzmann et al. (2011). This modelling system used the LES model FAST3D-CT to compute detailed velocity and turbulence fields. Two domain sizes representing an area of Hamburg were simulated in the study reported by Schatzmann et al. (2011): a 16km x 12km domain at 10m resolution, and a smaller 4km x 4km domain at 2.5m resolution. Simulations were carried out for 18 wind directions, with no buoyancy effects (i.e. for neutral atmospheric conditions).

The wind fields obtained by FAST3D-CT were then used to generate results for different wind directions, wind speeds, sources and source locations using a tool called CT-Analyst. It is assumed that the dispersing material is a passive and inert gas. In contrast to the sophisticated Lagrangian dispersion modelling in the previous models, a much simpler method is employed in CT-Analyst to estimate the plume envelope. The method involves first estimating the area over which the plume spreads. The plume edge is defined by the two boundaries starting from the source that follow the maximum 'instantaneous' directional deviations from the mean velocity vector. This gives the total area over which the material is able to spread. The local velocity then determines the speed at which the plume moves. Local mean concentrations within the plume are estimated (as a function of source strengths) simply by applying mass conservation together with a plume profile assumption. Fluctuations about the mean are roughly estimated from results from releases at sample locations in the LES.

Since the system relies solely on pre-computed data, it produces nearly instantaneous results in operational use. The system also boasts a user-friendly graphical user interface. Schatzmann et al. (2011) report initial validation studies with wind tunnel experiments and two short field campaigns that are generally good. However, the very simple plume envelope procedure inherent in this approach would be unable to capture the short-range features discussed in Section 3.

4.6 Canopy of tall buildings

Consider again an array of cuboids of height *H* and side *L* as in Figure 4.1. New dynamics comes into play when the height of the buildings, *H*, is about three times the width of the bases, *L*. The main difference is that the vertical mixing associated with building wakes no longer extends from the top of the buildings right down to the ground, and hence it is no longer a good approximation to assume that the scalar is well mixed through the depth of the urban canopy. Current approaches do not consider this case, but applications in very densely urbanised cities such as New York, Hong Kong and other dense city centres are identifying the pressing need for methods to address this regime (see lectures at recent Advanced Study Institute at Chinese University of Hong Kong,
http://www.arch.cuhk.edu.hk/asi2011/en/Home/news.htm).

An intuitively attractive approach is to model the buildings as a porous canopy that exerts a drag on the flow, see also Section 2.1. This approach has been used successfully for canopies of trees and vegetation; see the reviews of Finnigan (2000) and Belcher et al (2012). Belcher et al (2003) and Coceal & Belcher (2004) develop this approach for flow in canopies of cubical obstacles and demonstrate promising comparisons with measurements. The basic idea is to compute the flow field averaged over horizontal layers. This method then works when the canopy elements are relatively thin compared to their height.

Consider first the vertical profile of the wind speed in a canopy. For a homogeneous array of buildings the mean momentum equation reduces to

$$\frac{d\tau}{dz} - D = 0 \qquad (4.56)$$

where τ is the turbulent shear stress, and the drag per unit volume exerted on the wind induced by the canopy elements is defined to be $D = U^2/L_c$, where U is the mean velocity (which formally represents the time-mean velocity averaged over horizontal planes), and L_c is the canopy drag length scale, which represents the drag of the buildings and can be approximated by

$$L_c = \frac{2H}{c_d}\frac{1-\lambda_p}{\lambda_f} \approx H\frac{1-\lambda_p}{\lambda_f} \qquad (4.57)$$

where c_d is the sectional drag coefficient, whose value is approximately 2 for cubical obstacles (Coceal & Belcher 2004). Finally, if the turbulent shear stress is modelled using a mixing length model with a constant mixing length, $l_m \approx c_1 L_c$, where c_1 is a constant (see Finnigan, 2000, for a discussion of this model), then the wind profile in the canopy decreases exponentially from the top of the canopy according to

$$U = U_H \exp\left(\frac{z-H}{c_2 L_c}\right) \qquad (4.58)$$

where $c_2 = (2)^{\frac{1}{3}} c_1$ and $U_h \approx 0.3u_*$ (Finnigan & Belcher 2004). This solution makes it clear that this regime is important where $H \gtrsim L_c$, i.e.

$$\frac{H}{L_c} \approx \frac{\lambda_f}{1-\lambda_p} \gtrsim 1 \qquad (4.59a)$$

which for a uniform array of cuboids becomes

$$\frac{H}{L_c} \approx \frac{\lambda_f}{1-\lambda_p} = \frac{HL}{(L+W)^2 - L^2} \gtrsim 1 \qquad (4.59b)$$

This condition requires that the frontal area of the cuboids is larger than the plan area not occupied by buildings, which favours closely packed, tall buildings as expected.

The porous canopy approach can be extended to model the three-dimensional flow that develops through non-uniform canopies and to the edges of canopies (Belcher et al 2003; Coceal & Belcher 2004; Belcher et al 2012).

Hence in sufficiently dense canopies of tall buildings, the mean flow near the ground is very small and there is little turbulence there either to mix scalar (Finnigan 2000). Hence pollutant emitted near the ground remains trapped at low levels. Thus, as observed in central Hong Kong and some other cities, the concentration of vehicle pollution C_s is very large near the surface and decreases rapidly with height. The profile of concentration with height is sensitive to the profile of turbulence. Experience of forest canopies suggests that the density of the building array itself determines the turbulence conditions (Finnigan 2000) that drive mixing over a depth of order L_c into the canopy. Building wakes may also be important in organised regular arrays of tall buildings (Poggi et al 2004).

Dispersion processes in these dense arrays of obstacles are reviewed in Nepf (2012).

4.7 Dispersion of plumes impinging upon cities

The purpose of this section is to investigate how the boundaries (upwind, side and downwind) of an urban area might affect flow and dispersion patterns in that area. A typical application might be in city centres with clusters of high buildings as well as low buildings and streets (Li 2009). The processes have been studied in field and laboratory experiments with enough blocks that the transition zones around the boundaries were significant and could be studied.

In these cases it is appropriate to apply generalised plume formulae (Hunt, 1985) for dispersion relative to a streamline in terms of mean streamline, and turbulence σ_u, σ_v, σ_w along the streamline. This is the approach used for example in the complex terrain model of ADMS (FLOWSTAR).

Figure 4.12 shows 3 typical configurations of a plume impinging upon a large urban area. In case (a) the resistance to the flow is large. As explained in Section 2.1 the mean streamwise flow $\boldsymbol{U}$ decreases from $\boldsymbol{U}_o$ to $\boldsymbol{U}_c$ over an adjustment region, of length of order L_c (Belcher et al 2003). By continuity there is then a vertical motion out of the top of the canopy, which advects the plume upwards. In addition, the spanwise turbulence intensity σ_v / U increases and the plume rapidly spreads (see aerial photos and wind tunnel experiment of Davidson *et al.*, 1996). These effects tend to cancel and the mean concentration does not decrease significantly.

In case (b) the flow encounters a weaker resistance. The mean centreline streamline then passes around or over the canopy. If the distance between the centreline streamline and the canopy (n_o) is less than the mean plume width $(\sigma_y^2 + \sigma_z^2)^{\frac{1}{2}}$, then the plume impacts upon the canopy. The effective diffusivity within the canopy ($K_c \sim \boldsymbol{U}'_c W$) is much less than in the atmosphere ($K_o \sim u_o' H$), and so the plume does not significantly lose material into the canopy. However just as with dispersion into streets (considered in Section 4.2) there is a flux ($\sim u'_c\, C_c$) into the canopy which is spread and diffused over a distance $\ell_c \approx \sqrt{K_c L_p / \boldsymbol{U}_c}$, where L_p is a characteristic distance around the perimeter of the canopy region.

