

A Driver's Exposure to Traffic Pollution

by

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Abstract

The increasing use of road transport in the post war years has caused a substantial rise in the amounts of traffic related air pollutants and has raised concerns about urban air quality. This thesis describes a series of experiments designed to evaluate the level of traffic pollutants experienced by drivers in heavy traffic.

Field and wind tunnel measurements were initially carried out to assess pollutant transport and dispersion characteristics. These were followed by a series of experiments to determine the ventilation characteristics of a car derived van which was later used in two pollution surveys. The surveys were conducted over two heavily trafficked routes during the Nottingham morning and evening rush hours. Measurements of carbon monoxide and ozone concentrations were taken and the results analysed to give a time integrated exposure for the vehicle occupants.

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Glossary of Abbreviations and Symbols

α	Model yaw angle
τ	Time constant (the time taken for an exponential function to change by a factor e (=2.718))
τ_0	Shear stress of the air at the ground. (Pa)
ρ	Density of air (approximately 1.2kg m ³)
η	Dynamic viscosity of air (1.72 x 10 ⁻⁵ Pa s)
σ_i	The standard deviation of the instantaneous component of velocity at height z
A	Area (m ²)
a	dimensionless pollutant fraction
C _e	External concentration of tracer gas (ppm)
C _m	Measured concentration of pollutant (ppm)
C _s	Emitted concentration of pollutant (ppm)
C _(t)	Internal concentration of tracer gas at time t (ppm)
CFD	Computational Fluid Dynamics
d	Displacement height (m)
E _{cf}	Power lost due to wind chill factor (W)
E _{in}	Power being fed to equipment from a power supply (W)
F	The production rate of tracer by all sources within an enclosure
FEV	Forced expired volume (m ³)
FVC	Forced expiratory vital capacity (The volume of air expired in a forced expiration following a maximum inspiration) (m ³)
hv	The energy (E) of a quantum of radiation of frequency v is given by E=hv where h is Planck's constant
I _i	The instantaneous component of turbulence intensity
i'	The fluctuating component of velocity at height z
K	von Karman's constant (0.40)
k	k-factor

k_j	Jam density, the density of traffic at which the flow speed falls to zero (vehicles / km)
k_t	Traffic density (vehicles / km)
l	Characteristic length (m)
N	Air change rate per unit time
P_t	Total pressure (Pa)
PEF	Peak expiratory flow
PM_{10}	Particulate Matter (10 μm), term applied to suspended particulate matter with an aerodynamic diameter of less than 10 μm .
ppb	Parts per billion 1 part by volume in 10^9
ppm	Parts per million 1 part by volume in 10^6
Q	Specific airflow rate through an enclosure
q	Vehicles per unit time
R_{aw}	Airway resistance
R_e	Reynolds number
RH%	Relative humidity of air (%)
T_d	Temperature difference (K)
t	Time
TWA	Time weighted average
U	Mean velocity (m/s)
U_{car}	Vehicle velocity (m/s)
U_{ref}	Wind tunnel reference flow velocity (m/s)
U_{wind}	Wind velocity (m/s)
U_i	Instantaneous wind velocity (m/s)
$U_{(z)}$	The mean velocity at height z (m/s)
u	Friction velocity (m/s)
u_f	Free flow traffic speed (km/h)
V	Effective volume (m^3)
v	Flow velocity (m/s)
z_0	Roughness length (m)

Organizations

DoE	Department of the Environment, Transport and the Regions
DH	Department of Health
EPA	Environmental Protection Agency (US.)
EPAQS	Expert Panel on Air Quality Standards, reporting to the DoE
EU	European Union
MIRA	Motor Industry Research Association
PORG	Photochemical Oxidants Review Group, reporting to the DoE
QUARG	Quality of Urban Air Review Group, reporting to the DoE
WHO	World Health Organization

Gases and Chemical Compounds

CO	Carbon monoxide
CO ₂	Carbon dioxide
H	Hydrogen
H ₂ O	Water
NO	Nitrogen monoxide
NO ₂	Nitrogen dioxide
NO _x	Generic term for oxides of nitrogen
N ₂ O	Nitrous oxide
O ₂	Molecular oxygen
SF ₆	Sulphur hexafluoride
SO ₂	Sulphur dioxide
VOC	Volatile Organic Compound, a generic term for compounds such as benzene, ethanol and toluene.

Chapter 1. Introduction

1.1 Preamble

The growth of road transport over the last five decades is now seen as a major cause of global airborne pollution. In most of the major cities around the world, traffic pollution is causing a decline in public health, and damage to buildings and monuments. In the United Kingdom the growth in car ownership is causing congestion in our towns and cities which in turn results in increased levels of pollution.

Older, less well maintained vehicles have been shown to emit significantly more pollution than newer vehicles and exhaust emission checks have been introduced as part of the MOT test. This combined with the use of catalytic converters fitted into exhaust systems and “lean burn” engines has reduced the level of some pollutants emitted by a vehicle. However, reductions in pollution effected by these measures may be offset by increased vehicle usage, often for short journeys during which the engine and catalytic converter fail to reach their correct operating temperature. Catalytic converters when working correctly increase concentrations of pollutant gases such as CO₂, the chemical reactions themselves depend on the engine running with the fuel mixture set to the correct ratio (about 14.7:1 depending on the quality of the fuel) and even under perfect conditions they will only catalyse around 90% of the pollutant gases (Environmental Science 1997).

Medical research has shown that exposure to very low concentrations of some gaseous pollutants emitted by motor vehicles can have an impact on health (Environmental Research Inc. 1997 Indoor Air Quality Update 1995). Other pollutants emitted by vehicles (i.e. particulates) may have cumulative effects which manifest themselves during a persons later life. Roadside measurements of these pollutants are well documented, what is less well understood is the exposure of the vehicle occupants.

1.2 Statement of the Problem

As a vehicle travels along a congested road, exhaust pollution from the surrounding vehicles enters the vehicle cabin. The causes of high concentrations of pollutants building up in the vehicle cabin are as follows:

1. proximity of the ventilation air intake vents to the surrounding vehicle exhausts.
2. high external levels of pollutants due to the street “canyon” effect formed by tall buildings.
3. heavily trafficked road junctions where the overall traffic speed is low giving rise to queuing, causing external concentrations to rise.

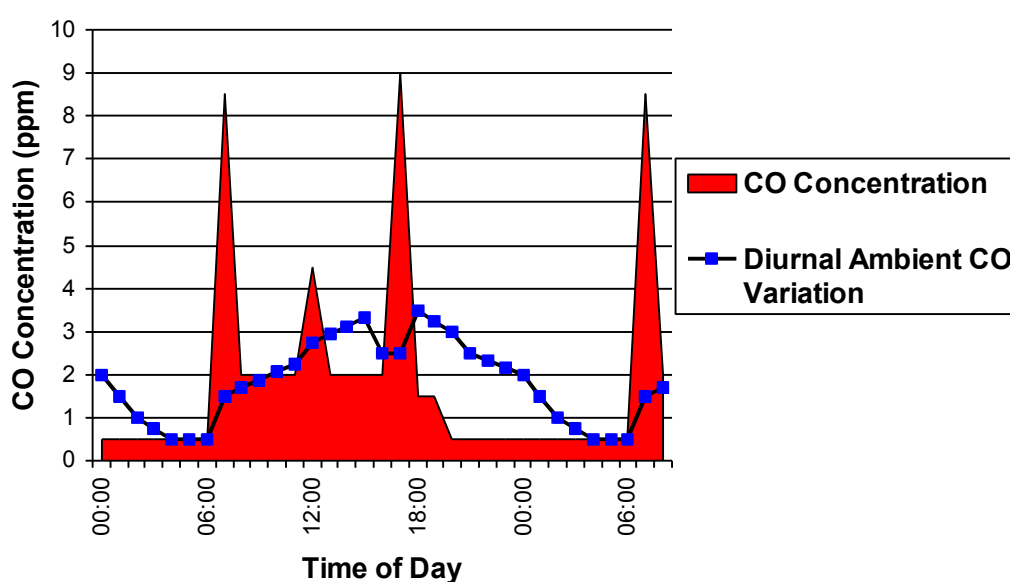


Figure 1.1 Hypothetical CO exposure profile for a commuter over a 24hr. period (After WHO 1982).

Because of this, the level of pollution experienced by the vehicle occupants can be several times higher than that measured at the kerbside. Commuters will often receive between one third to a half of their total daily exposure to vehicle exhaust pollution as they drive to and from work (Figure 1.1). Traffic pollution can give rise to a large number of respiratory problems, as well as causing sore throats and headaches. In addition there is evidence to suggest that exposure to very low concentrations of some of the pollutants emitted by motor vehicles may be detrimental to a persons health (Environmental Research Inc. 1997).

1.3 Background and Reasons for the Present Study

Since the second world war, car ownership has become an integral part of our lives. Indeed the lifestyle of the British people and that of the populations of all other developed countries has come to depend on the car. The growth of car ownership has seen more and more people living further and further away from their place of work, local amenities, schools and shops. During the 1960s the majority of school children walked or cycled to school, many are now transported by car. Out of town shopping malls rely on their customers arriving

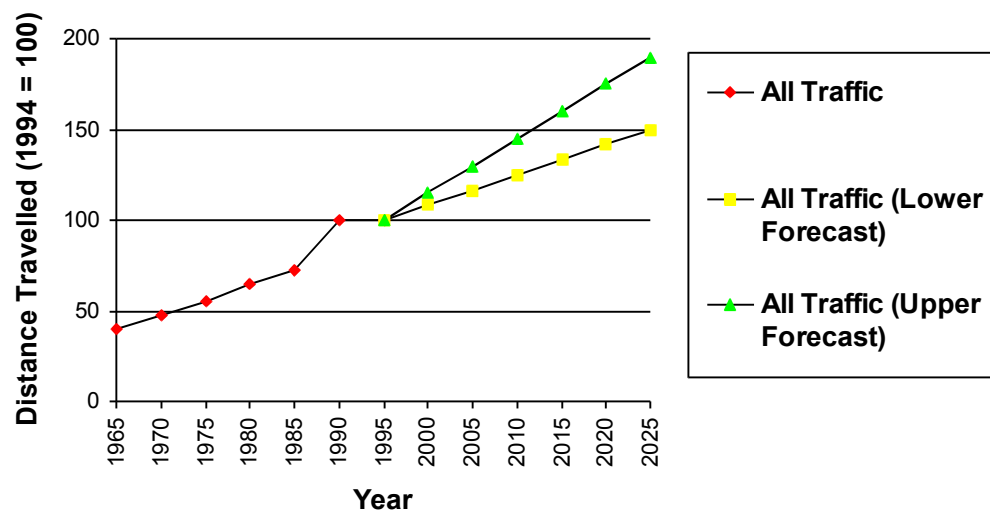


Figure 1.2 Motor vehicle traffic (1965 - 2025)
(Source DoE 1996a)

by car and the same is true of local amenities such as leisure centres. At present (1997) there are around twenty two million cars in the UK. Over the last five years this figure has been increasing by 1.3% to 1.6% per annum (Figure 1.2). During the immediate post war years, estimates of the growth in vehicle ownership did not take into account the dramatic impact that the car would have on our lives. In the 1950s road traffic increased at a rate of 7.8% per annum and during the 1960s growth was 6.0% per annum, grossly outstripping estimates made during the period. As more and more households achieve saturation car ownership the level of growth is falling, but even with a growth of only 1.4% per annum by the year 2005 there will be twenty six million cars in the United Kingdom.

These rapid increases in car ownership have lead to increased levels of congestion on roads, however, this is only a very small and insignificant part of the impact that the car may ultimately have upon all of our lives. Studies have shown that the pollutants emitted from motor vehicles may cause a range of respiratory diseases. The increased combustion of fossil fuels, which is largely due to the increase in road transport, is contributing to global warming and the residues left on the roads which contaminate water courses. Slow moving traffic, especially in urban areas, causes exposure to pollutants at levels that probably have an adverse effect on a person's health. This is the primary reason for this study.

1.4 Aims of the Work

The research project described in this thesis is concerned with investigating the movement of traffic pollution through slow moving traffic queues. The main aim was to quantify the level of traffic pollution experienced by a vehicle occupant during commuter journeys through a congested city. The concentration of gases such as carbon monoxide were measured inside a vehicle cabin under various ventilation conditions. The effect of wind and road speed on internal pollutant concentrations and of the cumulative effect of emissions from more than two vehicles in line were also studied. These were

evaluated using a combination of wind tunnel tests on scale models, field measurements using a vehicle and CFD modelling.

To enable these objectives to be met it was necessary to determine the airflow across a typical vehicle and the ventilation characteristics of the vehicle used for the field measurements. The airflow tests were conducted using 1:10 scale cars, made to a well documented design supplied by MIRA. The ventilation tests on the research vehicle were conducted using conventional tracer gas methods.

1.5 Summary of Contents

Chapter 2 contains the results from the literature search and briefly describes some of the important work carried out over the last two decades. Examples include pollution dispersion measurements conducted in Jerusalem and Tehran. In addition examples of research conducted to evaluate the exposure of commuters to vehicle exhaust and the concentration of pollutants in the vehicle passenger cabin are reviewed.

In chapter 3 the characteristics of the main constituents of traffic pollution are examined and their effects on health are discussed. The generation of atmospheric ozone is also described. This chapter also includes details of techniques used for monitoring gases and a description of the Automatic Air Monitoring Network.

Chapter 4 describes the techniques used to measure airflow, ventilation and pressure. Descriptions of the types of instruments used in the thesis are given, as are techniques for measuring air exchange rates using a tracer gas.

The use of the wind tunnel to investigate airflow over vehicles is described in chapter 5. Scale effects and Reynolds number are discussed as are the simulation of the atmospheric boundary layer and turbulence. Details of the wind tunnel and the various models used during the research are included.

Chapter 6 describes a typical vehicle ventilation system and gives an overview of vehicle ventilation. The chapter also presents the results from experiments to model ventilation through a single opening and to measure the effectiveness of the test vehicle's ventilation system. A brief description of the airflow around a vehicle body is also included.

Chapter 7 describes experiments conducted to evaluate the dispersion of carbon monoxide which comprises both wind tunnel and field measurements. A wind tunnel study of pollutant flow across a traffic queue is also included.

Chapter 8 presents the results from mobile pollution surveys carried out using the experimental vehicle. The method of estimating relative traffic density is explained and the survey results are presented.

Chapter 9 concludes the thesis by presenting the authors conclusions regarding the experimental work and possible future research.

Chapter 2. Review of Published Research

2.1 Introduction

Research to quantify the likely effects of, and exposure to traffic pollution has been carried out since the 1930's. American cities situated along the Western Seaboard such as Los Angeles suffered from increasing levels of photochemical smog. In 1950 Haagen-Smit stated that the smog's were due to increased levels of car usage (Nader 1966). This prompted research into the effects of breathing air heavily contaminated with traffic related pollutants, and has resulted in California having some of the most stringent vehicle emission legislation in the world. Nader (1966) states that the American automobile industry knew of atmospheric problems along the Western Seaboard during the 1920's when tyres and other rubber products were found to deteriorate rapidly. High concentrations of ozone, a product of photochemical reactions in smog, being the culprit for the degradation.

Due to the atmospheric problems in the western United States, research has tended to focus on this location with much of the published work coming from Californian universities. In the United Kingdom almost all of the published research is concerned with ambient levels of pollution and the concentrations experienced by pedestrians. This work has increased over the last two decades and data is now easily available to allow forecasts of air quality to be made with reasonable accuracy.

The papers selected for inclusion in this chapter have all been published over the last two decades, and have been subdivided to bring together papers and articles which cover similar areas of research.

2.2 Street Level Pollution

Traffic pollution is traditionally monitored using kerbside monitoring stations. Such stations may be located either on busy main thoroughfares or in quieter side streets. A problem with this technique is that it fails to account for the

spatial distribution of pollutants varies widely depending on the surrounding terrain, traffic density and atmospheric conditions.

Penn *et al.* (1996) monitored carbon monoxide (CO) in the area around Tottenham Court Road, Gower Street and University College London, UK using a specially developed instrument, the 'Street Box'. Based on a small CO sensor made by City Technology Ltd., this instrument had a detection limit of 0.1 part per million (ppm) with an accuracy of $\pm 5\%$. The research showed that on roads which were physically close together, but which had differing amounts of traffic, the spatial variation of CO concentration can be up to one order of magnitude. The levels of carbon monoxide were shown to be highest on heavily trafficked roads, dispersing rapidly as the distance from the road increased.

Rashidi and Massoudi (1980) measured the concentration of carbon monoxide at thirteen points between the centre of a busy road and the walls of a building abutting the pavement in Tehran. The measurements showed that across the carriageway the concentration was greatest in the centre and fell lineally to 0.8 of the centre concentration at the edge of the pavement. Across the width of the pavement the concentration fell more steeply to 0.6 of the centre concentration, despite the fact that it was only a quarter of the total road width. The primary aim of this study was to explore the effects of traffic congestion on air pollution by using monitors located on a number of main thoroughfares in Tehran. The streets used during the pollution survey, had all been the subject of an earlier traffic survey to select streets with similar traffic flow characteristics. The results showed a distinct correlation between traffic density and the level of carbon monoxide. At vehicle speeds of between 0 and 15km/h, corresponding to a high level of traffic congestion, the mean concentration of CO was around 93ppm. As the traffic flow velocity increased the number of vehicles moving through the streets also increased, causing CO concentrations to rise to 140ppm. This increase was found to be solely due to the increased number of vehicles traveling along the roads and occurred at a vehicle flow speed of 30km/h. As the flow velocity increased still further the concentration of CO fell

to a mean concentration of around 9ppm at 55km/h, the increase in velocity and the fall in concentration corresponding to lower numbers of vehicles traveling along the road.

Menachem *et al.* (1990) found a similar correlation between traffic density and pollution in Jerusalem. This survey was conducted to measure the concentration of CO and NO_x in the centre of the carriageway and was accomplished using a van equipped with a portable gas monitoring system. The survey showed that the concentration of both pollutants was significantly higher than corresponding concentrations recorded with kerbside monitoring equipment. Concentrations were also higher where the traffic was slow moving or waiting at road junctions. The measurements also showed that along most sections of the test route the concentrations exceeded the short term lower national Israeli air standard for CO and NO_x of 30 and 0.5ppm respectively. In addition the NO_x concentration data showed a significant number of violations of the higher national standard for NO_x of 1ppm. The peak recorded concentrations were 52ppm for CO and 2.2ppm for NO_x. These readings can be compared to the average of the recorded data which were 13ppm for CO and 0.63ppm for NO_x.

2.3 Passenger Compartment Ventilation

The method of ventilating a vehicle passenger compartment has a significant effect on the level of pollution experienced by the occupants. Methods to model the concentration of pollutants in the passenger compartment of a vehicle have been investigated by Ott and Willetts (1981). A method of measuring the dynamic response (τ) of a vehicle was explored. The passenger compartment was found to exhibit a first order response when subjected to a step change in the concentration of pollutant (Figure 2.1). This response was measured by purging the vehicle with clean air by driving on side roads and then driving onto heavily used roads where the measured level of pollution was several times higher. The ventilation system was found to behave in a similar manner to a low pass filter, removing short duration peaks of pollutant.

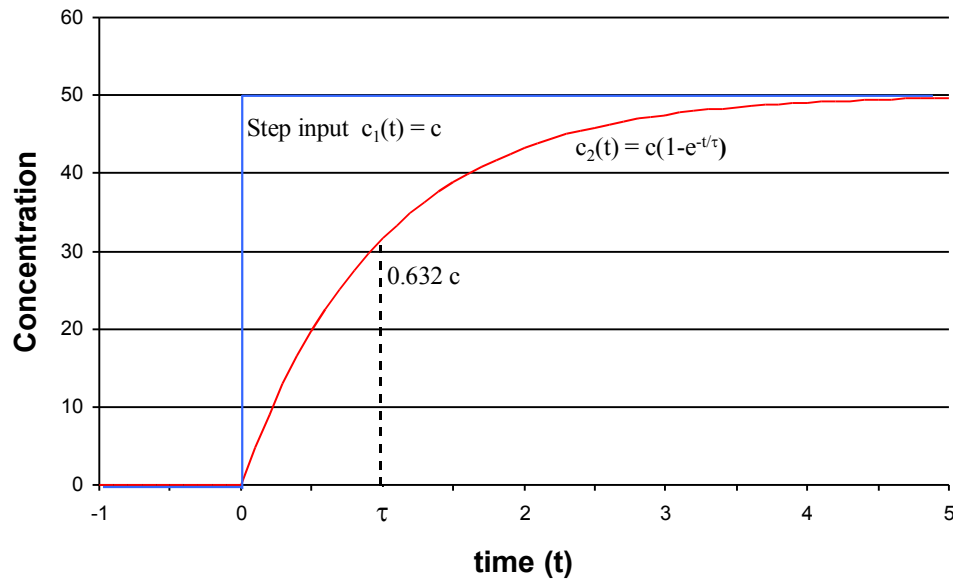


Figure 2.1 Response of a vehicle passenger compartment to a positive going step input of pollutant gas (After Ott and Willetts (1981)).

Heinsohn *et al* (1992) modelled the flow of cigarette smoke through a car using a sequential box model. A simplified car interior was modelled by dividing the volume into twenty four boxes. The flows into and out of each box were then computed, the volumetric flows either entering or leaving adjacent boxes. For all the ventilation cases studied, the model predicted that the smoke concentration would reach its maximum after about five minutes. The response of the compartment to CO entering the air inlets was found to be faster than that of the cigarette smoke. This was largely due to the better degree of mixing between the CO and the air. Carbon monoxide entering the model through the air inlet registers caused the concentration to rise to a maximum in around two minutes. Subjecting the compartment to a step change in pollutant level by extinguishing the cigarette exhibited a response similar to that measured by Ott and Willetts (1981) in a vehicle.

The air exchange rate of a stationary vehicle has been examined by Park *et al* (1996). The air exchange rates of the passenger compartments of three vehicles were measured and airflow over the vehicles was monitored. The air exchange rates of the three vehicles ranged between 1.0 and 3.0/h with the windows closed and no mechanical ventilation, and this rose to 1.8 and 3.7/h with the

windows closed and the fan turned on to recirculate the air. With the window open the air exchange rate rose to between 13.3 and 26.1/h with no mechanical ventilation and was between 36.2 and 47.5/h with the fan switched on to deliver fresh air. The measurements were conducted by facing the cars into a wind of between 0.9 and 4.5m/s. This research was prompted in part by concerns over the exposure of accident victims to combustion products released following the inflation of air bags.

2.4 The Exposure of Commuters to Traffic Pollution

Chan *et al* (1991a) examined the exposure of commuters to gasoline related volatile organic compounds (VOCs) as they journeyed to work using four different commuting modes (driving, subway, walking and cycling) in Boston, Massachusetts. Twenty five volunteers took part in the study, none of whom smoked. Nine of the volunteers drove to work, seven used the electric subway, six walked and three cycled. The substances monitored were, benzene, toluene, ethylbenzene, *m/p*-Xylene, *o*-Xylene and formaldehyde. Car commuters were found to be receiving the highest VOC exposure with passengers on the subway receiving the next highest exposure (Table 2.1). The exposure of the latter group was thought to be due to the release of toluene resulting from spillages of oil or solvents in the subway system. Commuters who walked or cycled were found to have a far lower exposure to all of the monitored VOCs than car drivers.

Chemical	Maximum Concentration $\mu\text{g}/\text{m}^3$ [Time]*	Mean Concentration $\mu\text{g}/\text{m}^3$	Journey Type and Mean Duration
Benzene	64.0 [22]	17.0	Car / 76min (40 samples)
Toluene	105.1 [22]	33.1	
Ethylbenzene	21.6 [87]	5.8	
<i>m/p</i> -Xylene	74.6 [87]	20.9	
<i>o</i> -Xylene	26.1 [87]	7.3	
Formaldehyde	19.7 [64]	5.1	
Benzene	13.2 [85]	6.9	Subway / 87min (38 samples)
Toluene	151.7 [190]	30.8	
Ethylbenzene	5.8 [95]	2.5	
<i>m/p</i> -Xylene	21.6 [80]	9.8	
<i>o</i> -Xylene	7.8 [57]	3.6	
Formaldehyde	14.1 [66]	4.5	
Benzene	24.2 [37]	10.6	Walking / 47min (31 samples)
Toluene	44.3 [37]	19.8	
Ethylbenzene	6.8 [35]	3.0	
<i>m/p</i> -Xylene	32.9 [37]	12.6	
<i>o</i> -Xylene	8.9 [35]	4.1	
Formaldehyde	15.1 [20]	5.5	
Benzene	28.0 [30]	9.2	Cycling / 54min (11 samples)
Toluene	45.1 [30]	16.3	
Ethylbenzene	7.1 [30]	2.4	
<i>m/p</i> -Xylene	28.3 [30]	10.0	
<i>o</i> -Xylene	8.9 [30]	3.0	
Formaldehyde	18.0 [40]	6.3	

- time associated with maximum exposure (minutes)

Table 2.1 Concentrations of VOCs for various commuting modes measured in Boston, Massachusetts (After Chan *et al* 1991a).

Flachsbart *et al* (1987) monitored fifteen commuter routes into Washington D.C. Eight of these were by car, four by bus and three by rail. The study period was from late January to early March 1983. Two hundred and thirteen trips were made by car, thirty five by bus and eight by rail: all the trips took place during the morning and evening rush hours. Five separate cars were used to record the car data, all of which were of the same model and age. This was a requirement of the survey procedure, as earlier studies had shown that differences in the ventilation systems of different makes of car can effect the pollution concentrations experienced in the passenger cabin. Commuting by car exposed the occupants to mean concentrations of CO between 9 and 14ppm, during journeys lasting between 40 and 60 minutes. The bus routes had mean concentrations of 4 to 8ppm and journey times of between 90 and 110 minutes and the train mean concentrations were between 2 and 5ppm with journey times of 30 to 45minutes duration. Data recorded during the car journeys showed a correlation with the mean road speed, which was being used to assess traffic density. At low mean journey speeds of 10mph the mean CO concentration inside the passenger cabin was around 13ppm, whilst at 60mph it had fallen lineally to 8.5ppm.

Ott *et al.* (1994) measured the concentration of CO during 88 standardised drives along the El Camino Real arterial highway around Palo Alto and Los Altos, Southern California over a 13 month period. Over the trips the mean concentration of CO was 9.8ppm with a standard deviation of ± 5.8 ppm. When the traffic density of the individual trips was used to adjust the recorded data a strong seasonal trend was found. CO exposure was found to rise during the winter months and to decay to a minimum in mid summer (the January/February mean exposure was around 17ppm and the July/August mean exposure was 7ppm). Correlation was also found between the concentration data and the average number of stationary vehicles at road junctions on the route, with the CO concentration rising from a mean of 6ppm with one stationary vehicle to 33ppm with 34 stationary vehicles.

Colwill and Hickman (1980) recorded the concentrations of CO inside eleven new vehicles on a 35km route around London. Mean concentration levels of CO varied between 12 and 60ppm and were between 30% and 80% of the recorded external levels. The buffering effect of the ventilation system was seen to damp short term peaks of pollutant. Differences in the concentrations measured over several journeys in the same vehicle were slight, but as with Flachsbart *et al* (1987) differences between vehicles were more marked and were attributed to differences in the design of the ventilation systems.

A comparison of the exposure of vehicle occupants and pedestrians to traffic pollution was made by Chan *et al* (1991b). In this study concentrations of CO, O₃, NO₂ and 26 gasoline-related VOCs were monitored. Five of the VOCs (1,3-Butadiene, benzene, isopentane, toluene and *m/p*-Xylene) were used for comparisons between in-car exposure, pedestrian exposure and the concentration recorded at a fixed-site monitor. The concentrations of all the selected VOCs, when measured inside or on the exterior of the test vehicle, were almost twice that measured on the pavement and almost four times the concentration measured at the fixed site. Time of day seemed to have little effect on the pollutant concentrations, except for ozone (O₃) which was at its highest concentration at noon and nitrogen dioxide (NO₂) which was marginally higher during the afternoon. This matches the diurnal pattern found in many U.S. cities. As with other surveys the concentration of pollutants was found to be greatest during urban driving.

2.5 Discussion

In all of the cited work the authors showed that the highest levels of pollution were experienced inside a car. Dispersion measurements have shown that the highest levels of pollution are experienced at the centre of the carriageway and decay rapidly across pavements. Pedestrians consistently experience lower concentrations of most pollutants and commuters travelling by bus also experience lower concentrations than car drivers. This may be attributed to the large volume of the passenger compartment and the different ventilation mechanisms employed. Most research also shows that high levels of congestion

cause the highest levels of pollution and that if traffic speeds were to be increased the concentrations experienced by drivers may fall due to the mixing effect of air flowing over the car, and greater distances between vehicles. The study of pollution dispersion conducted in Tehran, showed that as the flow of traffic along a road reaches its maximum, pollution can increase due to the number of vehicles using the roadway. Rashidi and Massoudi's (1980) findings also tend to show that atmospheric and geographical factors also influence the level of pollution experienced and to base estimates of pollution solely on traffic flow velocity, may be flawed when considering an urban environment.

The design of the ventilation system seems to effect the level of pollution experienced, with different models of car behaving differently in similar atmospheric and traffic conditions. All of the surveys which compared ventilation through the ventilation system with that through a window found that lower levels of pollution were measured with windows open. This may be due to the higher air exchange rate experienced when a window is opened and to the proximity of air inlet vents to pollution sources. Little seems to have been published evaluating the dispersion of traffic derived particulates. These have only recently been recognised as a problem and none of the cited work looks at particulate matter, concentrating solely on VOCs and other gaseous pollutants. Medical data is available which shows that particulate matter can cause damage to the respiratory system, and particulates are recorded as part of the data from the DoE automatic air monitoring network. Very little work seems to have been published about the ventilation of the passenger compartment of vehicles. In fact Park (1993) goes so far as to say that there was no published work on the ventilation rate of stationary vehicles, this may simply be due to manufacturers not releasing data for publication.

Chapter 3. Common Airborne Pollutants

3.1 Introduction

Airborne pollution is not a new phenomenon. Since the industrial revolution the air quality in towns and cities has been reduced due to industrial and domestic use of fossil fuel, and latterly the use of the internal combustion engine as a means of mass transport (QUARG 1993*a*). The early causes of air pollution in the United Kingdom have been all but eliminated. Coal, the main source of energy used in the United Kingdom since the latter part of the 18th century has been largely superseded in heating applications, the only large market remaining being electricity generation, where it is burnt under tightly controlled conditions.

