



Modelling air quality in street canyons: a review

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Abstract

High pollution levels have been often observed in urban street canyons due to the increased traffic emissions and reduced natural ventilation. Microscale dispersion models with different levels of complexity may be used to assess urban air quality and support decision-making for pollution control strategies and traffic planning. Mathematical models calculate pollutant concentrations by solving either analytically a simplified set of parametric equations or numerically a set of differential equations that describe in detail wind flow and pollutant dispersion. Street canyon models, which might also include simplified photochemistry and particle deposition–resuspension algorithms, are often nested within larger-scale urban dispersion codes. Reduced-scale physical models in wind tunnels may also be used for investigating atmospheric processes within urban canyons and validating mathematical models.

A range of monitoring techniques is used to measure pollutant concentrations in urban streets. Point measurement methods (continuous monitoring, passive and active pre-concentration sampling, grab sampling) are available for gaseous pollutants. A number of sampling techniques (mainly based on filtration and impaction) can be used to obtain mass concentration, size distribution and chemical composition of particles. A combination of different sampling/monitoring techniques is often adopted in experimental studies. Relatively simple mathematical models have usually been used in association with field measurements to obtain and interpret time series of pollutant concentrations at a limited number of receptor locations in street canyons. On the other hand, advanced numerical codes have often been applied in combination with wind tunnel and/or field data to simulate small-scale dispersion within the urban canopy.

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1. Introduction

1.1. Urban air quality and traffic

Urban air pollution was originally considered as a local problem mainly associated with domestic

heating and industrial emissions, which are now controllable to a great extent. Despite significant improvements in fuel and engine technology, present day urban environments are mostly dominated by traffic emissions (Fenger, 1999; Colville et al., 2001). It is now generally recognised that many of the substances directly emitted by vehicles in the ambient air or indirectly produced through photochemical reactions represent a serious hazard for human health (Hoek et al., 2000; Nyberg et al., 2000; Dab et al., 2001).

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The main traffic-related pollutants are CO, NO_x, hydrocarbons, and particles. CO is an imperfect fuel combustion product. Combustion also produces a mixture of NO₂ and NO, of which more than 90% is in the form of NO. A wide range of unburned and chemically transformed hydrocarbons (e.g. benzene, toluene, ethane, ethylene, pentane, etc.) is emitted by motor vehicles through a number of different processes (e.g. evaporation, fuel tank displacement, oil seep, etc.). Finally, particles of condensed carbonaceous material are emitted mainly by diesel and poorly maintained petrol vehicles.

Atmospheric pollutants are responsible for both acute and chronic effects on human health (WHO, 2000). CO is an asphyxiating pollutant that reduces the ability of blood to carry oxygen to the different organs (Burnett et al., 1998). Therefore, short-term exposure to high CO concentrations might cause an acute health impact. On the other hand, pollutants like benzene have a cumulative effect on human health. Long-term exposure to high benzene levels increases the risk for an individual to suffer from cancer (Cicoletta, 1997). Furthermore, there are gases like NO₂ that are responsible for both short- and long-term health effects. Depending on the effects related to each substance, atmospheric pollutants are regulated with respect to different exposure times. For example, in the European air quality guidelines, standards are set for benzene as 1 year averages, for CO as 8 h averages, and for NO₂ as both 1 h and 1 year averages (European Commission, 1999, 2000).

Particulate matter with aerodynamic diameter below 10 µm (PM₁₀) and especially the finer fraction with aerodynamic diameter below 2.5 µm (PM_{2.5}) was found to associate with increased daily mortality and asthma (Dockery and Pope, 1994; Anderson et al., 1992; Harrison and Yin, 2000). Furthermore, ultrafine particles (i.e. aerodynamic diameter <100 nm) are likely to represent a major health risk (Seaton et al., 1995). Nevertheless, current European legislation addresses only total PM₁₀ as 24 h and 1 year averages, while US legislation regulates both PM₁₀ and PM_{2.5} as 3 year averages (EPA, 1996).

Although roadside concentrations differ significantly from background levels, all outdoor environments are subject to the same regulatory standards for ambient air quality. This is due to the assumption that pollutant concentrations observed at a single or a few permanent monitoring stations within a city are representative of the exposure of the entire urban population.

In urban environments and especially in those areas where population and traffic density are relatively high, human exposure to hazardous substances is expected to be significantly increased. This is often the case near busy traffic axis in city centres, where urban topography and microclimate may contribute to the creation of poor air dispersion conditions giving rise to contamination

hotspots. High pollution levels have been observed in *street canyons*, which is a term frequently used for urban streets flanked by buildings on both sides. Within these streets, pedestrians, cyclists, drivers and residents are likely to be exposed to pollutant concentrations exceeding current air quality standards.

1.2. Air quality monitoring

Nowadays, automated monitoring networks operate in many European cities providing detailed air quality information on a regular basis. There are several techniques available for monitoring gaseous pollutants (e.g. continuous monitoring using standard gas analysers, diffusive and pumped sampling using tubes filled with an appropriate adsorbent, grab sampling using canisters) and particulate matter (e.g. filtration and impaction). Each one of them can be associated with a number of advantages and disadvantages that make it suitable or not for a specific application.

The response time, which is the time over which the sample is taken, is one of the major factors that determine the suitability of a method. Standard gas analysers are sufficiently sensitive and fast to give real time (i.e. typical response time: 1–2 min) measurements of CO, NO_x and O₃ concentrations. The results can then be averaged over short time periods and compared to the regulatory standards.

Diffusive samplers have a relatively long response time (e.g. typically from one/two days to four weeks), which makes them less suitable for observing atmospheric pollutants responsible for short-term health effects. On the other hand, long response times might be preferable when sampling substances like benzene, whose impact on human health is due to cumulative exposure. In these cases, peak concentrations are of minor concern and therefore diffusive samplers appear to be the ideal choice (Brown et al., 1999; Cocheo et al., 2000; Skov et al., 2001; Vardoulakis et al., 2001). Furthermore, diffusive samplers are portable devices and do not need electrical power supply, which makes them very suitable for spatial distribution measurements (e.g. vertical profiles within canyons), air quality mapping, personal exposure studies, and detection of long-term pollution trends.

The total number of air quality monitoring stations or sampling locations within a city is limited by practical constraints. Since pollutant concentrations might vary with a factor of 5 from a street canyon to an urban background area (Palmgren and Kemp, 1999), the selection of monitoring/sampling locations becomes fundamental. In addition, one should be cautious when comparing absolute values from different cities. The data are often based on one or few monitoring stations placed at critical sites and thus represent microenvironments rather than large urban areas (Fenger, 1999). In

general, monitoring stations and/or samplers should be located near places of expected air pollution hotspots, but also must be reasonable with respect to population exposure over the averaging times associated with the regulatory values. Permanent air quality stations within a city may be classified into two broad categories: (a) the roadside and (b) the urban background stations. Roadside stations are usually located on the pavement of busy streets, avenues or intersections, within few metres distance from the roadway and with their sampling head at 1.5–3 m height above ground. On the other hand, background stations are placed in parks or other urban locations away from road traffic.

1.3. Air quality modelling

The impact of air pollution on urban environments has become an important research issue (Georgii, 1969; Oke, 1988; Bitan, 1992), leading to numerous modelling studies related to the influence of buildings and other urban structures on pollutant accumulation/dissipation patterns. The main features of pollutant dispersion within urban canyons are well understood through the pioneering work of Johnson et al. (1973), Dabberdt et al. (1973), Hotchkiss and Harlow (1973), Nicholson (1975) and others.

Dispersion models are now widely used for assessing roadside air quality by providing predictions of present and future air pollution levels as well as temporal and spatial variations (Sharma and Khare, 2001). When used in a knowledgeable way, they can be very useful in giving insights into the physical and chemical processes that govern the dispersion and transformation of atmospheric pollutants. This article provides an overview of modelling techniques and available software for assessing air quality in street canyons. Within this framework, relevant experimental and theoretical studies are also briefly discussed.

2. Street canyon characteristics

2.1. Canyon geometry

The term *street canyon* ideally refers to a relatively narrow street with buildings lined up continuously along both sides (Nicholson, 1975). However, the same term has been used to refer to larger streets, also called avenue canyons. In the real world, a broader definition of the term has been applied, including urban streets that are not necessarily flanked by buildings continuously on both sides, allowing thus for some openings on the walls of the canyon.

The dimensions of a street canyon are usually expressed by its *aspect ratio*, which is the height (H) of the canyon divided by the width (W). A canyon might

be called *regular*, if it has an aspect ratio of approximately equal to 1 and no major openings on the walls. An avenue canyon may have an aspect ratio below 0.5, while a value of 2 may be representative of a *deep canyon*. Finally, the length (L) of the canyon usually expresses the road distance between two major intersections, subdividing street canyons into *short* ($L/H \approx 3$), *medium* ($L/H \approx 5$), and *long* canyons ($L/H \approx 7$). Urban streets might be also classified in *symmetric* (or *even*) canyons, if the buildings flanking the street have approximately the same height, or *asymmetric*, if there are significant differences in building height.

2.2. Wind flow

The climate of street canyons is primarily controlled by the micro-meteorological effects of urban geometry rather than the mesoscale forces controlling the climate of the boundary layer (Hunter et al., 1992). A clear distinction should be made between the *synoptic* above roof-top wind conditions and the *local* wind flow within the cavity of the canyon (Fig. 1). Depending on the synoptic wind (or *free-stream* velocity), three main dispersion conditions can be identified: (i) low wind conditions, for synoptic winds lower than 1.5 m/s, (ii) perpendicular or near-perpendicular flow for synoptic winds over 1.5 m/s blowing at an angle of more than 30° to the canyon axes, and (iii) parallel or near-parallel flow for winds over 1.5 m/s blowing from all other directions. In the case of perpendicular flow, the up-wind side of the canyon is usually called *leeward*, and the down-wind *windward*.

The emphasis has often been on the two-dimensional nature of the flow, studying vertical cross-sections at mid-canyon level. When the above roof flow is perpendicular to the canyon and the wind speed is > 1.5 –2 m/s, flow may be described in terms of three regimes, depending on the dimensions of the street (Oke, 1988) (Fig. 2): (a) *isolated roughness* flow, (b) *wake interference* flow, and (c) *skimming* flow. For wide canyons ($H/W < 0.3$), the buildings are well spaced and act essentially as isolated roughness elements, since the air travels a sufficient distance down-wind of the first building before encountering the next obstacle. As buildings become more closely spaced ($H/W \approx 0.5$), the disturbed air flow has insufficient distance to readjust before encountering the down-wind building, resulting in *wake interference* flow. In the case of regular canyons ($H/W \approx 1$), the bulk of the synoptic flow skims over the canyon producing the *skimming flow*, which is characterised by the formation of a single vortex within the canyon (Hunter et al., 1992).