There is also a third case (c) where a typical mean streamline penetrates the canopy, but tangentially so that the streamline is at a distance n_c from the edge of the canopy. But now, unlike the first case, n_0 is less than the plume width $(\sigma_y^2 + \sigma_z^2)^{1/2}$. In this case the effective mean streamline is now on the edge of the canopy, somewhat like a plume impacting onto a mountain. The concentration is determined by the exterior dispersion process (See Figure 4.13). We are not aware of experiments for this case.

5. Conclusions and recommendations for future work

The requirement reviewed here is to predict the dispersion of material released from line or point sources at short range in an urban area, where the dimensions and arrangement of buildings is paramount in determining the dispersion pattern. We have reviewed the city-scale context of the meteorology of urban areas (Section 2), the phenomenology of flow and dispersion in cities (Section 3) and methods for modelling dispersion in urban areas (Section 4). We now draw together some conclusions on our ability to predict dispersion in urban areas in an operational setting. For this reason we organise the conclusions first in Section 5.1 around classes of operational models and how they relate to the physical processes that control dispersion in different regimes, and then in Section 5.2 we identify priorities for future work.

5.1 Parameter regimes, modelling approaches and software packages

In Section 3.2.3 we described the empirical result that the scalar concentration along the centreline of the plume, C_{max}, falls off as an inverse square law, namely

$$C_{max}(R) = A\frac{Q}{UR^2}$$

where Q is the (steady) source strength, U is a reference wind speed, and R is the distance from the source, and $A = 6 - 12$ is an empirical coefficient. If this result could be shown to be more generally applicable, then it could form the basis for estimating dispersion in urban areas.

In order to focus physically based modelling we defined in Figure 4.1 a parameter space of urban geometry for regular arrays of cuboids. This parameter space was then divided into three regimes: sparse arrays, street networks, and tall-building canopies. The dominant physics that controls dispersion is different within each of these three regimes:

- Within *sparse arrays* the flow can be approximated as a superposition of wakes from the individual buildings, which then control the dispersion patterns (section 4.2)
- Within *street networks* the buildings are sufficiently close together that the regions between them form streets that join at intersections. Dispersion within the building array is controlled by the branching flow through the network of streets and intersections and the exchange of material with the boundary layer above (Section 4.4)
- Within *tall-building canopies* the buildings are so tall that the flow between them is restricted and does not mix pollutant effectively through their height. Pollutant then becomes trapped near the surface (Section 4.6).

So how do current software packages relate to these regimes? There are four basic classes of dispersion model currently in use: Gaussian plume models, street canyon models, street network models and building resolving models. Each of these is now discussed briefly in turn and assessed in their suitability to model dispersion in the different regimes Table 2 gives an example of each type of model together with brief details of some of their software characteristics and building and meteorological data input requirements.

5.1.1 Gaussian plume models

A number of Gaussian models have been developed where the plume spread and advection speeds are tailored to urban conditions. This tailoring may be entirely empirically based, with plume spreading rates estimated as functions of simple geometrical factors that describe the size and density of the buildings comprising the urban canopy. This approach provides the basis of the accidental release model UDM (Hall *et al.*, 2001), which was originally developed to take account of the interaction of wakes in areas with low building density by appropriate modelling of the changed cloud dimensions (see Section 4.2.5). These methods have been found to work quite satisfactorily in regular arrays of cuboids, but clearly lack the capacity to reproduce the local effects in street networks that have been discussed above in Section 4.4. Furthermore, in close-packed buildings and in street canyons with arbitrary source distributions the mean concentration distributions are significantly non-Gaussian at least on the building/street scale, and there is lateral displacement of plume centreline streamline (Section 4.2.4). Nevertheless, the UDM has been extended to quite high-density urban areas, through empirical adjustment, and by extensive wind tunnel and field validation. In such situations it is less accurate, but comparisons with field data have shown (Brook *et al.*, 2002) that it captures many of the observed plume features. UDM may also be applied to the large numbers of buildings over the neighbourhood scale (300 to 3,000m approximately) by applying cloud splitting and merging techniques in which groups of overlapping split plumes are merged and formed into new, larger plumes.

As described in Section 4 there are three situations when a Gaussian plume model has theoretical support:

- When the urban canopy is sparse, the flow can be considered as a superposition of building wakes. The plume then grows as a Gaussian, but with additional sources as is done within UDM and ADMS.
- For the special case of a uniform geometry, street network models tend to give lateral concentration profiles with a Gaussian variation after a ‘large’ number of intersections. Importantly, the parameters of the Gaussian variation are determined by the geometry of the streets and their intersections.
- In a far field, distant from the source, when more than say half of the plume material lies above the urban canopy, the plume dynamics are controlled by turbulence in the atmospheric boundary layer and the buildings merely provide roughness. The plume then grows as a conventional Gaussian plume – the processes special to urban areas sets the initial area of the plume. In this case the classical model of Briggs (1973), or some more recent incarnations (e.g. Hanna & Britter, 2003), suffice.

The Gaussian form of plume model is seductive because of its simplicity and prevalence in dispersion problems. However, the evidence analysed here shows that the spreading rates vary in complex ways, with the geometry and turbulence playing different roles in the spreading of the plume depending on the urban geometry and the distance from the source. In order for Gaussian plume models to capture this full richness of behaviour would require great empirical input and so the limits of its validity need to be very carefully documented. So, for sparse arrays modified Gaussian approaches are probably sufficient, but for more general situations we conclude that a simple Gaussian plume is probably an over-simplification.

5.1.2 Street canyon models

Street canyon models, such as OSPM (Berkowicz 2000), explicitly represent the flow field in a street canyon parametrically, including the canyon circulation and the effects this has on dispersion patterns. These models have been shown to work well when applied to traffic pollution as monitored in busy urban streets, for example in the application of ADMS-Urban to pollution in London. OSPM and its derivatives are designed to treat canyons with a range of cross-section shapes. The essential limitation of this approach is that intersections are ignored. Where the streets cross, they are treated as if independent of each other. On the whole, monitoring sites are positioned well away from the complex conditions found at intersections and are likely to be located where a street canyon model is not unreasonable. At such positions, local traffic sources dominate street-side pollution levels. Both effects contribute to the likelihood of reasonable model performance. The deficiencies of the models readily become apparent when point sources are at issue, rather than line emissions from traffic; dispersion through the street network is then important. Of course, OSPM was not designed with this application in mind.

5.1.3 Street network models

Street network models form a third class of dispersion model. In this approach the street-network geometry is represented explicitly and thus they provide natural extensions of the other two methodologies. Street network models are box models, with each street segment between an intersection treated as a well-mixed volume into and from which pollutant is carried by the mean flow along the canyon, whilst being exchanged at the top boundary with the external flow. Empirical algorithms define how fluxes are divided at intersections between the input and output streets.

There are currently three developments of the street network approach (Soulhac 2000, Hamlyn et al 2008; Belcher et al 2012), although only one has the software wrapping for operational air quality use, namely SIRANE, which is based upon Soulhac (2000). SIRANE has been successfully applied to air quality modelling in a number of European cities. Its main weakness for present purposes is the assumption of well-mixed conditions in the street segments that contain a source. The use of a single box to represent a long street segment is not essential to the methodology and sub-segments can, in principle, be used to provide more detail of concentration gradients along a street (see Section 4.4.2 above). Of course, because it treats the building configuration explicitly, SIRANE requires more complex input than the other two approaches where a few statistical properties of the urban district suffice. Recent development of SIRANERISK, the unsteady version of SIRANE, will allow the application of the model to emergency response modelling.

