

**MODELLING THE IMPACTS OF TRANSPORT POLICIES  
ON THE URBAN ENVIRONMENT**

by

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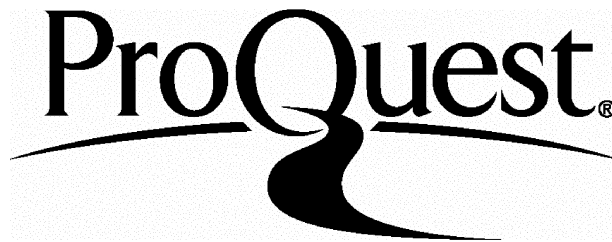
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## ABSTRACT

Road transport is by far the major source of environmental degradation in urban centres. Transport trends, planning policies and traffic management schemes can have significant impacts not only on local but also on global environmental conditions. This research analyses the extent to which a range of transport strategies can influence the total vehicle exhaust emissions and local levels of air pollution concentration and noise from traffic in an urban area.

The transport policies and trends considered may stimulate changes in the characteristics of traffic flows, which affect the patterns of traffic emissions, or alter the exhaust levels directly from emission sources. The changes produced in traffic emissions can have important impacts on environmental quality. The local and global environmental impacts from the various scenarios generated by the policies or trends analysed are compared with the environmental quality from the 'do nothing' situation.

The analysis is based on the application of a road traffic assignment model in conjunction with models for the estimation of environmental degradation, to the urban road network of Chester, England. The assignment model simulates the conditions of traffic flow generated by the demand for travel, in each link within the network. The outputs are utilised by an emission model, which estimates the total emissions produced by all traffic within the road network. The traffic data are also used as inputs to the air pollution dispersion and noise propagation models, in order to estimate the environmental conditions at the point in each road where the levels of degradation are expected to be maximum.

The last part of this research presents an assessment procedure, which addresses some further implications produced by the simulations carried out, such as the predominant economic and environmental costs. This procedure involves an evaluation of the costs associated with travel time, fuel consumption and environmental degradation.

The approach in this research can provide decision-makers with valuable information about the economic and environmental implications from changes in the characteristics of the transport system.

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# **1 INTRODUCTION**

## **1.1 Background**

The concern with environmental issues is not recent. The first smoke abatement law was set in 1306, during the reign of Edward I of England, and in 1307 a violator of this law was executed (Bach, 1972). Attempts to improve the air quality of urban air were made by the Smoke Nuisance Abatement Act of 1853 (Green, 1994), whereas the world's first comprehensive clean air act was passed in Britain in 1863. Such an early initiative demonstrated that industrial pollution could be reduced as a result of legislation. The Clean Air Acts of 1956 and 1968 had wide public support and over the following decade pollution from domestic and industrial sources was drastically reduced (Green, 1994). A number of measures for controlling environmental degradation has been imposed by the U.K. Government over the years (U.K. Government, 1990).

The necessity of maintaining the quality of the urban environment is becoming more and more widely recognised today, as environmental matters acquire growing public concern in all segments of modern society. This has been changing attitudes and behaviour towards environmental matters, and is likely to continue influencing political decisions.

The environmental problems produced by road transport are dominant amongst the factors affecting the quality of neighbourhoods and living conditions, and they also contribute to global environmental effects. Road transport has become by far the major and fastest growing source of environmental degradation in most cities. Air pollution and noise nuisance are the predominant detrimental impacts from road traffic. As a matter of fact, the current guidelines stipulating maximum acceptable levels of noise and pollutant concentration are exceeded from time to time, particularly at roadside sites in urban areas (Royal Commission on Environmental Pollution, 1994).

Despite the strict control and regulation of fuel composition and vehicle exhausts, transport still generates much more pollutant emissions than any other single human activity. The contribution of road vehicles in the U.K. has reached 90% of the total carbon monoxide emitted nationwide, about half of all volatile organic compounds, nitrogen oxides and black smoke, and over one fifth of the total carbon dioxide exhausts, the primary greenhouse gas

(Department of Transport, 1994a). Traffic is also the dominant source of noise levels, particularly in residential areas, where noise nuisance is considered a major cause of annoyance from traffic (European Conference of Ministers of Transport, 1990; TEST, 1991; Klaboe, 1992 and Royal Commission on Environmental Pollution, 1994). Thus, it seems clear that any strategy aiming to reduce the environmental impacts in urban developments should concentrate on managing road transport.

Air pollution causes various adverse effects such as damage to buildings, health problems, nuisance and deterioration on the living environment. The negative effects caused by traffic are more strongly perceived in urban centres, where the population density is high and the elevated levels of traffic flows interact closely with human activities. These effects are even more manifest in the peak periods, when the desire for mobility conflicts with the capacity of the urban system to accommodate the demand for transport. As a consequence of traffic saturation, flows often become interrupted and congested, and travel time and vehicle emissions tend to increase throughout the network.

As a result of the expansion of the economy in the U.K. over the past years and the improvement of general standards of living, there has been a substantial increase in the demand for transport of people and goods. Rising personal incomes have led to corresponding increase in private vehicle ownership and a decline in the use of public transport. According to the results of the most recent National Travel Survey, the growth in travel volume in Britain has involved both the number and length of journeys undertaken (Department of Transport, 1993). Car ownership has also increased considerably in the last decade. It has risen by nearly 27% between 1983 and 1993, and it is estimated to increase further from about 15% (low forecast) to 23% (high forecast) between 1995 and 2005. Car travel plays the dominant effect in traffic growth, and this has also been a consistent trend. Car traffic in terms of vehicle kilometres has risen by about 45% from 1983 to 1993 and is expected to further increase by about 19% (low forecast) to 31% (high forecast) between 1995 and 2005 (Department of Transport, 1994a). These figures indicate that car traffic has been growing fast. Such trends together with a consequent expansion in transport infrastructure have led to increasing pressures on the environment. Thus, the already massive contribution of traffic to the environmental problem will become even more critical, in view of the threat of such continuous growth in the demand for transport in the future. More cars will be running on streets and additional congestion and environmental problems will be created.

The impacts of transport on the environment have become key policy issues of the 1990s. For example, the Royal Commission on Environmental Pollution (1994) states that:

"there is widespread concern that continuing growth of transport will be damaging to the environment, to health and to the efficient functioning of the economy" (p xiii).

These concerns have been recognised by the U.K. Government (1994) in its sustainable development strategy:

"the impact of ever rising levels of transport on the environment is one of the most significant challenges for sustainable development (and) if people continue to exercise their choices as they are at present, the resulting traffic growth would have unacceptable consequences" (p 169).

In recent years, governments and industries have been looking into ways of controlling transport emissions. Technological improvements have occurred in several areas of the automobile industry. Vehicle manufacturers have been spending a great deal of time and resources in developing more sophisticated products. Cars are becoming faster, safer, more reliable, more comfortable, more economic and efficient in terms of fuel consumption and more environment-friendly. In fact, decreases in vehicle exhaust and noise emissions have been substantial in the last decade or so. Wachs (1993) suggests that the emission of some pollutants has reduced by 98% in new American cars when compared with new cars twenty years ago, and that nowadays 80% of the traffic air pollution is produced by only 10% of the vehicles, those which are very old or badly out of tune. This change is similar in the U.K. (Goodwin, 1994; Royal Commission on Environmental Pollution, 1994) and is to a large extent a result of the pressures of governmental and environmental organisations on vehicle manufacturers.

Before new cars and heavy vehicles can be marketed in the countries of the European Union, they must be tested and certified to conform with regulated limits on exhaust pollutants emitted. The limits for the emission of pollutants and noise are becoming more and more strict. Since January 1993 all new passenger cars have been compulsorily fitted with catalytic converters, the efficiency level of which is quite remarkable. Future regulations may require further technological changes in vehicle manufacturing. Emission checks have been

introduced into the annual MoT test for cars over three years old, and smoke emission tests have been included in the annual inspection of heavy duty vehicles. These regulations force vehicle owners to keep their engines in tune. The composition of fuel has also been regulated by the Government. Unleaded petrol is now widely available in Britain and its use is encouraged by a lower rate of taxation.

These initiatives indicate that within a few years the contribution of transport to total emissions should decrease substantially, particularly when the old cars have been replaced by new less polluting ones. Further technological progress is expected in the coming decade, which will continue to improve engine efficiency and effectiveness with additional environmental benefits.

New standards have recently been put forward by the Government-appointed Expert Panel on Air Quality Standards, which recommends maximum acceptable levels of pollutant concentrations in the atmosphere (Local Transport Today, 1994). Monitoring programs have been implemented and new sites are still being set up in many parts of the country. These will provide politicians and the community with continuous assessments of the local environmental quality and evidence of whether the air quality standards are being accomplished or not.

The relevance of the impacts of transport on the environment has been widely recognised by citizens, planners and scientists. Hence, there is a vast potential need for academic research and technological development in the field of transport and the environment.

## **1.2 Objectives**

Transport trends, planning policies and traffic management schemes can have significant impacts on local and global environmental conditions. The objective of this research is to analyse the extent to which a range of transport strategies can influence the total vehicle exhaust emissions and local levels of air pollution concentration and noise from traffic in an urban area. A further evaluation is made of the predominant economic and environmental costs incurred from the simulation of each strategy.

The ultimate objective of this research is to produce some scientific evidence about the detrimental impacts caused by vehicle pollution on society. This information can be used

responsibly to furnish decision-makers with information about the likely economic and environmental consequences from changes in the transport system.

### **1.3 Structure of thesis**

This thesis is divided into 13 chapters. Chapter 1 presents the introduction and objectives of this work, and stresses the importance of the subject on social, policy and academic contexts.

Chapter 2 looks at the range of environmental impacts brought about by the provision of road traffic and defines the scope of the study in terms of the impacts analysed.

Chapter 3 explores a variety of aspects about air pollution from traffic, such as the main traffic pollutants produced, the emission process, the factors influencing emissions, the dispersion of pollutants into the atmosphere, the methods for estimating air pollution concentrations and the current standards for air quality.

Chapter 4 discusses the most relevant points about traffic noise levels, including vehicle noise emission control, the factors influencing noise emissions, the methods for the estimation of traffic noise and the environmental standards for noise nuisance.

Chapter 5 provides the basis for the monetary valuation of the environmental impacts of traffic. This will be useful for the incorporation of the environmental dimension into the evaluation of the overall implications originated from changes in the transport system.

Chapter 6 describes the process for the modelling of road traffic air pollution and noise. Initially, it presents the modelling framework and introduces the grounds for road traffic assignment and modal share. Then, the air pollution emission and dispersion and noise propagation models are addressed in detail. Further, this chapter reports the model application to the study network.

Chapter 7 presents the modelling procedure for the assessment of the economic and environmental impacts incurred upon the transport system, as a result of each of the changes under consideration.

Chapters 8 and 9 examine the changes in environmental quality from the implementation of

a range of transport policies and the realisation of transport trends in the future. Indicators of the economic and environmental changes in monetary terms are also provided at the end of each section.

Chapter 10 reports the summary and principal conclusions from this research. Chapter 11 presents some recommendations for future academic research and for political initiatives in the field of the environment. The bibliographic references used throughout this work are listed in Chapter 12.

## **2 TRANSPORT AND THE ENVIRONMENT**

In the past, the term '*environment*' was understood simply as '*nature*', and consisted of the basic natural resources such as water, air, land, space and raw materials. However, the modern concept of '*environment*' has been widened in the last few years. Environmental conditions are not only determined by the long-term availability in sufficient quantity and of adequate quality of those natural resources, but also includes the totality of social, physical and built-up factors related to the quality of life of human beings and suitable habitats for animals and plants. The natural and cultural heritage have also been included in the notion of the environment (European Community, 1992c).

### **2.1 The environmental impacts produced by road traffic**

The provision of road transport produces a range of externalities at local and global levels, some of which are considered desirable or 'positive' and others undesirable or 'negative'. The main positive impact brought about by transport is an improvement in the quality of life, by enhancing accessibility. Transport is considered essential in modern societies, a means through which people are enabled to carry out most activities. Besides, transport facilitates the supply and distribution of goods and services, and proportionates a wider range of choices for both producers and consumers. It plays a major role in the economic life of industrialised countries and in the daily life of their citizens (European Conference of Ministers of Transport, 1990). Thus, transport is vital to both economic and social well being.

However, diverse negative impacts are also created by the provision of road transport facilities, sometimes affecting adversely the very same community who benefits from their utilisation. These impacts can be of different kinds (they may provoke economic effects, social consequences and physical nuisances), have distinct durations (intermittent, continuous or permanent) and affect areas in different scales (local, regional or global). Negative impacts arise from the process of maintenance and construction of infrastructure and vehicles, their physical presence, and most importantly, from traffic movements. The impacts of road transport activities include accidents (deaths, injuries and property damage), congestion, social segregation, local and global air pollution, noise nuisance, vibration, visual intrusion, water pollution (mainly through acid rain), land take and consumption of energy and other natural resources.



The environmental externalities produced by road traffic are disparate in nature. The various impacts have been identified in distinct ways in the literature available. Much contradiction and confusion has been made with the classification of the environmental impacts and their effects on people, animals, vegetation and materials, at local and global levels. The Manual of Environmental Appraisal (Department of Transport, 1983a) classifies the various groups affected by the environmental impacts of transport into travellers (drivers, passengers, cyclists and pedestrians), occupiers of property (residential, industrial, commercial, schools and hospitals) and users of facilities (shopping centres, public buildings and recreational areas). The Design Manual for Roads and Bridges (Department of Transport et al., 1993) actually replaces the Manual of Environmental Appraisal (Department of Transport, 1983a). However, the 1993 manual classifies the range of environmental impacts less explicitly than the 1983 report and it concentrates on providing techniques for the assessment of the environmental impacts on: air quality, cultural heritage, disruption due to construction, ecology and nature conservation, landscape effects, land use, traffic noise and vibration, pedestrians and cyclists, vehicle travellers, water quality and drainage, geology and soils and policy and plans.

The extent to which street traffic annoys and disturbs residents in terms of their perception of noise, air pollution, stress, risk and disruption of social interaction has been extensively verified by Appleyard and Lintell (1972). This study has had important influence on much succeeding research work in the field of transport and the environment.

A proper identification of the negative environmental impacts is the primary concern in environmental assessment studies, which have become compulsory for a number of projects that require planning permission. It also constitutes the basis for planning policies which pursue the very topical concept of sustainable development (more details about this concept are given in Chapter 8).

A comprehensive, yet not conclusive, classification of the main negative environmental impacts generated by road traffic and their consequences has been produced based on the author's perception of the cause-effect relationships. This classification is illustrated in Table 2.1.

**Table 2.1: The consequences from the environmental impacts produced by road traffic**

| Consequences                                   | Impacts       |                   |                  |             |
|--|---------------|-------------------|------------------|-------------|
|  | air pollution | noise & vibration | visual intrusion | segregation |
| human health problems                          | ***           |                   |                  |             |
| disturbance, discomfort                        | ***           | ***               | *                | **          |
| unpleasing surrounding                         | ***           | ***               | ***              | ***         |
| intimidation and accident risks                | *             | *                 | **               | ***         |
| changes in use, value and occupation of land   | **            | **                | **               | ***         |
| traffic congestion and delays                  |               |                   |                  | ***         |
| damage to plants and animals                   | ***           |                   |                  |             |
| damage to building facades and other materials | ***           | **                |                  |             |
| global warming (greenhouse effect)             | ***           |                   |                  |             |
| restriction of accessibility                   |               |                   |                  | ***         |

\*\*\* = substantial  
 \*\* = medium  
 \* = low

Each impact may produce a range of consequences on diverse groups in the community, and in different levels of importance. It is not simple to consider all the environmental implications brought about by the provision of road transport, and hence environmental studies are unlikely to provide full understanding of the environmental problem: first, because the primary environmental impacts of traffic may produce a wide variety of effects; secondly, a precise identification of the most relevant effects may prove complex; finally, some effects provide difficulty for quantification and measurement, whereas others can be assessed only on an essentially subjective basis.

The impacts particularly relevant in this work are those which are considered the main cause of dissatisfaction to communities and which generate most damage to the living environment

in urban areas. This study restricts the analysis of the environmental problem within two of the most relevant and immediate traffic-induced impacts: air pollution and noise. A number of studies have been undertaken specifically on such subjects and these impacts can be quantified, measured, predicted and related to environmental standards with relative ease. Ultimately, these impacts enable a picture of the environmental degradation in an area in quantitative terms.

Chapters 3 and 4 describe in more detail a variety of issues related to air pollution and noise originated from road traffic. These review chapters will demonstrate essential grounds for the subsequent developments of this work.

## **2.2 The importance of methods for environmental appraisal**

There has been a lack of progress on methods which further environmental considerations. The existing methods for environmental appraisal are still at a very early stage of both scientific development and public acceptance (Banister, 1990). In the field of road transport, these methods have achieved little progress and have only been applied to almost exclusively local or isolated situations (Ridley, 1992). They are only required to accompany the construction of roads or the implementation of major projects and are not really appropriate for application in actual urban situations, where traffic has gradually become a problem. In the past decades, research in this field was concentrated within various aspects of the provision of road transport, such as reduction of cost, time and accidents, while the environmental impacts generated used to be overlooked or accepted as the price the society had to pay for mobility. In fact, most actions in road transport schemes have been taken disregarding the environmental aspects and practical solutions for the mitigation of traffic-induced impacts have had limited success. Perhaps the principal reason for planners overlooking environmental issues is that driving is considered a basic individual right in modern society and tends to be the ultimate choice for intervention measures. There has been a considerable policy-level resistance to imposing restrictions on travellers for the sake of preserving the environment.

However, the outstanding and ever increasing contribution of transport to the environmental problem has not only focused public attention on all sorts of degradation arising from road traffic, but has also attracted increasing interest in the environmental dimension within transport planning. The emphasis in traffic management and control has turned towards

adapting road systems and their use to the aim of improving the balance between accessibility and environment (Allsop, 1990). Some effort has been made in the development of appropriate methods and techniques for furthering the consideration of the adverse environmental effects of road traffic into transport planning policies, alongside traditional criteria for planning appraisal. Several planning policy documents have been issued which emphasise the need for the environmental impacts of traffic to be considered at an early stage in transport plans (e.g. Department of the Environment and Department of Transport, 1992 and 1994). Some planning policies have already been pursued with the explicit objective of controlling traffic emissions in order to protect the living environment (e.g. Royal Commission on Environmental Pollution, 1994).

### 3 AIR POLLUTION FROM TRAFFIC

A pollutant is defined as a chemical or physical element present in the composition of the air at higher levels than those at which it is generally found. Such undesirable elements can alter the original characteristics of the atmosphere and become unpleasant or harmful to humans, animals, goods and natural resources. Air pollution can be assimilated or eliminated by nature up to a certain limit, through the processes of dispersion, deposition and absorption (mainly by plants). However, because of the great amount of polluting sources in urban centres, purification by natural process is often not complete. Air pollution builds up and concentrations may reach levels which exceed those considered acceptable.

The effects of air pollution on health are associated with persistent exposure to the degraded air, particularly in areas where pollutant concentrations have built up to dangerous levels. The magnitude of the effects will depend on the pollutant concentration level in the atmosphere, length of exposure, health condition and vulnerability of the individual towards pollution, age and activity performed by the time of inhaling. The resulting effect of a combination of distinct pollutants acting simultaneously may be more harmful than the addition of the isolated effects of each pollutant. Several studies have been carried out in order to establish the effects of air pollution on human health (a number of studies as such are reported in more detail in Sections 5.2.1 and 7.2.2). However, due to the complexity on the assessment of such effects and to the wide variety of interfering factors, often the results are only tentative and can be used merely as indicators of the likely damage produced. Further studies are necessary to provide more definite answers to the health effects of both the prolonged exposure to low levels of air pollution and occasional high level incidents.

Apart from effects on human health, air pollution can disturb, annoy, cause stress, discomfort and frustration both to drivers and pedestrians, contribute towards the formation of smog and reduction of visibility on roads (which increases the risk of accidents), provoke changes on the use, value and occupation of properties, cause damage to buildings and virtually any exposed material, and contribute to the greenhouse effect and acid rain. Some pollutants have localised effects on the environment, whilst other gases can accumulate in the atmosphere and produce impacts far away from the source. This division, however, is rather arbitrary, since some gases on their own or in reaction with others may contribute to both local and global air pollution.

### **3.1 The emission of pollutants**

Emissions from road vehicles arise mainly from the crankcase, the fuel system and the exhaust. The crankcase emissions from gasoline vehicles are formed when some of the burning gases in the engine cylinders escape into the crankcase (space underneath the engine pistons), during the compression and power strokes of the cylinders. In new vehicles this is eliminated by closing the crankcase vent to the air and recirculating the blow-by gases to the intake manifold. Crankcase emissions from diesel engines are minimal. Emissions from the fuel system consist mostly of evaporation caused by the gradual heating of the fuel tank during warm days and by spillage of vapours from the tank during refuelling. Hot soak emissions are caused by the rapid evaporation of the fuel left in the carburettor while the engine is still hot after having been turned off. Exhaust emissions from road traffic contain the products of combustion of fossil fuel. In an ideal combustion engine, fuel would be completely burned for the production of energy. In practice, however, this is never the case and most gaseous and particulate pollutants arise because, first, the internal combustion process is inevitably inefficient and, second, fuel may contain various impurities which are mostly expelled with the exhausts.

Emissions are key factors in air pollution modelling. They represent the rate at which an amount of pollutant (expressed in grams) is emitted by unit of time (minute), distance (kilometre) or fuel consumed (litre). Concentration levels in the atmosphere can be expressed as the mass of pollutant in a given volume of air, normally measured in microgrammes per cubic metre ( $\mu\text{g}/\text{m}^3$ ) or milligrams per cubic metre ( $\text{mg}/\text{m}^3$ ), or as the ratio of the volume of the pollutant to the volume of the air in which the pollutant is contained, normally measured in parts per million (ppm) or parts per billion (ppb). The relationships between  $\text{g}/\text{m}^3$  and ppm for various pollutants are given in Table 6.3 (Section 6.2.1).

### **3.2 The main pollutants from road traffic**

The primary traffic air pollutants analysed in this work are carbon monoxide (CO), hydrocarbons (HC), nitrogen oxides ( $\text{NO}_x$ ), particulate material (PM) and carbon dioxide ( $\text{CO}_2$ ). The first four pollutants have proved to be harmful to human health and have been subject of interest in most works respecting air pollution. PM is also the main pollutant emitted by diesel engines, responsible for the soiling of buildings, odour and unpleasant environments.  $\text{CO}_2$  is the primary greenhouse gas and is emitted in large quantity by road

vehicles. The following sections describe in detail each of the five pollutants under consideration in this work.

### **3.2.1 Carbon Monoxide (CO)**

#### *Description*

CO is an odourless, tasteless and colourless gas that is slightly lighter than the air and highly toxic. It results from the incomplete burning of materials which contain carbon, in particular fossil originated motor fuel. CO is an intermediate product through which all types of carbon must pass when combusted in oxygen. Its formation depends on the amount of available oxygen for combustion. CO is one of the most common and widely distributed air pollutant. It has been largely used in air pollution modelling as one of the main indicators of atmospheric contamination (Hickman and Colwill, 1982; Linaritakis, 1988; Matzoros, 1988).

#### *Occurrence in the atmosphere*

Natural background levels of CO fall in the range of 0.01-0.23 mg/m<sup>3</sup> (Whitelegg, 1993). CO levels in urban areas are highly variable, depending upon weather conditions and traffic density.

#### *Atmospheric behaviour*

CO is a relatively stable compound, presents localised dispersion behaviour and takes part slowly in atmospheric chemical reactions. CO participates in the formation of ozone and also contributes indirectly to global warming, not only by being further oxidised in the atmosphere to carbon dioxide, the major greenhouse gas, but also by slowing the destruction of methane, which is another greenhouse gas.

#### *Emission sources*

CO is considered to be the prime transport pollutant. The transport sources in the U.K. constitute the largest contribution of CO, most of which comes from petrol cars without catalytic converters (Department of Transport, 1994a).

The major source of atmospheric CO is the spark ignition combustion engine. More CO is emitted when there is a deficiency of air in the combustion chamber of an engine, or in other words, a low air/fuel ratio. In a diesel engine there is always excess air and CO emissions are relatively low. CO emission is high when the vehicle is cold and choked, and it is proportional to the engine operation effort and acceleration.

Smaller contributions come from all other processes involving the combustion of organic matter, such as power stations, industry and waste incineration. Tobacco smoking can be a significant source of CO in enclosed environments.

### *Human health effects*

Because CO is toxic, it can have very dangerous effects on human health. Its short term effects are relatively well known. CO can cause drowsiness, general discomfort, decreased physical performance and loss of working productivity. It can also induce angina and impair perception, thinking, concentration, neuro-behavioural function and reflexes<sup>1</sup>, even at low doses. CO increases the likelihood of exercise-related heart pain in people with coronary heart disease and may present risks to the foetus in pregnant women. Repeated episodes of impaired oxygen supply would be expected to damage the brain and possibly cause structural damage resulting in the reduced ability of the central nervous system to transmit information. High concentrations of CO can cause unconsciousness and convulsing, strokes, brain swelling and protrusions, and even death (According to Howard (1990), in confined areas such as enclosed garages and in tunnels with poor ventilation, only 0.3% by volume of CO is necessary to cause death within 30 minutes).

As CO is a relatively insoluble compound, it easily enters the lungs along with oxygen, diffusing through the walls of the alveoli thus entering the blood stream. Upon entering the blood stream CO combines with the haemoglobin in the same way as oxygen, but at a rate that is about 210 times more intensive. The carboxy-haemoglobin (COHb) formed reduces the blood capacity of conveying and supplying oxygen to the body tissues, which can give rise to a wide range of health effects. Table 3.1 present the percentage of COHb in the blood as a function of CO concentrations and exposure period at different activity levels.

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<sup>1</sup> There is suggestive but not conclusive evidence that drivers in fatal road accidents often have elevated levels of carboxy-haemoglobin (Whitelegg, 1993).



**Table 3.1: Percentage of carboxy-haemoglobin (COHb) in the blood**

| CO<br>(ppm) <sup>a</sup> | Exposure time (hour) |                      |         |                      |         |                      |
|--------------------------|----------------------|----------------------|---------|----------------------|---------|----------------------|
|                          | 1                    |                      | 4       |                      | 8       |                      |
|                          | resting              | moderate<br>exercise | resting | moderate<br>exercise | resting | moderate<br>exercise |
| 5.0                      | 0.6                  | 0.6                  | 0.8     | 0.8                  | 0.9     | 0.9                  |
| 9.0                      | 0.7                  | 0.8                  | 1.2     | 1.3                  | 1.4     | 1.5                  |
| 15.0                     | 1.0                  | 1.1                  | 1.8     | 2.0                  | 2.2     | 2.4                  |
| 20.0                     | 1.1                  | 1.4                  | 2.3     | 2.6                  | 2.9     | 3.1                  |
| 25.0                     | 1.3                  | 1.6                  | 2.8     | 3.2                  | 3.6     | 3.8                  |
| 35.0                     | 1.6                  | 2.1                  | 3.8     | 4.5                  | 4.9     | 5.3                  |
| 50.0                     | 2.2                  | 2.9                  | 5.2     | 6.3                  | 7.0     | 7.6                  |

a) 1 ppm of CO  $\approx$  1.2 mg/m<sup>3</sup>

Source: Environmental Protection Agency (1979).

Table 3.2 illustrates the effects of CO in typical individuals, as a function of the percentage of COHb in the blood.

**Table 3.2: Reported effects of CO exposure according to COHb levels**

| COHb (%)           | Effects  |
|--------------------|--|
| 0.4                | Normal physiologic value for non-smokers   |
| 2.5-3 <sup>a</sup> | Decreased exercise performance in patients susceptible to illnesses  |
| 4-5                | Increased oxygen debt in non-smokers and increased symptoms in traffic policemen   |
| 5-10               | Diminution of visual perception, manual dexterity or ability to learn;<br>Changes in myocardial metabolism and possible impairment |
| >10                | Headache and impairment manual coordination; Changes in visual evoked response   |

a) Maximum level recommended by the World Health Organisation (1987).

Source: Ferris (1978).

The figures from the tables above show that the level required to produce adverse effects on health is much higher than those normally found in the atmosphere (Watkins, 1991). Cigarette smoke has been found to produce higher exposure to CO than that experienced by pedestrians

exposed to exhaust emissions around urban streets (Department of Transport, 1983a). However, CO concentrations can build up to dangerous levels on heavily trafficked and congested urban roads, in particular situations of very poor dispersion conditions.

### **3.2.2 Hydrocarbons (HC)**

#### *Description*

HC belong to a larger group of chemicals known as volatile organic compounds. They are a mixture of approximately a hundred compounds of carbon and hydrogen, from which the hydrocarbons themselves, alcohols, aldehydes, organic acids, ketones, aromatic compounds and olefins are conspicuous (Holman et al. 1991).

#### *Emission sources*

In transport, they are formed through fuel evaporation and by unburnt or partially burnt fuel. The production of HC is particularly elevated when there is an excess of fuel in combustion. About 55% of HC emissions come from exhaust gases, 25% from 'blowby' (when unreacted gas leaks into the atmosphere) and the remaining 20% from evaporation. Bell (1990) found that HC from vehicles can make up to one third of all emissions in the atmosphere. The HC exhausts from transport represent 46% of the total emissions of this pollutant in the U.K. (Department of Transport, 1994a).

#### *Human health effects*

At the concentrations usually found in urban air, most HC are not directly harmful to health. In polluted areas, saturated HC can cause discomfort, drowsiness, eye irritation, coughing and sneezing. Benzene is carcinogenic and can cause leukaemia, chromosome anomalies and the suppression of blood cell formation in humans.

#### *Other effects*

HC are particularly harmful when reacting with NO<sub>x</sub> in presence of sun light, producing photochemical smog, which contributes to the visibility reduction on roads and damage to plants. They contribute indirectly to acid rain and global warming via surface ozone creation

(ozone is a powerful oxidizing agent and greenhouse gas). Methane contributes directly and non-methane hydrocarbons indirectly to the greenhouse effect.

### **3.2.3 Nitrogen Oxides (NO<sub>x</sub>)**

#### *Description*

NO<sub>x</sub> are inorganic gases, normally odourless, tasteless and colourless. NO<sub>x</sub> are a collection of compounds containing a combination of atmospheric nitrogen and oxygen. They are produced at high temperature and pressure, like the conditions found in the internal combustion chamber of an engine. NO is the main oxide of nitrogen formed, but much of it oxidises rapidly when emitted from the exhaust and then forms NO<sub>2</sub>. NO<sub>2</sub> is a reddish-brown gas, strong oxidant and soluble in water. It is considered the most important of the NO<sub>x</sub> from the point of view of human health and it has also become one of the most significant urban air pollutants in the U.K. Ambient concentrations and guidelines for accepted levels are generally expressed on its terms. They have detrimental health effects upon humans, vegetation and the global environment, especially when in combination with other pollutants.

#### *Emission sources*

In the U.K., transport represents the largest single source of NO<sub>x</sub>, accounting for 52% of the total emissions (Department of Transport, 1994a). All NO<sub>x</sub> emissions from traffic comes from vehicle exhausts and they may reach as much as 75% of the total emissions in some areas (Bell, 1990). The amount produced of NO<sub>x</sub> depends mostly on the proportion of air in the burned air-fuel mixture. The production of these pollutants is highest from diesel engines, particularly heavy goods vehicles. However, light petrol vehicles also emit them, above all when running in high speeds (Joumard, 1989), contrary to the emission profile of other pollutants.

#### *Human health effects*

The NO<sub>x</sub> may irritate the respiratory tract, cause sore throats and in high enough concentration can alter lung function and destroy lung tissue. The NO<sub>x</sub> also combine with the haemoglobin of the blood, being potentially toxic. The presence of NO<sub>x</sub> in the air may raise the human susceptibility to infections and contaminations by virus and bacteria. NO<sub>2</sub> may

exacerbate asthma and damage the immune defences of the respiratory system. The NO<sub>x</sub> can increase the incidence and seriousness of bronchitis and pneumonia and increase sensitivity to dust and pollen in asthmatics. Laboratory results on the effects of NO<sub>2</sub> exposure are given by the Parliament Office of Science and Technology (1994).

#### *Other effects*

In combination with SO<sub>2</sub> they become an important contributor to the acid rain. They also contribute to the smog formation when reacting with HC. Some of the products of reactions involving NO<sub>x</sub> are powerful greenhouse gases.

### **3.2.4 Carbon Dioxide (CO<sub>2</sub>)**

#### *Description*

CO<sub>2</sub> is a colourless and odourless gas, which is the product of complete combustion of fuel within the engine. The mass emitted is proportional to the amount of fuel consumed. As it is a stable compound, this pollutant may contribute to environmental problems on a global scale and exert an influence for many years (Department of Transport et al., 1993). CO<sub>2</sub> is a natural constituent of the air and elevated concentrations are very common in urban areas.

#### *Emission sources*

The emissions of CO<sub>2</sub> are rising rapidly. Although transport is responsible for only about a fifth of the total emissions of CO<sub>2</sub>, it is now the pollutant emitted in the largest amount in absolute terms. This is the case not only from road transport (over 121 million tonnes in 1992, compared with only 6 tonnes of CO, the second largest emitted), but also from all emission sources (566.6 million tonnes in 1992, compared with 6.7 tonnes of CO) (Department of Transport, 1994a).

#### *Other effects*

CO<sub>2</sub> concentrations are not directly harmful to health. It is not toxic and in the past it has not been considered as a major problem. However, the presence of CO<sub>2</sub> is threatening by causing alterations on the global heat balance of the earth's atmosphere through the increased

retention of thermal radiation. CO<sub>2</sub> is the largest single contributor to the greenhouse effect (Holman et al. 1991). CO<sub>2</sub> persists in the air long after it is produced and the accumulated present levels in the outer atmosphere are now higher than at any time in the past (Godlee and Walker, 1992). The catastrophic effects of major climatic changes have become crucial in current debates on the environmental impacts of transport.

### ***3.2.5 Particulate Matter (PM)***

#### *Description*

Airborne PM consist mainly of carbon and organic compounds derived from the incomplete combustion of fossil fuels and lubricating oil, particularly as a result of poor maintenance, when an excessive amount of fuel is burnt. PM are a mixture of fine solid fragments of soot and liquid particles such as dust and smoke. They can be emitted directly from sources or formed within the atmosphere either from condensation of vapours or a result of chemical reaction processes (Quality of Urban Air Review Group, 1993a). Other particulate matter include dust from brake linings, rubber from the abrasion of tyres to road surface and re-suspended roadside dust and dirt. Particles can remain air-borne for over a week and they are the most evident indicator of the presence of smog.

#### *Emission sources*

Road transport alone is responsible for the emission of 42% of the total black smoke in the U.K. (Department of Transport, 1994a), and about 94% of the emissions in London (U.K. Government, 1994). Over 85% of the diverse PM emissions from transport are produced by diesel engined vehicles, which emit approximately one hundred times more particles per kilometre than petrol engines (Quality of Urban Air Review Group, 1993b). PM is largely emitted from other sources including power plants, domestic coal burning, industrial processes and incineration.

#### *Human health effects*

Relatively little is known about the precise effects of PM on health. They irritate the respiratory system and may exacerbate morbidity and mortality from cardiovascular and respiratory dysfunctions. The presence of PM in the atmosphere slows the pulmonary

clearance mechanism, allowing toxic agents to remain in effective contact with susceptible tissues. They can be inhaled and deposited in the deepest regions of the lungs, increasing the incidence of lung cancer. They are probably carcinogenic (Friends of the Earth, 1989). Fine particulate matter may be toxic in itself or may carry or absorb up to 10,000 harmful or toxic chemicals into its relatively large surface area (Howard, 1990; TEST, 1991). Hospital admissions and death have been correlated with particulate levels. PM10 (the particulates smaller than 10 micrometres across) have emerged as by far the most worrying of the cocktail of pollutants pumped out by road traffic, as well as the most difficult to control (Bown, 1994). Recent research suggests that PM10 are killing 10,000 people a year in England and Wales (Bown, 1994), and in two American towns their presence has been associated with a 16-17% increase in death rate from all causes for every 100  $\mu\text{g}/\text{m}^3$  concentration rise. Death rates from heart and lung disease have been found up to 37% higher in American cities with higher levels of fine particulates (Walters, 1994).

#### *Other effects*

As far as aesthetics and public nuisance are concerned, PM are one of the most obvious forms of pollution, because of their unpleasant odour and unappealing smoky colour and dirt. PM in the air are particularly unpleasant at local level, have the potential to damage plants, cause hazy skies, reducing or impairing visibility conditions and soiling and spoiling the appearance of buildings and other exposed surfaces in the vicinity of traffic. PM can potentially contribute to the modification of climate and acid deposition.

#### **3.2.6 Secondary pollutants**

Certain conditions can favour the occurrence of photochemical reactions amongst some of the pollutants described above, deriving new compounds. These compounds are called secondary pollutants and their effects are different and in some cases more severe than those of the original pollutants. Ozone, an example of secondary compound, is sub-product of chemical reactions between nitrogen oxides and hydrocarbons, in the presence of sun light.

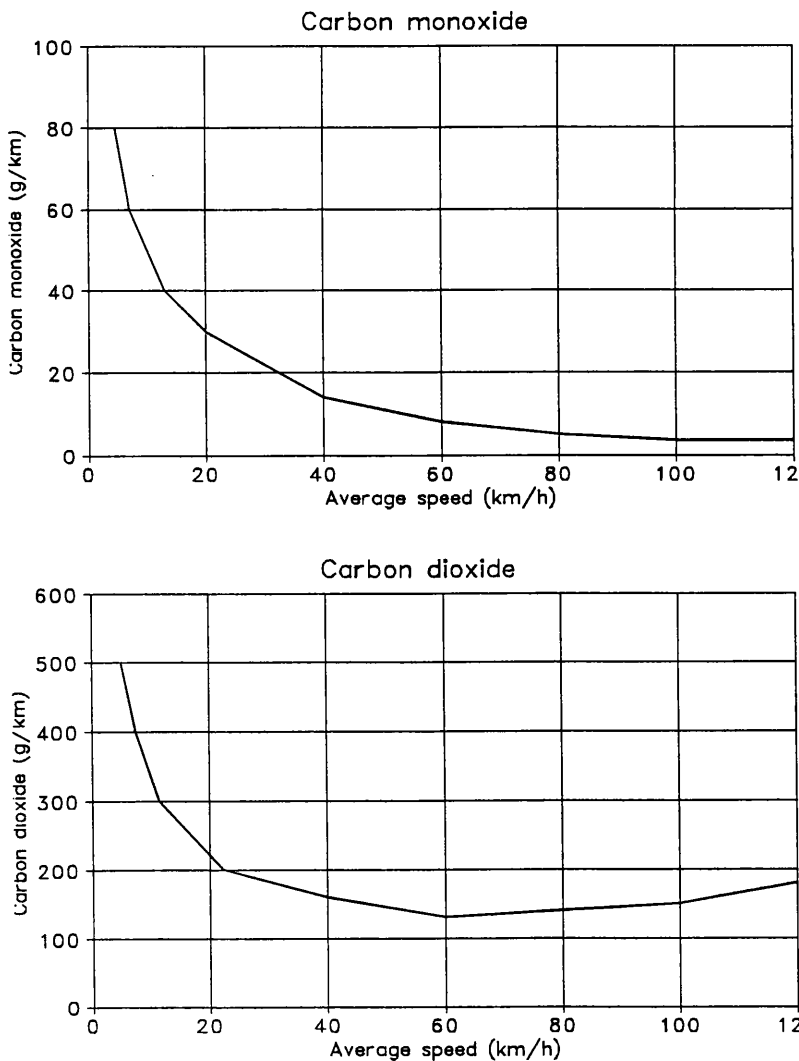
Because secondary pollutants disperse widely in the atmosphere during the time taken for reaction, their concentration are not always highest near the emission sources and their impact may spread over areas not confined to the locality of the traffic (Department of Transport et al., 1993).

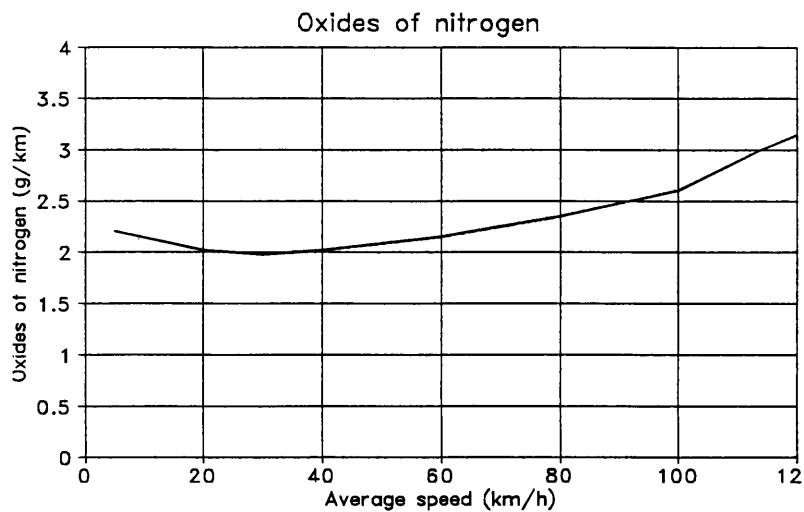
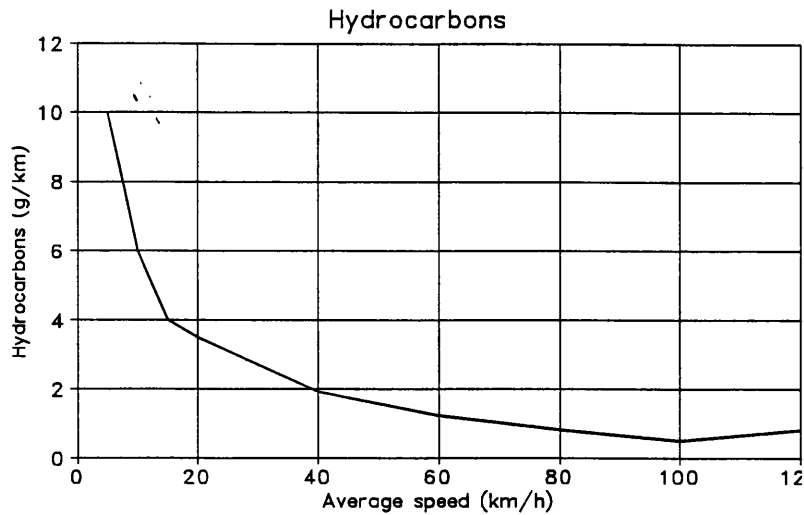
### 3.3 Factors influencing the emission of air pollutants

A large number of parameters influence vehicle emission rates. Vehicle exhausts vary considerably according to the type of fuel, level of traffic congestion, speed, driving patterns, driver attitude and behaviour, local traffic management, vehicle type, age and maintenance, fitting of catalytic devices, engine temperature (cold and hot start), loading and road gradient.

It has been widely verified that by increasing vehicle speeds within the urban speed range, CO, CO<sub>2</sub> and HC emissions decrease, but NO<sub>x</sub> exhausts actually increase (Potter and Savage, 1983; Joumard, 1989; Fergusson et al., 1989; Jost et al., 1992; Department of Transport et al., 1993; Eggleston et al., 1993). Typical speed related emission curves from light duty vehicles are indicated in Figure 3.1.

Figure 3.1: Speed-related emission factors for CO, CO<sub>2</sub>, HC and NO<sub>x</sub>





Source: Department of Transport et al., 1993.