From a three-dimensional point of view, a reflection of the wind off the windward wall of the canyon should be ideally observed in the case of skimming flow (Nakamura and Oke, 1988; Johnson and Hunter,

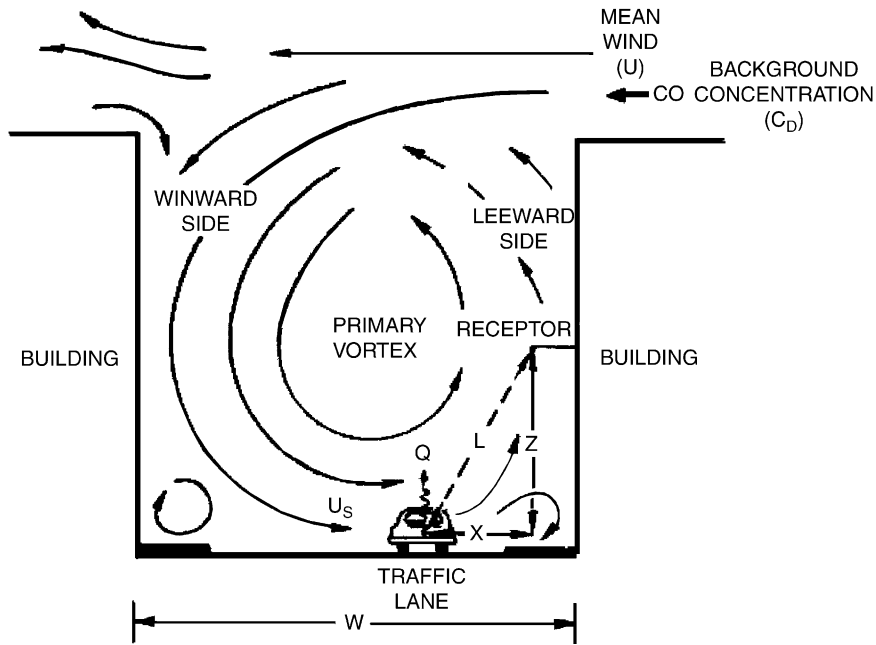


Fig. 1. Pollutant dispersion in a regular street canyon (Dabberdt et al., 1973).

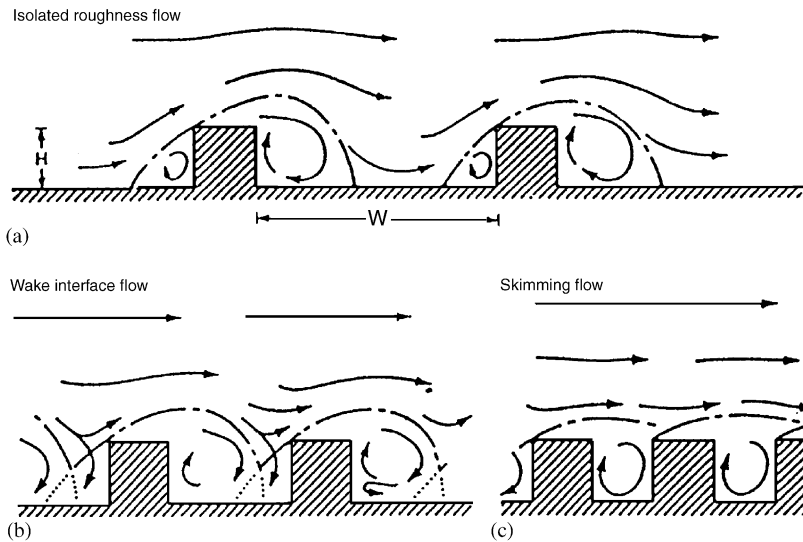


Fig. 2. Perpendicular flow regimes in urban canyons for different aspect ratios (Oke, 1988).

1999). For oblique roof-level winds, this reflection may induce a spiral wind flow through the canyon. Other complex channelling effects might be produced for winds parallel to the street axis. Additional low-pressure areas and wind circulation is created near intersections, resulting in horizontal corner vortices. In relatively short canyons, corner vortices might be strong enough to inhibit a stable vortex perpendicular to the street in the mid-section. This ventilation effect fades with increasing street length (Theurer, 1999).

The strength of the wind vortices inside the canyon mainly depends on wind speed at roof-top level. However, the local wind flow is also affected by the mechanical turbulence induced by moving vehicles (Eskridge and Rao, 1986) or created near urban roughness elements within the street (e.g. trees, kiosks, balconies, slanted building roofs, etc.) (Hoydysh and Dabberdt, 1994; Theurer, 1999). Furthermore, the shape and strength of the wind vortices might also be affected by the atmospheric stability and other thermal effects

induced by the differential heating of the walls and/or the bottom of the canyon (Sini et al., 1996; Kim and Baik, 2001).

In relatively deep canyons ($H/W > 1.3$), the main wind vortex is usually displaced towards the upper part of the cavity, with almost stagnant air below (DePaul and Sheih, 1986). As the aspect ratio increases ($H/W \approx 2$), a weak counter-rotating secondary vortex may be observed at street level (Pavageau et al., 1996). For even higher aspect ratios ($H/W \approx 3$), a third weak vortex might be also formed (Jeong and Andrews, 2002). In most cases, small weak vortices occupy the bottom side corners of the canyon.

Depending on the wind direction, asymmetric canyons may be sub-divided into two categories: (i) *step-up* canyons, when the down-wind building is higher than the up-wind building, and (ii) *step-down* canyons, when the down-wind building is lower than the up-wind building. In these cases, mid-section wind vortices might be displaced or reversed within the cavity.

2.3. Pollutant dispersion

The dispersion of gaseous pollutants in a street canyon depends generally on the rate at which the street exchanges air vertically with the above roof-level atmosphere and laterally with connecting streets (Riain et al., 1998). Skimming flow, a feature of regular canyons, provides minimal ventilation of the canyon and is relatively ineffective in removing pollutants (Hunter et al., 1992). Field measurements (DePaul and Sheih, 1985; Qin and Kot, 1993) show increased concentrations of traffic-related pollutants on the leeward side of the canyon, and decreasing concentrations along with height above the ground on both sides of the street. The increased leeward concentrations are due to the accumulation of pollutants locally advected by the large wind vortex that covers most of the canyon. Minor pollution hotspots might be also created in small cavities where additional recirculation phenomena can take place.

Street-level cross-road gradients observed in wind tunnel experiments (Hoydysh and Dabberdt, 1988) for perpendicular wind conditions indicate that concentrations are generally a factor of two greater for the leeward than for the windward side, except for step-down canyons where windward concentrations are slightly greater than leeward. Concentrations are generally lower in the step-up canyons relative to the even and step-down notches.

Flow visualisation experiments have shown that the strength of the canyon vortices varies. As a result, pollutants are periodically flushed out of the canyon (Pavageau et al., 1996). In relatively long canyons without connecting streets, field measurements have shown that maximum street-level concentrations are

more likely to occur when the synoptic wind is parallel to the street axis (Vardoulakis et al., 2002a). In that case, the accumulation of emissions along the line source outweighs the ventilation induced by the parallel winds (Soulhac et al., 1999; Dabberdt and Hoydysh, 1991).

Low synoptic winds create a well-known meteorological situation that favours air pollution built-up in urban areas (Qin and Kot, 1993; Vignati et al., 1996; Jones et al., 2000). There is evidence that when the synoptic wind speed is below about 1.5 m/s, the wind vortex within the canyon tends to disappear and the air stagnates in the street (DePaul and Sheih, 1986). In that case, the mechanical turbulence induced by moving vehicle as well as the atmospheric stability conditions might play a significant role in the dispersion of traffic-generated pollutants.

Fine and especially ultrafine particles are expected to disperse in the air like gases. The larger-sized particles, however, are greatly affected by gravity and thus have a shorter residence time in the air (Chan and Kwok, 2000). For this reason, the coarse fraction of the total suspended particles (TSP) exhibits larger vertical concentration gradients than those usually observed for gases or fine particles.

2.4. Pollutant transformation

Due to the very short distances between sources and receptors, only very fast chemical reactions have a significant influence on the measured concentrations within street canyons (Berkowicz et al., 1997). For this reason, most traffic-related pollutants (e.g. CO and hydrocarbons) can be considered as practically inert species within these distances. This is not the case either for NO₂, which dissociates extremely fast in the presence of light, or for NO, which also reacts very fast with O₃ (Palmgren et al., 1996). The time scales of these chemical reactions are of the order of tens of seconds, thus comparable with residence times of the pollutants in a street canyon.

It is expected that the relationship between relatively stable chemical species emitted by vehicles would not vary significantly within urban streets. This is very helpful for epidemiological studies, because a single or only few indicators can be identified for assessing population exposure to traffic-related air pollution (Kingham et al., 2000). A thorough discussion on the formation of photochemical pollutants in urban and rural areas is given elsewhere (Sillman, 1999; Jenkin and Clemitshaw, 2000).

2.5. Population exposure

From a population exposure point of view, air quality in street canyons is of a major importance, since the highest pollution levels and the larger targets of

impact are often concentrated in this kind of streets (Hertel et al., 2001). The so-called *canyon effect* (i.e. the reduced natural ventilation in urban streets) results in greater health impacts (e.g. indicated by an increased number of respiratory hospital admissions) and damage costs for the exposed population (Spadaro and Rabl, 2001).

Personal exposure can be calculated as the product of the pollutant concentration and time spent in a specific *microenvironment*, which is defined as a confined space (e.g. bedroom, office, car, parking, pavement, etc.) where pollutant concentrations are assumed to be uniform (Colls and Micallef, 1997). The total personal exposure will be then the sum of all such products. However, the assumption of spatial uniformity of air pollution might be erroneous for certain microenvironments like street canyons, where strong spatial concentration gradients are often observed. In these cases, exposure calculations should be refined by subdividing microenvironments into *sub-microenvironments*, taking into account pollution hot spots and refined human breathing zones (e.g. for residents, pedestrians, cyclists, drivers, etc.).

Relatively few examples of this approach can be found in literature. In a study attempting to quantify residential exposure to exhaust gases in Oslo (Larssen et al., 1993), a correction coefficient was introduced to account for changes in ambient concentrations with height over street level. Other authors (Croxford and Penn, 1998) have suggested that a side of the street factor should be introduced if the prevailing wind direction is perpendicular or near-perpendicular to the street axis.

3. Street canyon modelling

There is a plethora of dispersion models specially developed for or simply used in street canyon applications. They can be useful in air quality and traffic management, urban planning, interpretation of monitoring data, pollution forecasting, population exposure studies, etc. Although there are no clear-cut distinctions between different categories, models might be classified into groups according to their physical or mathematical principles (e.g. reduced-scale, box, Gaussian, CFD) and their level of complexity (e.g. screening, semi-empirical, numerical). Some of these (often overlapping) categories and corresponding models are presented in Table 1.