Street network models demonstrate the role of intersections in lateral dispersion within the urban canopy. This process can be represented in Lagrangian dispersion models (for example the Met Office NAME model), which represent the motion of many thousands of individual air parcels with a parameterised stochastic motion to represent the turbulent processes. A Lagrangian street network model can then be constructed by advecting particles randomly through the street network, with one step being a street length and then a random choice of exit at intersections. Such a model is currently being tested at the Met Office (David Thomson personal communication).

The limitations of the basic street network approach (common to all three model implementations mentioned above) relate to the assumption of well-mixed volumes (see Figure 4.1). Firstly, the streets must be narrow, so that $H/W > 0.3$, then the recirculation within the street fills the street and so the concentration across the street is well mixed. Secondly, if the buildings are sufficiently low that $H/L < 3$, so that the concentration is well mixed in the vertical. Provided these conditions are met the network approach looks promising.

5.1.4 Building-resolving models

In Section 4.5 we reviewed various operational methods of modelling dispersion using building resolving models. Most of the methods added elements to the flow field that represented the effects of the buildings, and so are similar in spirit to the street canyon models. Clearly they are as successful as the elements added to the flow. For these models there is a significant requirement to document carefully when the approach fails in order to help guide operational use.

The approach of Flow Air 3D and FAST3D-CT of pre-computing the flow and then doing only the dispersion calculation in real time is promising. It is clearly well suited to sensitive locations where the high cost of the flow calculations can be justified. But it is not without its limitations, as discussed in Section 4.5.3. This general methodology of pre-calculating the flow field or flow variables could be used more generally in conjunction with other dispersion models. For example, one could envisage the advection and detrainment velocities being pre-computed and stored (rather than the full flow fields) for different flow conditions as an alternative to using parametrizations of these velocities in the SIRANE model. However, such a procedure would only really be justified if it were demonstrated to yield better predictions.

5.2 Priorities for future research

Throughout this review the strengths and weaknesses of the various approaches have been described. Here we draw together themes that emerge from the review that set the priorities for the future:

1. The regime diagram in Figure 4.1 and the discussion of processes given in this review indicates that different simplified methods are likely to be useful in different urban settings, and that there is no single method that works for all urban areas. This delineation of regimes is new. A clear priority is therefore to demonstrate these differences by carefully selecting observed case studies that represent each regime. There is good wind tunnel or water channel data for the sparse array regime and the DAPPLE case is ideal for demonstrating the street network regime. The tall building canopy regime does not have a good data set. Wind tunnel studies with idealised geometries spanning a range of heights from moderate to extreme (such as Hong Kong and New York) would be a useful first step. Full-scale observations in a real city such as Hong Kong are essential, albeit expensive. Some field data for New York already exists – see the references in Section 3.2.5. It is then important to use the full range of models on data sets for all regimes to demonstrate when particular model approaches fail.

2. The street network approach is less mature than the other approaches. Further work is needed to establish the sensitivity of the results to the various details of the different implementations. The use of data from the DNS simulations of Branford et al (2011) provide an example where comprehensive numerical data can be used to evaluate these implementations. Such DNS or LES studies need to be extended to building configurations with longer streets that are more representative of the street network regime. The approach needs to be tested in more challenging geometries: the DAPPLE experiments provide ideal data that has not been fully exploited. It would be very useful for future experiments of this type to be designed with this type of application in mind, so that suitable measurements can be made – including simultaneous measurements of concentrations and velocities.

3. Robust methods of estimating the parameters of the various modelling approaches need to be developed. From a practical point of view, an important objective would be to automate the creation of the network model or the derivation of geometric parameters (such as building heights, street lengths and widths, and average morphological parameters such as λ_f and λ_p) from the national digital mapping

database. There is also a role for inverse modelling here. The combination of measurements of concentration from known sources with a model can be used to infer model parameters using formal inverse techniques. Such methods have been used to estimate the location and strength of sources (e.g. ADMLC-R6; Rudd et al. 2012), but can be generalised for parameter estimation.

4. Isolated tall buildings and open spaces have an important effect on the flow and dispersion in urban areas. We do not currently know how to represent their effects in either the Gaussian plume methods or the street network methods. The first goal should be to assess the significance of these deficiencies, then to develop suitable models, as appropriate. The development of such models will need to be informed by experiments and LES studies that can probe a range of parameters, such as building height. Xie et al. (2008) showed that even a “moderately tall” building with a height of 1.7 times the mean building height has a considerable influence on the flow.

5. Canopies of tall buildings are not catered for in operational models, indeed there is no proven methodology for attempting modelling. Measurements are needed as noted under point 1 above. The porous canopy approach looks to be a good possibility, but there is a long way to go before we have an evaluated operational tool.

6. The measurements described in this review are for neutrally stratified meteorology and the operational tools reviewed here all assume the basic meteorology (wind speed and stability) to be uniform over the domain of interest. These deficiencies in data can be addressed by making measurements in a wider variety of stabilities, although we note the finding of Section 2.2 that stable conditions are less prevalent in urban areas. These deficiencies in operational tools might be addressed by combining mesoscale meteorological models with a porous canopy approach to model the winds and with perhaps a street network model of dispersion.

7. Most dispersion models do not account for low winds. Low wind conditions arise in strongly convective conditions, when pollutant is mixed strongly by the convective turbulence, and in strongly stably stratified conditions – but as we saw in Section 2.2 stably stratified conditions are much rarer in urban than rural areas. In low wind conditions other processes come into play. For example, as explained in Section 3.2.1 there is evidence from the DAPPLE experiments for a role of traffic induced turbulence in transporting pollutant against the local wind direction. The basic properties of traffic induced mixing are understood, but they need to be integrated with the appropriate wind-driven dispersion model (e.g. street network model). This mechanism is likely to be particularly important in tall building canopies, when the wind-driven dispersion is weak.

References

Allwine, KJ, Shinn, HJ, Streit, GE, Clawson, KL and Brown, M, 2002. Overview of URBAN 2000: A multiscale field study of dispersion through an urban environment. Bull. Amer. Meteor. Soc., 83, 521–536.

Allwine, JK et al., 2004. Urban dispersion processes investigated during the Joint Urban 2003 study in Oklahoma City. Proc 5th AMS Urban Env. Conf, Vancouver, BC, 2004.

Allwine, JK and Flaherty, JE, 2007. Urban dispersion program overview and MID05 field study summary. Pacific Northwest National Laboratory Report, PNNL-16696.

Allwine, JK, et al., 2007. Urban dispersion program: urban measurements applied to emergency response. Proc 7th Symposium on the Urban Environment 10-13 September 2007, San Diego, CA.

Arnold, S et al., 2004: Dispersion of air pollution and penetration into the local environment, DAPPLE. Sci. Total Environ., 332, 139–153.

Bachlin, W, Theurer, W and Plate, EJ. 1991. Wind field and dispersion in a built-up area – a comparison between field measurements and wind tunnel data. Atmos. Environ., 25A, 7, 1135-1142.

Bachlin, W and Plate, EJ., 1988. Wind tunnel simulation of accidental releases in chemical plants. Environmental Meteorology, Kluwer, pp 291-303.

Balogun, AA et al., 2010. In-street wind direction variability in the vicinity of a busy intersection in central London. Boundary-Layer Met., 136, 489–513.

