MODELLING OF AIR QUALITY IN STREET CANYONS

by

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Abstract

Despite the recent technological improvements on vehicle emission control and the increasingly stricter regulations, traffic-related air pollution is one of the most pressing problems in modern urban agglomerations both in the developed and the developing world. Air quality limit values, whose objective is to protect public health, are frequently exceeded, particularly in busy streets and urban areas. Evidence is constantly emerging related to the human exposure to increased pollutant concentrations in densely populated urban areas in contrast with the proven adverse effects on human health. It is, hence, imperative to fully understand the pollutants behaviour within confined urban surroundings in order to achieve further improvements in urban air quality.

This study presents the results of a limited monitoring and modelling methodology, which is adopted in order to understand the predominant mechanisms of pollution dispersion in an urban street canyon. The examined canyon in Headingley area of Leeds is relatively narrow and flanked by closely spaced buildings on both sides. Evidence of the street canyon effect on the diffusion of Carbon Monoxide emissions, hence, has been attempted to be established via the analysis of the measured levels and the results of the application of OSPM dispersion model. The relationship between the engine emission rates, the ambient conditions and other factors affecting the pollutants dispersion could not be spherically examined because of the shortcomings in both the monitoring and the modelling procedure. Moreover, the model predictive performance of CO levels has been proven fairly poor, when evaluated against the observed concentrations. WinOSPM generally underpredicted the CO measurements and its sensitivity to wind speed and direction changes could not reveal lucid evidence of the street canyon effect.

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Chapter 1

Introduction

1.1 Background

The good condition of ambient air is a significant factor affecting quality of everyday life, particularly in large conurbations. The population living in modern cities is constantly increasing worldwide and, on the other hand, the quality of urban environment is deteriorating. Air pollution is not a new problem, but the types and behaviour of the various air pollutants have changed during the last decades. Severe environmental problems have been present since the Industrial Revolution, mostly due to the intense use of fossil fuels in the industry and domestic sector. However, since the UK Government's Clean Air Act implemented in 1956, these emissions have been significantly reduced and over the last two decades the main source of air pollution has been attributed to the motorised traffic. Increasing vehicle volumes, heavier congestion and more complex urban topographies outline the key causes of this alteration (Michail, 2003).

Nowadays, national and local authorities are trying to implement policy instruments and traffic management schemes to tackle poor air quality issues and locate the contamination 'hot spots', mainly in developed countries and large cities. Parallel to stricter legislations and reduced vehicle emissions, systematic air quality monitoring and modelling are essential tools in order to understand the behaviour of various toxic substances emitted by motor vehicles.

Monitoring campaigns can, however, be costly and time-consuming procedures if extensive and accurate results are to be achieved. That is why over the last years various types of air quality models have been developed, which are able to assess roadside air quality by predicting current and future pollutant levels and providing temporal and spatial variations for a wide range of topographies and conditions (Sharma and Khare, 2001). These mathematical models are able to study the physical and chemical processes related to the dispersion and transformation of gaseous pollutants and, furthermore, the effect of buildings and other urban structures on pollutant diffusion and accumulation patterns (Vardoulakis et al, 2003). This study makes use of a pollution dispersion model and involves its application and validation within a street canyon in a built-up area of Leeds.

1.2 Objectives and Scope of the Study

The central objective of this dissertation is to study the dispersion of air pollution within an urban street canyon and, specifically, along a section of Otley Road in the area of Headingley, in northern Leeds (Fig. 1.1).

The two main aims are to monitor and to model the air quality in this urban environment and, thus, evaluate the performance of a modern dispersion model. The first objective is to collect measurements of air pollutant concentrations from motor vehicles in the examined road as well as the associated traffic and meteorological data. Subsequently, the goal is to model the dispersion of the most important pollutants in the street canyon using the street canyon module of an advanced air quality model.

Finally, through the comparison of the monitored and the predicted pollution levels, the applied model can be validated and the presence of street canyon effect on the pollutants' dispersion can also be demonstrated.

The scope of this project is intended to include monitored data collection and dispersion modelling techniques in an urban street canyon in order to examine and interrelate a wide range of physical and environmental variables.

Figure 1.1 The location of the examined street canyon in Headingley, Leeds.
Source: Google Maps (2006)

Due to the lack of time and resources, the monitoring campaign will be limited to concentrations of Carbon Monoxide (CO) across the selected urban area. The installed equipment by the Institute for Transport Studies (ITS), University of Leeds, can measure CO concentrations for adequate period of time along the examined street canyon and at a background location. Meteorological and other useful background pollution data have been provided by the Leeds City Council. The required traffic data have been collected from relevant ITS projects and from manual classified vehicle counts in the area.

Regarding the modelling procedure, it is possible to study a wider range of pollutants and scenarios in the area. Hence, different meteorological and topographic characteristics can be analysed. For instance, various wind speeds and directions or receptor locations can be investigated and the effects on pollutants dispersion could thus be deduced. In this study a recently updated air quality model is used for the prediction of pollution levels within the examined urban area. The Windows based Operational Street Pollution Model (WinOSPM, Version 5.0.64) developed by the National Environmental Research Institute of Denmark (NERI, 2006) is believed to be a well accepted and widely applied model that is capable of assessing air quality in confined urban environments.

1.3 Structure of the Dissertation

The structure of this project is outlined below:

The second chapter contains the literature review on traffic pollution and, in particular, a brief description of the main air pollutants and the legislative framework for air quality standards in the United Kingdom (UK) and European Union (EU). The emission and dispersion of air pollution are also examined giving emphasis to the behaviour of the gases in constrained urban areas and street canyons. Moreover, the process of monitoring traffic pollution is reviewed along with the UK National Air Quality Strategy. The chapter closes with a description of the main types of air quality models and, in particular, the OSPM dispersion model, which has been used in this study of pollution dispersion in urban street canyons.

The methodology of the study is developed in Chapter 3. The first part includes the experimental method of the monitoring of the pollutants examined here. The selected site and the required equipment are described as well as the process of data collection, i.e. the meteorological data, the manual and automatic traffic counts and the in situ and background measurements of the pollutant concentrations. In the second section, the setting and the application of the air quality model is presented and the input and output data are also analysed.

Chapter 4 includes the results of the monitoring and modelling procedures. The observed data from the monitoring campaign are first analysed. The predictions of the AQ model are then compared against the measured concentrations in order to evaluate the performance of OSPM in the current situation.

Finally, the fifth chapter summarises the main conclusions of the study regarding the performance and the validation of the applied dispersion model. Recommendations for future work and applications are also mentioned at the end.

Chapter 2

Traffic Pollution

Although good quality of life and robust level of economy are strongly correlated with an efficient and flexible transport system, current situation shows that the radical increase in road transport causes an important and growing threat to natural environment and human health. The transport sector is the fastest growing consumer of energy and producer of greenhouse gases in Europe, jeopardising thus the EU reaching the 2010 emissions reduction target set under the Kyoto protocol. Vehicle technology and fuel improvements have resulted in noticeable decreases of emissions of particular pollutants. Nevertheless, air quality in most European cities is still poor and, hence, transport policies ought to aim at the control of traffic growth and the promotion of sustainable transport modes (EEA, 2006a).