Reductions in the use of coal have improved the air quality in all the major cities within the United Kingdom. The dense smogs, caused by coal smoke (DH 1992), are now thankfully a thing of the past, but as one major cause of pollution has been eliminated another has arisen. Increases in road traffic caused partly by the decline of the railway network during the 1950's and 60's, and increased car ownership, are now seen as the main cause of poor air quality within most major British cities.

The gases emitted by road vehicles include carbon monoxide (CO) and oxides of nitrogen (NO_x). Ozone (O₃) is also generated but unlike the other pollutants is not directly emitted from vehicles, rather it is a product of photo-chemical reactions in the atmosphere. Sulphur dioxide (SO₂), another pollutant gas, is also emitted from motor vehicles, though they are not the primary pollution source, which is the industrial use of coal and other sulphurous fossil fuels. These gases are the main constituents of gaseous air pollution within the UK. Many other substances are emitted both by industry and motor vehicles, but they are often present at no more than trace levels within the atmosphere. Particulates (PM₁₀) are another form of air pollution which has come to be seen as hazardous over the last few years. Although emissions from road vehicles are possibly the most noticeable form of this pollution they constitute only a

small percentage of the overall emissions, industry being the main source of particulate matter.

This chapter attempts to give an overview of the aforementioned pollutants and reviews medical research about their effects upon public health. In the main the pollutants are irritants and tend to reduce respiratory function, measured as the amount of air expired after a maximum inspiration (forced expired volume, FEV or forced vital capacity, FVC) or airway resistance (R_{aw}). The only pollutant known to be directly toxic is carbon monoxide which is generally fatal at a concentration of around 1000ppm. Of the pollutants reviewed, ozone is the one which generally causes most concern due to uncertainty over its effect on asthma sufferers. Ozone is unusual in that it has baseline concentrations only two to three times lower than concentrations found to cause adverse effects on living systems, whereas other pollutants have baseline concentrations several orders of magnitude lower.

3.2 UK. Monitoring of Airborne Pollution.

During the 1950s, the nature of pollutants such as SO_2 , CO and O_3 gave rise to concerns over their adverse effect on human health. These concerns prompted many national governments to set up programs to monitor and study ways of reducing the level of these pollutants. In the UK the government introduced the Clean Air Act of 1956 to curb the uncontrolled combustion of fossil fuels, primarily coal, in the inner cities. This legislation is now supplemented by the Environmental Protection Act of 1990. The establishment of a network of air quality monitoring stations to record the ambient levels of the main pollutants also began during the late 1950's: this network has been continually expanded and now has reasonably comprehensive coverage of the UK. Due to the ever increasing stringency of air quality legislation, air quality in the UK, particularly in the inner-city areas, has seen a continual improvement over the last three decades, despite the fact that road traffic pollution has increased.

At present the Automatic Air Monitoring Network is comprised of around ninety sites located in both rural and urban areas (Figure 3.1). Twenty of these

sites are situated in and around Greater London, the remaining sites are located such that they cover the remainder of the British Isles (DoE 1996b).



Figure 3.1 The Leicester Central DoE air pollution monitoring station.

The automatic sites, operated by the Department of the Environment, are supplemented with stations run by local authorities, either continuously, or as part of localised surveys conducted over periods of a few days or weeks.

The DoE sites measure a range of airborne pollutants, notably O_3 , NO_x , CO , SO_2 and PM_{10} . Not all the stations can measure this range of pollutants and only a few are equipped to measure other pollutant gases. Due to this variation in the sites it becomes difficult to plot nation-wide patterns of some pollutants due to lack of data, but the general trend is such that the highest levels of most pollutants are in the heavily populated South East with the levels decreasing toward the North and the West of the country.

In addition to the trends associated with population densities, the data from the automatic sites reveals seasonal and diurnal variations in pollutant levels. O_3 concentrations rise during the summer months, whereas CO levels fall during the summer and reach a peak in mid winter. Peaks of traffic-related pollutants occur during the morning and evening periods associated with commuter travel, and cumulative rises in ozone levels can be observed on a daily basis

during the summer. Observations such as these are possible because the data from the automatic stations are published as hourly mean concentrations. However short-lived but very high local concentrations (for instance, peak concentrations on pavements close to traffic), do not show up in the published data. To monitor these peak concentrations experienced by pedestrians and other road users, a continuous recording, or a short averaging time monitor needs to be used. This method of monitoring has its own set of limitations in that the spatial variability of most air pollutants is high (Penn *et al* 1996), especially around the inner city and so results gained from any such surveys are specific to that particular location and must be treated with caution.

3.3.1 Methods of Monitoring Gaseous Pollutants

Common methods of monitoring pollutant levels use either gas chromatography, or infra red absorption gas analysers. In both methods a sample of air is collected and fed into the analyser. In the case of the chromatograph the sample is passed over a hydrogen flame and the resulting spectrum is measured to determine and quantify the gases present in the air sample. In the case of the infra-red absorption analyser the sample is fed into a column with an infra-red source at one end and a detector at the other. Column length and the wavelength of the emitted radiation determine the concentration and type of gas that can be detected. Semi-conductor based detectors are now available to detect a wide range of pollutants. These detectors are robust, relatively inexpensive and reliable (Figure 3.2). Unfortunately detectors of this type are not sensitive enough to detect the very low concentrations of some pollutants present in ambient air. For the detection of sulphur dioxide, ultraviolet fluorescence can be used. This technique uses an optical sensor to detect the fluorescence given off by SO₂ molecules when they are subjected to



(a)



(b)



(c)

Figure 3.2 Typical types of gas monitoring equipment. (a) Diffusion tube. (b) Crowcon Triple Plus portable gas monitor. (c) Fixed site hazardous gas sensor.

ultraviolet light. Although this technique can be specific to SO₂, the sample must be passed through a scrubber to remove any hydrocarbons which can interfere with the detection process. With all of the techniques it is important that the system is zeroed and correctly calibrated. This is commonly done by purging the instrument with first, a zero gas (often nitrogen), to check the true zero on the instrument and secondly a span gas, to check and calibrate the actual reading given by the instrument.

Some pollutants have such low ambient concentrations that conventional gas monitors may be impractical, as revealed by a recent DoE survey of NO₂ (DoE 1994). Diffusion tubes can be employed in these cases. A diffusion tube is a short stainless steel tube packed with an absorbent medium. In use the tube is either left in the sampling location for a set time, or a known volume of air is pumped through the tube at a suitable flow rate (Cheong and Riffat 1994). When the tube has had the required exposure it is heated to de-sorb the pollutant gas which is then be passed into a gas chromatograph for analysis. This method has the advantage that the tubes are relatively inexpensive and robust, but is capable only of measuring a mean concentration over the sampling period. For some gases a measured sample may be bubbled through a suitable solvent, which when treated with an indicator can be analysed optically to quantify the pollutant present. This method also gives the mean concentration of the gas.

3.3.2 Monitoring of NO_x in the UK.

Forty seven of the DoE automatic monitoring stations (based on August 1996 figures) are equipped to measure NO_x; six of these are located in rural areas, the remainder being located in urban environments. During 1994 the data from the automatic stations was supplemented by the DoE UK Nitrogen Dioxide Survey (DoE 1994). The survey is conducted by local authorities and overseen by the DoE. Each authority sets up four diffusion tubes in a standard configuration.

This is summarised below :

Kerbside *1 to 5m from a busy road.*

Intermediate *20 to 30m from the same or an equivalent road.*

Background *>50m from a busy road (2 of).*

Each tube is exposed for a period of one month (Figure 3.3).

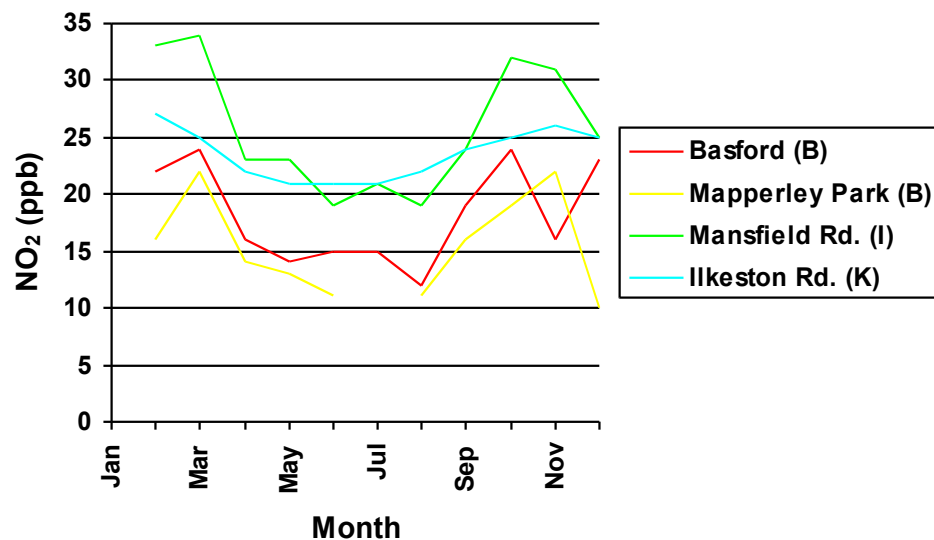


Figure 3.3 Monthly average NO₂ concentrations recorded during 1995 at the Nottingham City monitoring stations as part of the DoE diffusion tube survey.

3.4 Measurement of Particulate Concentrations

There are two methods for monitoring atmospheric particulate concentrations, namely gravimetric analysis and the smoke stain method. The two methods are not directly comparable and the distinction should be maintained when analysing data. In the case of the smoke stain method, a metered sample of air is pumped through a standard filter. The resulting stain is then measured using a reflectometer and the particulate concentration in $\mu\text{g} / \text{m}^3$ of standard smoke, is then gained from tables. Gravimetric measurement of particulate concentration is commonly performed using a micro-porous filter of known mass. A metered sample of air is then passed through the filter, to determine the increase in mass. This method measures the particulate concentration

directly in $\mu\text{g} / \text{m}^3$. Another technique which indirectly measures the gravimetric concentration of particulates uses diffraction of a beam of laser light. This technique, which is used in 'Grimm' portable dust monitors, measures the scattering of laser light by particles passing through the beam. This technique has the advantage that it gives an instantaneous reading of particulate concentration, rather than the mean concentration given by the two filter derived methods.

3.5 Overview of Common Atmospheric Pollutants

3.5.1 Carbon Monoxide

Carbon Monoxide (CO) is a colourless, odourless gas produced as a product of incomplete combustion. The World Health Authority have set time-weighted average (TWA) exposure guidelines for CO of 26ppm for 1 hour or 9ppm for 8 hours. The toxicity of CO at moderate to high concentrations is well known and has been the cause of innumerable fatalities. It stems from the ability of CO to bind with haemoglobin in place of oxygen molecules this reduces the ability of the blood to transport O_2 . The characteristics of CO poisoning are nausea, headache, confusion and tiredness. Acute poisoning may lead to rapid distressed breathing, cyanosis (blueness of the skin) and impaired consciousness. Moderate levels of exposure to CO cause drowsiness and reduce concentration. As measured levels of CO within a vehicle can exceed the 26ppm WHO limit, exposure of vehicle occupants to CO may have detrimental effects.

What is less well understood is the effect of low concentrations of CO, especially when experienced over a long period. The best evidence comes from a study in the US. Which correlated increases in hospital admissions for elderly patients suffering congenital heart failure with pollution episodes (Indoor Air Quality Update 1995). The findings of this study relating to CO are summarised in Table 3.1., where the relative risk is the increased likelihood of hospitalisation resulting from the elevation in CO level.

The single pollutant model only looked at the correlation between CO and hospital admissions. The multi-pollutant model took into account levels of other common airborne pollutants (i.e., NO_x and SO₂). The relative risk in both models is statistically significant and broadly similar, and by comparing the two

City	Single Pollutant Model		Multi-pollutant Model	
	Relative Risk	95% Confidence Interval	Relative Risk	95% Confidence Interval
Los Angeles	1.36	1.28-1.46	1.39	1.23-1.56
Chicago	1.29	1.16-1.44	1.23	1.07-1.43
Philadelphia	1.17	1.05-1.31	1.22	1.05-1.41
New York	1.10	1.03-1.18	1.05	0.97-1.14
Detroit	1.24	1.11-1.39	1.36	1.17-1.63
Houston	1.11	0.97-1.28	1.25	1.05-1.49
Milwaukee	1.29	1.07-1.57	1.26	0.89-1.77

Table 3.1 Relative risk for hospital admission due to a 10ppm increase in the ambient CO concentration (After Indoor Air Quality Update. 1995).

it seems that interaction between CO and the other pollutants is negligible. It must be borne in mind that the normal mean levels of CO in all of the cities surveyed are less than 6ppm, the highest being New York at 5.6ppm and the lowest being Milwaukee at 1.8ppm. Whilst exposure to CO can be shown to be harmful it does not appear to have any long term effect, and in the hospitalisation study cited, admissions fell sharply as the pollution events decayed.

3.5.2 Carbon Monoxide concentrations in the UK.

In Greater London 95% of the measured atmospheric CO can be attributed to motor vehicles (WHO 1992), the remainder coming from heating and power plants around the city. Peak CO pollution episodes occur during the winter months, when ground-based temperature inversions trap vehicle emissions close to the ground. Reductions in mean CO concentrations over recent years can be attributed to the use of catalytic converters on all petrol cars manufactured after 1993. However during winter months, when the likelihood

of problematic CO pollution is greatest, these reductions are probably offset by large numbers of vehicles being used for short journeys.

Thirty five of the DoE automatic monitoring stations (based on August 1996 figures) are equipped to measure CO and all are located in urban areas, often in shopping precincts. Due to the localised nature of traffic pollution these sensors can only monitor the background concentrations of CO. A recent survey carried out in London has shown that considerable spatial variation of CO occurs with only short distances between monitoring locations. The researchers concluded that pollutant levels were affected by local climatic conditions as much as by the traffic conditions at each location (Croxford *et al.* 1995). Consequently streets within short distances of each other could have very different levels of pollution. In a later survey by the same research team (Penn *et al.* 1996), considerable spatial variation was found using monitors located in a compact area around the Euston Road / Tottenham Court Road (London, UK) junction. During this survey, peak readings of 12ppm on Euston Road and 4 to 6ppm on Gower Street were measured, while only background concentrations were recorded in adjacent side streets.

In a typical inner city location the background concentration of CO is normally around 1ppm. In heavily trafficked streets, kerbside measurements have shown that peak concentrations may be many times higher than this, with concentrations of 10 to 20 ppm common. Occasionally the concentration may rise in excess of 100ppm. In rural areas background concentrations are lower and depend on the proximity of local industry, power generating stations and arterial roads.

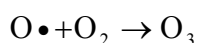
3.5.3 Oxides of Nitrogen

Nitrogen oxide (NO) and nitrogen dioxide (NO₂), collectively known as NO_x, are produced as a by-product of combustion systems. The dominant oxide produced is NO which makes up 90 to 95% of the total NO_x emission from most systems, the remainder is composed of NO₂. During the combustion process atmospheric nitrogen N₂ is oxidised into NO_x and so even fuels which contain no nitrogen will upon combustion emit NO_x.

In the atmosphere NO is progressively oxidised to NO₂ by reaction with ozone



As a consequence of this the amount of NO_x that is oxidised to NO₂ is limited by the availability of ozone. Close to the combustion source the majority of NO_x will comprise of NO and as the plume disperses and is mixed with unreacted atmospheric ozone the concentration of NO₂ will rise. In rural areas 80 to 90% of NO_x may be composed of NO₂. This process is complicated by photochemical decomposition of NO₂ by short wavelength light (<400nm).



These reactions define the so-called photostationary state, which in daylight is achieved in around a minute.

In urban areas where NO_x concentrations are highest, NO levels are generally higher than those of NO₂. This is due to the limited oxidising capacity of the urban atmosphere and generally 40% of urban NO_x is present as NO₂, but as with studies of CO large spatial variations can be found.

3.5.4 Sources of NO_x in the UK.

In the UK, the primary sources of NO_x are road transport (49%) and power generation (26%) the remainder being made up primarily by industry and other transport systems. Emissions by domestic fuel users account for around 3% of the total. In general the level of NO_x emissions has declined over the last decade; the only notable exception being emissions by road transport which increased from 810,000 tonnes in 1981 to 1,400,000 tonnes in 1991. Of the emissions by road vehicles, 30% was from petrol-driven vehicles and 21% was derived from diesel. The growth in the number of private cars over the 1980's can be shown to be the major cause of this increase; from 1981 to 1991 emissions from petrol driven vehicles rose from 394,000 tonnes to 816,000 tonnes, an increase of 107%. Emissions from diesel vehicles, which make up the large proportion of the commercial fleet, rose from 416,000 tonnes to 584,000 tonnes, a comparable increase of 40%.

Total emissions of NO_x from all vehicles are expected to have reached their maximum value around 1991 to 1992 as the fitting of catalytic converters to all new vehicles began to take effect shortly afterwards. The total NO_x emissions are predicted to fall to around 750,000 tonnes by the year 2000 (QUARG 1993a). Of these it is estimated that 450,000 tonnes will be emitted from commercial diesel vehicles, with the remaining 300,000 tonnes emitted from passenger cars.

3.5.5 Concentrations of NO_x in the UK.

Road transport produces 49% of the total emissions of NO_x, 24% is generated by power stations and the remaining 27% is produced largely by industrial processes. This is shown in the concentration of NO_x measured around large conurbation's. The annual mean concentration of NO₂ in UK cities is around 25ppb, this can be compared with rural areas where the annual mean concentration is 9ppb and remote areas where it is as low as 1ppb (based on data for 1994).

3.5.6 Epidemiological Effects of NO₂

Exposure studies of the effect of NO₂ on healthy individuals have shown no measurable effect upon lung function at concentrations below 1000ppb; although some slight increases in sensitivity to other compounds such as histamine have been noted after exposure to NO₂ (DH 1993)

In people with chronic obstructive pulmonary disease, studies have shown some reduction in FEV when compared with age matched control subjects. Further subsequent analysis of the data revealed that in chronic patients with more severe airflow obstruction, a reduction in sensitivity to NO₂ was apparent. Smoking was found to increase the responsiveness to NO₂ in the control subjects.

In asthmatic subjects controlled tests showed no change in lung function at concentrations of less than 3000 to 4000ppb. In tests at lower concentrations, increased sensitivity to some compounds was again found, particularly in subjects with more severe airway obstruction. NO₂ has not been found to cause any change in responsiveness to O₃. Adams *et al.* (1987) found no change in indices of lung function or in response to 300ppb of ozone following exposure to 600ppb of NO₂ for one hour, with heavy exercise. At doses below 300ppb some individual reductions in lung function have been observed in asthmatic subjects. It has been argued that low doses of NO₂ within a small range cause bronchial responsiveness, whereas higher doses do not.

Measured changes in lung function after exposure to NO₂ at concentrations of <1000ppb are both inconsistent and trivial in both normal and asthmatic subjects. There is some evidence of a small increase in airway resistance at higher concentrations.

3.5.7 Epidemiological Effects of NO

The daily intake of NO from outdoor sources is about 480 µg. Cigarette smokers, smoking twenty cigarettes per day would inhale four to five times this amount (1600 µg) at a concentration of around 1600ppb. Although NO is readily oxidized to NO₂ this does not occur when the gas is inhaled directly, or to any appreciable extent when administered acutely into the air. At low concentrations (below 5 ppm) there are no apparent changes in lung function.

3.5.8 Ozone

Most of the ozone in the Earth's atmosphere exists in the stratosphere between altitudes of 10 - 30km, where the 'ozone layer' has a beneficial effect. Ozone is unique as an atmospheric pollutant in two important respects. Firstly the baseline concentrations of ozone are only two to three times lower than concentrations which have been observed to cause adverse effects in some living systems. This can be compared with the baseline concentrations of other common pollutants which are often several orders of magnitude lower than the lowest concentrations seen to cause adverse effects (DH 1991*a*). Secondly there are no significant emission sources of ozone; it is produced by a series of complex photochemical reactions, occurring over time spans of minutes to months. A consequence of this is that the emission of precursor compounds, e.g. oxides of nitrogen (NO_x) and volatile organic compounds (VOCs), can influence production of ozone many kilometres away.

An assessment of UK. ozone concentrations was made in 1987 by the Department of the Environment's Photochemical Oxidants Review Group (PORG) which reviewed ozone concentrations up to and including 1985. In 1987 a network of monitoring stations was established by the Department of the Environment, to provide national coverage of ozone data for rural and urban areas. Much of our knowledge of O₃ is based on published data from these monitoring stations, forty six of which are equipped to measure ozone .

Ozone production during a pollution episode is superimposed onto a baseline concentration, which in northern latitudes around 50 ° N is approximately 25 -

35ppb. Although some of the ozone background is natural, the majority is man-made. The natural ozone arises from two sources, namely the movement of ozone rich stratospheric air into the troposphere, and production of ozone within the troposphere. The elevated episodes of ozone concentrations which persist for a few days at a time are thus peaks superimposed on a background concentration.

The 'ozone layer' is the section of the earth's atmosphere having the highest mixing levels of ozone, with concentrations reaching 8ppm (8000ppb). Mixing downward into the troposphere is estimated to account for 20% of tropospheric ozone. Chemical processes are estimated to account for the remaining 80%. These processes primarily involve the methane / carbon monoxide oxidation system and there is a consensus that ozone production is limited by the availability of NO_x . In a polluted atmosphere the chemical reactions take place over hours to days rather than the months appropriate to a free tropospheric chemistry.

Ozone production is largely governed by the following reactions :-



These reactions produce no net ozone but hold the system in equilibrium (the photo-stationary state), which in daylight is reached in around a minute. To increase ozone concentrations, the photolysis of NO_2 must be increased (i.e., an increase in the rate of the first equation). This is achieved by the reaction of hydrocarbons and other organic compounds (VOCs) which after an initial reaction with an OH radical, produce peroxy radicals (RO_2), NO can then be oxidised by these to NO_2 .



More O_3 can then be produced by the photolysis of NO_2 . The remaining RO radical can continue to react producing more radicals and oxidising more NO. In the free tropospheric state the RO radicals are produced from methane and carbon monoxide over a time scale in the order of months. In the polluted boundary layer a large number of VOCs are present due to the actions of our industrial society and they react to produce RO_2 in hours or days.

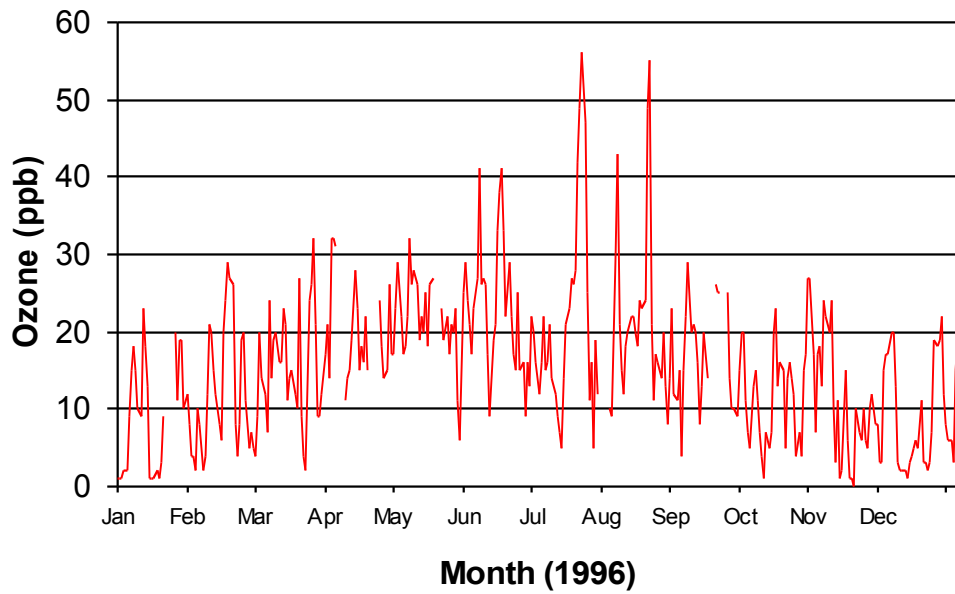
Elevation of ozone concentrations generally occurs during sunny anticyclonic weather. This may be explained in two ways; firstly one of the important reactions is photolysis of ozone itself to produce OH radicals. They in turn produce peroxy radicals which oxidise NO to NO_2 to produce ozone.



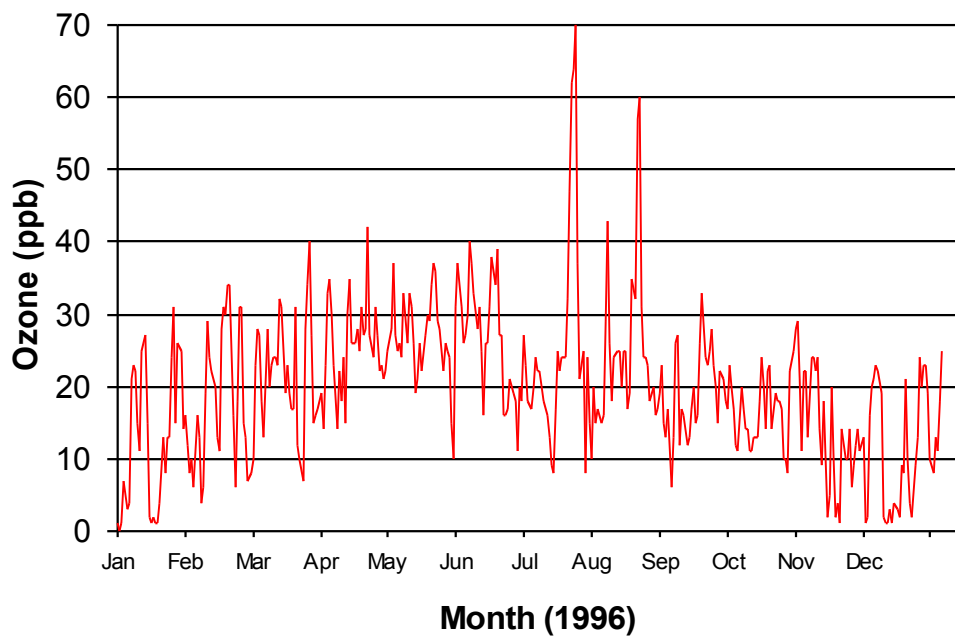
Secondly low wind speeds elevate the concentrations of the precursor VOCs and NO_x , which increases the rate of the ozone producing reactions. The main sources of the precursor VOCs are industrial solvents and processes (51%) and motor vehicles (37%). Motor vehicles also contribute 49% of current NO_x emissions in the UK. This is broadly similar to the rest of north-western Europe.

3.5.9 Ozone concentrations in the UK.

Ozone has been monitored using a network of fixed monitoring stations since 1987. Data gathered from the Bottesford (rural) and Leicester Centre (urban) monitoring stations are shown in Figure 3.4. It can be seen that the baseline concentration of ozone is generally lower in the urban environment than in a rural area. This is due to the destruction of ozone by nitrogen oxide (NO) which is emitted as a combustion product by motor vehicles. However this situation is complicated by measurements from other stations which suggest that in urban



(a)



(b)

Figure 3.4 Daily average concentrations of ozone recorded at the (a) Leicester Centre and (b) Bottesford automatic monitoring stations (DoE 1997).
 areas away from heavily trafficked roads, ozone concentrations are close to those found in a rural environment. This leads to the conclusion that there is

considerable spatial variability of ozone within urban areas which makes measurement of exposure difficult.

The concentration of ozone tends to follow a daily pattern, with peak concentrations occurring from noon onwards. This is readily explained by reference to the chemical reactions which produce ozone. If sinks for ozone are depleted for any reason, high concentrations may develop and can remain throughout the night when ozone production is negligible. This can cause a cumulative effect over several days. In addition to the diurnal cycle a yearly cycle exists and can be seen on the Figure 3.4. These show that ozone concentrations tend to peak during June, July and August and are at a minimum during December, January and February. With the small number of monitoring stations available, the drawing of robust contours or the undertaking of epidemiological studies is difficult. Patterns can however be deduced from the available data and these can be used to predict ozone concentrations within different regions of the UK. A broad North-South gradient is apparent, at the extremes of which, the number of hours when ozone concentrations are greater than 60ppb differ by a factor of between five and ten (DoE 1996e).

3.5.10 Epidemiological effects of ozone.

Ozone acts as an irritant upon the respiratory tract, giving short term reductions of lung function. The long term effects are more uncertain, as is the effect on people with acute respiratory disorders, such as asthma; this is due in part to the action of other pollutants and aeroallergens to which the subject may be sensitive. Trials conducted mainly in North America (DH 1991b) have monitored lung function and symptoms over days or weeks among panels of healthy subjects allowing individual variations due to changes in the ambient level of ozone to be established. Most of the studies have been carried out with children or adolescents: and the results are summarised in Table 3.2. A problem with the technique is that the ozone concentration may be measured several miles away from where the subjects are, in addition the exposure to ozone not only depends on the ambient outdoor concentration, but is also dependent on time spent indoors and the activities undertaken during the day.

<i>Location</i>	<i>Number and age</i>	<i>Physical activity level</i>	<i>Max 1- hr ozone (ppb)</i>	<i>FVC ml/ppb</i>	<i>FEV ml/ppb</i>	<i>PEFR ml/s/ppb</i>	<i>Notes</i>
Indiana	58 8-13yrs	moderate	110	-1.06	-0.78		YMCA summer day camp
Mendham	39 7-13yrs	low	185	-0.12	-0.28	-2.99	Summer day camp, evidence of persistence of effects
Fairview Lake	91 7-13yrs	moderate	113	-1.03	-1.42	-6.78	YMCA residential summer camp
Kingston	154 7-13yrs	low	78	-0.92	-0.99		Grades 5 & 6 from one school. Ozone measured 6mi. away

Table 3.2 Summary of field studies undertaken on healthy children to examine the effect of ambient ozone on lung function (After DH 1991b)

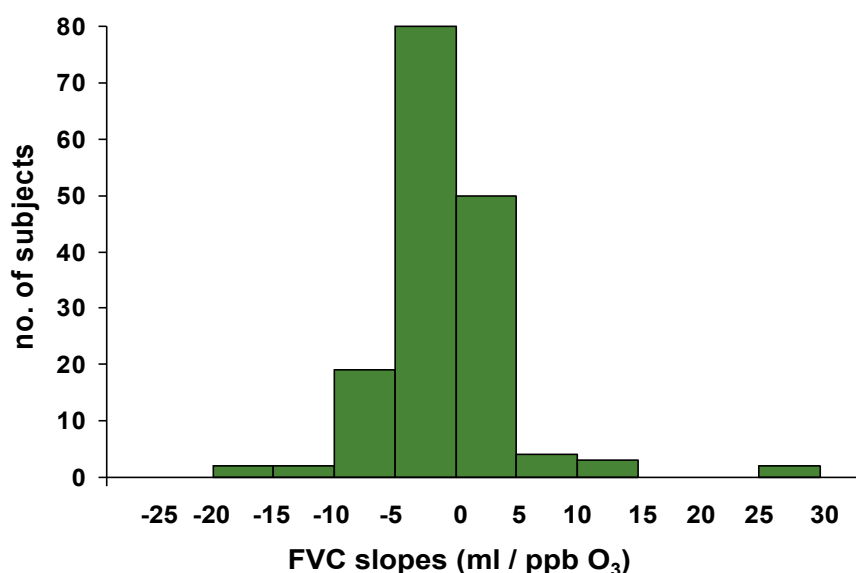


Figure 3.5 Distribution of 154 child-specific slopes (FVC by O₃) showing the relationship between ozone and lung function (After DH 1991b).