Barlow JF and Coceal O, 2009. A review of urban roughness sublayer turbulence. Met Office Technical Report No. 527, 68 pp. See also http://research.metoffice.gov.uk/research/nwp/publications/papers/technical_reports/reports/527.pdf

Barlow, J. F., Dobre, A., Smalley, R. J., Arnold, S. J., Tomlin, A. S. and Belcher, S. E. (2009) Referencing of street-level flows measured during the DAPPLE 2004 campaign. Atmospheric Environment, 43 (34). pp. 5536-5544.

Belcher, SE, 2005. Mixing and transport in urban areas. Phil Trans R Soc A, 363, 2947-2968.

Belcher, SE, Harman, I, Finnigan, JJ, 2012. The wind in the willows: flows in forest canopies in complex terrain. Annual Review of Fluid Mechanics 44, 479-504.

Belcher SE, Coceal O, Goulart EV, Rudd, A, Robins, A 2012 A network model for dispersion of a passive scalar through an urban street network. In preparation.

Belcher, S.E., Jerram, N. and Hunt, J.C.R. (2003) Adjustment of a turbulent boundary layer to a canopy of roughness elements. J. Fluid Mech., 488, 369-398.

Berkowicz, R., Hertel, O., Sørensen, N.N. and Michelsen, J.A. (1997) Modelling Air Pollution from Traffic in Urban Areas. In Proceedings from IMA meeting on Flow and Dispersion Through Obstacles, Cambridge, England, 28-30 March, 1994, eds. R.J.

Perkins and S.E. Belcher, pp. 121-142.

Berkowitz, R, 2000. OSPM – a parameterised street pollution model. J Env. Moni. Assess., 65, 323-331.

Biltoft, C, 2001. Customer report for Mock Urban Setting Test. Rep. No. WDTC-FR-01-121, US Army Dugway Proving Ground.

Biltoft, CA, 2001. Customer report for Mock Urban Setting Test. DPG Document No. 8-CO-1560-000-052.

Biltoft, CA, Yee, E. and Jones, CD, 2002. Overview of the Mock Urban Setting Test, in 4th Symposium on the Urban Environment, American Meteorological Society, Norfolk, VA.

Bohnenstengel, S. I., Evans, S., Clark, P. A. and Belcher, S. E. (2011) Simulations of the London urban heat island. Quarterly Journal of the Royal Meteorological Society, 137 (659). pp. 1625-1640

Branford, S., O. Coceal, T.G. Thomas, S.E. Belcher, 2011. Dispersion of a point-source release of a passive scalar through an urban-like array for different wind directions. Boundary-Layer Meteorol. 139, 367-394.

Briggs, G.A., 1973. Diffusion Estimation for Small Emissions. Atmospheric Turbulence and Diffusion Laboratory.

Britter, RE, Di Sabatino, S, Caton, F, Cooke, KM, Simmonds, PG and Nickless, G, 2002. Results from three field tracer experiments on the neighbourhood scale in the city of Birmingham UK. Water Air Soil Pollution: Focus, 2, 79–90.

Britter, RE and Hanna, SR, 2003. Flow and dispersion in urban areas. Ann. Rev. Fluid Mech., 35, 469-496

Britter, R.E. and Hunt, J.C.R. (1979) Velocity measurements and order-of-magnitude estimates of the flow between two buildings in a simulated atmospheric boundary layer. J. Ind. Aero., 4, 165-182.

Brixley, LA, et al., 2009. The effect of a tall tower on flow and dispersion through a model urban neighbourhood; Part 2. Pollutant dispersion. J. Environ. Monit., 11, 2171–2179

Brook, D.R., Beck, N.V., Clem, C.M., Strickland, D.C., Griffiths, I.H., Hall, D.J., Kingdon, R.D. and Hargrave, J.M. (2002) Validation of the Urban Dispersion Model (UDM). In Proceedings of 8th International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes, Sofia, Bulgaria, 14-17 October 2002, pp.8-12.

Brown, M., A. Gowardhan, M. Nelson, M. Williams, and E. Pardyjak, 2009. Evaluation of the QUIC wind and dispersion models using the Joint Urban 2003 Field Experiment dataset, AMS 8th Symp. Urban Env., Phoenix, AZ, 16 pp.

Brown, M.J. and Williams, M.D. (1998), An Urban Canopy Parameterization for Mesoscale Meteorological Models, LA-UR-98-3831. See also http://library.lanl.gov/cgi-bin/getfile?00418760.pdf.

Cai, X., J.F. Barlow, S.E. Belcher, 2008. Dispersion and transfer of passive scalars in and above street canyons – Large-eddy simulations. Atmospheric Environment 42, 5885-5895.

Carpentieri, M, Robins, AG and Baldi, S., 2009. Three dimensional mapping of air flow at an urban canyon intersection. Boundary Layer Met., 133, 277-296

Carpentieri, M and Robins, AG, 2010. Tracer flux balance at an urban canyon intersection, Boundary Lyer Met., 135, 229-242

Carpentieri, M, Salizzoni, P., Robins, A., Soulhac, L., 2012 Evaluation of a neighbourhood scale, street network dispersion model through comparison with wind tunnel data, Environmental Modelling & Software. In press.

Carruthers DJ, Blair J and Johnson K (2003) Validation and sensitivity study of ADMS-Urban for London. Topic report prepared for Defra, National Assembly for Wales, The Scottish Executive and The Department for the Environment, Northern Ireland.
http://www.cerc.co.uk/environmental-research/assets/data/CERC_2003_ADMS-Urban_validation_and_sensivity_study_for_London_10_TR-0191-h.pdf.

Carruthers D.J., S. di Sabatino S and J.C.R. Hunt (2012) Urban Air Quality: Meteorological Processes. In Encyclopedia of Sustainability Science and Technology. To be published

Carruthers, D.J., Edmunds, H.A., Lester, A.E., McHugh, C.A. and Singles, R.J. (2000) Use and Validation of ADMS-Urban in Contrasting Urban and Industrial Locations. Int. J. Environment and Pollution 14, no. 1-6, 364-374, DOI: 10.1504/IJEP.2000.000558

Carruthers, D.J., Holroyd, R.J., Hunt, J.C.R., Weng, W.-S., Robins, A.G., Apsley, D.D., Thomson, D.J. and Smith, F.B. (1994) UK-ADMS: A new approach to modelling dispersion in the Earth's atmospheric boundary layer. Journal of Wind Engineering and Industrial Aerodynamics, 52, 139-153.

Carruthers, D.J., Hunt, J.C.R. and Weng, W.-S. (1988) A computational model of stratified turbulent airflow over hills – FLOWSTAR I. Proceedings of Envirosoft. In Computer Techniques in Environmental Studies (editor P. Zanetti), pp. 481-492. Springer-Verlag.

Carruthers DJ, McHugh CA, Dyster S, Stidworthy A & Oates W (2001) ADMS: Fundamental Aspects, Validation and Comparisons to Other Models. AWMA Guideline on Air Quality Models: A New Beginning, April 2001, Rhode Island, USA

Carruthers DJ, McHugh CA, Vanvyve E, Seaton MD and Solazzo E (2009) Comparison of ADMS and AERMOD meteorological preprocessor and dispersion algorithms. Air & Waste Management Association, Guideline on Air Quality Models: Next Generation of Models, October 2009, Raleigh NC, USA.

Carruthers D, Seaton M, McHugh C, Sheng X, Solazzo E and Vanvyve E (2011) Comparison of the complex terrain algorithms incorporated into two commonly used local-scale air pollution dispersion models (ADMS and AERMOD) using a hybrid model. Journal of the Air and Waste Management Association, vol. 61, issue 11, pp. 1227-1235, DOI:10.1080/10473289.2011.609750.