3.1. Operational models

3.1.1. Gaussian plume models

These are sets of equations describing the three-dimensional concentration field generated usually by a

point source. They assume that the concentrations from a continuously emitting source are proportional to the emission rate, inversely proportional to the wind speed, and that the time averaged pollutant concentrations horizontally and vertically are well described by Gaussian (i.e. bell-shaped) distributions (Bouhel et al., 1994). In its simplest form, the Gaussian plume model assumes that there are no chemical or removal processes taking place and that pollutant material reaching the ground or the top of the mixing layer as the plume grows is reflected back towards the plume centreline.

Gaussian plume models rely on the appropriate selection of the plume spread sigma functions (in both the horizontal and vertical sense), which are generally expressed in terms of Pasquill atmospheric stability classes or Monin–Obukhov similarity theory parameters (Zannetti, 1990). Models using the latter approach dispose of height dependent sigma functions and are known as *second generation* Gaussian plume models (Carruthers et al., 1994).

Apart from industrial applications (i.e. point sources), specially designed Gaussian plume models can be used to calculate pollutant concentrations over urban agglomerations (i.e. area sources) and in the vicinity of highways (i.e. line sources). Gaussian models are not directly applicable to small-scale dispersion within the urban canopy, since they treat buildings and other obstacles only via a surface roughness parameterisation (Scaperdas, 2000). Nevertheless, in some cases, they include specialised *modules* for street canyons. This is the case of ADMS-Urban (Owen et al., 1999), a second generation urban-scale dispersion model that includes a street canyon module nested within the core Gaussian code.

3.1.2. CALINE4

CALINE4, the latest version of the CALINE series of pollutant dispersion models, is one of the most validated models available for assessing the impact of vehicle traffic on roadside air quality (Benson, 1984). It has been widely used in scientific and engineering applications mainly concerning highway development and management (Jones et al., 2000). Although CALINE4 is able to handle canyon or intersection situations, it has been used in relatively few urban air quality studies.

The model uses Gaussian plume theory to simulate the dispersion of pollutants emitted from a line source. This is divided in a series of elements, which are modelled as equivalent finite line sources located normal to the wind direction. The region directly over the road, called the *mixing zone*, is treated as a zone of uniform emission and turbulence. Within the mixing zone, vehicle induced turbulence (both mechanical and thermal) is taken into account (Benson, 1992).

The street canyon algorithm devised by Turner (1970) computes the effect of single or multiple horizontal

Table 1
Classification of commonly used dispersion models (some models appear only with the name of their authors)

Statistical	Parametric (operational)						Numerical		Lagrangian stochastic particle
	Receptor	Semi-empirical			Eulerian		Urban scale	MEMO	
		Screening	Box	Street canyon	Gaussian	Microscale (CFD)			
Derwent and Middleton (1996)	COPREM	CAR International	STREET-SRI	CPBM	ADMS-Urban	PHOENICS	GEM		
	Karim and Ohno (2000)	AEOLUS Screen	MAPS	OSPM	INDIC AIRVIRO	FLUENT	METRAS	Addison et al. (2000)	
Stedman et al. (2001)		AEOLIUSQ	STREET BOX	AEOLIUS	TNO-Traffic	STAR-CD	CALGRID	Jicha et al. (2000)	
		STREET	Nicholson (1975)	SLAQ	CAR-FMI	CFX-TASCflow		Xia and Leung (2001a, b)	
	UK DMRB		OMG		PROKAS-V	PANCHE			
			Hotchkiss and Harlow (1973)		HIWAY-2	MERCURE			
					CALINE4	CHENSI			
					APRAC	MISKAM			
					PUFFER	MIMO			
						MITRAS			
						ADREA-HF			
						FloVENT			

reflections of the plume on the walls of the canyon. In this case, the road axis and the wind velocity are assumed to be parallel to the horizontal topographic boundary (i.e. the walls of the canyon), resulting in equal concentrations on both sides of the street.

3.1.3. TNO and CAR

TNO-Traffic is a Gaussian plume-type model that describes the dispersion of traffic exhausts (Eerens et al., 1993). It is based on an extensive programme of wind tunnel experiments which covered a great number of different street configurations, including urban canyons and intersections (van den Hout et al., 1994). In this model, the traffic is represented by line sources divided into series of small point sources.

CAR (or CAR-International) is a simplified version of the same model, in which only the most representative street configurations were categorised (den Boeft et al., 1996). For each street type (e.g. highway, urban road, etc.) a source–receptor relationship is specified as a function of the distance between the receptor and the street axis. CAR uses annual average wind speeds and assumes that there is no prevailing wind direction. Thus, the user obtains the same yearly averages and percentiles on either side of the street. In all cases, the effect of trees and moving vehicles on street-level wind velocity is taken into account. This model should not be confused with CAR-FMI (Härkönen et al., 1995), which is a Gaussian line source model for calculating pollution from road networks.

3.1.4. STREET–SRI

Johnson et al. (1973) used a single box model, together with some simplified assumptions concerning initial dispersion and car induced turbulence, to derive a street canyon sub-model usually called STREET or SRI (i.e. Stanford Research Institute), which formed part of a multipurpose urban diffusion model for inert pollutants (APRAC). It is based on the assumption that concentrations of the pollutant occurring on the road-side consist of two components, the urban background concentration and the concentration component due to vehicle emissions generated within the specific street. Then, it calculates pollutant concentrations on both sides of the street, taking into account the height and distance of the simulated receptor from the kerb.

On the leeward side of the canyon, concentrations are assumed to be inversely proportional to the distance between the line source and the receptor point. On the windward side, the original expression given by Johnson et al. (1973) was revised by Dabberdt et al. (1973) to account for vertical decrease of concentrations due to entrainment of fresh air through the top of the canyon. For parallel or near-parallel synoptic winds, the average of the leeward and windward values may give the pollutant concentration on both sides of the street,

although the model is not specifically designed for this situation. STREET–SRI was parameterised using data from a regular street canyon and for this reason it might need re-calibration before being applied to other canyon geometries.

3.1.5. CPBM

The Canyon Plume Box Model (Yamartino and Wiegand, 1986) combines a Gaussian plume model for the direct impact of pollutants emitted in the street, with a box model that accounts for the additional impact of pollutants trapped within the wind vortex formed inside the canyon. The wind flow in the canyon is reproduced using the methodology proposed by Hotchkiss and Harlow (1973) for the two transverse components of the wind velocity and a logarithmic expression for the longitudinal component. An empirical model that takes into account wind generated turbulence as well as thermal effects induced by solar radiation and moving vehicles is used to calculate the turbulent sigma parameters representing the standard deviation of flow velocities about the mean flow.

The plume generated inside the canyon is divided into three segments, which are assumed to follow straight line trajectories and disperse according to Gaussian plume formulae. The impact resulting from the recirculation component is calculated from the consideration of the mass budget inside the canyon. On the leeward side of the street, the total impact is calculated by adding the direct plume to the recirculated fraction. On the windward side, where the only contribution arises from the recirculation component, the dilution of the concentrations due to the entrainment of fresh air is also taken into account. For winds parallel to the street axis or for very low wind speeds, a simpler plume model is used.

3.1.6. OSPM and AEOLIUS

AEOLIUS (Buckland, 1998) is based on concepts and techniques previously used for the development of the Operational Street Pollution Model (Hertel and Berkowicz, 1989a), which was evolved from the CPBM. AEOLIUS and OSPM are semi-empirical models that calculate concentrations of exhaust gases on both sides of a canyon assuming three different contributions: (a) the contribution from the direct flow of pollutants from the source to the receptor, (b) the recirculation component due to the flow of pollutants around the vortex generated within the recirculation zone of the canyon, and (c) the urban background contribution. A Gaussian plume algorithm is used for the calculation of the direct contribution and a simple box model for deriving the recirculation component.

The vortex is formed inside the canyon, if the synoptic wind is not parallel to the street axis. The length of the vortex (along the wind direction) is 2 times the up-wind

building height. For synoptic winds below 2 m/s, the length of the vortex decreases with the wind speed (Berkowicz, 2000a). The width of the recirculation zone cannot exceed the width of the canyon in any case. The relation between street- and roof-level winds in the canyon is given by a logarithmic relationship that takes into account the surface roughness length, the height of initial dispersion of car exhausts and the synoptic wind direction.

On the leeward side of the street, concentrations are calculated as the sum of the direct and recirculation contributions, while on the windward side only the direct contribution of emissions generated outside the recirculation zone are taken into account. If the recirculation zone extends throughout the whole canyon, then the windward concentrations are calculated from only the recirculation component. For near-parallel flow, emissions from outside the recirculation zone may contribute to the leeward concentrations. When the wind speed is near zero or parallel to the street axis, the concentrations on both sides of the canyon become equal. Finally, the mechanical turbulence in the street due to the wind and vehicle traffic is empirically derived.

AEOLIUS (the Full version) is based on the same formulation as OSPM. Nevertheless, some discrepancies between predictions from the two models cannot be excluded, due to differences in coding, parameterisation and data pre-processing techniques. There are also two screening versions of AEOLIUS, namely AEOLIUS Screen and AEOLIUSQ Emission, made available by the UK Met Office.

3.1.7. Receptor models

The models described in the previous sections may be also defined as *source-oriented* models. Such models rely on the use of best available emission estimates and meteorological data to predict pollutant concentrations at various roadside locations. An alternative approach is the *receptor-oriented* modelling, which is based on the detailed analysis of the pollutant collected at one or more monitoring sites. This analysis, also called *source apportionment* or *chemical mass balance* (Gordon et al., 1984), attempts to determine which sources contributed to the concentration measured at the receptor point. If the pollutant of interest is chemically inert (e.g. CO), there is no way to distinguish between different sources. But if the pollutant consists of a variety of chemical species (e.g. particulate matter), then from its chemical composition one can make inferences about the sources.

Receptor-oriented models, such as the Constrained Physical Receptor Model (CPRM) (Wählin et al., 2001), are mostly used to test the predictions made by source-oriented models as well as the accuracy of the emission estimates that are used in them (Karim and Ohno, 2000).

3.2. CFD models

Computational fluid dynamics (CFD) modelling is a general term used to describe the analysis of systems involving fluid flow, heat transfer and associated phenomena (e.g. chemical reactions) by means of computer-based numerical methods. It is a powerful modelling technique spanning a wide range of industrial and more recently environmental and biomedical applications (Gosman, 1999). What distinguishes CFD from other Eulerian models is their capability to deal with complex shaped walls and other boundary conditions (e.g. in aircraft and automobile design) using flexible fine-scale grids. Furthermore, they usually include advanced turbulence treatment schemes, which make them suitable for small-scale pollutant dispersion applications.