The most common statistical method for analysing these types of data is to use regression to establish the relationship between ozone and a measure of lung function (FEV, FVC, or flow rate) for each subject. This gives a regression coefficient which is usually expressed as ml/ppb ozone. A distribution graph can then be drawn (Figure 3.5). Problems arise due to other pollutants such as NO_x, masking the effect of ozone and it may be impossible to separate cause and effect. Despite this some definite conclusions can be drawn from the studies. All of the studies show a negative association between ozone and the

respiratory tests. This is all the more striking due to differences in the methodology and conditions for each test series. For the five studies of children in the table, the mean decrement in FEV was -0.77ml/ppb ozone. At 100ppb that amounts to -77ml, around 3% of a typical FEV for a child. The adult study found a decrement of FVC of -2.1ml/ppb ozone; a similar reduction considering the larger size of the adult lung. Given these results it seems that the change in lung function and exposure to ozone is causal. It is also consistent with laboratory controlled chamber studies on humans and animals.

In studies involving asthmatic subjects, no clear link has been found between ozone and hospital admissions or mortality. Studies have shown that mortality amongst asthma patients rises during August and there is a tendency for hospital admissions to rise during September and October, but this does not correlate with the peak in ozone concentrations which occurs in May and June. It has been postulated that ozone sensitises the respiratory tract and it is this sensitivity combined with falling temperatures, infections and other pollutants which cause these trends. No studies have been able to directly link ozone with the mortality of respiratory patients. Ozone acting as it does as an irritant probably, contributes to respiratory disorders but acts with other pollutants and particulates rather than on its own.

No safe threshold of ozone exposure has been defined. The World Health Organisation guidelines for exposure to ozone give the 1hour time weighted average (TWA) concentration as 75ppb, and the 8hour TWA as 55ppb. During pollution episodes these concentrations can be exceeded for several days. At concentrations below 100ppb most people will not notice any ill effects, sensitive people may notice some impairment of their ability to exercise. As the concentration rises there is a gradual increase in the number of people who will suffer some impairment of lung function, limiting their ability to perform physical tasks. At concentrations of 200ppb impairment of lung function will be measurable in most individuals: this is the maximum concentration expected to occur in the United Kingdom.

3.5.11.1 Volatile Organic Compounds

VOCs are compounds such as benzene and 1.3-butadiene and are present in the atmosphere due to evaporation or as products of combustion. Although the concentrations present in the atmosphere are very small, many of these compounds are known to have effects detrimental to health and some are classed as carcinogenic. There is some evidence to suggest that exposure to low concentrations of VOCs can cause fatigue and mental confusion (Lundberg 1996). The discussion will be confined to benzene and 1.3-butadiene as they are the only VOCs routinely monitored in the UK.

3.5.11.2 Benzene

Benzene is emitted from motor vehicles due to both evaporation and incomplete combustion of petrol of which it is a minor constituent. In the EU up to 5% by volume of petrol may comprise of benzene, but the level is normally around 2% by volume. The evaporative emissions occur due to fuel being lost from the fuel system and fuel vapour being expelled from the tank as the vehicle is refuelled. In the UK, during 1994 around 67% of the emitted benzene (approximately 26100 tonnes) was due to petrol driven vehicles (DoE 1996f). In addition a further 11% (4290 tonnes) of emitted benzene can be attributed to the refining of petrol.

Benzene is accepted as a human carcinogen and can cause leukaemia. It has not been possible to determine a safe exposure for benzene and therefore all policies concerning exposure are based on keeping the risk to health very small. The highest concentrations are found around large urban conurbation's where the mean concentration of benzene may reach 1.7ppb and in rural districts the mean concentration is approximately 0.5ppb, the risk to health at such concentrations is felt to be minimal (DoE 1996f).

3.5.11.3 1.3-Butadiene

1.3-Butadiene is like benzene carcinogenic. It is used in the production of synthetic rubber and studies have shown that workers in this industry have a slightly higher risk of cancer to the bone marrow. Road vehicles account for

77% of all 1,3-butadiene emissions (68% petrol vehicles and 9% diesel vehicles) due to combustion processes.

Like benzene there is no agreed safe level of exposure to 1,3-butadiene, but a maximum annual mean exposure of 1ppb would be a minimal risk to health (DoE 1996g). In the UK, the highest concentrations are in urban conurbation's where the mean concentration may rise to around 0.5ppb in rural areas the mean concentration is approximately 0.2ppb.

3.5.12 Sulphur Dioxide and Particulates

Unlike CO and NO_x, motor vehicles are not the primary source of either sulphur dioxide (SO₂) or particulates (PM₁₀). Emissions arise from a wide range of sources and for this reason only a brief overview is given.

Sulphur dioxide is a gas which dissolves in water to give an acidic solution which is readily oxidised to sulphuric acid. In the UK it is primarily derived from the combustion of sulphur contained in fossil fuels. Since the Clean Air Act of 1956 concentrations of SO₂ have decreased due to the use of cleaner industrial and domestic fuels such as gas (Figure 3.6). Emissions are now dominated by fossil fuelled power stations which contribute 64% of the national total and industrial plant which is responsible for 25% of emissions. Road transport accounts for less than 3% of the total emissions. SO₂ is an irritant and has been shown to cause asthma attacks and breathing difficulties in susceptible people. It also causes environmental damage due to its oxidation to sulphuric acid (DoE 1996c).

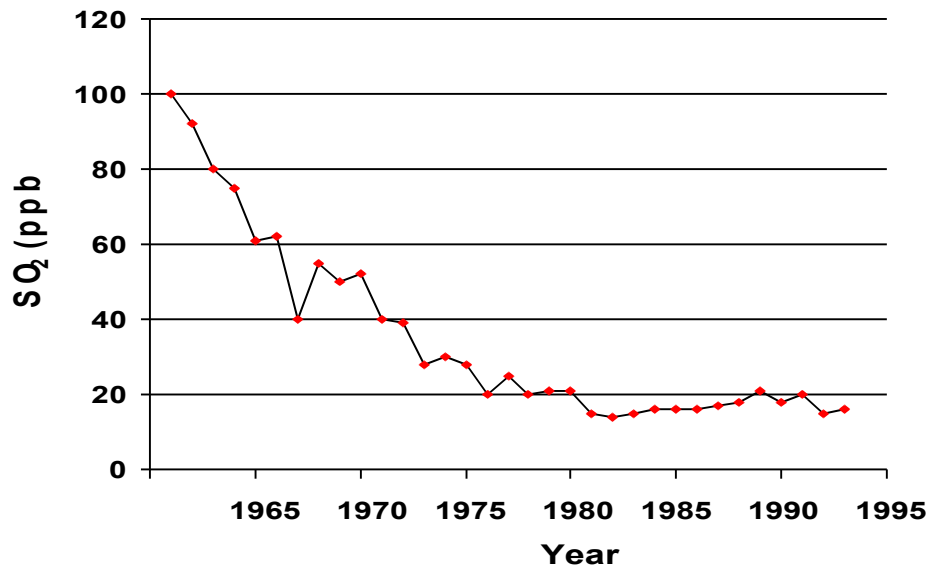


Figure 3.6 Annual mean concentrations of SO₂ measured at the Lambeth monitoring site (After DoE 1996c).

The highest concentrations of sulphur dioxide are found in heavily industrialised regions such as the Midlands and South Yorkshire. These areas are the location of a large number of coal burning power stations and industries such as smelting and refining. Large urban conurbations also generate high concentrations of SO₂ due to domestic burning of coal, the presence of industry and the density of road transport. The lowest levels of sulphur dioxide are found in rural areas such as the north of England, Scotland, Wales and the West Country.

Particulates are generated by many of the sources that produce SO₂. Power generation and industrial processes contribute 57% of measured emissions, domestic and commercial use of fuels accounts for 16% and road transport for 26%. The road transport emissions can be broken down and show that 73% of the emitted particulate matter comes from diesel engines, 19% from petrol engines and 8% is emitted from the tyres, brakes and clutches of vehicles (DoE 1996d). PM₁₀ is primarily emitted from vehicles under acceleration due to the engine being over-fuelled, which results in incomplete combustion. This is most noticeable in diesel exhausts where it manifests itself as a cloud of black

smoke as the vehicle accelerates away. This smoke is largely unburned carbon, it does however contain traces of sulphur, aldehydes and other potentially toxic or carcinogenic compounds. Many vehicle manufacturers are now developing filters to reduce the levels of particulate matter in exhaust gases (Auto Express 1994). Research is also being undertaken to improve combustion and the injection of the fuel. As a result of these measures vehicle derived PM₁₀ emissions are falling in the UK (Figure 3.7).

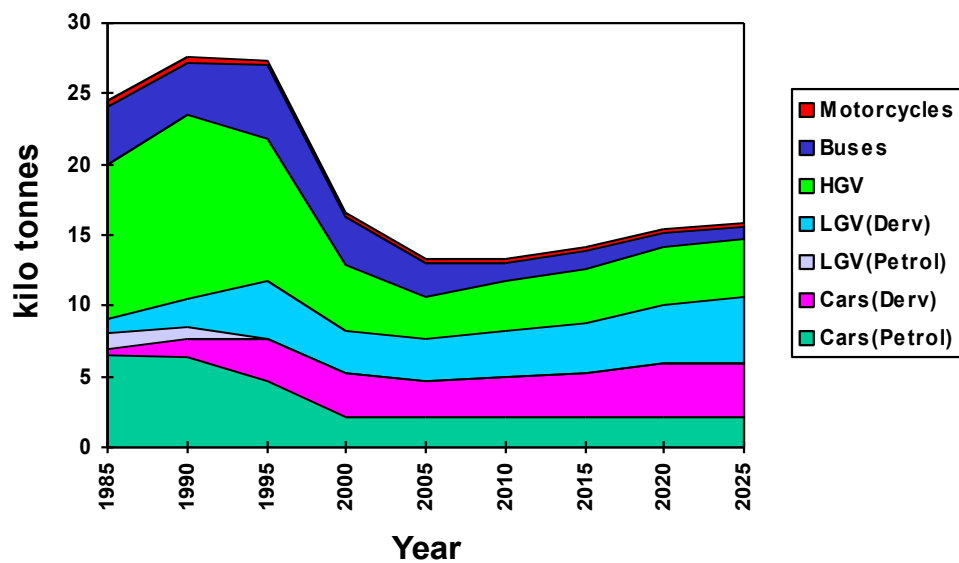


Figure 3.7 PM₁₀ emissions by road transport projection, based on diesel car sales being approximately 20% of all new car sales (After DoE 1996d)

Approximately 2.8% of deaths in urban areas may be due to PM₁₀ (Auto Express 1994). Research conducted in seventeen cities around the world has shown a correlation with mortality and PM₁₀, in London almost 2,000 deaths a year (3% of the total) are thought to be caused by particulates. The causes of death are respiratory or cardio-vascular disease and of these, cardiac diseases are twenty times more likely.

3.6 Conclusion

The effects of atmospheric pollution have clearly been shown to exacerbate respiratory disease. Legislation and improvements in combustion technology are reducing the level of pollution, but close to heavily trafficked roads the level of pollutants such as NO_x and O₃ are high enough to cause concern. Exposure to these pollutants has been shown to cause sensitivity to other pollutant gases such as sulphur dioxide. Particulate emissions from vehicles are also a cause for concern particularly in busy town centres. The effect on health of particulate matter is not greatly understood and cumulative effects cannot be ruled out.

Occupants of motor vehicles travelling through congested towns and cities are exposed to the highest concentrations of motor vehicle exhaust pollution. Whilst studies have shown that the reduction in lung function due to a single pollutant has no apparent long term effect, the consequences of being exposed to a high concentration of the cocktail of pollutant gases emitted by a motor vehicle are not fully understood. Psychiatric disorders have also been linked to air pollution (Lundberg 1996). Experimental evidence has shown that exposure to VOCs can cause fatigue, mental confusion and sensory irritation. In addition stress caused by exposure to air pollution can cause mood changes, loss of concentration and anxiety.

The amount of pollution entering a car travelling through a typical European city has not been evaluated as most of the research conducted has been centred on the west coast of America. In Europe the composition of the fuel differs to that used in the United States and the atmospheric conditions of northern Europe are not directly comparable with those of America. Taking these factors into account it is reasonable to assume that there may be differences in the concentrations of pollutants present in the atmosphere and in the dispersion characteristics of the streets.

Chapter 4. Airflow Measurement Techniques

4.1 Introduction

This chapter presents an overview of techniques and the instruments used to measure airflow through and around objects and the ventilation rate of spaces. The discussion has been limited to the methods and instruments used during the wind tunnel and field experiments which performed as part of this study are described in chapters six and seven. As this discussion is intended to give an overview of the techniques used rather than descriptions of specific instruments, more detailed information relating to the commercial instruments used during the experimental work is included in the relevant chapters.

4.2 Pressure Measurement Techniques

The measurement of the air pressure across surfaces or in ductwork is a useful tool when considering airflow and there are a number of ways that such measurements may be accomplished. Pressure is measured as either static, or dynamic (velocity) pressure. Static pressure is the pressure present due to fluid flow across a surface and for environmental studies is commonly measured with respect to the ambient atmospheric pressure. Dynamic pressure is caused by the forward motion of a fluid and is measured by placing a sensor into the fluid and measuring the pressure developed across a surface facing into the flow (Figure 4.1). The difference between the two readings gives the total pressure and this can be used to calculate the flow rate of the fluid. For air at room temperature and atmospheric pressure the equation is:-

$$U = \sqrt{1.74 P_t} \quad (4.1)$$

Where P_t is the total pressure in Pascals and U is the flow rate in m/s.

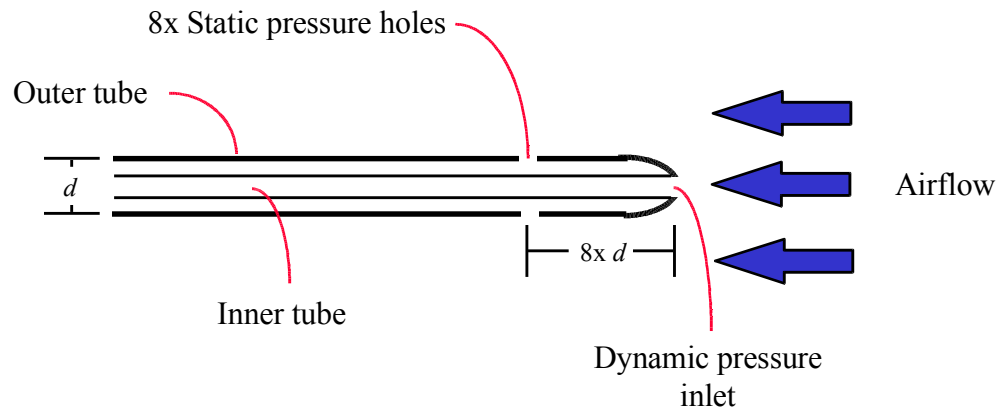


Figure 4.1 The tip of a ‘standard’ pitot-static tube. Dynamic pressure compresses the fluid inside the central tube, whilst static pressure reduces the pressure in the outer tube.

The most suitable instrument for use in conjunction with a pressure probe in wind tunnel work is the manometer, this is a differential instrument and can be used for both static and dynamic pressure measurements. For calibration and experimental work where high accuracy is required this will often take the form of an inclined tube of paraffin or mercury. Positive pressure is measured by connecting the probe to the lower end of the tube and for negative pressure measurements the probe is connected to the upper end.

Electronic manometers have been available for many years and are comprised of a differential pressure transducer and a suitable display. In this type of instrument the pressure is applied across a diaphragm and the displacement is measured using strain gauges. The electronic instrument has the advantage over the inclined tube manometer in that it is easier to set up and use, however the accuracy that may be obtained with an inclined tube manometer is high and consistent when the instrument is correctly installed. Single ended piezo-electric pressure transducers are also widely available, these may be calibrated to read absolute or gauge pressure and can be used to make either static or dynamic pressure measurements. These transducers work in a similar manner to differential pressure transducers in that the pressure is applied across a diaphragm, however unlike the differential transducer where pressure may be

applied to both sides of the diaphragm, the rear of the diaphragm is sealed and the test pressure is applied to one side only.

4.3.1 Anemometry

For air velocity measurements direct readings may be taken using either vane or hot wire anemometers (Figure 4.2). Vane anemometers utilise the flow of air to turn a small turbine, the rotational speed being proportional to the flow velocity. Cup anemometers used for recording wind speed at meteorological stations use the same principal in a more robust form. Hot wire anemometers use the flow to chill a small heated element and monitor the energy used to maintain the element at a constant temperature this being proportional to the velocity. In the most sensitive of these instruments the element is a short length of extremely fine ni-chrome resistance wire, whereas in the more general purpose instruments the element takes the form of a bead thermistor. Both types operate in a similar manner in that the electrical resistance of the element varies with temperature.

Both types of instrument find applications in wind tunnel studies. Vane anemometers have the disadvantage that the instruments are quite bulky and need a clear unobstructed flow to work correctly. For this reason they are generally used during the calibration of a wind tunnel and to monitor the overall flow speed. For detailed measurements the hot wire probe is less intrusive due to the small size of the sensing element. The hot wire instrument has a number of disadvantages the most significant of which is that the response of the probe is non linear, and either the electronic controller or the display system must correct the readings. Hot wire anemometer probes are also dependent on the ambient temperature of the air being measured. This is generally overcome by having a sensor integrated into the probe to monitor the air temperature the output of which is then used to correct the signal from the probe control



Figure 4.2 Forms of anemometer. (a) Airflow Instruments, hot wire anemometer and (b) vane anemometer.

system. Most of the general purpose hot wire probes use an integral amplifier unit and give a linear output which overcomes all of problems associated with this type of instrument. The more sensitive wire ended probes generally have a separate amplifier unit with controls to linearise the probe output.

4.3.2 The Experimental Directly Heated Airflow Probe

As part of the experimental work conducted for this thesis, a Rover Maestro light van was fitted with instrumentation to record the ventilation conditions of the passenger compartment. As part of this instrumentation package a simple, robust airflow probe was constructed and mounted in the heater inlet duct (Figure 4.3). The probe consisted of a small, low voltage heating element, which was fed from a stabilised power supply, the power consumed by the heater was a constant 3.6W. Two bead thermistors were used to monitor the temperature of the air flowing through the inlet duct and the temperature of the sensor element. This type of probe design was chosen as no corrections for changes in the ambient air temperature are required. The parameter being

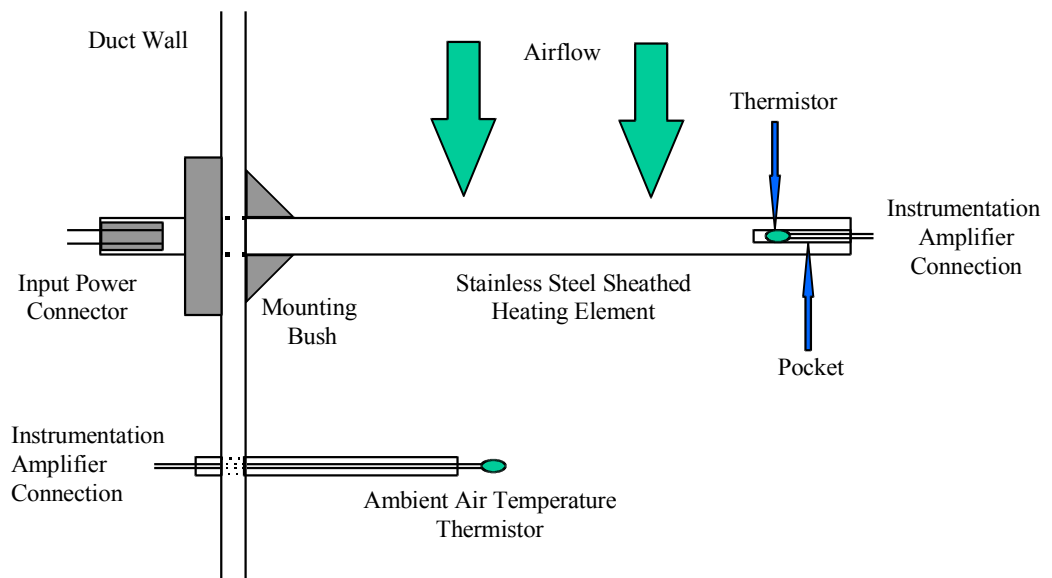


Figure 4.3 The directly heated airflow probe used in the Rover van

measured is the temperature difference between the probe and the air flowing through the duct, changes in the ambient temperature will be reflected in the probe surface temperature and may be ignored.

The probe was initially calibrated against a Furness Controls digital manometer which was set to measure flow speed rather than pressure. This initial calibration allowed the performance of the probe to be evaluated at a known temperature and flow speeds, these measurements were then used to calculate the k-factor for use in the chill factor power equation :-

$$E_{cf} = E_{in} - kT_d^{5/4} \quad (4.2)$$

where:-

E_{cf} = The power lost from the probe due to the flow chill factor (W)

E_{in} = The input power to the probe (3.61W)

T_d = The temperature difference between the probe and the surrounding air (K)

k = The k-factor of the flow measuring system (0.0343)

$10^{E_{cf}}$ is directly proportional to the flow speed and only needs to be scaled :-

$$U = \frac{10^{E_{cf}}}{343} - 2.08 \quad (4.3)$$

The equation gives the flow rate of the air in metres per second, by calculating the power being dissipated by the probe due to the chill factor of the air passing over the probe surface. This technique will only work in a non condensing atmosphere as the evaporation of water from either of the two temperature sensors will lower the measured temperature. The probe was found to work satisfactorily with air temperatures of between 5° and 25° centigrade and with flow speeds of between 1.0 and 6.0m/s, the time constant of the probe was around 2.5 seconds. As the sensor had a non linear characteristic curve (Figure 4.4), large measurement errors were found at very low flow speeds (below 0.5m/s). This was felt to be acceptable as the flow velocity in the duct was

normally above 1m/s due to the action of the inlet fan. Across the normal working range the accuracy of the probe was found to be better than $\pm 5\%$.

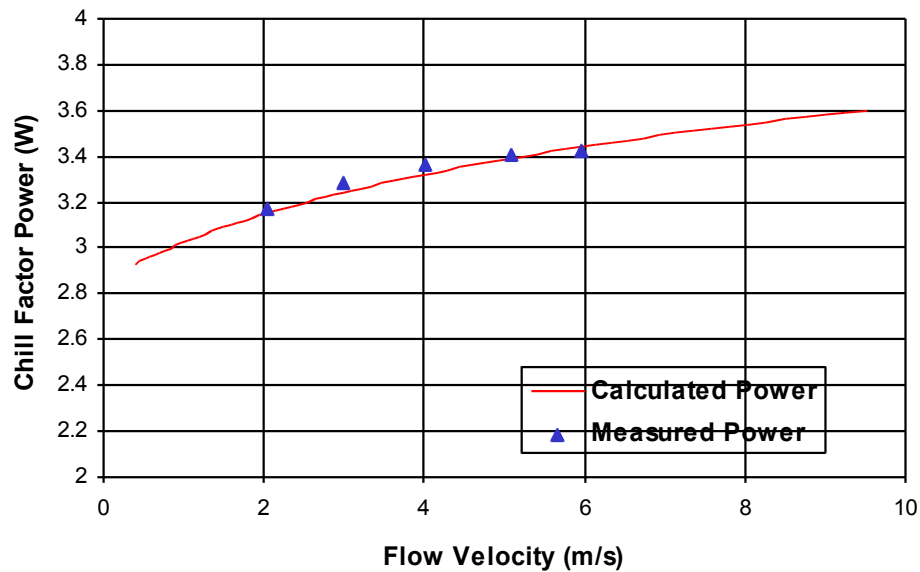


Figure 4.4 Calibration graph for the directly heated airflow probe used in the Rover van.

4.4.1.1 Measurement of Ventilation

The ventilation of an enclosure may be measured by recording all of the flows into and out of the space, or by injecting a tracer gas and monitoring changes in its concentration. Both methods allow calculation of the air exchange rate of the enclosure but tracer gas techniques have advantages where the flow through openings cannot be conveniently measured. During the experimental work conducted for this thesis tracer gases were used in all of the measurements involving the ventilation rate of the experimental vehicle cabin and during some of the wind tunnel simulations which modelled ventilation through openings.

The accuracy of tracer gas measurements depend on the complete mixing of the gas with the air to be measured. The effects of incomplete mixing of the sample gas can be overcome by injecting the tracer gas through a number of injection

points spaced around the area of the test, sampling from a number of points and recording the aggregate concentration. Alternatively the air can be mixed mechanically using fans. The mixing technique used depends largely on the tracer gas technique chosen for the ventilation test.

4.4.1.2 Tracer Gases

A tracer gas for use in ventilation measurements must have certain properties, an ideal tracer gas would possess the following:-

1. The gas must be safe to use, it should not be flammable, explosive, toxic or an irritant.
2. The gas must be non-reactive, it must not be absorbed by, or react with, any material in the area to be monitored or the measurement system.
3. The presence of the tracer gas should not affect the flow or density of the air being measured.
4. The gas must be distinctive, it should not normally be present in the air being measured. A tracer with a non-zero background level may be used as long as the background level is stable and that the tracer concentration to be measured is significantly larger than the stable background level.
5. The gas must be easily available.
6. The gas concentration must be quantifiable using an established experimental technique of known accuracy and reliability.

No easily available gases fulfil all the requirements of the ideal gas, but a number are used successfully as tracer gases. These include nitrous oxide, sulphur hexafluoride and carbon dioxide. Table 4.1 gives a broad outline of the properties of some of the most commonly used tracer gases.

<i>Tracer Gas</i>	<i>Formula</i>	<i>Density</i>	<i>8hr. Exposure</i>	<i>Remarks</i>
		<i>Relative to air</i>	<i>Limit (ppm)</i>	
Carbon dioxide	CO ₂	1.53	5000	Highly unstable background levels aggravated by building or vehicle occupants and combustion.
Nitrous oxide	N ₂ O	1.53	25	Anaesthetic gas, widely used as a tracer.
Sulphur hexafluoride	SF ₆	5.11	1000	Widely used as a tracer, detection is affected by other halogenated compounds in air.

Table 4.1 Properties of some commonly used tracer gases
(after Kohal 1995)

4.4.2.1 Tracer Gas Measurement Techniques

The essence of all tracer gas ventilation measurements is the tracer continuity equation :-

$$V \frac{dC}{dt} = Q[C_e - C_{(t)}] + F \quad (4.4)$$

Where :-

V = The effective volume of the enclosure, m³

Q = Specific air flow rate through the enclosure, m³/s

C_e = External concentration of tracer gas

$C_{(t)}$ = Internal concentration of tracer at time t

F = The production rate of tracer by all the sources within the enclosure

Some of the various methods of performing tracer gas measurements are discussed below. The derivation of the equations is more fully explained by Charlesworth (1988).

4.4.2.2 Tracer Concentration Decay

With this technique a single injection of tracer is made into the measurement area. It is important that the air is adequately mixed with the tracer and this is generally accomplished by the use of fans. A schematic diagram of the equipment required to conduct tracer decay measurements is shown in Figure

4.5. The concentration of the tracer gas is recorded as it decays due to the airflow through the space.

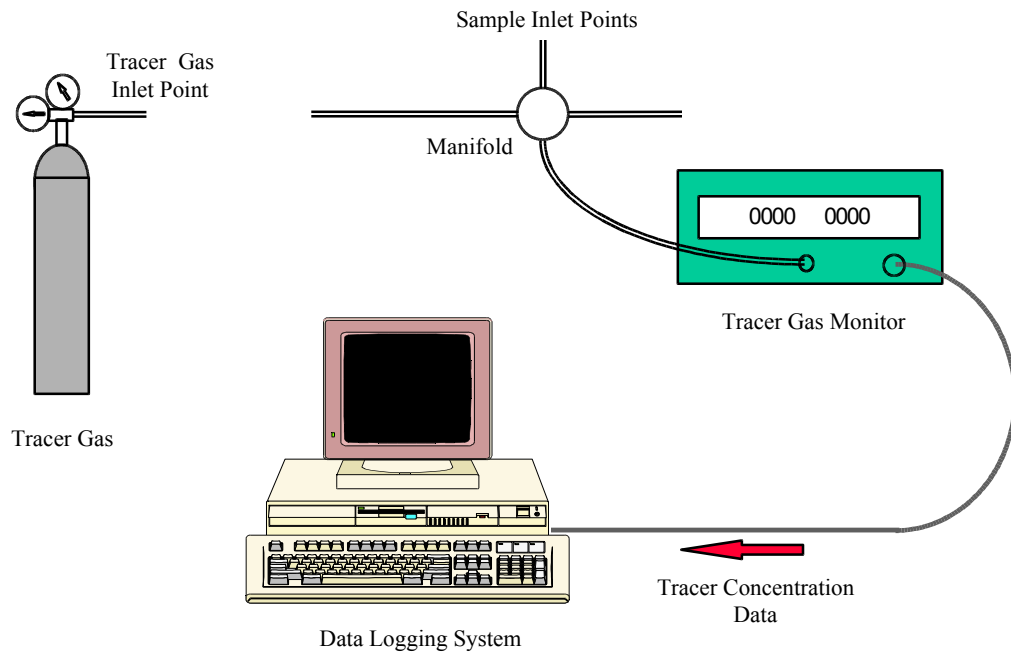


Figure 4.5 Layout of equipment required for conducting Tracer Decay measurements

Assuming that after the initial injection of gas the production of tracer is zero then the continuity equation reduces to:-

$$V \frac{dC}{dt} = -QC_{(t)} \quad (4.5)$$

And can be rearranged to give:-

$$\frac{dC}{C_{(t)}} = -\frac{Q}{V} dt \quad (4.6)$$

This equation may be solved by integration if the flow rate is assumed to be constant:-

$$\int_{C_{(0)}}^{C_{(t)}} \frac{dC}{C_{(t)}} = -\frac{Q}{V} \int_{t=0}^{t=t} dt \quad (4.7)$$

where $C_{(0)}$ = the concentration of tracer at time = 0

Hence:-

$$\ln C_{(t)} - \ln C_{(0)} = -\frac{Qt}{V} \quad (4.8)$$

This equation may be solved to give the variation of tracer gas concentration with time:-

$$C_{(t)} = C_{(0)} e^{-\frac{Q}{V}t} \quad (4.9)$$

$\frac{Q}{V} = N$ = Air change rate per unit time. Therefore if N remains constant over the measurement period the tracer gas concentration will exhibit a negative exponential decay.