CERC, 2001. ADMS-Urban User Guide. Cambridge Environmental Research Consultants.

CERC (2010) ADMS Technical Specification (2010) www.cerc.co.uk/environmental-software/model-documentation.html#technical

Chang, J.C., Hanna, S.R., 2004. Air quality model performance evaluation. Meteorol. Atmos. Phys. 87, 167-196.

Chatwin, PC, 1968. The dispersion of a puff of passive contaminant in the constant stress region. Quart. J. R. Met. Soc., 94, 350-360.

Cierco, F.X., Soulhac, L., Méjean, P., Lamaison, G., Salizzoni, P., Armand, P., 2010. SIRANERISK: an operational dispersion model for urban areas incorporating a new method to account for concentration fluctuations. In: 13th International Conference on Harmonisation Within Atmospheric Dispersion Modelling for Regulatory Purposes, Paris, France, 1-4 June 2010.

Cimorelli, A.J., Perry, S.G., Venkatram, A., Weil, J.C., Paine, R.J., Wilson, R.B., Lee, R.F. and Peters, W.D. (1998) AERMOD: Description of Model Formulation. (12/15/98 Draft Document) Prepared for Environmental Protection Agency, Research Triangle Park, NC. (Docket No. A-99-05; II-A-1). 113pp.

Claus, J., Coceal, O., Thomas, T.G., Branford, S., Belcher, S.E. & Castro, I.P., 2012. Wind direction effects on urban-type flows, Boundary-Layer Meteorol. 142, 265-287.

Claus, J, Krogstad, PA and Castro, IP, 2012. Some measurements of surface drag in urban-type boundary layers at various wind angles. Boundary Layer Meteorology, DOI 10.1007/s10546-012-9736-3,

Coceal, O. and Belcher, S.E. (2004) A canopy model of mean winds through urban areas. Q. J. R. Meteorol. Soc., 130, pp. 1349–1372.

Coceal, O., Thomas, T.G., Castro, I.P. & Belcher, S.E., 2006. Mean flow and turbulence statistics over groups of urban-like cubical obstacles, Boundary-Layer Meteorol. 121, 491-519.

Coceal, O., Dobre, A., Thomas, T.G. & Belcher, S.E., 2007. Structure of turbulent flow over regular arrays of cubical roughness, J. Fluid Mech. 589, 375-409.

DAPPLE, 2011. Final reports to the Home Office; DAPPLE 2009/13 v3. www.dapple.org.uk/Live_Resources/Reports/DAPPLE%20final.pdf

Davidson, M.J., Mylne, K.R., Jones, C.D., Phillips, J.C., Perkins, R.J., Fung, J.C.H. and Hunt, J.C.R. (1995) Plume dispersion through large groups of obstacles – a field investigation. Atmospheric Environment, 29, 3245-3256.

Davidson, M.J., Snyder, W.H., Lawson, R.E., Hunt, J.C.R. (1996) Plume dispersion from point sources upwind of groups of obstacles – wind tunnel simulations. Atmospheric Environment, 30, 3715-3725.

Di Sabatino, S, Kastner-Klein, P, Berkowicz, R, Britter, RE and Fedorovich, E, 2003, The modelling of turbulence from traffic in urban dispersion models – Part I: Theoretical considerations. Environ. Fluid. Mech. 3, 129–143.

Dixon, N, Boddy, J, Smalley, R and Tomlin, A., 2006. Evaluation of a turbulent flow and dispersion model in a typical street canyon in York, UK, Atmos Environ., 40, 958-972.

Dobre et al., 2005. Flow field measurements in the proximity of an urban intersection in London, UK. Atmos. Environ. 39, 4647-4657.

Duchenne C., P. Armand, O. Oldrini, C Olry and J. Moussafir (2011), “Application of PMSS, the parallel version of MSS, to the micrometeorological flow field and deleterious dispersion inside an extended simulation domain covering the whole Paris area”, 14th International Conference on Harmonisation within Atmospheric Dispersion Modelling fir Regulatory Purposes, Harmo’14, Kos (Greece), Oct. 2-6, 2011.

Dupont S, Brunet Y. 2008. Edge flow and canopy structure: a large-eddy simulation study. Bound.-Layer Meteorol. 126:51–71.

Fackrell, JE, 1980. A flame ionisation detector for measuring fluctuating concentration. J. Phys. E: Sci. Instrum., Vol. 13 (1980), 888-893.

Fackrell JE, Robins AG (1982) Concentration fluctuations and fluxes in plumes from point sources in a turbulent boundary layer. J Fluid Mech 117:1–26.

Feddersen, B, Leitl, B, Rotach, MW and Schatzmann, M, 2004. Wind tunnel modeling of urban turbulence and dispersion over the City of Basel (Switzerland) within the BUBBLE project. Fifth Symposium on the Urban Environment (AMS), Vancouver, Canada, 23–28 August 2004

Fernando, HJS (2010) Fluid Dynamics of Urban Atmospheres in Complex Terrain. *Ann. Rev. Fluid Mech.* **42** 365-389

Finnigan JJ (2000) Turbulence in plant canopies. Annu Rev Fluid Mech 32:519–572

Gayev, Y.A. and Hunt, J.C.R. (2007) Flow And Transport Processes With Complex Obstructions: Applications to Cities, Vegetative Canopies, and Industry – NATO Science Series II: Mathematics, Physics and Chemistry v.236.

Goulart (2012), Flow and dispersion over urban areas. PhD Thesis. University of Reading.
Gowardhan, A., Pardyjak, E., Senocak, I., Brown, M, 2011. A CFD-based wind solver for an urban fast response transport and dispersion model, Environ Fluid Mech, doi:10.1007/s10652-011-9211-6.

Gowardhan, A. and Brown, M, 2012. A Study of the Effects of Different Urban Wind Models on Dispersion Patterns Using Joint Urban 2003 Data, 92nd AMS Annual Meeting, New Orleans, LA, 8 pp.

Griffiths, IH, Brook, DR, Hall, DJ, Barry, A, Kingdon, RD, Clawson, K, Biltoft, C, Hargrave, JM, Clem, CM, Strickland, DC and Spanton, AM, 2002. Urban dispersion model (UDM) validation. American Met Soc 4th Symposium on the urban environment, Davis, Ca, May 2002.

Grimmond, C. S. B. and Oke, T. R. 1999 Aerodynamic properties of urban areas derived from analysis of surface form. J. Appl. Meteorol., 38, 1262–1292

Grimmond, C.S.B., M. Blackett, M. Best, J. Barlow, J.-J. Baik, S. Belcher, S. E. Bohnenstengel, S.I. Calmet, F. Chen, A. Dandou, K. Fortuniak, M. Gouvea, R. Hamdi, M. Hendry, H. Kondo, S. Krayenhoff, S. H. Lee, T. Loridan, A. Martilli, S. Miao, K. Oleson, G. Pigeon, A. Porson, F. Salamanca, L. Shashua-Bar1, G.-J. Steeneveld, M. Tombrou, J. Voogt, N. Zhang. 2010 The International Urban Energy Balance Models Comparison Project: First results. *J. Appl. Met. Climatol.* **49** 1268-1292

Grimmond, CSB, M Blackett, MJ Best, J-J Baik, SE Belcher, J Beringer, SI Bohnenstengel, I Calmet, F Chen, A Coutts, A Dandou, K Fortuniak, ML Gouvea1, R Hamdi1, M Hendry, M Kanda1, T Kawai1, Y Kawamoto, H Kondo, ES Krayenhoff1, S-H Lee, T Loridan, A Martilli, V Masson, S Miao, K Oleson, R Ooka, G Pigeon, A Porson, Y-H Ryu, F Salamanca, GJ Steeneveld, M Tombrou, JA Voogt, D Young, N Zhang. 2011 Initial Results from Phase 2 of the International Urban Energy Balance Comparison Project. *Int. J. Climatology* **31** 244-272.