CFD codes are structured around numerical algorithms that can tackle fluid flow problems. In order to provide easy user access, most commercial CFD packages include sophisticated input and output interfaces. Hence, they contain three main elements: (I) The pre-processor, which serves to input problem parameters, generate the grid of the computational domain, select the physical and chemical phenomena that need to be treated, define the fluid properties, and finally specify the appropriate boundary conditions. (II) The solver, which first approximates numerically the unknown flow variables, then discretises the governing flow equations using these approximations, and finally solves the resulting system of algebraic equations. (III) The post-processor, which displays the grid and geometry of the domain, plots vectors (e.g. wind velocity) and contours (e.g. pollutant concentration) and may even provide animation facilities for dynamic result display.

3.2.1. Physical and numerical principles

CFD modelling is based on the numerical solution of the governing fluid flow and dispersion equations, which are derived from basic conservation and transport principles: (a) the mass conservation (*continuity*) equation, (b) the three momentum conservation (*Navier–Stokes*) equations in x, y, z , and (c) the transport equation for pollutant concentration. The equations of state (obtained through the thermodynamic equilibrium assumption) and the Newtonian model of viscous stresses are also enlisted to close the system numerically. Initial and boundary conditions have to be specified. Furthermore, atmospheric turbulent processes need to be modelled. Existing turbulence models can be classified in two broad categories: (I) the classical models based on Reynolds Averaged Navier–Stokes (RANS) flow equations (e.g. the $k-\epsilon$ model, which is by far the most used and validated); (II) the Large Eddy Simulation (LES) models, which are computationally very

demanding and therefore mainly used in research applications (Versteeg and Malalasekera, 1995).

There are three different streams of numerical solution techniques: finite difference, fine element, and spectral methods. The main differences between them are associated with the way in which the flow variables are approximated and with the discretisation processes. The finite volume method, which was originally developed as a special finite difference formulation, is now the most well established and thoroughly validated method (it is central to most popular CFD codes: PHOENICS, FLUENT, STAR-CD).

3.2.2. Applications and codes

When pollutant dispersion is examined within a street canyon, the computational domain should be sufficiently extended to stabilise the air inflow and outflow through the geometrical boundaries of the area. The relief of the buildings (e.g. due to the presence of balconies) or the street (e.g. due to the presence of vegetation, parked cars, etc.) can be taken into account by introducing a roughness coefficient for each surface of the domain. The space discretisation is usually not uniform, since a higher resolution is required near the canyon walls and the roadway. Finally, fields of pollutant concentrations, wind velocity and other physical quantities (e.g. turbulent kinetic energy and eddy diffusivity) may be reproduced.

The commercially available general-purpose CFD codes PHOENICS, FLUENT, STAR-CD, CFX-TASCflow and Fluidyn-PANACHE have been used in a number of street canyon applications. Other numerical models like MERCURE (Carissimo et al., 1995), CHENSI (Levi Alvares and Sini, 1992) and MISKAM (Eichhorn, 1995) were specially designed to simulate pollutant dispersion at local scale. MISKAM was used to create a database of numerical three-dimensional simulations that was integrated in a screening model called STREET (Petit et al., 2000). Furthermore, the street canyon module PROKAS-B, which forms part of the Gaussian urban-scale model PROKAS-V, was also based on dimensionless concentrations calculated using a version of MISKAM. Finally, the microscale models MIMO and MITRAS were also specially designed for street canyon applications and nested within the mesoscale MEMO and METRAS, respectively (Ehrhard et al., 2000).

3.3. Reduced-scale models

The reduced-scale (or *physical*) models are based on the *similarity principle*, which means that by reducing the geometrical scale of a given flow domain and adjusting the reference parameters (e.g. flow velocity), the original full-scale conditions can be reproduced. Reduced-scale

modelling can be carried out in a wind tunnel or a water tank facility. Although wind tunnels have been more widely used for simulating pollutant dispersion than water tanks, the same principles and considerations apply to both the methods.

Three monitoring techniques have been often used in wind tunnel experiments: (a) flow visualisation, which helps to explore the range of possible flow and dispersion patterns obtained for different building arrangements, (b) tracer dispersion, which is used to quantify concentrations at receptor locations within the canyon, and (c) Laser Doppler Anemometry (LDA), which is used to study in more detail the patterns observed during flow visualisation experiments.

Although wind tunnel modelling might involve scaling difficulties, it can efficiently approximate real atmospheric conditions in urban streets. Furthermore, it allows isolating and studying separately each one of the phenomena involved in microscale pollutant dispersion. Reduced-scale modelling has often been used as a complementary tool to numerical modelling and been proved especially useful in model development and validation (Baker and Hargreaves, 2001). Nevertheless, differences between wind tunnel and full-scale experimental data should be carefully considered when validating numerical models (Schatzmann et al., 1999).

3.4. Model requirements

Dispersion model predictions are in most cases a function of meteorology, street geometry, receptor location, traffic volumes and emission factors. The acquisition and pre-processing of these data is an important part of any modelling study, since the performance of a model greatly depends on the quality of the inputs.

3.4.1. Traffic data

Detailed traffic information, including traffic volumes, fleet composition (e.g. ratio of light/heavy duty vehicles) and average vehicle speeds, is normally required for running street canyon models. Part of this information (e.g. traffic volumes and average vehicle speeds) might be obtained from automatic detectors permanently or temporarily operating in the street of interest. The vehicle fleet composition, however, is rarely available for a specific location and time period and for this reason has often to be estimated from on site spot measurements. At least few manual traffic counts should be always taken to assure the quality of data obtained from automatic traffic networks.

3.4.2. Emissions

All street canyon models require vehicle *emission factors* (e.g. g/km per single vehicle) or *emission rates* (e.g. g/km per hour) as input, although some operational

models (e.g. CAR) might include default national emission factors. In certain models (e.g. AEOLIUS), separate emission factors for small and large vehicles need to be specified. The emission rates in a street can be derived from the traffic volumes and the composite emission factors of the pollutants. A number of methodologies and models may be applied to determine the appropriate fleet-average emission factors.

The CORINAIR working group (sponsored by the European Commission) developed a methodology for calculating emissions, including appropriate emission factors, from road traffic (Eggleston et al., 1993). The methodology was transformed into the computer program COPERT (Ntziachristos and Samaras, 1997, 2000). The same methodology was adopted by ADEME (1998) to develop IMPACT, a road traffic emission model which quantifies fuel consumption and atmospheric releases of a specified vehicle fleet in a given year in France. Emissions are calculated for two vehicle operating modes: *hot* and *cold start*. The required input parameters are traffic composition, average vehicle speed, length and slope of the road segment of interest. In addition, the month of the year is used to estimate average ambient temperatures, which are further used for calculating evaporative and cold running emissions. The model provides default values for the average travelling distance and the fraction of this distance run with a cold engine in France.

The US EPA has developed and regularly revised MOBILE, which is also a mobile source emission factor model. MOBILE distinguishes moving vehicles into three operating modes: *cold start*, *hot stabilised* and *hot start*. The model inputs include the vehicle miles travelled by each specified type of vehicle, ambient temperature, terrain altitude, calendar year, average vehicle speed, etc. Default values applying to the US vehicle fleet are provided within the model. A related EPA model, PART5, can be used to calculate emission factors for particulate matter. In California, the Air Resources Board's EMFAC model is used in place of MOBILE.

An important aspect that differentiates MOBILE from COPERT is that the latter bases its assumptions on fuel sale statistics, while MOBILE assesses annual mileage accumulation rates by using data from traffic surveys. Other significant differences lie in the way the two models account for the effect of low ambient temperatures and cold start emissions (Zachariadis and Samaras, 1999). A detailed comparative analysis of MOBILE5a and COPERT was presented by Samaras and Zachariadis (1994).

The German MOBILEV emission model can also be used to calculate yearly or hourly average emissions for a single street or a street network using available information on emission factors, traffic mode, street characteristics and vehicle fleet composition. Congested

traffic and cold start emissions are also taken into account. Casella Stanger in association with AEA Technology developed a spreadsheet model for NO_x and PM₁₀ emission factors on behalf of the UK Department of the Environment (DEFRA), to assist local authorities in the air quality review and assessment process. Furthermore, the protocol used by Buckland and Middleton (1999) can be applied for estimating composite emission factors for most regulated pollutants in the UK. This methodology is based on predefined emission factors specific to each vehicle category. The fleet composition is then used to derive a composite emission factor for the road segment of interest. Finally, inverse modelling can be applied to estimate actual fleet emissions from roadside measurements using an operational street canyon model (Palmgren et al., 1999).

Although not exhaustive, the above discussion gives an idea of the existing emission calculation methodologies and available models. It should be stressed that emission factors have to be regularly updated to reflect changes in fuel standards, vehicle fleet composition and engine technology (Stedman et al., 2001). It is generally recognised that emission factors represent one of the most important source of uncertainty in modelling traffic pollution (Kühlwein and Friedrich, 2000).

3.4.3. Meteorological data

The amount of required meteorological information for air quality modelling is proportional to the complexity of the selected model. Simple models for screening applications (e.g. CAR and AEOLIUS Screen) only require the average wind speed over a period of time, assuming that there is no prevailing wind direction. Relatively more complex street canyon models (e.g. OSPM and AEOLIUS Full) require time series of wind speed and direction for the dispersion calculations, ambient temperature and global radiation for the photochemistry algorithm, and (in some cases) atmospheric pressure for unit conversion. In addition to this information, CFD codes (e.g. PHOENICS) require certain specifications concerning atmospheric turbulence and wind profiles. Finally, the atmospheric stability and mixing height need to be specified in order to run Gaussian plume models (e.g. CALINE4).

It should be remembered that meteorological data obtained simultaneously at different weather stations located within few kilometre distances from each other might differ significantly, especially for short averaging periods. When the first street canyon models were developed a few decades ago, it was assumed that the local roof-top wind information needed as an input would not be generally available, and airport 10-m wind data would have to be used. For this reason, empirical expressions relating airport and local winds were derived (Johnson et al., 1973). However, recent research studies have shown that model simulations carried out using

airport winds generally produce lower and less accurate air quality predictions compared to those produced using local wind data (Manning et al., 2000).

3.4.4. Street geometry

Again, more advanced models require a larger amount of input information. Simple models may only need the height and the width of the canyon as input (e.g. AEOLIUS Screen), or just the *type* of the street and the distance between simulated receptor and road axis (e.g. CAR). Semi-empirical models (e.g. OSPM) may additionally require the length and the orientation of the canyon and allow for some gaps between the buildings. A surface roughness coefficient might also be provided by the user. Although there are both experimental and theoretical methods for estimating the roughness length of an urban surface (Pal Arya, 1988), arbitrary values (≈ 0.6 m) are often used.

Relatively simple mathematical models are generally not able to capture the details of the urban canopy (e.g. trees, slanted building roofs, balconies, parked cars, etc.), which might have a significant influence in small-scale pollutant dispersion within street canyons (Gayev and Savory, 1999; Rafailidis, 2000). By contrast, CFD models are able to closely reproduce the details of the urban canopy, if the necessary input information is available. Four main types of boundary conditions imposed to the physical limits of the simulated area have to be specified: (a) the walls, (b) the inlets, (c) the outlets, and (d) the planes of symmetry (i.e. normal velocities are set to zero at a symmetry boundary and the values of all other properties just outside the solution domain are equal to their values just inside the domain). Most commercial CFD codes provide the necessary graphical and numerical tools for treating complex street configurations, including fixed and moving obstacles (Theurer, 1999; Venetsanos et al., 2001).