The risk of human exposure to high pollutant concentrations is significantly increased in densely populated urban locales. Air quality limit values are set in order to care for public health, but they are repeatedly exceeded, especially in busy roads and other urban hotspots. The pollutants emission and dispersion modelling is a powerful tool for determining which pollutants' reductions are needed in certain areas, so that they remain below the predefined emission ceilings. The analysis of air quality scenario projections at street level and the impacts of particular policies and measures are possible with the use of street scale models. Thorough local traffic data, air pollution measurements, meteorological and topographical data of the problematic sites are normally required as inputs to these models in order to obtain accurate air quality prognoses at street level (EEA, 2006b).

This chapter begins with the main components of air pollution and the legislative framework of air quality control. Subsequently, the mechanisms of pollutants' emission and dispersion are reviewed, laying emphasis on urban areas and street canyons, as well as the principles of relevant monitoring and modelling techniques.

2.1 Main types of air pollutants

The most important air pollutants, which are related to motorised traffic and mostly concern local authorities nowadays, are carbon monoxide, nitrogen oxides, hydrocarbons and particulate matters. The UK Department for Environment, Food and Rural Affairs (DEFRA, 2006) has listed the main air pollutants, their sources and impacts on human health as well as some characteristic time-series data and trends of emissions, as summarised below. These substances are mostly produced by fossil fuel burning and in Table 2.1, they have been categorised in terms of contribution to greenhouse phenomenon and acidification, toxicity and involvement in Local Air Quality strategies.

- **Carbon monoxide (CO)** is a toxic gas produced by combustion processes and by the oxidation of hydrocarbons and other organic compounds. It reduces the capacity of the blood to transport oxygen and deliver it to the tissues. Figure 2.1 illustrates the trends in CO emissions by emission source from 1970 to 2004. The introduction of catalytic converters on petrol vehicles and, to some extent, the increasing numbers of diesel cars are the main reasons for the dramatic reduction of total emissions after 1990. The diagram in Figure 2.2 shows that the prevailing vehicle type has logically been private car and consists, thus, the predominant source of traffic-related CO pollution.

Table 2.1 Air pollutant types (including greenhouse gases).

Pollutant	Greenhouse gas	Acid gas	Ozone precursor[1]	Toxic pollutant[2]	Local Air Quality
Carbon dioxide CO_2	x				
Methane CH_4	x		x		
Nitrous oxide N_2O	x				
Hydrofluorocarbons *HFC*	x		x		
Perfluorocarbons *PFC*	x		x		
Sulphur hexafluoride SF_6	x				
Nitrogen oxides *NOx* ($NO_2 + NO$)	*indirect*	x	x		x
Sulphur dioxide SO_2	*indirect*	x			x
Particulates PM_{10}					x
Black smoke *BS*					x
Carbon monoxide *CO*			x		x
Ozone O_3					x
Non-methane volatile organic compounds *NMVOC*	*indirect*		x		x
Benzene					x
1,3 butadiene					x
Ammonia NH_3		x			
Hydrogen chloride *HCl*		x			
Hydrogen fluoride *HF*		x			
Arsenic *As*				x	
Cadmium *Cd*				x	
Chromium *Cr*				x	
Copper *Cu*				x	
Mercury *Hg*				x	
Nickel *Ni*				x	
Lead *Pb*				x	x
Selenium *Se*				x	
Vanadium *V*				x	
Zinc *Zn*				x	
Persistent organic pollutants *POPs*				x	

1 Ozone is produced by photochemical reactions involving VOCs & NO_x in the lower atmosphere.
2 Includes heavy metals & POPs.

Source: Department for Environment, Food and Rural Affairs (DEFRA, 2006)

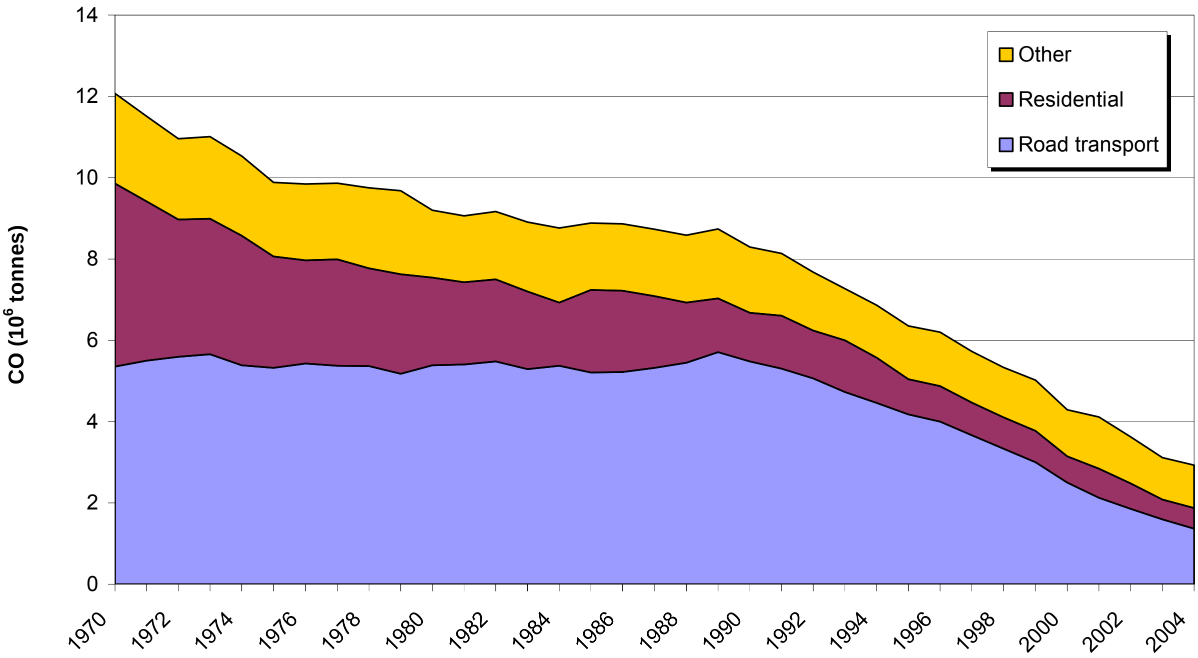


Figure 2.1 Estimated CO emissions by source. UK, 1970-2004.
(adapted from DEFRA, 2006)