Hall, DJ, Macdonald, R, Walker, S and Spanton, AM, 1996. Measurements of dispersion within simulated urban arrays – a small scale wind tunnel study. Building Research Establishment, BRE Report CR178/96.

Hall, DJ, Macdonald, RW, Walker, S and Spanton, AM, 1998. Measurements of dispersion within simulated urban arrays – a small scale wind tunnel study. BRE UK Report No. CR244/98.

Hall, DJ, Macdonald, R, Walker, S, Mavroidis, I, Higson, H and Griffiths, RF, 1997. Visualisation studies of flows in simulated urban arrays. BRE Report CR39/97.

Hall, D.J., Spanton, A.M., Griffiths, I.H., Hargrave, M., Walker, S. and John, C. (2001) The UDM: a puff model for estimating dispersion in urban areas. In Proceedings of 7th International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes, Belgirate, Italy, pp.256-260.

Hamlyn D, Hilderman T, Britter R (2007) A simple network approach to modelling dispersion among large groups of obstacles. Atmos Environ 41: 5848-5862.

Hanna, S, Fabian, P, Chang, J, Venkatram, A, Britter, R, Neophytou, M & Brook, D, 2004. Use of Urban 2000 field data to determine whether there are significant differences between the performance measures of several urban dispersion models. Proc. 5th AMS Conference on the Urban Environment, Vancouver, August 2004.

Harman, I.N. and Belcher, S. E. (2006) The surface energy balance and boundary-layer over urban street canyons. Quarterly Journal of the Royal Meteorological Society, 132 (621). pp. 2749-2768.

Heist, DK, et al., 2009. The effect of a tall tower on flow and dispersion through a model urban neighbourhood; Part 1. Flow characteristics. J. Environ. Monit, 11, 2163–2170

Hertel, O., Berkowicz, R. and Larssen, S. (1990) 'The Operational Street Pollution Model (OSPM).' 18th International meeting of NATO/CCMS on Air Pollution Modelling and its Applications. Vancouver, Canada, pp741 749.

Hilderman T, Chong R (2007) A laboratory study of momentum and passive scalar transport and diffusion within and above a model urban canopy final report. Contract Report DRDC Suffield CR 2008-025, 70 pp.

Hilderman, T, Chong, R and Kiel, D, 2004. Urban dispersion modelling data. Coanda R&D Report CRDC00327, March 2004, Coanda R&D Corp, BC, Canada.

Hoydysh, WR and Dabberdt, W, 1988. Kinematics and dispersion characteristics of flows in asymmetric street canyons. Atmos. Environ., 22, 677-689

Hoydysh, WR and Dabberdt, W, 1994. Concentration fields at urban intersections: fluid modelling studies. Atmos. Environ., 28,1849-181860

Huber A. (1991) Wind tunnel and Gaussian plume modeling of building wake dispersion. Atmospheri c Environment, 25, 1237-1249.

Hunt, J.C.R. (1985) Turbulent diffusion from sources in complex flows. Annual Review of Fluid Mechanics, 17, 447-485.

Hunt, J.C.R., Carruthers, D.J., Daish, N. and Britter, R., (2004) Dispersion from Accidental Releases in Urban Areas. See also http://www.admlc.org.uk/documents/ADMLC20023_000.pdf.

Hunt, J.C.R., Fernando, H.J.S., Princevac, M., (2003) Unsteady thermally driven flows on gentle slopes. Journal of the Atmospheric Sciences, 60(17), 2169–2182.

Hunt, J.C.R. and Mulhearn, P.J. (1973) Turbulent dispersion from sources near two-dimensional obstacles. J. Fluid Mech., 61, 245-274.

Hunt, J.C.R., Puttock, J.S. and Snyder, W.H. (1979) Turbulent diffusion from a point source in stratified and neutral flows around a three-dimensional hill. Part I. Diffusion equation analysis. Atmospheric Environment, 13, 1227-1239.

Hunt, J.C.R., Vrieling, A.J., Nieuwstadt, F.T.M. and Fernando, H.J.S., (2003) The influence of the thermal diffusivity of the lower boundary on eddy motion in convection. Journal of Fluid Mechanics, 491, 183-205.

Hunt, JCR, Carruthers, DJ, Britter, RE and Daish, NC, 2004. Dispersion of accidental releases in urban areas. ADMLC Report ADMLC/2002/3.Kastner-Klein, P, Fedorovich, E, Ketzel, M, Berkowicz, R and Britter, RE, 2003,
The modelling of turbulence from traffic in urban dispersion models – Part II: Evaluation against laboratory and full-scale concentration measurements in street canyons. Environ. Fluid. Mech. 3, 145-172.
Kastner-Klein P , Fedorovich E and Rotach MW, 2001. A wind tunnel study of organised and turbulent air motions in urban street canyons. J Wind Eng Ind Aero., 89, 849–861.

P. Kastner-Klein, P, B. Leitl, B, Pascheke, F and Schatzmann, M, 2004. Wind tunnel simulation of the Joint Urban 2003 tracer experiment. Proc. Symposium on Planning, Nowcasting, and Forecasting in the Urban Zone, 84th AMS Annual Meeting (Seattle, WA), Jan 10-16, 2004.

Kastner-Klein, P., Plate, EJ and Fedorovich, E., 1997. Gaseous pollutant dispersion around urban-canopy elements – a wind tunnel study. I J Env & Poll., 8, 727-737.
Klein, PM and Young, DT, 2011. Concentration fluctuations in a downtown urban area. Part 1: analysis of Joint Unban 2003 full-scale fast-response measurements. Environ. Fluid Mech., 11:23-42.

Klein, PM, Leitl, B and Schatzmann, M, 2011. Concentration fluctuations in a downtown urban area. Part 2: analysis of Joint Unban 2003 wind-tunnel measurements. Environ. Fluid Mech., 11:43-60.

Lamaison, G., Soulhac, L. and Armand, P., 2011a: Presentation of SIRANERISK-2.0 – A decision-support oriented computationnal tool adapted to the dispersion of deleterious RBC agent in the urban atmospheric environnement – Examples of application. 14th International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes. Kos Island, Greece, 2-6 October 2011.

Lamaison, G., Soulhac, L. Cierco, F.-X., Salizzoni, P. and Armand, P., 2011b: Validation of siranerisk-2.0 operational model against a lagrangian particle dispersion model and a new campaign of dispersion experiments performed in the lmfa-ecl wind tunnel in an idealized urban mock up. 14th International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes. Kos Island, Greece, 2-6 October 2011.

LCSQA 2009. Evaluation de modèles pour la simulation de la pollution à proximité des axes routiers, 2009 http://www.lcsqa.org/.

Leitl, B, Bezpalcova, K and Harms, F, 2007. Wind tunnel modelling of the MUST experiment. Proc. 11th International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes, 435-439.

Li, X-X, Liu, C-H, D.Y.C. Leung (2008), Large-eddy simulation of flow and pollutant dispersion in high-aspect ratio urban street canyons with wall model. Boundary-Layer Meteorol. 129, 249-268.

Li, X-X, Liu, C-H, D.Y.C. Leung (2009), Numerical investigations of pollutant transport characteristics inside deep street canyons. Atmospheric Environment 43, 2410-2418.