3.4.5. Background concentrations

Even the simplest urban canyon models require a background pollution value as input, to account for the fraction of the pollutant that is not emitted within the simulated street. The urban background concentration can be defined in several ways. Ott and Eliassen (1973) suggested the existence of an urban CO background as a relatively constant concentration that would be observed at a number of locations throughout the city, providing that the observer was at least 200 ft from the nearest street. Other authors have suggested the use of roof-top measurements as an estimate of the urban background levels. An alternative approach is to simulate the entire urban area using a larger-scale model in order to determine background levels contributed by non-localised sources. The main disadvantage of this method is that it requires additional input data, which are subject to uncertainties (Cooper, 1987).

Berkowicz (2000b) developed a simple model for urban background pollution that can be used in combination with OSPM. ApSimon et al. (2001) adopted ADMS-Urban within the integrated assessment model USIAM to define contributions from different sources at different background receptor locations. CAR includes a simple algorithm for deriving urban background using the regional background (i.e. background due to distant sources) and the diameter of the built-up city area. Nevertheless, the commonest (and more reliable) modelling practice is to use background concentrations obtained from measurements at urban locations that are not directly affected by local sources.

3.5. Special features

3.5.1. Deposition and resuspension

The mass of TSP and PM₁₀, usually measured in fixed monitoring stations, is dominated by the coarser fraction of airborne particulate matter. The PM₁₀ fraction may be transported over long distances (Vignati et al., 1999). It is unlikely for an all-purpose street canyon model to be able to reproduce atmospheric aerosol concentrations measured on the kerbside, unless long range transport, local and regional non-traffic sources, relative humidity, deposition and resuspension processes are adequately taken into consideration.

The number of measured particles is dominated by ultrafine particles (i.e. the smaller fraction of PM_{2.5}). Significant correlation at street level was observed between traffic-related gases (NO_x, CO) and ultrafine particle numbers detected in a street canyon in Copenhagen (Wählin et al., 2001), indicating that traffic was their major source in the urban air. Ultrafine particles are generally expected to behave like inert gases within short distances from their sources. Therefore, their concentration may be successfully calculated using urban canyon models (e.g. OSPM) originally developed for gaseous pollutants (Le Bihan et al., 2001).

3.5.2. Vertical profiles

Relatively simple street canyon models (e.g. AEOLIUS, CAR, etc.) only calculate street-level pollutant concentrations, without giving the user the possibility of choosing the height of the simulated receptors. It has been suggested by several authors (Capannelli et al., 1977; Huang, 1979; Dabberdt and Hoydysh, 1991; Zoumakis, 1995; Vardoulakis, et al., 2002a) that the vertical concentration profiles in a street canyon generally satisfy a law of exponential reduction with height, although more complex patterns depending on the side of the street, the distance from the walls, and the small-scale features of the buildings may be also observed (Kastner-Klein and Plate, 1999; Jicha et al., 2000). The exponential reduction law might not be applicable to traffic-related substances with very short

chemical lifetime (e.g. NO, NO₂). It has been experimentally demonstrated (Väkevä et al., 1999) that NO₂ concentrations may even increase with height within a street canyon, when the weather conditions favour photochemical activity.

3.6. Chemistry sub-models

Simple street canyon models (e.g. STREET–SRI) may only calculate concentrations of passive compounds (e.g. CO). On the other hand, models that are used in regulatory applications need to take fast photochemical reactions into account in order to calculate NO_x concentrations. OSPM uses a simplified chemistry algorithm to account for the transformation of reactive species (i.e. NO_x and O₃) inside a street canyon. AEOLIUS includes a subroutine that calculates statistically NO₂ from NO_x concentrations by means of an empirical curve-fit formula derived from measurements in London (Derwent and Middleton, 1996). In that case, a maximum ratio is set to prevent the NO₂ exceeding 25% of the NO_x at high concentrations. CAR uses an empirical relationship derived from the more elaborated TNO model to calculate street-level NO₂ concentrations, depending on O₃ levels, the fraction of total NO_x directly emitted as NO₂, and the type of the street. Finally, CALINE4 includes the Discrete Parcel Model for NO_x chemistry.

General-purpose CFD models are only able to provide concentrations of inert pollutants, since they do not usually take photochemistry into account. However, specially designed microscale models may combine CFD codes with simplified chemistry algorithms. For example, WinMISKAM adds to MISKAM a simple NO–NO₂ conversion model. A simple photochemistry algorithm linked with is also implemented in the street and neighbourhood scale MICRO-CALGRID model, MISKAM as an alternative to the full chemistry scheme implemented in the urban scale CALGRID model (Stern and Yamartino, 2001).

4. Street canyon studies

Several modelling and experimental field studies aiming at establishing pollutant dispersion and transformation patterns within street canyons have been carried out in the past. Depending on their objectives, different modelling and monitoring techniques have been adopted. Some of these studies were purely experimental, which means that they were exclusively based on full- and/or reduced-scale measurements. At the other end of the spectrum, some purely theoretical studies mainly focusing on the investigation of different wind flow and pollutant dispersion regimes using mathematical models can be also found in the literature.

Most commonly, street canyon studies have combined both mathematical modelling and experimental work.

Recently, the European research network TRAPOS (Optimisation of Modelling Methods for Traffic Pollution in Streets) gave new insights in a number of street canyon related issues: (a) the influence of moving vehicles on pollutant dispersion and turbulence in urban streets (Kastner-Klein et al., 2000, 2001; Vachon et al., 2001); (b) the thermal effects on flow and dispersion within street canyons especially under low wind conditions (Kovar-Panskus et al., 2001a; Louka et al., 2001); (c) the sensitivity of flow and turbulence characteristics to the geometry of the street and its surroundings (Kovar-Panskus et al., 2001b; Kastner-Klein and Rotach, 2001; Leidl et al., 2001; Chauvet et al., 2001); (d) the dispersion and transformation of traffic-related particles (Le Bihan et al., 2001; Wåhlin et al., 2001). TRAPOS included field and wind tunnel measurements, as well as mathematical simulations carried out with advanced numerical (MISKAM, CHENSI, MIMO, CFX-TASCflow) and a simpler parametric model (OSPM). A significant part of the work within the network was devoted to the inter-comparison and evaluation of these models (Louka et al., 2000; Sahn et al., 2001; Ketzler et al., 2001) (Figs. 3 and 4).

In another recent comparison exercise, 24 modellers from 21 different institutions used a range of models to predict pollutant concentrations in the same street canyon in Hannover (Germany). Large discrepancies were identified in the emission and dispersion modelling results obtained from different participants. The fact that individual modellers obtained different results even when they used the same model revealed the influence of the human factor on the quality of the simulations as well as the need for establishing standard operational procedures (Baechlin et al., 2001; Lohmeyer et al., 2002). In the following sections, representative studies covering all aspects of street canyon research are briefly discussed.

4.1. Full-scale experiments

DePaul and Sheih (1985, 1986) carried out a tracer gas (SF₆) experiment in an urban street canyon in Chicago (USA) in order to obtain measurements of pollutant retention times and resulting concentrations within the canyon. The mean wind velocities were determined by analysing trajectories of air balloons that were released in the street. Nakamura and Oke (1988) studied the climate of urban canyons using field observations of wind and temperature from a street canyon in Kyoto (Japan). These observations were used to derive simple algorithms relating the above roof-level to the within-canyon meteorological conditions. Pfeffer et al. (1995) presented measurements of NO₂, CO, benzene, soot and other atmospheric pollutants carried out in two busy street canyons in Dusseldorf and Essen (Germany), as a

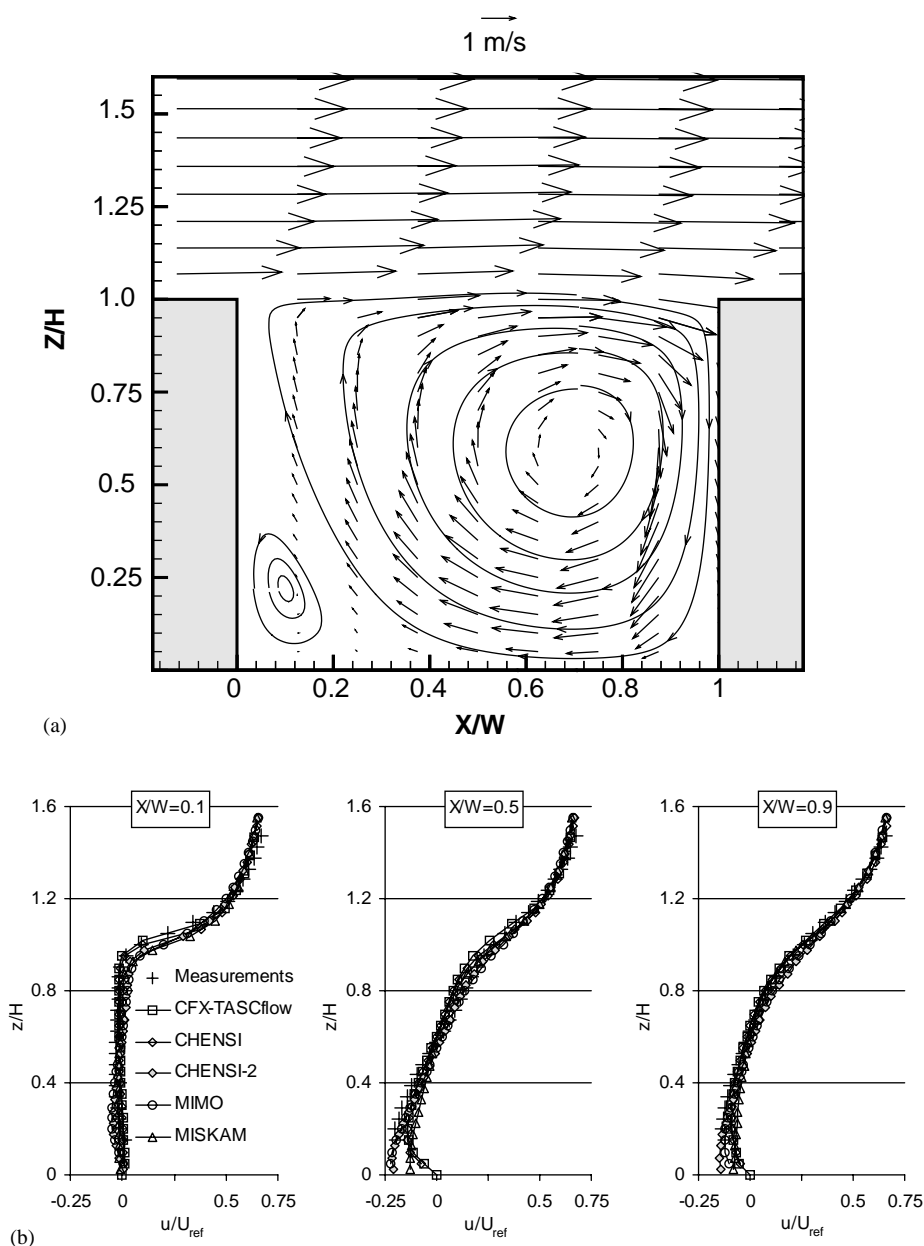


Fig. 3. (a) Flow field within and above a wide street canyon as it was reproduced using CHENSI, and (b) vertical profiles of the normalised horizontal wind component (u/U_{ref}) measured in a wind tunnel (BLASIUS) and predicted using five different CFD models. H and W are the height and the width of the canyon, and z and x the height and the distance of the receptor from the canyon wall, respectively (Sahm et al., 2001).

part of a pilot study preparing the implementation of new regulations included in the German Federal Clean Air Act. The correlation between different pollutants and the influence of the wind conditions on measured concentrations were investigated. Namdeo et al. (1999) presented results from a monitoring study on traffic-related particulate pollution in urban areas. Field measurements of airborne fine and

coarse particulate matter were taken in an urban street canyon in Nottingham (UK) and the correlation of the observed concentrations with traffic was studied. Venegas and Mazzeo (2000) reported CO concentrations measured in a deep street canyon in Buenos Aires (Argentina).