Vehicle emissions are higher when the engine is cold. Cold starts and cold weather conditions contribute for higher emission rates because when the engine is cold the fuel tends not to vaporise well and as a result unburned or incompletely burned fuel may pass through the engine and into the exhaust stream. Besides, the engine has to run richer under cold conditions. Recent surveys have been carried out by the Transport Research Laboratory (Gover et al., 1994) and by the Warren Spring Laboratory (Farrow et al., 1993a, 1993b and 1993c) with different vehicle categories and start temperatures. Emission rates originated from on-road tests conducted with 'as received' cars after cold and hot start in urban driving conditions are given in Table 3.3:



**Table 3.3: Average cold and hot urban emission rates from current technology cars**

| Vehicle type            | Engine temperature | Pollutant emissions (g/km) |                  |                 |                 |                 |
|-------------------------|--------------------|----------------------------|------------------|-----------------|-----------------|-----------------|
|                         |                    | CO                         | HC               | NO <sub>x</sub> | CO <sub>2</sub> | PM <sup>a</sup> |
| Petrol without catalyst | cold               | 45.0                       | 5.4              | 1.7             | 213.2           | -               |
|                         | hot                | 27.3                       | 2.7              | 1.7             | 186.5           | -               |
| Petrol with catalyst    | cold               | 19.6                       | 1.8              | 0.5             | 318.4           | -               |
|                         | hot                | 2.1                        | 0.2              | 0.4             | 292.8           | -               |
| Diesel                  | cold               | 1.9                        | 2.6 <sup>b</sup> | 1.4             | 264.5           | 0.7             |
|                         | hot                | 1.2                        | 2.0 <sup>b</sup> | 1.1             | 229.1           | 0.6             |

a) PM emissions are not routinely measured in petrol cars, being typically very low.

b) HC from diesel were taken from EU dynamometer surveys.

Source: Compiled from Farrow et al. (1993a), (1993b) and (1993c).

According to the figures in the table above, if engines are fully warmed up, the emissions of CO<sub>2</sub>, CO and HC can be reduced by about 8-13.5%, 36-89.5% and 22-91%, respectively, depending on the vehicle type, and to a lesser extent in relation to NO<sub>x</sub> and PM. Thus, short trips are proportionally more polluting than long ones. Hence, the beginning of a trip produces much more emissions per kilometre than the rest of the journey. On a hot day a petrol car may have to be driven for about 10 kilometres in an urban area before the engine is fully warmed up and operating efficiently. In similar conditions, a diesel car may warm up after about 5 km (Quality of Urban Air Review Group, 1993b). Air pollution episodes can be critical in winter due to the lower atmosphere boundary layers and temperatures, higher humidity and presence of fog. On the other hand, some of the worst air pollution incidents may be recorded in summer, when sun light causes NO<sub>x</sub> and HC to react together producing ozone.

The pattern of emission for each pollutant originated from the combustion of petrol is quite different than from diesel engines. Diesel engines operate at very high air/fuel ratios, have superior thermal efficiency and lower combustion temperatures. Therefore, they tend to emit less HC, CO and CO<sub>2</sub>, but more PM than the petrol counterparts. Besides, diesel is not as highly volatile as gasoline, and evaporative emissions from diesel vehicles are lower than from petrols (Hassounah and Miller, 1994). Based on bench test emission data, diesel cars

and buses have proved to emit about seventy and one hundred times more PM than petrol cars, respectively (Chiquetto, 1995; Chiquetto and Mackett, 1994a and 1994b).

Catalytic converters are devices installed in the exhaust stream of petrol vehicles with the aim of reducing emissions. They consist of a mixture of noble metals, such as platinum and palladium. When heated, the metals catalyse the oxidation of HC and CO to CO<sub>2</sub> and water. Unleaded fuel is needed to operate vehicles equipped with catalytic converters because lead additives can destroy the activity of noble-metal catalysts. The fitting of three-way catalysts is remarkably effective for reducing CO, HC and NO<sub>x</sub>, but is worthless for reducing PM. Catalytics actually cause an increase in CO<sub>2</sub> emissions, since some CO is oxidised into CO<sub>2</sub> in the catalysation process. (Chiquetto, 1995; Chiquetto and Mackett, 1994a and 1994b). As the molecular weight of CO<sub>2</sub> is considerably higher than of the originating compound, there is a significant emission increase in absolute terms. The efficiency of catalysts is significantly reduced when the engine operates at cold temperature (more details about the efficiency of catalytic converters are given in Section 9.2). The Royal Commission on Environmental Pollution (1994) shows that emissions from petrol engines with catalytic converters vary considerably and present a high tendency to deteriorate with use.

Motor vehicles operate under a variety of driving conditions, specially in urban areas (Chiquetto and Santos, 1993). Santos (1981) concluded that the principal factor influencing exhaust emissions in urban conditions is the pattern of driving. When accelerating, the engine operates at an elevated rotational rate and consumes more fuel, which is necessary in order to take the vehicle from inertia or from lower speed ranges. Thus, the emission rates of most pollutants during the acceleration mode are higher than those rates produced during other driving patterns. Queuing vehicles produce a concentrated amount of emissions since they are much closer together than in moving conditions. Besides, the emissions from queuing are prolonged at a particular site and hardly affected by the dispersion imposed by moving vehicles. Road junctions have proved to have a dominant effect on pollutant emissions (Cohen, 1977). Clagget et al. (1981) found that CO concentrations at mid-block locations were about four times lower than those around the junction zone. Driver behaviour is also a large source of variation on vehicle emissions. Careful driving also reduces fuel consumption and emissions.

### 3.4 Vehicle exhaust emission control

In recognition of the magnitude of the air pollution problem caused by road traffic, measures have been taken to control the quantities emitted. Limits on vehicle emissions were first set in Britain in 1973 under the Road Traffic Act 1972 by the Motor Vehicles Regulations 1973 (SI 1973 No 1347). Such statutory limits have been applied to permissible levels of CO, HC and NO<sub>x</sub> in petrol vehicle exhausts. The limits have been reduced several times since they were first introduced and changes have been made to the test method to make it more realistic and effective. However, there was no obligation to comply with the limits and the standards based on European regulations were only made compulsory for both petrol and diesel vehicles only from 1 April 1991 (Haigh, 1992). Nowadays, before cars and heavy vehicles can be marketed in Europe, they must be tested and certified to conform with regulated limits on exhaust pollutants emitted. The most recently proposed standards applied to new vehicles were implemented in the beginning of 1993. They show a reduction of about 80 percent compared with the limits from two decades before. This does not mean, however, that emissions from vehicles in use remain at the design level, especially when the vehicle becomes old or is poorly maintained (Department of Transport et al., 1993). Table 3.4 shows the past, current and future regulation emission limits for light vehicles.

As illustrated in Table 3.4, additional restrictions in vehicle emissions are expected in the future. The European Community (1992a) has established new emission limits for 1996, separated for petrol and diesel cars, which will require new cars to reduce further their emission levels. The Royal Commission on Environmental Pollution (1994) suggests that there is no evidence that manufacturers will have difficulty in complying with the 1996 regulations for petrol cars, since many new cars tested on the prescribed EU cycle already achieve these standards. The 1996 restrictions may oblige engine manufacturers to use far more effective emission control systems in some models of their range. The European Commission has not yet put forward the proposals which are due to come into effect in the year 2000.

**Table 3.4: Historic of the European regulation for new light vehicle exhaust emissions**

| Year                | Emission limit |          |                 |                    |      |      |
|---------------------|----------------|----------|-----------------|--------------------|------|------|
|                     | CO             | HC       | NO <sub>x</sub> | HC+NO <sub>x</sub> | PM   |      |
| 1970 <sup>a,b</sup> | 100-220        | 8-12.8   | -               | -                  | -    |      |
| 1974 <sup>a,b</sup> | 80-176         | 6.8-10.9 | -               | -                  | -    |      |
| 1977 <sup>a,b</sup> | 80-176         | 6.8-10.9 | 10-16           | -                  | -    |      |
| 1978 <sup>a,b</sup> | 65-143         | 6-9.6    | 8.5-13.6        | -                  | -    |      |
| 1983 <sup>a,b</sup> | 58-110         | -        | -               | 19-26              | -    |      |
| 1988 <sup>a,b</sup> | 25-45          | -        | 3.5-6.0         | 6.5-15             | 1.1  |      |
| 1989 <sup>a,b</sup> | 19             | -        | -               | 5                  | 1.1  |      |
| 1991 <sup>c,b</sup> | 2.72           | -        | -               | 0.97               | 0.14 |      |
| 1993 <sup>c,d</sup> | 3.16           | -        | -               | 1.13               | 0.18 |      |
| 1996 <sup>c,e</sup> | petrol         | 2.2      | -               | -                  | 0.5  | -    |
|                     | diesel         | 1.0      | -               | -                  | 0.7  | 0.08 |
| 2000 <sup>c,f</sup> | petrol         | 1.0      | 0.1             | 0.1                | -    | -    |
|                     | diesel         | 0.5      | 0.1             | 0.3                | -    | 0.03 |

a) Standards expressed in g/test (according to the European Regulation test).

b) Source: Department of Transport et al. (1993).

c) Standards expressed in g/km.

d) Source: European Community (1991a).

e) Source: European Community (1992a).

f) European Parliament proposal; Source: European Community (1994).

Although the emission of some pollutants tend to decrease by control regulations, the emissions of other pollutants such as CO<sub>2</sub> are not expected to be reduced in the future. On the contrary, as shown in Section 3.3, with the increase of catalytic fitted cars and increase in traffic volumes, the emissions of CO<sub>2</sub> tend only to rise continuously.

Table 3.5 shows the emission limits for light goods vehicles which came into effect on 1 October 1994.

**Table 3.5: EU emission limits for light goods vehicles**

| Type of vehicle<br>(gross vehicle weight) | Emission limit (g/km) |                      |      |
|---|-----------------------|----------------------|------|
|   | CO                    | HC + NO <sub>x</sub> | PM   |
| up to 1.25 tonnes                         | 3.16                  | 1.13                 | 0.18 |
| 1.25 to 1.7 tonnes                        | 6.0                   | 1.6                  | 0.22 |
| 1.7 to 3.5 tonnes                         | 8.0                   | 2.0                  | 0.29 |

Source: European Community (1993).

The limits for the lightest of the light goods vehicles are identical to the 1993 limits for cars. Higher limits have been set for heavier light goods vehicles, which are likely to require the fitting of catalytic converters in petrol engines or oxidation catalysts in diesel engines, in order to meet the limits (Royal Commission on Environmental Pollution, 1994). In order to control emissions from light duty petrol driven vehicles in service, a CO and HC emission check has been included in the annual MoT test for these vehicles since November 1991.

Control over emissions from heavy goods vehicles are also being progressively tightened. The EU has committed itself to more stringent limits to take effect by the year 2000. Table 3.6 shows the European limits for heavy goods vehicles.

**Table 3.6: EU emission limits for heavy duty engines**

| Stage | Emission limit (g/kilowatt-hour) |          |                 |           |
|-------|----------------------------------|----------|-----------------|-----------|
|       | CO                               | HC       | NO <sub>x</sub> | PM        |
| 1993  | 4.9                              | 1.23     | 9.0             | 0.68-0.40 |
| 1996  | 4.0                              | 1.1      | 7.0             | 0.15      |
| 2000  | 2.0-2.8                          | 0.6-0.77 | 4.9-5.0         | 0.1       |

Source: European Community (1991b).

The emission limits for heavy goods vehicles are obviously more tolerant than for lighter engines, but they are also being progressively lowered. Heavy duty diesel vehicles are tested annually for smoke emissions and smoke tests for light duty diesel vehicles have been included in the MoT test since January 1993 (Department of Transport et al., 1993). Like diesel cars, heavy diesel vehicles can meet present emission limits without pollution control

devices. However, the achievement of future limits will require advances in engine technology.

Emission rates from in-service vehicles have been controlled through the MoT annual test since 1991. All cars over three years old are subject to the test and a smoke emission test has also been included in the annual inspection of heavy duty vehicles. The specific emission limits set in the test are:

- no vehicles are permitted to emit 'excessive' smoke
- cars produced from 1975 onwards are required to emit less than 1200 ppm of HC during an idle test
- cars produced between 1975 and 1983 are required to emit less than 6% of CO
- cars produced from 1983 onwards must meet a CO limit of 4.5%

The requirements of the MoT were tightened in 1993 and will be further restricted in 1996. This will ensure that vehicles are kept well maintained and emissions will not exceed established maximum limits.

Emissions are also controlled through fuel specifications. The most important fuel change has been the introduction of unleaded petrol, which is now widely available in Britain and its use is encouraged by a lower taxation rate. This will eventually lead to the complete elimination of lead emissions from traffic. There is also a limit to the amount of benzene in unleaded petrol and to sulphur in diesel fuel.

Vehicle emissions have been reduced in recent years as a result of these control measures allied to the compulsory implementation of catalytic converters. They will continue to be reduced further as new vehicles replace old ones. Other changes in vehicle technology are also of great importance in the accomplishment of future air quality goals.

### **3.5 The dispersion of pollutants into the atmosphere**

The air pollution process starts with the emission from exhaust sources. Immediately after pollutants are released into the atmosphere, they scatter widely and rapidly in all directions, mainly by wind carriage and dilution. The concentration falls off dramatically with distance from the source (Sistla et al., 1979). The dispersion of pollutants in the atmosphere is a very

complex process.

Pollutant concentration levels are the result of the balance between the amount emitted and the degree to which it is diluted and scattered in the atmosphere. Thus, it is necessary to translate a specific pattern of discharge into the resulting pattern of concentration. Pollution concentrations are the best indicators of air quality, since the exposure to different levels of which indicates the extent to which people's health and material deterioration may be affected.

It is extremely difficult to predict air pollution concentration, because of the large number and complexity of factors which can affect directly or indirectly its levels. Dispersion depends on numerous and combined effects of traffic emissions, meteorological conditions and surrounding characteristics (presence of natural and man-made barriers). Concentration levels are directly proportional to emission rates and reversely proportional to the dispersion factors. The capacity of the atmosphere to assimilate and scatter residual discharges varies with time, space, meteorological, climatic and topographic conditions and the nature of the material emitted. Some other weather conditions, such as solar radiation, temperature, pressure, humidity, atmosphere stability and occurrence of temperature inversions also exert significant influence on the period of permanence of pollutants in atmosphere, through preventing or hastening their dilution. Wind speed and direction are the most relevant dispersion factors. Increases in wind speed accelerate the dispersion of pollutants, whereas wind direction defines the direction of pollutant dispersion from the source to the receptor. Thus, pollutant concentrations are usually higher downwind than upwind of the emission source. Leeward concentrations are generally higher than windward ones, due to the turbulence induced by moving vehicles themselves. Other factors affecting the atmospheric dispersion of exhaust gases are time, distance from the source and road lay out (existence of trees, buildings and other urban apparatus), by interfering with the speed, intensity or direction of the wind. Complex aerodynamic effects make modelling rather complicated and even impracticable in situations when the pattern of variations cannot be predicted with reasonable accuracy.

Levels of pollutant concentration may vary considerably within distinct segments of the same road and even from one side of the road to the other. Moreover, it may change over different times of the day, days of the week and seasons. This can be a result of the combination of distinct factors affecting either vehicle emissions (traffic flow, composition, speed and driving pattern) or the dispersion conditions (wind speed and direction, distance source-receptor and

position of exhaust pipe<sup>2,3</sup>). As a consequence, pollutants tend to achieve different concentration levels in the air at different points and time periods.

### **3.6 Methods for the prediction of air pollution concentration**

The production of environmental data is essential for studies in which the magnitude of degradation is concerned. Measurements of air quality require expensive equipment and maintenance, as well as a great deal of work. Unless continuous monitoring equipment is available, direct measurement is an ineffective, hard to perform and time-consuming task. Measurements are most commonly performed for deriving statistical models or for calibrating and validating existing models for the prediction of air pollution concentration levels.

The methods for predicting pollutant concentrations provide a number of advantages over the measurement approach, of which the main ones are greater practicality and capability of simulation. They describe the likely effects of prevailing traffic emissions combined with weather conditions on the environmental quality, at a specific point and time. By inputting the traffic and meteorological variables required, which are far easier to collect than the environmental ones, models estimate the level of environmental deterioration. Predictive models are specially attractive as a planning tool to help policy formulation, by testing the impact on air quality from various traffic policy options or from changes in the characteristics of the effluent and meteorological factors.

An important consideration when predicting environmental degradation concerns the definition of the method to be applied. There are three main approaches to dispersion modelling: Eulerian, statistical and Lagrangian (Horowitz, 1982a). The Eulerian approach uses the continuity equation of mathematical physics to develop a description of the processes that govern the relations between emissions and concentrations. Such models, however, are complicated and rarely used in transport air pollution modelling. The statistical approach of dispersion modelling makes use of direct observations to infer functional relations between

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<sup>2</sup> From the 20 models of car which sell in largest numbers in the U.K., 16 have their exhaust pipe on the pavement side, apparently because they were originally designed for left-hand drive (The Times, 1994). This induces higher pollutant concentrations at the locations where pedestrians are exposed.

<sup>3</sup> Babies are most exposed to health risks, as pushchairs put them and their delicate lungs in line with car exhaust pipes.



pollutant concentrations and emission rates, under the influence of meteorological variables. Statistical techniques, such as regression analyses, are required to establish such relationships. This approach, however, is limited to a range of specific conditions. It cannot be used reliably when explanatory variables other than those considered in the model exert significant influence on the results or when the values of the variables are beyond the ranges represented in the data sets used to develop the model. The Lagrangian approach is the most frequently used in estimating pollutant concentrations near roadways. It uses a probabilistic description of the motions of pollutant particles in the atmosphere to derive an expression for pollutant concentrations. The most widely used Lagrangian models for predicting air pollution concentration follow the Gaussian plume dispersion theory for stationary sources, which is addressed in more detail later in this section.

A variety of methods have been developed in the U.K. to predict pollutant concentrations at roadside locations, for specific ranges of conditions. In the U.K., Crompton and Gilbert (1972) made the first attempt to quantify air pollution levels at kerbside, in which traffic density per unit width of carriageway was found the most significant variable. A further variety of traffic and lay-out characteristics were included.

Two empirical regression predictive models were developed by Joyce et al. (1975) for predicting average concentration levels of carbon monoxide and smoke in one-hour periods. Since these models were based on measurements taken at specific sites in London and far from the main intersections, their applicability is limited to very specific situations and they do not account for the effects of urban traffic driving.

Canyon is the dominant layout configuration in urban areas. The dispersion process in canyon conditions is particularly complex, since the building facades prevent a natural dispersion and the wind can create a helical vortex which produces distinct concentrations at both sides of the canyon. Pollutants may build up or be quickly dispersed depending on the direction of the wind in relation to the buildings. The modelling of pollutant concentrations from traffic emissions in street canyon conditions presents major complications (Menard et al., 1987), and Gaussian models tend not to perform such predictions well. Linaritakis (1988) developed a multiple regression analysis model to estimate CO and PM concentration levels between the kerbside and building facades, from interrupted traffic flow in urban street canyon conditions. In order to avoid the complex influence of diverse driving patterns near to road junctions on emission rates, pollutant concentrations have been predicted at mid-block locations. As is the

case for all statistical models, its range of applicability is limited.

A method has been specifically developed for predicting pollutant concentrations at interrupted flow situations. This research has used models for dispersion, queuing, emission and traffic assignment in order to predict CO, HC, NO<sub>x</sub> and lead concentrations at any point along a road and their distribution around junctions (Matzoros, 1988; Matzoros and Van Vliet, 1992). Pollutant concentrations are predicted on signalised and priority intersections. As is the case for most approaches based on the plume dispersion theory, the emission model included in the method is not very robust and, because it uses actual vehicle emission data, it is inevitably outdated. The flow of lorries and buses are not accounted for and the pollutant PM is not included in the emission model. Furthermore, currently important variables such as the proportion of diesel cars in the traffic and the fitting of catalytic converters on petrol cars are not considered.

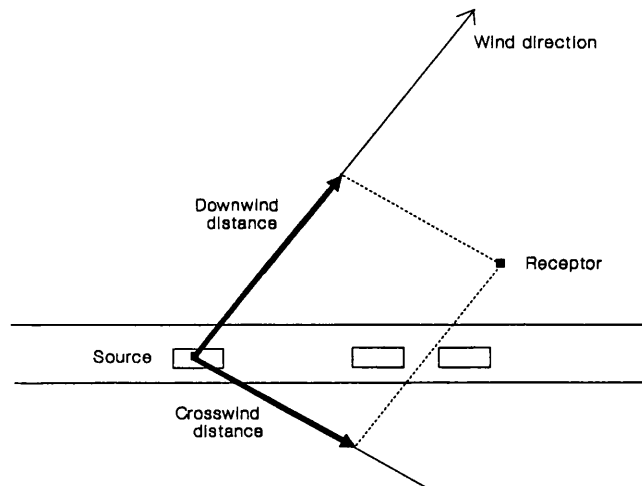
Most existing methods for predicting air pollution use a common approximation, in which the concentration across the plume follows a Gaussian distribution. The Gaussian plume dispersion theory is fully described in Horowitz (1982a). It assumes a continuous emission regime and a normal dispersion distribution on horizontal and vertical directions, and it is suitable to assess the diffusion prediction of non-reactive pollutants. It was first conceived to predict pollutant dispersion from industrial chimneys, but it has proved to enable a large spectrum of applications, including traffic pollution modelling (Horowitz, 1982a). In fact, the Gaussian dispersion theory has been extensively used in modelling air pollution from road transport sources, and the main modifications to the original concept are in relation to the height, number of contribution sources and the effects from their movement. A critical appraisal on the Gaussian approach for the prediction of traffic air pollution and the results of various applications are reported in Section 6.2.3.

Hickman and Colwill (1982), on behalf of the Transport Research Laboratory (TRL), have developed an extensively-used method based on the Gaussian theory for predicting pollutant concentrations (the main applications of this method are reported in Section 6.2.3). The dispersion equation is described by Equation 3.1 and the modelling of air pollution dispersion is represented in Figure 3.2.

$$C = a \frac{E F}{\pi \sigma_y \sigma_z \mu} \exp\left(-\frac{y^2}{2 \sigma_y^2}\right) \quad \text{Equation 3.1}$$

- $C$  = pollutant concentration at a receptor with coordinates  $x, y, z$  in relation to the source (ppm),  
 $E$  = emission rate (g/sec),  
 $F$  = traffic flow (vehicle/h),  
 $\mu$  = wind speed in  $x$  direction (m/s),  
 $y$  = crosswind distance from source (m),  
 $\sigma_y, \sigma_z$  = standard deviations of concentration distribution in  $y$  and  $z$  directions at downwind distance  $x$  from the source (m).  
 $a$  = adjustment factor which accounts for background concentrations and converts the units of concentration  $\text{g/m}^3$  to ppm (more details in Section 6.2.1).

**Figure 3.2: The modelling of air pollution dispersion**



Since most traffic generated pollution is emitted at ground level, the method assumes that there is no difference in height between the source and receptor. Therefore, the method simplifies the problem into two dimensions only.

Although this method is based on the Gaussian theory, it is to a large extent an empirical approach. A number of empirical formulations for diverse variables were developed by Hickman and Colwill (1982) to be included in the method. These relationships are described below and are expressed by Equations 3.2 to 3.5.

Analyses of several extensive data surveys showed that the turbulent air movement caused by the traffic usually enhances the effect of the wind, increasing the dispersion. This effect

is most significant at low wind speeds, and as the wind speed increases the effect becomes less important. An empirical relationship has been determined to express the effective wind speed ( $\mu$ ) in terms of measured one ( $\mu'$ ), which is expressed by Equation 3.2.

$$\mu = \frac{\mu'}{0.59 + 0.11 \mu'} \quad \text{Equation 3.2}$$

Generally, the influence of the wind speed is overridden locally by the complex air turbulence produced by traffic. Thus, the method assumes the lowest wind speed at roadside sites of 1 m/s, below which level the results produced are less accurate.

Although in theory pollutants disperse only in downwind directions, the air turbulence produced by traffic flows scatters pollutants upwind as well. Upwind concentrations are simulated by the model as if they were situated downwind, but at a much greater distance from the source. The effective downwind distance for points upwind in relation to the source is given by Equation 3.3.

$$D = 1.3 \times d^{1.5} \quad \text{Equation 3.3}$$

D = effective downwind distance,  
d = actual upwind distance.

The wind direction determines the orientation of the pollutant plume from the source. This is taken into account by the plume standard deviation, in terms of an angle measuring cross wind spread ( $\sigma_y$ ) and a height establishing vertical spread ( $\sigma_z$ ). The downwind concentrations of a pollutant along the vertical and crosswind axes of a plume are assumed to be normally distributed. Roadside observed concentrations resulting from traffic emissions do not conform to the standard dispersion patterns developed for stationary sources. Thus, roadside Gaussian standard deviations have also been defined from empirical experiments, which have presented acceptable agreement over a range of conditions. The standard deviations along the z and the y axes are shown in Equations 3.4 and 3.5.

$$\sigma_z = 1.85 [1 + \exp[0.39(\ln D)^3 - 4.76(\ln D)^2 + 20.95(\ln D) - 32.67]] \quad \text{Equation 3.4}$$

$$\sigma_y = 12.5 \sigma_z \quad \text{Equation 3.5}$$

The TRL model is named PREDCO and its functioning is described in detail in the user guide (Department of Transport, 1983b). The main assumptions upon which this model is based include the restrictive movement of traffic in a single line, all sources produce a unique and constant emission profile under every driving pattern along the road and all vehicles drive at the average speed. These assumptions will be dealt with in Section 6.2.1.

### 3.7 Environmental standards for pollutant concentrations

Environmental standards provide reference guidance for the maximum acceptable extent of environmental degradation. They aim to ensure public health and well-being, and to protect people who are particularly sensitive to different types of pollution, such as children, asthmatics, elderly and pregnant women. Standards are very useful for the identification of locations where potential problems may exist. They are helpful as guidelines for transport and planning policy decisions, as to ensure successful pollution control strategies, and for giving focus to environmental objectives, which represent the first stirrings of progress towards sustainability (Whitelegg, 1993). When not regulated or legislated, the non-mandatory limits may be merely proposed as recommended guidelines. Standards may be expressed in terms of the amount of effluent being dispersed into the environment (emissions), but they are usually more useful if expressed in terms of the quality of the receiving environment (pollutant concentration).

The United States Federal Air Quality Standards were pioneers in setting recommended limits for ambient levels of airborne pollutants, according to different exposure periods. These criteria have been employed for many years in several countries, including the U.K., and are among the most stringent in the world. Data from U.K. traffic air pollution surveys have shown that the standard for 8 hour period is more stringent and difficult to meet than the limit for 1 hour (Department of Transport, 1983a).

The World Health Organisation (1987) have established air quality guidelines for Europe,

whereas European Community Directives (European Community, 1980 and 1985) have been implemented in the U.K. by regulations which ensure that concentrations are monitored and kept below specified levels. New standards are being put forward by the Expert Panel on Air Quality Standards, appointed in 1994 by the Department of the Environment. It recommends maximum acceptable levels of pollutant concentrations in the atmosphere (Local Transport Today, 1994). The panel has produced reports on benzene, ozone, carbon monoxide and sulphur dioxide, and plans to publish reports for other pollutants in the near future. No safe levels of HC (as a group of pollutants) have been recommended so far. Table 3.7 shows the air quality standards and guidelines which have been established by international bodies.

**Table 3.7: Air quality standards and guidelines**

| Pollutant                               | NAQS          |                          | WHO           |                      | ECD           |                      |
|---|---------------|--------------------------|---------------|----------------------|---------------|----------------------|
|   | Exposure time | Concentration levels     | Exposure time | Concentration levels | Exposure time | Concentration levels |
| CO<br>(mg/m <sup>3</sup> )              | 1 hour        | 40 <sup>a</sup> existing | 15 min        | 100                  | 1 hour        | 200 <sup>c</sup>     |
|   |               | 29 proposed              | 30 min        | 60                   |               |                      |
|   | 8 hour        | 10 <sup>a</sup>          | 1 hour        | 30                   |               |                      |
|   |               |                          | 8 hour        | 10 <sup>b</sup>      |               |                      |
| NO <sub>2</sub><br>(µg/m <sup>3</sup> ) | 1 year        | 100                      | 1 hour        | 400                  | 1 hour        | 200 <sup>c</sup>     |
|   |               |                          | 24 hour       | 150                  |               |                      |
| PM<br>(µg/m <sup>3</sup> )              | 24 hour       | 260                      | 24 hour       | 120                  | 24 hour       | 80 <sup>d</sup>      |
|   | 1 year        | 75                       |               |                      |               | 130 <sup>e</sup>     |
|   |               |                          |               |                      |               | 250 <sup>f</sup>     |

NAQS: United States National Air Quality Standards (U.S. Government, 1971).

WHO: World Health Organisation (1987).

ECD: European Community Directives (European Community, 1980 and 1985).

a) limits which should not be exceeded more than once a year.

b) limit identical to the standard recommended by the Expert Panel on Air Quality.

c) limit expressed as 98<sup>th</sup> percentile of hourly means over a year.

d) limit expressed as median of daily means over a year.

e) limit expressed as median of daily means over the winter, October to March.

f) limit expressed as 98<sup>th</sup> percentile of daily means over a year.

Monitoring data show that many people in the U.K. are exposed from time to time to concentrations of pollutants which exceed the WHO guidelines, particularly in urban sites (Royal Commission on Environmental Pollution, 1994). This information can be used to indicate the locations where polluting sources might be adversely affecting people's health.

It is clear that the imposition and enforcement of environmental standards and guidelines are likely to produce benefits to the environment. However, the realisation of the environmental standards can be very difficult in practical terms. The lower the standard setting the greater are the costs associated with the resulting environmental benefits, and this relationship is far from linear. Such costs can be incurred from the need to limit traffic flows (and restrict mobility) or from the requirement of further reductions of emission rates by individual vehicles (implementation of devices or development of other technological improvements).

## 4. TRAFFIC NOISE

### 4.1 Basis and concepts

Sound is defined as a regular disturbance resulting from mechanic vibrations through an elastic means (specially the atmosphere), for which energy is transmitted from a source by progressive sonorous waves. These sonorous waves are detected by the human ear, and within the audible range, produce the sensation of hearing (Department of Transport, 1983a, TEST, 1991).

Noise is characterised as any acoustical phenomenon which produces a sensation perceived as disturbing, unpleasant or unwanted. Therefore, the concept of noise is quite subjective and depends above all on however people may perceive its effects. Some people may consider to be noise maybe an enjoyable sound for others.

According to TEST (1991) noise is any sound that:

- advertises the presence of undesired activities;
- interferes with desired activities, including communication, relaxation, leisure and sleep;
- causes annoyance or disturbance;
- causes physical or mental harm.

Noise can provoke a wide range of effects, depending on its intensity, the vibration frequency and exposure time. The effects caused by noise can be of physical or psychological nature, including annoyance, stress, agitation, behavioural changes, activity disruption, sleep disturbance<sup>4</sup>, hard of hearing and loss of intelligibility and ease of communication. The auditory fatigue caused by noise exceeding a certain level can be reflected in a transitory decrease in auditory acuity. Long-term exposure to such noise levels can result in permanent hearing loss (European Conference of Ministers of Transport, 1990). Deafness is the effect

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<sup>4</sup> Research has shown that exposure to noise during sleep diminishes not only its duration but also its quality, which is not necessarily perceived by the sleeper (European Conference of Ministers of Transport, 1990).



of noise which can be pinpointed with most certainty.

The standard unit of measurement of the sound pressure is the decibel (dB). The decibel is expressed as the ratio of the pressure fluctuations due to the passage of a sound wave and a small reference pressure. Equation 4.1 describes the measurement of the sound pressure level.

$$SPL = 20 \log \left( \frac{P}{P_o} \right) \quad \text{Equation 4.1}$$

SPL = Sound Pressure Level (dB),  
P = Measured sound pressure (N/m<sup>2</sup>),  
P<sub>o</sub> = Reference sound pressure, normally considered as the lowest audible pressure (2 x 10<sup>-5</sup> N/m<sup>2</sup>).

Noise levels are measured in a logarithmic scale which provides two advantages. First, the logarithmic scale is well suited to human hearing, which is also logarithmic in its behaviour rather than linear, and therefore it gives a more realistic indication of noise level changes. Second, since the amplitude of audible pressure is very large, the logarithm reduces the scale conveniently.

The human ear can respond to very small pressure fluctuations and the audible range is very large. The sound pressure realised by the human ear can be as low as 0 dB, which characterises the inferior limit of audibility (threshold of hearing), and as high as 120 dB, when sound becomes so loud that it begins to hurt (threshold of pain) (Department of Transport et al., 1993).

Since the human ear presents different sensitivities to the wide range of existing frequencies, a good indication of annoyance has to adjust the sound pressure level as to give more weight to the frequencies which are detected most readily by the human ear. The best correlation between perceived and actual loudness is made in A-weighted decibels, dB(A), which produces a single weighted measurement of the likely annoyance of most types of noise (Department of Transport, 1983a). Social surveys on disturbance due to traffic noise have shown that annoyance best correlates with L<sub>10</sub> sound levels in dB(A) (P. Nelson, 1980). The L<sub>10</sub> level is the sound exceeded for 10% of the time over a specified period, and it is usually established for 1 and 18 hour intervals. L<sub>10</sub> has the advantage that it is a straightforward

measure and can be determined with accuracy, since prediction and measurement techniques are well developed.

Traffic is one of the dominant sources of noise nuisance, and it is distinctly perceived by individuals in dwellings situated in urban areas (Plowden and Sinnott, 1977; European Conference of Ministers of Transport, 1990; TEST, 1991; Klaboe, 1992 and Royal Commission on Environmental Pollution, 1994). The noise from traffic stream is not constant, but varies from time to time according to the number, category and moving conditions of the sources at each moment. Traffic noise is often intermittent, because periods of quiet are interspersed with periods of noise, in which the intensity also varies widely.

Traffic noise can be originated from several sources. Table 4.1 shows the diverse sources of traffic noise and their dominant occurrence.

**Table 4.1: Main sources of traffic noise and their dominant occurrence**

| Types of noise   | Source   | Dominant occurrence                 |
|------------------|--|-------------------------------------|
| power noise      | engine, transmission, gas exhaust and inlet cooling              | interrupted traffic                 |
| rolling noise    | tyre in contact with the ground, transmission and wheel bearings | free flow traffic, high speeds      |
| electronic noise | horns, sirens and alarms   | if electronic devices are activated |

Source: Adapted from TEST (1991) and Department of Transport et al. (1993).

Other sources of noise include the broadcast system operation, gearbox, vehicular vibration, aerodynamic friction, hooters and brakes. Since traffic noise is produced in so many different ways, the reduction of noise emissions at the source can be very complex.

The engine noise comes as a result of combustion within the cylinder and mechanical operation, which vary with engine speed and power output. Traffic noise is dominated by power train noise at low speeds which are typical in urban areas. At speeds over about 40 km/h noise from light vehicles is dominated by rolling noise, while at highway speeds the rolling noise from heavy goods vehicles contributes substantially to the total noise level (European Conference of Ministers of Transport, 1990).

The adoption of an appropriate index is necessary for the purposes of assessment of the overall noise level. The index adopted by the British Government to assess traffic noise is  $L_{10, 18\text{-hour}}$ , which is the arithmetic mean of  $L_{10}$  levels between 6:00 am and 12:00 pm. The statistical averaging process used to derive the  $L_{10, 18\text{-hour}}$  index has proven to give a good correlation with the scale of residents' dissatisfaction over a wide range of exposures (Department of Transport, 1983a). Measures of night-time noise, however, do not correlate as well with dissatisfaction as the  $L_{10, 18\text{-hour}}$  index. Further research is needed to produce a better indicator of night-time noise, since no index is currently available to provide adequate correlation within this period.

The correlation of noise levels with disturbance is not straightforward, since it depends on how different people realise the effects of noise and on the period of exposure. Different noise levels can disturb people in various degrees, interfere with communication, affect sleep quality, cause stress reactions, damage the inner ear and provoke risks of deafness.

An index of noise nuisance which has been used in many surveys is the percentage of people who say they are bothered very much by traffic noise (Department of Transport et al., 1993). The percentage of people likely to be bothered very much by a certain level of traffic noise before an increase or a decrease in noise levels is illustrated by the curves given in Figure 4.1. These relationships have been derived from the National Environmental Survey data (Department of Transport et al., 1993). The best fit equations to the curves in Figure 4.1 were derived from multiple regression, and are given by Equation 4.2 and 4.3 below. The *R squared* parameter are 0.89 and 0.97, respectively, which reflect a good agreement of these equations to the curves.

$$P_i = - 54 + 1.2062 \times N \quad \text{Equation 4.2}$$

$$P_d = - 77 + 1.8530 \times N \quad \text{Equation 4.3}$$

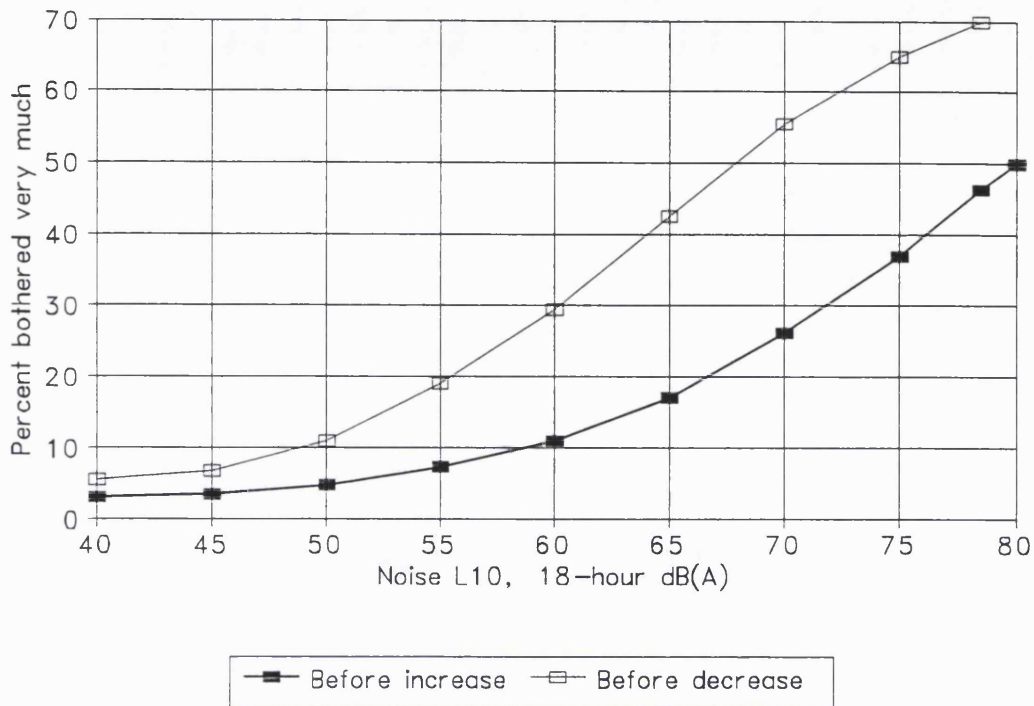
$P_i$  = Percentage of people bothered before the increase.

$P_d$  = Percentage of people bothered before the decrease.

$N$  = Noise level  $L_{10, 18\text{-hour}}$  dB(A).

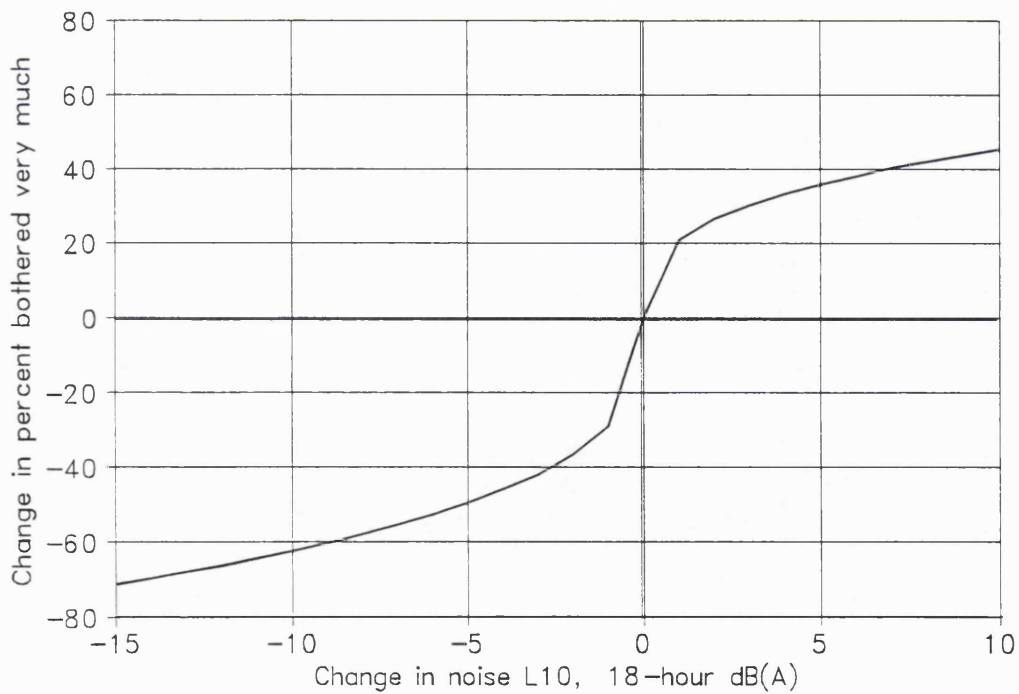
The changes in the percentage of people bothered by traffic noise can be predicted from the respective change in traffic noise using the relationship in Figure 4.2.

**Figure 4.1: Estimation of the percentage of people bothered by traffic noise**



Source: Department of Transport et al. (1993).

**Figure 4.2: Change in the percentage of people bothered by traffic noise**



Source: Department of Transport et al. (1993).

The curve in Figure 4.2 is mathematically expressed by the Equations 4.4 and 4.5 (Department of Transport et al., 1993).

$$C_p = 21 C_n^{\frac{1}{3}} \quad \text{if } C_n \text{ is positive} \quad \text{Equation 4.4}$$

$$C_p = 29 C_n^{\frac{1}{3}} \quad \text{if } C_n \text{ is negative} \quad \text{Equation 4.5}$$

$C_p$  = Change in the percentage of people bothered.

$C_n$  = Change in traffic noise  $L_{10, 18\text{-hour}}$  dB(A).

The Department of Transport et al. (1993) recommends the use of the correlations shown in the figures and equations described above to forecast the effects of noise changes. In fact, these correlations can be very useful for the assessment of the impacts produced by changes in the patterns of traffic flows, as a result of the implementation of transport policies or traffic schemes.

#### 4.2 Vehicle noise emission control

The first official report on the problem of noise suggested that standards should limit traffic-induced noise levels (U.K. Government, 1963). A limit of 85 dB(A) was proposed for noise emission from all vehicles, excepting motorcycles which were limited to 90 dB(A). Noise regulations were first introduced in 1969 in the U.K. and are contained in the Road Traffic-Motor Vehicles Regulations (House of Commons, 1969). These included regulations governing audible warning devices, the need for silencers and provisions against the emission of excessive noise, as well as specifications on sound emission limits and test conditions not only for new vehicles, but also for vehicles in use. In 1973, the Motor Vehicles Regulation also stated the maximum permitted levels of noise for various classes of vehicles (Bell, 1990). These limits were still very similar to those set a few years earlier.

More recently, however, new vehicles became subjected to further restrict noise level limits which were set in order to comply with the European Union standards. The limits established for 1988-1989 were laid down in the EEC Directive 84/424/EEC (European Community, 1984), and range from 77 to 84 dB(A), according to the type of vehicle. The new limits which are being put into practice in 1995-1996 have been set by the EC Directive 92/97/EEC

(European Community, 1992b) and restrict noise emissions to the range of 74 to 80 dB(A). The limits imposed on new designs of road vehicles are shown in Table 4.2.