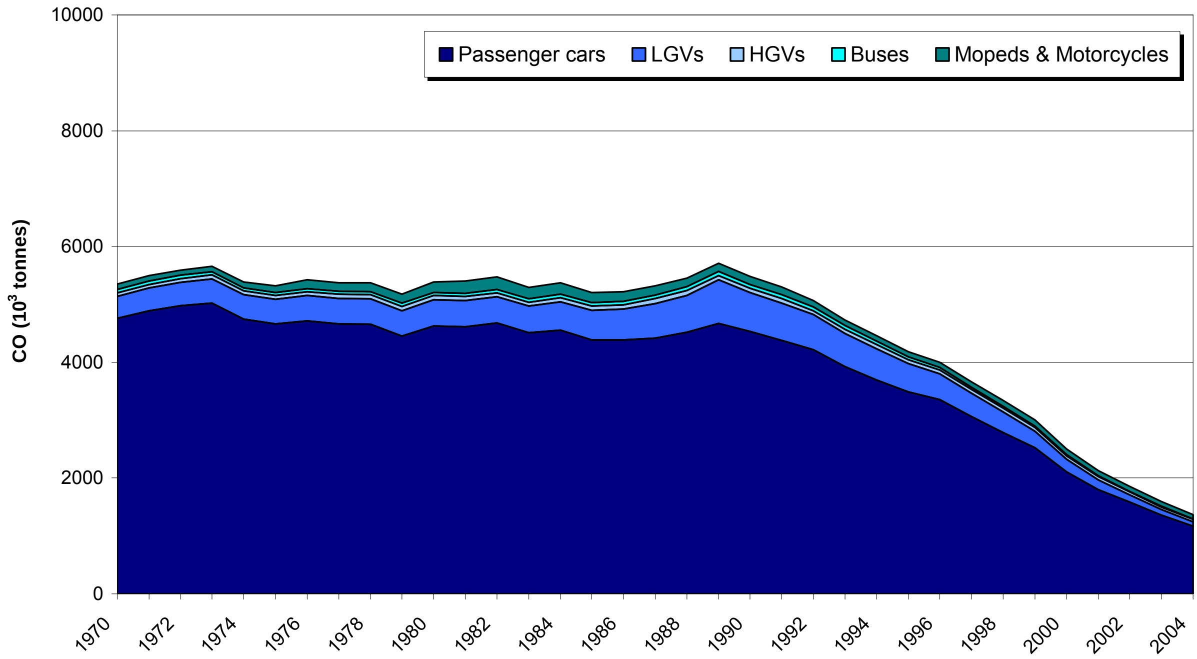
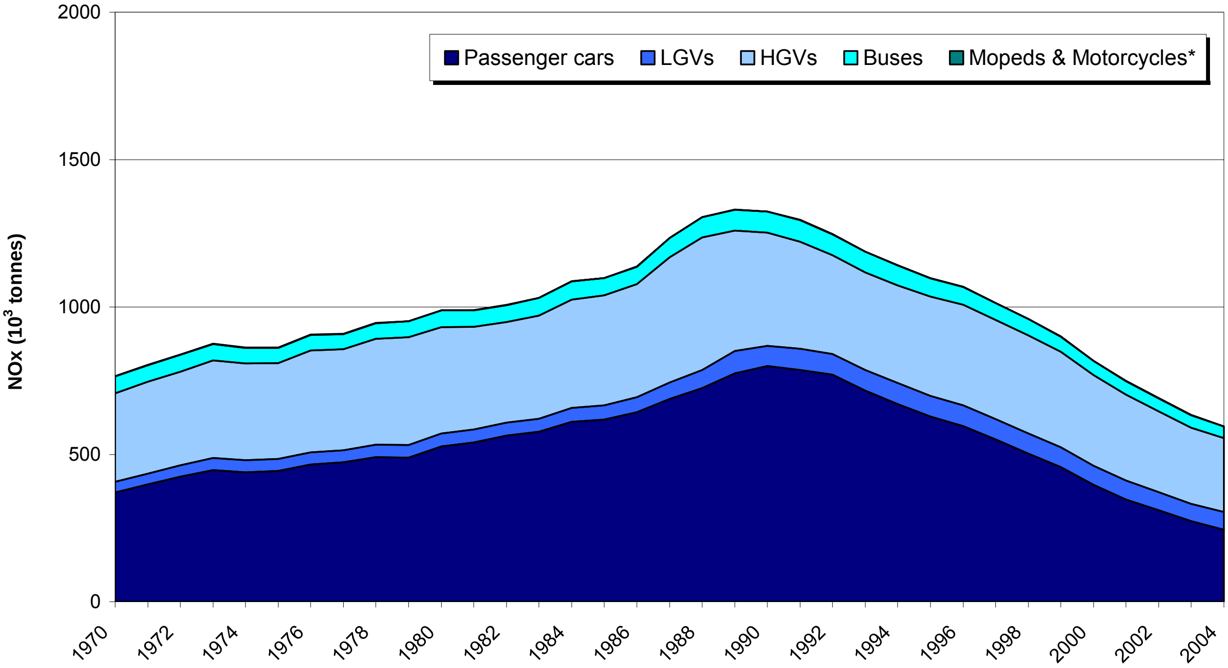


Figure 2.2 Estimated CO emissions by vehicle type. UK, 1970-2004.
(adapted from DEFRA, 2006)

Nitrogen oxides (NO_x) are acid gases and ozone precursors contributing largely to local air pollution. The emissions can have negative effect on human health and vegetation. Nitrogen dioxide (NO_2) is believed to affect airways and lung function both acutely and chronically, particularly in people with asthma. Figure 2.3 shows estimated emissions of NO_x by source category for the period 1970-2004. Total emissions reached their maximum in late 1980s, but then considerably declined between 1990 and 2004, mainly due to the use of three way catalysts for petrol cars and also cleaner operation of major combustion plants. The shift from some non-catalyst petrol to diesel cars has also played a smaller part.



* Approximately 1,000 tonnes per year

Figure 2.3 Estimated NO_x emissions by vehicle type. UK, 1970-2004. (adapted from DEFRA, 2006)

- **Ground level ozone (O_3)** is a natural atmospheric component, but its concentration can rise, when reactions between NO_x, oxygen and Volatile

Organic Compounds (VOCs) occur in the presence of sunlight. The formation of ozone as a secondary pollutant is widely affected by the weather and episodes of high levels usually occur during summer due to the prolonged sunshine, high temperatures (>20°C) and calm winds. Ground level O_3 can persist for several days and can be transferred long distances. Ozone concentrations are generally higher at rural sites than in urban areas, because nitric oxide (NO), which is quite common in cities, can react with and deplete O_3 to from nitrogen dioxide (NO_2). Possible effects on human health include eyes and nose irritation and even damage of the airway lining in case of exposure to high levels.

- **Airborne Particulate Matters (PM)** is a very diverse category of air pollutants, including microscopic compounds with various physical and chemical properties and of many forms. Apart from motor traffic, the particles can be formed from power generation, construction work, gas-to-particle conversion and other photochemical reactions. Road transport in the UK now contributes 23% of all PM_{10} emissions with the main sources being exhaust of diesel vehicles, automobile tyre and brake wear, and resuspension of road dust and soil particles. Figure 2.4 shows trends in PM_{10} emissions by vehicle type for the last decades. The drastic reduction since 1990 is generally due to modern low-emission engines and particulate traps installed in heavy duty vehicles. The airborne particles size plays important role in their behaviour and the potential hazards against human health. Fine particles, 10μm in diameter or smaller, can penetrate deep into the lungs and their accumulation might cause premature deceases among people with pre-existing lung and heart disease. At present, ultrafine (<0.1 μm) and nanoparticles (<0.05 μm) constitute a growing field of research, as they might consist the main fraction of modern engine emissions and their environmental and health impacts may be stronger than that of fine particles (Le Bihan et al.).