Vertical concentration gradients of CO were observed by Zoumakis (1995) in a busy street canyon in Athens

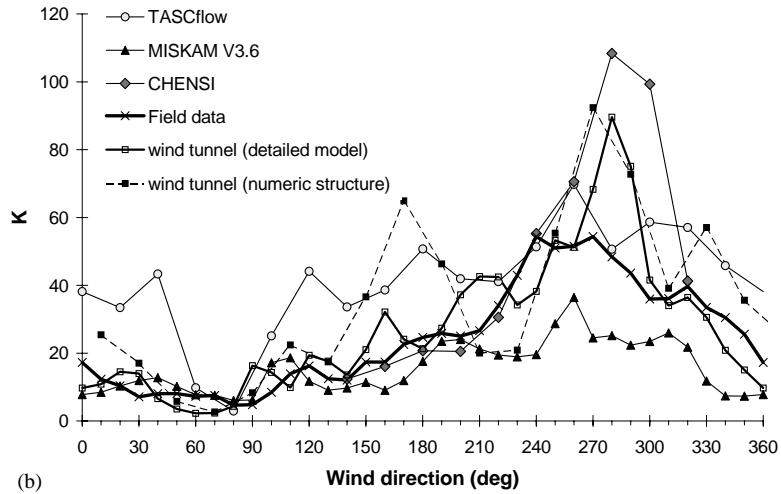
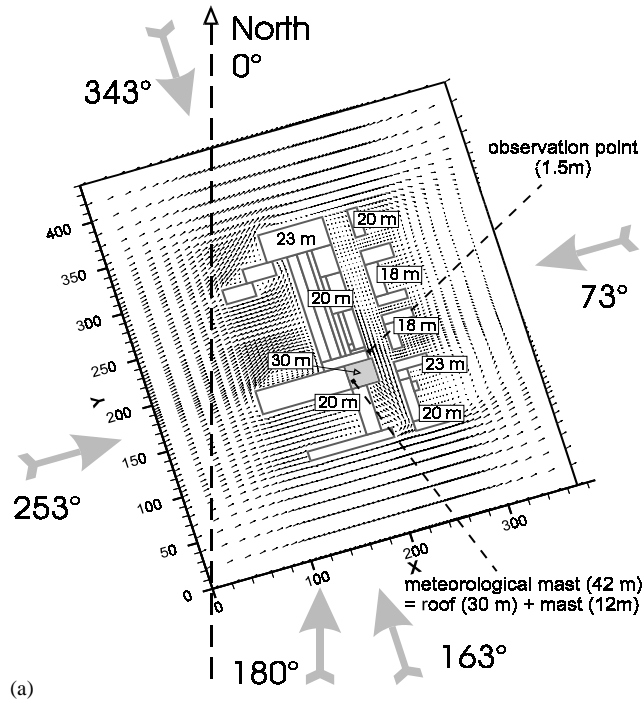


Fig. 4. (a) Flow field in a complex street canyon in Hannover reproduced using MISKAM (wind direction: 250°), and (b) calculated normalised concentrations (K), field data, and wind tunnel measurements (EnFlo) using the exact shape of the real buildings (detailed model) or building shapes adapted to the resolution of the grid used in the CFD simulations (numerical structure) (Louka et al., 2000).

(Greece). The monitoring results were used to derive an empirical expression relating pollutant concentration and height above the ground. Gaseous pollutants (CO , NO_x , O_3) and aerosol particle concentrations were measured at two different heights within an urban street canyon in Lahti (Finland) by Väkevä et al. (1999). The main objective of this study was to investigate the factors leading to the formation of vertical concentra-

tion profiles within the canyon. TSP, PM_{10} and $\text{PM}_{2.5}$ concentrations were measured in two open streets and two canyon sites in Hong Kong by Chan and Kwok (2000). These measurements showed that the dispersion of particulate matter was affected by the prevailing wind direction and the aspect ratio of the street. An exponential reduction of TSP and PM_{10} with height was observed.

Vachon et al. (2000) reported results (i.e. concentration, temperature and wind fields) from a full-scale experiment carried out in a street canyon (Rue de Strasbourg) in Nantes, France. This was the first campaign of the URBCAP project, which has the aim of assessing pollutant transformation processes within the urban canopy and validating small-scale dispersion models. Finally, within the framework of the LIFE RESOLUTION project (Wright, 2001), benzene and NO₂ measurements were taken in four European cities (Dublin, Madrid, Paris and Rome) in order to assess pollution levels with reference to established air quality standards, optimise the design of monitoring networks, and provide experimental data to support the validation of urban dispersion models. The sampling, carried out with diffusive tubes, covered a wide range of urban and suburban locations, including a number of street canyons.

4.2. Reduced-scale experiments

Leisen and Sobottka (1980) made a comparison between field observations from two street canyons in Cologne (Germany) and wind tunnel measurements in order to investigate pollutant dispersion within urban streets and develop simulation models. Meroney et al. (1996) presented a wind tunnel study of car exhaust dispersion from street canyons in an urban environment. The main objective of this study was to investigate how pollution dispersion is affected by street geometry and particular emphasis was put on the design of a line source to realistically represent traffic emissions. The experiments were performed in the atmospheric boundary layer wind tunnel (BLASIUS) of the Meteorological Institute of Hamburg University (Germany). In a later study, Pavageau and Schatzmann (1999) investigated the concentration fluctuations in a reduced-scale urban canyon simulated within BLASIUS. Experimental data sets of wind tunnel measurements carried out in this facility for the validation of microscale dispersion models are available on the Internet (<http://www.mi.uni-hamburg.de>).

The differences between reduced- and full-scale experiments were illustrated by Liedtke et al. (1999), who compared field measurements obtained in a street canyon in Hannover with wind tunnel results. The generic effect of using a simplified model of a street canyon in the wind tunnel was studied by taking measurements using different scaled models that included various levels of detail of the real canyon geometry. Significant differences were found in the results. In a later study, Schatzmann et al. (2000) showed how wind tunnel data can be used to supplement and enhance the value of field measurements for model validation purposes.

Rafailidis (1999) investigated the influence of atmospheric thermal stratification on urban street canyon ventilation in the EnFlo wind tunnel of the University of Surrey (UK). The measurements indicated that stable stratification conditions result in trapping the pollutants within the canyon. In a number of other wind tunnel studies (Rafailidis, 1997, 2000), the influence of building area density and roof shape on the wind field above and inside the urban canopy were highlighted. Uehara et al. (2000) also investigated the effects of thermal stratification on the wind flow in and above urban street canyons using the atmospheric diffusion wind tunnel at the Japanese National Institute for Environmental Studies (Ogawa et al., 1981). The results showed that the wind vortex within the canyon becomes weaker when the atmosphere is stable. Kastner-Klein and Plate (1999) presented results from tracer dispersion experiments performed in a neutrally stratified atmospheric boundary layer wind tunnel in the Institute of Hydrology and Water Resources of the University of Karlsruhe (Germany). The influence of systematic parameter variation (i.e. building configuration, roof shape, wind direction) on the concentration field within a street canyon was studied.

Gerdes and Olivari (1999) studied the wind and concentration fields generated within even and asymmetric street canyons under perpendicular winds using the L-2B wind tunnel of the von Karman Institute (Belgium). A strong influence of the surrounding landscape on pollutant dispersion was observed. The ratio of the height of the walls flanking the street was found to have a significant effect on the concentration patterns, while the width of the canyon was proved to be of less importance.

4.3. Parametric modelling using field and/or wind tunnel measurements

Johnson et al. (1973) developed STREET-SRI using data from the San Jose Street Canyon Experiment in California. Sobottka and Leisen (1980a, b) created a modified version, called MAPS, which is quite similar in form and performance with the original model. Nicholson (1975) developed a simple box model that yields street-level average CO concentrations in urban canyons under perpendicular and parallel wind conditions. Model results were proved to be in reasonable agreement with field data obtained in Frankfurt (Germany), Madison and Chicago (USA). Yamartino and Wiegand (1986) developed CPBM using data from an extensive field monitoring programme in Bonner Strasse (Cologne). Part of the experimental data (CO and NO_x) was used to evaluate the model. In the same study, the performance of CPBM (only for CO predictions) was compared with the performance of STREET-SRI and its modified version MAPS.

Hoydysh and Dabberdt (1988, 1994), and Dabberdt and Hoydysh (1991) carried out flow visualisation and tracer concentration measurements in the atmospheric boundary layer wind tunnel (ABLWT) of the Environmental Science and Services Corporation (USA). Pollutant dispersion was simulated using reduced-scale models for street canyons (both even and asymmetric) and intersections. Wind tunnel results were compared with concentrations calculated using STREET–SRI and the analytical model developed by Hotchkiss and Harlow (1973). Finally, a simple exponential law describing vertical concentration profiles was established.

Hertel and Berkowicz (1989a) developed OSPM using measurements obtained in Jagtvej street in Copenhagen (Denmark). An intensive monitoring site was established in connection with a permanent air quality station operating in this street. A selection of the obtained wind and turbulence data was analysed by Nielsen (2000). OSPM has been applied to several street canyons in Copenhagen, Utrecht, Oslo, Helsinki, Beijing and other major cities (Berkowicz et al., 1996; Hertel and Berkowicz, 1989b, c; Kukkonen et al., 2001; Fu et al., 2000). Based on the same principles, Buckland (1998) formulated AEOLIUS, which has been mainly used in the UK (Manning et al., 2000). Other authors (Sacré et al., 1995; Vardoulakis et al., 2002b) have presented slightly modified versions of OSPM. Finally, Jensen et al. (2001) developed a decision-support tool (AirGIS) based on OSPM, which applies a geographic information system (GIS) for mapping traffic emissions, air quality and human exposure levels at residential/professional addresses and in streets.