**Table 4.2: Sound level limits for new vehicles**

| Vehicle type                                  | Sound limit <sup>a</sup> dB(A) |             |             |
|---|--------------------------------|-------------|-------------|
|   | 1983                           | 1988 - 1989 | 1995 - 1996 |
| cars  | 80                             | 77          | 74          |
| buses < 150 kW                                | 82                             | 80          | 78          |
| buses > 150 kW                                | 85                             | 83          | 80          |
| small buses, delivery lorries < 2 tonnes      | 81                             | 78          | 75          |
| small buses, delivery lorries: 2 - 3.5 tonnes | 81                             | 79          | 77          |
| lorries > 3.5 tonnes, < 75 kW                 | 86                             | 81          | 77          |
| lorries 75 - 150 kW                           | 86                             | 83          | 78          |
| lorries > 150 kW                              | 88                             | 84          | 80          |

a) If the engine is diesel powered, the limit values are 1 dB(A) higher for cars, buses and goods vehicles of less than 3.5 tonnes.

Source: European Community (1984 and 1992b).

Noise limits were reduced by 2-4 dB(A) in 1988, which is equivalent to halving the sound intensity, and are being further reduced by the same amount between 1995 and 1996. From 1995 new cars will be required not to exceed 74 dB(A) when accelerating in low gear at full throttle. Reductions in the sound produced at the source will have an important role in achieving the targets for the abatement of road traffic noise as proposed by the Royal Commission on Environmental Pollution (1994). These targets are reported in Section 4.5.

It is very difficult to reduce noise emission levels, since the sound produced by vehicles comes from several sources. In principle, sound can be neutralised by generating waves which are an exact inversion of those produced by the original source. However, this technology is too complex and costly to apply to ordinary ranges of small vehicles (Royal Commission on Environmental Pollution, 1994). The achievement of the targets for noise will depend on the accomplishment of the noise limits legislation and on the use of quieter surfacing materials and screening.

### 4.3 Factors influencing traffic noise

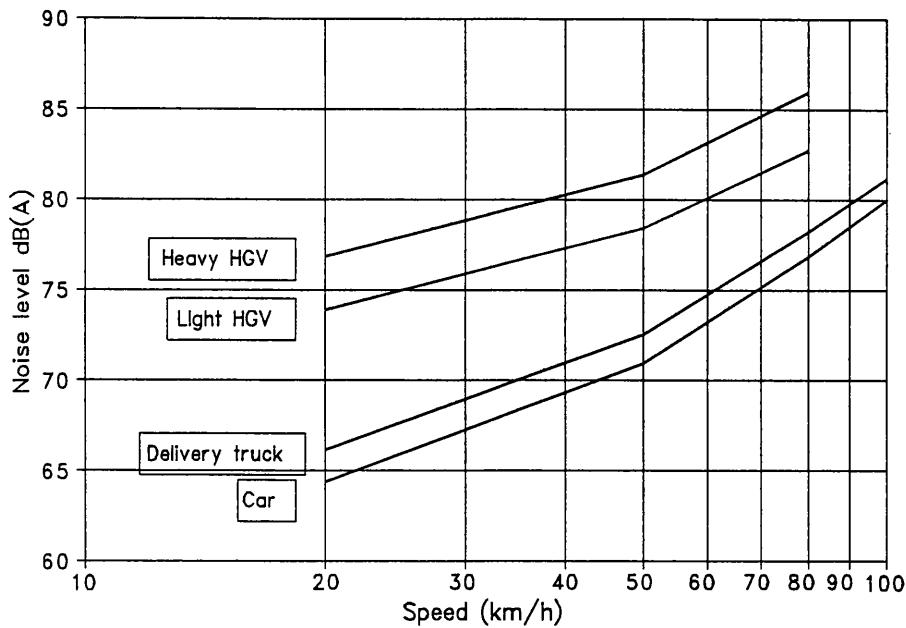
The level of traffic noise at a certain point is a composition of many different source contributions at various distances. The characteristics of noise may vary according to traffic flow, vehicle and engine type, operating conditions, vehicle speed and engine rotation rate, state of maintenance and age, nature of the ground surface and reflection from building facades and other obstructions. Since traffic patterns are random, so are the noise fluctuations, which vary due to variations in traffic flow and composition throughout the day.

The volume of traffic obviously exerts a significant influence on the level of noise produced. However, this correlation is far from linear. Pressure changes are proportional to the square of the energy, and by duplicating the energy level there is an equivalent change of only 3 dB. Similarly, there is a logarithmic relationship between traffic volume and noise level. In theory, when the volume is doubled or halved there is a corresponding 3 dB(A) increase or reduction in noise level  $L_{10}$  (Department of Transport, 1994d). In practice, when the traffic volume is doubled, the corresponding noise level has proved to increase by about 2.3 dB(A) in highways and 2.7 dB(A) in urban roads (Working Group Research into Road Traffic Noise, 1970).

Vehicle speed has proved to be a significant factor influencing traffic noise  $L_{10}$  levels from freely flowing traffic (Gilbert et al., 1980). The maximum noise produced by a vehicle in steady speed is proportional to the logarithm of its speed. Thus, an increase in noise level of 10 dB(A) can be noticed when vehicle speed becomes twice as high, the engine being kept in the same gear. Figure 4.3 shows the variation of traffic noise with speed for different vehicle types.

Although traffic speed is not as significant for interrupted traffic flow conditions, speed control alone has enormous potential to reduce noise levels in cities.

**Figure 4.3: Variation of traffic noise with speed for different vehicle types**



Source: Whitelegg (1993).

The percentage of heavy vehicles in the flow strongly influences the global noise level around roadways (P. Nelson, 1980), since heavy vehicles produce higher levels of noise than other vehicles at all speeds (see Figure 4.3). It also can be noticed in Figure 4.3 that the difference in noise levels between heavy and light vehicles is lower on highways, where high speeds are found. When the proportion of heavy vehicles in the flow becomes twice as high an increase of 2 to 3 dB(A) has been noticed (Departamento Nacional de Tránsito, 1980).

In the speed range typically found in urban areas, the rolling noise caused by the friction between tyre and the road surface contributes significantly to the total noise produced, especially when the road surface is wet. The frictional properties of tyre and road surface noise are function of the tyre design and the nature of the road surface. Safe road surfaces and tyre with a good skid resistance are known to increase vehicle noise. Thus, there is a conflict between the requirements for noise abatement and the need for a reasonable standard on road surface skidding resistance (P. Nelson, 1980).

The perception of noise is much greater in residential dwellings, where peace and quiet are most appreciated. Recent research indicates that people are sensitive to changes in traffic noise associated with the implementation of new road schemes. In the period following a



change in traffic flow, people may find appreciable benefits or disbenefits where noise variations are as small as 1 dB(A) (Department of Transport et al., 1993). Such small change in noise level is equivalent to a change in traffic flow of about 20% or a change in speed of about 25%.

The characteristics of road lay-out, such as the height of building facades and the presence of barriers and other street furniture, can affect noise levels considerably. Although the effects of barriers in highways are well understood, in urban roads it is still difficult to determine them. It is known, however, that the reflection of the sound from building facades can rise noise levels up to 3 dB(A).

Since noise propagates through sonorous waves which disperse three dimensionally, its intensity diminishes as the receptor departs from the source of noise. When the distance between the traffic flow and the receptor is doubled, there is a decrease of only 3 dB(A). This effect is strongest for distances ranging from 15 to 50 meters.

Noise levels tend to increase with the road slope, mainly from heavy vehicles, as a result of the higher engine work employed for the movement.

The introduction of insulating devices like double glazing can be very efficient in reducing indoor noise levels, whereas noise barriers are usually designed to enclose noise into traffic corridors.

#### **4.4 Methods for the estimation of traffic noise**

In the U.K., a reliable method for predicting traffic noise levels is required not only to determine eligibility for noise insulation under the 1975 Noise Insulation Regulations, but also as part of environmental impact assessments of alternative transport and land use planning schemes. Methods for the prediction of noise levels are particularly useful in comparisons between situations before and after the implementation of traffic management measures.

Several models have been developed to estimate noise levels produced by traffic. The empiric models have achieved the highest level of development and are the ones most extensively used in noise estimation. Like other empirical models developed in other areas, these models

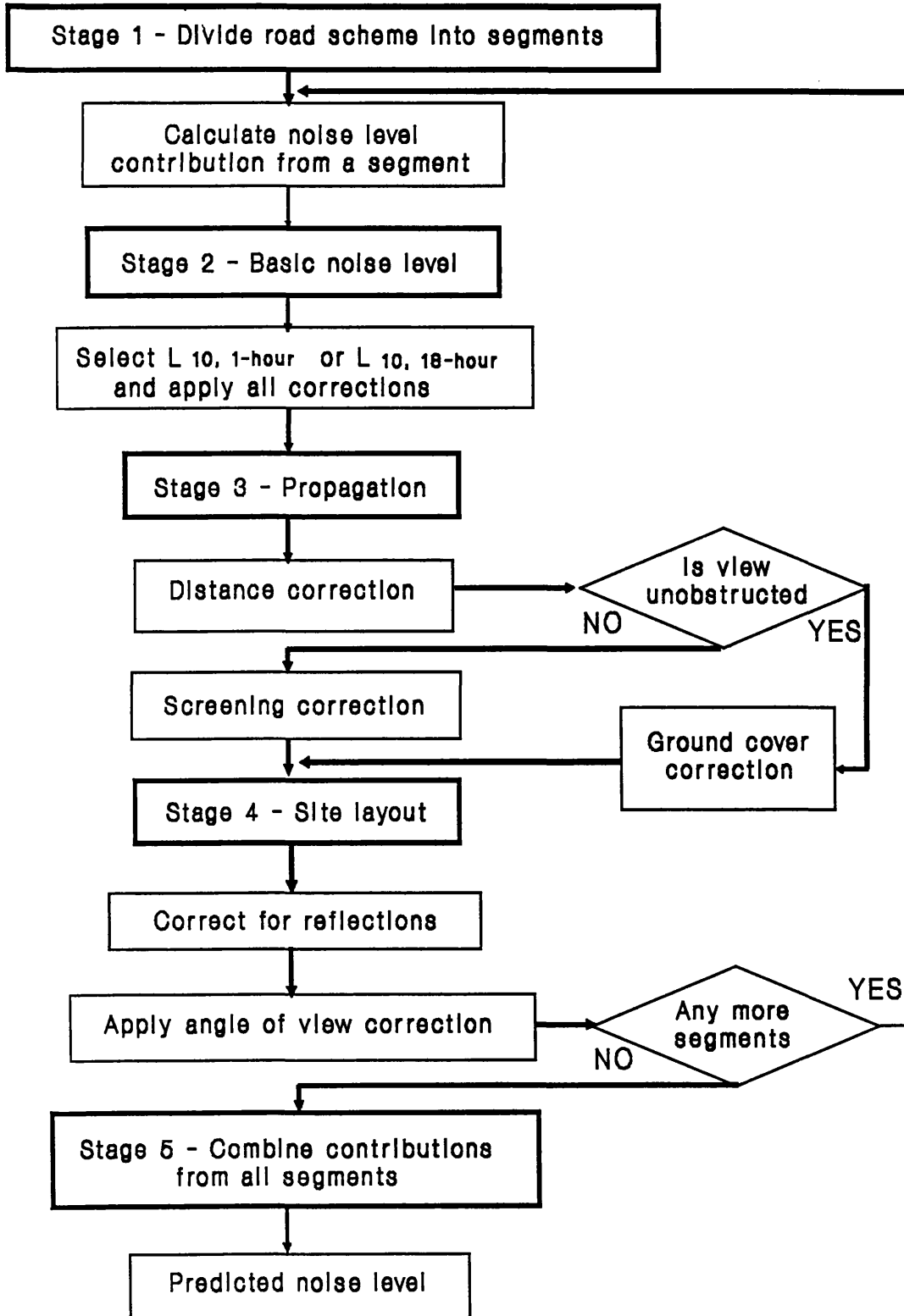
require a considerable amount of measures in the field in order to establish a relationship between traffic flows, sound propagation conditions and noise levels.

Traffic noise from vehicles travelling under free flow conditions has been widely modelled, since 1969. However, relatively little research has been carried out about situations where vehicles are operating under congested or interrupted flow conditions, which are characteristic in urban roads. Gilbert (1977) developed an empirical model for predicting traffic noise, which was pioneering in dealing with urban traffic conditions. Later, Gilbert et al. (1980) produced a more sophisticated method to predict noise levels from interrupted flows.

The Calculation of Road Traffic Noise (CRTN) method was first developed in 1975, and its latest version was published in 1988 by the Department of Transport and Welsh Office. This method provides guidance for the prediction and measurement procedures of  $L_{10}$  traffic noise levels and can be applied to both congested and non-congested traffic situations. The CRTN is the most comprehensive, detailed and accurate method available for predicting noise levels in urban roads and it has been recommended for many years by the British government for the estimation of noise levels from traffic sources. The CRTN method is simple to use, even by professionals without any acoustic knowledge. The expected accuracy of the method is reported in Section 6.3.1.

The CRTN method predicts the traffic noise level at a reception point from combined road sources. Figure 4.4 illustrates the various stages of the method, which are described in more detail below.

Figure 4.4: The prediction of road traffic noise



- Stage 1: division of the road into segments such that the variation of noise within the segment is small.
- Stage 2: calculation of the basic  $L_{10}$  noise level for the period of either one or eighteen hours, as a function of the total traffic flow in that period. A series of corrections are then carried out and added to the basic estimate, in order to account for other factors influencing noise levels, such as traffic speed, percentage of heavy vehicles, road gradient and road surface texture.
- Stage 3: estimation of the propagation correction, which takes into account distance attenuation, screening from the source line and pavement and ground absorption.
- Stage 4: estimation of the layout correction, which takes into consideration the size of the source segment and site layout features including reflections from barriers, building facades and other reflecting or obstructing surfaces.
- Stage 5: combination of the contributions from all segments to give the predicted noise level at the reception point for the whole road scheme.

After the road has been divided into segments (Stage 1), the  $L_{10}$  noise level for the periods of one and eighteen hours is predicted as indicated by Equation 4.6.

$$L_{10}^i = a + (10 \log F) + C_D + C_{SH} + C_F + C_{FO} + C_S \quad \text{Equation 4.6}$$

$L_{10}^i$  = noise level on segment  $i$ , in dB(A),

$a$  = 42.2 for one-hour predictions,  
29.1 for eighteen-hour predictions,

$F$  = Total traffic flow during the one or eighteen hour period (in both directions including heavy vehicles),

$C_D$  = Correction for distance between source and receptor, in dB(A),

$C_{SH}$  = Correction for traffic speed and percentage of heavy vehicles, in dB(A),

$C_F$  = Correction for facade effect, in dB(A),

$C_{FO}$  = Correction for reflection from opposite facades, in dB(A),

$C_S$  = Correction for surface, in dB(A).

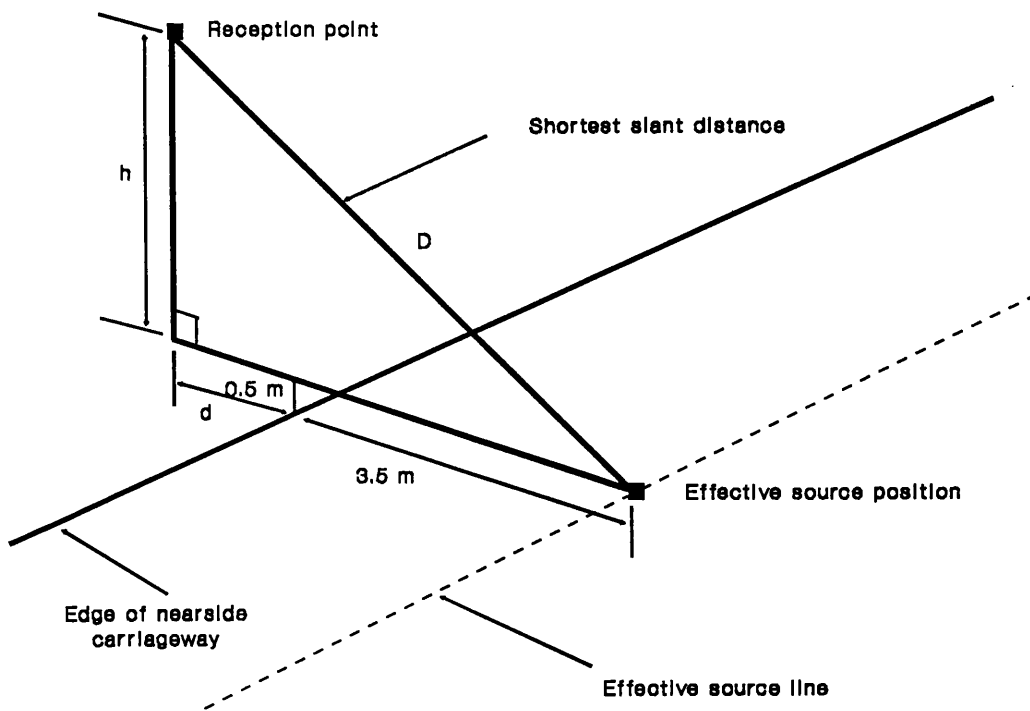
The corrections made to the basic noise level are described in Equations 4.7, 4.9, 4.10, 4.11 and 4.12. The correction for distance from the source line (Stage 3) is given by Equations 4.7 and 4.8 below and the distance parameters are illustrated in Figure 4.5.

$$C_D = - 10 \log \left( \frac{D}{13.5} \right) \quad \text{Equation 4.7}$$

$$D = [(d + 3.5)^2 + h^2]^{\frac{1}{2}} \quad \text{Equation 4.8}$$

- D = Shortest slant distance from the effective source position (m),
- d = Shortest horizontal distance from the edge of the nearside carriageway to the receptor point (m),
- h = height of the reception point relative to the source line (m).

**Figure 4.5: Distance from source line to the reception point**



Equation 4.9 presents the correction for speed and percentage of heavy vehicles in the flow.

$$C_{SH} = 33 \log \left( S + 40 + \frac{500}{S} \right) + 10 \log \left( 1 + 5 \frac{P}{S} \right) - 68.8 \quad \text{Equation 4.9}$$

S = Average traffic speed (km/h),  
P = Percentage heavy vehicles (%).

Equation 4.10 gives the correction for facade reflection, which represents the Stage 4.

$$C_F = 2.5 \quad \text{if receptor is one metre from facade} \quad \text{Equation 4.10}$$

If the receptor is more than one metre from building facades or if there is no facade beyond the receptor point, the reflection effect is not considered. In such cases,  $C_F$  becomes zero and no correction is made. The correction for the angle of view is given by Equation 4.11.

$$C_{FO} = 1.5 \frac{ra}{va} \quad \text{Equation 4.11}$$

ra = sum of angles subtended by all reflecting facades, opposite to traffic stream (degrees).  
va = angle of view in relation to each link (degrees).

Equation 4.12 shows the correction to be applied concerning road surface.

$$C_S = -1.0 \quad \text{in impervious bituminous and} \quad \text{Equation 4.12} \\ \text{concrete road surfaces when speed} < 75\text{km/h}$$

In situations where traffic speed is higher than 75 km/h or the road surface is of any other nature but impervious bituminous and concrete, no surface correction should be made.

The method does not perform well in situations of low traffic volumes. For traffic flows between 50 and 200 vehicles per hour, a correction factor (K) must be added to the predicted noise level. Equation 4.13 expresses the correction factor for low traffic flows.

$$K = -16.6 \left(\log_{10} \frac{30}{D}\right) \left(\log_{10} \frac{F}{200}\right)^2 \quad \text{Equation 4.13}$$

If traffic flows are lower than 50 vehicles per hour, however, then the method becomes unreliable and should not be utilised at all.

For the prediction of noise levels in the vicinity of road junctions, each arm of the junction should be treated as a separate road segment. The combined contribution of noise levels  $L_{10}$  from 'n' segments around a road junction is defined by the expression given in Equation 4.14, which represents the Stage 5.

$$L_{10} = 10 \log_{10} [10^{(0.1 L_{10}^1)} + 10^{(0.1 L_{10}^2)} + 10^{(0.1 L_{10}^3)} + \dots + 10^{(0.1 L_{10}^n)}] \quad \text{Equation 4.14}$$

The major contribution to the overall predicted noise level is made by the segment adjacent to the receptor point.

The predictions of  $L_{10, 18\text{-hour}}$  noise level require the collection of traffic flow data over the 18-hour period, which is usually too labour-intensive to perform. Thus, the CRTN method recommends a shortened measurement procedure which can be used to estimate the  $L_{10, 18\text{-hour}}$  based on the measurements of  $L_{10, 1\text{-hour}}$  made over any three consecutive hours, between 10:00 and 17:00. Using  $L_{10, 3\text{-hour}}$  as the arithmetic mean of the three consecutive values of hourly  $L_{10, 1\text{-hour}}$ , the value of  $L_{10, 18\text{-hour}}$  can be calculated by the expression given in Equation 4.15.

$$L_{10, 18\text{-hour}} = L_{10, 3\text{-hour}} - 1 \text{ dB(A)} \quad \text{Equation 4.15}$$

This shortened procedure is particularly useful to enable comparisons of the results of short-period measurements with the environmental standards, which are set for 18 hour periods.

#### 4.5 Environmental standards for noise levels

The need to alleviate noise pollution levels has long been recognised and has led to legislation in the form of the Noise Insulation Regulations 1975. In addition, several planning policy documents have been issued by the Department of Environment which also emphasize

the need for the effects of traffic noise to be considered at an early stage in transportation plans. The Control of Pollution Act 1974 and the Environmental Protection Act 1990, as amended by the Noise and Statutory Nuisance Act 1993, actually contain powers to control environmental noise nuisance (Department of the Environment, 1994).

Several worldwide organisations have been trying to establish the maximum noise level acceptable as a standard. For instance, the United States Department of Housing and Urban Development established in 1979 regulations for the degree of acceptability of noise levels. A noise level of 65 dB(A) was defined as '*normally acceptable*', the range between 65 to 70 dB(A) was regarded '*normally unacceptable*' and levels above 75 dB(A) were considered '*unacceptable*' (Beranek, 1988).

In the U.K., there is no absolute agreement on the maximum acceptable noise level. The first report on noise in this country (U.K. Government, 1963) made tentative recommendations for noise level standards. It states that indoors noise levels  $L_{10}$  should not exceed 50 dB(A) during day times and 35 dB(A) at night times. The Noise Advisory Council (1980) recommended that residential areas should in no circumstances be subjected to a  $L_{10, 18\text{-hour}}$  noise level of more than 70 dB(A) at one metre from the house facades facing the traffic stream. This noise limit has been widely applied in Britain. The Department of the Environment (1973) also considers 70 dB(A) as the absolute limit of acceptability for road traffic noise in residential developments, although acknowledges that such level is not desirable. The Land Compensation Act 1973 enabled the Noise Insulation Regulations 1973 and 1975 to be made for providing insulation against increased noise at or above the level of 68 dB(A), measured as the  $L_{10, 18\text{-hour}}$  index (Nutley and Beaumont, 1974; Acton et al., 1984). The Royal Commission on Environmental Pollution (1994) has recently proposed to reduce exposure to noise nuisance from transport, measured as  $L_{eq}^5$ , to not more than 65 dB(A) during day-time and 59 dB(A) at night-time, at the external walls of housing. None of these limits are mandatory, as they were set purely as guidelines. However, many local authorities have chosen to provide noise control measures for levels in excess of 68 dB(A) (Bell, 1990).

The standards for noise levels should perhaps have been defined differently for distinct land

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<sup>5</sup>  $L_{eq}$  is the energy level equivalent to a measured fluctuating sound under consideration over a given period.



uses. At places where peace and silence are most expected and appreciated, such as schools, churches, libraries, hospitals, noise limits should be more rigorous, especially at night time. Noise limits should also be sensitive to other land uses where quietness is also important, such as residential areas, hotels, leisure and recreational areas, parks and other open spaces. Tentative limits for noise levels  $L_{10}$ , have been contemplated by an academic research. Table 4.3. illustrates various noise level limits according to different land uses and standards.

**Table 4.3: Suggested noise limits for different land uses and standards**

| Land use                  | Noise level (dBA) |                 |              |                   |
|---------------------------|-------------------|-----------------|--------------|-------------------|
|                           | high standard     | medium standard | low standard | very low standard |
| Residential               | 66                | 70              | 73           | 74                |
| Shopping                  | 72                | 75              | 78           | 81                |
| Offices                   | 68                | 72              | 76           | 80                |
| Open space and recreation | 66                | 70              | 74           | 78                |

Source: Chadwick (1989).

Although the setting of suggested limits as given in the above table seems to represent a very interesting approach, these limits have not been applied in practice. Further research is necessary to better assess the acceptable noise limits for different land uses under various standards.

## **5 MONETARY VALUATION OF THE ENVIRONMENTAL IMPACTS OF TRAFFIC**

A diagnosis of the best use of scarce public funds together with an estimate of the likely social costs and benefits may be essential for supporting decisions on the approval of projects. A rational decision is supposed to weigh up the advantages and disadvantages involved in the decision-making process, and to whom the diverse costs and benefits accrue. In order to enable a range of estimated effects from a change in the transport system to be taken into account, it is desirable for the various effects to be summarised in an assessment procedure. The elementary but useful proposition of cost-benefit analysis (CBA) seems the most appropriate framework for assessing the diverse aspects of traffic schemes altogether. It is based on the common sense concept of rationality (Pearce and Nash, 1981), it presents a clear and rational description of the accounted effects and it is considered to be a measure of overall efficiency in the use and allocation of economic resources (Pearce, 1976). There are few practicable alternatives to CBA, and the COBA manual (Department of the Environment, 1971) adopts it to assess the costs and benefits from road traffic schemes.

The optimum balance between the overall costs and benefits is achieved through an economic evaluation of all outcomes of a scheme on a single scale. The most useful scale for integrating all diverse dimensions of evaluation into a unique measuring rod is money, which is the natural unit of markets. Environmental economics provides the basis for monetary valuation of environmental assets (Pearce, 1990, OECD, 1992). Environmental valuation has become important in transport planning decisions for appraisals of projects and schemes. It can play a relevant role in the design of policies which seek sustainable development (Pearce, Markandya and Barbier, 1989), as well as for an efficient management on the use of natural resources. It provides information that may assist in decisions concerning environmental regulation and preservation. It is also valuable for establishing cost responsibilities, road user charges and road pricing, whenever environmental degradation is an issue.

The use of money as the standard measuring rod for comparative assessments has a variety of attractions as an aid to decision making. One of the most significant advantages over the qualitative approach is the allowance of great consistency in benefit-cost assessments. This enables environmental considerations to be incorporated into economic appraisals, reducing the risk of subjective decisions. The extent to which transport schemes are worth being implemented, or when it is the case, preserving and improving the environment is socially

and economically worthwhile, can be indicated by the monetary outputs. Monetary valuation is a valuable tool in decision making, but it will not replace the need for judgement in CBAs.

The rationale of economic valuation for environmental attributes can be derived conceptually. Although environmental goods, assets and amenities are finite and scarce resources, they usually have no explicit prices in the market, since no trade is made. Nevertheless, according to the economic theory, value arises whenever satisfaction is derived to individuals. When environmental attributes are considered a source of satisfaction, certain intrinsic economic values can be associated to them. Hence, monetary value arises on social grounds, as long as a want or desire is satisfied or a preference is met.

However, there are non-marketed components for which monetary valuation is not trivial and may depend upon extensive economic and theoretical basis. The translation into money terms can even be impracticable in some cases. The normative significance of environmental valuation may be highly questionable, if it fails to reflect realistically social-economic costs.

Monetary valuation is considered an advance for the most sophisticated and comprehensive cost-benefit analyses, but it must be utilised carefully in order to ensure a proper representation of the environmental costs produced.

### **5.1 Economic instruments on environmental damaging activities**

A great deal of human activities imply that one good is transformed into another. This process very often generates damage to the environment. Environmental damage involves complex social costs, which directly or indirectly have been or will have to be paid by someone.

According to the 'polluter-pays' principle<sup>6</sup>, the cost of environmental preventive or corrective measures should be inserted in the cost that the production and consumption of goods and services impose on society (Pearce, 1989). Thus, the polluting activity would pay for all the resources it consumes, like any other economic activity in the market, plus the marginal damage generated. The implementation of economic instruments should act as a regulatory framework to control environmental damage and is likely to create an environmental

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<sup>6</sup> This principle is also known as 'environmental charge' and 'cost-responsibility'.

awareness.

Monetary valuation for negative externalities allows a proper setting of economic instruments on environmental damage. The existing economic instruments for controlling environmental pollution are taxes, charges or subsidies (Pearce, 1976). Subsidies are financial assistance for activities that provide incentive to ecological preservation. Contrarily, taxes and charges are imposed upon polluting activities<sup>7</sup>. These can be conceived as economic instruments for preventing misuse of scarce environmental resources, or alternatively as suggested by Baumol and Oates (1971), as the price for the private use of social resources.

The optimum level of taxes and charges should be equal to the monetary value derived from the environmental damage generated. This level should be settled high enough to provide a sufficient incentive for polluters to reduce their discharges so as to achieve a desired ambient quality within a specified time period, but not so high as to increase the activity costs to undesirable levels. This can be settled by trial and error, firstly setting a charge and observing its effects, and then adjusting it up or down until the ambient quality reaches the desired goal (Anderson et al., 1977). The taxation system would be most effective if it allows prices to be adjusted in order to encourage producers and consumers to operate constantly in an environmentally friendly manner. The payment of charges should be economically less advantageous to the polluter than investments or measures to reduce environmental impacts, otherwise, contrarily to the implicit objectives, the polluter may prefer to pay and continue polluting. In this respect, economic instruments should allow the trade of pollution rights<sup>8</sup>.

The obligation to pay for the environmental harm produced provides an incentive not to cause that harm (Anderson et al., 1977). Firms will choose to pay the charge or invest in new and less environmentally damaging methods of production. The development of cleaner technologies would be promoted, as demand for less polluting processes increases. The public

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<sup>7</sup> *Effluent charges* are based on the quality or quantity of discharged pollutants. *User charges* are payments for the cost of treatment of pollutants. *Product charges* are incident on the price of products which pollute in the manufacturing or consumption. *Administrative charges* represent payments for implementation and enforcement of regulations.

<sup>8</sup> Some authors propose the Government to sell 'tradeable pollution permits' to individuals or companies. Those who exceed the target environmental damage reduction can trade their excess permits to others who not managed or wished to comply with regulations. This would introduce an element of flexibility into the market-based instruments. Further discussions on this subject can be found in Hughes (1992) and Pearce, Markandya and Barbier (1989).

should be informed of the magnitude of pollution caused by production and how much extra is incurred on anti-pollution costs.

Evidences presented by Pearce, Markandya and Barbier (1989) suggest that market-based approaches to pollution control can be highly cost-effective. The main aim of this principle is to alter behaviour, but it can also raise revenue (Pearce, 1989). The revenues raised may be reverted into funds for preventing further environmental degradation or used where they are best needed, provided they are used to achieve the objectives for which the charges were conceived.

In relation to transportation, the price actually paid for dislocations does not yet reflect such inherent costs. In this sense, transport policy has to decide whether the transport users themselves should pay for the external social costs incurred from the use of environmental resources, or if the whole society should eventually shoulder them. The compensation for the cost, damage or disutility generated by road traffic pollution can be imposed through measures to prevent or discourage car usage, such as road user charges, road pricing, fuel taxation or subsidised fares. The charges on environmental impacts could help to reduce vehicle miles travelled, by encouraging the use of more environmental-friendly modes of travel. They can promote a shift to public transport and discourage unnecessary trips, if set high enough to deter car journeys. These charges should regard appropriately all external effects within and outside the transport market. Such instruments should reflect not only the manageable pollution costs but also the implicit costs of irreversible effects, like ozone layer depletion or the greenhouse effect.

In the U.K. the differential between the tax on leaded and unleaded petrol can be seen as an example of a market-based instrument, where the polluter pays more for more harmful emissions. The result of such a measure introduced in 1987, together with the wider availability of cars suitable for unleaded fuel, was the rise in the sales of unleaded petrol to about 35% of the market in three years (U.K. Government, 1990) and the drastic reduction of lead concentrations throughout the country. In Sweden, economic instruments were introduced into the environmental policy in 1988 in order to attribute social marginal responsibility charges for road and rail air pollutant emissions (Hansson, 1990 and 1991). Another example of economic instruments on transportation can be found in Germany, where tax reductions are given for catalyst equipped vehicles and tax on leaded petrol has been differentially rising since 1989 (Blum and Rottengatter, 1990).

Despite the very sensible theoretical grounds for economic instruments on environmental damage, there are several practical difficulties for implementing such principles. The incorporation of environmental costs into polluting activities tend to make them more expensive, and the consumer would end up paying for the damage produced. Thus, taxation raises concern about economic growth, inflation and social equity, which must be outweighed in the balance.

## **5.2 Techniques for environmental valuation**

The techniques for estimating monetary values for environmental attributes are still at an early stage of technical development and public acceptance. The existing techniques are subject to great deficiencies and difficulties of application. The more disaggregation that is attempted the greater the difficulties become.

The first official reference to environmental valuation included in the U.K. Government guidelines was provided by H.M. Treasury (1991). The Department of the Environment (1991) presents more specific guidelines on valuation techniques, which offers advice as to ensure that significant environmental effects are fully considered in policy appraisals. Further, the Standing Advisory Committee on Trunk Road Assessment (Department of Transport, 1992a) supports the idea that monetary techniques will assist the assessment of the various environmental effects from transport schemes. The Department already includes some proxies to the expected monetary costs from environmental effects on its cost-benefit assessment, like mitigation measures and compensation payments.

A number of methods and techniques have been developed in an attempt to secure money measures for diverse non-priced environmental attributes, such as goods, assets and amenities. The existing approaches search for distinct dimensions of the underlying economic theory for incorporating environmental evaluation into economic analyses. Many applications of these techniques have been carried out for actual money estimates (the following sections report the most important applications of the various techniques to different environmental effects).

The idea of attaching monetary values to environmental damage is relatively recent and the world experience in doing so is still limited. Nevertheless, many attempts have been made for deriving a proper monetary value, price or cost for environmental attributes. Results from the practical experience of environmental monetary valuation in some developed nations are

widely available in the literature (e.g. Barde and Button, 1990; Barde and Pearce, 1991; Danish Ministry of Energy, 1991; Coker and Richards, 1992; Navrud, 1992; Banister and Button, 1993 and European Conference of Ministers of Transport, 1994). Data for total annual costs from pollution damage and for the benefits of pollution control have also been derived (e.g. Pearce and Markandya, 1989). Strong evidence suggests that the valuations of external costs actually represent considerable monetary sums. The efforts in Germany and the United States towards monetary valuation have given outstanding contributions to the state-of-the-art. A modest number of studies in Scandinavia have also resulted in valuable experience.

The application of monetary techniques is not trivial and presents a number of practical problems. They inevitably rely upon large number of assumptions, hypothetical statements, or complex, extensive and expensive data requirements. As a result, even the best technique and data base available are unlikely to generate accurate and reliable estimates. Hence, it is difficult to evaluate how successful the existing attempts have been to attribute value to environmental attributes.

Environmental attributes can be valued monetarily through distinct techniques. Since different techniques have proper characteristics, requirements, advantages and disadvantages, it is important to determine which ones are able to carry out the type of valuation needed<sup>9</sup>. The success of valuation depends a great deal on the choice of the technique, which should consider its suitability for each specific application. It must be borne in mind, however, that both the conception and application of monetary techniques usually present several practical problems.

Some assets can be fully valued through a particular technique, but usually no general technique is capable of reflecting a comprehensive valuation of all the diverse degradation effects. Complementary valuation may be recommended in situations where the application of distinct techniques can be combined in order to estimate the total valuation<sup>10</sup>. Results

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<sup>9</sup> A summary of advantages and limitations of the principal monetary techniques can be found in Turner and Bateman (1990) and in OECD (1992).

<sup>10</sup> Additional care should be taken in adding the values associated to effects from different techniques, as an element of double counting is likely to be introduced into the analysis. In some situations, however, this apparent overestimation could be considered an offset of the omission of other less tangible factors.

from different techniques can also be compared in order to ascertain the consistency and accuracy of the valuation process. However, this might cause a significant increase in the application costs.

The literature reveals a wide range of concepts, classifications and terminologies in relation to the monetary techniques for environmental valuation. Five main techniques have been identified as appropriate for valuing the impacts produced by road transport. These are: damage costs, avoidance costs, compensation costs, surrogate market and preference surveys. Several detrimental effects arise from traffic air pollution and noise. After careful appraisal of the literature, four predominant effects have been identified, namely: noise nuisance, damage to health, deterioration of the living environment and damage to materials<sup>11</sup>. Table 5.1 shows a summary of the suitability of the techniques available to value each of these predominant detrimental environmental effects. The entries in Table 5.1 are a result of the author's appraisal of the level of applicability of the monetary techniques, which is described in more detail along the next sections.

**Table 5.1: Suitability of techniques to value detrimental effects**

| Technique          | Noise          | Air pollution    |                              |                     |
|--------------------|----------------|------------------|------------------------------|---------------------|
|                    | noise nuisance | damage to health | deterioration of environment | damage to materials |
| Damage costs       | no             | yes              | no                           | yes                 |
| Avoidance costs    | limited        | no               | limited                      | no                  |
| Compensation costs | limited        | limited          | no                           | no                  |
| Surrogate market   | yes            | no               | yes                          | no                  |
| Preference surveys | yes            | yes              | yes                          | no                  |

The following sections present a review of the principal techniques useful for deriving values to the predominant environmental effects produced by the provision of road transport.

<sup>11</sup> There is a range of regional (e.g. acid rain and damage to crop production, fauna and flora) and global effects (e.g. global warming), which are not included in the scope of this work as they provide much greater difficulties in terms of valuation, although their relevance to some aspects of transport planning is acknowledged.



### *5.2.1 Damage costs*

As stated previously, any productive system can potentially impose damage on certain segments of society, and any damage can be directly or indirectly associated to costs. The damage caused by pollutant exhausts and noise levels from traffic includes loss of amenity, costs of materials, health, local and global environment, fauna and flora. The damage costs incurred may be used as a proxy measure for economic valuation.

The damage cost technique is the most comprehensive and absolute within the existing mechanisms for environmental valuation. It makes straightforward use of market prices, where all incurred costs are accounted by the 'enumeration-and-valuation' methodology. Thus, the monetary assessment of damage costs requires a series of estimations:

- the amount of physical environmental degradation and subjective annoyance produced.
- the quantity of material, size of population and environmental resources exposed to different levels of impacts.
- the extent to which exposure affects people's satisfaction, health, materials, plants, animals and other resources.
- the money value associated to each unit of degradation or damage.

The determination of the amount of damage produced is relatively straightforward, since measurement techniques are well developed. However, while some negative externalities can be easily quantifiable, others less tangible or more subjective may provide great difficulty for quantification and even identification. This technique is likely to provide underestimated results if some elements cannot be properly accounted for and are missing from the calculation.

The number and the nature of activities, people, goods and other factors involved in the process of damage can be so great that estimates may become subject to a high degree of uncertainty. Besides, the damage originating from traffic air pollution can be quite difficult to separate from the damage produced by other sources.

In relation to the effects of exposure, these can be determined by damage functions or dose-response relationships. However, dose-response relationships are neither simple or fully understood, because of the complex interactions which are usually involved in the process of cause and effect. Relationships as such have not yet been sufficiently developed for most environmental effects, and there are still considerable gaps in knowledge about the adverse effects of exposure levels of pollution on human health and material deterioration.

Considering the substantial complexity in estimating the previous stages of the 'enumeration-and-valuation' technique which account for the total environmental damage, there is much less prospect for the evaluation of the financial marginal costs of the environmental degradation produced.

The damage costs technique may depend on a number of assumptions about maintenance. Repair prices may vary immensely, according to the quality of materials and labour employed. Further assumptions have to be made about the period for which polluting activities have contributed to the damage since it is most likely to be felt over a number of years, due to the gradual and cumulative effect of pollution. Current and past environmental degradation are also an inheritance to the next generations. This includes the additional complexity of how to evaluate the cost produced by an effect that will take place in the future and of which not enough is known about its consequences (Hansson, 1991). The estimation of costs related to irreversible effects acquires a further degree of complexity.

The most important monetary consequences of pollution imposed on society are the associated health diseases. Although in theory damage cost is a comprehensive and detailed technique to assess the health damage caused by air pollution, there are several practical problems involved with its application. The main difficulty is that empirical work in this area is limited and, as indicated by Pearce and Markandya (1989), sophisticated data required are very rarely available.

The overall costs related to health damage have been estimated through the damage costs technique and the figures are alarming (Rice, 1966; Cooper and Rice, 1976; Danish Ministry of Energy, 1991). The economic burden of health damage on society has been estimated by accounting the incurred costs on hospitalisation and health care (Silverman, 1973; Cooper and Rice, 1976; Bhagia and Stoevener, 1978 and Shechter, 1992), morbidity and productivity reduction (Cooper and Rice, 1976; Lave and Seskin, 1977; Ostro, 1983 and Kneese, 1984)

and mortality (National Academy of Sciences, 1974; Cooper and Rice, 1976; Lave and Seskin, 1977; Hartunian et al., 1980; Schwing et al., 1980; Kneese, 1984; Pearce, Markandya and Barbier, 1989; Turner and Bateman, 1990; H.M. Treasury, 1991; Danish Ministry of Energy, 1991 and Small and Kazimi, 1995).

Empirical data are required to produce damage functions relating a particular marginal health effect of air pollution to measures of air quality, which are indicators of actual exposure. However, current observations of ambient quality and exposure to air pollution are not adequate indicators of the actual level of degenerative diseases or death, as air pollution usually provokes long term and cumulative effects. For example, respiratory cancer can be caused by exposure to carcinogenic substances over a prolonged period of time and, according to Kneese (1984), it may occur as much as two decades after exposure. Therefore, estimates of the number of diseases and deaths due to air pollution may require extensive historic data collection.

The main problem with estimating health damage is the absence of information on the increase of severity of existing diseases and non-reported incidence of illness. Epidemiological studies only investigate the health effects from normal living exposure to pollution, so they provide difficulty in distinguishing the contribution of diverse sources to health problems. Clinical studies can only account for physical or psychological effects, as it is not possible to expose people deliberately to pollutants over long periods of time or at concentrations that are likely to result in harmful effects. Horowitz (1982a) reports a number of dose-response relations between concentrations of diverse pollutants and clinical symptoms.

A number of estimates have been made on the costs incurred from diseases of the respiratory system (Cooper and Rice, 1976; Lave and Seskin, 1977; Ostro, 1983 and Kneese, 1984), but no concrete evidence has been found in the literature about the number of life years lost per patient specifically due to air pollution-induced cardiovascular conditions. Limiting health damage to respiratory cancer can underestimate results. The estimation of earning losses through this technique strictly evaluates the sickness effects on the productive population. Thus, the health effects on non-productive people, such as elderly, housewives and children, who are usually the most strongly affected, are not accounted for. In the case of death, the monetary loss can be estimated from the net value of the expected lifetime earnings.

The damage cost technique is also the most suitable procedure for estimating in monetary terms the harm caused by air pollution to materials. The economic costs incurred from material damage can reach considerable amounts (National Academy of Sciences, 1974). Nevertheless, little practical research has been found in the literature on the costs of material damage. Most of the existing empirical studies have concentrated on the effect of sulphur compounds and particulate matter on specific materials and buildings (Michelson and Tourin, 1966; Ball, 1984 and Jeanrenaud et al., 1993). Fewer studies have focused material damage on a disaggregate level and as suggested by Winpenny (1991), damage on only part of the spectrum of material at risk has been researched so far. Assessment of material damage is complex due to the threshold levels of deposition, before which corrosion may be neglected and after which it may be accelerated.

### *5.2.2 Avoidance costs*

The avoidance costs technique applied to environmental valuation refers to the expenditure incurred from various preventive or corrective measures taken to eliminate or mitigate environmental externalities. The need for such expenditures indicate the minimum valuation of satisfaction or benefit gained from adopting measures for environmental improvements (Starkie and Johnson, 1975). Examples of corrective costs include the expenses from the implantation and maintenance of noise barriers and double glassing to reduce indoors noise levels, and catalytic converters to reduce pollutant emissions. This technique is more likely to be applied purely on a theoretical basis. The derivation of monetary valuation relies upon the available technology.