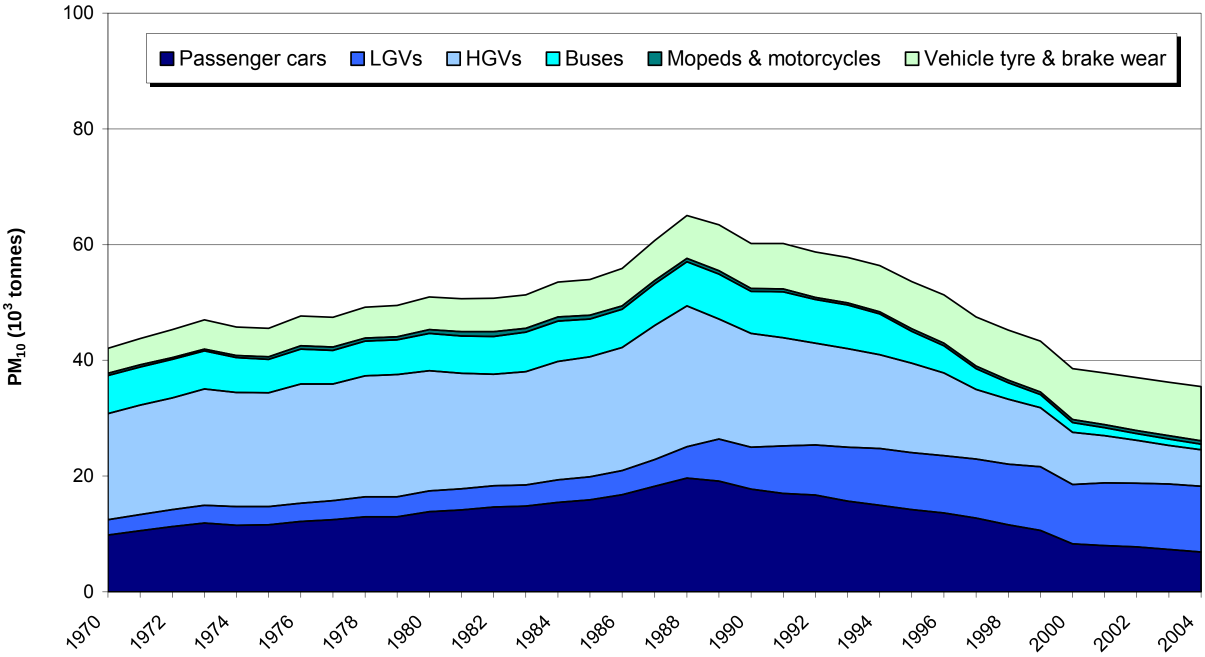


Figure 2.4 Estimated PM_{10} Emissions by vehicle type. UK, 1970-2004. (adapted from DEFRA, 2006)

- **Volatile Organic Compounds (VOCs)** are ozone precursors and include a wide variety of chemical compounds, such as hydrocarbons (alkanes, alkenes, aromatics), oxygenates (alcohols, aldehydes, ketones, ethers) and halogen containing species. The notable decrease of road transport VOC emissions since 1990 is mainly due to the introduction of catalytic converters for petrol cars and, to some degree, to replacement of non-catalyst petrol cars with diesel vehicles. Except for fossil fuels burning, a significant proportion of VOCs come from sources, such as motor fuel and solvent evaporation, refining of petrol and other industrial processes, and natural sources. The most important environmental impact of non-methane VOCs relates to their contribution to ground level O_3 formation, but they may also cause various health effects. Most VOCs are non-toxic, but others like benzene and 1,3-butadiene are carcinogenic and have damaging effects on human health.

- **Sulphur dioxide (SO_2)** is an acid gas and is principally produced by fossil-fuelled power stations, but also from fuel use in manufacturing industries and construction. Total SO_2 emissions in the UK fell dramatically between 1990 and 2004. Emissions from power stations declined by 82%, largely as a result of a reduction in coal use and introduction of outlet gas desulphurisation plants. Emissions from motorised traffic have decreased by 85% since 1998 due to reduction in the sulphur content of fuel. Sulphur dioxide can have negative effects on human health and particularly the lining of the nose, throat and airways of the lung, and among people with asthma and chronic lung disease.

- Among the main air pollutants belong also: (i) **Ammonia**, which causes nitrogen enrichment and potentially acidification, and may also contribute to the creation of particulate matters through atmospheric chemistry. (ii) **Hydrogen chloride** and **hydrogen fluoride**, which are both acid gases. (iii) **Lead** and other **heavy metals**, which can cause, among a range of health problems, deterioration of the immune, the metabolic and the nervous system, and many are known or suspected carcinogenic substances. (iv) **Persistent organic pollutants**, which include: polycyclic aromatic hydrocarbons, polychlorinated biphenyls, dioxins and furans; trace quantities can be found in all areas of the environment and they have varying levels of toxicity and an accumulative effect in humans and vegetation.

2.2 National Air Quality Strategy

This section incorporates the general legislative framework regarding vehicle emissions and the acceptable limit values for pollution levels. As illustrated above, emissions and concentrations of most airborne pollutants have declined extensively as a consequence of the implementation of national and local measures in UK.

According to the 1995 Environment Act, the UK Government and the devolved administrations for Scotland and Wales are responsible for policy and legislative issues related to the environment and air quality. The transboundary nature of many air pollutants, however, encouraged the formation of a UK-wide air quality strategy with common aims for all parts. The 2000 National Air Quality Strategy (NAQS) for England, Scotland, Wales and Northern Ireland was produced on that basis and included standards and objectives for improving ambient air quality. The Act also initiated the concept of local air quality management (LAQM), where local authorities are required regularly to review and evaluate the current and future pollution levels in their areas against the prescribed targets in the Strategy (DEFRA, 2003).

The National Air Quality Strategy contains the governmental policies for reducing air pollution and introduces standards for certain air pollutants, which are set to be achieved by certain target dates. These pollutants were revised in the NAQS Addendum (published in 2003) and now comprise of: sulphur dioxide (SO_2), fine particles (PM_{10}), nitrogen dioxide (NO_2), carbon monoxide (CO), lead (Pb), benzene, 1,3-butadiene, ground level ozone (O_3) and polycyclic aromatic hydrocarbons (PAHs). The updated objectives are summarised in Table 2.2 (DEFRA, 2006).

The main goals of this initiative as outlined in the updated NAQS Addendum (DEFRA, 2003) are to:

- map out future air quality policy in the UK in the medium term and to the greatest possible extent;
- provide realistic protection to human health by setting health-based limits for air pollutants;
- set objectives to protect the natural environment, i.e. vegetation and ecosystems;
- describe present and future levels of air pollution; and
- provide a framework to help recognise what can be done at local, national and international level to improve air quality.

Table 2.2 New air quality objectives included in the NAQS for protecting human health.