4.4.2.3 Constant Emission Rate

Another approach to the measurement of ventilation is to produce tracer gas at a constant rate and to then measure the concentration as it settles to a steady value. The continuity equation is then of the form:-

$$V \frac{dC}{dt} = -QC_{(t)} + F \quad (4.10)$$

This equation may be solved in terms of the concentration of tracer gas:-

$$C_{(t)} = \frac{F}{Q} + \left[C_{(0)} - \frac{F}{Q} \right] e^{-\frac{Q}{V}t} \quad (4.11)$$

If the airflow into and out of the space remains constant this equation becomes:-

$$C_{(t)} = \frac{F}{Q} [1 - e^{-Nt}] \quad (4.12)$$

If N remains constant a finite time is required for the tracer concentration to reach equilibrium, once the tracer reaches equilibrium then the airflow rate is given by:-

$$Q = \frac{F}{C_{(t)}} \quad (4.13)$$

Hence by setting the generation rate of tracer to an appropriate value and monitoring the concentration, the ventilation rate may be determined. A diagram of a typical measurement system is shown in Figure 4.6. A critical section of this equipment is the mass flow controller which meters the generation of tracer gas within the test space.

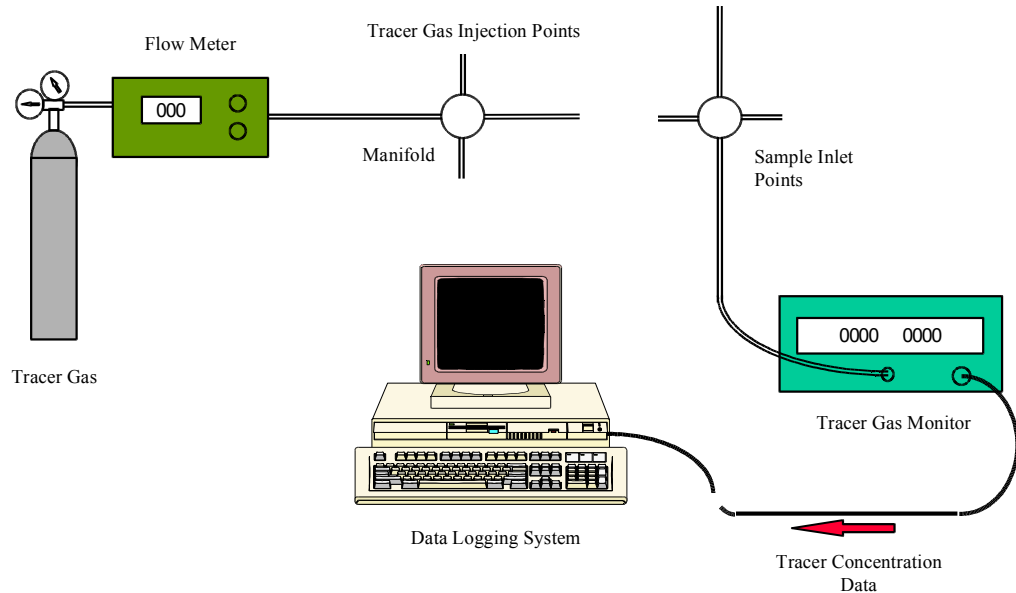


Figure 4.6 Layout of the equipment required to conduct Constant Emission Rate measurements

4.4.2.4 Constant Concentration

A third approach is to control the generation of the tracer gas so as to maintain a constant concentration within the measurement area. This immediately reduces the continuity equation to its simplest form:-

$$F_{(t)} - QC = 0 \quad (4.14)$$

Where $F_{(t)}$ is the generation rate of tracer at time t .

This may be solved to give:-

$$Q = \frac{F_{(t)}}{C} \quad (4.15)$$

Which is the same equation used for the constant emission method, only now the variable is the rate of tracer gas generation. This is the most demanding of the various measurement methods.

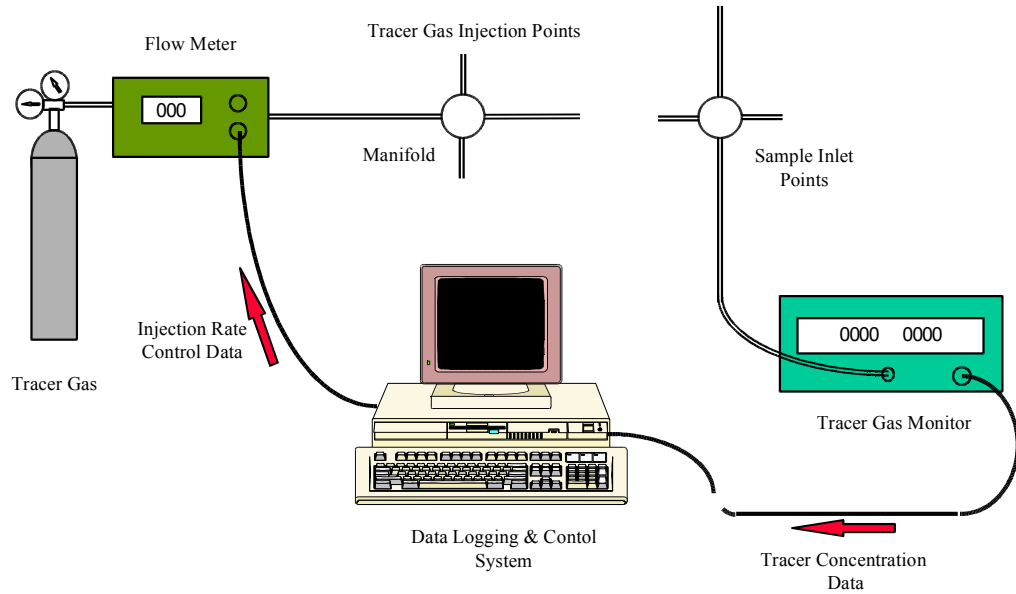


Figure 4.7 Layout of the equipment required to conduct Constant Concentration measurements.

In order to hold the gas concentration stable a control mechanism must be introduced between the concentration measurement system and the tracer gas injection system (Figure 4.7), the practical realisation of which invariably incorporates a micro-computer to control and monitor the flow of tracer gas.

Although this method is the most complex to set up, it has the advantage that it may be completely automated and is suitable for continuously monitoring the ventilation rate of large areas.

Chapter 5. Wind Tunnel Techniques

5.1 Introduction

Wind tunnel studies have become accepted as the most cost efficient and practical method of solving most aero-dynamic design problems. The Wright brothers aircraft the 'Wright Flyer' first 'flew' in a wind tunnel during 1902 and over the course of the 20th century wind tunnels have been used not only to design aircraft, but to study areas as diverse as the flow of water around power-boat hulls, smoke deflection on railway engines and the effect of wind on tall buildings (Rae and Pope 1984). From these examples it can be seen that the wind tunnel is a truly versatile design tool when used correctly. That is not to say there are no limitations on what may be achieved, for example, the tunnel must be large enough to accommodate models of a scale big enough to allow observation of the flow, the air flow through the tunnel must be representative of the real world and any relevant scale effects due to the relative density and viscosity of the air must be overcome.

In this chapter the techniques used to overcome variations in airflow due to model size are discussed. Simulation of the atmospheric boundary layer and turbulence are explained and the techniques used during the experimental work are described, although most of the experimental work was conducted without simulation of either effect, these simulations not being considered relevant to the outcome of the experiments.

5.2 The IBT Open Jet Wind Tunnel

The wind tunnel used for the model tests is based on a small open jet tunnel developed for teaching purposes by the BRE. The tunnel has a maximum flow velocity of 6m/s and a working section of width 1m, height 0.75m and length 2.25m. The dimensions of the working section effectively limit the size of model which may be satisfactorily tested (Figure 5.1).

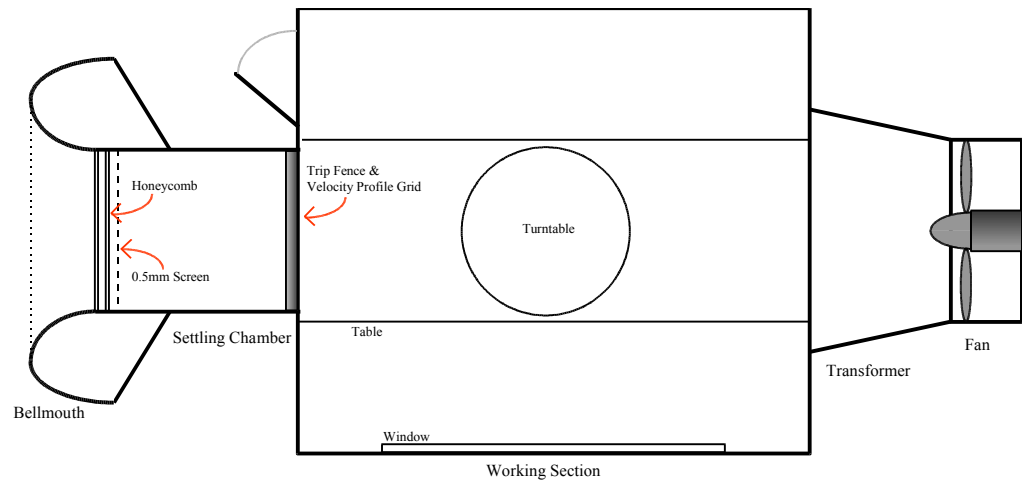


Figure 5.1 Layout of the IBT open jet wind tunnel (drawing not to scale).

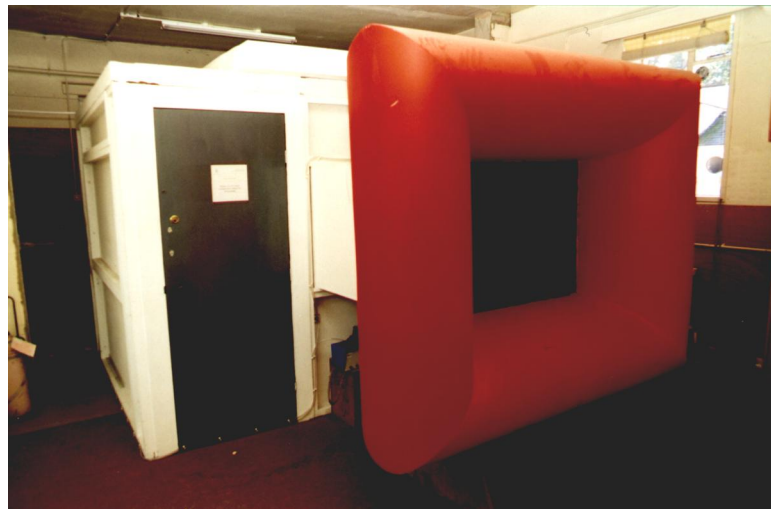


Figure 5.2 The entry section of the IBT wind tunnel.

The air enters the tunnel through a bell-mouth, shaped so as to minimise uncontrolled turbulence in the inlet section (Figure 5.2). This is immediately followed by two layers of honeycomb to straighten the flow and a 0.5mm mesh screen to further reduce turbulence. Before the air enters the working section it passes through a settling chamber in which a velocity profile grid and trip fence may be fitted, these allow simulation of the lower part of the atmospheric boundary layer and are used mainly during model building studies.

The working section of the tunnel consists of a table across which flows the jet of air drawn through the inlet by the fan. A turntable is fitted in the centre of the table to allow a model to be rotated. As the jet expands slightly as it flows through the working section the fan is preceded by a transformer with an inlet cross section slightly larger than that of the settling chamber outlet. The fan has a diameter of 900mm and gives a maximum flow velocity of 6.0 m/s across the table. Its speed is controlled by a variable frequency motor drive. The air is normally discharged through a large pair of doors situated immediately beyond the fan.

5.3 Scale Effects and Reynolds Number

Due to limitations on tunnel size and fan power many wind tunnel experiments are conducted with scale models. For many applications scale effects are not significant and may be ignored (Pope 1954a), tests involving pressure readings are one example. However for most model tests, scale effects will have a marked significance and will require correction. In order to minimise scale effects, tests involving models should ideally be conducted with the Reynolds number equal to that calculated for the full scale object.

Where:-

$$Re = \frac{vl\rho}{\eta} \quad (5.1)$$

and where:

ρ = Density of air (approximately 1.2kg m³)

η = Dynamic viscosity of air (1.72×10^{-5} Pa s)

v = Flow velocity (m/s)

l = Characteristic length (m)

As the Reynolds number for the model will be lower than that of the full scale object by the scale factor, the speed of the tunnel is generally increased to compensate for the reduction in characteristic length. This technique is suitable for objects, such as buildings, that only experience low or moderate flow velocities in the outside world. Another technique which may be used to decrease the flow speed, is to increase the air pressure inside the tunnel, this has the effect of increasing the density of the air and thus increases the Reynolds number (Pope 1954b).

When conducting wind tunnel experiments the Reynolds number should always be calculated. It serves as a guide to the conditions prevalent in the tunnel at the time of the test and is generally expected to be quoted in the results of any experimental work. During the wind tunnel tests involving model vehicles described in Chapter 7, the Reynolds number was between 40,000 and 85,000 depending on the tunnel speed. At full scale these Reynolds numbers would correspond to wind speeds of between 0.15 and 0.3 m/s (0.5 and 1.0 km/h).

5.4 Simulation of the Atmospheric Boundary Layer

The atmospheric boundary layer extends to an altitude of between 400 and 600m. Due to the limitations of the wind tunnel it is rarely practicable to model its entire depth. Fortunately to obtain satisfactory results it is sufficient to model only the lower portion of the boundary layer (Bauman *et al.* 1988). For a full simulation variations in the longitudinal, lateral and vertical components would need to be modelled, this is generally not required and simulation of only the longitudinal flow gives good results

A suitable model of the lower layers of the atmosphere is:-

$$U_{(z)} = \left(\frac{u}{K} \right) \ln \left[\frac{z-d}{z_0} \right] \quad (5.2)$$

Where:

$U_{(z)}$ = Mean velocity at height z

u = Friction velocity

K = von Karman's constant (0.4)

z_0 = Roughness length

d = Displacement height

The friction velocity may be defined as $\tau_0 = \rho u^2$ where τ_0 is the shear stress at the ground. Displacement height defines the height above ground at which the airflow conforms to the model and in smooth terrain can be assumed to be zero. Roughness length is a measure of the retarding effect on the wind by the terrain. For low-rise urban areas suitable values of z_0 and d are $0.3\text{m} \leq z_0 \leq 0.7\text{m}$ and $5\text{m} \leq d \leq 10\text{m}$ (Bauman *et al.* 1988).

To simulate the boundary layer a velocity profile grid is introduced into the inlet section of the tunnel (Figure 5.3). The grid shown gives a logarithmic profile to the airflow through the tunnel and is suitable for model building studies. During the model tests described in Chapter 7 the velocity profile grid was removed.

5.5 The Simulation of Turbulence

Modelling of turbulence in the wind tunnel can be achieved by using trip fences, turbulence grids or surface roughness. In small wind tunnels the usual method is to use a combination of techniques to give the required turbulence intensity and power spectra. In the IBT wind tunnel a trip fence may be fitted in place of the lower section of the velocity profile grid, this is normally followed by a short length of roughness made up of 50mm cubes (Figure 5.3). This technique gives a reasonable simulation of turbulence for undergraduate

teaching. In order to accurately simulate atmospheric turbulence in a tunnel, a long settling chamber is normally built immediately beyond the tunnel inlet section (often 10 or 12m in length) to allow turbulence to build up and develop over long lengths of roughness. The power spectra of the turbulence may then be modified by inserting blocks of varying dimensions or shape. Due to the length of the settling chamber in the IBT wind tunnel, only a simple approximation of atmospheric turbulence may be obtained.

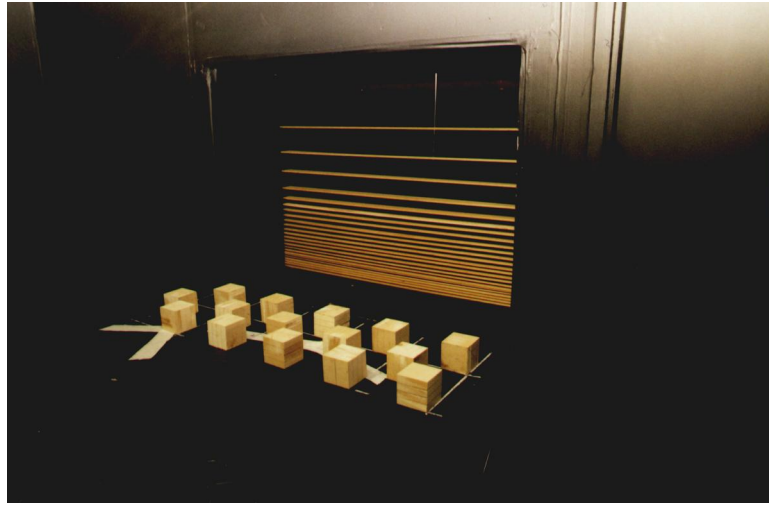


Figure 5.3 The velocity profile grid and roughness during a model building test in the IBT wind tunnel.

Turbulence intensity is a measure of velocity fluctuations compared to the mean flow velocity, it is a non dimensional quantity and can be defined as:-

$$I_i = \frac{(i'^2)^{\frac{1}{2}}}{U_{(z)}} = \sigma_i / U_{(z)} \quad (5.3)$$

Where:-

σ_i = The standard deviation of the instantaneous component of velocity at height z

I_i = The instantaneous component of turbulence intensity

i' = The fluctuating component of velocity at height z

$U_{(z)}$ = The mean velocity at height z

Turbulence intensity is always greatest near the ground where the airflow interacts with the surface roughness and decreases with increasing height. Typical values for low-rise urban terrain are around 0.4 close to the ground falling to around 0.2 at a height of 60 to 70 metres. For the scale model tests turbulence was created by the use of a trip fence and placing model vehicles in line, although this is not ideal it was felt that it gave a reasonable simulation of queuing traffic, the turbulence intensity being in the order of 0.3.

5.6 Modelling Cross Winds

For model vehicle tests a simple model was used to simulate the effect of cross winds. It was assumed that the effective wind would be a vector sum of the vehicle's forward velocity and the wind speed, which will be at an angle θ , to

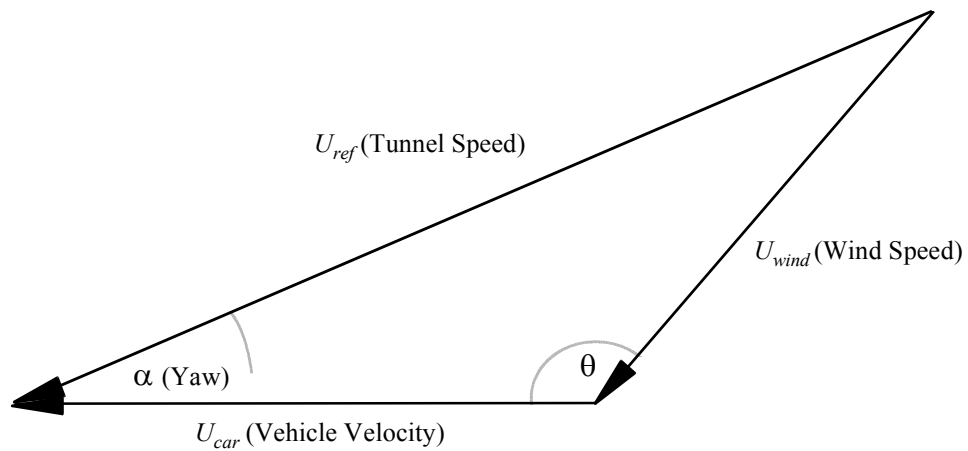


Figure 5.4 Vector diagram showing the velocities and angles required to calculate wind tunnel speed and model yaw, during cross-wind simulations.

the direction of motion of the vehicle (0° being a tail wind). Figure 5.4 shows the model as a vector diagram. The quantities to be found are the tunnel speed (U_{ref}) and the angle of model yaw (α).

The tunnel speed may be calculated thus:-

$$U_{ref}^2 = |U_{car}^2 + U_{wind}^2 - 2U_{car} U_{wind} \cos \theta| \quad (5.4)$$

The angle of yaw may then be calculated:-

$$\alpha = \arccos \left(\frac{U_{ref}^2 + U_{car}^2 - U_{wind}^2}{2U_{ref} U_{car}} \right) \quad (5.5)$$

This model was found to be adequate for the modelling of slowly moving traffic queues in a prevailing wind.

5.7 The Model Vehicles

The model vehicles used in the wind tunnel were based upon the MIRA simplified car, a well documented model shape (Shaw and Simcox 1988). Five models were constructed using 8mm medium density fibreboard (MDF) three at 1:10 scale and two at 1:5 scale (Figures 5.5 and 5.6). Two models were fitted with static pressure tappings (one of each scale) in order to measure not only the pressure distribution across the body but also the concentration of tracer gases during the experimental work.

The pressure tappings were formed from 0.5mm ID brass capillary tube and care was taken to ensure that they were fitted absolutely flush to the model surface. Tappings were fitted along the centreline of the vehicle, down both sides and across the front. The tappings were then connected by PVC tubing to a Furness Controls pressure scanner, located under the wind tunnel table.

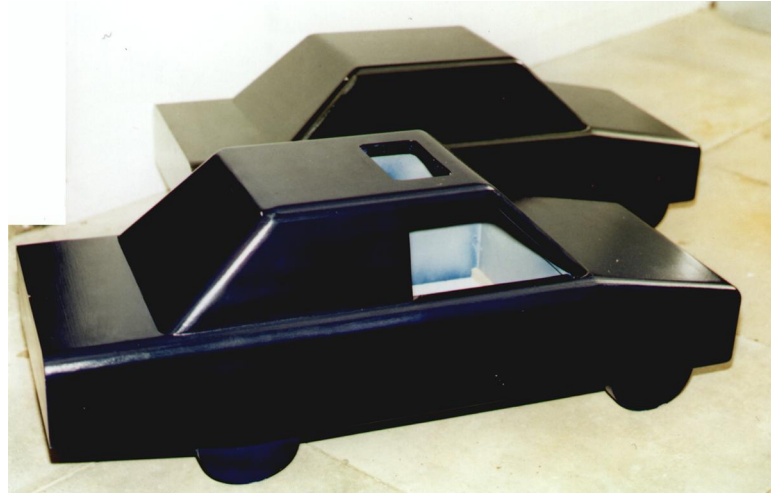


Figure 5.5 Two of the models used in the wind tunnel tests

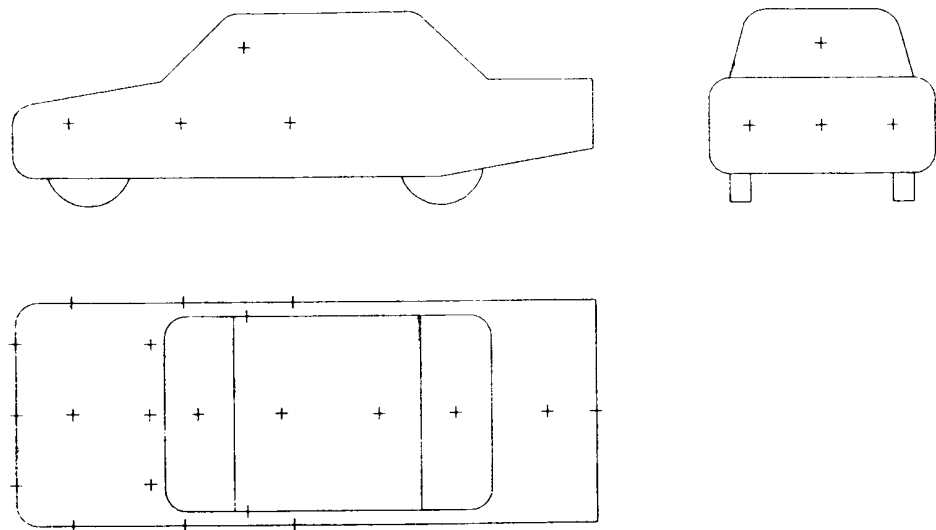


Figure 5.6 The MIRA simplified car shape showing the position of the static pressure tapings.

Three of the models (one of the 1:10 scale and two of the 1:5 scale) were fitted with tubes to allow the injection of tracer gas or smoke. The tubes were positioned at the rear of the model to simulate the engine exhaust pipe. Using

these tubes dispersion of the exhaust plume could be studied and the likely effects of having two or more vehicles in line evaluated.

Chapter 6. Vehicle Ventilation.

6.1 Introduction

Ventilation of a motor vehicle is achieved by forcing air into the cabin of the vehicle through vents at the front and exhausting the air through grilles towards the rear. This gives rise to the Ram Effect due to the air pressure differences across the body which is used by manufacturers to reduce the size of the ventilation fan needed to circulate air through the passenger cabin. Using the Ram Effect for ventilation has two inherent problems, the first is that no airflow takes place when the vehicle is stationary, or moving slowly, unless the heater fan is running. Secondly the air inlet vents have to be located in areas of the body which experience a rise in pressure as the vehicle moves forwards, this means that the air inlets are positioned in possibly the worst locations for use in slow moving traffic.

Ventilation air inlets are normally fitted to a vehicle either behind the radiator grille, or at the junction where the bonnet meets the windscreen, both are areas which experience high levels of exhaust contamination in slowly moving traffic queues. The bonnet vents also suffer from stagnation of the airflow across the vehicle body, which under certain conditions can cause short term peaks of pollutant to be trapped over the vent giving rise to high concentrations in the passenger compartment. Grille mounted vents due to their proximity to the engine exhaust pipes of other vehicles are also prone to drawing in high concentrations of pollutants.

Secondary ventilation is achieved through the opening of windows or a sun roof. Approximately half of the airflow through the vehicle under these conditions is due to single sided ventilation mechanisms and may be modelled as such. Sun roofs of the 'tilt opening' kind may lead to pollution entrained in a vehicle wake entering the vehicle cabin in moderate to high concentrations. This effect is due to the formation of eddies behind the lid of the sun roof trapping pollution not only from the vehicle immediately in front, but also in some cases, from a vehicles own exhaust.

This chapter endeavours to explain the various ventilation mechanisms relevant to light motor vehicles and presents the results of experimental work conducted to ascertain the performance of the ventilation system fitted to a typical vehicle.

6.2 Airflow Around the Vehicle Body

In order to understand the ventilation mechanisms involved more fully, it is helpful to examine the airflow across the vehicle body as it moves forward.

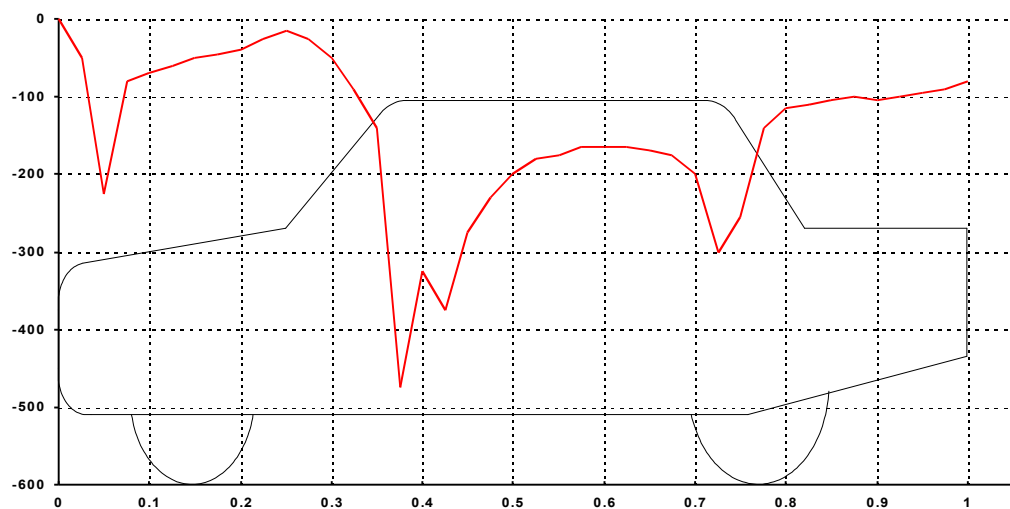


Figure 6.1 Pressure distribution along the centreline of a 'MIRA' car due to it's forward motion.

As a car moves forward the airflow over the bonnet and windscreen give rise to increases in air pressure. These occur across the front of the vehicle and at the junction of the bonnet and windscreen, it is this increased pressure that allows the Ram Effect to work. Figure 6.1 shows the pressure distribution measured along the centreline of one of the wind tunnel models and the increase in pressure at the base of the windscreen may be seen. The sharp dips in the distribution curve are due to areas of separated flow. Air is generally exhausted through vents in the rear doors or the boot of the car. These are placed in areas which experience a negative differential pressure with respect to the interior of the vehicle. As it is driven by pressure differences the Ram Effect does not have enough power to force a significant quantity of air through the passenger compartment unless the vehicle speed is relatively high. This can necessitate

having the fan running continuously in slow moving traffic in order to ventilate the vehicle adequately.