Liu, C-H and M.C. Barth (2002), Large-eddy simulation of flow and scalar transport in a modelled street canyon. J. Appl. Meteorol. 41, 660-673.

Liu, C-H, M.C. Barth, D.Y.C. Leung (2004), Large-eddy simulation of flow and pollutant transport in street canyons of different building-height-to-street-width ratios. J. Appl. Meteorol. 43, 1410-1424.

Louka P, Belcher SE, and Harrison RG, 2000. Coupling between air flow in streets and the well-developed boundary layer aloft. Atmos Environ 34, 2613–2621.

Macdonald, RW, 1997. Physical modelling study of flow and dispersion around groups of buildings. PhD Thesis, UMIST, Manchester, July, 1997.

Macdonald, R.W., 2000. Modeling the mean velocity profile in the urban canopy layer. Boundary Layer Meteorology 97, 25–45.
Macdonald RW, Griffiths RF, Cheah SC (1997) Field experiments of dispersion through regular arrays of cubic structures. Atmos Environ 31:783–795.

Macdonald, RW, Griffiths, RF and Hall, DJ, 1998. A comparison of results from scaled field and wind tunnel modelling of dispersion in arrays of obstacles. Atmos. Environ., 32, 3845-3862.

Macdonald, R. W., Griffiths, R. F. and Hall, D. J. 1998 An improved method for the estimation of surface roughness of obstacle arrays. Atmos. Environ., 32, No. 11, 1857–1864.

Meinders, E.R. and Hanjalic, K., (2002) Experimental study of the convective heat transfer from in-line and staggered configurations of two wall-mounted cubes. Int. J. of Heat and Mass Transfer, 45, 465-482.

The Met Office (2010) The Met Input Module, ADMS Technical Specification www.cerc.co.uk/environmental-software/assets/data/doc_techspec/CERC_ADMS4_P05_01.pdf.

Moulinec, C., Hunt, J.C.R. and Nieuwstadt, F., (2004) Disappearing wakes and Dispersion in Numerically Simulated Flows Through Tube Bundles. J. Flow. Turb. and Comb., 73(2), 95-116. (In ERCOFTAC Bulletin, 56, 5-9).

Moussafir J., Oldrini O., Tinarelli G, Sontowski J, Dougherty C., (2004) A new operational approach to deal with dispersion around obstacles: the MSS (Micro-Swift-Spray) software suite. Proc. 9th Int. Conf. on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes, vol. 2, 114-118.

Moussafir J., C. Olry, P. Castanier, G. Tinarelli , S. Perdriel, 2010: Applications of the mss (micro-swift-spray) model to long-term regulatory simulations of the impact of industrial plants, 13th International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes, Paris (France) Jun 1-4, 2010.

Nepf H.M. (2012) Flow and Transport in Regions with Aquatic Vegetation. Annu. Rev. Fluid Mech. 44:123–42.

Oke T (1988) Boundary Layer Climates. Routledge.

Oldrini, O., C. Olry, J. Moussafir, P. Armand, and C. Duchenne, 2011: Development of PMSS, the parallel version of Micro-SWIFT-SPRAY, 14th International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes, Harmo'14, Kos (Greece), Oct. 2-6, 2011.

Olesen, H.R., Løfstrøm, P., Berkowicz, R. and Jensen, A.B. (1992) An improved dispersion model for regulatory use: the OML model. In Air Pollution Modelling and its Applications IX, Plenum Press, New York (eds. H. van Dop and G. Kallos).

Pasquill, F. (1962). Atmospheric diffusion. The dispersion of windbourne material from industrial and other sources. D.van Nostrand Company Ltd.

Pasquill, F., Smith, F.B., 1983. Atmospheric Diffusion, third ed. John Wiley & Sons.

Pavageau M, Rafailidis S and Schatzmann M, 2001. A comprehensive experimental data-bank for the verification of urban car emission dispersion models. Int J Environ Pollut., 15, 417–425.

Pavageau, M and Schatzmann, M, 1999. Wind tunnel measurements of concentration fluctuations in an urban street canyon. Atmos. Environ., 33 (1999), 3961-3971.

Perkins, R.J., Carruthers, D.J., Drayton, M.J. and Hunt, J.C.R. (1991) Turbulence and diffusion at density interfaces. Proc. of Int-Seminar on Phase-interface

Phenomena in Multiphase Flow. Dubrovnik. In: Phase-interface Phenomena in Multiphase Flow, 1991, Hemisphere, New York. P 21.

Poggi, D., Porporato, A., Ridolfi, L., Albertson, J.D., Katul, G.G. (2004). The effect of vegetation density on canopy sublayer turbulence. Boundary-Layer Meteorology 111: 565–587.

Porson, A., Clark, P. A., Harman, I. N., Best, M. J. and Belcher, S. (2010a) Implementation of a new urban energy budget scheme in the MetUM. Part I: Description and idealized simulations. Quarterly Journal of the Royal Meteorological Society, 136 (651). pp. 1514-1529.

Porson, A., Clark, P. A., Harman, I. N., Best, M. J. and Belcher, S. (2010b) Implementation of a new urban energy budget scheme into MetUM. Part II: Validation against observations and model intercomparison. Quarterly Journal of the Royal Meteorological Society, 136 (651). pp. 1530-1542

Puttock, J.S. and Hunt, J.C.R. (1979) Turbulent diffusion from sources near obstacles with separated wakes. Part I. An eddy diffusivity model. Atmospheric Environment, 13, 1-13.

Reynolds, R. M. (2006), The Madison Square Garden Dispersion Study (MSG-05) Meteorological Data Report, Technical report, Brookhaven National Laboratory.Robins, A.G. and Apsley, D.D. (2000) Modelling of building effects in ADMS. ADMS Techinical Specification, Paper P16/01N/00.

Robins AG, Carruthers DJ and McHugh CA, **(1997)**: The ADMS building effects module. Int. J. Environment and Pollution, vol. 8, no. 3-6, pp. 708-717, DOI: 10.1504/IJEP.1997.028223.

Robins, AG, Cheng, H and Hayden, P, 2010. DAPPLE wind tunnel studies. Final report to the Home Office, DAPPLE 2009/08 Version 3, 29th March, 2010, available from www.dapple.org.uk.

Robins, AG and Fackrell, JE, 1998. An experimental study of the dispersion of short duration emissions in a turbulent boundary layer. Air Pollution VI (Brebbia, Ratto & Power, Eds), 697-707, WIT Press, Southampton, 1998.

Robins A, Savory E, Scaperdas A and Grigoriadis D, 2002. Spatial variability and source-receptor relations at a street intersection. Water Air Soil Pol. Focus 2:381–393.

Röckle, R., 1990. Bestimmung der stomungsverhaltnisse im Bereich Komplexer Bebauugsstrukturen. Ph.D. thesis, Vom Fachbereich Mechanik, der Technischen Hochschule Darmstadt, Germany.

Rodean, Howard C., 1996, “Stochastic Lagrangian Models of Turbulent Diffusion,” American Meteorological Society, 82 pages.