The practical viability of this approach is limited to cases where individuals spend money to offset environmental hazards. Thus, the total avoidance cost would fail to capture cost where remedial measures cannot be implemented either for technical, political or personal reasons. Another problem with the conception of this technique is that the satisfaction gained from the implementation of avoidance devices may be perceived to exceed or not to reach the actual damage costs incurred. Besides, the extent to which these costs are actually accepted by individuals is difficult to estimate (World Bank, 1991).

The avoidance cost technique has been applied to derive the value of noise nuisance through the costs of implementing sound insulation devices in buildings. The insulation costs were conceived as the monetary values necessary to provide the benefit of reducing the disbenefits

suffered from indoor noise nuisance (Starkie and Johnson, 1975). Insulation costs depend upon a number of variables, such as type and size of the windows, material employed (aluminium, PVC, etc.) and type of insulation (simple secondary glassing panels on existing windows or replacement for purpose built double glassing windows).

This technique has also been applied for the monetary valuation of the deterioration of the living environment provoked by traffic air pollution in Sweden (Danish Ministry of Energy, 1991). The estimates of the disbenefits and loss of amenity caused by traffic air pollution (like odour, smoke, visibility reduction and the spoiling of aesthetics) were derived through the costs of implementation and annual replacement of technical devices for emission control, such as catalytic converters, in private vehicles. Estimates on the costs of purchasing catalysts can be found in Ball (1984), Schwing et al. (1980) and Danish Ministry of Energy (1991).

### *5.2.3 Compensation costs*

The compensatory costs represent the set of expenditures necessary to compensate monetarily the harm caused by an imposition or an increase of environmental degradation. This principle is most commonly applied on assessments of road construction, from which properties lose amenity or have to be demolished. The application of this technique usually provides information only on a small segment of the population.

In specific cases, the value of the inconvenience caused by noise nuisance can be derived from the amount of compensation due, for instance, to workers who have become hard of hearing due to noise at workplace or to residents who have lost amenity due to indoor traffic noise nuisance. The sum of money required to make residents as satisfied after the imposition of a noise nuisance as they were before has been assessed by Plowden and Sinnott (1977).

Valuation for damage to health and loss of life can also be estimated through the amount of compensation demanded as a result of exposure to air pollution or through the compensating wages in high accident or death risk jobs. The costs from pain and suffering associated with respiratory injuries imposed by air pollution have been evaluated by the National Academy of Sciences (1974).

#### 5.2.4 Surrogate market

Individuals express daily several preferences which originate decisions for or against goods and services to be exchanged for money. Surrogate pricing methods use the information revealed by actual market transactions on the purchasing price and demand of goods, assets or services, to infer indirectly the money value of environmental attributes which do not have market prices.

The consequences of changes on environmental quality can partly be expressed in the market values through an approximate value of a change on the potential welfare (Pearce, Markandya and Barbier, 1989). The surrogate market approach is based on the realms of economic theory. The main techniques based on this approach are *property prices* and *travel costs*.

The property value technique is conceptually based on the hedonic price theory, in which the monetary value associated to some environmental conditions can be derived from the demand for housing. Thus, the disbenefits of environmental degradation can be felt on the housing market through the marginal reduction provoked in the value of land, rent or property.

This technique is capable of estimating the proportion of the difference in property prices separately associated with differences in environmental quality at distinct situations. An academic illustration for this technique is the hypothetical situation of two properties identical in all aspects, except in relation to a specific environmental asset. The difference between the prices of these properties indicates the valuation of such asset. This situation, however, is unrealistic since a large number of factors may influence the property market price.

This technique usually applies multiple regression in which data are taken either on a small number of properties over a period of time (time series) or on a larger number at a point in time (cross section). Best results are obtained when properties are in the same geographic area, with similar household, accessibility and neighbourhood characteristics.

Although technically feasible, the main deficiency of this technique lies in its very conception. Since house market prices reflect a very large range of attributes, it is extremely difficult to isolate the influence of a specific impact on the price. This technique is only applicable to impacts which are likely to be capitalised into property values. The consideration of more than one environmental attribute on the property value is even more

complex. This method is not suitable for long term analyses due to the likelihood of further factors affecting the property market. It also works poorly for unclear effects of environmental degradation. Another great disadvantage of this approach is that, in practice, the necessary data on house price changes are almost impossible to be regularly obtained. It depends on an adequate sample of market transactions, but usually only a very small percentage of the total housing stock is exchanged at the same period.

This technique is more appropriate to reflect the environmental losses from the implantation of projects than to value marginal disbenefits caused by an increase on traffic flows. This technique would achieve best results if an extensive degree of mobility existed, because in reality, few households move for an increase in environmental degradation. Besides, this method does not assess the nuisance suffered by non-property owners, visitors, casuals and local workers. Pearce, Edwards and Harris (1979) claim that this method almost certainly is not capable of estimating environmental valuations.

Despite the problems relating to the application of this technique, several studies have derived monetary values for the environmental impacts of transport.

The presence of air pollutants in the atmosphere has proven to induce property depreciation. Several studies have found significant correlation between fall in property values and increase air pollution levels in urban areas: Freeman (1979), Schulze et al. (1981), Brookshire et al. (1982), Chestnut and Violette (1984), Shechter (1992) and Römer and Pommerehne (1992).

Extensive experience has also been gained by valuing the capital loss from a fall in property prices as a consequence of an increase in noise nuisance. The property value approach on noise nuisance has been applied most extensively to the effects of aircraft noise (Commission on the Third London Airport, 1970 and 1971, Walters, 1975; J. Nelson, 1980; O'Byrne et al., 1985). Noise nuisance produced by aircraft is not equivalent to that caused by traffic. In fact, evidence on the literature suggest the former to be far more intense, due to the characteristics of intermittence and high frequency of aircraft noise emissions. Valuation of road traffic nuisance, although a little more scarce, has also been consistently derived (Pearce, 1977; Nelson, 1982; Römer and Pommerehne, 1992; Navrud and Strand, 1992; Jeanrenaud et al., 1993). Most studies on the valuation of traffic noise nuisance produce estimates on the percentage of property price change for each marginal decibel variation in traffic noise.

The other technique based on the hedonic price approach is *travel costs*, which assumes that the costs incurred by travelling can be taken as a proxy for the monetary value of an amenity. This technique relies on the observed behaviour of people, by assessing the extent to which travellers are willing to trade their journey length or time, in order to avoid stress and bother, or to gain satisfaction from a more attractive and pleasant route (Rendel Planning and Environmental Appraisal Group, 1992). This technique does not require a great deal of information, relying on the results of surveys on the time and expenses from travelling. On the other hand, its applicability field is rather restricted to situations in which travel costs can reflect the demand for environmental attributes. It is difficult to capture this influence from multi-purpose trips.

### 5.2.5 Preference surveys

The experimental approaches for monetary valuation simulate a market by placing respondents in a position in which they are asked to express their individual subjective preferences of perceived aspects in their welfare, utility or satisfaction. Preference survey techniques are, therefore, means of measuring preference on non-marketed goods. The monetary values for the predominant external effects from road traffic can be estimated through distinct experimental survey techniques, namely: *contingent valuation*, *stated preferences* and *transfer prices*.

Contingent valuation methods (CVM) or revealed preferences techniques are derived from a social sample interview which assesses the maximum '*willingness to pay*' (WTP) for improving the ambient conditions or for mitigating environmental deterioration down to a certain target level. Alternatively, it can be based on the '*willingness to accept*' (WTA) compensation for a loss, at the very least amount. Thus, CVM are based on the principle that, if people are willing to pay a certain amount for a non-priced benefit or accept a certain compensation for a specific disbenefit, then this attribute is supposed to be worth at least the amount stipulated. In practical experiments, bids for improving the environment have proven to be lower than those to prevent further deterioration. This is contrary to the economic theory, which suppose WTP and WTA valuations to be equal. The divergence between practice and theory can be explained by the fact that individuals tend to dislike more the idea of losing an attribute already owned than acquiring further satisfaction. Besides, WTP respondents tend to underestimate the amount they would be prepared to pay for the attribute under consideration whereas WTA respondents are strongly inclined to overestimate their



demands (Cummings et al., 1986).

The WTP criterion was one of the first means used for valuing time savings. It is indeed consistent with the general theory of market economics and it is a clear market test of user preference (Howe, 1971). This technique is comparable to psychological tests, and it still is in many ways the most direct and simple method to understand. Interview techniques provide great versatility, because they are potentially capable of valuing a broad range of environmental attributes, which can be directly and separately addressed. Besides, they are sensitive enough to capture diverse attributes which are subjective and have no natural unit of measurement, such as annoyance, disturbance, risk, fear and other psychologic effects, and usually cannot be valued by traditional measures of individual benefit. Thus, CVM provide an extraordinary freedom for environmental valuation and do not depend on extensive data base, like most techniques. Several variations of this method have been developed (Dixon et al, 1988 and 1994). The Department of Transport (1992a) considers these techniques as "*the greatest early scope for monetary valuation*" and Mitchell and Carson (1989) support the idea that CVM are the most powerful and versatile tools for valuing non-market goods.

CVM have proved potentially capable of providing important information about environmental valuation, as long as the effects are clearly perceptible. As stated by Langdon (1978), "*monetary evaluation by direct questioning is feasible in practice, yields actual results and produces values not inconsistent with those obtained by means of different methods*". Cummings et al. (1986) and Bishop and Heberlein (1986) support that properly carried out CVM give meaningful values for environmental goods and produce results with an accuracy of  $\pm 50\%$ . Such an approximation is considered reasonable for the nature of the experiment and magnitude of the values involved. Bishop and Heberlein (1986) acknowledge that "*while CVM appear to be biased even under the best of circumstances, the degree of bias does not appear to be sufficiently high to rule out the use of the results in public decision-making*".

Contingent valuation is the most appropriate technique to assess monetarily the marginal noise nuisance from traffic. A good deal of research has been done on the contingent valuation of peace and quiet in monetary terms (Starkie and Johnson, 1975; Langdon, 1978; Römer and Pommerehne, 1992; Baughan and Haddart, 1992).

The valuation of health damage has also been estimated through the amount people are

willing to pay in order to avoid all the losses associated with illnesses (such as incurred costs, pain and suffering), averted morbidity or reduced mortality risks. Some researches have been carried out to analyse how people value their health conditions, mainly in the United States (Loehman et al., 1979; Chestnut and Violette, 1984 and Gerking and Stanley, 1986).

Further and more recent attempts have been made through CVM to assign economic values for deterioration of the living environment due to air pollution. The developments in this matter include the works done by: Dorfman (1977), National Academy of Sciences (1974), Schulze et al. (1981), Brookshire et al. (1982), Chestnut and Violette (1984), Shechter, Kim and Golan (1989), Navrud (1991), Shechter (1992) and Navrud and Strand (1992).

Despite the revealing promise of survey techniques, a large part of the literature on CVM is taken up with discussion about their accuracy problems. Since CVM are based on individual statements of revealed preference they do not necessarily reflect social costs. WTP bids are usually overestimated, since respondents do not have, in reality, to pay the amounts stated. There are further numerous practical and theoretical issues which need to be carefully addressed on their design and application, in order to reduce the serious threat to CVM's validity. Monetary valuation through CVM depends on the extent to which respondents properly realise the effects of environmental pollution and are able to value them. In many cases, they might not capture the full and precise values and preferences of non-marketed effects, and only a broad range of the plausible valuation may be estimated. CVM are largely vulnerable to misuse and biases.

An important variation on the preference survey techniques is the *stated preference*. It has been shown that people find it easier to order their preferences than state directly how much they would be willing to pay for environmental attributes (Hopkinson et al., 1990). The respondent is initially provided with a base level of an environmental attribute and is offered an alternative in which the attribute is increased, but at a price which is varied until the respondent sees no advantage in an alternative over the other (Dixon et al., 1988 and 1994).

Questions can be asked directly about the stated preference among a number of hypothesized choice scenarios, on the basis of multiple combination of differing levels of environmental attributes, time and money inherent to each alternative. Alternatives are ranked or chosen through the observation and comparison of options, and by making trade-offs of monetary values placed on the non-monetary attributes (Fowkes, 1991). However, this variation of the demand-revealing techniques is strongly dependent upon the choice set of scenarios available

to individual respondents (Bates and Roberts, 1983). The quality of results also depend on the extent to which the structured alternatives represent real choices.

The last variant of preference techniques is *transfer prices*. The transfer prices survey techniques are based on the amount by which the cost of one option would have to be varied to equalise its overall attractiveness or disutilities towards that of another predefined alternative (Gunn, 1984). Respondents are asked to assess the indifference point in the choice, through the amount that the cost of a preferred option would have to increase in order to induce them to switch to a rejected alternative (MVA Consultancy et al., 1987 and Broom et al., 1983).

### 5.3 Conclusion

The provision of road traffic produces a range of costs to the system, some of which are not easy to evaluate. In particular, the derivation of the costs originating from the environmental impacts still represent a major challenge for cost-benefit analyses.

This chapter has presented the state-of-the-art concerning the monetary valuation of the environmental impacts. The five techniques discussed in this chapter constitute the best existing devices to translate environmental damage into monetary terms, as far as the impacts of traffic are concerned. Each technique is consistent with a different underlying aspect of the economic theory, and hence, presents particular advantages and disadvantages for practical applications, as reported in this chapter.

Despite the many practical problems involved with monetary valuation, techniques have become more and more accepted and a number of attempts have been made for deriving proper values for environmental damage. Techniques have been extensively applied to a wide range of situations, the most important of which have been reported in the preceding sections. Although better reliability would be desirable, rough estimates may be valuable for policy decisions in many situations and represent an advance in relation to usual assessment procedures. Further progress in the practical application of the theoretical grounds of environmental valuation is expected in the near future, as the importance of the environmental dimension continues to increase in the overall appraisal of transport schemes.

A procedure which assesses the economic and environmental costs incurred from the

implementation of a range of transport policies was created for the purposes of this research. The literature review reported in this chapter forms the theoretical basis of the environmental dimension of the assessment procedure, and some of the results of international applications of monetary techniques, also mentioned in this chapter, will be adopted in this work as the cost parameters for environmental degradation.

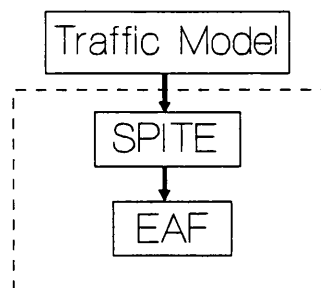
Chapter 7 will describe in detail the results of specific applications of these techniques and how such results have been incorporated into the assessment framework, as to represent the costs incurred by road traffic on the environment. The results of the assessment procedure will constitute relevant information on the actual overall costs incurred and will furnish politicians and planners with further grounds for supporting decision-making.

## 6 MODELLING ROAD TRAFFIC AIR POLLUTION AND NOISE

The analysis in this work is based upon the application of a suite of models for the prediction of environmental impacts followed by an environmental assessment framework. A traffic model provides the input data to the Suite for the Prediction of the Impacts of Traffic on the Environment (SPITE) and the outputs of the suite are applied to the Environmental Assessment Framework (EAF). These acronyms will be used frequently throughout this work.

The overall structure of this research is illustrated in Figure 6.1 below. The boundaries of the dotted line represent the developments of this research.

**Figure 6.1: Overall structure**

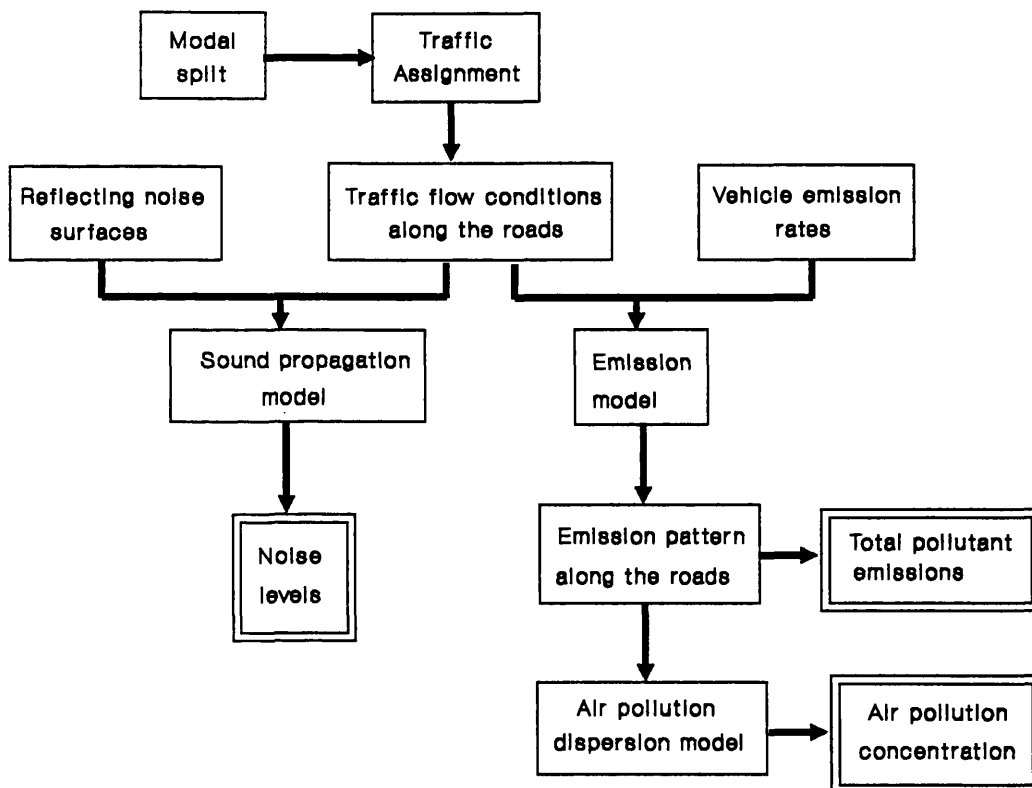


As this work is based on the application of a series of models, it is important to draw some remarks about modelling. Modelling is defined as any process created to describe the link between cause and effect, usually by a mathematical expression. A model is a representation of reality and it can be described in more or less sophisticated fashions, depending on the objective and capability of different applications. Models are largely used in transportation to simulate the demand for travel given the available supply and infrastructure. Models have also had increasing use in the field of the environment to estimate the impacts caused by degrading sources.

The approach for the modelling of air pollution and noise is illustrated in more detail by Figure 6.2. A combined modal split-traffic assignment model is initially applied to simulate the patterns of traffic flow conditions throughout the road network. This traffic simulation procedure produces a range of information on the characteristics of traffic flows along each

road within the network. The traffic flow data produced by the traffic model, together with assumptions about the reflecting noise surfaces (building facades) along the roads, are used as input for the application of a sound propagation model, which estimates local traffic noise levels. The traffic data is also used in conjunction with data on actual vehicle emission rates as input to the emission model, which predicts the emission patterns along each road. This information is used to predict the total emissions produced in the network as well as an input to the air pollution dispersion model, which estimates the levels of pollutant concentrations at a number of points in the vicinity of road intersections.

**Figure 6.2: Modelling the environmental impacts of traffic**



This chapter starts with reporting the basis of the traffic modelling process utilised in this work. Section 6.1 reports the aspects of transport demand which are relevant to this work. The traffic assignment process is addressed in detail in Section 6.1.1. This section reports some considerations of the method used and includes the determination of the cost function parameters adopted to represent travel choices. The modal split process is reported in Section 6.1.2.

The chapter then continues with describing the SPITE and how the suite of models has been used to carry out the analysis of the impacts of transport policies on the environment. The SPITE is composed of models which have been developed for estimating the total vehicle exhaust emissions and local levels of air pollution concentration and noise from road traffic. The emission model in the SPITE is reported in Section 6.2.1, the air pollution dispersion model is in Section 6.2.2 and noise modelling is in Section 6.3.

Because the nature of each model which compose the suite is very complex, and many of the factors involved in the process may not be entirely accounted for or precisely explained by mathematical equations, it is important that the level of accuracy provided by each model is verified. The expected accuracy of each model is described at the end of the following sections in this chapter. The application of the suite of models in the case study is reported in Section 6.4. Chapter 7 will address the main considerations about the EAF.

## **6.1 Transport demand**

The demand for travel is a result of many individuals decisions. Travellers are motivated to make their travel choice for many reasons. They decide if and when to travel, which mode to use, where to go and which route to take to get to the desired destination. The basic role of transport demand modelling is to represent in mathematical formulations the operation of a system composed of individuals' travelling decisions. These decisions depend upon a large number of variables, such as cost, travel time, comfort, convenience, accessibility and other factors which may influence the preferences of individuals.

Transport models represent the demand for travelling between origin-destination zones, which form the geographic boundaries of trip matrix behaviour. The pattern of demand is usually derived from a comprehensive household survey. As far as transport supply is concerned, most urban transport models employ a discrete approximation to represent urban space (divided into travel zones) and the transport supply network. The respective centroid of each zone represents the population gravitational centre, where all trips are assumed to be generated or attracted to the zone. The road network is portrayed by links representing road sections, and nodes representing intersections.

### 6.1.1 Traffic assignment

Traffic flow is the principal factor influencing the level of environmental degradation in urban areas. In order to estimate the magnitude of the environmental impacts produced, it is essential to determine the prevailing characteristics of traffic throughout the road network. Road traffic assignment models are capable of generating information as such (in particular, traffic flow, speed and level of congestion) which can be used as input in air pollution and noise modelling. Traffic simulation has been previously applied in conjunction with air pollution models (Cohen, 1977; Matzoros, 1990).

Traffic assignment is a process designed to estimate how traffic would be distributed over a network as a result of the demand for travel and subject to the characteristics of the road system. A common way of modelling this process is on the basis of the perceived costs of travel, which basic premise is that all travellers have complete information concerning travel costs and time on every possible route. Furthermore, all individuals are assumed identical in their behaviour and they make rational decisions on the basis of this information, always selecting the route which offers the minimum perceived individual travel cost.

There is a number of methods conceived for assigning traffic onto a road network (Ortúzar and Willumsen, 1990). It is widely accepted that *user equilibrium* is one of the best and most commonly used approaches for modelling route choice. This assignment framework follows the Wardrop's *user equilibrium* principle, which simulates the behaviour of a rational traveller who attempts to maximize his or her utility subject to available resources (Wardrop, 1952). Equilibrium is reached when no traveller can reduce his or her path generalised costs by switching routes, i.e., the equilibrium distribution of traffic is achieved when each vehicle travels along its minimum journey cost route through the network. The set of traffic volumes on each road is such that the travel cost on all routes between an origin-destination pair are equal or less than on any unused route. *User equilibrium* takes account of the fact that traffic congestion occurs, beyond which point both speed and flow start declining. Thus, *user equilibrium* finds traffic loadings under conditions of capacity restraint, in which flow is incrementally loaded onto the network allowing congestion to build up gradually and travel cost estimates to adjust in response. This in itself simulates the spread of trips on the network.

Traffic does not operate under steady state conditions throughout a road network, especially



in urban areas. In fact, speed is usually reduced and additional travel time (delay) occurs when traffic flow increases. Vehicles often have to stop fully when approaching junctions. The actual traffic operation at junctions may limit road capacity, in which case, additional congestion effects may occur. Such congestion effects around intersections imply that more time and higher costs are spent on travelling.

Consideration of traffic congestion is essential for the purposes of this research, because not only travel costs (including travel time and fuel consumption), but also air pollution and noise levels tend to increase after traffic flows reach the saturation point. Besides, the diverse driving operation patterns imposed by congestion, like decelerating, idling and accelerating, produce much greater air pollution<sup>12</sup> and noise emissions.

In the U.K., a number of techniques have been developed in recent years to represent traffic route and operation. The available techniques to represent both traffic management changes and the consequent changes in drivers' choices of routes include: CONTRAM, HINET, JAM, SATURN, TRIPS, URBASS and TRAFFICQ (Allsop, 1990). Amongst these models, SATURN (Simulation and Assignment of Traffic in Urban Road Networks) and CONTRAM (CONtinuous TRaffic Assignment Model) have been developed specifically in the traffic management context (Allsop, 1990), and hence are most appropriate for traffic management applications and assessments of scheme options.

SATURN it is a sophisticated and powerful model for simulating urban traffic assignment. It has an exceptional capability for simulating travel patterns and flexibility in assessing the impacts of a variety of traffic management schemes and transport policies. SATURN simulates in detail traffic conditions, delays and the formation and dissipation of queues at junctions by means of cyclic flow profiles<sup>13</sup>, for given levels of flows on each stream. The advanced junction delay models in SATURN permit very accurate representation of traffic

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<sup>12</sup> Further evidence of the air pollution effects from driving patterns at junctions can be found in Claggett et al. (1981), Horowitz (1982) and Potter and Savage (1982 and 1983). Matzoros and Van Vliet (1992) also show that congested conditions are disproportionately more polluting than non-congested. This indicates the usefulness of traffic management and control for air pollution improvements.

<sup>13</sup> For simulation purposes, a cyclical behaviour is imposed upon flows by traffic signals operating with a common cycle time (typically in the range of 60 to 120 seconds). Within each cycle, traffic is represented by flow profiles rather than individual vehicles or packets of vehicles.

route choices, particularly in congested urban areas. SATURN is able to deal with networks composed of several hundred zones. SATURN was produced at the University of Leeds and has been further developed by Steer Davies Gleave. Its description can be found in (Hall, Van Vliet and Willumsen, 1980 and Van Vliet, 1982). SATURN was kindly provided by Steer Davies Gleave for use in this work. It was applied in the study area to estimate the characteristics of traffic along each road of the network.

SATURN requires a range of input data, namely:

- origin-destination matrix;
- bus routes and frequencies;
- road network coding (zones, nodes, links, centroids, gateways);
- link data (length, capacity, number of lanes, flow directions, free-flow traffic speed);
- turn data (saturation flow, turns allowed and turning capacity);
- junction types (priority, signalised or roundabout);
- node co-ordinates;
- parameters such as minimum gap acceptance between vehicles;
- traffic signal data: stage lengths, offsets and cycle time.

SATURN assigns traffic under the *user equilibrium* approach, in which travellers make their decisions based upon their perceived travel costs. The cost function in the assignment model is defined by the expression in Equation 6.1.

$$C = VT \times T + CK \times D \qquad \text{Equation 6.1}$$

- VT = Value of time (pence per minute);
- T = Time (minutes);
- CK = Cost per kilometre (pence per kilometre);
- D = Distance (kilometres).

The cost function is defined by default in SATURN as a function of travel time only. Therefore, the parameters VT and CK are given 1 and 0, respectively. For the purposes of this work, however, the running costs CK have also been accounted for in the assignment process, but only in terms of the expenses with fuel consumption (the other vehicle operational costs not being considered). CK is defined as shown in Equation 6.2.

$$CK = PP \times FC$$

Equation 6.2

PP = Petrol price (pence per litre);  
FC = Fuel consumption (litre per kilometre).

It is important that the costs related to travel distance are included in the cost function in order to represent the influence it exerts on trip making and route choice. The introduction of the distance component into the definition of the travel cost has been found to improve the performance of SATURN (Matzoros et al., 1987). In this work, the cost incurred as a function of the distance travelled is particularly useful to permit analyses of the effects of fuel prices changes in the assignment of traffic and therefore on the levels of environmental degradation produced. Thus, the parameters VT and CK have been appropriately modified in the cost function. The value of the parameter CK was based on official statistics, whereas the value of VT was based on valuations derived from the literature.

According to the U.K. Transport Statistics (Department of Transport, 1994a) the average price of unleaded petrol was 50.1 pence per litre in 1993, and the average fuel consumption for in-service cars was 0.0935 litres per kilometre. This information input in Equation 6.2 makes the cost per kilometre (CK parameter) equal to 4.68 pence per kilometre.

The value of time was taken from the figures published in 1988 by the Traffic Appraisal Manual (Department of Transport, 1991). The average value of working time for a car driver was quoted at 849.7 pence per hour. This figure was updated to 1993 prices through the Gross Domestic Product (GDP) per head index, taken from International Monetary Fund (1995) and from OECD (1995) publications. This index has been used throughout this work (see Chapter 7) to update diverse economic and environmental monetary values to the base year (1993). The value of time converted to the valuation in the base year becomes 18.67 pence per minute, which enters the cost function (Equation 6.1) directly in the place of the parameter VT.

Considering the parameters CK and VT as described above, the cost function has been defined as given by Equation 6.3.

$$C = 18.7 \times T + 4.7 \times D \quad \text{Equation 6.3}$$

The critical time of the day in relation to both traffic conditions and environmental impacts coincides with periods in which vehicle flow and density are maximum. In urban areas, the greatest deal of travel demand is most commonly concentrated at the peak times. The analysis of travel demand was undertaken in the study area during the afternoon peak period, reflecting the most severe levels of traffic congestion and, probably, the highest levels of noise and exhaust emissions.

SATURN assigns buses first in the network, based on the data related to local bus routes and frequencies in the simulation period. Each bus is considered to be equivalent to 3 passenger-car-units (pcus) in terms of the road space occupied. Bus routes and frequencies are assumed fixed. These simplifications imply that, in the simulation process, variations in car flows will not affect the travel time of buses and bus operators will not adjust supply due to changes in travel demand. Cars are next assigned onto the road network.

The flow of heavy goods vehicles has also been taken into account. These vehicles have been added to the network as a proportion of the car flow, according to their participation on the flow composition on each of the main radials entering Chester. The percentage of heavies in the flow was taken from the results of a series of traffic counts carried out in 1993 by Cheshire County Council (1995). The proportion of heavy goods vehicles in relation to the total number of vehicles was in average 4.14%. This percentage is in agreement with the national average figure, as published in the U.K. Transport Statistics (Department of Transport, 1994a), estimated at 4.5% in relation to all traffic which travel within trunk and principal built up roads and minor roads. Like buses, each goods vehicle is considered 3 pcus for the purposes of assignment. Thus, an appropriate loading factor has been applied to the network to represent the flow of lorries in pcus.

The outputs of SATURN that were used in the environmental predictive models are: vehicle queues at junctions, travel times and the flows of cars, buses and lorries on each road.

### ***6.1.2 Modal split***

Mode choice is determined by the utility provided to travellers by various modes of transport. The utility function for each alternative mode is a composite of the generalised costs

influencing individuals in their mode choice. A number of factors influence people's choice for travelling. The explanatory factors include travel, social and behavioral characteristics represented by factors, such as travel cost (petrol price, public transport fares and parking costs), time in each mode and car ownership. Other more subjective factors may also be included, like individual tastes, quality of service, attractiveness, comfort, security, convenience and status. The mode which presents the maximum utility determines the traveller's choice.

The transport package utilised for the analysis of modal changes in the network is SATCHMO (SATurn Travel CHOice MOdel), which has been recently developed to complement SATURN. SATCHMO has been developed by Steer Davies Gleave and WS Atkins Planning Consultants and has also been provided for use in this research. SATCHMO's main attribution, as the name itself suggests, is describing multi-modal choice, and it provides the facilities required to model a wide range of behaviour responses by trip-makers (Willumsen et al., 1993). SATCHMO addresses the various interactions between road schemes and public transport.

In a disaggregate level, the probability of a traveller to choose a certain mode within a range of alternatives is given by a multinomial logit model, in which the utility of the mode considered is exponentially contrasted with the sum of the utilities of all alternatives. However, the aggregate modal split model incorporated in SATCHMO distributes trips amongst the available modes through estimates of the proportion of individuals choosing one over a defined set of alternatives. SATCHMO performs simultaneous travel choice and assignment using a modified Frank-Wolfe algorithm (more details about this algorithm can be found in Sheffi, 1985). Its full description is given in Willumsen et al. (1993), but the main relationships are reported below. The proportion of travellers choosing alternative of transport  $k$  is given by the expression in Equation 6.4.

$$P_k = \frac{\exp(-\beta GC_k)}{\sum_k \exp(-\beta GC_k)} \quad \text{Equation 6.4}$$

- $P_k$  = proportion of travellers which choose the alternative  $k$
- $GC_k$  = generalised cost of choosing alternative  $k$
- $\beta$  = coefficient which weighs the explanatory variables and accounts for particular tastes and non-rational behaviour representative of the study area.

In order to cope with incremental and classic distribution and mode choice outside an explicit equilibration network, an incremental model is used as shown in Equation 6.5.

$$P_k = \frac{P_k^* \exp(-\beta \Delta GC_k)}{\sum_k P_k^* \exp(-\beta \Delta GC_k)} \quad \text{Equation 6.5}$$

$P_k^*$  = proportion of travellers which choose the alternative  $k$  in the base scenario  
 $\Delta$  = indicate changes in generalised costs between the base and a new scenario

Analogously, the singly constrained incremental model can be written to represent the number of travellers choosing a particular option as indicated in Equation 6.6.

$$T_{ij} = \frac{G_i T_{ij}^* a_j \exp(-\beta \Delta GC_{ij})}{\sum_{ij} T_{ij}^* a_j \exp(-\beta \Delta GC_{ij})} \quad \text{Equation 6.6}$$

$T_{ij}$  = number of travellers between zones  $i$  and  $j$   
 $T_{ij}^*$  = number of travellers between zones  $i$  and  $j$  in the base scenario  
 $G_i$  = total trips generated at zone  $i$   
 $a_j$  = growth factor reflecting changes in the destinations  $j$

The formulations represented by Equations 6.5 and 6.6 are convenient because they only account for changes in the generalised costs, rather than their complete values, thus any modal penalties are cancelled out in  $\Delta GC$ .

Transport policies may exert substantial influence on travel mode choice, and this can affect the characteristics of traffic flows, which eventually alter the patterns of vehicle emissions. Besides, each mode of transport produces distinct impacts on the environment. The introduction of a modal split model in this work enables the assessment of how travellers will respond to changes in various transport policies which affect modal choice, and the extent to which mode choice can affect the levels of environmental impacts produced.

A great part of the input data necessary to run SATCHMO is related to the characteristics of travel demand (provided by the origin-destination matrix) and transport supply (expressed by

the road network coding), as used by SATURN. Further data required as input to the mode choice model include the public transport trip matrix, costs of travelling by public transport modes and estimates of walking time to the locations where public transport is provided (usually bus stops). The outputs from SATCHMO are the number of trips which should be made by each of the transport modes under consideration.

### *6.1.3 Accuracy of the traffic model*

SATURN is a widely accepted assignment model and has been applied in various studies worldwide. Most consultancies and local authorities in the U.K. utilise SATURN for diverse traffic planning and management purposes, and the number of users amounts to over two hundred. SATURN has also been used in a number of transport studies in Britain and around the world.

Both traffic assignment and modal split models have been previously applied to the case study. SATURN was applied by the Cheshire County Council in a traffic project for the justification of the construction of the Chester Western Bypass. SATCHMO was also applied to Chester in a project in which Steer Davies Gleave was commissioned to determine the changes in modal share caused by the implementation of a light rail transit. The outputs of the assignment model have actually been compared with actual traffic counts in Chester. Unfortunately, such results have not been publicised and could not be obtained to illustrate the level of accuracy of the assignment model at the case study conditions.

Several pieces of work have been carried out on the validation of SATURN. In particular, Matzoros et al. (1987) carried out a series of tests using data collected in central Manchester to assess the accuracy of the predictions provided by this assignment model. Both model predictions and manual measurements were made for the morning peak hour of weekdays. The model inputs were improved in two ways: first, best fit generalised cost parameters were used in the cost function and, second, the trip matrix was updated using a matrix estimation procedure. The percentage differences between modelled flows and observed counts, given by the mean absolute difference divided by the average observed flow, were around 25% using the original trip matrix, but were reduced to about 8-16% using the updated trip matrix. Given that the level of accuracy inherent in the traffic counts is probably in the order of 10% (due to counting errors, day-to-day variations, etc.), the fit was considered to be very good.

However, it must be borne in mind that the simulations performed by SATURN are likely to underestimate not only the levels of traffic flow but also congestion and delays. This underestimation can originate from the effect of the journeys not incorporated within the trip matrix (such as those made by taxis and visitor vehicles), which are not considered in the traffic assignment process.

## **6.2 Air pollution modelling**

Air pollution has been modelled by various approaches (as reported in Section 3.6) and for diverse purposes, notably for assessing the impacts caused by the construction of new road transport infrastructures. In this work, air pollution modelling was addressed specifically to provide information about the likely changes in air quality provoked by a range of traffic management schemes, transport policies and future traffic trends. Thus, the model was developed to be sensitive enough to the impacts of small changes in the characteristics of traffic flows.

The air pollution model in the SPITE initially incorporates the input data required as produced by the assignment model, namely flow of cars and buses, travel times, number of lanes, number of vehicles in the queue and road lengths. It also reads the data file containing the coordinates of all nodes in the network. Since the coordinates are given in a virtual scale and two-way roads are represented on the same link, the model calculates the real coordinates for both ends of one-way roads, accounting also for the road widths. The model also calculates the queue length (idling distance upstream from the junction), average traffic speed, flow of heavy goods vehicles and the angle of each road in relation to the grid of node coordinates. Based on the characteristics of flows, the model estimates the speed at the beginning and end of each road. It also takes into account vehicle acceleration and deceleration rates in order to estimate the distances along the road within which vehicles operate under each typical driving pattern (this is explained in forthcoming sections and is illustrated in Figure 6.4).

The assumptions utilised in the air pollution model in the SPITE are:

- the average length of vehicles in a queue is 5.75 metres, including the gap between vehicles.



- the acceleration and deceleration rates are constant along the road, equal to the average rates from the European urban driving cycle.
- vehicles decelerate down to the middle of the queue and re-start the movement also from the middle of the queue, accelerating until they reach the average speed of each road.
- all vehicles are assumed to slow down at give-way junctions where queues are not formed. In these cases, speed is reduced by half.

The modelling of air pollution has been based on the theoretical and empirical relationships developed by the Transport Research Laboratory, and described as the TRL method in Section 3.6. A number of modifications have been made in this research to the original method for estimating air pollution concentrations, in an attempt to improve the accuracy of the predictions. The principal change relates to the emission model which has been entirely re-developed in order to represent with more precision the actual traffic emissions in congested networks.

### ***6.2.1 Emission modelling***

Vehicles operate under a variety of conditions, specially in urban areas, and the emission patterns vary considerably according to the conditions of traffic flow. The emission model used in the TRL method is simplistic and not completely adequate for the urban case, since it does not take into account the effects of traffic congestion. In the TRL method, a unique emission rate is assumed for all vehicle types in all driving modes, depending only on the average traffic speed. Besides, the empirical relationship for traffic emissions in the TRL method is rather outdated, as it represents the 1980 vehicle fleet emissions. The relationships which use CO as a proxy to estimate concentration levels of other pollutants are also no longer reliable, because traffic emissions have changed dramatically since this method was conceived. Thus, although no other predictive model has proved to provide significant advantages over the TRL method at the current state-of-the-art, and hence its utilisation is still recommended by the British Government, it must be borne in mind that the nature of its empirical relationships may produce inaccurate results, mainly when applied to interrupted traffic flow conditions.

A new emission model was developed to compose the SPITE, particularly with the purpose of addressing the weakness of existing models for predicting traffic air pollution in urban areas. The traffic emission model tries to reproduce the traffic emission pattern along each road of the network. Initially, the results of the emission model are utilised for the estimation of the total vehicle exhaust emissions within the study area. They are subsequently used as input data in the dispersion model, which predicts local levels of pollutant concentration. Other standard predictive models do not produce data on total emissions. In fact, since most pollutants have principally local effects, their total emission levels may not have great relevance for policy decisions, other than functioning as mere indicators. However, the most important effect of CO<sub>2</sub> emissions is global, as it contributes to the greenhouse effect. The total emission level of this pollutant has become a very meaningful piece of information for sustainable transport policies, especially after international agreements have been set for the reduction of global emission levels.

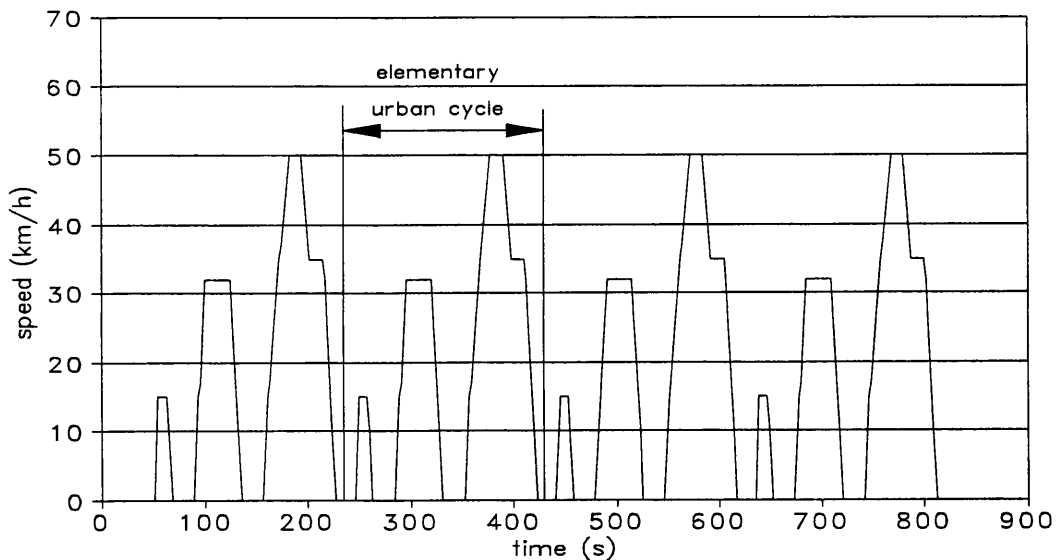
The emission rates of all pollutants have been modelled using the data from actual vehicle exhaust emissions. It was not feasible to take into account in the emission model all the factors influencing emission rates (see Section 3.3). Exhausts from petrol cars were modelled for four different driving patterns (accelerating, decelerating, cruising and idling) and the fitting of catalytic converters. The idea of modelling the typical traffic operating patterns is to represent the spatial variability of vehicle exhaust emissions, particularly around road junctions, which accounts for much of the effects of traffic congestion. This is considered an advance in relation to other existing emission models. Some emission models (for instance Barlow et al., 1992) have adopted the instantaneous vehicle speed and the product of speed and acceleration as the best indicators of emissions in relation to driving patterns. Emissions from diesel vehicles have been modelled as a function of vehicle type: cars, buses and lorries.

Emission tests are expensive to carry out. In addition, a large number of surveys ought to be made in order to derive a representative figure for overall vehicle emissions. As a result, there are only a few emission data inventories available in the literature. The most recent large scale emission surveys have been carried out by the Warren Spring Laboratory (Farrow et al. 1993a, 1993b and 1993c), Commission of the European Communities (Eggleston et al., 1993) and in the Drive Project (Jost et al., 1992). However, there are no recently published data on emissions as a function of the pattern of driving, which is an important model parameter. The latest published data as required by the emission model were produced by the Warren Spring Laboratory over a decade ago (Potter and Savage, 1982 and 1983). These rates were

representative of the vehicle fleet at the time and are, in general, much higher than those found in recent emission test surveys. The main reasons for such decline are the technological improvements in the automobile industry and the emission limits imposed upon new vehicles.

The emissions from petrol engines, with the exception of PM, were obtained from a series of bench test emission surveys on in-service cars, carried out by Tickford Limited (1994). These emission rates are derived from surveyed vehicles driving at patterns which comply with the EU urban cycle (European Community, 1991a). Figure 6.3 shows the speed profile of the urban phase of the driving cycle.

**Figure 6.3: Urban phase of the European driving cycle**



Source: European Community (1991a).

The sample of emission test results was taken as the representative rates for the total fleet of petrol cars in the study area. The emission rates of PM from petrol cars are not routinely measured, as they represent a very small fraction of the exhausts produced. Since emission surveys have not derived emission factors of PM from petrol cars as a function of the patterns of driving along the roads, these are modelled as the average rate for the whole driving cycle. The emission rate of PM was taken from data in recent literature.