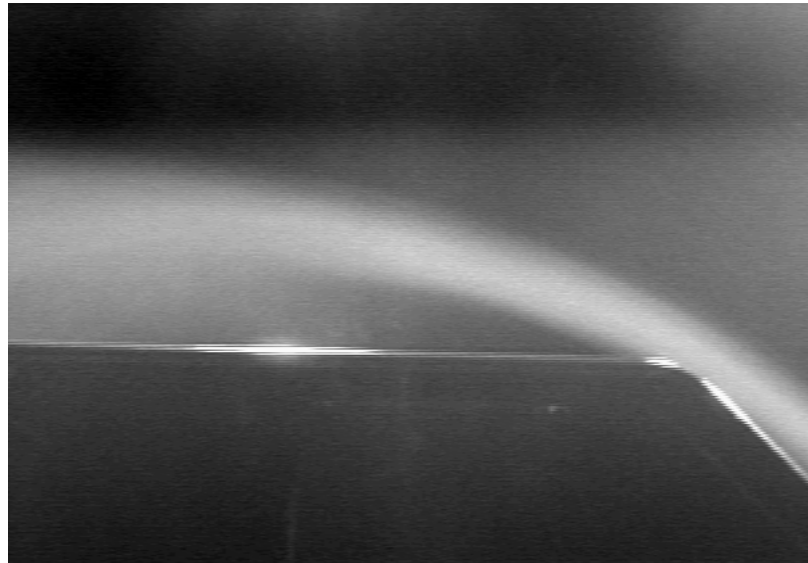


Figure 6.2 Smoke visualisation of flow separation at the leading edge of the roof of a 'MIRA' car in the IBT wind tunnel.

If a car is travelling directly into a head-wind the air will flow smoothly around and over the vehicle, with zones of separated flow around the nose of the vehicle, the forward edge of the roof and the rear window (Figure 6.2). As the cars direction veers away from a direct head-wind and the vehicle starts to experience a cross-wind, the separated flow on the leeward side of the vehicle extends further along the body and on the windward side reduces toward the vehicle nose. Airflow across the bonnet of the vehicle is deflected by the wind and can start to form a vortex at the junction of the bonnet and windscreen (Figure 6.3), the air no longer flowing smoothly over the windscreen and roof. The air is then trapped along the leeward side of the vehicle.

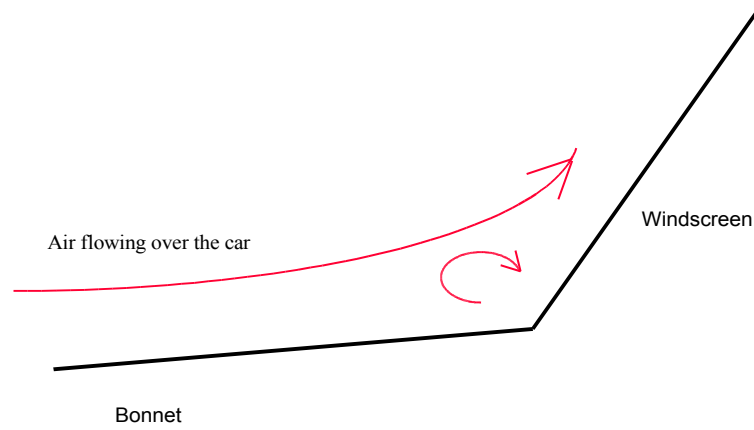


Figure 6.3 The formation of a vortex at the junction of the bonnet and windscreen.

The instantaneous wind speed across the vehicle will be the magnitude of the vector sum of the vehicle velocity and the wind velocity :-

$$U_i^2 = |U_{car}^2 + U_{wind}^2 - 2U_{car}U_{wind}\cos\theta| \quad (6.1)$$

Where U_{car} is the forward velocity of the vehicle, U_{wind} is the wind velocity and $\cos \theta$ is the angle of incidence of the wind. The effects vary in magnitude according to the geometry of the vehicle body and the magnitude of the resultant of the vector sum.

Computational Fluid Dynamics (CFD) although a useful tool for visualising flow around objects and across surfaces, must be used with caution when considering the extremely turbulent flows encountered around vehicle bodies and in a vehicle wake. The modelling of the vehicle wake in particular poses a number of problems, the greatest of which (and probably for many researchers the most insurmountable) being the sheer complexity of the CFD grid and the level of computing power needed to calculate the result. For both these reasons no attempt was made to study the vehicle wake in detail using CFD. The relative velocities of the air flowing over the vehicle body may be successfully modelled using CFD techniques (Figures 6.4 and Figure 6.5). It can be seen that the velocity of the air flowing across vents situated at the junction between

the windscreen and bonnet is extremely low and may stagnate. As a vehicle moves forward the air flowing over the body originates from a point approximately 0.5m ahead of the front bumper. The air divides evenly between the upper surfaces and underside of the vehicle, this may also be seen using CFD. Figure 6.6 shows a simulation of the flow of pollution emitted from a car in a traffic queue. In this simulation the streaklines show the path of pollution over the second vehicle. It must be remembered that this is an idealised case and that in reality the pollutant would be more diffuse due to the turbulence encountered in the vehicle wake.

6.3 Ventilation Using the Heating and Ventilation System

The ventilation system fitted to a modern car has two main functions, to maintain the interior temperature at a comfortable level and to demist the windscreen and side windows. In order to fulfil these requirements the system draws external air into the vehicle passenger compartment, a proportion of which is passed through a small heat exchanger connected to the engine cooling system. The heated air is then directed through vents to demist the windows or directly into the passenger compartment through vents generally situated in the footwells. Some vehicle heaters have a re-circulation system, in which the incoming air may be mixed with air drawn from the passenger compartment, the advantage being a faster heater response and more stable temperature control.

The heater fitted to the test vehicle is of the non re-circulating type (Mead 1994) and is fitted in all models of Rover Maestro and Montego vehicles manufactured after 1986 (Figure 6.7). Air is drawn in through an inlet situated below the windscreen and enters the fan housing. The air flows through the fan and is then directed by a flap valve either through or around the heater matrix.

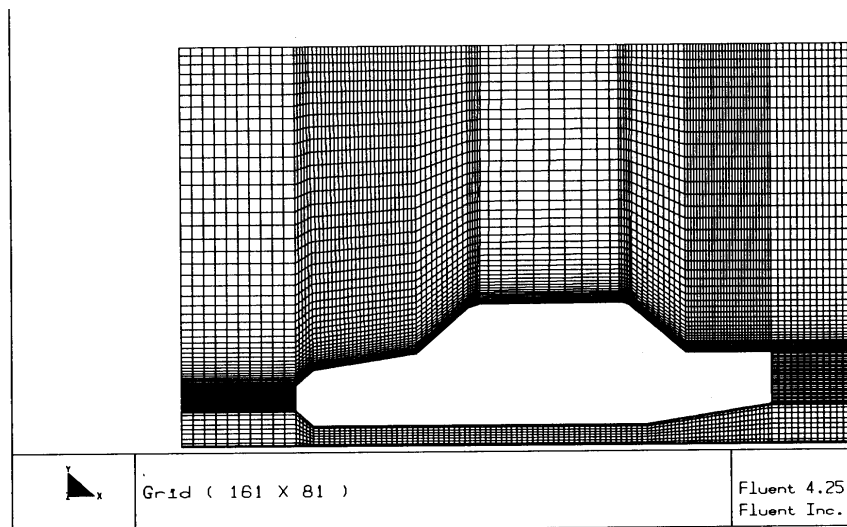


Figure 6.4 Computational grid around one MIRA vehicle

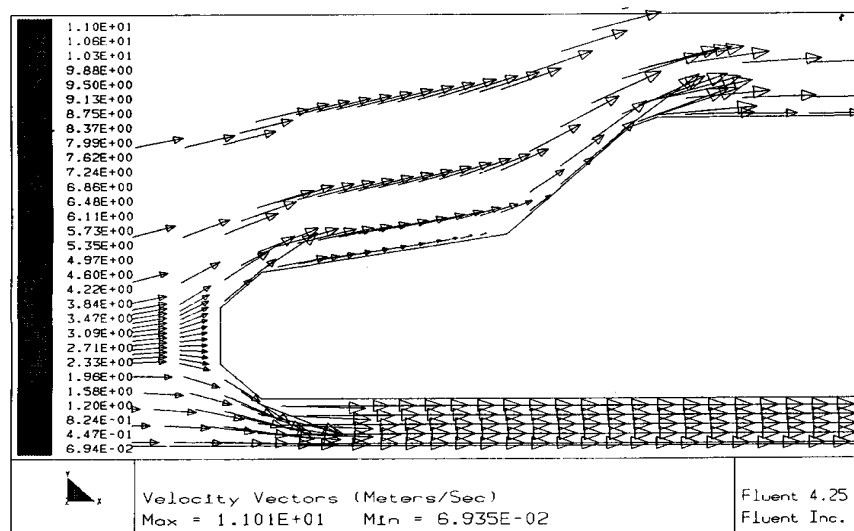


Figure 6.5 Velocity vectors around the front of the vehicle



Figure 6.6 Streaklines from around the exhaust of the first vehicle cover the surface of the second.

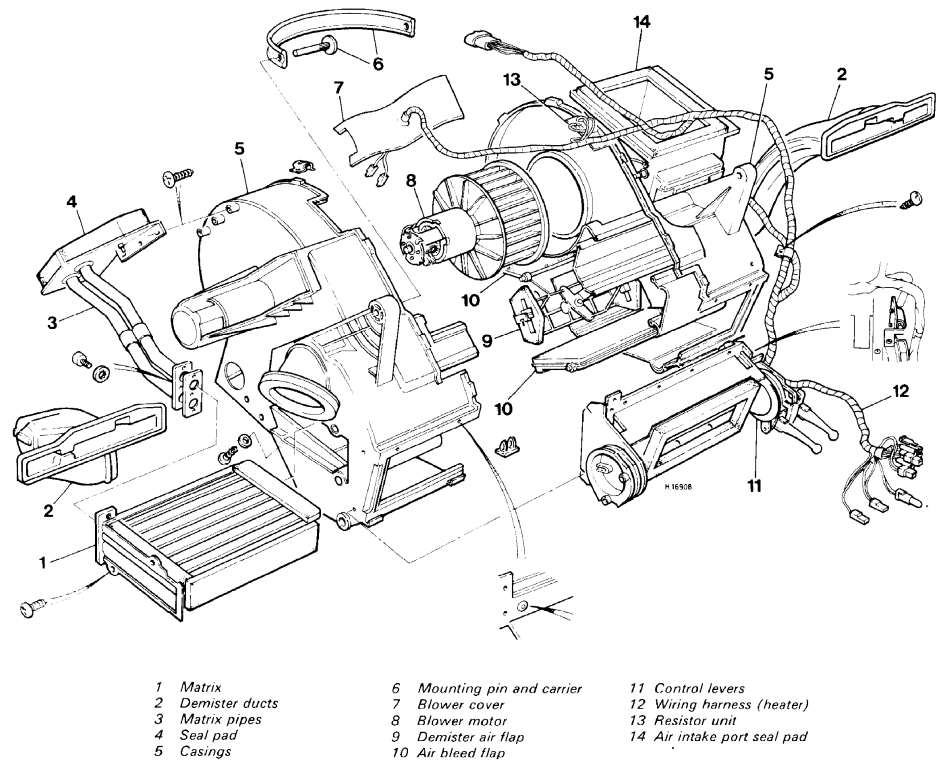


Figure 6.7 Exploded view of the heater fitted to the Rover Maestro test vehicle.

(Drawing © Haynes Publishing, Sparkford, UK.)

This allows the ratio of heated to unheated air to be varied and so allows control of the temperature. The heated air is then directed, by means of another flap valve to either the foot level vents or the windscreen demister vents. Face level vents are provided on the dashboard, these are fed with air directly from the outlet of the fan.

Control of the fan is achieved with a four position rotary switch, which controls the fan speed by means of a wire-wound resistor unit mounted on the fan casing. The heater flap valves are controlled directly by means of two mechanical slider controls mounted on the dashboard.

6.4 Ventilation Through Windows or a Sun-Roof

6.4.1 Single Sided Ventilation Theory

Ventilation using the windows and sun-roof of a motor vehicle is similar to the mechanism used to naturally ventilate buildings. Turbulence causes eddies in the flow of air across the opening, which in turn cause positive and negative pressure variations in the plane of the opening relative to the pressure inside. The flow through the opening is thus a function of the pressure distribution across the opening at any given time (Figure 6.8).

Experimental measurements (Crommelin *et al.* 1988) have shown that the geometry of the opening plays little part in the air exchange mechanism and the opening can be thought of in terms of area alone with the air exchange rate being nearly proportional to its size. The air exchange rate was also shown to be dependant upon the speed of the flow across the opening this again being nearly proportional to the flow velocity.

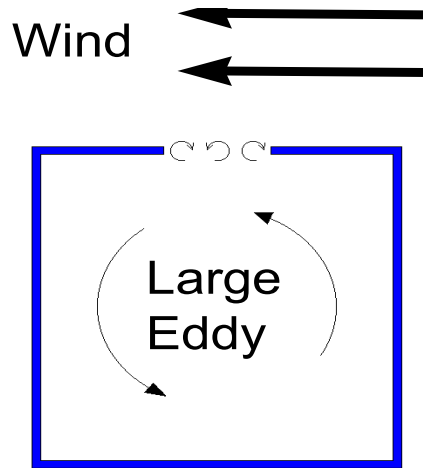


Figure 6.8 Single sided ventilation

Because ventilation increases with the flow velocity and opening size nearly proportionally, a dimensionless ventilation (Q) can be defined:-

$$F = \frac{Q}{AU} \quad (6.2)$$

Where A is the area of the opening and u is the flow speed across the opening. F is dependent on Reynolds number but in practice may be considered as a constant with a value of about 0.03.

6.4.2 Wind Tunnel Simulation of Single Sided Ventilation

Experiments were conducted with a 0.025m^2 opening in one wall of a 0.125m^3 box placed below the wind tunnel table, the opening being flush with the table surface. The opening was divided by a cotton grid to allow consistency in the positioning of an air velocity probe across the plane of the opening. The velocity magnitudes across the opening are shown in Figure 6.9. Although the differences are small it can be seen that the velocity rises towards the rear of the opening, with the lowest velocities being present in the centre and toward the leading edge. There was considerable fluctuation in the velocity magnitudes with time, however the time constant of the probe tended to damp these and consequently a mean value was recorded.

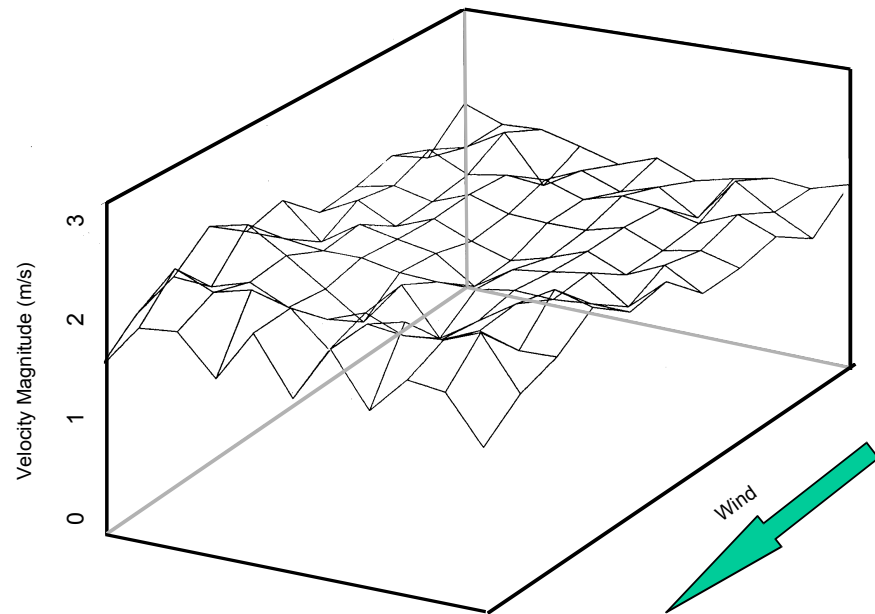


Figure 6.9 Velocity magnitudes across the box opening

In order to record the higher frequency components of the flow, a piezo-electric pressure transducer connected to a short length of 0.5mm bore copper tube was used to monitor pressure fluctuations due to eddies being formed across the opening. The data was recorded using a storage oscilloscope and analysed using a computer software program entitled 'MathViews'. As the frequency component of the signal was to be observed, it was important that the sampling frequency was at least twice the highest required frequency present in the signal. During the tests the sampling frequency was set to either 20 or 50Hz. The raw data was analysed using a Fourier transform algorithm which allowed the frequency spectrum of a signal to be observed, the Nyquist frequency being exactly half of the sampling frequency (Figure 6.10). The shedding frequency of eddies from the rear edge of the hole was found to rise proportionately to the tunnel flow speed (Figure 6.11) confirming the findings of Crommelin (1988).

The flow inside the box was then examined, the flow across the opening produces a rotating flow in the plane of the external flow. With the square opening this was found to have a maximum velocity across the width of the

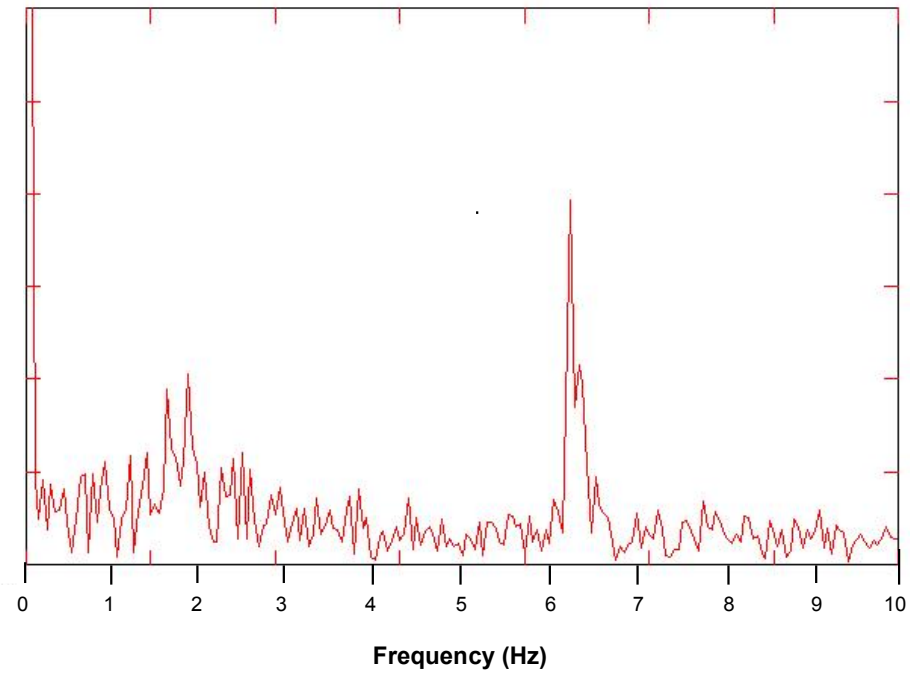


Figure 6.10 Fourier analysis of a typical data set, the large peak occurs at 6.2Hz. This data set was recorded with the tunnel speed set at 2.5m/s.

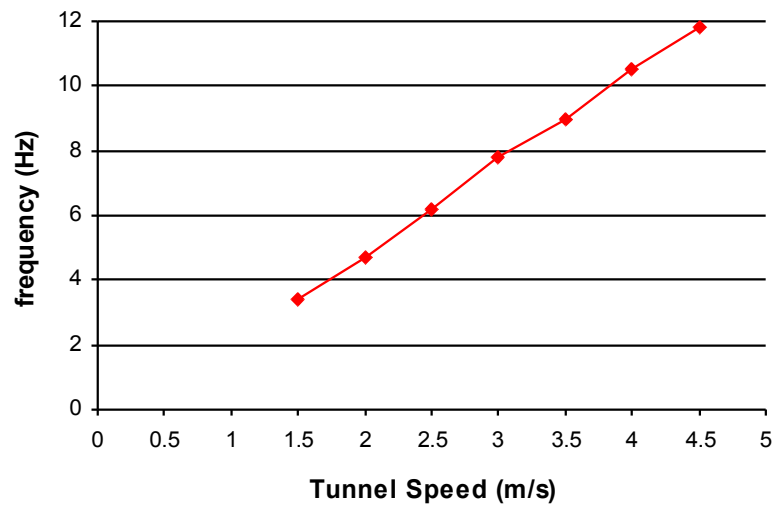


Figure 6.11 Vortex shedding frequency by wind tunnel speed.

opening, which decayed towards the edges of the box. At the edges of the box small vortices were observed perpendicular to the main flow. The velocity of

the main rotating flow was found to be directly proportional to the flow velocity, this was expected as the driving force of all the flows observed in the box is the flow across the opening (Figure 6.12).

After the flows inside the box had been observed, tracer gas (N_2O) was injected to measure the air exchange rate through the square hole using the constant emission rate technique. Taking F as 0.03 calculation of the theoretical values of Q/V gave a good approximation of the actual measured value (Figure 6.13).

6.5 The Ventilation Rate of the Test Vehicle.

The test vehicle was used to measure the ventilation of a “real” vehicle as it travelled at a constant speed on a public road. To ensure that it was possible to maintain a steady speed the experiments were undertaken during the early hours of three Sunday’s during early summer. A multi-point tracer injection system was fitted into the vehicle and the resulting concentration was continuously monitored using a Miran 2b multi-gas analyser. The readings were recorded using a Grant Instruments 1200 series ‘Squirrel’ logger with the time interval set at two seconds.

Tracer decay tests were first undertaken to determine the leakage from the vehicle when it was stationary, both with the drivers’ window open and with it closed. With the window closed the air change rate of the vehicle was found to be around two air changes per hour and with the window open this rose to three air changes per hour. With the fan running at its lowest speed the air change rate in both instances rose to around six air changes per hour, the fan being the dominant ventilation mechanism whilst the vehicle is stationary.

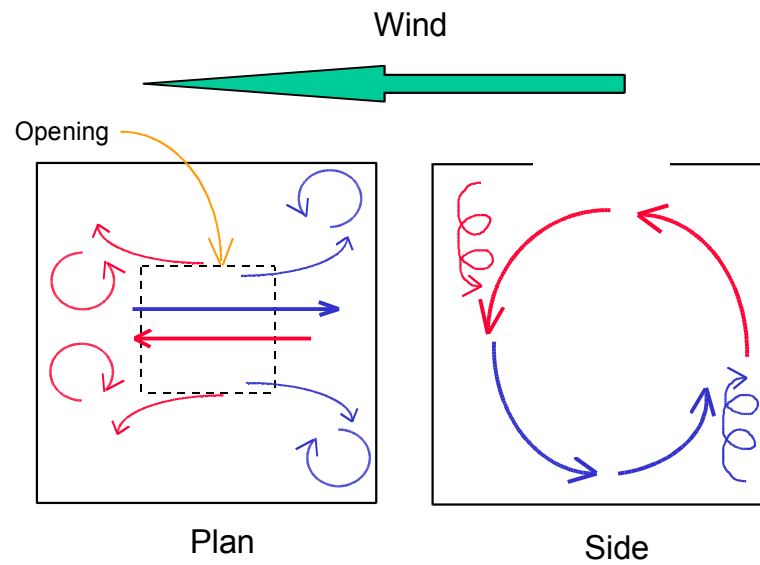


Figure 6.12 The main flows observed in the box.

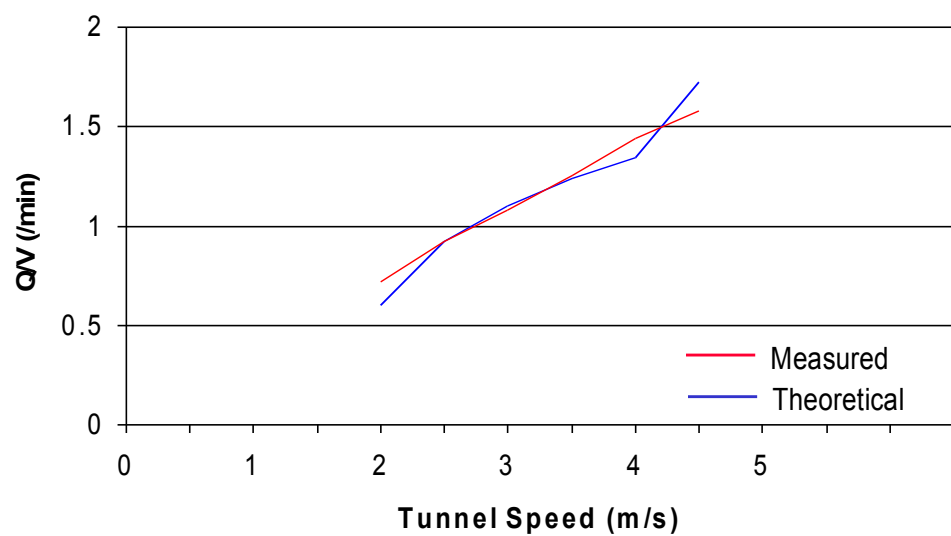


Figure 6.13 Calculated and measured values of the box ventilation rate, by wind tunnel speed.

The moving vehicle tests were then undertaken, using the constant emission method. The tracer gas (SF_6) was injected into the van cabin at a constant rate,

through the multi-point injection harness, the gas inlet being metered by a Bronkhorst mass flow controller which was set to give an injection rate of 0.5 l min^{-1} . The gas concentration was initially allowed to build up to around 250ppm and the van was then driven at as constant a speed as possible for five to six minutes. A plot of the calculated ventilation rate from data recorded at four different road speeds is shown in Figure 6.14. This set of results clearly shows the different ventilation rates calculated directly from the 'raw' gas concentration data. The time taken for the tracer concentration to stabilise can also be seen particularly in the 80km/h section of the plot. Prior to this journey the vehicle had been stationary for a short period whilst the logging system was adjusted and the concentration of tracer gas had increased due to the low ventilation rate. In order to remove the variation caused by small fluctuations in speed and those due to turbulence around the vehicle body, the arithmetical means of the data recorded at each speed were calculated. These are plotted in Figures 6.15 to 6.17 and show the variation of ventilation rate due to the velocity of the vehicle.

With the window closed and with the fan turned off (Figure 6.15) the airflow into the vehicle is solely dependent upon the Ram Effect. The ventilation rate is highly speed dependent at low vehicle speeds, whilst at higher vehicle speeds the ventilation rate stabilises at approximately 2.25 air changes per minute. With the window open the ventilation rate rises and becomes more speed dependent. Figures 6.16 and 6.17 show data recorded both with and without the heater fan running. The fan has little effect at low speeds, but does give a slight increase in ventilation at higher road speeds.

By subtracting the data recorded with the window closed from that recorded with the window open, the effect of the window opening can be seen. This is

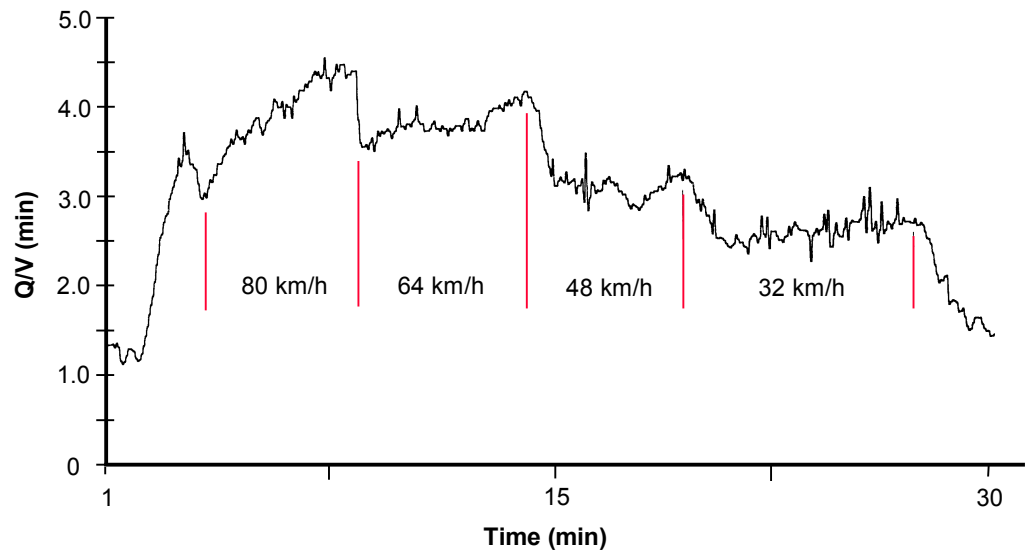


Figure 6.14 A typical set of ventilation data. Several such data sets were recorded for each ventilation condition.

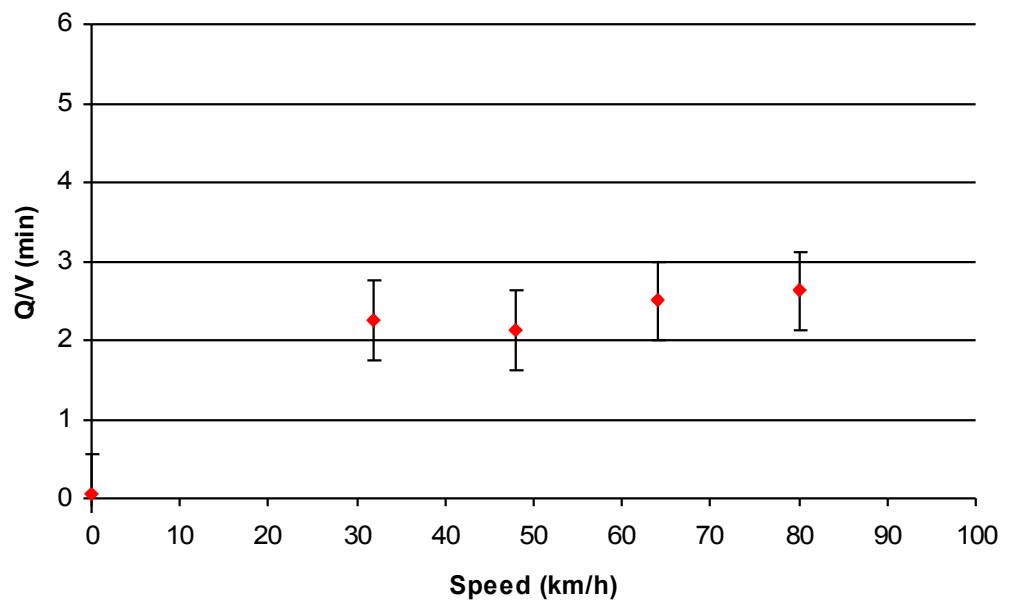


Figure 6.15 Van ventilation rate (window closed, fan off).