Kono and Ito (1990a) presented the OMG VOLUME-SOURCE model, a microscale dispersion model that estimates concentrations of traffic-related pollutants in an urban area within 200 m from the roadside. The model parameters were determined using experimental data from five locations in Osaka (SF_6 was released as a tracer gas). Model results were compared by the same authors (Kono and Ito, 1990b) with concentrations calculated using three line source dispersion models, namely the JEA model, the TOKYO model, and the HIWAY-2 model (Peterson, 1980).

Qin and Kot (1993) took measurements of CO and NO_x at different heights and distances from the kerb within three asymmetric street canyons in Guangzhou City (China). STREET–SRI and a Gaussian plume model were used in this study to obtain CO and NO_x estimates, which were found to be in reasonable agreement with the observed values. Lanzani and Tamponi (1995) presented the microscale Lagrangian particle model GEM for the atmospheric dispersion of primary pollutants in the urban canopy. GEM was validated against field measurements and compared with CPBM and STREET–SRI. The STREET–SRI model

was also used in two independent studies in Argentina to calculate CO concentrations in street canyons in Cordoba (Stein and Toselli, 1996) and Buenos Aires (Bogo et al., 2001). A reasonably good agreement was found between measurements and predictions.

Hargreaves and Baker (1997) developed a Gaussian puff model, called PUFFER, to simulate the dispersion of vehicular pollutants in urban street canyons. This model, which explicitly takes into account vehicle induced turbulence, enables the user to investigate realistic transient situations such as traffic congestion and non-steady above canyon wind fields. A short sensitivity analysis and a comparison with STREET–SRI were also included in the same study. Gualtieri and Tartaglia (1998) developed a comprehensive air quality model, including traffic, emission and dispersion sub-models, for assessing pollutant concentrations in urban areas. A semi-empirical street canyon algorithm based on STREET–SRI and field measurements from Firenze (Italy) (Tartaglia et al., 1995; Gualtieri and Tartaglia, 1997) was included in this model, which was finally integrated in a GIS.

Hassan and Crowther (1998a) developed a single box model for calculating first estimates of pollutant concentrations within urban canyons. The model parameters were derived using field CO measurements taken in Hope Street, Glasgow (UK). Furthermore, Hassan and Crowther (1998b) used PHOENICS to simulate wind flow and pollutant dispersion within the same canyon. The accuracy of the two-dimensional steady state numerical simulations was examined by comparing the predicted results with the field measurements.

Micallef and Colls (1999) developed a semi-empirical emission–dispersion model for predicting the temporal and spatial distribution of airborne particulate matter in street canyons. This model called SLAQ includes emission modules, meteorological pre-processors, modules for within-canyon processes, dispersion modules, and modules accounting for external influences. The dispersion module is mainly based on CPBM. Model features include a correction for the heat island effect, dry and wet deposition, particle settling, etc. SLAQ was evaluated against measurements obtained in a street in Loughborough (UK) using the automated near real time kinetic sequential sampling (KSS) system (Colls and Micallef, 1999).

Coppalle et al. (1999, 2001) measured NO_x and CO concentrations at different background and kerbside locations, including a street canyon (Rue Crevier), in a medium size French city (Rouen) during 4 weeks in winter. Using the obtained experimental data for parameterisation, Coppalle (1999) developed a simple mathematical model that calculates the vertical pollutant distribution inside a street canyon under low wind conditions. Vardoulakis et al. (2000, 2002a,b) carried out field experiments in two even (Bd. Voltaire and Rue

de Rennes) and one asymmetric (Av. Leclerc) street canyons in Paris, France. Three semi-empirical dispersion models (STREET-SRI, OSPM and AEOLIUS) were tested using the observed CO and benzene values (Fig. 5). Furthermore, an empirical relationship for calculating vertical concentration profiles within the

streets was derived as well as a practical methodology for estimating model uncertainty.

In the framework of AUTO-OIL II programme (Skouloudis, 2000), a large number of air quality modelling simulations were carried out in order to assess the compliance with the new EU air quality

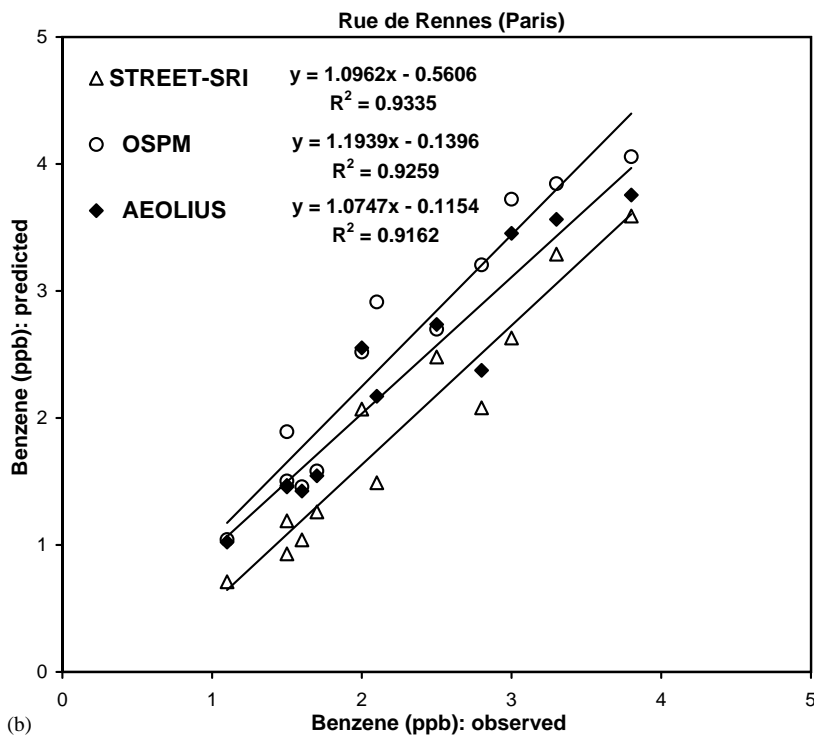
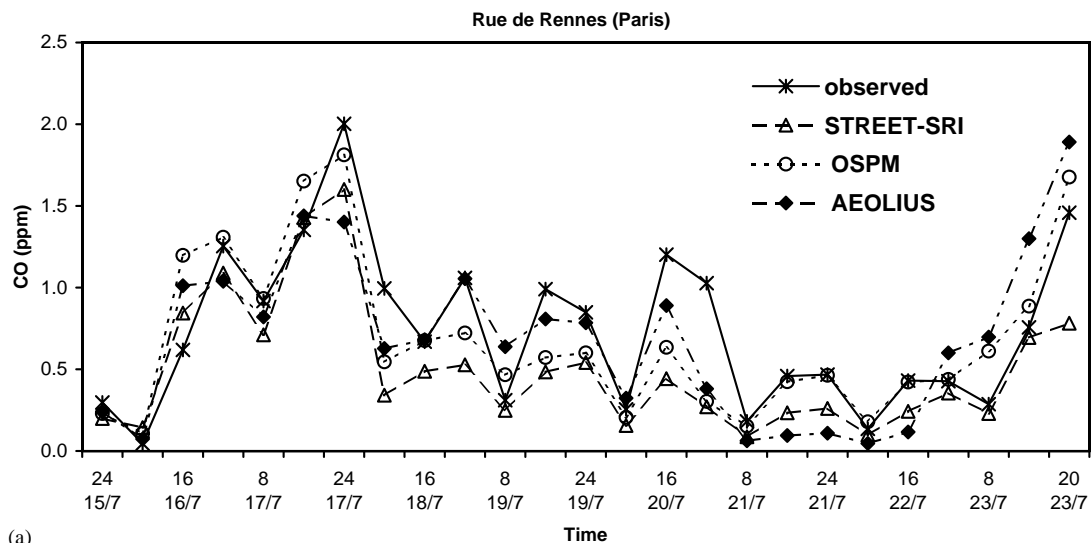


Fig. 5. STREET-SRI, OSPM and AEOLIUS predictions against (a) roadside CO (ppm) measurements obtained with a standard infrared analyser and (b) benzene (ppb) measurements obtained with passive samplers at different locations in Rue de Rennes, Paris (Vardoulakis et al., 2002a).

standards for NO₂, O₃, CO, benzene, and PM₁₀. An advanced methodology was established, incorporating models of different spatial scales, to provide air quality simulations down to street level. Two urban canyon models, namely OSPM and MICRO-CALGRID, were evaluated using measurements from Viale Murrilo (Milan) and Schildhorn Strasse (Berlin).

Mensink and Lewyckyj (2001) developed the simple mathematical model STREET BOX, which assumes a uniform concentration distribution within a street canyon and is based on the concept of a turbulent intermittent shear flow shed from the roof of the up-wind building. Model results were compared with benzene concentrations measured in 10 streets in Antwerp (Belgium). Furthermore, benzene, CO and NO_x concentrations calculated with STREET BOX were compared with values predicted using OSPM for a street canyon in Hannover. A discrepancy of 30% between predictions from the two models was reported.

Mukherjee and Viswanathan (2001) used the street canyon and Gaussian line source modules of the regional-scale dispersion model INDIC AIRVIRO to simulate ambient CO concentrations on two major roads in Singapore. The street canyon module based on STREET-SRI gave predictions comparable to the measured values at both the sites, despite the significant differences in street geometry.

Finally, Addison et al. (2000) presented an integrated method for predicting the spatial pollutant distribution within a street canyon. This method was based on a Lagrangian stochastic particle model superimposed on a known velocity and turbulence field. A traffic simulation package (Paramics) was used to model the flow of vehicles in realistic traffic conditions. This model is expected to be calibrated in the future using roadside measurements.

4.4. CFD modelling using field and/or wind tunnel measurements

Okamoto et al. (1994, 1996) developed a two-dimensional numerical air quality model that can be applied to street canyon cross-sections under perpendicular wind conditions. It contains a wind field and a diffusion sub-model; the latter based on a Monte Carlo particle scheme. The model was evaluated using databases from field measurements carried out in three typical roadways surrounded by tall buildings in Tokyo (Japan). Furthermore, the predictive performance of the model was compared with the performance of STREET-SRI and of the APPS three-dimensional numerical model.