Pollutant	Objective	Concentration[1] measured as:	To be achieved by
Benzene	**16.25μg/m³** (5ppb) in England & Wales	running annual mean	31/12/2003
	5μg/m³ (1.54 ppb) in England & Wales	annual average	31/12/2010
	3.25μg/m³ (1 ppb) in Scotland & N. Ireland	running annual mean	31/12/2010
1,3-Butadiene	**2.25μg/m³** (1ppb)	running annual mean	31/12/2003
Carbon monoxide	**10 mg/m³** (8.6 ppm) in England, Wales & N. Ireland [2]	maximum daily running 8-hour mean	31/12/2003
	10 mg/m³ (8.6 ppm) in Scotland	running 8-hour mean	31/12/2003
Lead	**0.5μg/m³**	annual mean	31/12/2004
	0.25μg/m³	annual mean	31/12/2008
Nitrogen dioxide	**200μg/m³** (105ppb) not to be exceeded more than 18 times a year [3]	1-hour mean	31/12/2005
	40μg/m³ (21ppb)	annual mean	31/12/2005
Ozone	**100μg/m³** (50ppb) not to be exceeded more than 10 times a year [3]	maximum daily running 8-hour mean	31/12/2005
Polycyclic aromatic hydrocarbons	**0.25ng/m³** B[a]P [3],[5]	annual average	31/12/2010
Sulphur dioxide	**266μg/m³** (100ppb) not to be exceeded more than 35 times a year	15-minutes mean	31/12/2005
	350μg/m³ (132ppb) not to be exceeded more than 24 times a year	1-hour mean	31/12/2004
	125μg/m³ (47ppb) not to be exceeded more than 3 times a year	24-hour mean	31/12/2004
Particles	**50μg/m³** not to be exceeded more than 7 times a year in UK (apart from London) & 10 times a year in Greater London [3],[4]	24-hour mean	31/12/2010
	20μg/m³ in England (apart from London), Wales & N. Ireland [3],[4]	annual mean	31/12/2010
	18μg/m³ in Scotland [3],[4]	annual mean	31/12/2010
	23μg/m³ in Greater London [3],[4],[6]	annual mean	31/12/2010

(1) Conversions of ppb and ppm to μg/m³ and mg/m³ at 20°C and 1013mb
(2) During 2002 the original strategy objective for CO was replaced by a more stringent one
(3) Objectives are provisional
(4) Measurements are in gravimetric units
(5) N. Ireland adopted the same PAHs objective as England, Wales and Scotland in April 2004
(6) It is proposed that London should work towards a 20μg/m³ annual mean target after 2010, with the aim of achieving it by 2015, where cost effective and proportionate local action can be identified.

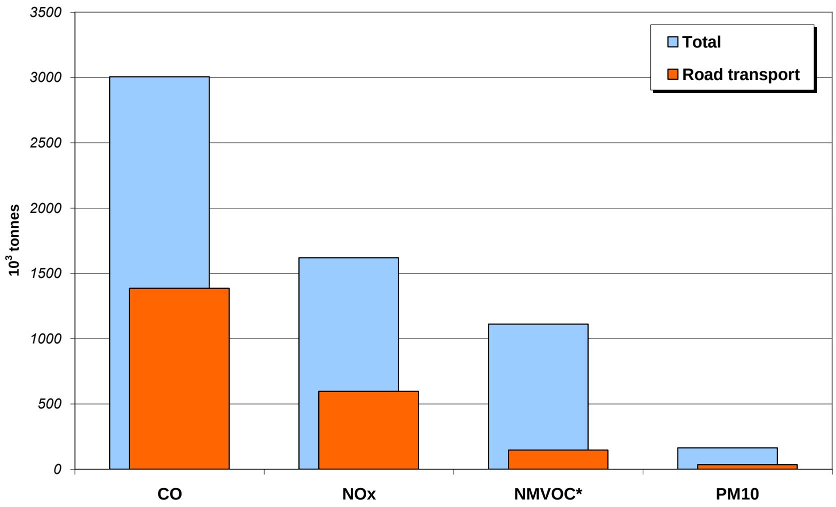
Source: DEFRA, 2003 & 2006

2.3 Vehicle emissions

Vehicle emissions are mainly originated from the fuel burning engine operation, which transmits to the tail pipe a variety of elements and chemical compounds. The primary ones are: carbon dioxide, a mixture of NO_x (of which more than 90% is in the form of NO and the rest is NO_2), carbon monoxide from incomplete burning, as well as multiple unburned and chemically transformed hydrocarbons, such as benzene and 1,3-Butadiene, methane (CH_4) and other alkanes, and other complex polymeric aromatics. Additional products are particulate matters of different size and composition, mainly of condensed carbon material, emitted by diesel and by poorly maintained petrol vehicles. Road traffic emissions also include other processes, such as direct evaporation of volatile fuel components and leaks from fuel tank, as well as dust particles from tyre and brake wear and from re-suspended material of road surface erosion. (Vardoulakis et al., 2003; Ropkins, 2006).

Such traffic-related emissions were considerably high in the last decades due to the vast growth of road transport. Motorised traffic has, thus, been converted into the primary source of air pollution, when studying congested urban networks or industrialised regions. On the other hand, at the same time the air pollution emitted from manufacturing and domestic sources has generally declined to a great degree due to the various governmental Acts and the stricter regulations introduced in the majority of the developed countries (Sharma and Khare, 2001).

As shown earlier though, emission levels of main pollutants have been reduced significantly over the recent years mostly as a result of the stricter vehicle emission and fuel quality EURO standards and in spite of the further growth of motor traffic (15% since 1990). The graph in Figure 2.5 is illustrative of the current atmospheric emission levels in the UK and of the mitigated share of traffic induced emissions. Technological innovations can contribute to further emissions reduction over the long-term, but the large proportion of older vehicles that remain on the road networks and the increasing trends of traffic volumes are likely to have an adverse effect on such improvement (DEFRA, 2006).



*Non-methane VOCs incl. benzene and 1,3-butadiene.

Figure 2.5 Total and traffic-related emissions of key air pollutants in UK for 2004. (Source: NETCEN & Office for National Statistics; ONS, 2006)

The basic principle in the emissions calculation method is that the core vehicle emissions comprise of the various exhaust gases and the VOCs produced by fuel evaporation. An additional fundamental assumption is that engine emissions are higher when an engine is started, because it runs below its normal operating temperature and the fuel is used inefficiently. The total emissions are, thus, the sum of the hot engine and cold start emissions plus the possible evaporative losses (EC-MEET, 1999):

$$E_{TOTAL} = E_{HOT} + E_{COLD} + E_{EVAPORATIVE}$$

Each of these components is, however, affected by several parameters and modification factors:

- The vehicle type, model and speed, the engine size and pollution control, the age, mileage and maintenance level of the vehicles are important factors specifying the quantity and quality of the emissions. The passenger cars versus heavy duty vehicles or the use of three-way catalytic converter in the exhaust system are relevant classification criteria.

- The type and grade of fuel used, e.g. petrol, diesel, LPG or biodiesel, and standard versus high octane fuel, also play important role in engine emissions.
- The driving activity includes: urban, rural or motorway trips (i.e. average speed, congestion, stops/starts, trip length); driver's behaviour (e.g. normal, passive or aggressive driving); high or low engine load demand (driving uphill or downhill, vehicle load or use of in-car air conditioning system); and 'cold start' versus 'warm/hot' vehicle operation. Each of these variables can differentiate the amount of emissions from the same vehicle.
- External factors, such as road condition (quality of road surface, road gradient etc.) and ambient air temperature, pressure, humidity and other environmental parameters also determine the exhaust emissions (Ropkins, 2006; Namdeo, 1995).

The calculation of vehicle emissions is clearly a complex and multipart procedure. In general, exhaust gas emissions are the most easy to monitor and, thus, to accurately model. On the other hand, pollutants emitted from other points of the vehicle (e.g. vaporisation of fuel spirit during refuelling, diurnal breathing, hot soak and running losses) or from the passage of vehicles (brake/tyre/road-surface wear, re-suspended road dust etc.), cannot be easily quantified and they are, hence, poorly represented in emissions modelling.