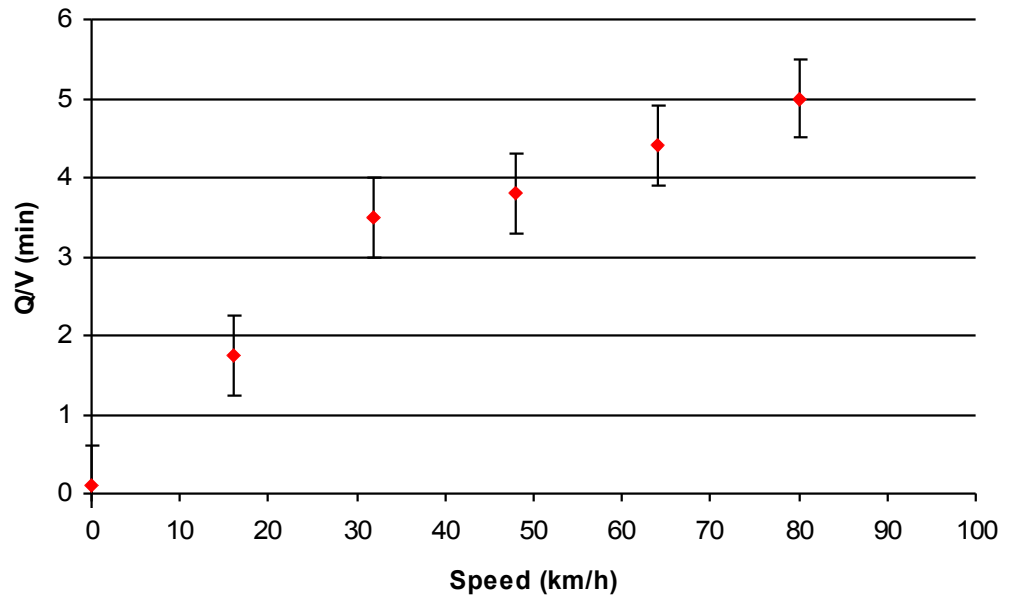


Figure 6.16 Van ventilation rate (window open, fan off).

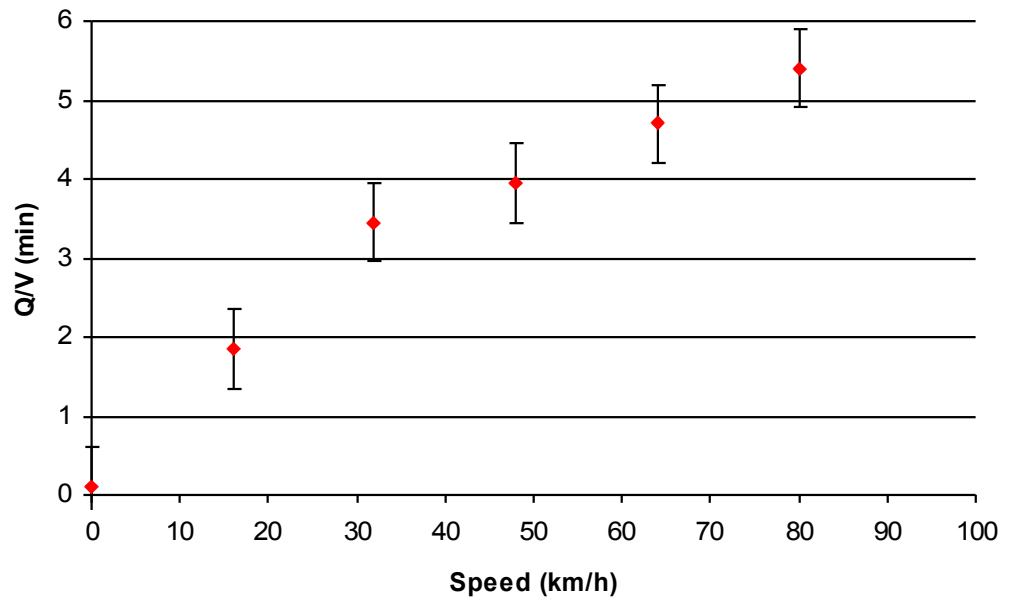


Figure 6.17 Van ventilation rate (window open, fan on).

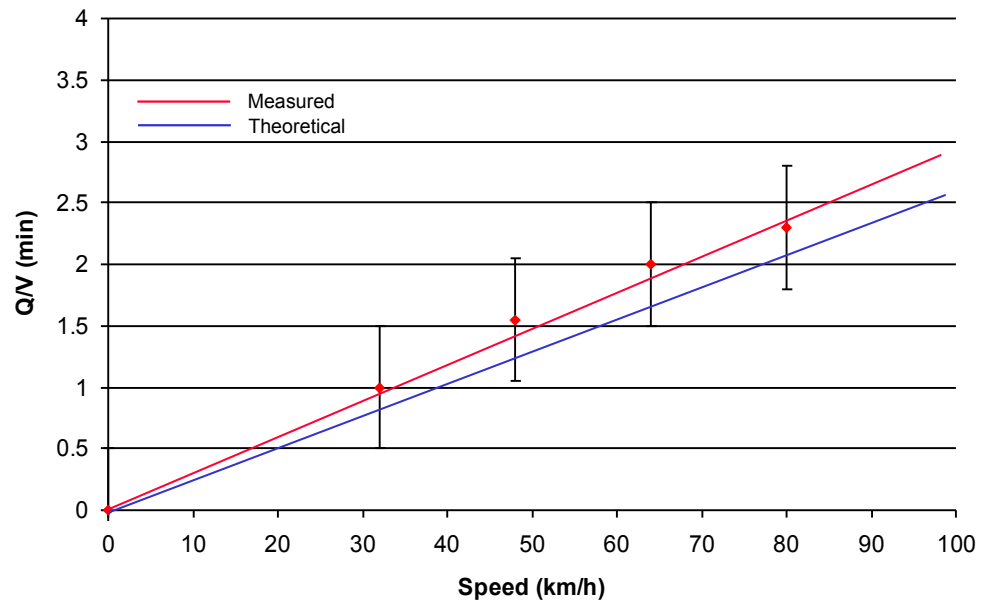


Figure 6.18 The actual (red) and theoretical (blue) effect of the window opening.

shown in Figure 6.18 is plotted alongside the theoretical values. These were calculated from the window open area (0.25m^2), and measured road speed, taking F as being 0.03 (equation 6.2). The measured data show a distinct correlation with the calculated theoretical values. The slightly raised values of the measured data are probably due to the highly turbulent nature of the flow around the vehicle body.

During the moving vehicle experiments it was felt that atmospheric effects could be ignored. All the experimental data were gathered during the early hours of the morning, the experiments being started at around six-o'clock and continuing until approximately half past seven, when the presence of other vehicles began to make a continuous speed untenable. Wind speeds were low relative to the vehicle velocity for all the experiments and so could be ignored and on the days that the experiments were taking place the weather was consistently fine. The roads used for the experiments dictated the upper speed at which it was possible to reliably measure the vehicle ventilation rate. It was found that a constant 80 km/h (50 mph) could be maintained for a long enough

time to allow a reliable measurement to be made. Measurements were attempted at 96 km/h (60 mph) but a suitably long stretch of continuous, straight road could not easily be found.

Several conclusions can be drawn from the ventilation experiments. At slow vehicle speeds the Ram Effect is ineffective as a means of ventilating a vehicle cabin necessitating continuous use of the heater fan to ventilate the vehicle interior adequately. At high vehicle speeds the Ram Effect becomes the dominant ventilation mechanism and the fan has no significant effect. Due to the fan having to be run continuously in slow moving traffic, pollution passing over the front of the vehicle will tend to be drawn into the vehicle cabin. This effect is made more severe due to the location of the vents which are positioned so that the Ram Effect may be used. These locations are prone to contamination from surrounding vehicles. If windows are opened, approximately half of the air flow through the cabin is due to single sided ventilation across the window openings. This can be predicted satisfactorily as a simple single sided ventilation model. This permits the geometry of the window opening to be ignored, and relies purely on the area of window opening.

Chapter 7. Dispersion of Traffic Pollution

7.1 Introduction

The dispersion and spatial distribution of traffic pollutants around urban areas has been well researched. Hall *et al.* (1996) discusses the variation of atmospheric pollutants measured at sites around Birmingham and Walsall and concludes that at a very localised level there are a large number of spatial variations in pollutant concentrations. Although this work was concerned with pollutants flowing over buildings, studies in London have found similar spatial variations in traffic pollutant concentrations (Croxford *et al.* 1995). These studies are interesting as they show that although the background concentration of pollutant gases may be very low, higher concentrations can develop in very localised pockets. This phenomena is apparent on a congested road. Measurements conducted in both Jerusalem (Menachem *et al.* 1990) and Tehran (Rashidi and Massoudi 1980) both found considerably higher concentrations of traffic pollution measured at the centre of the carriageway than at the kerbside.

As the concentration of pollutant at the centre of busy roads can be shown to be higher than that of the surrounding area, it is reasonable to assume that the concentration experienced by vehicle passengers may be higher than that experienced by pedestrians. Research undertaken in the United States tends to back up this argument. Chan *et al.* (1991a) found that the exposure of car commuters in Washington to various atmospheric pollutants was greater than that of cyclists or pedestrians. Ott *et al.* (1994) also found that at busy road junctions, the concentration of traffic related pollutants rose in the vehicle cabin.

In this chapter the results of experimental work conducted to evaluate the concentration of pollutant present at points across a vehicle body whilst travelling in congested traffic are presented. The work takes the form of a series of wind tunnel experiments designed to simulate a queue of slow moving vehicles. These experiments are supplemented by measurements of traffic

pollution at varying distances from a busy road which were conducted in part to validate work done by other researchers.

7.2 Dispersion of Carbon Monoxide from Roadside

In order to measure the rate at which traffic pollutant concentrations decay, an experimental system was constructed to measure the concentration of carbon monoxide at intervals perpendicular to a busy road. The traffic on the road chosen for the survey is generally free flowing but does become stationary for short periods of time during the rush hours. The system consisted of seven sampling points located 1m above the ground, fed back to a switching box, gas sensor and data logging system (Figure 7.1). Samples from each point were drawn across the gas sensor at 15 minute intervals and after allowing time for the dead air to be purged from the tubes the concentrations were recorded and logged.

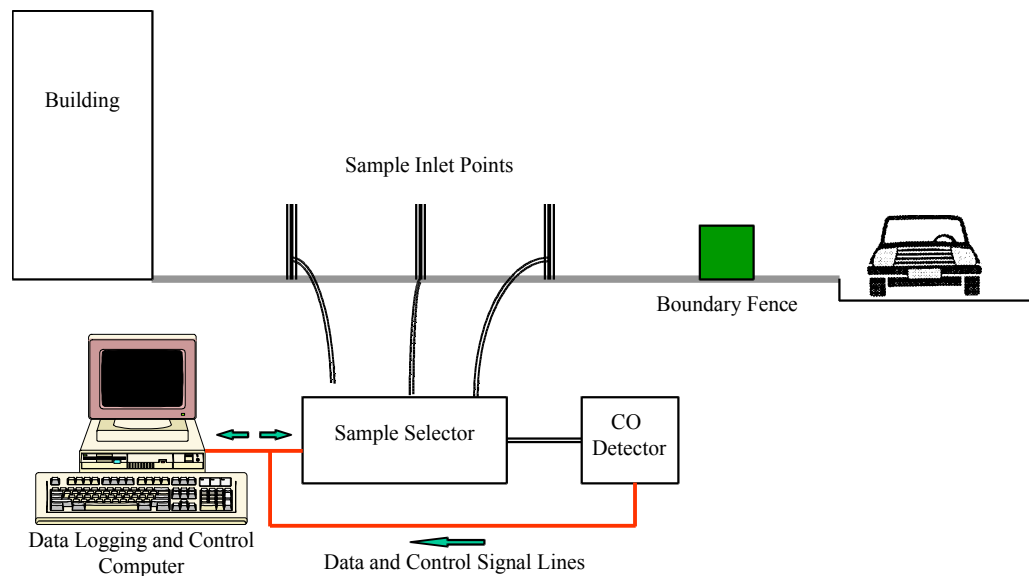


Figure 7.1 Schematic diagram of the experimental system used to record pollution dispersion.

Two sets of data were collected from two different locations adjacent to the A52 Nottingham to Derby trunk road, over a period of two months. The first location chosen was relatively overgrown and the sampling points were set at

1m intervals. The averaged data are plotted in Figure 7.2, and shows that the concentration of CO falls almost linearly as the distance from the road increases, with the mean decay being -0.14 ppm/m.

The second set of data was recorded at a location with a more open aspect and the sampling points were set at 2m intervals (Figure 7.3). In this set the concentration again decays with increasing distance. The overall rate of the decay is markedly less than that of the first data set, being -0.045 ppm/m, this may be due to the lack of bushes or trees to diffuse and absorb the pollutant gases. The differences in initial concentration may be attributable to the traffic accelerating as it passed the first location, which was situated at the brow of a hill and about 300m from a busy road junction.

It is interesting to compare this data with concentration measurements taken from a moving vehicle during the experimental period. Over the course of the experimental period the roadside readings varied between 0 and 3ppm, dependent on the time of day and wind direction. The readings recorded from a vehicle travelling past the survey site varied between 10 and 15ppm. These findings are in agreement with those of Rashidi and Massoudi (1980), who found that the concentration of pollutants fell sharply at the kerb and then decayed gradually across the pavement. This experiment therefore, tends to reinforce the view that vehicle passengers must experience higher concentrations of pollution than other road users. The reason being that they are not simply close to the flow of traffic, but are an integral part of the flow and hence are in the worst location.

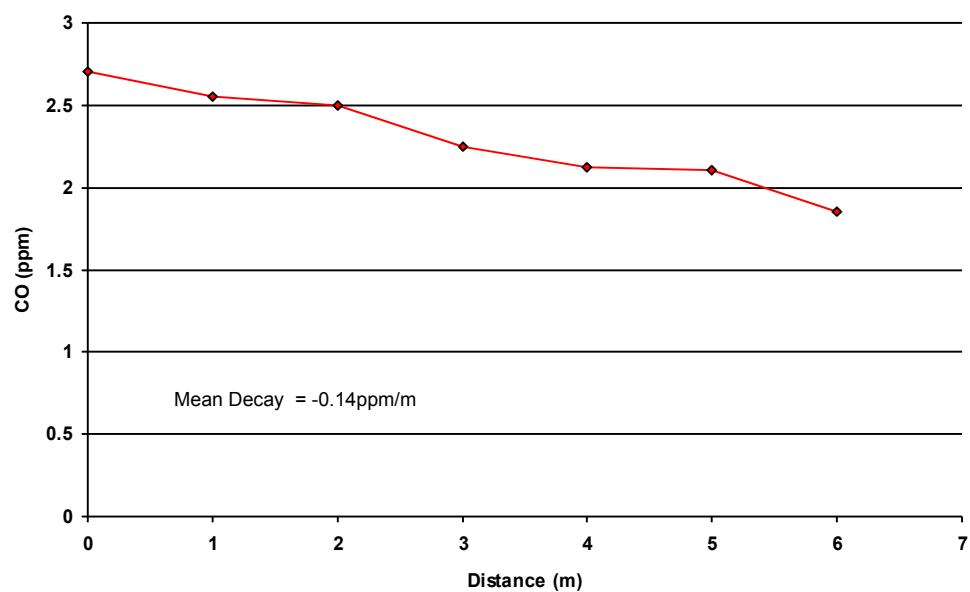


Figure 7.2 Average CO concentrations recorded at 1m intervals from the roadway for the densely overgrown site (0 = boundary fence).

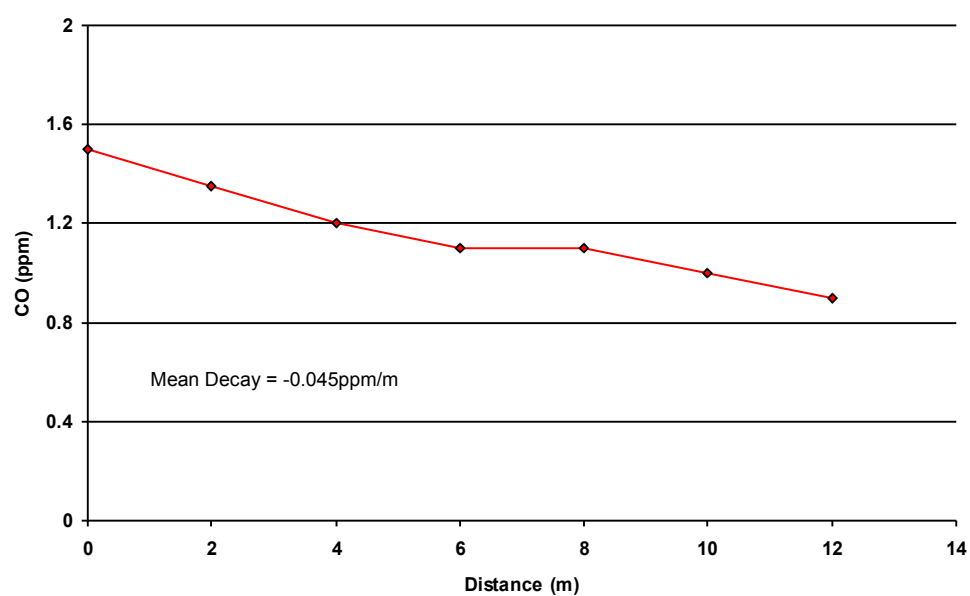


Figure 7.3 Average CO concentrations measured at 2m intervals from the roadway for the open site (0 = boundary fence).

7.3.1 Model Simulations of Traffic Pollution

In order to obtain an understanding of the structure and the behaviour of vehicle wakes, a series of wind tunnel simulations were conducted. The tests were planned so as to simulate the airflow around queuing and very slow moving traffic. No attempt was made to simulate turbulence or any boundary layer effects as the IBT wind tunnel is not suitable for turbulence modelling at the scale of the models being used (1:5 and 1:10).

7.3.2 Smoke Visualisation of Traffic Queues

The initial series of simulations consisted of a series of smoke visualisations to show the path of pollutants across vehicle bodies whilst in queuing traffic. Two of the 1:10 models were placed 200mm apart, in line on the tunnel turntable and oil smoke was injected using a probe situated beneath the right hand rear corner of the 'leading' vehicle (Car 1) (Figure 7.4). The wind tunnel was run at a speed of 2.5m/s and the flow of smoke was observed as it passed over the 'following' car (Car 2). The Reynolds number was calculated to be 43,600 and would correspond to a full scale wind speed of approximately 0.25m/s (0.9km/h).

With the vehicles facing directly into the wind (0 ° yaw) the smoke flow was densest along the right hand side of Car 2, with some mixing taking place in the leading vehicle wake. Very little of the smoke was observed to flow down the left hand side of the following car. This would seem to indicate that ventilation through the side windows would lead to high concentrations of pollutant gases entering the cabin, this does not seem to be the case. Chan *et al.* (1991b) found that pollution concentrations inside a vehicle were lower when the windows were opened due to the high ventilation rate. When the angle of yaw set to 15 ° the amount of mixing in the wake of the Car 1 increased and a distinct flow pattern was visible in the vehicle wake. Smoke was observed to cover the boot and rear window of the leading car and the second car was running in a corridor of well mixed smoke. The smoke flow was now travelling predominantly along the left hand side of Car 2 flowing around the

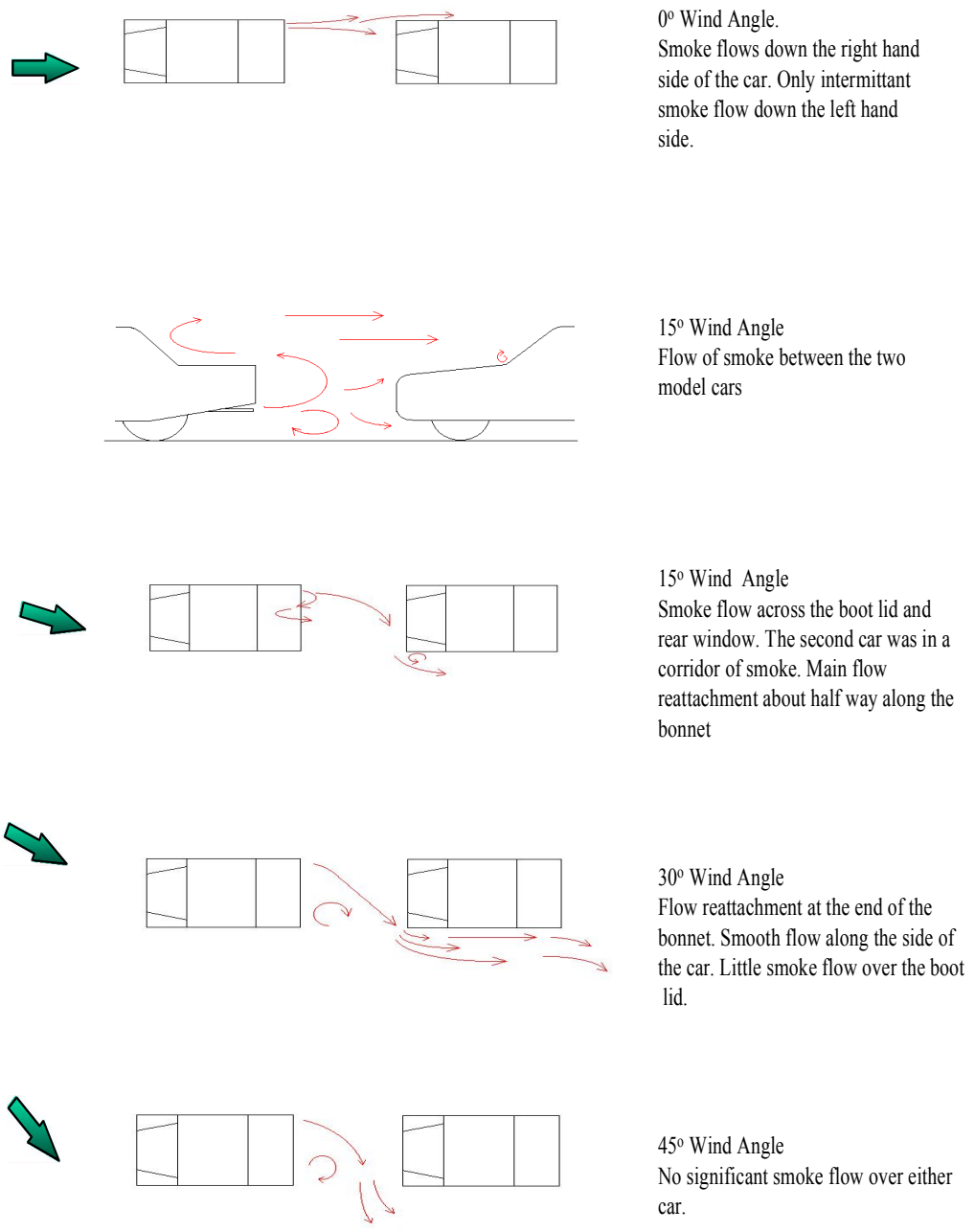


Figure 7.4 Wind tunnel visualisation of the airflow around two 1:10 scale model cars.

front and diagonally across the bonnet and windscreen. Flow separation was visible at the leading edge of the front left hand wing with re-attachment occurring approximately half way along the wing.

With the angle of yaw set to 30° very little of the smoke passed over either the leading cars boot lid or the bonnet of Car 2. The area of separated flow along the front wing of the following car was observed to have lengthened with the re-attachment point now level with the bonnet / windscreen junction. At 45° yaw there was no smoke flow over either Car 1 or Car 2, the smoke being dispersed by the wind flowing between the two models.

The visualisation tests, although simple, show if a vehicle is moving slowly in a queue of traffic on a calm day most of the exhaust emitted from the car in front will flow around the front of the following vehicle and across its bonnet. The worst case occurs when the airflow is very slightly offset to the centreline of the car. In such a case, if the leading car's exhaust is located on the windward side of the vehicle, the following car will be travelling in a plume of exhaust gases. These results are only valid at relatively low road speeds. At higher speeds not only does the distance between vehicles increase, but the mixing in the vehicle wake becomes very much greater.

7.3.3 Tracer Gas Simulation of the Exhaust Plume

Whilst visualisation using oil smoke gives a subjective view of the dispersion characteristics of the exhaust plume, tracer gas techniques can give a quantitative view. The configuration of the models in the wind tunnel was the same as for the smoke visualisation with a 200mm gap between the cars and with the tracer gas injection tube positioned beneath the right hand rear corner of the leading model (Car 1) (Figure 7.5). Sampling tubes were connected to sixteen static pressure tappings on the surfaces of the 'following' vehicle (Car 2) and fed through a Furness Controls selector box to a Binos 1000 gas monitor.

In order to simplify the data obtained from the simulation the measured data are expressed as the ratio of the measured pollutant to the emitted pollutant:-

$$\alpha = \frac{C_m}{C_s} \times 100$$

Where:

α = Dimensionless pollutant fraction

C_m = The measured concentration of pollutant (ppm)

C_s = The emitted pollutant concentration (ppm)

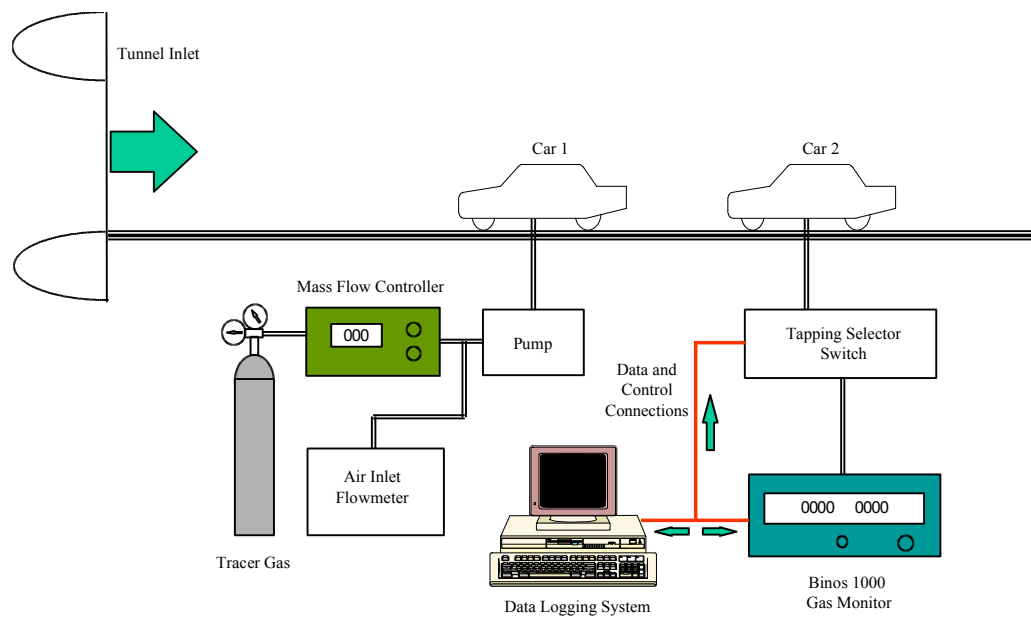


Figure 7.5 The wind tunnel setup used for the tracer gas measurements. Two cars used for the initial tests, a third car and a second tracer injection system were added for the test described in section 7.3.4.

The tracer gas (nitrous oxide N_2O) was mixed with air so that the emitted concentration from Car 1 was 500,000ppm at a flow rate of 1l/min. Experimental data were obtained at tunnel flow speeds of 2.0, 3.0 and 4.0m/s at Reynolds numbers of 41,000, 63,000 and 84,000 respectively. The behaviour of the tracer gas plume is similar to that of the smoke. Figures 7.6 and 7.7 show the fraction of the emitted pollutant (α) recorded at the tappings with a flow speed of 2.0m/s.

At 0° yaw the flow of tracer is predominantly over the right hand side of the trailing model (Car 2), although due to the turbulence caused by the leading vehicles wake there is a small amount of tracer measured down the left hand side. The highest levels of tracer are shown to be present at the rear of the trailing vehicles bonnet Tappings were purposely located at this location as it corresponds to the position of the ventilation inlet vents on most modern cars.

With a yaw angle of 15° the plume of tracer gas flows predominantly around the left hand side of Car 2, the right hand side experiencing a negligible concentration of tracer. There are however measurable concentrations of tracer at the left hand bonnet tapping. At yaw angles of 30° and 45° no tracer gas could be measured on the model at all.

It is apparent that there is considerable variation of the concentration of pollutant due to small changes in the prevailing wind direction and these can conveniently be shown as graphs. Figure 7.8 shows the pollutant fraction recorded at the right hand bonnet tapping at several model yaw angles. It can be seen that the concentration rapidly falls to zero as the yaw angle increases and that there is very little variation due to the flow speed.

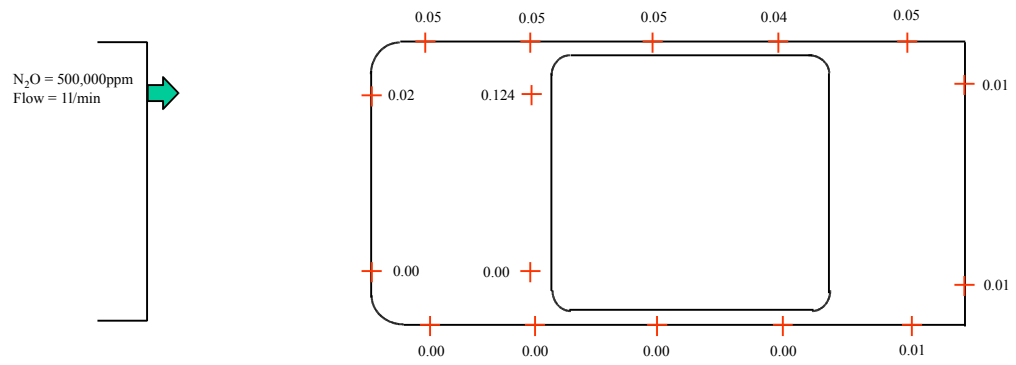


Figure 7.6 Tracer gas as a percentage of the emitted concentration (α) at a yaw angle of 0° .