Table 6.1 illustrates the representative emission rates for U.K. petrol vehicles as applied in the emission model, according to the driving patterns and the fitting of catalytic converters.

**Table 6.1: Emission rates from petrol cars**

| Driving pattern           | Catalytic converter | Pollutant emissions (g/km) |      |                 |                 |                   |
|---------------------------|---------------------|----------------------------|------|-----------------|-----------------|-------------------|
|                           |                     | CO                         | HC   | NO <sub>x</sub> | CO <sub>2</sub> | PM                |
| Acceleration <sup>a</sup> | without             | 17.83                      | 3.64 | 3.00            | 373.95          |                   |
|                           | with                | 6.55                       | 0.77 | 0.96            | 400.77          |                   |
| Deceleration <sup>b</sup> | without             | 7.03                       | 2.09 | 0.62            | 133.60          |                   |
|                           | with                | 1.68                       | 0.27 | 0.11            | 153.82          | 0.02 <sup>d</sup> |
| Cruise                    | without             | 8.78                       | 1.89 | 1.43            | 204.18          |                   |
|                           | with                | 3.36                       | 0.38 | 0.32            | 216.87          |                   |
| Idle <sup>c</sup>         | without             | 1.73                       | 0.36 | 0.06            | 33.47           |                   |
|                           | with                | 0.44                       | 0.09 | 0.01            | 35.41           |                   |

Source: compiled from Tickford Limited (1994).

a) average acceleration rate: 0.75 m/s<sup>2</sup>.

b) average deceleration rate: 0.70 m/s<sup>2</sup>.

c) emission rates for the idling mode are given in g/min instead of g/km.

d) representative of unleaded, non-catalyst petrol cars. Source: Gover et al. (1994).

As described in Section 3.3, it can be noticed from the results of this actual emission survey that cars accelerating produce much higher emissions of all pollutants than in any other driving pattern. Thus, these emission rates capture much of the spatial details of pollutant emissions in interrupted traffic conditions. It can also be seen that the emission rates of CO<sub>2</sub> in absolute terms are many times higher than the emission of other pollutants. Some of the effect of cold starts, which has proven to be an important factor influencing emission level, has been incorporated on the emission rates from petrol vehicles, since the emission rates for each driving pattern were averaged during the four elementary cycles within the urban phase (see Figure 6.3). Emissions from the first elementary cycle were consistently higher than from the subsequent runs, what indeed reflects the effect of cold starts.

Table 6.1 also reinforces that the fitting of catalytic converters reduce the emission of all pollutants but CO<sub>2</sub> (such results are in agreement with the findings reported in Section 3.3). The Society of Motor Manufacturers and Traders (1994) has estimated that about 11% of the total petrol passenger cars in circulation had been equipped with catalytic devices at the end of 1993. This figure has been utilised in the modelling process for the estimation of the patterns of vehicle emissions, accounting for the effect of catalytic devices on emission rates.

However, such devices have proven to degrade and fail with use. For the purposes of this work catalysts have been assumed to fail at a rate of 5% a year, since failed devices have been detected in the annual MoT test and repaired with a 95% success rate (Quality of Urban Air Review Group, 1993b). This rate has been subtracted from the estimates of the participation of catalytic-fitted vehicles, in order to consider device failures in the analysis.

The emission data from diesel engines used in the emission model were taken from U.K. test results carried out by the Transport Research Laboratory and represent the most up-to-date and comprehensive set of data currently available (Gover et al., 1994). Table 6.2 shows the diesel emission rates for urban driving conditions.

**Table 6.2: Emission rates from diesel vehicles**

| Vehicle type       | Pollutant emissions (g/km) |      |                 |                 |      |
|--------------------|----------------------------|------|-----------------|-----------------|------|
|                    | CO                         | HC   | NO <sub>x</sub> | CO <sub>2</sub> | PM   |
| Car <sup>a</sup>   | 0.62                       | 0.12 | 0.66            | 153.80          | 0.23 |
| Bus                | 18.80                      | 5.80 | 16.20           | 1148.00         | 1.60 |
| Lorry <sup>b</sup> | 4.92                       | 0.74 | 9.58            | 630.00          | 1.53 |

Source: Gover et al. (1994).

a) emission rates from medium diesel cars.

b) medium heavy goods vehicles (between 7.7 and 17 tons).

The effect of cold and hot engine temperatures has also been incorporated into the emission rates from diesel cars, since the proportion of all journey distances driven with cold and hot vehicles in the U.K. is taken into consideration in the emission estimates (Eggleston et al., 1993). For the purposes of the emission model, all lorries and buses have been assumed to be diesel fuelled.

The emission contributions from diesel cars, buses and lorries were considered uniform along the link, according to the average emission rate during the driving cycle. Thus, the emissions from diesel vehicles have not been modelled in accordance with the patterns of driving, like the emissions from petrols, because emission data as such have not been produced from emission test surveys.

The exhaust emission rates utilised in the emission model (given by Tables 6.1 and 6.2) are

assumed equal for all vehicles of the same type (petrol and diesel cars, buses and lorries). The traffic composition on the simulated network was estimated according to the proportion of petrol and diesel cars within the British vehicle fleet. Based on the U.K. Transport Statistics for 1993, 5.9% of the cars on the road have been assumed diesel powered (Department of Transport, 1994a).

As reported in Section 3.4, all new vehicles from 1996 must conform with new emission limit regulations (see Tables 3.4, 3.5 and 3.6). The emission model takes into account the future emission limits imposed by this regulation, in the analyses of the impacts originated from transport trends which affect the projected traffic circulation conditions (these trends are dealt with in Chapter 9).

Other aspects of vehicle emissions, such as maintenance and age, are not accounted for in this study because there are no emission data available on the quantification of these aspects. Besides, the deterioration in catalyst efficiency with use is probably under-represented in the emission data.

The empirical adjustment coefficient used in the dispersion model (see Equation 3.1) not only accounts for background concentrations and contributions from other sources, but also converts concentration units from  $\text{g/m}^3$  to ppm. The conversion factors for these two commonly used measures of concentration are, as a matter of fact, different for each pollutant. The conversion factors shown in Table 6.3 have been incorporated within the adjustment coefficient for the estimation of concentrations of each pollutant in ppm.

**Table 6.3: Conversion factors for pollutant concentrations**

| Pollutant  | $\text{g/m}^3$ | ppm     |
|--|----------------|---------|
| CO <sup>a</sup>                                    | 1              | 829.19  |
| HC <sup>b</sup> (as methane)                       | 1              | 1508.29 |
| NO <sub>x</sub> <sup>a</sup> (as NO <sub>2</sub> ) | 1              | 504.79  |
| PM <sup>c</sup>                                    | 1              | 384.61  |
| CO <sub>2</sub> <sup>b</sup>                       | 1              | 549.45  |

a) derived from Greater London Council (1983);

b) derived from Hesketh (1972);

c) derived from Perkins (1974).

### *6.2.2 Dispersion modelling*

Dispersion modelling represents the process of the scattering of air pollution and predicts the eventual concentration levels at particular locations and time periods. The dispersion model in the SPITE estimates the concentration levels of the five air pollutants under consideration.

The concentration of most air pollutants is far the best indicator of air quality. However, the estimation of CO<sub>2</sub> concentrations originated from traffic emission sources is not of great meaning for air quality assessment purposes. CO<sub>2</sub> does not have local health effects since it is not a poisonous gas, and it is found in abundance in the atmosphere as background concentrations. Besides, it is emitted in highest quantities by sources other than traffic (only about one fifth of the total emissions of this pollutant is produced by transport sources). Nevertheless, CO<sub>2</sub> concentrations are estimated by the dispersion model in the same way as the other pollutants, but transport planners should regard such estimations as indicators of less relevance in air quality. The dispersion of PM emissions is estimated in the same way as gaseous pollutants, since particles are small enough to behave like gases.

Air pollutants disperse rapidly in the atmosphere and the concentration falls off considerably with the distance from the source. The 'breathing zone' in urban areas, i.e. where most outdoor human activities take place, is usually a limited space between kerbsides and building facades. The definition of the precise location for the estimation of pollutant concentrations is an important element in the dispersion process. There is not a consensus about the location where concentrations should be measured or estimated, and this has caused incompatibility of concentration data produced by different studies (Linaritakis, 1988).

In this work, concentrations have been estimated at locations in the network where traffic emissions are highest, in order to represent the maximum pollutant concentration levels that can be found where pedestrian activities are expected. Such locations are in the vicinity of road junctions, where flows are often interrupted. The driving conditions which produce highest vehicle emission rates, such as acceleration and idling, are most frequently found around intersections. Furthermore, vehicles usually spend longer time and more vehicles utilise the road space around junctions than at any other mid-block location. The predictions of air pollution have been made at points situated downstream of each link, on the nearside pavement of the lanes where queues may form, at one metre from each kerb of the junction.

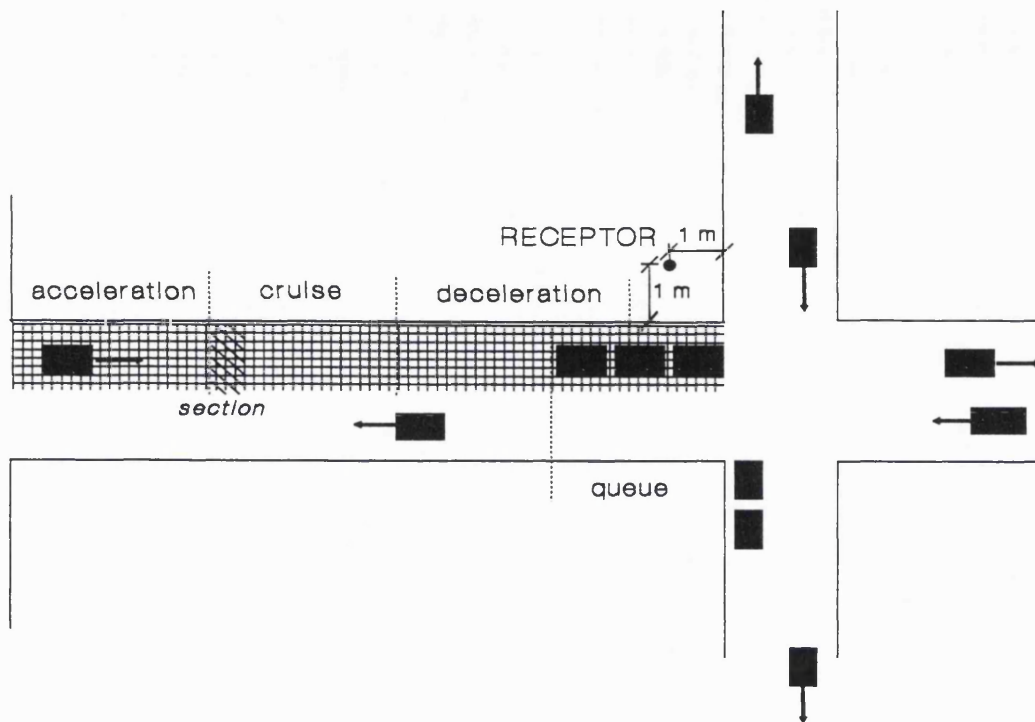
A procedure was created within the air pollution model in order to determine the coordinates of all the points throughout the road network where the concentrations are estimated. The mechanism for determining such points is not as simple as it may seem to be. The procedure initially finds the end of each road (where queues may form) and the road which joins it on the left (this defines the pavement area which is bounded by both roads). Then, the procedure calculates the road widths and the angle between both roads at the junction. Finally, using a series of straight line equations, it calculates the coordinates of the crossing point of the two straight lines parallel to each road, at the pre-established distance of one metre from each kerb. These locations where concentrations are estimated have been defined as 'receptor points'. As far as air pollution modelling is concerned, the receptor points are assumed to be at the same height as the polluting sources (see Section 3.6).

The traffic in each road is assumed to produce emissions at the centre of the road (two-way roads are modelled as two separate roads running next to each other in opposite directions). There is scope for some inaccuracy in this modelling assumption in relation to roads with more than one lane, since the inner lane tends to be used more intensively by slower and heavier vehicles. The vehicle emissions produced in the inner lane (which is closer to the receptor points) are likely to differ from the exhausts in other lanes, since the emission patterns from heavy vehicles are very distinct from the exhausts emitted by most light vehicles (see Section). Parking lanes are not considered in the model.

Figure 6.4 illustrates the way in which the exhaust emissions produced by traffic in various road segments around a road junction, and subject to the dispersion conditions, contribute for the total pollutant concentration at a specific receptor point.



Figure 6.4: The emission contributions from road segments to a receptor point



The air pollution model divides each road into segments representative of the typical driving patterns, which produce specific emission patterns. This is shown in the illustration by the shaded road divided within the distances in which vehicles are most likely to queue, accelerate, decelerate and cruise during the period of analysis. The emission rates presented by Tables 6.1 and 6.3 have been applied to road segments as a function of the patterns of driving which are typical on each of them. These segments are further divided into smaller sections, which represent emission contribution units and emissions are calculated individually for each of these sections (a road section is shown in hachured in the figure). The concentration level at the receptor point is calculated by summing up the contributions from all emission sections, at the distance between the point and the middle of each section. All the roads situated around the receptor point contribute to the built up level of air pollution concentration<sup>14</sup>.

As far as the dispersion of air pollutants is concerned, the most unfavourable situation occurs

<sup>14</sup> Past experience and modelling experiments have shown that the emission contributions become negligible when road sections are situated at distances over 200 metres from the receptor.

when the wind speed is low and when it scatters the emissions towards the receptor point. Following guidance from the TRL method regarding the conditions which give reliable 'worst case' results, wind speed was assumed to be low at roadside sites and to be distributed evenly in all directions. The wind speed in every site was set to 1 m/s, which is the lowest level normally found around road sites, accounting mostly for the air turbulence produced by traffic itself. As predictions were made for a grid of points in the network, the wind in each point was distributed in eight directions during the modelling period.

Three important points have been made earlier in this chapter about estimating the highest expected levels of air pollution concentration. The first point was in relation to the time period. The analysis of travel demand was undertaken for the afternoon peak period, when demand for transport is at maximum, traffic is most concentrated, flows are most often congested and the environmental impacts are therefore most serious. The second point referred to the locations where concentrations should be estimated. The receptor points were located close to where traffic emissions along each road are highest. Finally, the meteorological dispersion conditions have also been assumed unfavourable. These three points should ensure that the concentrations found represent the maximum levels to which the community may be exposed.

A change in the characteristics of traffic, as produced by various transport policies and trends, affects not only traffic volumes but also speed, queue lengths at junctions and the distances travelled on the remaining driving patterns. All these factors affect the exhaust emission patterns on the sections along the links and as a consequence, their relative contribution to the concentration levels at receptor points. Thus, although the dispersion parameters used are constant, pollutant concentrations do not correlate linearly with changes in traffic flows.

### ***6.2.3 Accuracy of the air pollution model***

No method for short-term predictions has been successful for producing accurate results in air pollution modelling. Dispersion methods applied to long-term predictions are likely to produce much better results.

There are three main sources of potential errors in pollution dispersion modelling under the Gaussian approach (further discussion can be found in Chiquetto, 1993). The first concerns the emission factors, which cannot be obtained for each individual source with precision.

Average emission factors are used in emission modelling, but as reported in Section 3.3, there is a large scope for variance in emissions from different vehicles. A modest error of 10% at each node in a Gaussian plume calculation has been shown to produce an outcome uncertain by 40 to 400% (Benarie, 1987). The dispersion process is the second source for potential errors, which can result from the complexity in modelling the various influencing factors as described in Section 3.5. Atmospheric dispersion comprises a deterministic component which may be modelled with all the precision allowed by the experimental input, and a random element which gives margin to a range of errors. The third source of error is related to the level of background concentration, which is extremely difficult to assess.

Given the variety of uncertainties in the estimation of the modelling parameters and in the description of the modelling process, a good level of accuracy cannot realistically be expected. Even when predictive methods are applied to very simple dispersion conditions, such as around a stationary source in an open space, they have proven to be unable to provide very accurate results. In particular, Gaussian models have proven worthless in conditions of extreme stability, very low wind speed, variable wind direction at different levels and plume rise from the heated exhaust (Chock, 1977).

A high level of uncertainty has been reported in the literature on air pollution modelling. A number of studies have found that models based on the Gaussian plume theory frequently provide estimations which differed from the observed concentration levels by a factor of two, and occasionally the differences may reach a factor of three or more (Chock, 1977; Noll et al., 1978; Sistla et al., 1979; Rao and Keenan, 1980; Rao et al., 1980; Verkatram, 1984; Menard et al., 1987; Chiquetto, 1993 and Larssen et al., 1993). In particular, Rao et al. (1980) have evaluated the goodness-of-fit of four Gaussian and three empirical models. Of all the models tested, that which provided the best simulation predicted within a factor of 3 of the observed concentrations during 87% of the time. Rao and Keenan (1980) used improved dispersion curves to the Gaussian model and found predicted concentration to lie within a factor of 2 of the observed during 82% of the time. This can be considered a substantial improvement in the estimation accuracy.

The TRL method was a result of several years of research. A great deal of pollution monitoring has been carried out in order to calibrate empirically several relationships within the method. It is considered not less accurate than any other existing method and long experience from its developers has proven that the method provides plausible results. The

TRL method is mostly suitable for highways, where traffic flow is predominantly uninterrupted and sources emit uniform rates of air pollutants, but it also provides satisfactory results when applied to urban sites (empirical evidence can be found in Hickman and Colwill, 1982; Matzoros and Van Vliet, 1992).

The calibration and validation of the TRL model have been carried out by its developers and they were based on measurements of CO. In order to illustrate the typical accuracy of this model at urban sites, the TRL has compared predictions with measured figures (Hickman and Colwill, 1982). In this experiment, about 91% of estimates fell within  $\pm 4$  ppm of the observed concentrations and 74% of estimates were within  $\pm 50\%$  of measured values. The validation for other compounds has not been demonstrated, but there seems to be no reason why the method should not be applied to simulate the dispersion of other pollutants.

The TRL model has also been applied at urban sites in Leeds and Manchester, and performed satisfactorily in tests against observed pollutant concentrations (Matzoros, 1988 and 1990). The results of this study have shown that two-thirds of the modelled values, excluding the predictions of NO<sub>x</sub>, fell within  $\pm 50\%$  of the observed ones and 96% of them were within  $\pm 100\%$  of the observed. McCrae et al. (1988) also compared observed concentrations with the results of the TRL method and the accuracy found ranged from -52% to +45%, considering distances from the road edge of 5 and 10 metres, respectively.

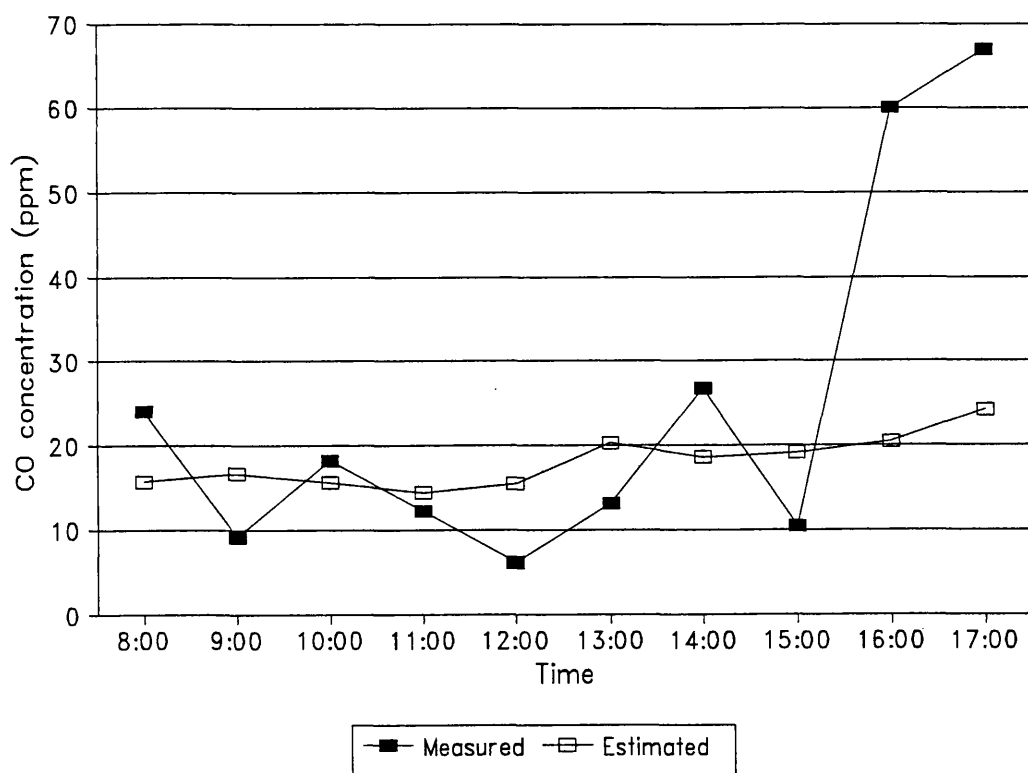
A simplified version of this method is recommended for the prediction of pollutant concentrations by the British Government in the Design Manual for Roads and Bridges (Department of Transport et al., 1993). The TRL method has been used in several consultancies in the U.K., principally in assessments of the impacts of road traffic commissioned by local authorities. It was also used as part of a model suite, which has been applied to predict pollution episodes in Athens.

A further series of air pollution concentration, traffic and meteorological data were necessary to test the level of accuracy of the dispersion model in the SPITE, principally after changes have been made to the original TRL model. A data set was obtained from the Nottingham University Transport Research Group, who has been carrying out an automatic air pollution monitoring programme in urban sites in Leicester. The author has participated in the data collection from roadside surveys. The data include observations of carbon monoxide concentrations, wind speed, wind direction and traffic flow and composition in both

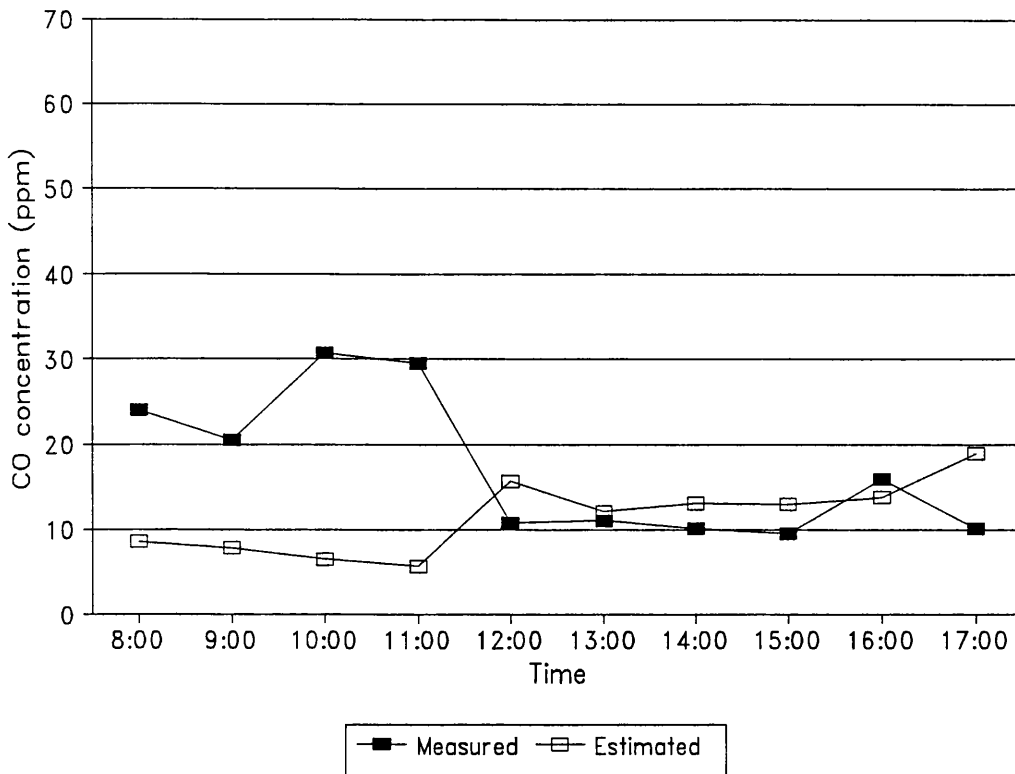
directions. Since the set of data in each survey was very often incomplete, only two surveys could be taken for the application of the dispersion model. The sites selected are at University Road and Regent Street, and they were surveyed in 16 June and 18-20 August 1992 from 8:00 am to 5:00 pm.

The concentration levels of CO were measured in the field and compared to estimations carried out using the air pollution model. Figures 6.5 and 6.6 show the comparisons between measured and estimated levels of CO concentrations in urban sites in Leicester.

**Figure 6.5: Measured and estimated CO concentrations at University Road, Leicester**



**Figure 6.6: Measured and estimated CO concentrations at Regent Street, Leicester**



The air quality in the locations surveyed could not be considered at all satisfactory. The actual levels of CO concentration exceeded by far the standards for one-hour period on two occasions in the first survey and almost reached the standards on two other occasions in the second survey. The standards for eight-hour periods were exceeded in every interval within both field surveys. These standards should not be exceeded more than once a year (see Table 3.7), but they were exceeded during times of the surveys. This suggests that the actual levels of CO are likely to exceed the standards many more times throughout the year and severe measures are needed to ensure that traffic emissions are restrained in order to comply with the air quality standards.

The data set collected in Leicester is neither comprehensive nor accurate. It consists of only two one-day surveys and it does not include two variables necessary in air pollution modelling, namely traffic speed and queue at junctions. Some assumptions had to be made for estimating such variables. Another problem related to the reliability of the data collected

is that traffic counts and further readings were made manually and during very long periods, which could have given a margin of inaccuracy in the observation. Nevertheless, the discrepancies found between measured and estimated figures are not any worse than those from experiments in the literature (Claggett et al., 1981; Venkatram, 1984; Matzoros, 1988; Matzoros and Van Vliet, 1992). All but one model results, which represents 95% of the estimations, lied within  $\pm 100\%$  of measured values and about half of the calculated figures were within  $\pm 50\%$ .

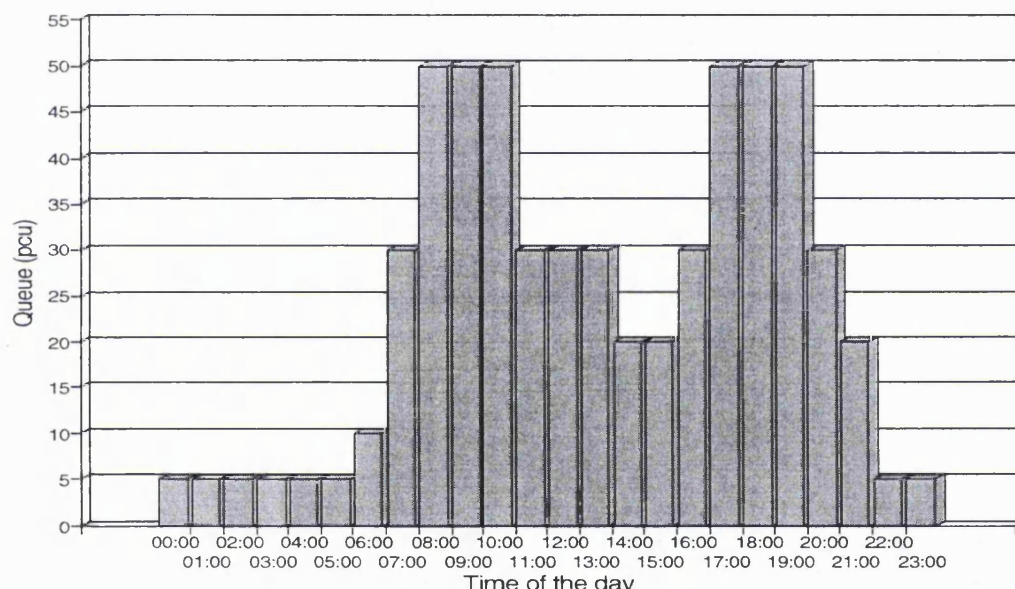
Since the data from Leicester did not provide a very good correlation with the estimates, a more comprehensive set of data was obtained in order to ascertain the validation of the air pollution model. The TRL was able to provide a series of air quality and traffic data recorded at a kerbside location on the High Street in Ealing, London, for the purposes of testing this modelling procedure. The air quality data contains measurements of hourly average levels of CO, HC and NO<sub>x</sub> concentrations (in ppm) whilst the traffic data comprises hourly counts of the total traffic flow. This data was recorded between May and July 1991. The meteorological data was supplied to the TRL by the Meteorological Office. It comprises of hourly average measurements of wind speed (in knots) and wind direction (in degrees), recorded during the same period at the Heathrow Meteorological Station, which is the closest site to the Ealing monitoring location, at a height of 25 metres.

The hourly sets of data in which measurements of air quality, traffic or meteorological data have failed for technical reasons were excluded from the analysis. There remains 770 sets of complete hourly data.

Some adjustments had to be made to the model so that it could be run with the data available. First, the data relates to measurements carried out in 1991. Thus, the percent of cars fitted with catalytic converters in the car fleet was set to 2%, which is the estimate from the Department of Transport (1994b) for that year. The percent of diesel cars in the total fleet in 1991 was 3.72% (Department of Transport, 1992b). Wind speed was transformed from knots to metres per second and the direction of the wind was adjusted to the position of the road in relation to the meteorological station.

Some assumptions were also necessary due to the lack of precise data as required by the model. A correction factor of 0.75 was applied to wind speed, since it was measured much higher up than the locations where measurements of air pollution concentrations took place. Wind speed is supposed to decrease with the reduction in height, due to the friction with a range of natural and man-made obstructions, particularly in urban areas. Queues have been assumed to form at the junction closest to the measuring location, but at different levels throughout the different hours of the day. A queue profile was created, as shown by Figure 6.7, to be input in the model. The queue profile was estimated from the profile of traffic flows.

**Figure 6.7: Queue profile in High Street, Ealing**

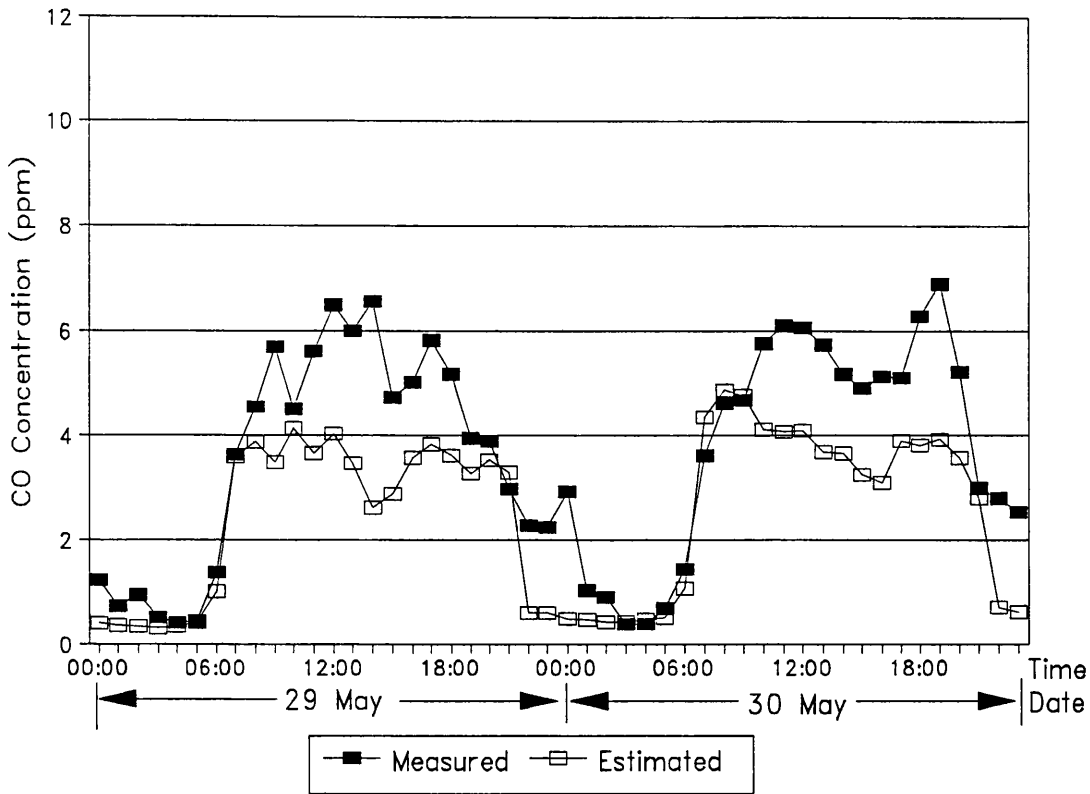


Considering the adjustments and assumptions described above, the air pollution dispersion model was run for the 770 hourly intervals for which the full set of air quality, meteorological and traffic data were available. This enabled comparisons of estimated and measured concentration levels not only of CO, but also HC and NO<sub>x</sub>.

Figures 6.8 to 6.23 illustrate the differences between measured and estimated values of average hourly CO concentrations in High Street (Ealing, London), from 29 May to 30 June 1991.



**Figure 6.8: Measured and estimated CO concentrations - 29 and 30 May 1991**



The pattern of variations between measured and estimated concentration levels of the other two air pollutants follow similar trends to CO. The parameters indicating the performance of the dispersion model in relation to the three pollutants analysed are shown in Table 6.5.

The occasions in which the line of measurements and estimations is discontinued in the graphs below (e.g. Figure 6.10) indicate that the observation of at least one of the input data to the air pollution model was unavailable.

Figure 6.9: Measured and estimated CO concentrations - 31 May and 1 June 1991

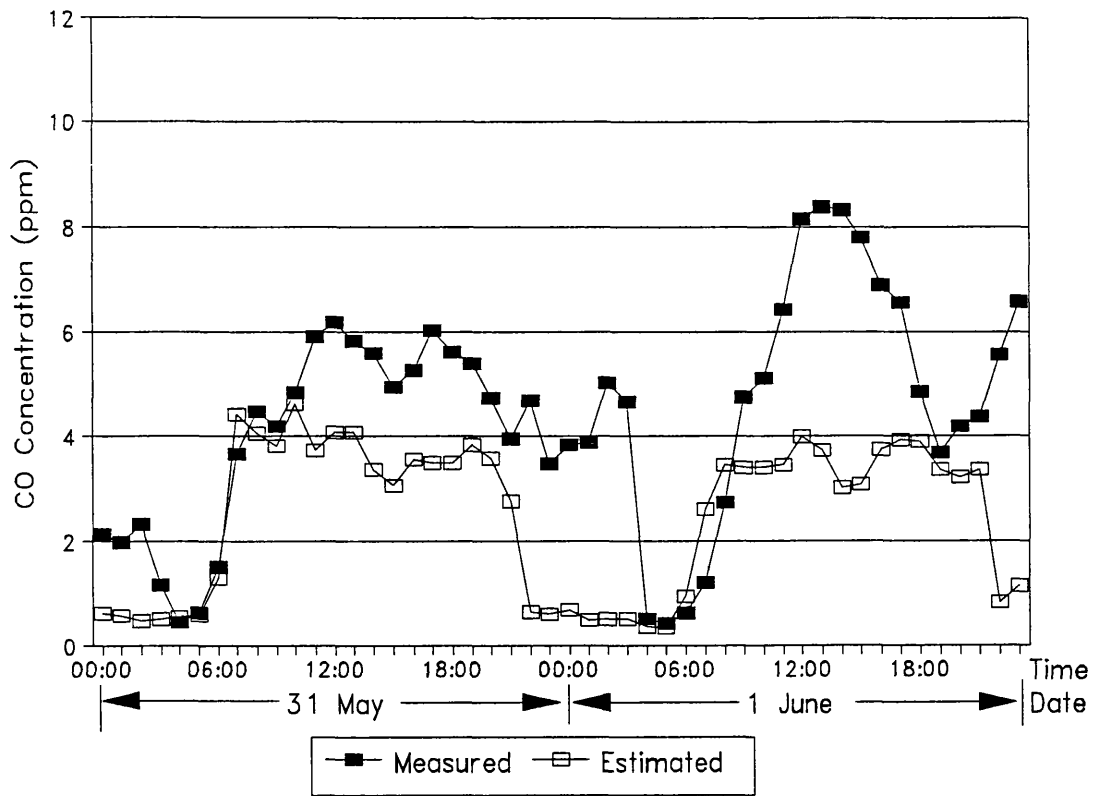
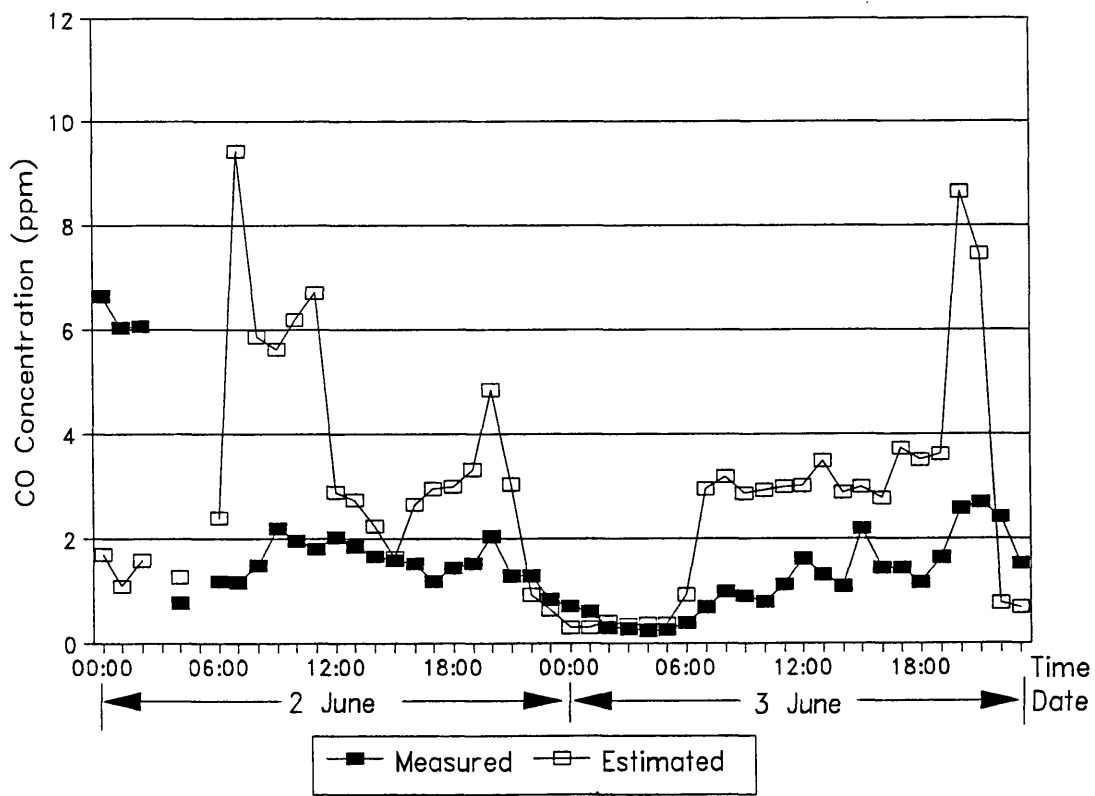
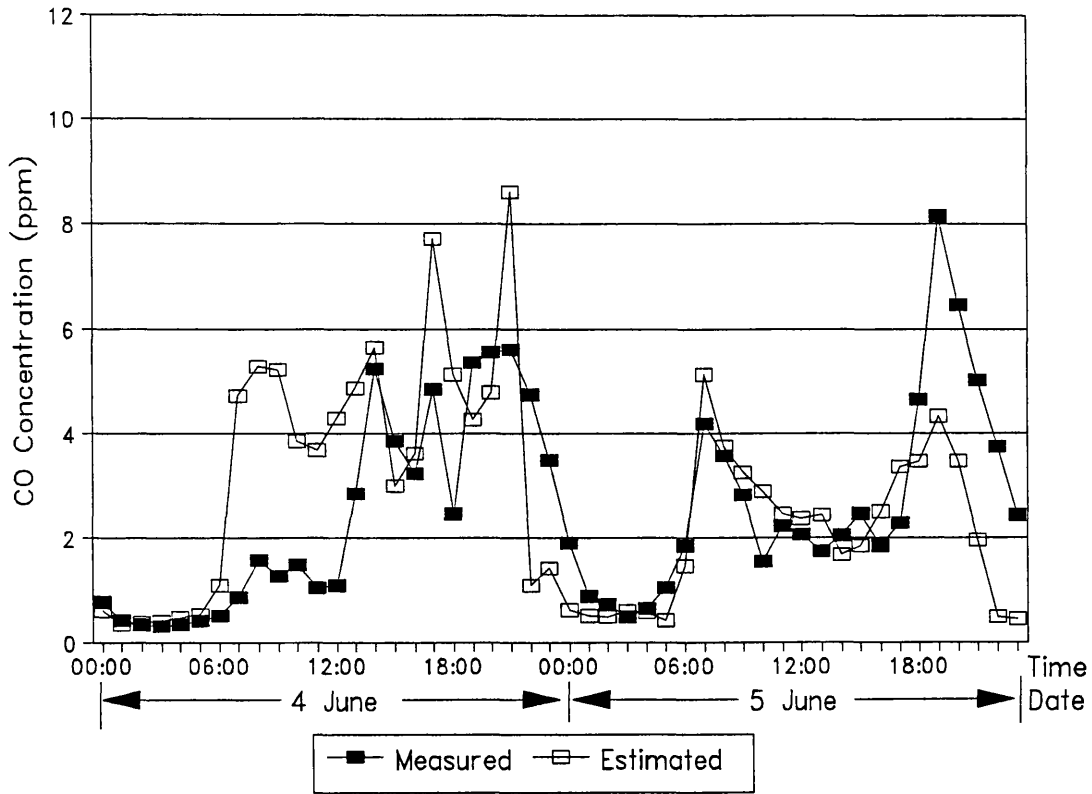


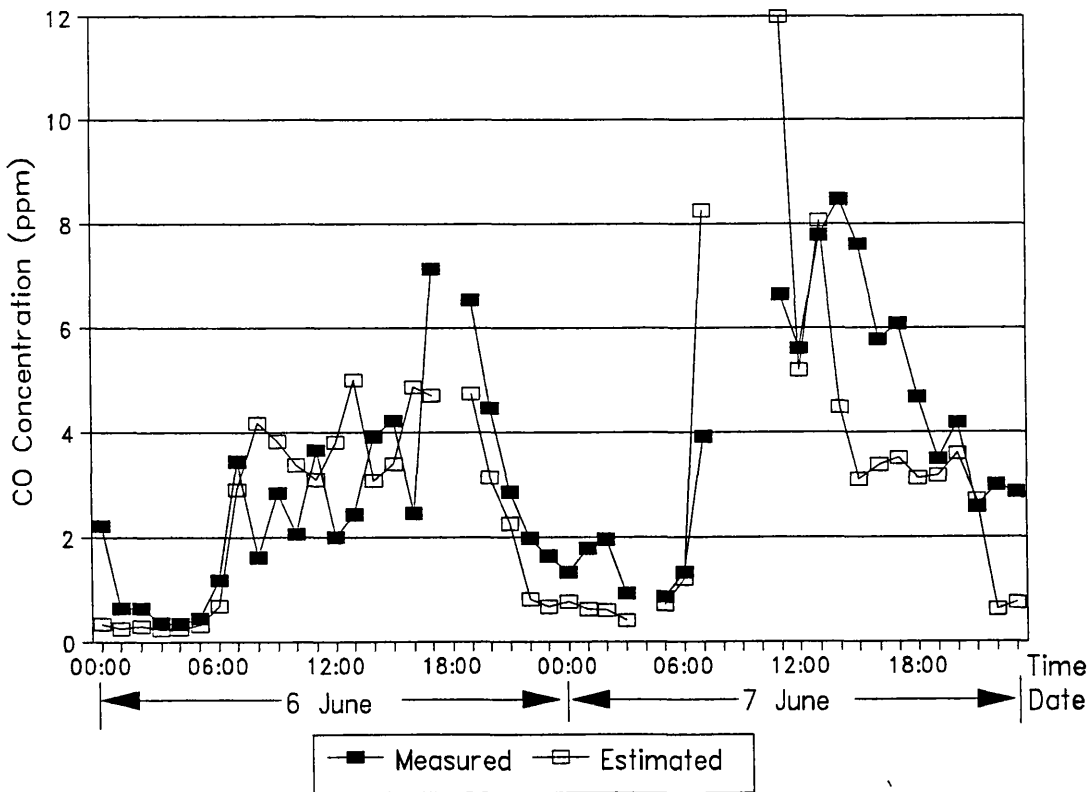
Figure 6.10: Measured and estimated CO concentrations - 2 and 3 June 1991