Rotach, MW, 2002. Overview of the Basel Urban Boundary Layer Experiment – BUBBLE. Proc 4th AMS Urban Env. Conf, Norflok, VA, 2002

Rotach, MW, Gryning, SE, Batchvarova, E, Christen, A and Vogt, R, 2004. Pollutant dispersion close to an urban surface – the BUBBLE tracer experiment. Metorol Atmos Phys, 87, 39–58

Rotach, MW, et al., 2005. BUBBLE: An urban boundary layer meteorology project, Theoretical and Applied Climatology 81, 231–261

Roth, M (2000), Review of atmospheric turbulence over cities, Quarterly Journal of the Royal Meteorological Society 126, Issue 564, pages 941–990

Rudd, A., Robins, A. G., Lepley, J. J. and Belcher, S. (2012) An inverse method for determining source characteristics for emergency response applications. Boundary-Layer Meteorology. ISSN 0006-8314 doi: 10.1007/s10546-012-9712-y (In Press)

Scaperdas, A, Robins, AG and Colville, RN, 2000. Flow visualisation and tracer dispersion experiments at street canyon intersections. I J Env. Pollution, 14, 526-537

Scaperdas A-S, 2000. Modelling flow and pollutant dispersion at urban canyon intersections. PhD thesis, Imperial College of Science, Technology and Medicine, University of London

Schatzmann, M., B. Leitl, D. Hertwig, F. Harms, C. Peeck, G. Patnaik, J. Boris, K. Obenschain, S. Fischer, P. Rechenbach, 2011: An LES-based microscale airborne hazard model. 14th Int. Conf. on Harmo. within Atmos. Disp. Modell. for Regul. Purposes, Kos, Greece.

Scire, J.S., Strimaitis, D.S. and Yamartino, R.J (1999) A user's guide for CALPUFF dispersion model (Version 5). Earth Tech Inc., Concord, MA. See also http://www.src.com/calpuff/calpuff1.htm.

Smethurst, H, 2012. Improving the management and assessment of the health aspects of odour exposure. EngD Thesis, University of Surrey, July, 2012.

Snyder, WH, 1981. Guideline for fluid modeling of atmospheric dispersion. USEPA, EPA-600/8-81/009, April, 1981.

Soulhac, L., Weber, B. and Perkins, R.J. (1998) A practical model for dispersion in a network of streets. In Proceedings of International Conference on Air Pollution Modelling and Simulation (APMS '98), Champ-sur-Marne, France, 1998, pp. 253-262.

Soulhac L, 2000. Modélisation de la dispersion atmosphérique á l'intérieur de la canopée urbaine. PhD thesis, École Centrale de Lyon, France

Soulhac, L., Perkins, R.J., Salizzoni, P., 2008. Flow in a street canyon for any external wind direction. Boundary-Layer Meteorology 126, 365e388.

Soulhac L, Garbero V, Salizzoni P, Mejean P, Perkins RJ (2009) Flow and dispersion in street intersections. Atmos Environ 43:2981–2996

Soulhac L, Salizzoni P, Cierco F-X, Perkins R (2011) The model SIRANE for atmospheric urban pollutant dispersion; part I, presentation of the model. Atmospheric Environment 45, 7379-7395.

Soulhac L, Salizzoni P, Mejean P, Didier D, Rios I (2012) The model SIRANE for atmospheric urban pollutant dispersion; part II, validation of the model on a real case study. Atmospheric Environment 49, 320-337.

Taylor, GI, 1921. Diffusion by continuous movements. Proc. London Math. Soc., 20, 196-210.Taylor, A.M.K. and Whitelaw, J.H. (1984) Velocity Characteristics in the Turbulent Near Wakes of Confined Axisymmetric Bluff Bodies. J. Fluid Mechanics, 139, 391.

Thomson D.J. 1987: Criteria for the selection of stochastic models of particle trajectories in turbulent flows. J. Fluid Mech., 180, 529-556.

Thomson D.J., 1990: A stochastic model for the motion of particle pairs in isotropic high Reynolds number turbulence and its application to the problem of concentration variance. J Fluid Mech, 210, 113-153.

Thomson D.J., 1992: The fluctuations module, ADMS documentation. See also at http://www.cerc.co.uk/environmental-software/assets/data/doc_techspec/CERC_ADMS4_P13_01.pdf

Thykier-Nielsen, S., Deme, S. and Mikkelsen, T. (1999) Description of the Atmospheric Dispersion Module RIMPUFF. Report RODOS(WG2)-TN(98)-02. Risø National Laboratory. See also http://www.rodos.fzk.de/Documents/Public/HandbookV5/Volume3/4_2_6_RIMPUFF.pdf

Tinarelli G., Brusasca G., O.Oldrini, D.Anfossi, S.Trini Castelli, J.Moussafir (2007) “Micro-Swift-Spray (MSS) a new modelling system for the simulation of dispersion at microscale. General description and validation”. Air Pollution Modelling and its Applications XVII, C. Borrego and A.N. Norman eds., Springer, 449-458.

Vardoulakis, S, 2003. Modelling air quality in street canyons: A review, Atmos Environ., 37(2), 155–182.

Vendel, F., G., Lamaison, L. Soulhac, L. Donnat, O. Duclaux and C. Puel, 2010: A new operational modelling approach for atmospheric dispersion in industrial complex area. 13th Int. Conf. on Harmo. within Atmos. Disp. Modell. for Regul. Purposes, Paris, France.

Vendel, F., G., L. Soulhac, P. Mejean, L. Donnat and O. Duclaux, 2011: Validation of the safety Lagrangian atmospheric model (SLAM) against a wind tunnel experiment over an industrial complex area. 14th Int. Conf. on Harmo. within Atmos. Disp. Modell. for Regul. Purposes, Kos, Greece.

Venkatram, A., 1992. Vertical dispersion of ground-level releases in the surface boundary layer. Atmospheric Environment 26 (5), 947-949.

Venkatram, A., Karamchandani, P., Pai, P. and Goldstein, R. (1994), ‘The Development and Application of a Simplified Ozone Modelling System.’ Atmospheric Environment, Vol 28, No 22, pp3665-3678.

Wang X and McNamara KF, 2007. Effects of street orientation on dispersion at or near urban street intersections. J Wind Eng Ind Aerodyn, 95, 1526–1540

Weil, J.C., 1985. Updating applied diffusion models. Journal of Climate and Applied Meteorology 24 (11), 1111-1130.

Williams, M., M. Brown, B. Singh, & D. Boswell, 2004. QUIC-PLUME Theory Guide, LA-UR-04-0561, 22 pp.

Wood CR, Arnold SJ, Balogun AA, Barlow JF, Belcher SE, Britter RE, Cheng H, Dobre A, Lingard JJN, Martin D, Neophytou M, Petersson FK, Robins AG, Shallcross DE, Smalley RJ, Tate JE, Tomlin AS and White IR, 2009. Dispersion experiments in central London: the 2007 DAPPLE project. Bull Am Met Soc 90:955–969.

Wood CR, Lacser A, Barlow JF, Padhra A, Belcher SE, Nemitz E, Helfter C, Famulari D, Grimmond CSB, 2010. Turbulent Flow at 190m Height Above London During 2006–2008: A Climatology and the Applicability of Similarity Theory. Boundary-Layer Meteorology 137: 77-96.

Xie, Z-T (2011) Modelling street-scale flow and dispersion in realistic winds – towards coupling with mesoscale meteorological models. Boundary-Layer Meteorology 141: 53-75.

Xie Z-T, Hayden P, Robins AG, Voke PR (2007) Modelling extreme concentrations from a source in a turbulent flow over a rough wall. Atmos Environ 41:3395–3406.

Xie, Z-T, Castro, IP (2009) Large-eddy simulation for flow and dispersion in urban streets. Atmospheric Environment 43: 2174-2185.

Xie, Z-T, Coceal, O, Castro, IP (2008) Large-eddy simulation of flows over random urban-like obstacles. Boundary-Layer Meteorology 129: 1-23.

Yee, E and Biltoft, CA, 2004. Concentration fluctuation measurements in a plume dispersing through a regular array of obstacles. Boundary Layer Meteor., 111, 363-415.