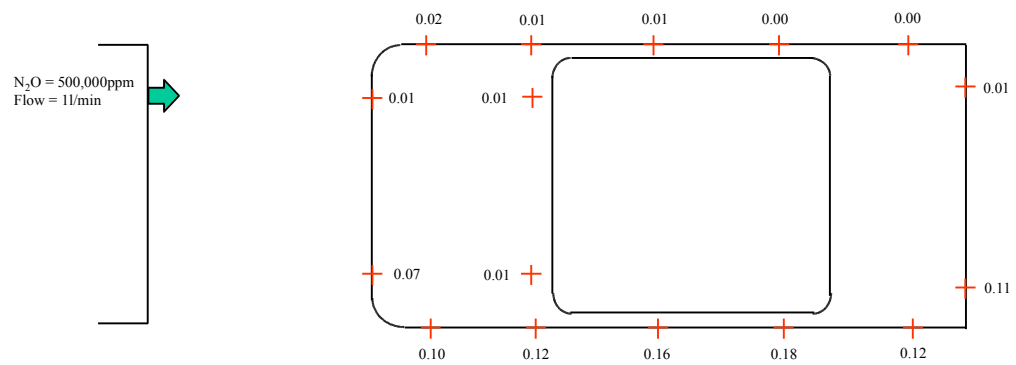


Figure 7.7 Tracer gas as a percentage of the emitted concentration (α) at a yaw angle of 15° .

Figure 7.9 shows the pollutant fraction recorded at the left hand bonnet tapping of Car 2. At 0° there is a negligible concentration of pollutant present at the tapping. As the yaw angle increases, however the concentration rises as gas flows diagonally across the bonnet. At an angle of 30° the pollutant fraction has fallen back to a negligible level as the gas flows through the gap between the two vehicles. The concentration of tracer gas is also significantly lower than that measured on the right hand side of the model, this is attributable to dispersion of the gas in the vehicle wake. The increased concentration at higher flow speeds is due to the airflow across the bonnet, which tends to form a vortex at the junction of the bonnet and windscreen, trapping the tracer gas plume and reducing the mixing effects of the vehicle wake.

7.3.4 The Dispersion of Two Exhaust Plumes.

A logical progression of the previous experiment is to consider the effect of having more than two vehicles in line (Clifford *et al.*1997). In order to model the dispersion of pollutant two 1:10 scale models were fitted with tracer injection tubes, again positioned at the right hand side rear. SF₆ tracer was injected from the exhaust of the first car (Car 1) and N₂O was injected from the exhaust of the second car (Car 2) both at a concentration of 250,000 ppm and at a flow rate of 1l/min. A third car (Car 3) was fitted with pressure tapings connected through a Furness controls selector box to a dual channel Binos 1000 gas analyser. The models were again set 200mm apart on the wind tunnel table and the tunnel started at an initial flow velocity of 1.5m/s. Since the flow around the models was turbulent it was necessary to average the readings over 20s on each tapping. When the test had been completed the tunnel speed was altered and the experiment repeated. In this manner data were recorded for a range of flow speeds. The tapings used during the series of tests were located along the centre line of Car 3. The car was expected to be running in a well mixed plume of the tracer gases caused by the increased turbulence due to the presence of the third car and all of the tests were conducted with a zero yaw angle.

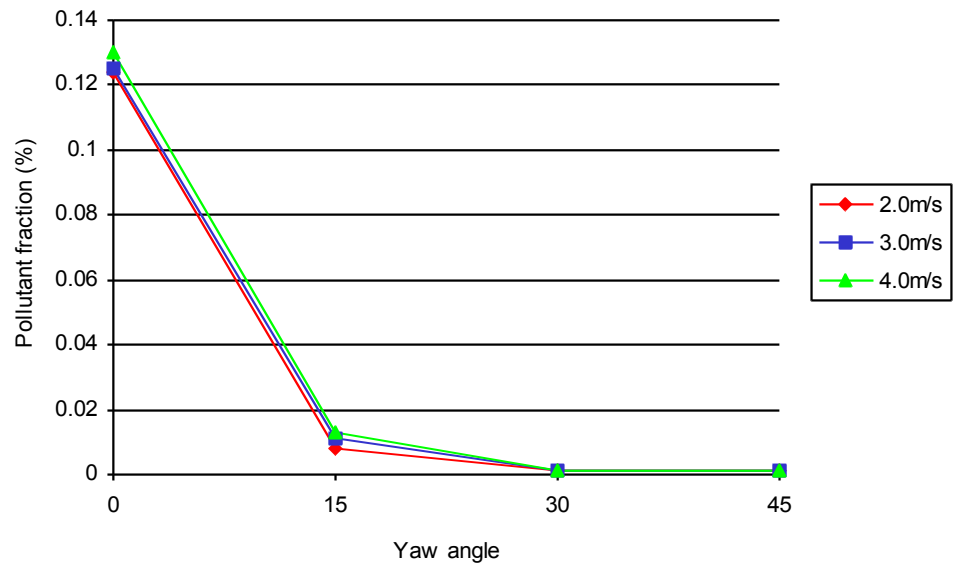


Figure 7.8 Tracer gas concentration shown as a percentage of that emitted at the right hand bonnet tapping by model yaw angle.

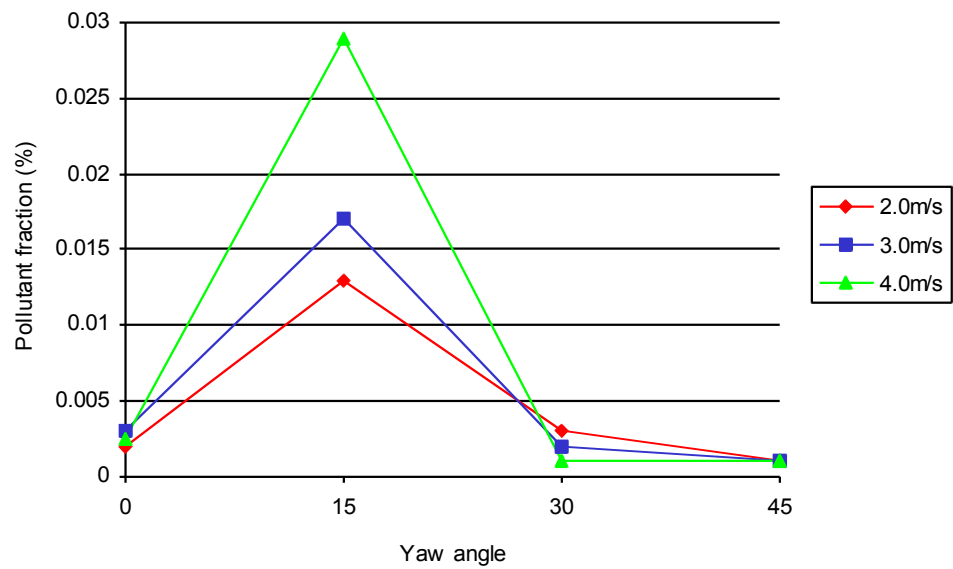


Figure 7.9 Tracer gas concentration shown as a percentage of that emitted at the left hand bonnet tapping by model yaw angle.

The data from each of the tests at a constant tunnel speed shows that there is considerable variation in the tracer gas concentration over Car 3 from Car 2. This can be seen in Figure 7.10 where the fraction of N_2O recorded along the centre line of Car 3 is plotted. It can be seen that the three curves follow a similar pattern and that the concentration of tracer gradually falls along the length of the vehicle but recovers slightly at the rear. This trend is more pronounced at the lowest wind speed. The reason for this pattern may be deduced by considering the flow around the vehicle, the concentration falls along the length of the vehicle due to the increasing distance from the source. The rise at the rear is due to tracer flowing along the sides and being mixed in the turbulent wake.

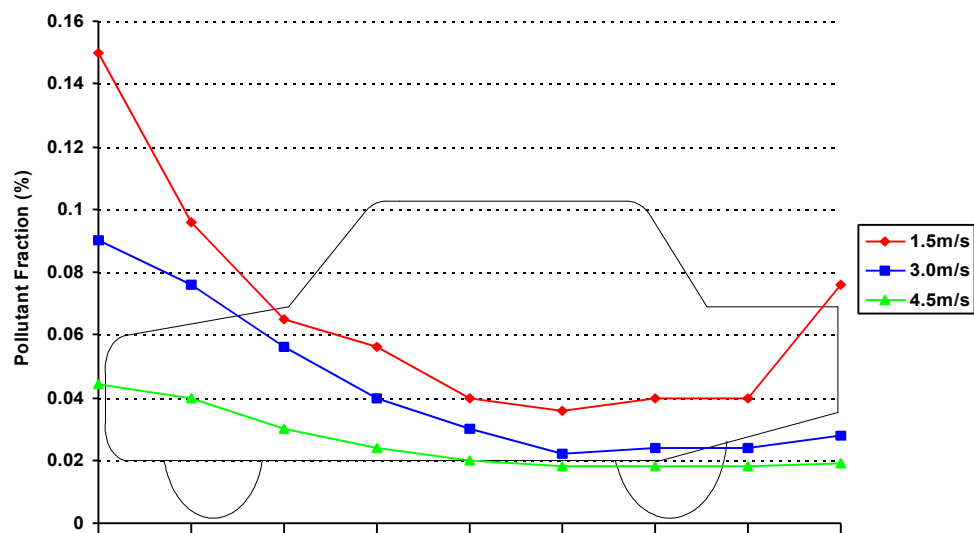


Figure 7.10 The fraction of N_2O (Car 2) along the centre line of Car 3.

Figure 7.11 shows the concentration of SF_6 emitted by Car 1 along the centre line of Car 3, again at three tunnel speeds. The concentrations are lower due to the increased distance from the source and the dissipative effect of the turbulent flow around Car 2. The curves again show a gradual fall in concentration along the length of the vehicle until the rear screen where the concentration rises markedly before falling off again. Again the trend is most pronounced at the lowest wind speed. The overall variation of the concentration is considerably

less than in the previous case indicating that the flow is relatively well mixed due to the turbulence encountered around Car 2.

The proportion of pollution received from Car 1 and Car 2 by Car 3 will now be considered. Figure 7.12 shows the fraction of each tracer gas received along the centre line of Car 3. The graphs are all very similar and show that the proportion of tracer received by Car 3 from Car 2 to that from Car 1 decreases from about 4:1 to 3:1 along the bonnet. The concentrations continue to converge over the roof, but diverge at the rear of the vehicle. The majority of the pollution experienced by Car 3 can be seen to be emitted by Car 2.

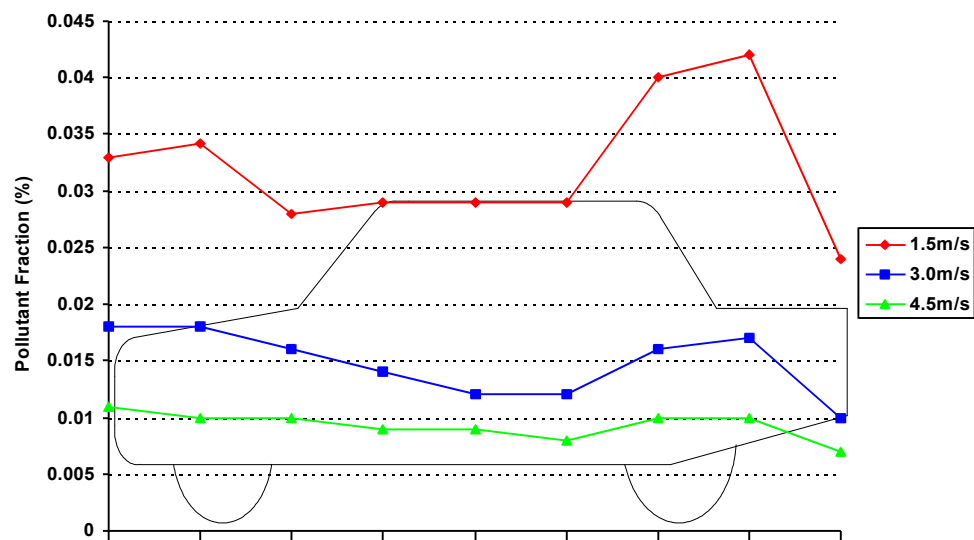


Figure 7.11 The fraction of SF_6 (Car 1) along the centre line of Car 3.

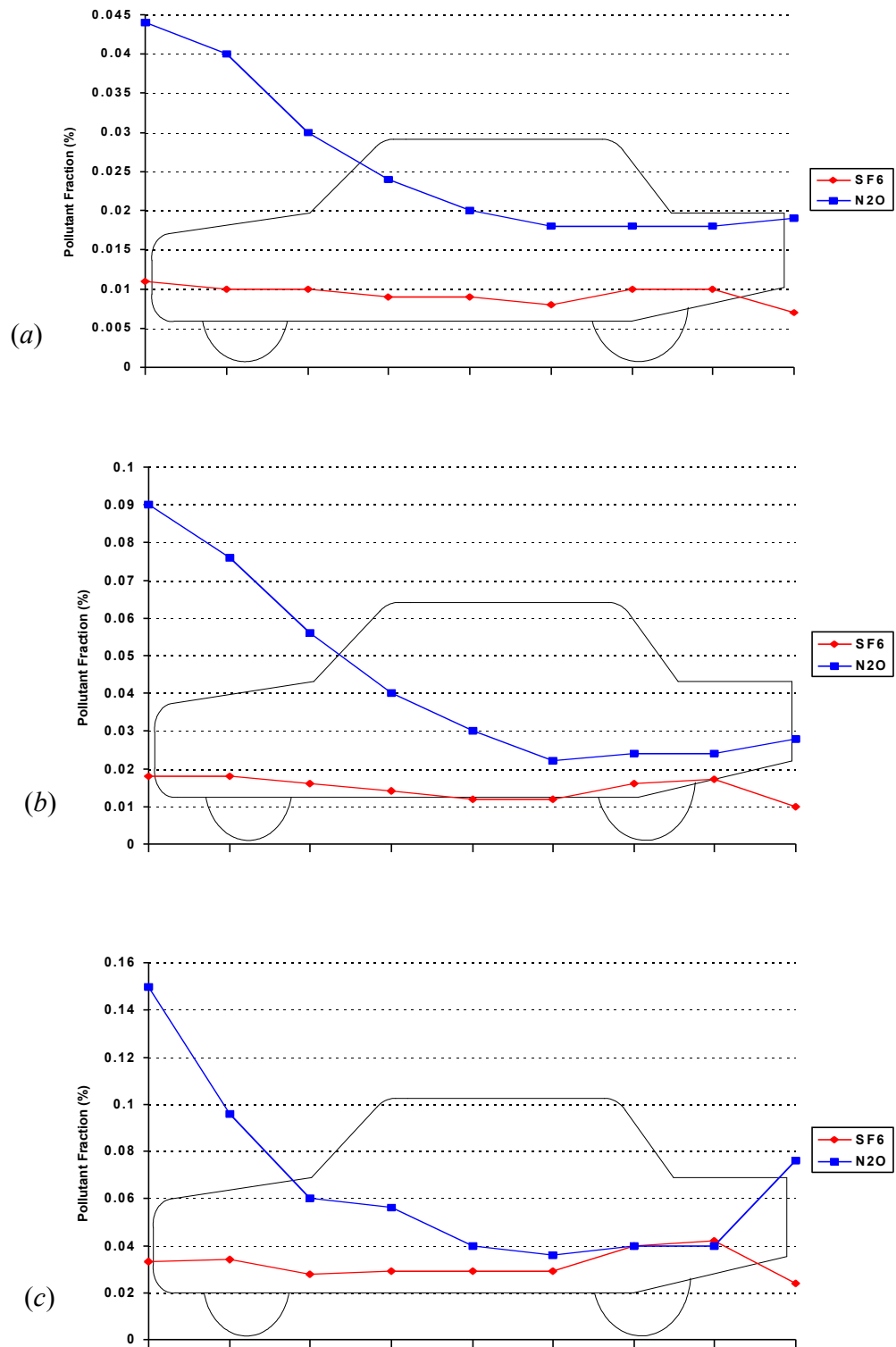


Figure 7.12 The fraction of SF₆ (Car 1) and N₂O (Car 2) along the centre line of Car 3 at (a) 4.5m/s, (b) 3.0m/s and (c) 1.5m/s.

The effect of wind tunnel speed can be seen in Figure 7.13. where the fraction of tracer gas present at the two tappings representing the bonnet inlet vents on

Car 3 are plotted against the wind tunnel speed. The results show that the concentrations of both pollutants tend to fall with increased wind tunnel speed. Proportionally the concentration of pollutant from Car 2 to that from Car 1 is about 2.5:1 and seems to be independent of the wind tunnel speed.

7.3.5 Conclusions Drawn From The Wind Tunnel Studies.

The studies conducted in the wind tunnel are representative of a slowly moving queue of traffic having a road speed in the order of 1.0km/h. Under such conditions, which may be experienced at a road junction in a busy town, 75% of the pollution experienced by a vehicle may come from the car immediately in front. The effect of crosswinds is uncertain as they are difficult to model accurately but at shallow angles of incidence they appear to increase the level of pollution flowing across sections of the vehicle body, primarily the bonnet.

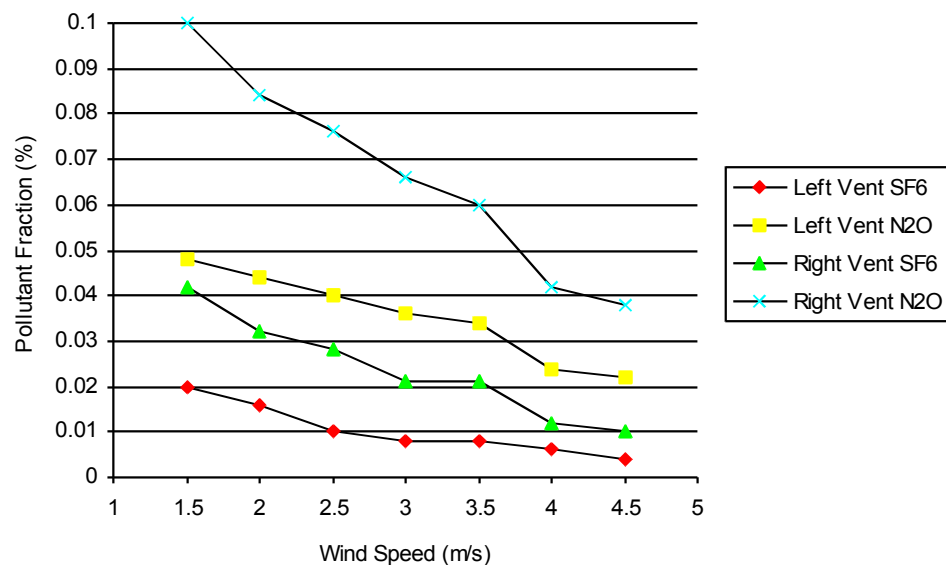


Figure 7.13 Fraction of tracer gas emitted from Car 1 (SF_6) and Car 2 (N_2O) measured at the bonnet vents of Car 3.

Chapter 8. Measurement of Pollution Within Vehicles

8.1 Introduction

Traffic pollution is generally quantified by sampling air adjacent to busy roads or by recording the background concentration of pollutants. It has been shown that localised pockets of pollution may have concentrations several times higher than such measurements would lead us to suppose, this phenomena is apparent if concentration measurements are taken across or at several locations adjacent to a busy road (Croxford *et al* 1995 and Menachem *et al* 1990). A vehicle travelling along such a road is surrounded by the exhaust emissions of other vehicles and it is therefore reasonable to assume that a proportion of those emissions will be drawn into the vehicle cabin by the ventilation and heating system.

In order to quantify the concentration of pollutant likely to be encountered within the vehicle under different driving conditions a ‘Rover Maestro’ light van was equipped with instrumentation to record pollutant concentrations, air temperature and vehicle speed. The vehicle was then used to collect pollution and traffic density data during a series of surveys on two standard routes across Nottingham. In this chapter the data obtained during the surveys are presented. The chapter also explains the method used to estimate traffic density and details of the instrumentation fitted into the test vehicle.

8.2 Estimation of Traffic Density.

Traffic flow, density and speed are the three variables which form the basis of traffic analysis. Traffic flow is the number of vehicles, q , passing a fixed point over a time interval, t , or:-

$$q = \frac{n}{t} \quad (8.1)$$

q is expressed as vehicles per unit time.

Density, k_t , is the number of vehicles occupying a given length of road and is expressed as vehicles per kilometre, speed, U , is the mean velocity of the traffic flow (Mannering and Kilareski 1990).

In order to visualise how these variables are related, consider an empty road. If a vehicle travels over this stretch of road its maximum speed will be uninhibited by other vehicles and will be governed by any speed limits that are in force, this is known as the free flow speed. Under these conditions the traffic density will be very low (close to 0), as more vehicles use the section of road, the traffic density will rise and the speed will start to fall due to cautious drivers and manoeuvring vehicles slowing the traffic stream. If the number of vehicles continues to increase eventually the road will become so congested that the traffic flow will come to a stop. In this condition the density is governed by the length of the vehicles and the distance between them, and is known as the jam density. Between these two extreme conditions the flow speed can be used to estimate the density using a simple model.

The model takes the speed / density relationship as a linear function:-

$$k_t = k_j \left(1 - \frac{U}{u_f} \right) \quad (8.2)$$

where:-

k_t = traffic density (veh/km)

k_j = jam density (veh/km)

U = mean flow speed (km/h)

u_f = free flow speed (km/h)

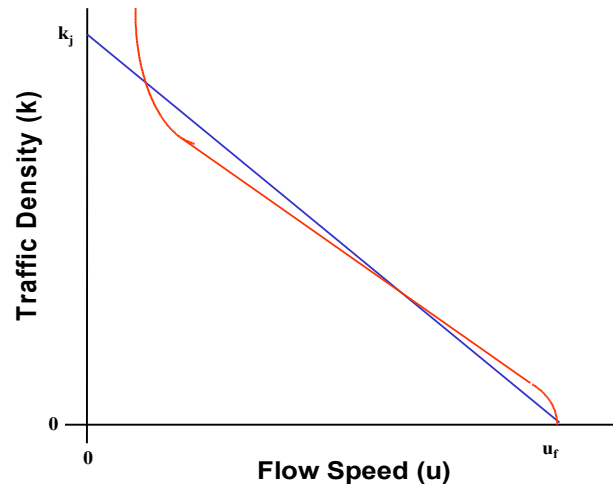


Figure 8.1 The relationship between traffic density and flow speed. The red line shows the nature of the deviation from the simple linear model.

This simple model is comparatively accurate providing the traffic flow does not approach either of the two extreme conditions, where the relationship tends to break down. This occurs because the traffic flow rarely becomes stationary as the traffic density equals the jam density, but continues to move slowly. At the opposite extreme the flow speed will not immediately begin to fall rapidly as the traffic density rises, but will initially fall away slowly (Figure 8.1). However the model is useful in that over a generally free flowing road, or route, the mean flow speed may be used to estimate traffic density.

8.3 Instrumentation.

The vehicle used for the field measurements (a Rover Maestro 600 diesel van) was fitted with instrumentation to enable measurements to be made as the vehicle was being driven. To accomplish this a small data logging system was used. Made by Grant Instruments of Cambridge UK., the 1200 series Squirrel logger had a 12 bit resolution and a maximum sampling frequency of 1Hz.

Carbon monoxide measurements were made using a CO sensor and a self contained pollution monitor called a Triple Plus both instruments being made by Crowcon Instruments of Abingdon UK (Figure 8.2). The CO sensor had a span of 0 to 250ppm and was fed with air drawn from the inlet to the

ventilation system by a small pump. Pollutants inside the van cabin were monitored using the Triple Plus. This instrument measured three common airborne pollutants CO, O₃ and NO_x at concentrations of 0 to 250ppm for CO, and 0 to 1000ppb for O₃ and NO_x. The sensitivity was too low for reliable measurements of NO_x to be made during the survey periods but concentration data was obtained for the other two pollutants. The gas sensors were calibrated at weekly intervals using span gases obtained from BOC Special Gases of Morden UK.

Road speed was recorded throughout the survey period using a specially designed optical sensor fitted onto the speedometer cable of the van (Figure 8.3 and Appendix A). Airflow into the vehicle cab through the heater inlet duct, was monitored using the airflow sensor described in Chapter 4 section 3.2.

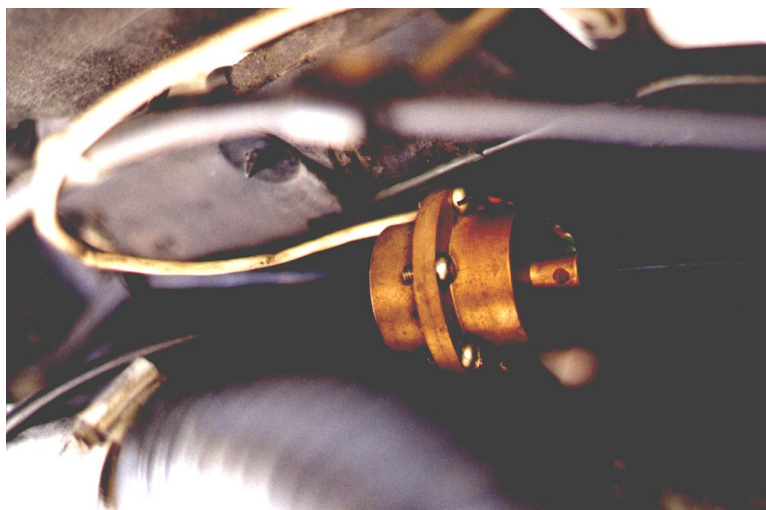


Figure 8.3 The road speed sensor installed in the van.

8.4 Experimental Procedure

The experimental data were gathered over three winter months (October, November and December 1995) and three summer months (June, July and August 1996). These periods were chosen as they are at the extreme points of the seasonal cycle and it was hoped that any seasonal pollution trends would manifest themselves as differences in the recorded pollution concentrations.

Data was to be recorded over two predetermined routes across the centre of Nottingham. Routes were chosen that would encompass a variety of road and traffic conditions. The morning route (Figure 8.4) encompassed one of the main arterial roads into the city (Carlton Hill, Carlton Road), Nottingham's inner ring road (Canal Street, Maid Marian Way and Upper Parliament Street) and a portion of the outer ring road (Aspley Lane to Derby Road) a total distance of 9.5 miles. Traffic conditions along the route vary from congested inner city traffic to free flowing traffic on the outer ring road. The tests conducted on the evening route (Figure 8.5) again encompassed free flowing ring road traffic (Dunkirk to Clifton Bridge), more congested traffic around the town centre (Queens Drive to London Road) and often very slow moving traffic on the main road out of the city towards Colwick (Daleside Road), a total distance of 11.5 miles. The average journey time on both routes was approximately 35 minutes.

In order to allow the equipment to stabilise the instrumentation was switched on five minutes before the beginning of each survey trip. The data logger would then be started and the van driven over the prescribed route. During a test run the van was driven in as normal a manner as possible with the heater and ventilation controls set to suit the conditions prevalent at the time. It was found however that the heater fan needed to be running to adequately ventilate the cab and so this was set to run on the lowest speed during the majority of the test runs.

8.5 Experimental Results

8.5.1 Winter Survey (October - December 1995)

During the autumn and winter months carbon monoxide (CO) was the only pollutant that could be reliably measured. A typical set of recorded CO data is

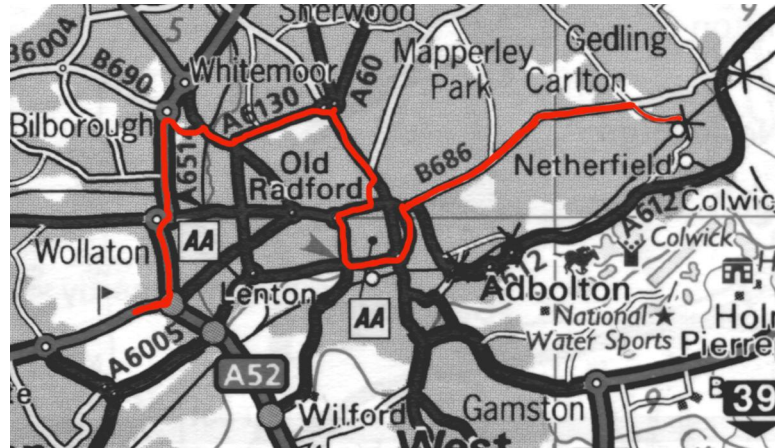


Figure 8.4 The morning route (Carlton to University Park).

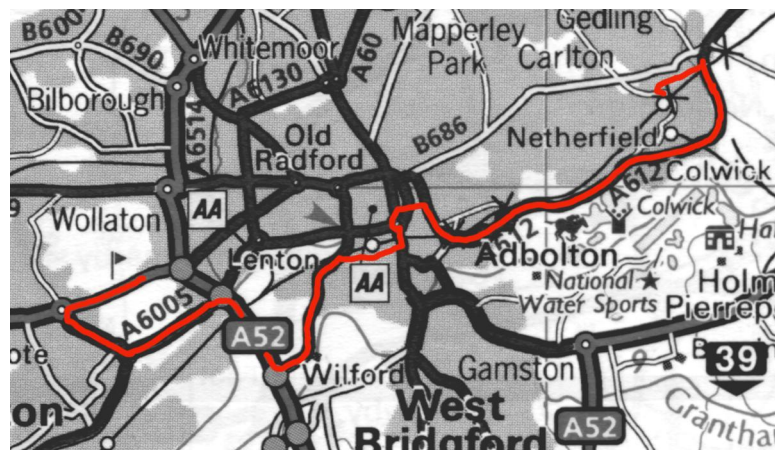


Figure 8.5 The evening route (University Park to Carlton).

shown in Figure 8.6. and shows the concentration of CO in both the air entering the vehicle through the heater and the air inside the vehicle cabin.