Furthermore, the above mentioned parameters that can be quantified or accurately estimated, are only a small proportion of all the possible combinations of factors that influence vehicle emissions. Most of the emission models use different emission factors[1] for only certain conditions and apply appropriate quantifiable correction factors to the basic calculations (Ropkins, 2006).

[1] Emission factor is the amount of emission produced in a given period under certain conditions divided by that time period or the by distance travelled/fuel consumed during that period (Ropkins, 2006).

A major category of emission models contains those that utilise average speed-emission relationships to estimate emissions during a trip. The European COPERT III (COmputer Programme to calculate Emissions from Road Transport III) is a widely used model, which calculates total emissions by combining driving activity data for each vehicle class with emission factors suitable for each driving situation, climatic conditions etc. An outline of the parameters required and the intermediate computed values is provided in the flow chart of Figure 2.6. The varying driving conditions play an important role, as they influence engine operation conditions and, thus, cause divergent emission profiles. Hence, the erratic driving and emission performance in each trip is further represented by distinguishing urban, rural and motorway driving situations. Each category has different average speeds and, thus, differing activity data and emission factors. The cold start emissions are calculated additionally to the 'hot emissions', which occur after the engine and the catalyst have warmed-up. The impact of cold start over-emissions is mainly attributed to urban driving, because the large majority of trips are assumed to start within urban areas contributing, thus, to higher pollution levels there (Ntziachristos and Samaras, 2000). When calculating emissions from urban driving, it is also important to consider the effect of driver behaviour, which can be quantified by dividing the modelled journey according to driving mode, i.e. idling, accelerating, constant-speed cruising and decelerating. The associated emission factors for each operating mode are different. This modal method can show the effect of traffic conditions, particularly at congested intersections, as well as the impacts of traffic management schemes (e.g. traffic signal coordination).

Instantaneous emission models are another key tool used to estimate vehicle emissions. Measuring campaigns of instantaneous emission data are quite rare and elaborate, so by use of these models it is possible to predict second-by-second exhaust emissions for a variety of vehicle types, engine technologies and emission control devices. Such models can accurately estimate emissions during the acceleration mode, which is the period with the highest emission rates for modern engine vehicles. However, they require a large amount of input data and the appropriate vehicle and engine operation parameters must be specified or calculated in advance (Tate, 2006).

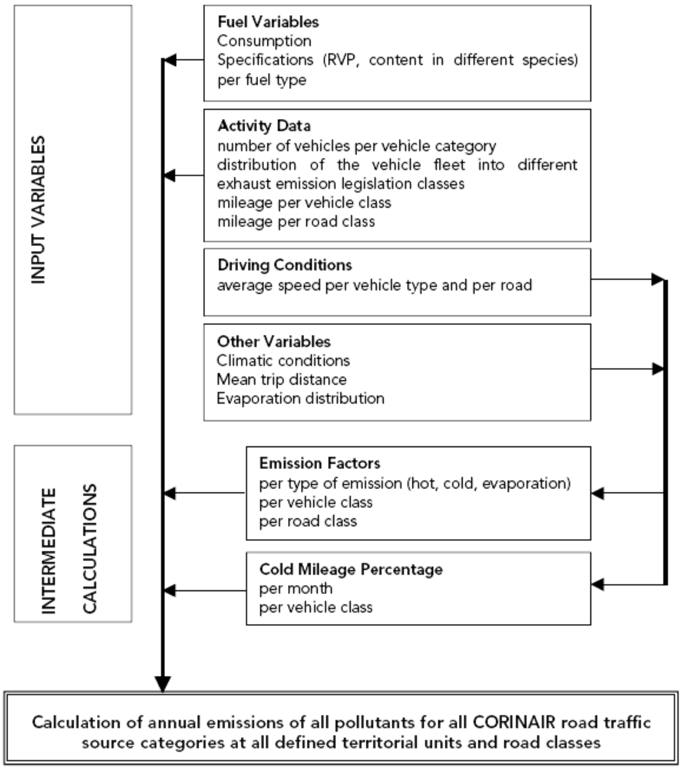


Figure 2.6 Flow chart of the application of COPERT-III baseline methodology. (Source: Ntziachristos and Samaras, 2000)

2.4 Dispersion of air pollution in urban areas

Having estimated the vehicle emissions, the next step is to examine the pollutants dispersion in order to calculate their concentrations across the examined area. Air pollution problems have typically occurred at local scale, i.e. in the surroundings of sporadic point or area sources. More recently, environmental policy has had to address also global scale problems, such as the greenhouse effect, climate change, ozone depletion etc. Table 2.3 contains some key policy issues related to the atmospheric environment including, among others, acidification, photo-oxidant formation, air toxics and the modern problem of urban air pollution.

Table 2.3 Environmental policy issues and analogous scales of dispersion phenomena.

Policy issue	Scale of dispersion phenomenon			
	Global	Regional-to-continental	Local-to-regional	Local
Climate change	x			
Ozone depletion	x	x		
Tropospheric ozone		x		
Tropospheric change		x		
Acidification		x		
Nutrification		x		
Summer smog		x	x	
Winter smog		x	x	
Air toxics		x	x	x
Urban air quality			x	
Industrial pollutants			x	x
Nuclear emergencies		x	x	x
Chemical emergencies		x	x	x

Source: Moussiopoulos et al., 1999

The dispersion of air pollution largely depends on processes in the atmosphere which are differentiated with regard to their spatial and temporal scale. These scales extend from macroscale (typical lengths >1,000km), including global and most of regional-to-continental phenomena, where the air flow is largely related to synoptic phenomena (e.g. the geographical distribution of pressure systems) to microscale (typical lengths <1km), where atmospheric flow is very complex and depends strongly on the detailed surface features. Local scale dispersion incidents belong to microscale and can be well described in appropriate simulation modelling tools (Moussiopoulos et al., 1999).

In such models the factors that typically have to be considered are: (i) the shape and size of buildings and other obstacles, (ii) their position with regard to the wind direction, (iii) the variability in wind speeds and directions, (iv) the amount of mechanical turbulence mainly dependent on the vehicle induced mixing and the formation of air vortices because of the surface roughness (see Fig. 2.7), (iv) the thermal turbulence effect, which is determined by the temperature structure of the lower atmosphere and relates to the vertical convection of heat.

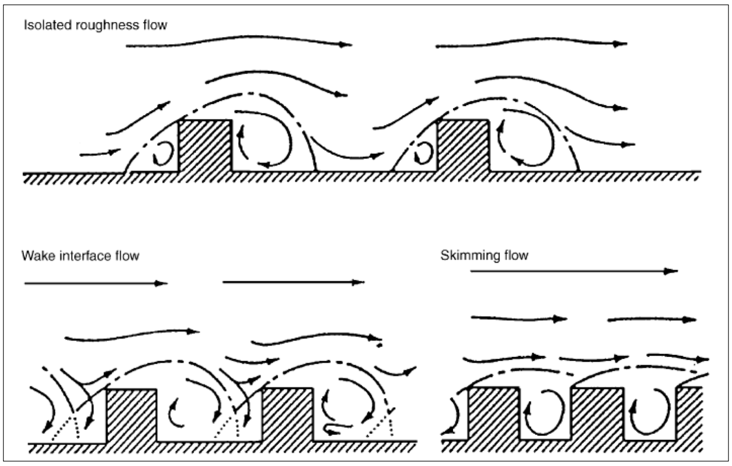


Figure 2.7 Perpendicular flow systems in urban areas for different road widths. (adapted from Vardoulakis et al., 2003)

Street Canyons

An interesting microscale dispersion phenomenon occurs at urban street canyons, which are ideally narrow streets between buildings arrayed continuously along both sides. This distinctive configuration impedes good ventilation when wind is perpendicular to the street axis and, thus, results in poorer dispersion and often higher pollutant concentrations than current air quality standards (Nicholson, 1975).