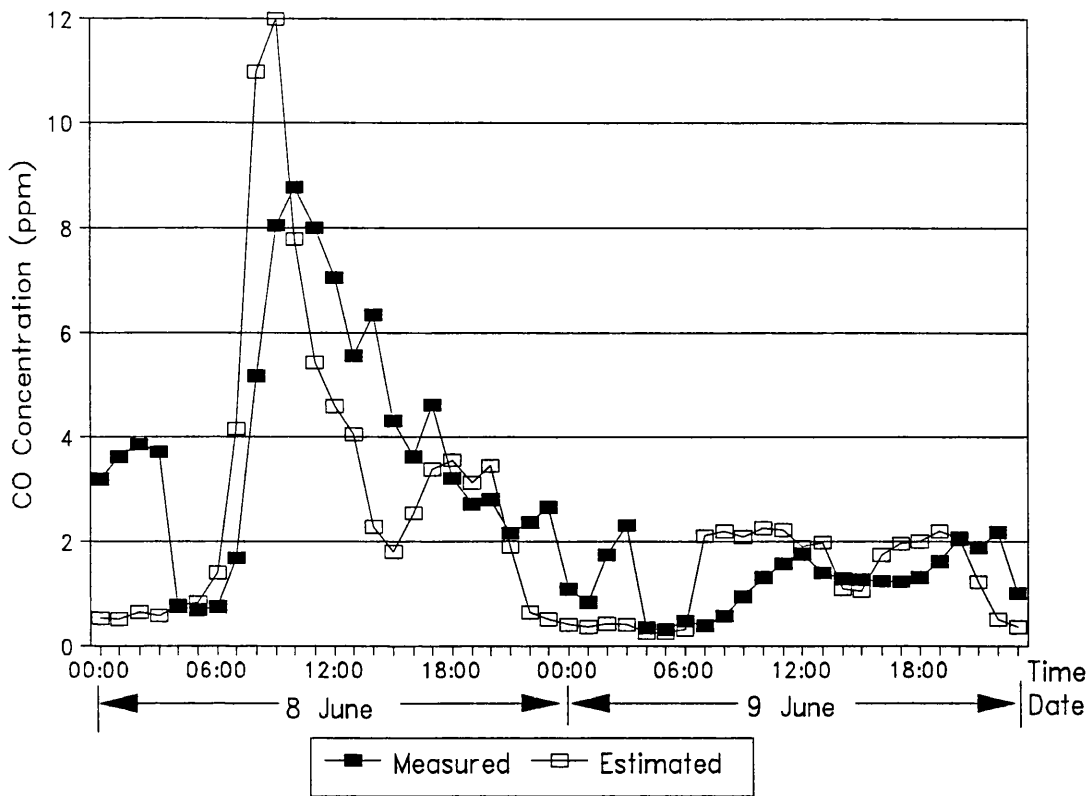
**Figure 6.11: Measured and estimated CO concentrations - 4 and 5 June 1991**



**Figure 6.12: Measured and estimated CO concentrations - 6 and 7 June 1991**



**Figure 6.13: Measured and estimated CO concentrations - 8 and 9 June 1991**



**Figure 6.14: Measured and estimated CO concentrations - 10 and 11 June 1991**

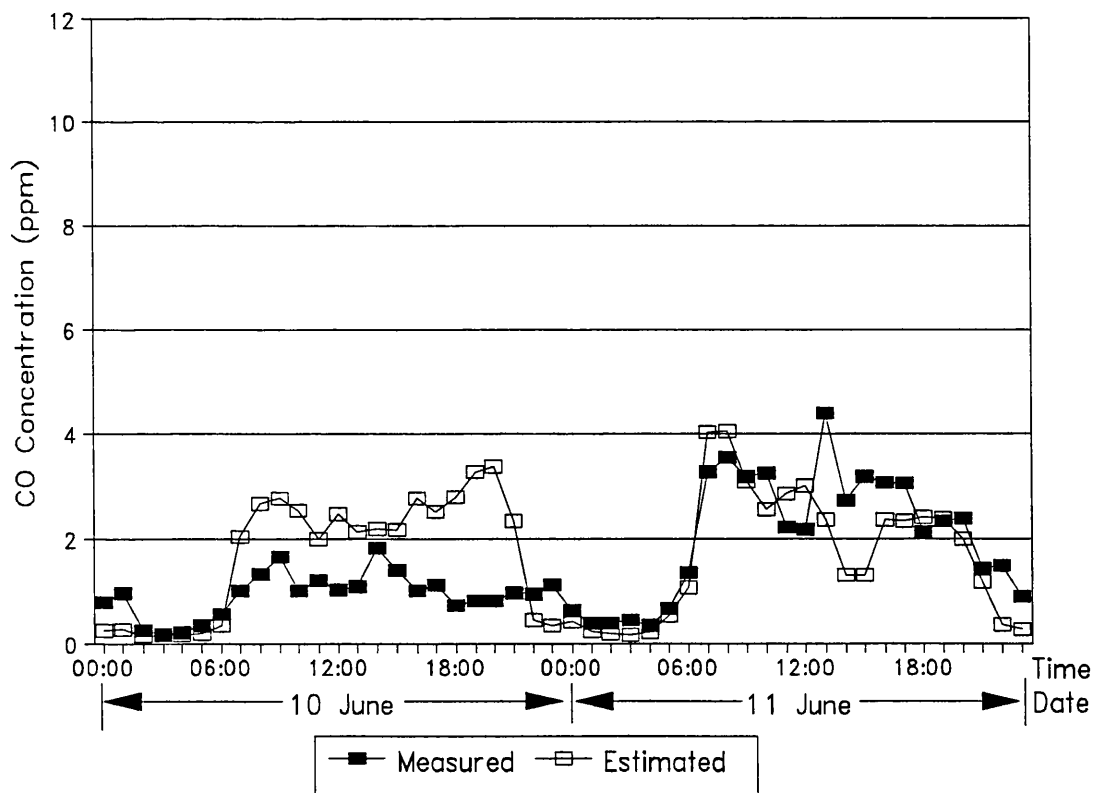


Figure 6.15: Measured and estimated CO concentrations - 12 and 13 June 1991

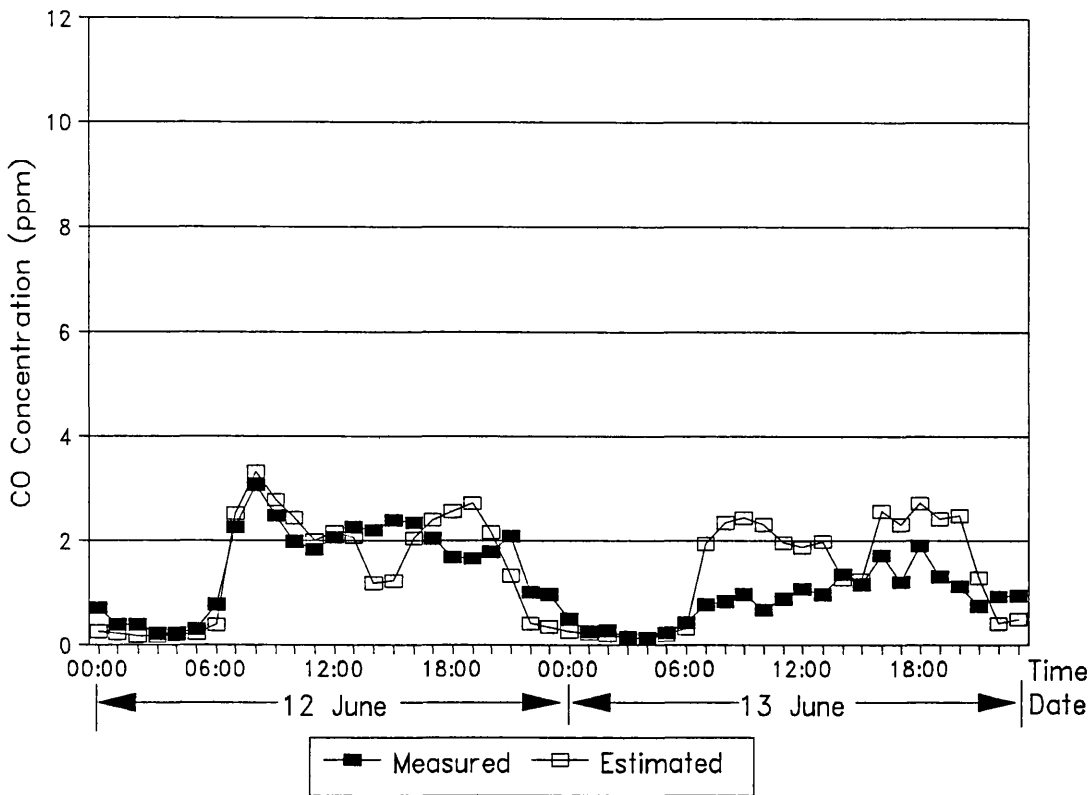


Figure 6.16: Measured and estimated CO concentrations - 14 and 15 June 1991

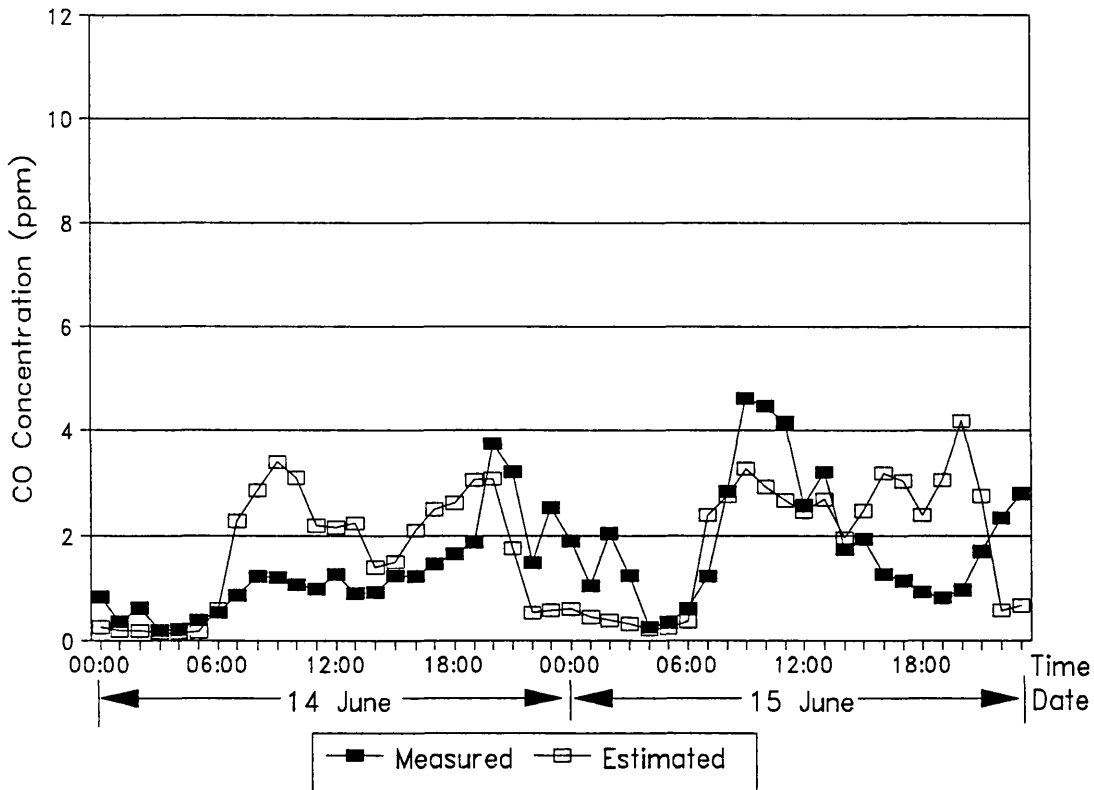


Figure 6.17: Measured and estimated CO concentrations - 16 and 17 June 1991

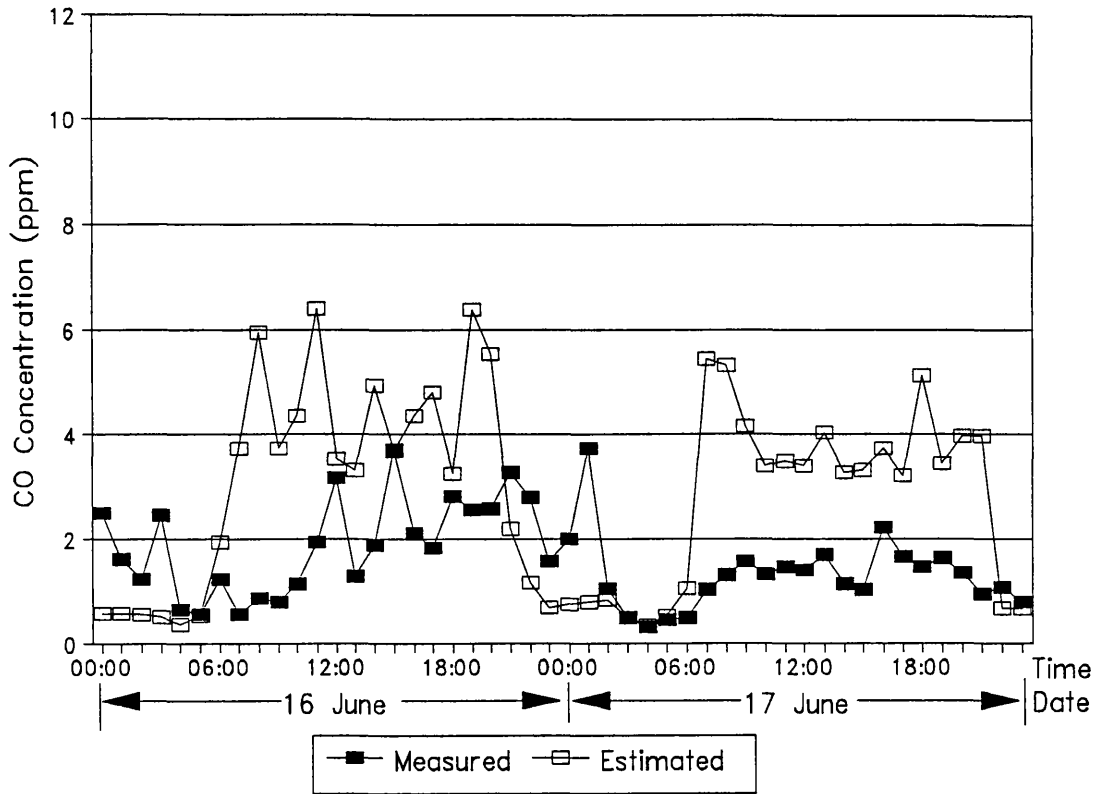


Figure 6.18: Measured and estimated CO concentrations - 18 and 19 June 1991

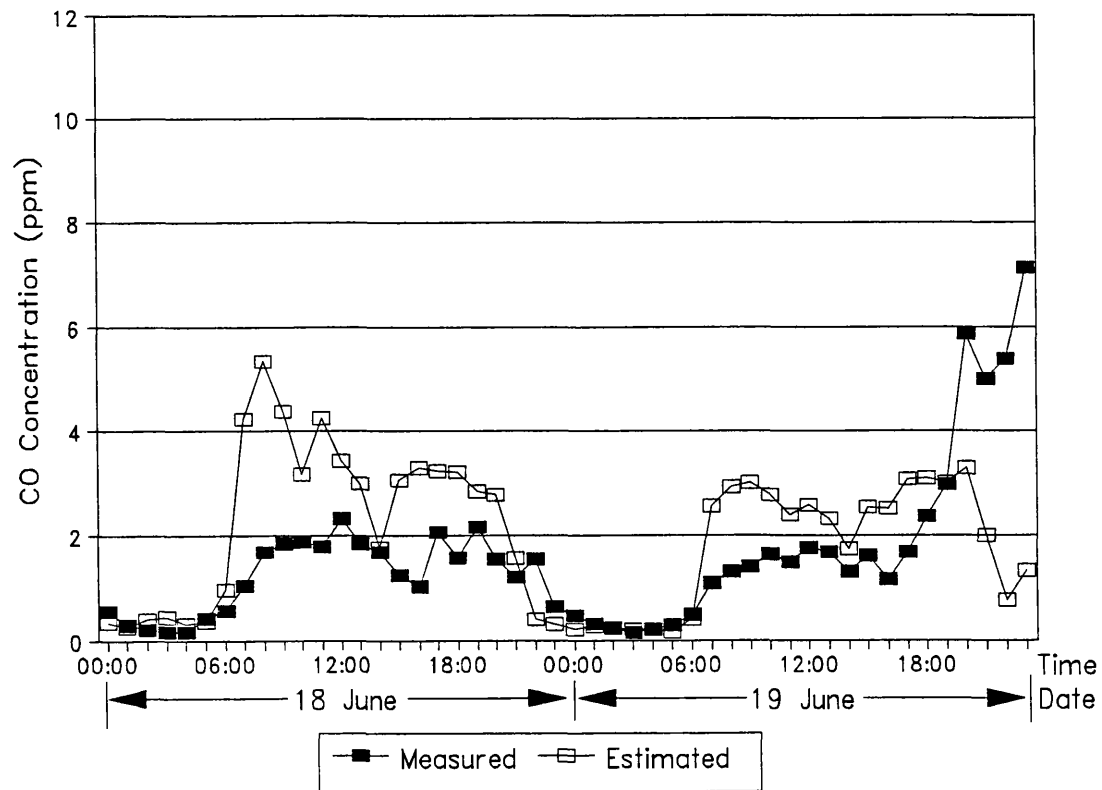


Figure 6.19: Measured and estimated CO concentrations - 21 and 22 June 1991

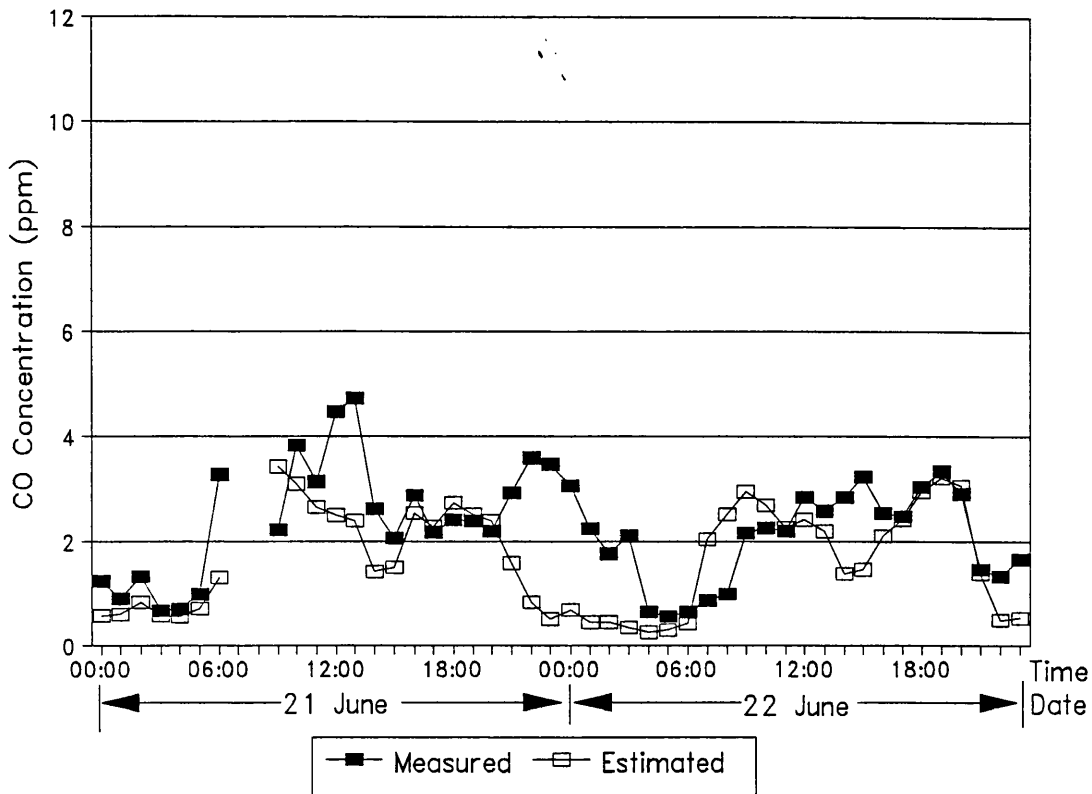


Figure 6.20: Measured and estimated CO concentrations - 23 and 24 June 1991

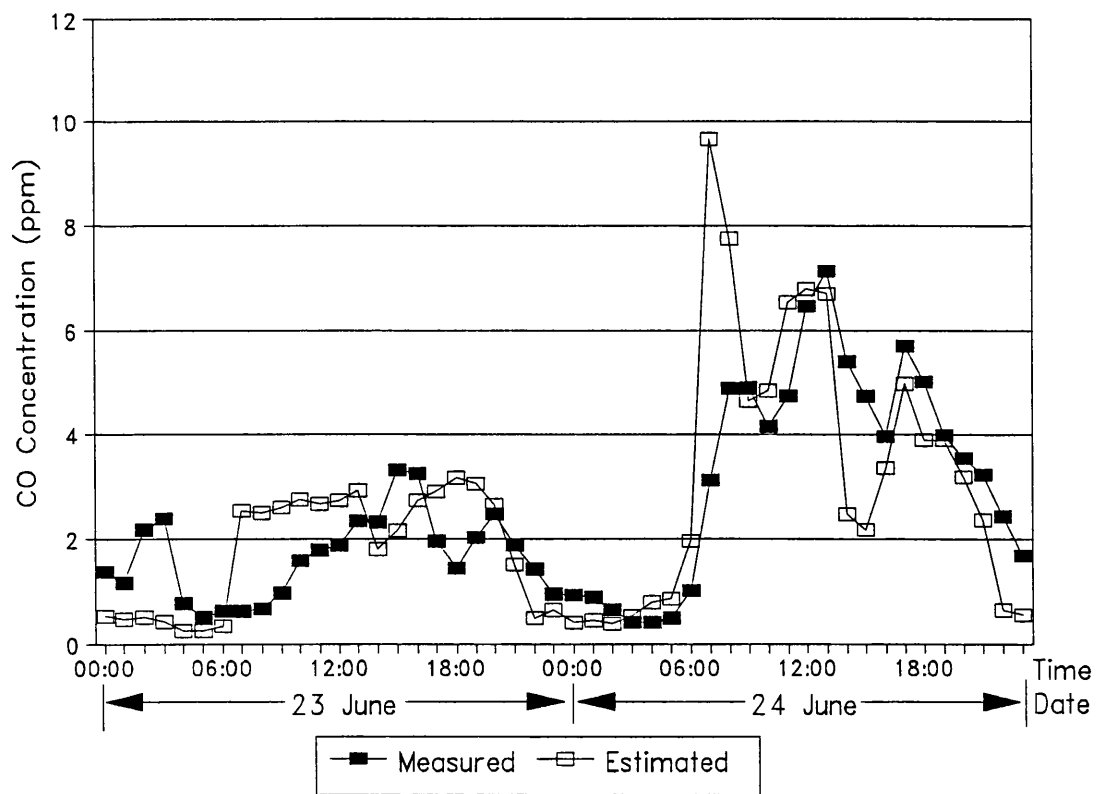


Figure 6.21: Measured and estimated CO concentrations - 25 and 26 June 1991

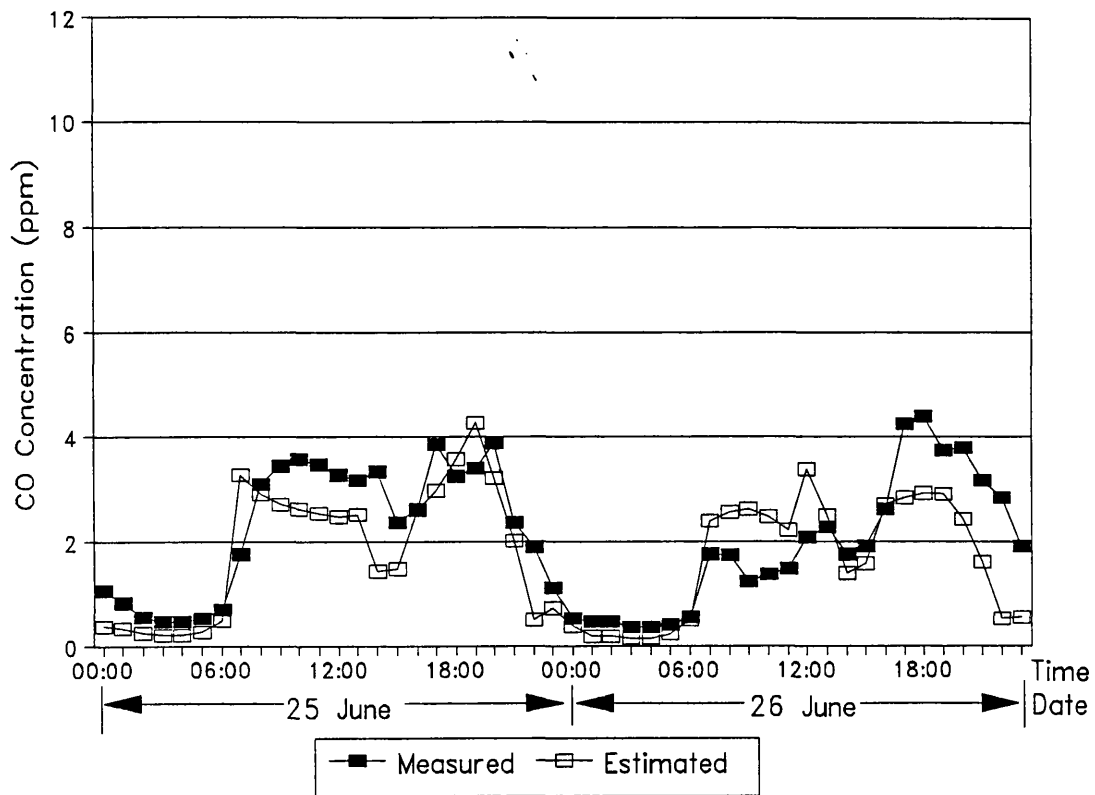
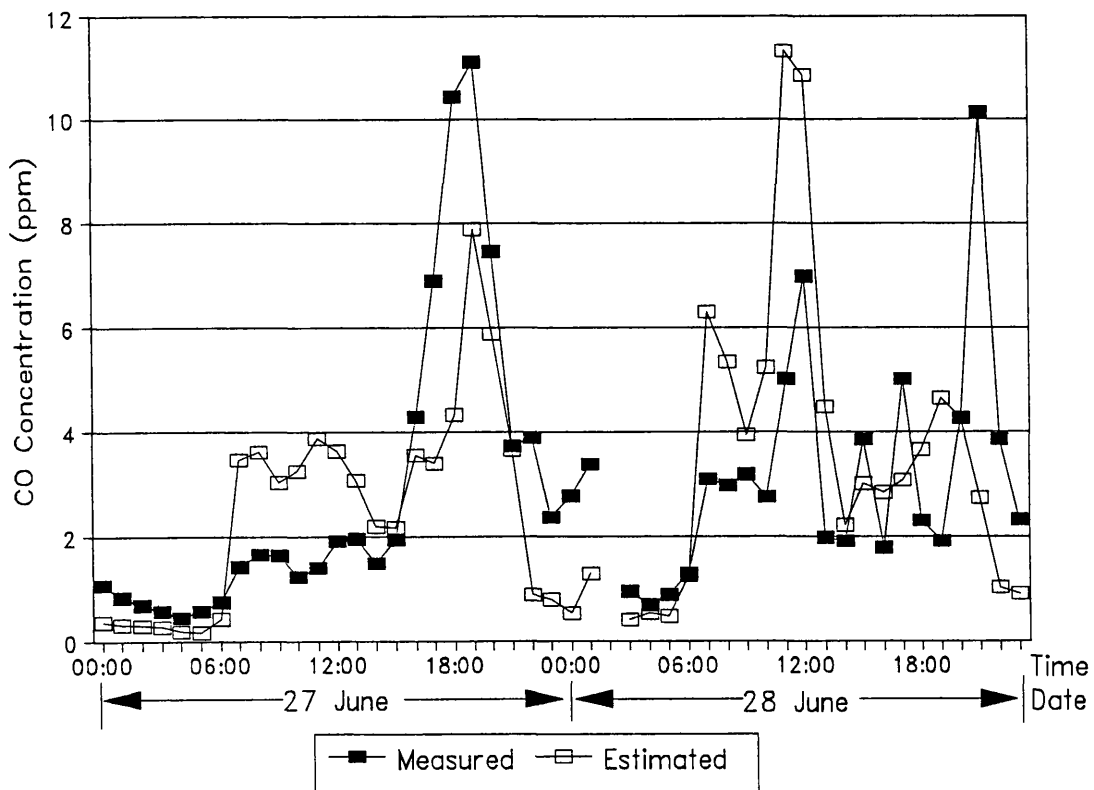
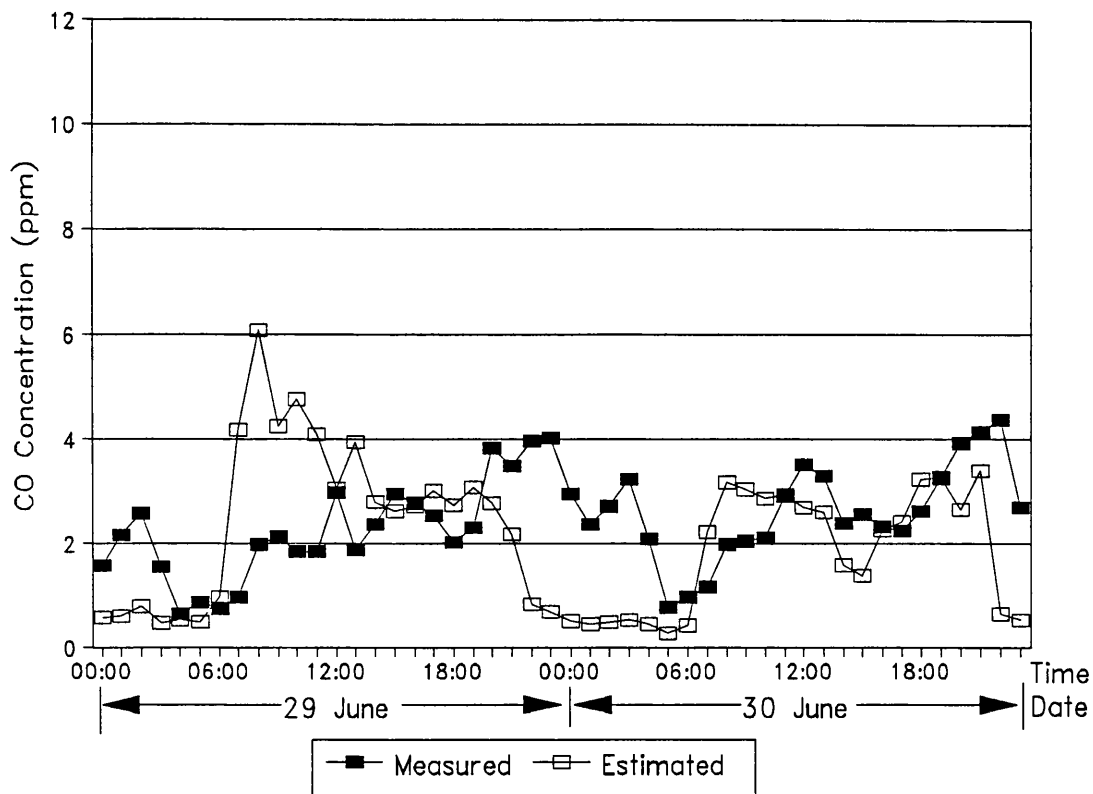


Figure 6.22: Measured and estimated CO concentrations - 27 and 28 June 1991





**Figure 6.23: Measured and estimated CO concentrations - 29 and 30 June 1991**



As can be seen in the illustrations above, most estimations have presented a fair correlation with measurements, and this is particularly evident in Figures 6.8, 6.11, 6.12, 6.13, 6.14, 6.15, 6.16, 6.18, 6.19, 6.20, 6.21 and 6.23. The fit was not so good in much of the time period shown in Figures 6.9, 6.10, 6.17 and 6.22. It can also be noticed that better fits are obtained in time periods when there is not much oscillation in the observation values, what suggests that the air pollution model performs better at more stable conditions.

One of the reasons why the performance of the air pollution model varies for different time periods is that meteorological factors, such as rain, humidity, temperature, pressure, atmosphere stability and occurrence of temperature inversions (see Section 3.5), which are not accounted for in the model, may have varied considerably along the period analysed. Such meteorological variations can have an important influence on the actual levels of air pollution concentration.

The parameters which illustrate the goodness-of-fit of the results of the air pollution model in relation to air quality measurements for the 770 observations are presented in Table 6.4.

**Table 6.4: Parameters of goodness-of-fit of the air pollution model**

| Parameter   | Pollutant |       |                 |
|---|-----------|-------|-----------------|
|   | CO        | HC    | NO <sub>x</sub> |
| Mean of the differences (ppm)                             | 0.135     | 1.262 | 0.032           |
| Mean of the absolute differences (ppm)                    | 1.327     | 1.309 | 0.060           |
| Standard error (ppm)                                      | 1.876     | 1.870 | 0.084           |
| Percentage of estimates within ± 50% of measurements (%)  | 51        | 37    | 52              |
| Percentage of estimates within ± 100% of measurements (%) | 83        | 100   | 96              |

The mean of the differences represents the average discrepancy between measurements and estimations within all the observations. It is, therefore, the expected value of any observation. As shown in Table 6.4, it is very low for all pollutants, what would suggest that the fit is very good. However, since there are positive and negative differences, the mean can be misleading as a parameter of performance. The mean of the absolute differences was then derived in order to provide a measure of real divergence between estimates and measurements. These figures are also considerably low. The estimates of CO and HC are in average about 1.3 ppm far from the observations and estimates of NO<sub>x</sub> diverge from the observations by an average of 0.06 ppm.

The mean alone is not enough to indicate the variation of a distribution. The most meaningful measure of the degree of scatter of the differences between measurements and estimates is the standard error. This parameter provides information on the absolute dispersion of a certain distribution in relation to the expected distribution, the higher the standard error the greater the amount of scatter. Therefore, standard error is a measure of the quality of fit. The mathematical expression for standard error is given in Equation 6.7.

$$Standard\ error = \sqrt{\frac{\sum_{i=1}^n (E_i - M_i)^2}{n}} \quad \text{Equation 6.7}$$

E<sub>i</sub> = Estimated values

M<sub>i</sub> = Measured values

n = number of observations

The percentages of estimates which lie within  $\pm 50$  and 100% of the observations are not considerably elevated, but they are still in line with the findings from previous studies on the validation of air pollution modelling procedures.

In conclusion, it has been emphasised that there is much scope for inaccuracy in air pollution modelling and high levels of precision cannot be expected. Despite the constraints in the accuracy of predictive models, they are likely to produce more reliable results when used to compare the effects of different policies, i.e. when relative rather than absolute concentrations are derived. This is indeed the case for the estimates produced in this work, and many of the inaccuracies and simplifying assumptions used in the application of the air pollution model in different situations should be cancelled out. Thus, the relative differences between estimates of air quality from the base scenario in comparison with the predictions related to other scenarios, which consider various changes in transport policy, can represent relevant information for supporting decision-making.

### **6.3 Noise modelling**

Most of the existing models for the estimation of traffic noise have been developed and applied to highway conditions. This suggests, first, that highway traffic noise is perceived as a major problem and, second, it is much more straightforward to predict. However, as reported in Section 4.1, urban traffic noise has become one of the main causes of disturbance, principally in residential areas.

The model in the SPITE developed to estimate local noise levels in urban areas follows the recommendations contained in the CRTN method (Department of Transport and Welsh Office, 1988), which were described in Section 4.4. The model is focused on situations typical to urban areas, particularly in relation to the complex geometry of the road network, the existence of reflecting facades on buildings along the roads, the short distance between source and reception and multiple contributions of traffic at road junctions. Another particular feature of the model is the prediction of traffic noise at a grid of receptor points throughout any road network, which is an advance in relation to the original method. This is useful for the analysis of the impacts of traffic changes in different parts of the network.

The level of noise predicted is  $L_{10-1 \text{ hour}}$ , which is the time period compatible with the traffic data produced by the assignment model and with the predictions of air pollution

concentrations. The principal assumptions utilised in the noise modelling are:

- traffic speed is the average speed on the road for all vehicles.
- no facade correction is applied, since there is no facade behind the estimating points (see Equation 4.10 and Figure 6.24).
- no noise obstructions (barriers, trees, etc.) are situated between the source (traffic) and receptor (on the pavement).
- the noise level contribution from road segments screened by buildings is assumed negligible.
- all roads surfaces are either concrete or impervious bituminous and all speeds in the road network are under 75 km/h, which means that 1 dB(A) is subtracted from the basic noise level (see Equation 4.12).

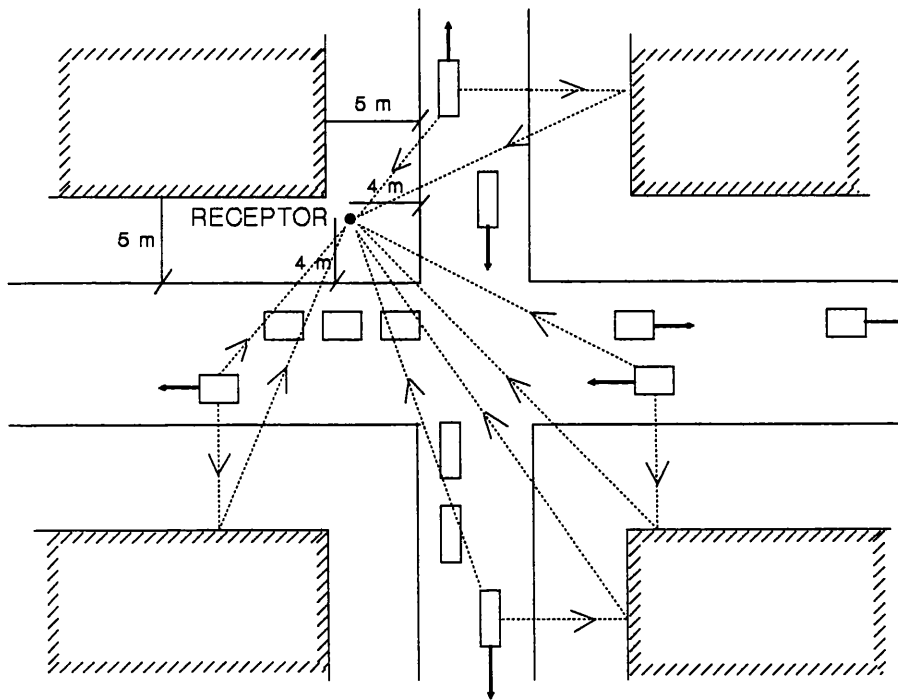
Like the air pollution model, the noise model in the SPITE initially incorporates the traffic data produced by the traffic assignment model, namely: car and bus flows, road lengths, number of lanes and travel times related to each road, and reads the coordinates of all road junctions. The noise model then calculates traffic speed, road widths and the percentage of heavy vehicles on the flow (this consists of the proportion of buses and lorries on the total flow), as required by the predictive method. The model also calculates the angles of all roads in relation to the grid of node coordinates.

The locations where noise levels are estimated could not be the same as where pollutant concentrations have been predicted (at one metre from the kerbsides, see Section 6.2), because the noise method is not valid for distances less than four metres from the traffic stream. For predictions at distances less than four metres from the kerb, the method recommends that predictions are made at the minimum distance of four metres. Thus, the locations where noise levels are estimated have been defined as the points situated downstream of each road, on the nearside pavement of the lanes where queues may form, at four metres from each kerb of the junction. Like the air pollution model, the noise model finds the coordinates of all receptor locations, which are defined as the crossing points of the lines parallel to the roads at four metres from the kerb, on the pavement (this is illustrated

by Figure 6.24 below).

As urban areas are in general fully built up, it was assumed that there are building facades on both sides along all roads in the network. For purposes of the noise model, continuous noise reflecting facades were assumed to be five metres from all kerbs (one metre further in from the points where noise levels are predicted). Figure 6.24 illustrates the position of a receptor point and the geometry of reflecting facades on buildings around a symmetric road junction. This figure also shows the various traffic contributions to noise levels, both directly and via reflection, at the receptor point.

**Figure 6.24: Traffic contributions to noise levels at a receptor around a road junction**



The noise contributions at the receptor point from particular vehicles travelling on different roads around the junction are indicated in Figure 6.24. However, as vehicles move constantly, the model considers road segments the sources of noise rather than individual vehicles. Since building facades reflect the sound waves and contribute to the increase in the noise level at the receptor point, the sound propagation correction is applied to all road segments by considering the geometry of the building facades at each intersection.

In order to apply the sound propagation corrections (see Equation 4.11), the noise model determines the coordinates of building facades along all roads and calculates the angles of view and reflection from each receptor point in relation to all building reflecting facades and road segments around the junction. The model also verifies whether the angles of reflection are obstructed by other facades (this happens very often when the road junction is asymmetric). When this is the case, the model recalculates the unobstructed angles of reflection. As a result of various geometry combinations within the network, the angle of reflection can sometimes be bigger than the angle of view, and the model sets the former equal to the latter in all situations as such.

When accounting for the noise contribution from various roads at a particular junction, the noise model requires that all source lines (road segments) are extended so that the perpendicular distance between receptor and all road segments can be determined. However, the extended source line of some road segments often passes through or very close to the receptor point. The distance correction in the noise method is valid only for distances above or equal to four metres from the extended edge of the nearside carriageway along source lines. If such distance is less than the lowest valid gap, the receptor point is artificially replaced at the recommended minimum distance and all the angles of view and reflection are recalculated.

The height of the points where traffic noise levels have been estimated was defined at 1.2 metres, as recommended in the CRTN method. Since cars are usually assumed 0.5 metres high, the relative height between source and receptor (which is an input in Equation 4.8) becomes 0.7 metres.

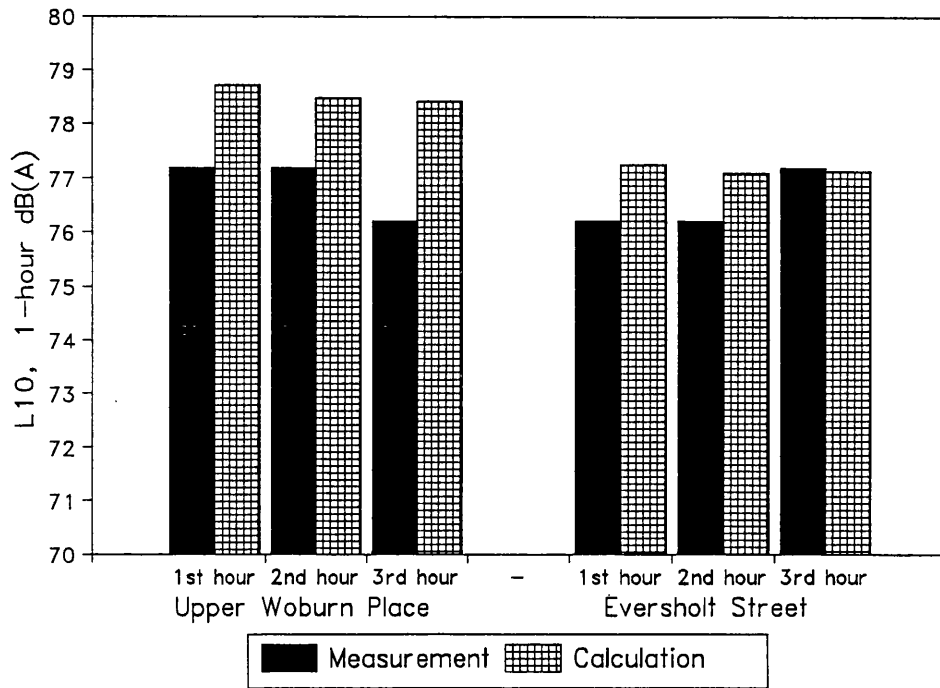
### *6.3.1 Accuracy of the noise model*

The CRTN method is recommended by the British Government in the Design Manual for Roads and Bridges (Department of Transport et al., 1993), for the measurement and estimation of traffic noise levels. It also enables entitlement under the Noise Insulation Regulations 1975 (Great Britain Statutory Instruments, 1975).