It is interesting to compare the mean concentration of the internal and external data sets. The internal readings are related to the external readings by the mass

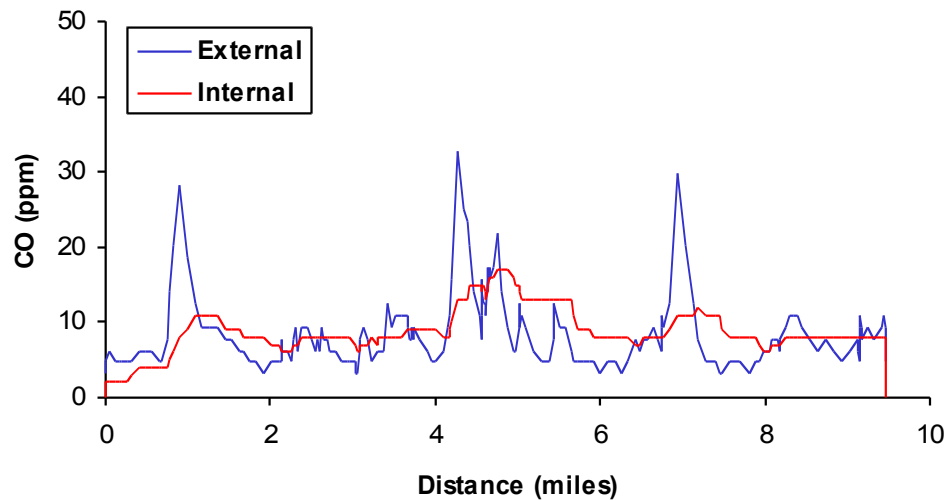
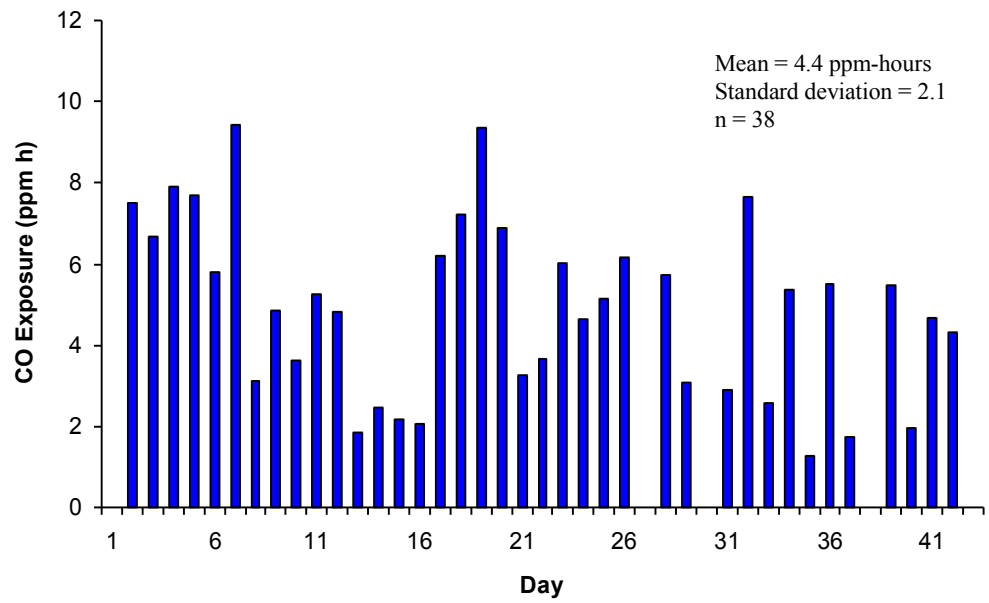
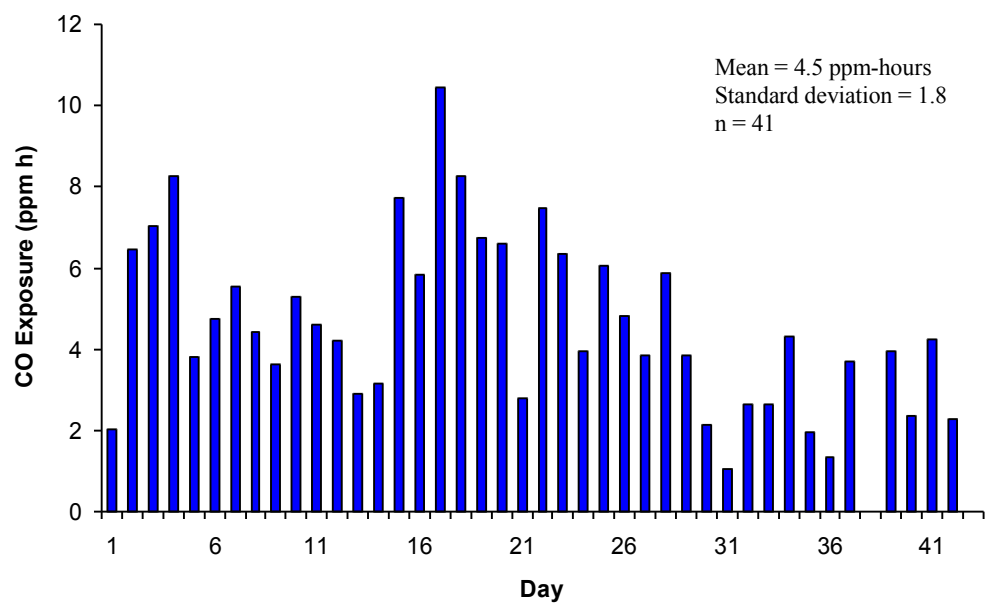


Figure 8.6 A typical set of CO concentration data.

balance equations often used to predict indoor air quality (Ott and Willits 1981); suffice it to say that the air inside the vehicle acts as a buffer and damps the peaks in the external concentration. Over a suitably long time scale the average values for internal and external concentration will be the same, but the time constant of the vehicle can vary considerably with differing ventilation conditions. For instance in a moving vehicle with a window open the air in the cabin may be exchanged every 10s, but if the vehicle is in slow moving traffic with the windows and vents closed it may take up to 10min for the air to be replaced. As the journeys reported here took between 29 and 47 minutes the buffer effect cannot be ignored and in this case proved to be beneficial, as the van was parked away from heavy traffic between the test runs. The air inside the cabin was virtually free from CO at the start of each journey and during the winter months this led to the mean internal concentration being 7.9ppm (morning) and 8.0ppm (evening), compared with average external levels of 11.1ppm (morning) and 11.3ppm (evening).



(a)



(b)

Figure 8.7 Mean integrated CO exposure for the morning (a) and evening (b) runs during the winter survey.

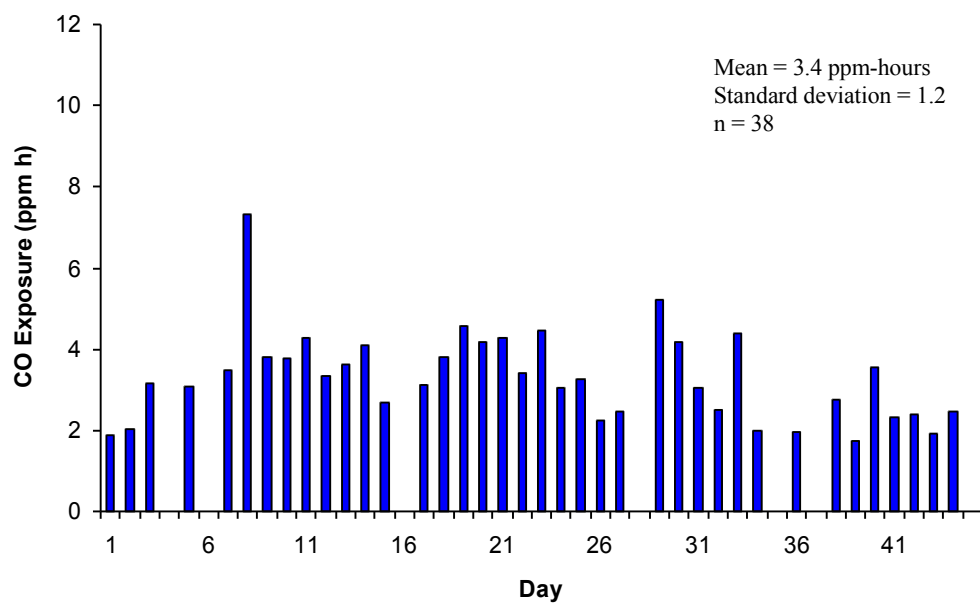
The average daily internal CO concentrations have been multiplied by the journey time to give time integrals of CO exposure for each journey. The results are shown in Figure 8.7. Over the sampling period the mean time integral exposure was 4.4 and 4.5 ppm-hours for morning and evening journeys respectively.

8.5.2 Summer (June - August 1996)

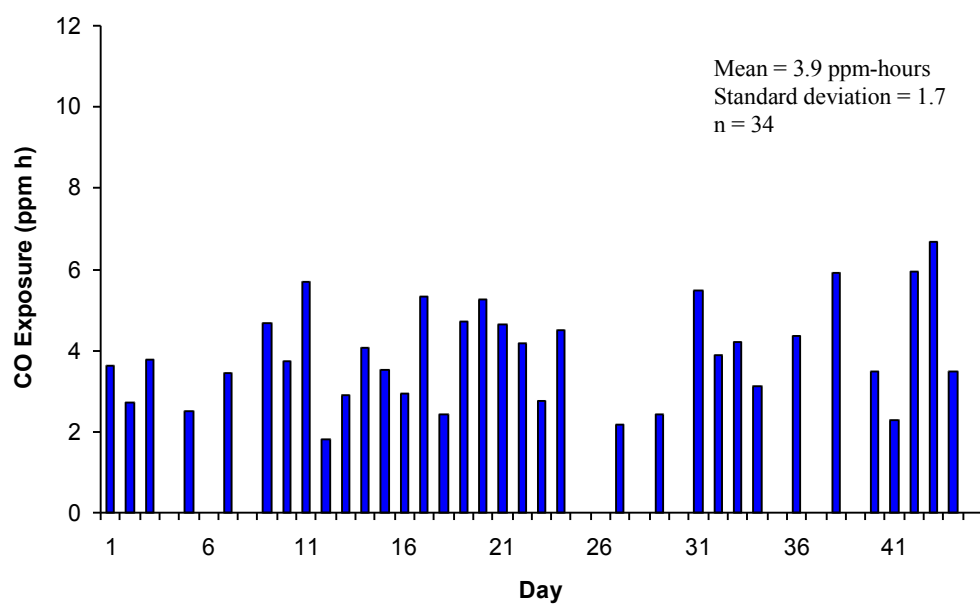
During the summer months in addition to the CO measurements data was obtained which showed the ozone concentration of the air circulating within the vehicle cabin. The concentration data recorded for both pollutants will be discussed in turn.

8.5.2.1 Carbon Monoxide

Over the summer months the background level of carbon monoxide present in the atmosphere falls, this is partially due to more efficient combustion taking place in motor vehicle engines. External concentrations of CO measured from the test vehicle, over the summer months had a mean value of 9.4ppm (morning) and 11.5ppm (evening) and may be compared to the winter mean concentrations of 11.1ppm (morning) and 11.3ppm (evening). Although the morning runs show a reduction of the external CO concentration there is no significant variation in the mean evening concentrations. The internal concentration of CO however does show a significant reduction during both the morning and evening test runs. The mean concentration of CO inside the vehicle was 6.3ppm (morning) and 7.39ppm (evening) these may be compared to the mean winter readings of 7.9ppm (morning) and 8.0ppm (evening). This difference becomes more apparent when the time integrated exposures (Section 8.5.1) are calculated (Figures 8.8) and which have mean values of 3.4 ppm-hours (morning) and 3.9 ppm-hours (evening), compared with those calculated for the winter months which have mean values of 4.4 ppm-hours (morning) and 4.5 ppm-hours (evening). This reduction is probably attributable to the use of the side windows for ventilation, in addition to the vehicle heater.



(a)



(b)

Figure 8.8 Mean integrated CO exposure for the morning (a) and evening (b) runs during the summer survey.

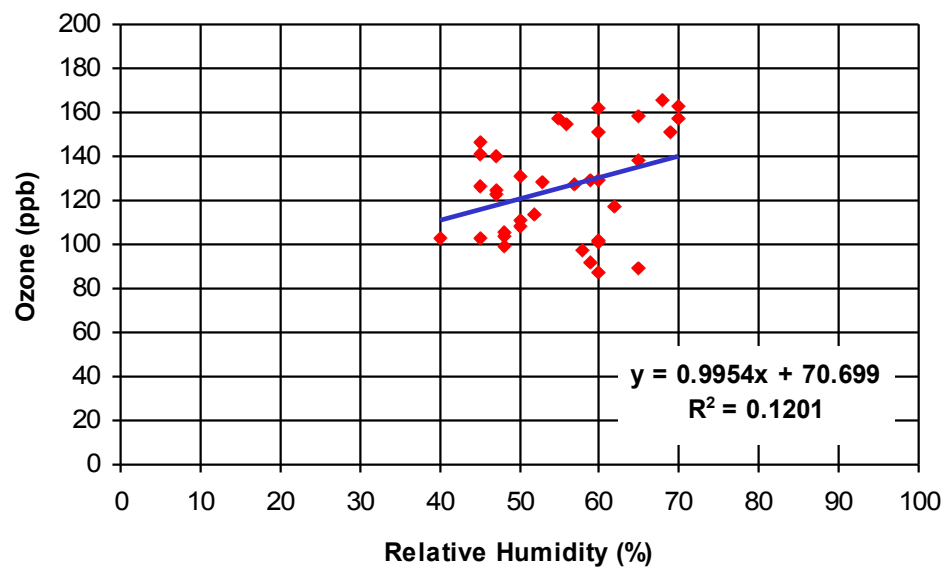
8.5.2.2 Ozone

During the winter runs there was no measurable ozone present in the vehicle cabin. During the summer however ozone concentrations rise due to photochemical reactions in the atmosphere and measurements were possible. The data showed that the highest concentrations of ozone during the morning runs occurred when the relative humidity (%RH) of the air was high. During the test period the humidity was monitored and varied between 40%RH and 70%RH (Figure 8.9). Although there is a considerable scatter a distinct trend can be seen, with ozone concentrations tending to be up to 50% greater at 70%RH than at 40%RH. During the afternoon the relative humidity tends to fall and no significant trend is apparent.

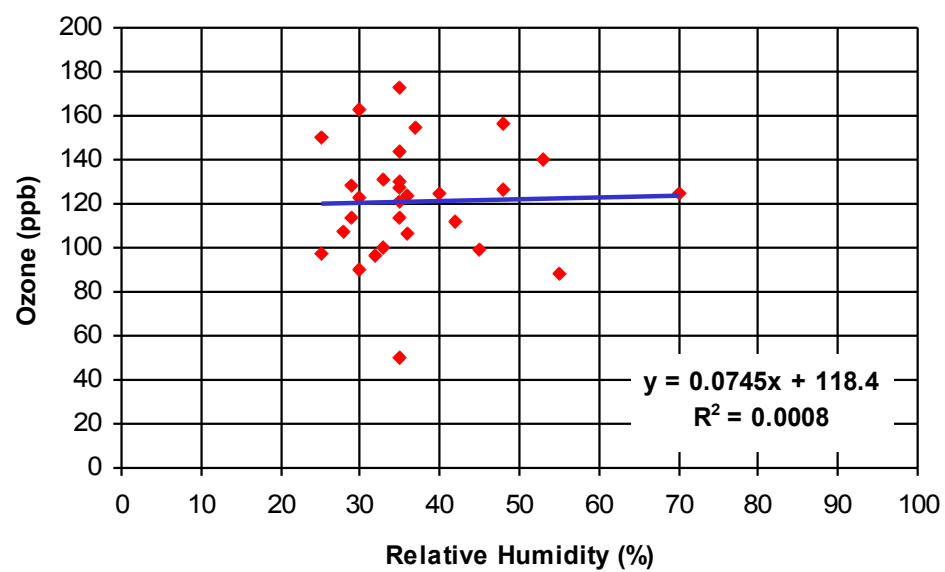
A typical set of ozone concentration data is shown in Figure 8.10. The most striking feature of all the data sets is their lack of sharp peaks and troughs. This is due to the absence of spatial variations of the external ozone concentration, the concentration of ozone in the vehicle is therefore almost equal to the external concentration. Time integrated exposures have again been calculated for the test runs and have mean values of 64.8 ppb-hours (morning) and 62.2 ppb-hours (evening) (Figure 8.11).

8.6 Discussion

The experimental data clearly shows that over the two sampling periods the mean internal concentration of CO, taken over a journey, varied between 3 and 12ppm. There was considerable day-to-day variation but the daily levels and integrated time exposures were within the EPQAS standard of 10ppm taken as an eight hour average. However the effect of emissions from individual cars or at busy intersections can cause increases in the internal CO concentration and increase the integrated exposure. In general the levels of CO recorded were lower than in previous studies in the UK. (Brice and Roesler, 1966; Colwill and Hickman, 1980) and this may be due to reductions in the amount of CO



(a)



(b)

Figure 8.9 Mean concentrations of ozone recorded during the morning (a) and evening (b) runs against relative humidity.

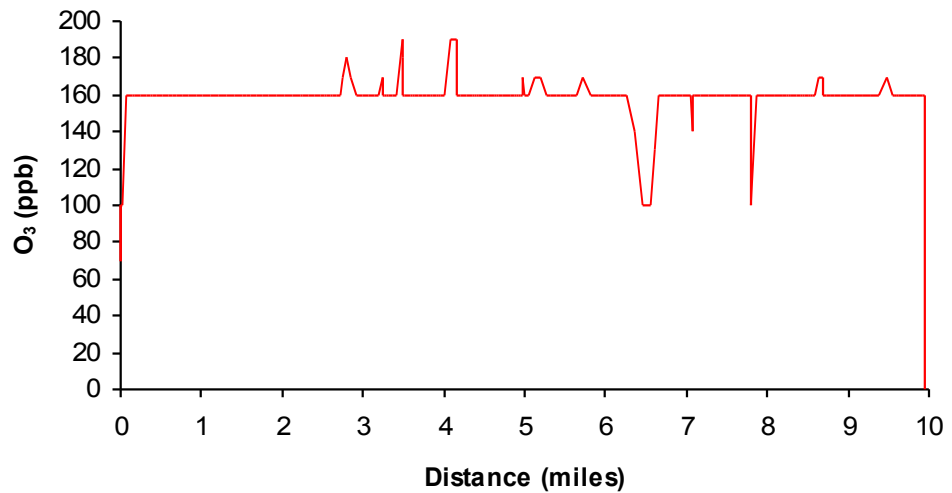
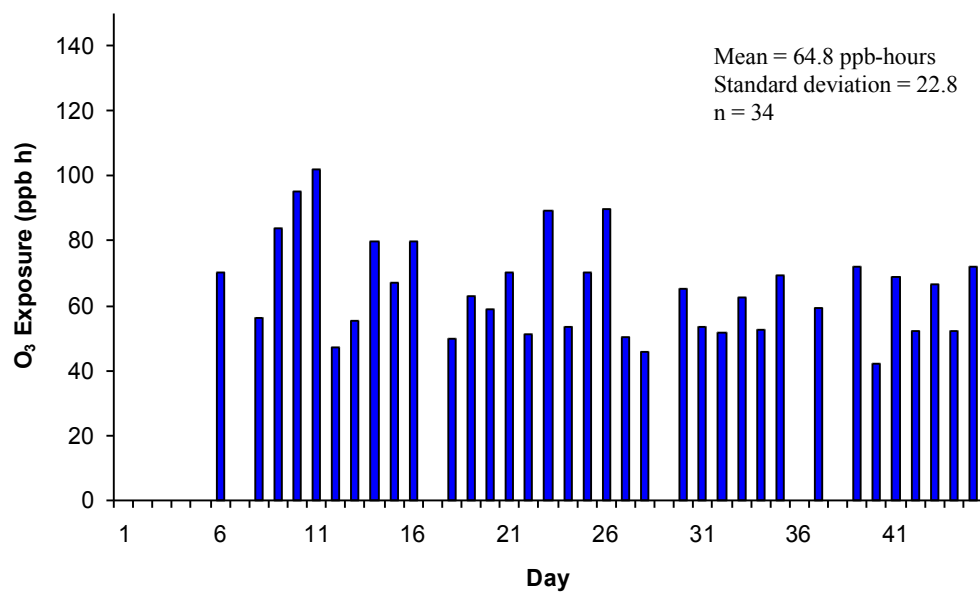


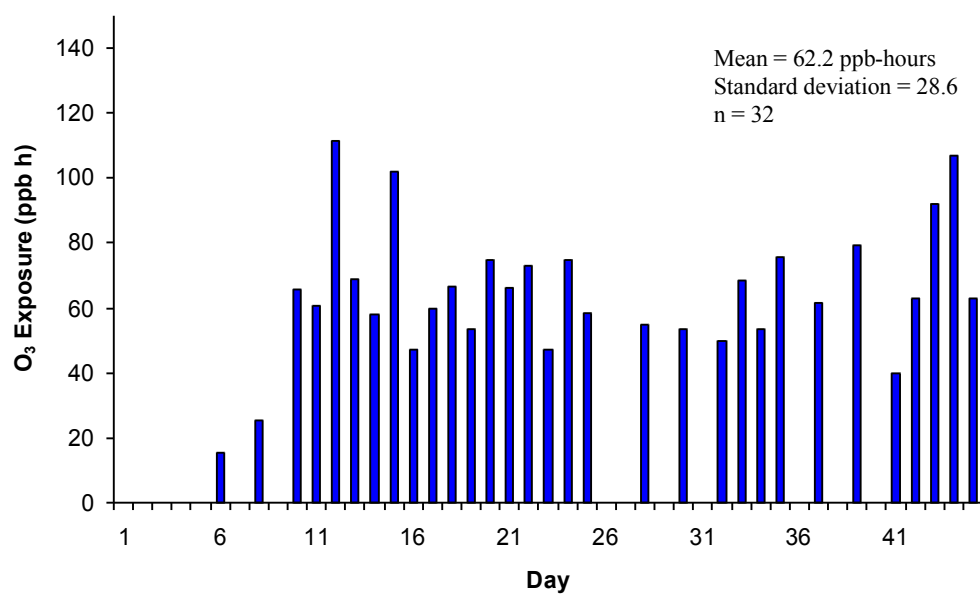
Figure 8.10 A typical set of ozone concentration data.

produced per kilometre travelled. Meteorological conditions seem to have had little effect on the level of CO recorded, higher wind speeds tended to reduce the recorded concentration due to more rapid dispersion but the effect was minimal, precipitation appeared to have no effect.

The density of traffic was estimated during both test periods and over both test routes by calculating the average journey speed. On the morning route the overall average speed 16mph and on the evening route the overall average was 20mph. Heavier traffic tends to lower the average speed due to increased queuing time and lower overall road speeds and this is an accepted method of estimating the traffic density. The effect of traffic density is shown in Figure 8.12, the mean internal CO concentration has been plotted against the average journey speed and a negative trend can be seen; this is most pronounced in the data for the morning runs. The degree of scatter makes day-to-day prediction of exposure impossible and reinforces the argument that the concentration of CO is strongly influenced by the surrounding vehicles.

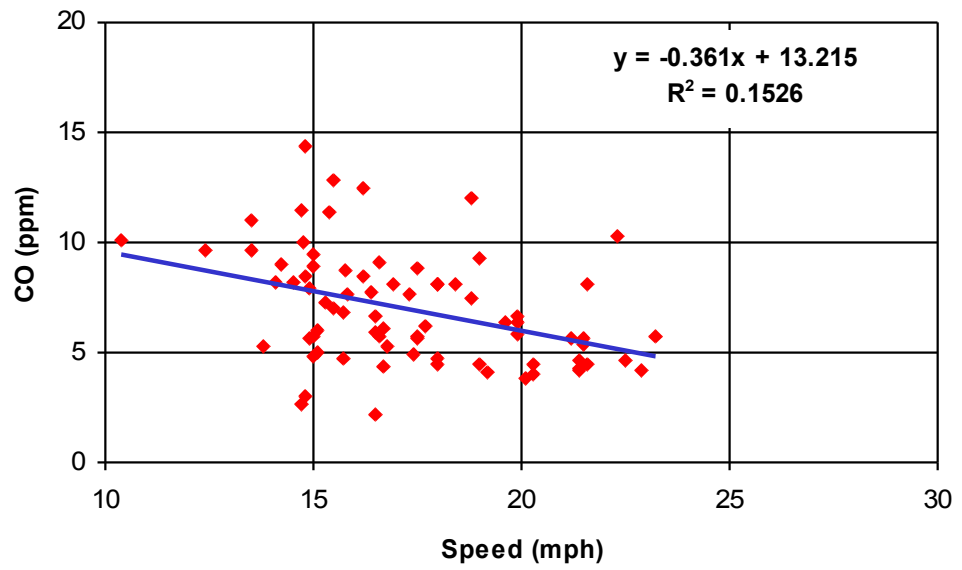


(a)

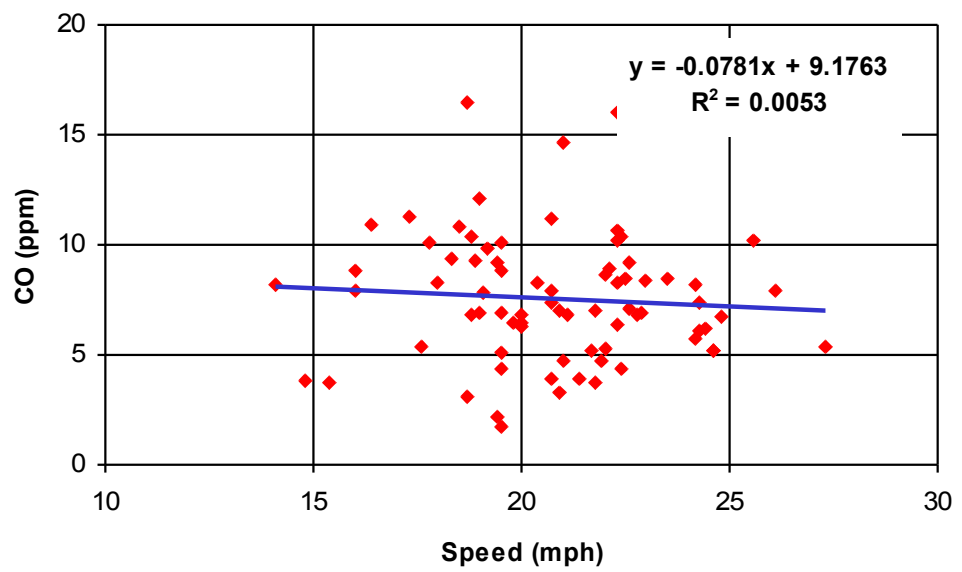


(b)

Figure 8.11 Mean integrated ozone exposure for the morning (a) and evening (b) runs during the summer survey.



(a)



(b)

Figure 8.12 Mean internal CO plotted against average speed for the morning (a) and evening (b) runs of both survey periods.

The measurements of ozone recorded during the summer months are possibly of greater concern with mean concentrations of 123ppb being recorded in the vehicle cabin over the summer period. As there is little apparent spatial variation in the concentration of ozone and vehicle occupants probably experience similar concentrations as pedestrians and other road users. There is evidence that concentrations of 120ppb are high enough to cause increased sensitivity to inhaled allergens such as pollen and increased sensitivity to other gaseous pollutants (Section 3.5.8). In addition drivers of vehicles such as taxis and buses may be receiving a daily exposure in excess of the EPAQS standard of 50ppb expressed as a running eight hour average.

Chapter 9. Conclusions and Recommendations for Further Work

9.1 Conclusions

This work is concerned with the concentrations of gaseous traffic pollutants being experienced by road vehicle occupants whilst travelling in heavy traffic. The literature search showed that although there was a substantial amount of research undertaken during the 1960's and 70's, most of this was conducted along the western seaboard of the United States. As both the type of automotive fuel and the atmospheric conditions differ in the UK. from those in the US. it is reasonable to assume the level of pollutants experienced by road users will also differ. The experimental work carried out allows the following conclusions to be drawn:

- 1) Ventilation tests on an instrumented research vehicle driven on the public highway showed that with the windows closed the ventilation of the vehicle was largely dependent on the road speed due to Ram Effect. Air exchange rates varied between 2/hour when the vehicle was stationary to 4/min at fifty mph. The ventilation fan was only effective at low road speeds when it boosted the air exchange rate to around six / hour whilst the vehicle is either stationary or moving slowly. The effect of the fan was found to be minimal at higher road speeds.
- 2) When the vehicle was driven with a window opened then almost half of the air was exchanged by single sided ventilation mechanism. Experimentation in the wind tunnel and using the research vehicle showed that ventilation through the window could be simply modelled using the open window area. The geometry of the opening appearing to have little or no effect on the ventilation rate and confirmed Crommelin *et al.* (1988).
- 3) Wind tunnel modelling showed that the flow of air over a typical vehicle body traps pollutant at positions where the air inlets for the ventilation

system are often located. Air inlets situated in the front grille draw in exhaust from the vehicle immediately in front, and vents below the windscreen can draw in polluted air trapped there by the flow regime over the bonnet. Model tests have also shown that the majority of pollutants present inside a car originate from the car immediately in front, with the other surrounding vehicles contributing to the overall background concentrations.

- 4) Field measurements using the test vehicle to drive around set routes in an urban environment, showed a large spatial variation in the concentration of carbon monoxide. Peaks in the concentration measurements tended to occur at road junctions and in slow moving traffic queues. Individual, heavily polluting vehicles were also seen to produce large peaks in the concentration measurements, a consequence of such vehicles being to raise the concentration of pollutant inside a following vehicle very rapidly.
- 5) The field measurements also showed that the mean concentration of CO experienced by vehicle occupants was very much higher than kerbside measurements would suggest. Dispersion of CO was found to be comparatively rapid with even high concentrations falling to near the background level over a distance of only a few metres.
- 6) Ozone measurements did not show the large spatial variation experienced with CO and therefore the concentration experienced by vehicle occupants will be very close to that of pedestrians and other road users. The concentration of ozone in the vehicle cabin (a mean concentration of 123ppb) was high enough to cause reductions in lung function and sensitivity to other pollutants.
- 7) The experimental data reinforces the findings of other researchers, who have concluded that car commuters experience higher levels of traffic pollution

than commuters travelling by any other means (Chan *et al* 1991a, Flachsbart *et al* 1987).

9.2 Recommendations for Further Work

The exposure of vehicle drivers to traffic pollution has serious consequences for those professional drivers who routinely cover many miles in heavy traffic. Their total exposure to pollutants such as CO may be in excess of the guidelines set out by organisations such as the WHO. More work needs to be conducted to evaluate the total exposure to traffic pollutants employees, such as taxi and bus drivers, experience in the course of their work.

The wind tunnel experiments suggest that relocating the air inlets to the vehicle ventilation system may give reductions in the concentrations of pollutants experienced by the occupants. Both this and the application of filters to remove pollutant gases from the air being drawn in, need further developmental research.

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Appendix 1. Publications Arising From the Research

Riffat S.B. Shao L. Clarke R. Green N. Movement of traffic pollution around and inside vehicles and its control. *Proc 1st APTE 13th - 15th May 1998, Singapore* (to appear).

Clifford MJ, Everitt PJ, Clarke R, Riffat S.B. Using computational fluid dynamics as a design tool for naturally ventilated buildings. *Building and Environment*. 1997, **32**, (4), pp.305-312.

Clifford MJ. Clarke R. Riffat S.B. Drivers' exposure to carbon monoxide in Nottingham, UK. *Atmospheric Environment*. 1997, **31**, (7), pp.1003-1009.

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Clifford M.J. Clarke. R Riffat. S.B. Reducing drivers' exposure to carbon monoxide. *Proc 29th ISATA 3rd - 6th June 1996, Florence, Italy*.

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