The cross-section, illustrated in Figure 2.8, is of a typical urban street canyon flanked by buildings on both sides. In reality, a wide range of geometries can be found with varying structure heights and shapes along the street, with gaps between the buildings or even with buildings only on one side. In this case, the effect of a regular, symmetric street canyon is showed, with an aspect ratio approximately equal to one[1] and buildings of similar height on either side. When the above roof-level wind direction is perpendicular to the street, the wind flow skims over the canyon producing a primary air vortex between the buildings, which is the main dispersion mechanism for the air pollutants. The diffusion is generally affected by the rate of the fresh air exchange

[1] i.e. Average Buildings Height/Average Street Width ≈ 1

between the street and its outlets, i.e. the above roof level and the connecting roads. The recirculating wind flow cannot remove effectively the street-level pollutants and, in addition, it contributes to the formation of substantial pollution gradient across the street canyon, as also verified by field measurements. The leeward side of the canyon (see Fig. 2.8) shows overall higher concentrations than the windward due to the transferred pollutants by the main wind vortex. Secondary weaker vortices can be created in the bottom side corners of the canyon or in small cavities causing, thus, localised pollution hotspots. The strength of the canyon vortices largely depends on the speed of the roof-level wind, but it is also affected by the vehicle induced mechanical and the thermal turbulence or the reflection off the various obstacles within the street (e.g. trees, kiosks, balconies etc.).

More complex wind flows occur near the ends of the canyon at intersections with other streets. There, the low-pressure corners and the wind circulation create horizontal air vortices which bring fresh air into the canyon. This ventilation mechanism, however, becomes weaker as the street length increases. Similarly complex flow channelling effects occur near gaps between the canyon buildings and also within asymmetric canyons, where the down-wind and up-wind buildings have different heights (Vardoulakis et al., 2003).

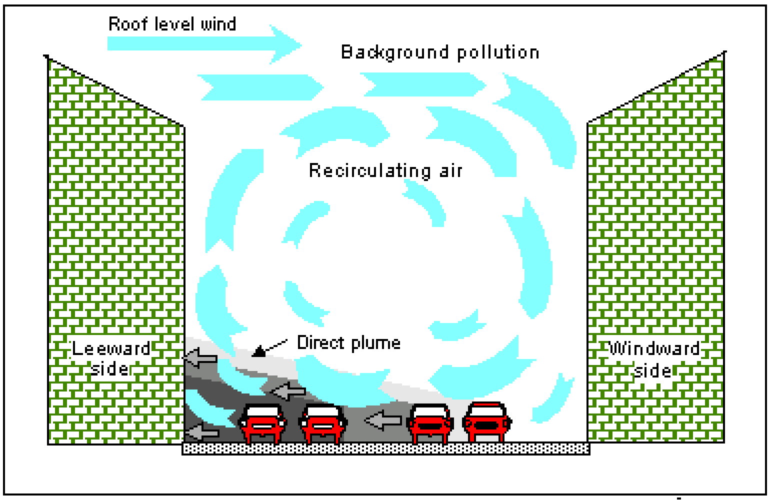


Figure 2.8 Vertical cross-section of a typical symmetric urban street canyon. The recirculating wind flow is shown in the case of perpendicular roof level wind. (Source: NERI, 2006)

2.5 Monitoring traffic pollution

The objectives of air quality strategy are based on constantly updated air pollution measurements from the national air pollution monitoring network and the most up-to-date modelling tools of air quality level predictions. More than 1600 national AQ monitoring sites are located around the UK. They are divided into automatic and non-automatic network monitors and collect several types of data depending on the local requirements. Most recent developments include the dissemination to the public of hourly collected data from approximately 120 automatic monitoring points, the operation of sites, which are capable of collecting detailed information on ultra-fine particles, as well as an increased number of monitoring sites of PAHs (DEFRA, 2003).

Nevertheless, the amount of permanent air quality monitoring stations in a city is practically limited because of the initial and operational costs, the size and shape of the equipment, the power supply possibility etc. It is, hence, important to ensure the optimal utilisation of the available monitoring equipment and of the obtained data, which can be supplemented via alternative measurements and modelling techniques in order to fully assess air quality and population exposure in dense built-up areas. According to the EU legislation on air quality, monitoring stations should be sited where the highest concentrations and human exposure risks occur. Confined urban environments should be avoided, so that the receptor measurements can represent the air pollution levels in a surrounding area of at least $200m^2$. Other guidance requires that the height of the monitoring point is between 1.5m (i.e. human breathing zone) and 4m, no less than 25m away from major junctions and than 4m from the middle of the nearest traffic lane. For NO_2 and CO, the sampling inlet should be less than 5m from the kerbside, and for PM_{10} and benzene it should be placed near the building façade (but no less than 0.5m from the nearest wall) (Vardoulakis et al., 2005).

Several techniques have been developed for monitoring air pollutants (e.g. continuous monitoring using standard gas analysers or diffusive and pumped sampling using tubes filled with an appropriate adsorbent, etc.) and particles (e.g. filtration and impaction), each of them with advantages and disadvantages that make it proper or not for a specific use. The response time (i.e. the time interval in which the sample is taken), is a key criterion for the suitability of the selected method. Standard gas analysers are sensitive and fast enough for real time measurements of CO, NO_x and O_3 concentrations. Appropriate averaged results can then be produced over short time periods so that they are comparable with the regulatory standards. On the other hand, diffusive samplers have a fairly long response time (e.g. one/two days to four weeks), which makes them preferable when sampling substances with cumulative effects on human health (e.g. benzene) or for spatial variation measurements, air quality mapping and personal exposure studies, since they are portable devices (Vardoulakis et al., 2003).

Having compared monitoring data from two streets in Copenhagen, Berkowicz et al. (1996) demonstrated that roadside measurements are highly site dependent, even within the same street. Other studies have also shown the dependence on local wind flows and the interaction with street and buildings geometry. So, it is very important to focus on the ability of sampling points to characterise traffic pollution in complex urban areas and also to be combined with modelling tools. The ideal urban air quality management would avoid costly monitoring campaigns as well as excessive dependence on models, by maximising the representativeness of AQ monitoring stations and following the existing location criteria for roadside receptors (Vardoulakis et al., 2005).

Vardoulakis et al. (2003) suggest two main categories of permanent AQ stations in a city: (a) the roadside and (b) the urban background stations. Roadside stations are usually located on the pavement of busy streets, avenues or junctions (as described earlier), whereas background stations are placed in parks or other urban locations away from road traffic.