In order to demonstrate the degree of accuracy of the CRTN method, a series of field surveys were carried out in urban sites in London. The sites surveyed were at Upper Woburn Place and Eversholt Street, on 14 and 16 December 1992 respectively. The data collected consisted

of traffic flows, speed, road layout (reflecting facades on buildings) and noise levels  $L_{10,1\text{-hour}}$ . Traffic noise levels were also estimated by inputting the data collected to the CRTN method. The comparison of measured and calculated figures is shown in Figure 6.25.

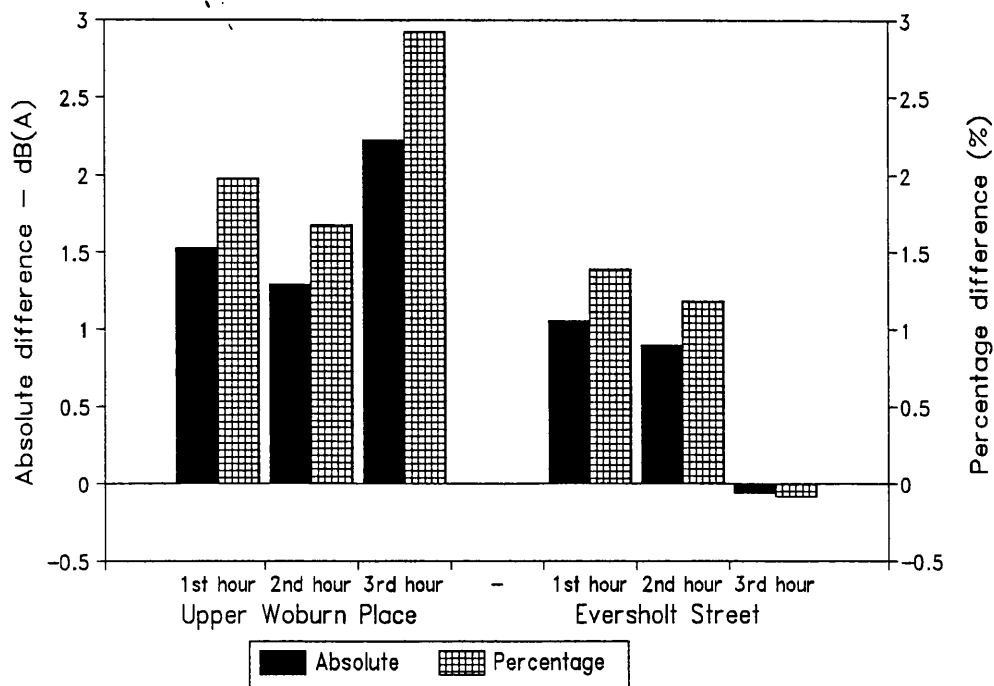
**Figure 6.25: Comparison between measured and calculated noise levels**



The shortened procedure provided by the CRTN method (Equation 4.15) can be used to produce estimates of  $L_{10, 18\text{-hour}}$  based on the predictions of  $L_{10, 1\text{-hour}}$  over three consecutive hours. This procedure was used to enable comparisons of the results of short-period predictions with the environmental standards, which are set for 18 hour periods. It was verified that the recommended maximum noise levels were exceeded at all intervals in the sites surveyed, reflecting unacceptable levels of environmental degradation.

The absolute (in dB(A)) and relative (in percentage terms) differences found between measurements and calculations of noise levels are illustrated in Figure 6.26.

**Figure 6.26: Differences between measured and calculated noise levels**



The results from this somewhat limited survey show that the CRTN method tends to overestimate the measured noise levels. The prediction level is under the measurement level in only one interval, but the magnitude of the difference is negligible. The highest over-prediction reached 2.2 dB(A), or 2.9%, which suggests that the predictions in this study can be subject up to +3% inaccuracy. The results of the noise model in the SPITE are considered to have met closely the figures found from measurements. The level of accuracy can be considered quite satisfactory, principally if compared with the average differences of about 10 dB(A) in absolute terms, or about 20% in relative terms, between measured and calculated noise levels in 17 locations, as described in Barlow et al. (1992).

#### 6.4 Model application in Chester

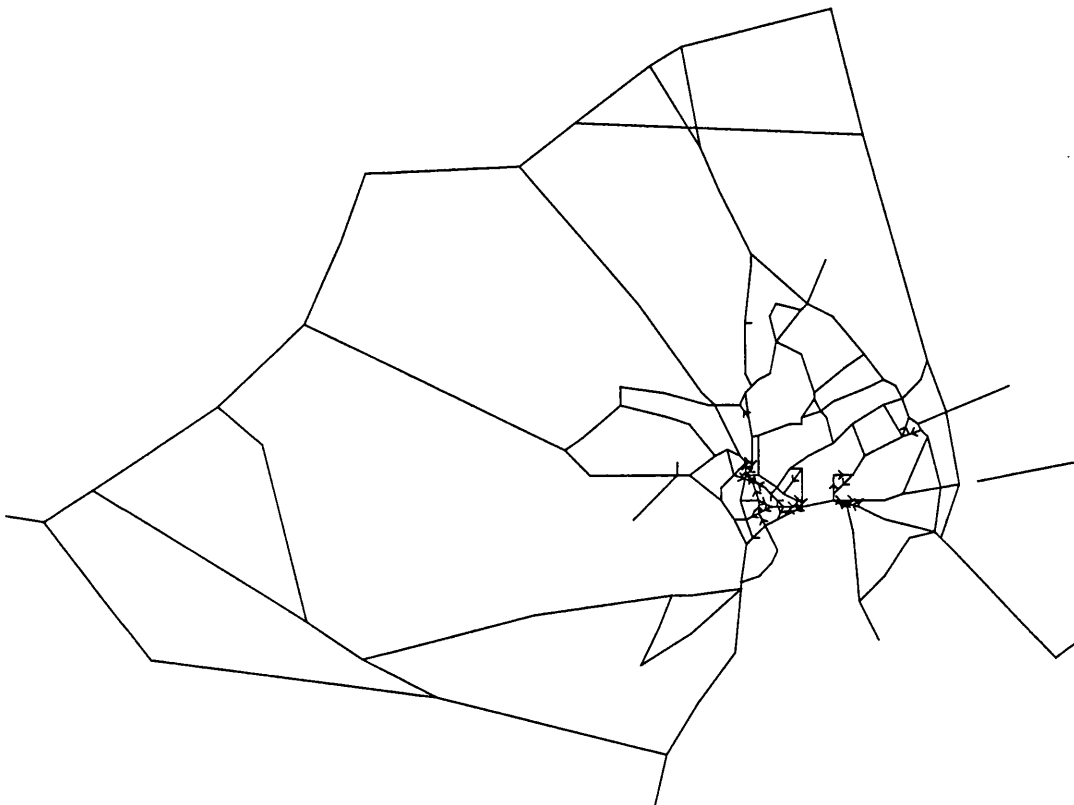
The models for simulating traffic assignment and for estimating air pollution and noise levels, as described in Sections 6.1, 6.2 and 6.3, have been applied to a case study. The case study aims to test the application of the SPITE for assessing the environmental impacts of various transport policies in urban areas. The urban area selected is in Chester, an historic city situated in the north-west of England, with population of about 78,000 inhabitants.



The urban area of Chester has been chosen as the study case mainly because most of the data necessary for modelling were readily available. A great deal of the road network has been coded and the survey which produced the origin-destination trip matrix was carried out in recent years. Besides, the data related to bus routes and frequencies in Chester have also been produced recently.

The road network in Chester comprises 94 zones, 284 nodes and 415 links. Some changes have been made to the initial network coding, particularly in relation to the addition of links in which traffic calming and pedestrianisation schemes are being implemented by the County Council. The addition of such links is essential for the assessment of the local impacts generated by traffic. Figure 6.27 illustrates the road network of Chester produced by the traffic assignment model.

**Figure 6.27: The Chester road network**



The increasing demand for private transport in Chester has caused problems such as congestion, delays, traffic queues and air pollution, mainly in the peak periods. Traffic

congestion has had a depressing effect on the commercial development and on the quality of life of residents and tourists. The Chester County Council has been concerned with improving the quality of life of residents and conserving the quality of the architectural fabric of buildings and other features in the historic centre (Cheshire County Council, 1990). Thus, the commercial and residential attractiveness of the city may well be enhanced by proper air pollution control.

The actual levels of environmental degradation in Chester allied to its historical characteristics have provoked political and social interest in the environmental problems created by traffic. The concern with environmental issues in Chester is featured in a recent study about the future developments and the resulting traffic impacts in the historic city (Building Design Partnership et al., 1994 and Local Transport Today, 1995b). This study includes the design of a methodology to identify the city's environmental capacity and the application of this approach for the noise nuisance criterion in relation to diverse land uses, in twelve radial links crossing the Chester cordon. Another recent report reveals that Chester is in serious danger of becoming a pollution blackspot, with levels of NO<sub>2</sub> in the city centre above the recommended safe level (Davies, 1995).

The application of the assignment model has loaded the trips comprised in the origin-destination matrix (which represents the travel pattern in the afternoon peak hour) onto the road network of Chester. This procedure has produced the main indicators of traffic characteristics in the whole road network, within the simulation period. These indicators include the total travel time, total distance travelled, average speed and fuel consumption. They are shown in Table 6.5 below, and will be compared with the traffic indicators from the simulations of transport policies and trends carried out in Chapters 8 and 9.

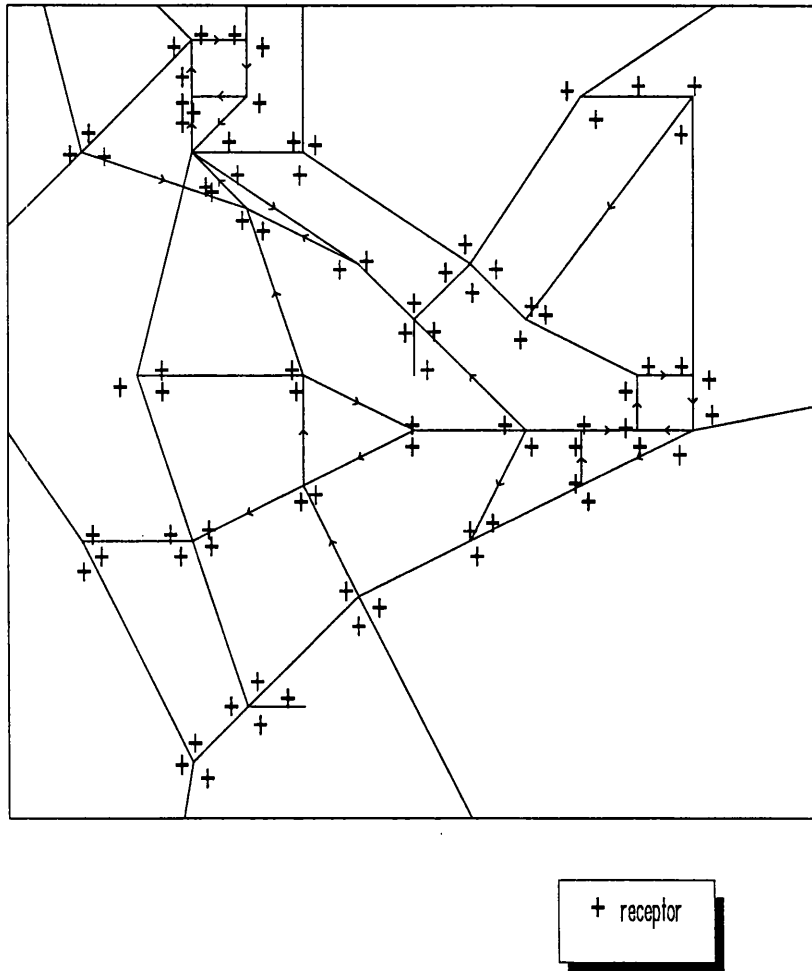
**Table 6.5: Main traffic indicators in Chester**

| Travel time (pcu.h) | Distance (pcu.km) | Speed (km/h) | Fuel consumption (litres) |
|---------------------|-------------------|--------------|---------------------------|
| 3,392.7             | 44,119.6          | 13.0         | 3,491.3                   |

The results of the assignment model have also indicated the spatial variability of traffic congestion throughout the road network. Similarly, the application of the SPITE has shown the spatial variability of degradation levels in the network. As expected, it was very

noticeable that the highest levels of environmental degradation occurred consistently in several locations of the central area. The central area can be easily identified in Figure 6.27 by the elevated density of roads. Figure 6.28 presents the central area at a larger scale and also shows the receptor points, where air pollution concentrations and noise levels have been estimated.

**Figure 6.28: The central area in Chester**

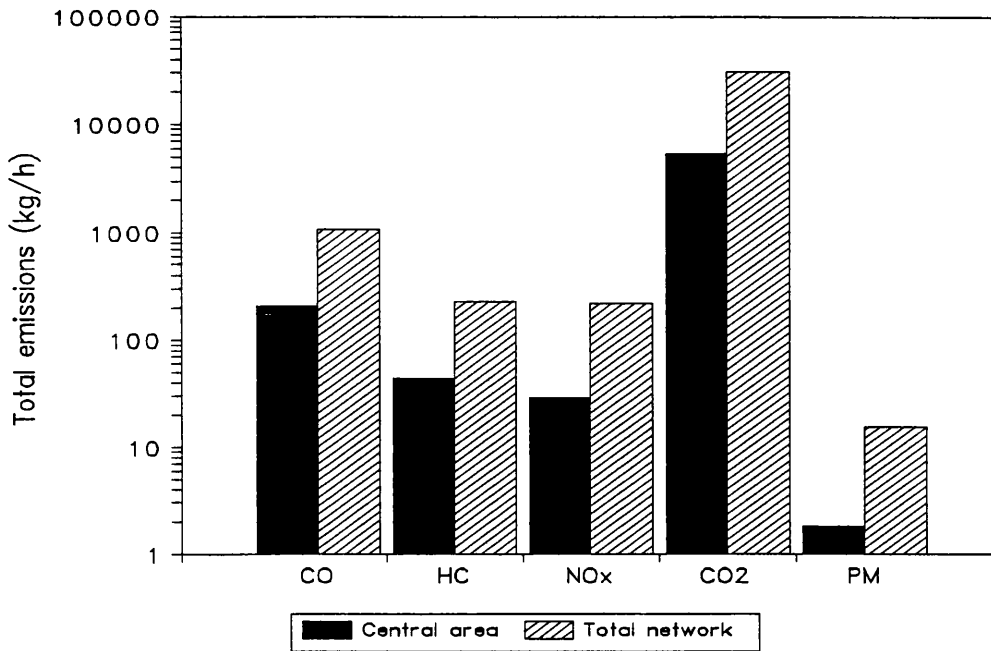


The total traffic emissions produced in the study area have been calculated by the emission model in the SPITE. Figure 6.29 shows the total exhausts (in kilograms within the one-hour modelling period) emitted in the central area and in the total urban network of Chester.

The magnitude of the total emissions are significantly different for each air pollutant. This graph is given in a logarithmic scale, in order to make it possible to represent the total

emissions of different pollutants on the same graph. It can be seen, however, that the total emission of PM from traffic sources is under 2 kg/h in the central area and just over 15 kg/h in the total network, whereas the emissions of CO<sub>2</sub> from traffic exceed 5,300 kg/h in the central area and 30,600 kg/h in the entire network.

**Figure 6.29: Total emissions in Chester**



The dispersion model in the SPITE produced data on the concentration level of each air pollutant in locations throughout the road network. However, since the road network of Chester comprises a large number of links and therefore has to be represented in a small scale, it is difficult to illustrate the spatial variability of environmental degradation throughout the whole city. Thus, the various levels of environmental degradation are shown for the central area only. Figures 6.30 to 6.34 illustrate the concentration levels of the different air pollutants analysed in this work, estimated by the application of the air pollution model (reported in Section 6.2) in the central area of Chester.

Figure 6.30: Concentrations of CO in the central area

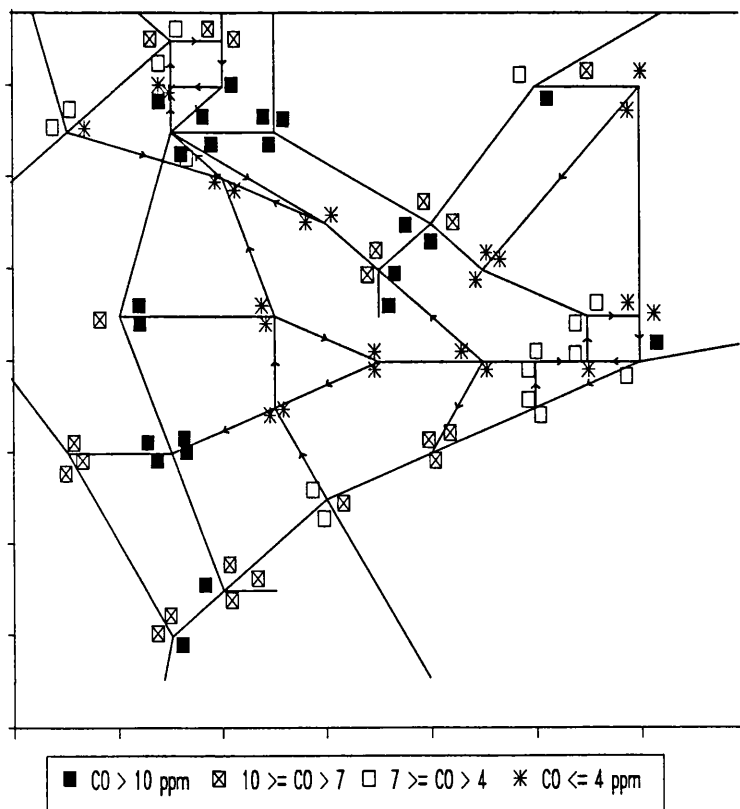


Figure 6.31: Concentrations of HC in the central area

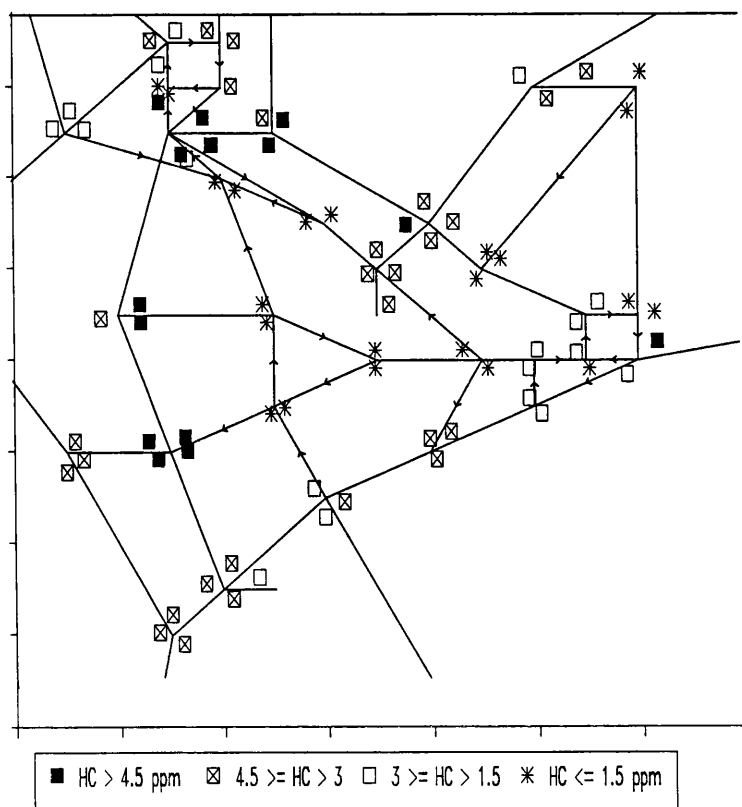


Figure 6.32: Concentrations of NO<sub>x</sub> in the central area

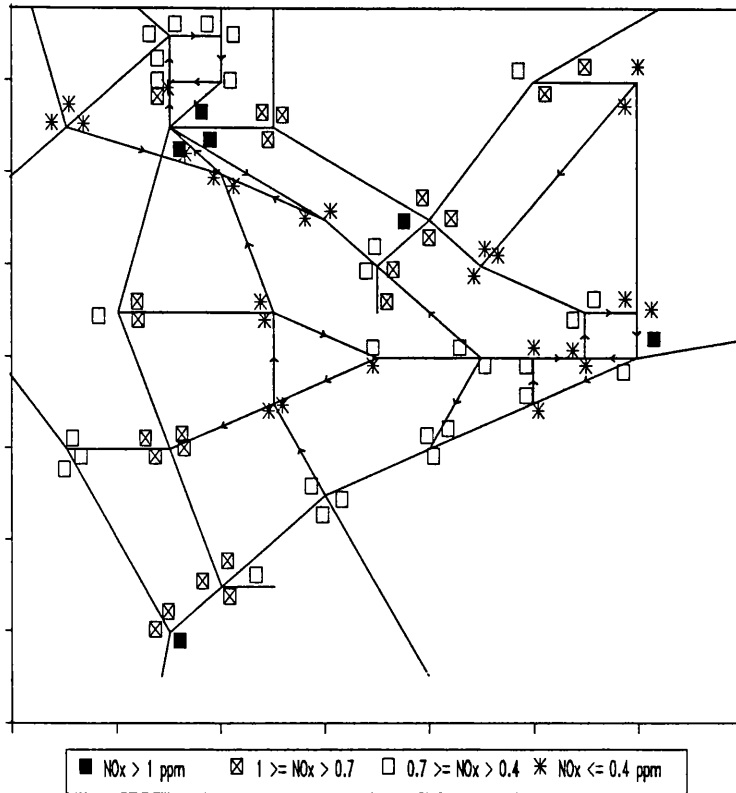
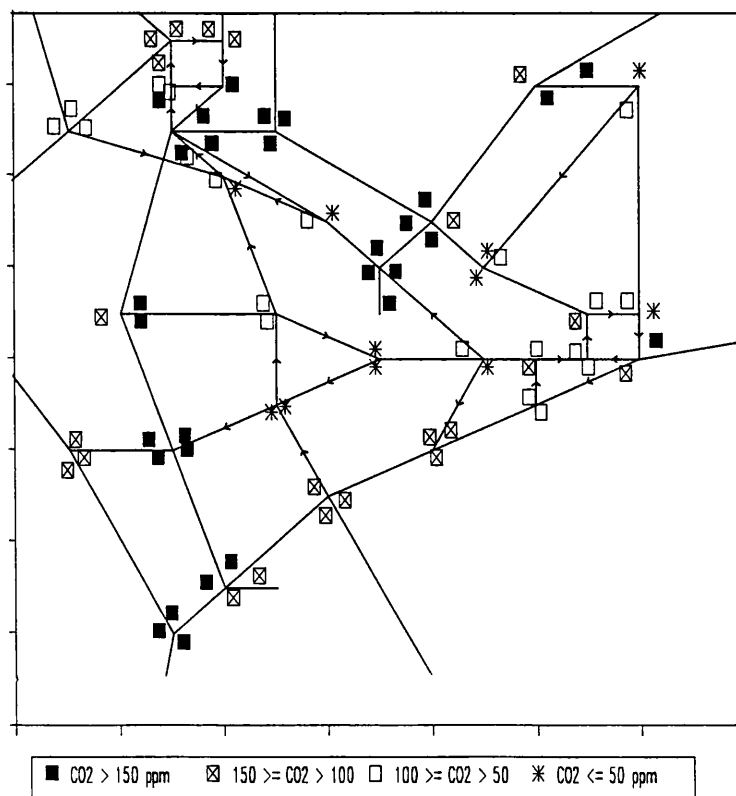
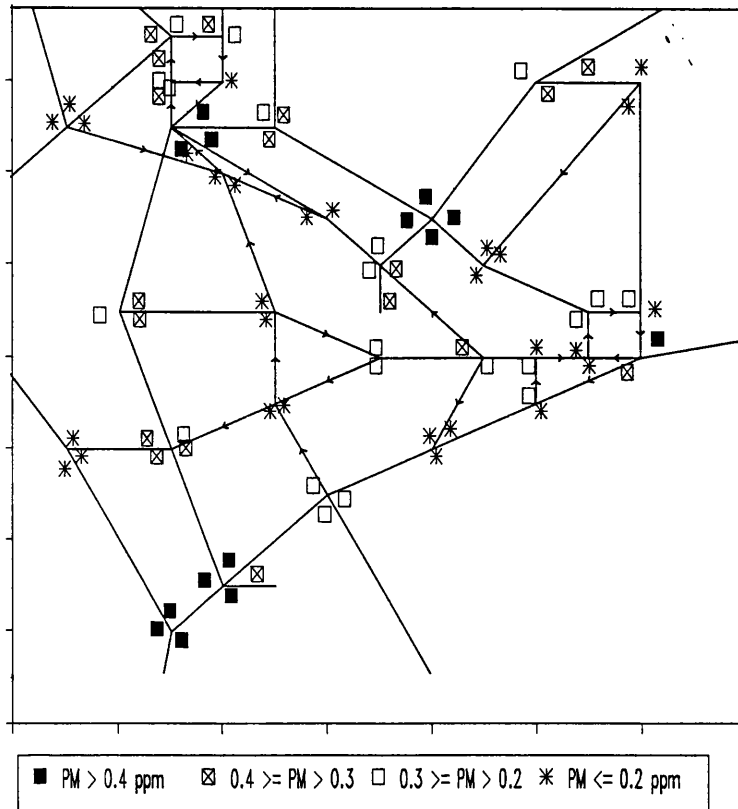


Figure 6.33: Concentrations of CO<sub>2</sub> in the central area



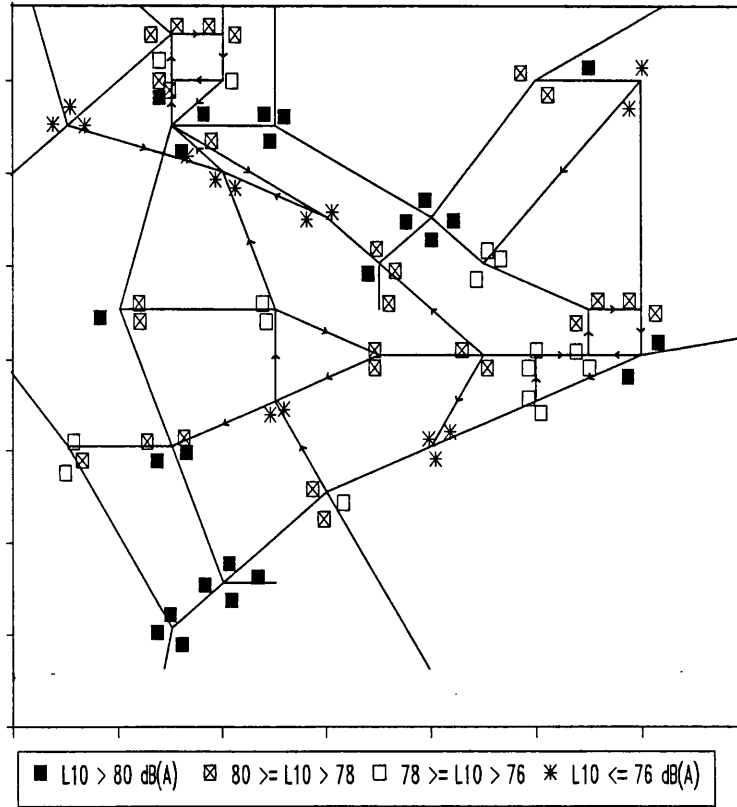
**Figure 6.34: Concentrations of PM in the central area**



The predicted levels of air pollution concentration in the central area can be considered very elevated, especially around certain busy road junctions. The concentration of CO exceeded the level of 10 ppm in over twenty locations (as shown in Figure 6.30) and also exceeded the recommended standard for one-hour period (as given by Table 3.7) in a few places. The concentration levels of the other pollutants cannot be related to the air quality standards. The HC exceeded the concentration level of 4.5 ppm in fourteen locations in the central area (Figure 6.31), often coinciding with the locations where levels of CO are most elevated. Levels of NO<sub>x</sub> and PM concentrations exceeded 1 and 0.4 ppm, in six and fourteen locations, respectively (Figures 6.32 and 6.34). The concentrations of CO<sub>2</sub> in the central area surpassed the level of 150 ppm in nearly 30 locations (Figure 6.33). The elevated levels of CO<sub>2</sub> concentrations found (tenths of times higher than the predicted levels of the other pollutants) were caused by the amount of this pollutant emitted by traffic (Section 6.2.1). As reported in Section 3.2.4, CO<sub>2</sub> is a natural constituent of the air and elevated concentrations in the atmosphere are very common. However, the predictions of the concentration levels of this pollutant account only for the traffic-induced emissions and therefore are likely to be underestimated.

Figure 6.35 shows the  $L_{10, 1\text{-hour}}$  traffic noise levels in the central area, predicted by the application of the noise model in the SPITE (reported in Section 6.3).

**Figure 6.35: Traffic noise levels  $L_{10}$  in the central area**



The  $L_{10, 1\text{-hour}}$  predicted noise levels were consistently high throughout the central area. Noise levels very often exceeded 80 dB(A) and have occasionally reached 82 dB(A). These levels, however, cannot be compared to the environmental standards which are only set for 18-hour periods. The  $L_{10, 18\text{-hour}}$  noise levels could not be predicted, since traffic flow data are unavailable for 18-hour periods. The shortened procedure to estimate  $L_{10, 18\text{-hour}}$  noise levels, as recommended by the CRTN method (Equation 4.15), could not be used either, because it requires three consecutive hours of traffic data. Assuming the characteristics of traffic flows over the two preceding hours the same as the conditions within the simulation period, the shortened procedure indicates that the noise standards for the 18-hour period (70 dB(A)) would not be exceeded in only 4 out of the 83 locations in the central area. This indicates that the environmental quality in central Chester can be considered quite unsatisfactory under the noise criterion.



## 6.5 Conclusion

This chapter has presented the way in which the traffic-induced air pollution and noise have been modelled within the context of this work.

It has been shown that the application of the SPITE relies on the input data produced by the traffic model. The cost function on which the assignment model is based to perform the trip allocation throughout the road network has been redefined as to account not only for the costs related to travel time but also for those originating from the expenses on fuel. The cost parameters (value of time and fuel prices) have been set as to represent the valuation on the base year (1993).

The application of the SPITE in Chester has produced a great deal of information related to a range of traffic and environmental conditions in the network. The traffic model has simulated the traffic conditions and generated information such as travel time, fuel consumption, flows, distances travelled and average speed.

The air pollution dispersion model in the SPITE, which is based on the Gaussian plume theory, has been redeveloped in a great extent, particularly in relation to the emission modelling. The emission model provides a detailed pattern of vehicle emissions as a function of the spacial distribution around reception points, the driving patterns, the type of vehicle, the fitting of catalytic converter and it also accounts for future emission limits which will come into effect after the implementation of the 1996 emission regulations. These considerations into the emission model constitute an important advance in relation to existing air pollution models. The emission model has shown the magnitude of the total emissions produced within the network, and this is particularly relevant in terms of the contribution of the study area to the greenhouse effect.

The dispersion model has estimated the concentration levels of the air pollutants under consideration in various locations. The predicted levels of air pollution concentration in the central area of Chester were very high, particularly at points around certain busy junctions. The locations where the highest concentration levels have been found are not necessarily the same for all pollutants.

The noise model in the SPITE, which is based on the CRTN guidelines, predicts noise levels

at a grid of receptor points, those which have been defined as to represent the locations most degraded in each road of the network. The advancements of the noise model in relation to the original method include: the prediction at a grid of points situated around road junctions, the consideration of the geometry of every receptor point in relation to all building facade reflections, and the fact that the inputs to the noise model are provided by a traffic model.

The results of the application of these models show the levels of local environmental degradation throughout the network and the locations where the environmental standards are likely to be exceeded. Assuming that the standards have been set with the aim of protecting human health, these findings can be potentially helpful to aid decision-making to promote public well-being.

The level of accuracy expected for each of the models composing the suite has been verified at the end of each section. It was found that, amongst the models tested, the noise model provided estimates which had the lowest deviation from the observed values. The air pollution model is likely to produce an elevated level of uncertainty, and this is due to the complex nature of the relationships represented in the model and to the difficulty to represent further relationships into it. A considerable amount of data was obtained to test the accuracy of the air pollution model and the results of the comparisons between measurements and estimations have shown that the model performs well in most of the time periods analysed. Some distortions between measurements and estimations have also been found, which are probably due to the influence of other meteorological parameters not considered in the modelling process.

Next chapter describes the way in which the economic and environmental costs incurred by road traffic are evaluated in an assessment procedure. The aim of this procedure is to provide an estimate of the overall economic implications from various transport policies. The two following chapters will report the effects of transport policies and future traffic trends on the local environmental quality. The economic and environmental assessment for each policy tested will be produced at the end of each section.

## 7 ASSESSING THE ECONOMIC AND ENVIRONMENTAL COSTS

It has been reported in Chapter 5 that the environmental externalities originated by road traffic may result in costs to the system. Thus, any change in the strategies for transport will affect the total costs incurred. This chapter describes the way in which the predominant negative impacts of traffic in an urban area are assessed in monetary terms, in the scope of this work. A procedure is created to estimate the economic and environmental costs that will occur to the system from the simulation of transport policies or trends, which will alter the pattern of traffic emissions in the road network. This procedure has been named EAF (Environmental Assessment Framework) in Chapter 6 and its interaction with the SPITE is illustrated in Figure 6.1.

The EAF accounts for the principal indicators of the costs accrued by travellers and by the entire community within the study area. The economic costs considered in the framework are those incurred from the total time spent in travelling and from the consumption of fuel, which are the same costs considered in the cost function of the transport model (Section 6.1.1). Other economic implications, such as accident costs and vehicle operational costs have not been considered in the framework. The local environmental costs from the damage provoked by road traffic include the costs of nuisance, the damage to health and the damage to materials. These will also be expressed in monetary terms. The components of the total costs as accounted for in the EAF are outlined in Equation 7.1 below.

$$TC = TTC + EFC + EDC \quad \text{Equation 7.1}$$

TC = Total costs  
TTC = Travel time costs  
EFC = Expenses with fuel consumption  
EDC = Environmental damage costs

The values used in the EAF have been derived in different time periods and countries. However, the estimates of different costs need to be directly comparable, and in order to express all value terms in the same unit (British pounds) and base year (1993), they were adjusted to:

- *Gross Domestic Product (GDP) per head.* The GDP per head index was used to bring up to the base year all money values originating at different dates. This index is a measure of the population's state of wealth instead of a measure of prices (inflation). Thus, the wealthier people get the more capable they become to bear economic and environmental costs. The GDP and population indices were taken from International Monetary Fund (1995) and OECD (1995).
- *Purchasing power parity.* This index was applied to adjust the prices derived from other nations to the equivalent valuation in sterling. This index incorporates the exchange rate in relation to the American dollar, and appropriate conversions were made to represent the value of distinct currencies into sterling. The 1993 average purchasing power parities were taken from OECD (1995).

The economic and environmental indicators are provided for each policy and trend tested in Chapters 8 and 9. This is particularly valuable for comparing the monetary consequences incurred from different traffic management and transport planning policies.

## **7.1 The economic costs incurred**

The assessment of the economic costs in the EAF include the two most important factors perceived by travellers and which affect their transport decisions. These costs consist of the value of time (TTC) and the total expenses with fuel consumption (EFC) incurred to all travellers in the case study, as indicated in Equation 7.1.

The cost associated to the value of travel time is considered of dominant importance on the assessment of transport planning schemes. However, in order to account for the costs incurred in terms of travel time within the EAF, these have to be converted into monetary terms. A wide theoretical basis has been developed to derive monetary values to travel time savings. Quantitative measures for values of time savings among different individuals have been established for different trip purposes, mode of transport and revenue (MVA Consultancy et al., 1987). The most recent estimates of typical values of time for several categories of travellers have been described in the Traffic Appraisal Manual (Department of Transport, 1991). The value of time for car drivers and bus passenger were quoted at 14.16 and 11.69

pence per minute in 1988 prices. These figures have been updated to 1993 prices according to the GDP per head index and became 18.67 (car driver) and 15.41 (bus passenger) pence per minute.

The estimation of costs related to fuel consumption have been based on current market prices. Fuel prices have increased over the last few years and the average 1993 figures were quoted at 50.1 and 49.3 pence per litre for petrol and diesel, respectively (Department of Transport, 1994a).

The parameters used in the estimation of the total economic cost, as described in the two previous paragraphs, are outlined in Table 7.1.

**Table 7.1: Economic cost incurred per minute travelled and litre of fuel consumed**

| Travel time (pence/minute) |      | Fuel price (pence/litre) |        |
|----------------------------|------|--------------------------|--------|
| Car                        | Bus  | Petrol                   | Diesel |
| 18.7                       | 15.4 | 50.1                     | 49.3   |

It is important to notice that the cost parameters related to travel time and fuel price applied in the EAF are the same as those expressed in the cost function, on which travellers have based their route and mode choices in the assignment stage (see Section 5.1.1).

The total travel time costs (TTC) and the expenses with fuel consumption (EFC) have been estimated separately for each vehicle type namely: petrol cars, diesel cars, lorries and buses. The economic costs in the EAF (TTC and EFC) are shown in Equations 7.2 and 7.3.

$$TTC = \frac{TT \times VT \times NP}{60} \quad \text{Equation 7.2}$$

- TT = Travel time in each type of vehicle (h)
- VT = Value of time (pence/min), as given in Table 7.1
- NP = Number of passengers per vehicle

$$EFC = L \times PF$$

Equation 7.3

L = Number of litres consumed by each type of vehicle (l)

PF = Price of fuel (pence/l), as given in Table 7.1

Other economic changes may well occur to the system, such as the subjective costs or 'disbenefits' perceived by travellers, like inconvenience, welfare losses and dissatisfaction from changes in accessibility conditions and travel behaviour (suppression of trips, change in route, mode or destination). However, such costs will not be accounted for in this framework.

## 7.2 The environmental costs incurred

The derivation of precise money values for the environmental impacts of traffic is not the main objective of this research. This would require a great amount of effort, time and resources, and as described in Chapter 5, monetary valuation is not yet completely developed and still provides major theoretical and practical difficulties.

However, some type of assessment is highly desirable for taking into account the indirect costs imposed on the system. For this purpose, practical experience on the monetary valuation from studies undertaken so far has been gathered in order to originate approximate cost estimates for the local degradation caused by traffic in the study area. It is acknowledged that money estimates which have been derived for particular circumstances may not be quite appropriate for application elsewhere. Nevertheless, they should provide a broad indication of the actual costs incurred, which can be considered to be a progress in relation to usual assessment procedures. Although better accuracy would be desirable, order-of-magnitude estimates may be valuable for policy decisions in many situations, and certainly preferable to no information at all.

The framework for the valuation of the environmental costs in the EAF is presented with great reservations in light of the considerable level of uncertainty involved with environmental valuation. Its aim is to provide some guidance rather than a definite rule for the calculation of the costs of traffic-generated impacts. The estimation of the environmental costs in the EAF is tentative and should produce indicative outputs, since it is almost

impossible to reach absolute conclusions on this subject. Whether environmental valuation is considered, it is necessary to accept a substantial degree of uncertainty, which is characteristic of the various shady frontiers of knowledge involved. Therefore, the results must be treated with caution and scepticism.

The valuation of the environmental impacts caused by road traffic is restricted to the estimation of the costs derived from the local effects which have predominant economic implications. The local environmental effects considered in the EAF are noise nuisance, damage to health and damage to materials. The monetary valuations of these effects will be compatible with the other costs (which are also expressed in monetary terms) included in the EAF. The various techniques described in Chapter 5 have been applied previously to produce monetary values to these environmental effects. The following sections describe in detail the economic implications of the environmental effects considered in the EAF, the techniques applied for their valuation, the results from the application of these techniques and the final figures adjusted to the valuation on the base-year and to the national currency, as utilised in the EAF.

### *7.2.1 Noise nuisance*

Noise nuisance is physiologically harmless to a large extent, but further hazards may develop through symptoms such as effects on hearing and communication, increased blood pressure, headache, sleep disturbance and insomnia. Although rare, serious impairment in human well-being, such as mental and physical health problems and damage to auditory organs, may also occur. However, the overwhelming and most common effect of road traffic noise is nuisance, annoyance or disturbance. More details about noise nuisance can be found in Chapter 4.

Noise nuisance is liable to cause costs in form of disbenefits to individuals who spend some of their time in the proximity of traffic flows. Experience on the valuation of noise nuisance through different monetary techniques has been reported in Chapter 5.

The valuation of traffic noise nuisance in this work will rely upon estimates of the annual social cost incurred in Neuchâtel, Switzerland (Jeanrenaud et al., 1993). These cost estimates have been based on individual willingness to pay for reducing noise levels, according to different vehicle categories. These estimates are particularly suitable for the purposes of this work because they have been derived in a disaggregate level in terms of costs per vehicle-

kilometre. The adoption of estimates as such makes the monetary assessment of noise nuisance from transport policies and trends fairly straightforward, since the costs incurred are directly proportional to the number of kilometres travelled by different vehicles. Table 7.2 shows the costs of traffic noise nuisance given in 1990 Swiss currency, as derived in the above mentioned study, and the equivalent values adjusted to sterling. The adjustments were made to the indices ‘*GDP per head*’ and ‘*purchasing power parity*’ (which accounts for the exchange rate between the different currencies), as reported earlier in this chapter.

**Table 7.2: Traffic noise nuisance costs**

| Vehicle category | Cost per vehicle-kilometre |                    |
|------------------|----------------------------|--------------------|
|                  | 1990 Swiss centimes        | 1993 British pence |
| Car, van         | 2.8                        | 0.85               |
| Bus, lorry       | 5.0                        | 1.52               |

These figures indicate that each kilometre travelled by light vehicles causes a noise nuisance cost of approximately 0.85 pence, which is equivalent to only about 18% of the costs spent with fuel supply, considering an average engine efficiency rate of 10.7 km/l (Department of Transport, 1994a). The monetary estimate for noise nuisance from heavy vehicles is roughly twice as high as from light ones.

### 7.2.2 *Damage to health*

Health damage is one of the most important elements of the total environmental damage cost produced by road traffic pollution, principally in populated areas. The chronic diseases caused by air pollution may contribute to both higher morbidity and mortality rates, which produce a range of costs to the economic system.

Traffic air pollution can cause or contribute to the increase in the incidence of a number of respiratory and cardiovascular diseases, but the extent to which this happens is widely uncertain. The main diseases associated with the presence of compounds originated from exhaust gases are lung cancer, pneumonia, tuberculosis, asthma, bronchitis and emphysema, whereas the main symptoms are headaches, cough, cold, sore throat and eye irritation. Further psychological effects, such as pain, suffering and emotional distress may also occur. Illness



and death from air pollution can be caused by various types of emissions. Smoking habits and proximity to industries should be considered when evaluating the health effects of pollution. More details about the effects of air pollution on health can be found in Section 3.2.