Leitl and Meroney (1997) used FLUENT to simulate numerically wind tunnel experiments conducted by Rafailidis et al. (1995) in the BLASIUS facility of the University of Hamburg. Several simplified two- and

three-dimensional simulations were carried out to study the effect of emission rate and source design on flow structures and pollutant dispersion within the canyon. The advantages of using numerical CFD codes for optimising wind tunnel experiments were highlighted. In a later study, Meroney et al. (1999) compared numerical simulations carried out with FLUENT against other wind tunnel data from BLASIUS corresponding to several building shapes including a street canyon. Johnson and Hunter (1998) carried out a preliminary comparison between wind tunnel data from BLASIUS and simulations of wind flow and pollutant dispersion within street canyons using SCAM. This is a numerical code that consists of the wind model CITY and the dispersion model SCALAR (Johnson and Hunter, 1995).

Yoshikawa and Kunimi (1998) reported the development of an air quality simulation system, which calculates traffic volumes, evaluates the effects of building structures on pollutant dispersion along roadways, and takes into account photochemical reactions. The dispersion model, which serves as the platform of the overall simulation system, is a standard CFD code slightly modified to take into account vehicle induced turbulence inside street canyons. The model was validated against field data from an earlier tracer gas (SF₆) diffusion experiment carried out in Tokyo.

Riain et al. (1998) measured CO concentrations at different heights within an asymmetric canyon in central London (UK). The FloVENT code was used in that case to simulate the concentration and wind fields created in the street. Soulhac et al. (1999) studied pollutant dispersion within street canyons (both even and asymmetric) and intersections using wind tunnel and numerical simulations carried out with MERCURE and CHENSI. The results were compared with two simple models: CARMEN for flow in a single street canyon and SIRANE for flow in a street network (Soulhac and Perkins, 1998; Soulhac et al., 2001).

Both CFD simulations using STAR-CD and wind tunnel measurements in EnFlo were carried out on a model arrangement of two intersecting street canyons, allowing the accuracy of predictions to be assessed (Scaperdas, 2000). It was found that even small changes in building alignment had a significant effect in the dispersion of pollutants in the street. Monitoring data from a permanent air quality station in central London were also used in this study (Scaperdas and Colville, 1999).

Ketzel et al. (2000) carried out mathematical simulations of pollutant dispersion within street canyons using the relatively simple OSPM and the more complex MISKAM code. The results were compared with wind tunnel simulations and field measurements from two permanent monitoring stations in Copenhagen and Hannover.

Huang et al. (2000) developed a two-dimensional numerical code, which was evaluated using data sets from tracer gas dispersion experiments carried out in an asymmetric street canyon in Tokyo. Chan et al. (2002) used FLUENT to simulate the wind flow and pollutant dispersion within an isolated street canyon. The validation of the numerical model was carried out using an extensive experimental database obtained from BLAS-IUS. Different turbulence models and street canyon configurations were studied. It was found that wider streets and lower buildings are favourable to pollutant dilution within canyons.

Garcia Sagrado et al. (2002) studied the two-dimensional wind flow and pollutant dispersion within urban canyons by means of wind tunnel measurements (L-2B wind tunnel, von Karman Institute) and numerical simulations carried out with FLUENT. It was observed that pollutant concentrations decreased with increasing the height of the down-wind canyon wall. The influence of a third building situated up-wind was also investigated.

4.5. Theoretical CFD modelling

Hunter et al. (1992) carried out a numerical investigation of typical three-dimensional flows within urban canyons in order to identify the key parameters that determine the transition between the different flow regimes for synoptic winds perpendicular to the street axis (see Section 2.2). Lee and Park (1994) developed a parameterisation scheme whereby the pollutant concentrations in an urban street canyon can be estimated from the source term, the meteorological conditions, and the street geometry using a two-dimensional time-dependent flow model. Sini et al. (1996) used CHENSI to study the influence of the geometrical aspect ratio of a street, which led to a refinement of the Oke (1988) classification into three flow regimes. In addition, it was shown that the differential heating of street surfaces (e.g. building facades) can influence the dispersion conditions within the canyon. Assimakopoulos et al. (1999) used MIMO to assess the influence of the numerical treatment of the wall boundary on the wind field and concentration patterns within two different two-dimensional street canyons.

Jicha et al. (2000) adopted a three-dimensional Eulerian–Lagrangian approach to study pollutant dispersion in an idealised street canyon taking traffic-induced turbulence into account. The Eulerian approach was based on the CFD code STAR-CD into which a Lagrangian model was integrated. Craig et al. (2001) also used STAR-CD coupled with a mathematical optimisation algorithm to identify the configuration of an idealised urban geometry that minimises pollution peaks. This methodology may be used to optimise traffic patterns or to modify street geometry for air pollution

control. Xia and Leung (2001a, b) used a Lagrangian particle together with a two-dimensional wind field model to simulate flow patterns for different building configurations within the urban canopy. The flow was visualised numerically by discharging a large number of particles into the computational domain. It was found that the higher concentrations did not always appear on the leeward side of the canyon and that the flow pattern was highly dependent on the configuration of the buildings and surrounding urban canopy.

Theodoridis and Moussiopoulos (2000) investigated the influence of building density and roof shape on the wind and dispersion characteristics in an urban area using CFX-TASCflow. In a later study, Theodoridis et al. (2001) applied the same model for simulating wind fields and pollutant dispersion in a complex urban area. In that case, two advanced turbulence models (namely the k - ϵ and the RSM) were adopted and two grid refinement levels were tested. Venetsanos et al. (2001) carried out flow and dispersion calculations using ADREA-HF, a CFD code for simulating vapour cloud dispersion in complex terrain (Bartzis, 1991), to study the effects of moving vehicles on air pollution patterns within street canyons. The calculations were performed in a moving co-ordinate system with the car and site geometry fully resolved.

Chan et al. (2001) carried out a number of three-dimensional numerical simulations using CFX-5 in order to study flow regimes and corresponding pollutant dispersion characteristics for various types (i.e. aspect ratios) of urban street canyons. Some guidelines related to the geometry of the canyon were established for efficient pollutant dispersion. Finally, Jeong and Andrews (2002) used two numerical codes (TEACH-2E and HEATX) to study the two-dimensional flow structure of skimming flow fields in a street canyon at high Reynolds number. The critical aspect ratios of the transition between different vortex regimes were identified.

5. Conclusions

Considerable effort has been made in recent years to improve the scientific understanding of dispersion and transformation phenomena governing urban air quality. A large number of research studies have focused on street canyons, where the highest levels of air pollution often occur and the larger targets of impact are concentrated. The natural ventilation of urban streets is reduced mainly due to the presence of buildings. Within the urban canopy, wind vortices, low-pressure areas and channelling effects may be created under certain meteorological conditions, giving rise in some cases to air pollution hotspots. For example, high concentration levels have been often observed on the leeward side of regular canyons under perpendicular

wind conditions. Furthermore, photochemical activity especially during sunny days may induce high street-level concentrations of secondary pollutants.

Most authors have adopted different combinations of monitoring and modelling techniques for assessing air quality in urban street. There are several methods for monitoring roadside particulate and gaseous pollutants, each one of them having a number of advantages and drawbacks. Passive sampling can be used to obtain air quality data of high spatial resolution (both vertically and horizontally). On the other hand, active sampling can provide high temporal resolution. Hence, a combination of passive and active methods may be ideally used to capture short-term air pollution episodes and hotspots within a canyon. Mathematical and physical models are needed to optimise air quality monitoring, provide estimates for regulatory purposes, study different street geometries, and finally test future emission and traffic scenarios. Depending on their mathematical/physical principles, they may be more or less suitable for a number of applications.

5.1. Gaussian plume models

They are popular because of their relative simplicity and the possibility of easily including special features like deposition, source buoyancy, etc. Although they are mainly designed to simulate point or line sources in open terrain, they may also include complex terrain, street canyon and intersection modules (e.g. CALINE4). They can produce time series of pollutant concentrations and are used in a wide range of engineering and scientific applications. The main disadvantages of this method are the restricted number of different canyon configurations allowed and the relatively large amount of input information required.

5.2. Operational street canyon models

Semi-empirical parametric models (e.g. STREET-SRI, OSPM, AEOLIUS, etc.) are the most commonly used tools in regulatory street canyon applications. They are specially designed to produce time series of pollutant concentrations within near-regular canyons, and they require a relatively small amount of input information and computational resources. On the other hand, they are based on a number of empirical assumptions and parameters that might not be applicable to all urban environments. For this reason, they should be recalibrated against a small (at least) number of field measurements, if they are to be applied to new locations.

5.3. CFD models

CFD is a numerical modelling technique that can be applied to many different fields of engineering and

scientific research. As far as roadside air quality is concerned, the main advantage of the method is that it can reproduce the entire flow and concentration fields within urban canyons of any configuration, if the necessary input data are available. Furthermore, the details of urban canopy can be efficiently taken into account, thanks to the fine grid generation capabilities of modern CFD models.

Traditionally, CFD has been seen as a modelling technique requiring long computational times and expensive hardware/software resources. However, recent computer hardware developments have contributed to the spread of CFD modelling, since the speed and memory capacities of PCs are now sufficient for relatively small applications. Furthermore, CFD codes have become much easier to use due to improvements in interface facilities (although they still require a reasonable level of knowledge of flow physics). It should be kept in mind that the main objective of an environmental CFD exercise is to improve the understanding of the behaviour of a system, rather than obtaining results readily comparable with regulatory standards.

5.4. Reduced-scale models

Physical modelling in wind tunnels has been proved very useful in investigating specific characteristics of pollutant dispersion within the urban canopy (e.g. effects of roof shape, moving and fixed obstacles, etc.). Although wind tunnel experiments have the advantage of providing controlled dispersion conditions (e.g. wind velocity, stability, etc.), they might be seen as relatively expensive and difficult to set up. Wind tunnel measurements are often used in the development and validation of mathematical models.

All modelling techniques described in the previous paragraphs are complementary. For the selection of the appropriate dispersion model, one should be aware of the capabilities, underlying assumptions and limitations of the available software. Although they vary greatly in terms of complexity, simple and advanced models can be both useful in different air quality applications (Berkowicz, 1997). For regulatory purposes, it is recommended to use a simple screening model initially, before adopting a more sophisticated approach that will include simulations with a more advanced code and a limited (at least) number of measurements. For quick air quality surveys and traffic planning, simulations with a simple model like CAR might suffice. On the other hand, air quality monitoring network design may require both parametric modelling for an initial selection of the streets to be implemented and then CFD simulations to identify representative locations within these streets.

It should be stressed that all mathematical models need thorough validation against experimental data.

The accuracy of their predictions is bounded by the accuracy of input data such as emission factors, traffic and meteorological data, street geometry, etc. Therefore, decision-makers should use modelling results cautiously, especially when relevant field measurements are not available.

Future research on urban air quality is expected to focus on topics related to low wind conditions, thermal effects due to solar radiation, microscale dispersion around fixed and moving obstacles, and pollutant dispersion in irregular canyons and other complex urban microenvironments (e.g. intersections, parking spaces, etc.). Despite the large number of existing codes, there is still a need for scientifically sound, user-friendly and well-documented air quality models, as well as for high quality experimental data sets.

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