2.6 Modelling traffic pollution

Several modelling studies, related to the effect of buildings and other urban structures on pollutant concentrations and dissipation patterns, were driven by the increasing need for research on the impacts of air pollution in urban environments. Dispersion models are now extensively used for assessing roadside air quality by predicting current and future pollution levels, as well as their temporal and spatial variations. When used in a knowledgeable way, they are able to provide most detailed information about the physical and chemical processes that govern the diffusion and transformation of atmospheric substances (Vardoulakis et al., 2003).

2.6.1 Types of air quality models

Models describing the dispersion of air pollution in the atmosphere can be distinguished in many ways, such as: their spatial (global, regional-to-continental, local-to-regional or local) and temporal scale (episodic or statistical, long-term models), the treatment of the transport equations (Eulerian, Lagrangian models etc.) and of the different processes (chemistry, wet and dry deposition). Table 2.4 summarises the main existing model types and their brief descriptions.

Since late 1950's atmospheric dispersion models based on Gaussian distribution and Pasquill-Gifford classes have been used for legislative reasons in Europe. During the last two decades dispersion models have been developed based on boundary layer parameterisation occurred due to increasing understanding of both the structure of the boundary layer and the dispersion science. The meteorological input data to these models are produced via new methods where the vertical profiles of speed, temperature and turbulence are dependent on the height of the boundary layer and a Monin-Obukhov length scale determined by the temperature, the friction velocity and the heat flux. More reliable and accurate modelling results are required due to the expanding utilisation of practical operational models for regulatory and planning purposes. The air quality guidelines are becoming increasingly stricter and detailed in many countries and evaluated models are required to meet the needs of modern air quality management, especially in urban areas (Moussiopoulos et al., 1996).

Table 2.4 Main types of existing air quality models.

Model type	Description
Plume-rise models	In most cases, pollutants injected into ambient air possess a higher temperature than the surrounding air. Most industrial pollutants, moreover, are emitted from stacks or chimneys and possessing, thus, an initial vertical momentum. Both factors (thermal buoyancy and vertical momentum) contribute to increasing the average height of the plume above that of the smokestack. Plume-rise models calculate the vertical displacement and general behaviour of the plume in this initial dispersion phase. Both semi-empirical and advanced plume-rise formulations are available.
Gaussian models	The Gaussian plume model is the most common air pollution model. It is based on the assumption that the plume concentration, at each downwind distance, has independent Gaussian distributions both in the horizontal and vertical direction. Almost all the models recommended by the U.S. Environmental Protection Agency are Gaussian. Gaussian models have been modified to incorporate special dispersion cases. A simplified version of Gaussian model, the Gaussian climatological model, can be used to calculate long-term averages (e.g. annual values).
Semi-empirical models	This category consists of several types of models which were developed mainly for practical applications. In spite of considerable conceptual differences within the category, all these models are characterised by drastic simplifications and a high degree of empirical parameterisations. Among the members of this model category are box models and various kinds of parametric models.
Eulerian models	The transport of inert air pollutants may be conveniently simulated by the aid of models which solve numerically the atmospheric diffusion equation, i.e. the equation for conservation of mass of the pollutant (Eulerian approach). Such models are usually embedded in prognostic meteorological models. Advanced Eulerian models include refined sub-models for the description of turbulence (e.g. second-order closure models and large-eddy simulation models).
Lagrangian models	As an alternative to Eulerian models, the Lagrangian approach consists in describing fluid elements that follow the instantaneous flow. They include all models in which plumes are broken up into elements such as segments, puffs, or particles. Lagrangian models use a certain number of

	fictitious particles to simulate the dynamics of a selected physical parameter. Particle motion can be produced by both deterministic velocities and semi-random pseudo-velocities generated using Monte Carlo techniques. Hence, transport caused by both the average wind and the turbulent terms due to wind fluctuations is taken into account.
Chemical modules	Several air pollution models include modules for the calculation of chemical transformation. The complexity of these modules ranges from those including a simple, first-order reaction (e.g. transformation of sulphur dioxide into sulphates) to those describing complex photochemical reactions. Several reaction schemes have been proposed for simulating the dynamics of interacting chemical species. These schemes have been implemented into both Lagrangian and Eulerian photochemical models. In Eulerian photochemical models, a three-dimensional grid is superimposed to cover the entire computational domain, and all chemical reactions are simulated in each cell at each time step. In the Lagrangian photochemical models a single cell (or a column of cells or a wall of cells) is advected according to the main wind in a way that allows the injection of the emission encountered along the cell trajectory.
Receptor models	In contrast to dispersion models (which compute the contribution of a source to a receptor in effect as the product of the emission rate multiplied by a dispersion coefficient), receptor models start with observed concentrations at a receptor and seek to apportion the observed concentrations at a sampling point among several source types. This is done based on the known chemical composition of source and receptor materials. Receptor models are based on mass-balance equations and are intrinsically statistical in the sense that they do not include a deterministic relationship between emissions and concentrations. However, mixed dispersion-receptor modelling methodologies have been developed and are very promising.
Stochastic models	Stochastic models are based on statistical or semi-empirical techniques to analyse trends, periodicities, and interrelationships of air quality and atmospheric measurements and to forecast the evolution of pollution episodes. Several techniques are used to achieve this goal, e.g. frequency distribution analysis, time-series analysis, Box-Jenkins and other models, spectral analysis, etc. Stochastic models are intrinsically limited because they do not establish cause-effect relationships. However, statistical models are very useful in situations such as real-time short-term forecasting, where the information available from measured trends in concentration is generally more relevant (for immediate forecasting purposes) than that obtained from deterministic analyses.

Source: Zanetti, 1993

Regarding the requirements for input data into the modelling tools, there are generally two main categories of information required (Moussiopoulos et al., 1996):

- Emissions. For the traffic models, typical descriptive information for the road network work are required as input data. This could be number of cars per day, number of lanes, average driving speed, road gradient and explanation of the surroundings, i.e. open terrain, scattered buildings or street canyon.
- Meteorology. The screening type models use default meteorology data as input, describing a critical meteorological situation. Other models use a set of meteorological cases as input, described in wind and stability classes. The new types of models make use of pre-processed meteorological data based on similarity theory for the atmospheric surface layer. The measurements that are required include variables, such as wind and temperature profiles, cloud cover or solar radiation, surface roughness etc.

The Operational Street Pollution Model

The Operational Street Pollution Model (OSPM) was developed for supplementation to standard monitoring activities and for assessment of abatement strategies at the National Environmental Research Institute (NERI) of Denmark. The model contains a simplified description of flow and dispersion conditions in urban roads. Concentrations of exhaust emissions are calculated using a combination of a plume model for the direct contribution and a box model for the recirculating pollution part in the street. Despite the simplified parameterisation, OSPM is capable of a proper simulation of the dependence of air pollution levels on meteorological conditions, such as wind speed and wind direction. A recent improvement is the modelling of turbulence in the street by taking into consideration the effect of atmospheric turbulence due to wind velocity, but also due to vehicle induced mixing, which dominates for low and calm wind cases. The model includes also a chemical sub-model which is used to calculate the transformation of NO to NO_2. Using actual meteorological data and estimations of emissions, as well as *a priori* assumptions regarding flow and dispersion characteristics, the model provides hourly values of concentrations at predefined receptor locations in the examined street (Gokhale et al., 2005).