As a result of the impacts on health, air pollution has been reported as responsible for restricting activity, and therefore provoking productivity reduction (for instance: Ostro, 1983). Death and sickness depend on how exposure to pollutant concentrations during a certain period of time is connected with health harm. A 1970 study reported in Danish Ministry of Energy (1991) estimates that 50% of all respiratory diseases are associated with air pollution. Whitelegg et al. (1993) found a clear link between traffic volumes and everyday health symptoms in people living in the proximity of roads. Bown (1994) reports that fine particles in exhaust fumes are responsible for the death of 10,000 people per year in England and Wales.

The valuation of health damage in this work will rely upon the derivation of the total marginal costs of health diseases in Sweden, by the Danish Ministry of Energy (1991). The health damage costs have been estimated on the basis of evaluations of the relative toxicity of various pollutants emitted by different vehicle categories. The relative share of the total toxicity has been considered to represent the damage responsibility per vehicle group for each group of diseases. These estimates have also been derived in terms of vehicle-kilometre, which is compatible with the valuations of the other environmental effects under consideration. Table 7.3 presents estimates of the average marginal health damage costs in 1987 Danish currency and in the equivalent values adjusted to 1993 sterling (according to the indices GDP per head and purchasing power parity, as reported in the beginning of this chapter).

**Table 7.3: Health damage costs**

| Vehicle category | Cost per vehicle-kilometre |                    |      |
|------------------|----------------------------|--------------------|------|
|                  | 1987 Danish kroner         | 1993 British pence |      |
| Car              | petrol                     | 0.075              | 0.64 |
|                  | diesel                     | 0.050              | 0.43 |
| Lorry            |                            | 0.365              | 3.11 |
| Bus              |                            | 0.495              | 4.22 |

These figures show that each kilometre travelled by petrol cars, diesel cars, lorries and buses produce health damage costs of about 0.64, 0.43, 3.11 and 4.22 pence, respectively. The health damage costs produced by petrol and diesel cars are actually lower than the cost estimates for noise nuisance from light vehicles. On the other hand, the health damage costs produced by lorries and buses are many times higher than the respective noise nuisance costs.

In terms of health damage, buses are about seven times more harmful than cars, per kilometre travelled. In order to compare the order-of-magnitude of the different costs involved, the health damage costs produced by cars and buses represent about 14 and 31% of the costs spent on fuel consumption. These estimates were made based on the fuel price and average consumption rate for each vehicle type (Department of Transport, 1994a). This means that the environmental costs in terms of health damage are still only a fraction of the actual expenses incurred from fuel consumption.

These figures can be compared with the costs imposed by petrol cars and lorries in terms of the mortality and morbidity caused from particulates in Los Angeles (Small and Kazimi, 1995). The results of this study indicate that the total costs are 2.55 and 49.28 American cents per vehicle-mile, for petrol car and lorry, respectively. These amounts were adjusted for GDP per head and purchasing power parity at the study's base year (see discussion about adjustment factors in the beginning of this chapter), in order to be compared with the results from Table 7.3. The health damage costs are equivalent to 1.06 and 20.4 British pence per kilometre, which means that the American valuation is substantially higher than the Danish estimates, particularly for lorries.

### *7.2.3 Damage to materials*

Apart from health problems, road traffic air pollution can cause harm to numerous material goods. Pollutant deposition provokes deterioration of building facades and other materials, mainly through the erosion of surface coatings and corrosion of metals. The principal pollutant compounds which cause damage to materials are sulphur, particulate matter, oxidants and nitrogen oxides.

The materials in closer contact with exhaust emissions are likely to suffer most from damage. Atmospheric particulate levels are 3 to 4 times higher in situations exposed to road traffic than otherwise (Ball, 1984) and therefore buildings situated in roads with busy traffic streams

are subject to much faster deterioration. Soiling due to particulate deposition is also aesthetically unappealing and diesel emissions are the principal responsible for soiling effects in urban areas.

For the purposes of this work, the monetary valuation of damage to materials is based on estimates of cost damage to buildings in Neuchâtel, Switzerland (Jeanrenaud et al., 1993). This study estimated initially the average restoration expenses on building damage per square metre. The damage cost per vehicle-kilometre has also been estimated according to the category of vehicle. The units derived from these estimates are compatible with the previous estimates of monetary valuation of environmental attributes. The costs incurred from pollution damage to buildings, expressed in 1990 Swiss centimes, and the equivalent values in British pence adjusted to the base year (according to the indices GDP per head and purchasing power parity) are shown in Table 7.4.

**Table 7.4: Damage to buildings costs**

| Vehicle category | Cost per vehicle-kilometre |                    |
|------------------|----------------------------|--------------------|
|                  | 1990 Swiss centimes        | 1993 British pence |
| Car              | 0.6                        | 0.18               |
| Bus              | 16.4                       | 4.97               |
| Lorry            | 14.5                       | 4.40               |

Considering the figures in Table 7.4, it can be noticed that cars contribute very little to damaging materials. The cost of degradation to buildings caused by a car is estimated at only 0.18 pence per each kilometre travelled. This represents less than 4% of the expenses incurred from fuel consumption. The cost imposed by cars on building damage is considerably lower than the costs incurred through noise nuisance and health damage. These findings are in line with the profile of pollutant emissions originated from petrol cars (see Tables 3.3 and 5.1), which are typically minimum in terms of particulate matter.

However, the monetary costs imposed by heavy vehicles on the maintenance of building facades are comparable to the health damage costs and are a great deal higher than the noise nuisance costs provoked by these vehicles. This is explained by the high levels of particulate emitted by heavy vehicles and the strong effect of the exposure to particulates on the soiling

of buildings. According to the figures shown in Table 7.4, the damage costs produced by the circulation of buses are almost 28 times higher than the costs caused by cars. Therefore, the total costs from damage to buildings will largely depend on the number of cars, lorries and buses in the traffic stream.

#### 7.2.4 Total environmental costs

This section outlines the total environmental cost estimates as discussed in detail the previous sections. The cost estimates are given in monetary terms, according to 1993 sterling prices, caused by different vehicle categories per kilometre travelled.

Table 7.5 summarises the costs imposed by each environmental criterion considered and shows the estimates for the total environmental costs incurred by each vehicle category. In the cases where no difference has been made for the cost produced by petrol and diesel cars, these have been assumed to be equal. This assumption underestimates the material damage produced by diesel cars, which has not been specifically derived (as shown in Table 7.4).

**Table 7.5: Total environmental costs**

| Vehicle category | Cost (pence per vehicle-kilometre) |               |                 |              |             |
|------------------|------------------------------------|---------------|-----------------|--------------|-------------|
|                  | noise nuisance                     | health damage | material damage | Total        |             |
| Car              | petrol                             | 0.85          | 0.64            | 0.18         | <b>1.67</b> |
|                  | diesel                             | 0.85          | 0.43            | 0.18         | <b>1.46</b> |
| Lorry            | 1.52                               | 3.11          | 4.40            | <b>9.03</b>  |             |
| Bus              | 1.52                               | 4.22          | 4.97            | <b>10.71</b> |             |

The environmental damage costs (EDC) have been estimated as shown in Equation 7.4.

$$EDC = D \times EC \quad \text{Equation 7.4}$$

D = Distance travelled by each type of vehicle (km)

EC = Environmental costs incurred by each type of vehicle (pence/km)

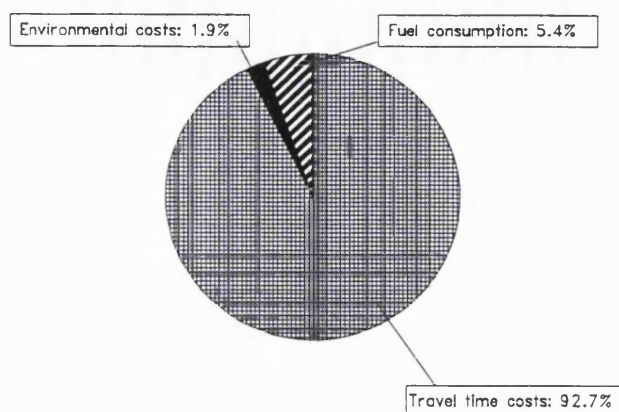
The cost estimates for noise nuisance, as shown in Table 7.5, may seem to be high in comparison to the costs of health and material damage from exposure to air pollution, particularly for cars. Noise nuisance costs have been derived from the CVM, which as reported in Section 5.2.5, tends to overestimate bids of WTP. The local costs imposed by air pollution have been estimated from the application of the damage cost technique, which as stated in Section 5.2.1, is likely to provide underestimated results because it is virtually impossible to account for the totality of the effects from health and material damage.

The results from Table 7.5 show that the total environmental costs produced by heavy vehicles are many times higher than from light ones. Buses generate environmental costs about seven times higher than cars, in absolute terms. However, the degradation per passenger produced by buses will depend largely on the occupancy rate. Considering the average occupancy rate of cars at 1.2 passengers, it can be surmised that buses ought to carry at least eight-to-nine passengers per kilometre in order to be considered more attractive than cars in environmental terms. The same comparative view can be applied in terms of energy efficiency between the private and public modes of transport.

The total environmental cost imposed by buses correspond to about 80% of the costs incurred with diesel consumption from these vehicles, whereas the total costs produced by cars are about one third of the expenditure on fuel (these estimates were based on fuel prices and consumption rates as reported in Section 7.2.2). This suggests that the environmental costs can represent amounts which are comparable to other typical economic costs (in a smaller extent in relation to value of travel time).

A comparison of the environmental cost estimates (Section 7.2.4) with the economic costs (Section 7.1) reveals that the costs related to the value of time spent on travelling are noticeably higher than the other costs considered in the assessment framework. The expenses with fuel consumption represent only a modest proportion of the total costs incurred, whereas the environmental costs amount to a much smaller fraction. Figure 7.1 shows the proportion of the environmental costs in relation to the total costs incurred from a petrol car per kilometre travelled.

**Figure 7.1: Percentage of the total costs per kilometre generated by a petrol car**



The environmental costs, fuel consumption and travel time are equivalent to about 1.9%, 5.4% and 92.7% of the total costs incurred per kilometre travelled by a petrol car, respectively. These estimates have considered an average car occupancy rate of 1.2, engine efficiency of 10.7 km/l and speed of 16.6 km/h, as in the evening peak period in central London (Department of Transport, 1994a). The total environmental costs produced by cars are comparable to about 36% of the costs incurred from the expenditure on petrol, or to only about 2% of the costs incurred from the time spent in travelling.

It is very important, however, to acknowledge some of the probable reasons for the large contrast in the magnitude of the various cost estimates included in the framework. The estimates of the value of travel time are likely to be inflated, because many travellers do not perceive their travelling times as money expenditure. Estimates of the environmental costs in this work have probably been underestimated, since neither many effects of pollution damage (such as the impacts on fauna and flora and the long-term impacts on health and materials) or other environmental impacts (such as global implications of air pollution) have not been accounted for.

### **7.3 Conclusion**

This chapter has presented a framework for the assessment of the economic and environmental costs produced within a certain area (EAF). The costs comprised in the EAF are those incurred from the time spent in travelling, the expenses with fuel consumption and the environmental damage costs. The travel time costs were based on well-developed research on the value of time. The derivation of the fuel costs is straightforward, based on current

market prices. The environmental costs include the valuation from three effects which have economic implications, namely noise nuisance, damage to health and damage to materials. The costs from the environmental damage are subject to various uncertainties inherent to the nature of the valuation. However, their incorporation in the EAF can be considered a progress in relation to other assessment procedures.

The magnitude of the figures derived for the environmental costs are very low if compared to the valuation of the economic costs, particularly for light vehicles. The total costs incurred on the environment by a petrol car are equivalent to about a third of the costs incurred from the consumption of fuel, or comparable to only about 2% of the costs incurred from the time spent in travelling. However, the environmental costs produced by heavy vehicles are a great deal more significant within the assessment framework, since these vehicles generate much more environmental damage than light ones. The environmental costs generated from buses correspond to about 80% of the costs incurred from the consumption of diesel by these vehicles, whereas the total costs produced by cars are about one third of the expenditure on fuel. This suggests that the environmental costs can represent amounts which are comparable to other typical economic costs and, therefore, should not be neglected in the assessment of transport schemes. It has been concluded that, in environmental terms, buses must carry at least eight-to-nine passengers per kilometre in order to be considered more attractive than cars.

It has been pointed out that the value of travel time used in the framework is likely to have been overestimated, whereas the appraisal of the environmental costs would have been underestimated. These facts emphasise that the environmental costs probably represent a much greater share of the total costs generated by the provision of transport.

Further research is necessary in order to incorporate the environmental costs into an assessment framework in a more comprehensive fashion. However small these costs may be, it is important to acknowledge their existence and continue making efforts to account for them into the assessment of traffic schemes in the best possible way.

The following two chapters present the applications and results of the SPITE (Chapter 6) and the EAF (Chapter 7) in the study area. Chapter 8 will investigate the environmental consequences of short-term traffic management measures, whereas Chapter 9 will estimate the environmental impacts from future transport trends in the longer term.

## **8 THE IMPACTS OF TRANSPORT POLICIES ON THE ENVIRONMENT**

The need to maintain traffic capacity, safety, accessibility and the environment at desirable levels often generates a wide range of conflicts of interest amongst the various segments of society. Transport planning projects and traffic management schemes are usually conceived in order to deal with one or more of these problems, which ultimately aim to enhance aspects of the quality of life of local or regional communities.

Traffic management schemes are considered one of the most effective and flexible ways of promoting better use of the available transport facilities. They have been shown to provide practical and short term solutions, with great benefits at relatively low cost. Many traffic schemes have been conceived with the specific objective of reducing traffic congestion, but these schemes may have other purposes such as giving priority to special routes or modes of transport, reducing fuel consumption and accident rates, and improving mobility. Traffic schemes in the context of transport planning can also be used for improving the overall efficiency of travel in order to reduce the environmental impacts produced by its provision.

Many changes in the transport system can influence travel patterns and the characteristics of traffic flows, by facilitating or restricting traffic flow conditions. Because traffic flow and speed are the principal explanatory variables for the levels of air pollution and noise emissions, any change in these parameters will directly affect environmental quality (Chiquetto, 1994 and 1995; Chiquetto and Mackett, 1994a and 1994b). Most of the environmental impacts from traffic changes are realised at the local level, since noise and pollutant concentrations are produced by emissions in the immediate vicinity of locations where pedestrian activities take place. However, they may also have global effects, since traffic pollution constitute a significant portion of the greenhouse emissions.

The management of road capacity may have two objectives: to reduce the traffic level or to speed up the existing flow, which although often contradictory in nature, both aim to reduce congestion. Changes to road capacity are directly correlated with variations in traffic flow and speed, the evidence of which is acknowledged from the speed-flow relationships. The most common measures for enhancing capacity and speed are the elimination of parking spaces, road widening, transformation of roads into express or one way, installation of reversible lanes, deviating or re-routing flows and banning turning movements. Amongst the traffic



restriction schemes, one can propose speed limits<sup>15</sup>, road narrowing or closure, humps, creation of bus lanes, pedestrianisation, traffic calming schemes and restriction on cars or lorries from circulating in environmentally sensitive areas. Investments in new capacity or improvements to the quality of service provided by existing capacity should reduce congestion, as long as they do not attract new demand.

Invariably, environmental benefits (like the mitigation of pollutant and noise emissions) are achieved by decreasing traffic flow and density. However, as far as traffic speed is concerned, some environmental benefits may be achieved by increasing speeds, which within the urban speed range produces a considerable decrease of most air pollutant emissions. In other cases, benefits such as the reduction of noise levels and NO<sub>x</sub> emissions can be obtained by lessening traffic speed.

In some cases, concentrating traffic within the main routes may be preferred to spreading the environmental impacts throughout the network. This is particularly true for the case of noise level, which is a function of the logarithm of traffic flow and by doubling the flow an increase of only 3 dB(A) is achieved (see Section 4.3). In the case of air pollution, however, the concentration of traffic can be disastrous, particularly if the main routes are already saturated. This can be harmful because most exhaust emissions build up, so does their concentration in the air, and hence air pollution is higher the more congested the traffic routes are (Section 3.3).

A number of studies have been carried out recently on the investigation of the impacts of transport policies for mitigating vehicle emissions. DiRenzo (1979) and Krawack (1993) analysed changes in local and regional air quality from the implementation of a wide range of traffic management strategies. Hughes (1992) presented a range of strategies for reducing the long-term emissions of greenhouse gases from personal travel. The Department of the Environment jointly with the Department of Transport (1993) have also published guidelines on the reduction of emissions through land use planning. Wegener (1993) presents strategies for the reduction of CO<sub>2</sub> emissions from transport through the reorganisation of urban activities.

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<sup>15</sup> In the U.K., the speed limit of 20 miles per hour have been imposed in some zones which are particularly sensitive. Speed limiters on coaches and new heavy goods vehicles are compulsory since August 1992 and on the existing heavy lorries since August 1993.

The Government has already set out transport policy objectives which underlie the improvement of the environment and help local authorities to meet sustainable development strategies. This concern has been expressed in a number of White Papers and other publications: U.K. Government (1990 and 1994), Department of the Environment (1993) and Royal Commission on Environmental Pollution (1994). In particular, the Planning Policy Guidance 12 (PPG 12) requires authorities to regard environmental considerations in preparing their general policies (Department of the Environment and Department of Transport, 1992) and PPG 13 recommends the reduction of the need to travel and encourages the use of less environmentally damaging modes of transport (Department of the Environment and Department of Transport, 1994).

The idea of sustainable planning originated from the need to make sure that development is achieved without compromising the ability of future generations to meet their own needs (World Commission on Environmental Development, 1987). A transport policy compatible with sustainable development objectives should strike the right balance between meeting the needs of present economic development and protecting the future ability to sustain a high quality of life. In order to achieve this, the Government will need to provide a framework in which people can exercise their transport choice in ways which are compatible with environmental goals (U.K. Government, 1994).

An idea similar to 'sustainable transport' originated some thirty years earlier with the Buchanan Report (Ministry of Transport, 1963), from the conflict between the accessibility conditions and the need to control the local environmental impacts of transport. The concept of 'environmental capacity' was then introduced, which can be summarised as follows: environmental capacity is the maximum traffic flow which produces up to a certain level of environmental degradation compatible with the maintenance of a set of arbitrary standards for acceptable environmental conditions. This concept has been applied to a range of studies (Buchanan, 1971, Appleyard and Lintell, 1972, Crompton and Gilbert, 1976 and 1978, Gilbert, 1988, Chadwick, 1989, Chiquetto and Santos, 1993) to which traffic management schemes played the ultimate role in ensuring the maintenance of environmental quality.

Thus, it is important to ascertain the levels of environmental impacts which can be expected from various changes in the transport system or from the implementation of new transport policies. The understanding of the magnitude of environmental changes can be relevant for providing theoretical grounds to assist the decision-making process, for enabling entitlement

under compensation acts against environmental deterioration and as an aid to long-term overall Governmental goals.

This chapter examines in detail the extent to which the implementation of a range of short-term transport planning policies and traffic management schemes can influence total pollutant emission levels as well as local levels of road traffic air pollution concentration and noise within the central area of Chester. Scenarios are created in order to represent the sensitivity of changes in the transport system on the environment, when compared to the base-scenario or 'do-nothing' situation.

The combined modal split-assignment model (Sections 6.1.2 and 6.1.1, respectively) is run for each policy analysed. Each new run results in new generalised costs, tree-building, flows, speeds, congestion levels and other traffic features. Then, the SPITE is applied for the estimation of the environmental impacts produced by each new traffic scenario. The transport policies simulated in this chapter are:

- the implementation of a pedestrianisation scheme in the central area of Chester,
- the implementation of traffic calming devices in a road outside the central area, and
- the exclusion of heavy goods vehicles from the central area.

The particulars of these policies as well as the results from the SPITE and the EAF are reported in Sections 8.1, 8.2 and 8.3, respectively.

### **8.1 The implementation of a pedestrianisation scheme in central Chester**

Pedestrianisation is defined as the removal of traffic from existing city streets, which may be followed by suitable treatment in terms of paving, street furniture and other details (Hall and Hass-Klau, 1985). Pedestrianisation is a measure applied to areas of predominantly pedestrian use, particularly suited to central and high density areas, along main shopping streets. Delivery traffic may be possible at special daily hours and in some cases it is applied with the exclusive provision of public transport facilities. This scheme usually aims to provide better accessibility and mobility to pedestrians, enhance the volume of shopping and other

business activity in the centre<sup>16</sup> and improve the attractiveness of the local environment in terms of aesthetics, air pollution, noise and accident involving pedestrians. Pedestrianisation schemes tend to increase the number of pedestrians using the city centre, and bring about an increased appreciation of the pedestrianised area and an awareness of the historic environment of the city. Pedestrianisation can induce an increase in the values of properties situated within the traffic-free areas.

On the other hand, pedestrianisation can also provoke land use, rental and property devaluation. This scheme tends to worsen the accessibility to car users, and often generates an increase in the traffic flow in the surrounding areas, which means increased travel time and fuel consumption for travellers. Pedestrianisation can also discourage car users to travel to the traffic-free areas and induce changes to more accessible destinations, usually situated out-of-town.

A large number of streets have been pedestrianised in Britain (Roberts, 1981; Hall and Hass-Klau, 1985), and traffic air pollution and noise levels have had important considerations in the pedestrianisation proposal in many cases. Before the implementation of the pedestrianisation scheme in Durham, the concentration of lead in the atmosphere in some locations of the city centre approached the recommended maximum level and noise levels were exceeding considerably the acceptable limit of 68 dB(A) (Durham City Pedestrianisation Joint Working Party, 1982). In the 'after' scenario, the level of lead decreased by almost half and noise dropped well within the limit. In Leeds, noise levels have been reduced from 65-75 to 60-65 dB(A) after pedestrianisation (Hall and Hass-Klau, 1985), which also meant a reduction to levels below the environmental standard. Some studies have been carried out abroad showing the changes in air pollution and noise levels before and after pedestrianisation (Roberts, 1981). In these studies, the reductions in air pollution ranged from about 40% to 92% in relation to different pollutants, and noise reductions were as high as 25 dB(A).

A pedestrianisation scheme was proposed for implementation in the central area of Chester a few years ago by the Cheshire County Council (1990). The project has passed the public inquiry stage and has been recently implemented coming into operation in May 1995. The vehicle restrictions caused by this scheme have affected the circulation conditions for most

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<sup>16</sup> Evidence of the effects of pedestrianisation on trade can be found in Roberts (1981) and TEST (1988).

travellers using the city centre, including motorists, pedestrians, cyclists, bus passengers, taxi users and delivery drivers.

Figure 8.1 shows the previous traffic restrictions and Figure 8.2 illustrates the current pedestrianised scheme in central Chester.

The SPITE was applied to the situations 'before' and 'after' the implementation of the pedestrianisation scheme in Chester, in order to simulate the changes in the environmental impacts produced.

As a result of the implementation of the pedestrianisation scheme, traffic was diverted from the pedestrianised roads and reassigned in the network. The simulation of traffic conditions after pedestrianisation was carried out by modifying the road network data file and setting the parameters 'number of lanes' and 'saturation flow' of the turns banned due to the pedestrianisation scheme both to zero. The bus routes have also been removed from the pedestrianised roads. These artificial settings guaranteed that no motor vehicles would travel in the roads closed to traffic.

The simulation of the traffic conditions has accounted for the shifts in transport mode and variations in traffic route which are likely to occur when travellers are faced to accessibility changes in the road network. Matzoros et al. (1987) have demonstrated that the assignment model SATURN has predicted post-pedestrianisation traffic flows in a particular application with an accuracy level of around 12%.

Figure 8.3 shows the percentage changes in four traffic indicators for the Chester road network. These overall changes are related to the base-scenario figures (or 'before' situation), which were presented in Table 6.5 (Section 6.4).

Figure 8.1: Previous traffic restrictions in Chester

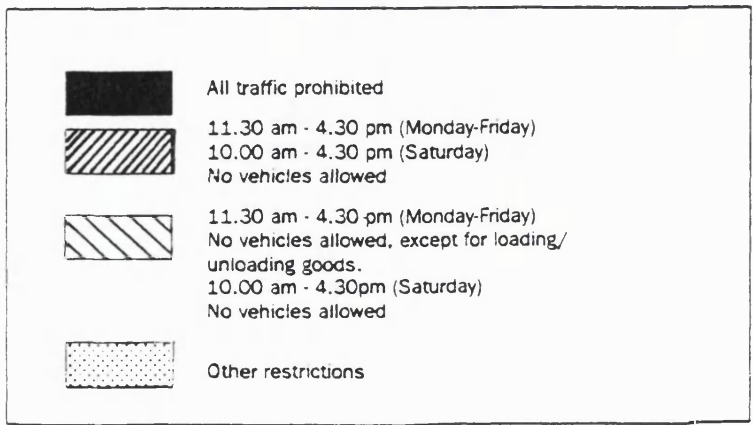
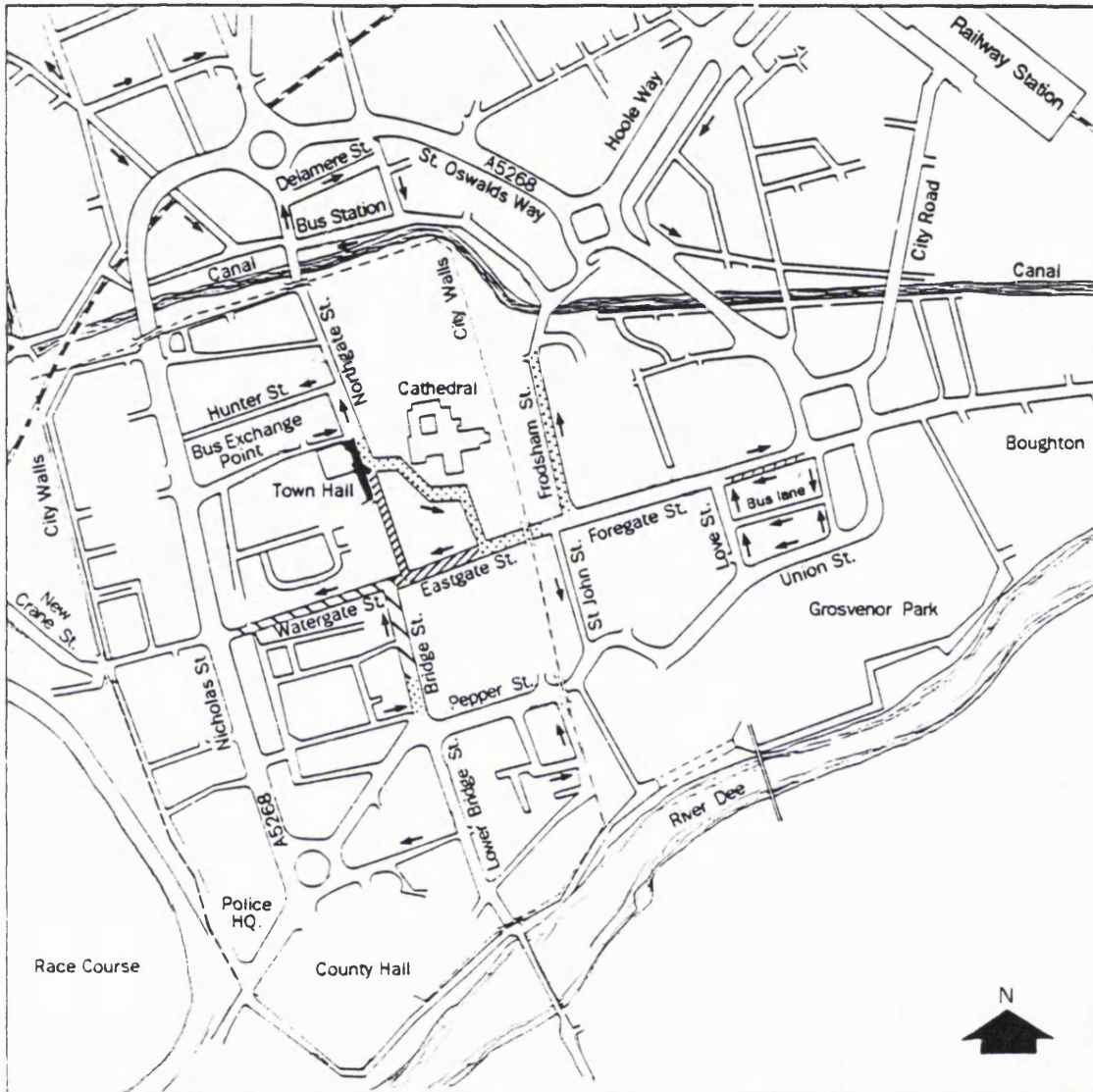
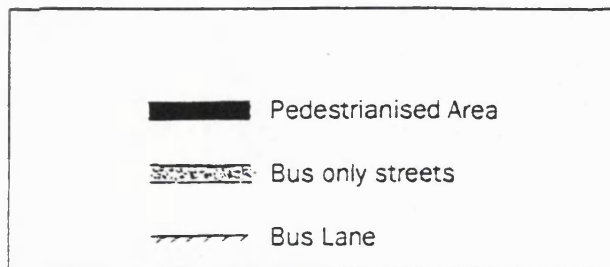
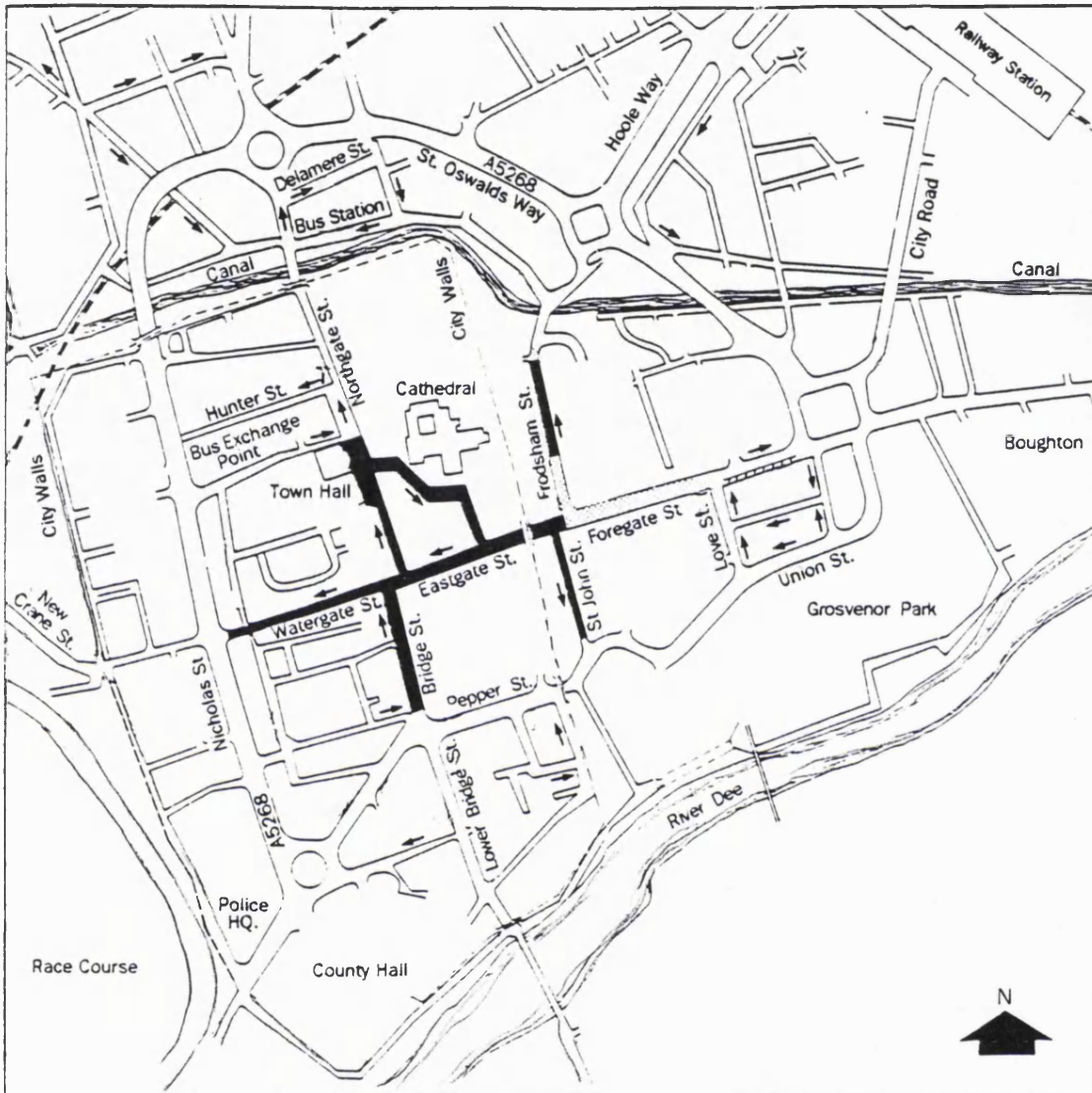
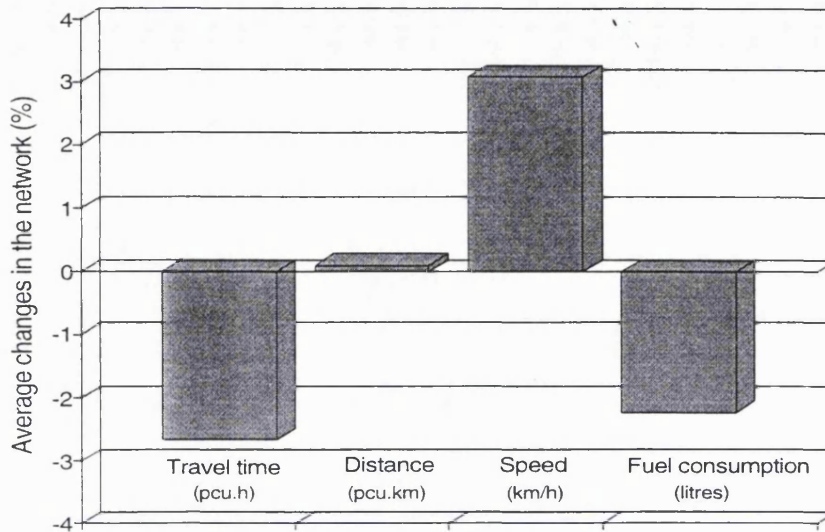


Figure 8.2: Current pedestrianised area in Chester





**Figure 8.3: Percentage changes in the traffic indicators due to pedestrianisation**



As result of the implementation of the pedestrianisation scheme, traffic was removed from the closed roads and some routes became longer. Thus, one would expect that the total distance travelled in the whole network (and probably the total travel time and total fuel consumption) should increase. However, the modelling results indicate an opposite trend. The closure of the pedestrian roads to traffic has made the overall circulation conditions better, with reduced congestion, travel times and fuel consumption.

This apparently counter-intuitive situation is known as 'Braess's paradox' (Sheffi, 1985). The reason for such improvement in flow conditions is rooted in the essence of the user equilibrium approach used by SATURN for flow distribution. This approach describes drivers' behaviour and each driver minimises his or her own travel costs (Section 6.1.1). The individual route choice is carried out with no consideration of the effect of this action on other network users, which defines the non-cooperative behaviour represented by the *user equilibrium*. In this particular application, although some routes have become longer and the travel times of some travellers have increased, the total time spent in travelling in the whole network has decreased. This paradox illustrates the fundamental difference between the *user equilibrium* approach, in which costs are minimised for each traveller, and the normative *system optimum* approach, in which costs are minimised for the system as a whole. If flows had been distributed according to the *system optimum* approach, the total travel time would have certainly increased. From a more general perspective, the Braess's paradox emphasises

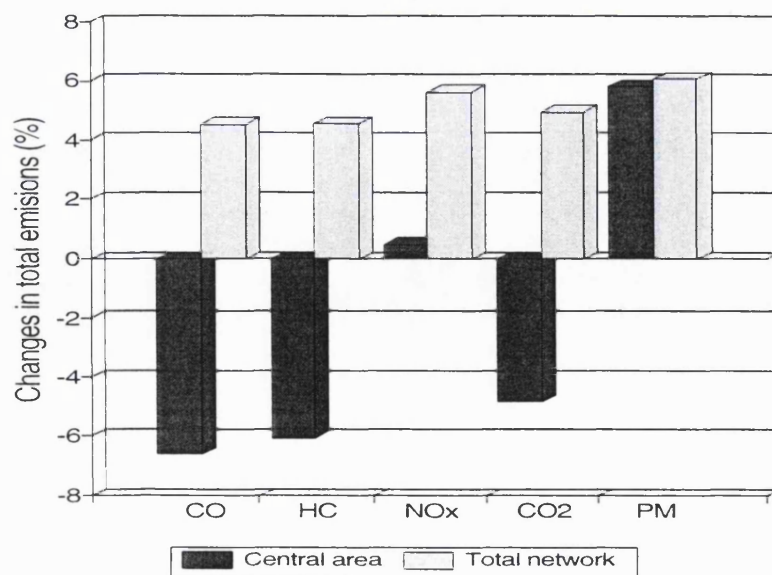


the importance of a careful and systematic analysis of transport planning schemes in urban networks. Sheffi (1985) states that: "...not every addition of capacity can bring about the anticipated benefits and in some cases the situation may be worsened. In fact, traffic engineers have known for a long time that restrictions to travel choices and reductions in capacity may lead to better overall flow distribution patterns. This is the underlying principle behind many traffic control schemes".

In conclusion, although the total distance travelled in the whole network increased slightly, the average traffic speed has actually increased by about 3% and the total travel time decreased by just under 3%. This means that traffic is in general running more freely. The pedestrianisation scheme in Chester has also provided benefits in economic terms, by causing an overall reduction in the consumption of fuel over the entire network. These findings, however, are particular to the case study and similar trends are unlikely to be found elsewhere.

The following step was the application of the SPITE, using the data produced by the traffic model. The estimations of the total emissions produced 'before' and 'after' the implementation of the pedestrianisation scheme were made by the emission model in the SPITE, as reported in Section 6.2.1. Figure 8.4 illustrates the percentage changes of the total emissions in the central area and in the total network, in relation to the absolute amounts produced in the base-scenario which are shown in Figure 6.29.

**Figure 8.4: Changes in total emissions due to pedestrianisation**



The total emissions of CO, HC and CO<sub>2</sub> produced in the central area of Chester have decreased slightly (about 5-7%) after pedestrianisation. Although the pedestrianised roads produce no emissions at all, some traffic has been re-diverted to other roads still in the central area and thus contributing to offset some of the environmental gains which would be obtained within the city centre. The buses diverted from the pedestrianised roads very often had their routes extended within the central area, what contributes for increased emissions of PM and NO<sub>x</sub> in the central area.

Despite the fact that the traffic circulation conditions improve after pedestrianisation, the total emissions in the total network (including the central area) have increased by about 5% for all pollutants. Although the total distances travelled in the Chester network have increased very little in percentage terms, this change in absolute terms (43.2 km) is more significant, as far as traffic emissions are concerned, than the effect from the increase in traffic speed by about 3% (from 13 km/h to 13.4 km/h). In conclusion, this simulation exercise has shown that even when pedestrianisation schemes produce benefits in terms of traffic circulation, a small increase in the total emissions within the city's network is likely to occur.

The air pollution dispersion and the noise propagation models in the SPITE were then applied to the 'after' situation at the reception locations in the study area. The changes in the levels of environmental degradation were analysed separately at three different scales: in the pedestrianised roads, within the central area (which includes the pedestrianised roads) and in the total network (which includes the central area).

Initially, the changes in the levels of environmental degradation were analysed in detail in the pedestrianised roads. Figure 8.5 shows a computer representation of central Chester produced by the traffic assignment model. This figure also indicates the pedestrianised roads and the points for which the environmental impacts have been estimated in the pedestrianised area.

Figure 8.5: Representation of the pedestrianisation scheme in Chester

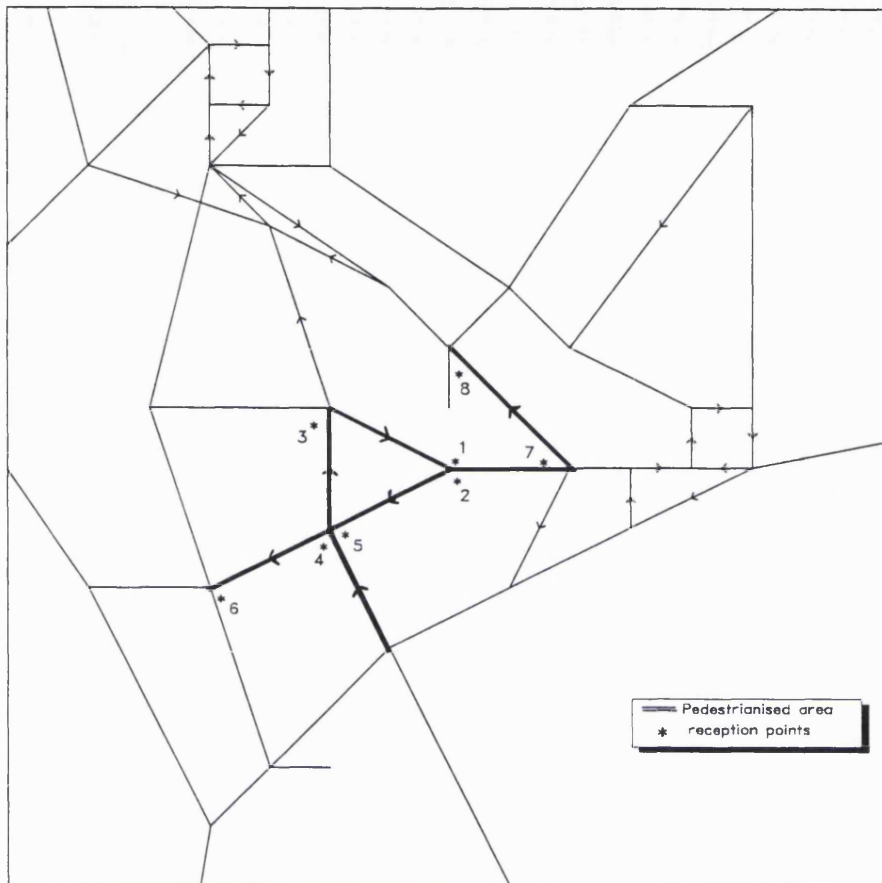


Table 8.1 shows the percentage changes in air pollution concentration and noise levels at the reception points in the pedestrianised area in Chester, which are indicated in Figure 8.5 above. These changes are related to the base-scenario estimates, as reported in Section 6.4. The negative values represent a decrease in the percentage changes of environmental degradation levels, whereas the positive figures indicate that increases have occurred.

**Table 8.1: Percentage changes in environmental impacts in the pedestrianised area**

| Location | Pollutant concentration (ppm) |        |                 |                 |        | Noise L <sub>10, 1-hour</sub><br>dB(A) |
|----------|-------------------------------|--------|-----------------|-----------------|--------|--|
|          | CO                            | HC     | NO <sub>x</sub> | CO <sub>2</sub> | PM     |  |
| 1        | -100.0                        | -100.0 | -100.0          | -100.0          | -100.0 | -100.0                                 |
| 2        | -100.0                        | -100.0 | -100.0          | -100.0          | -100.0 | -100.0                                 |
| 3        | -99.5                         | -99.3  | -97.3           | -98.9           | -95.8  | -9.4                                   |
| 4        | -100.0                        | -100.0 | -100.0          | -100.0          | -100.0 | -100.0                                 |
| 5        | -100.0                        | -100.0 | -100.0          | -100.0          | -100.0 | -100.0                                 |
| 6        | -26.4                         | -24.0  | +1.3            | -21.6           | +38.3  | +0.7                                   |
| 7        | -92.3                         | -90.1  | -81.1           | -86.4           | -76.9  | -11.3                                  |
| 8        | -25.9                         | -25.4  | -17.0           | -24.2           | -10.7  | -0.8                                   |

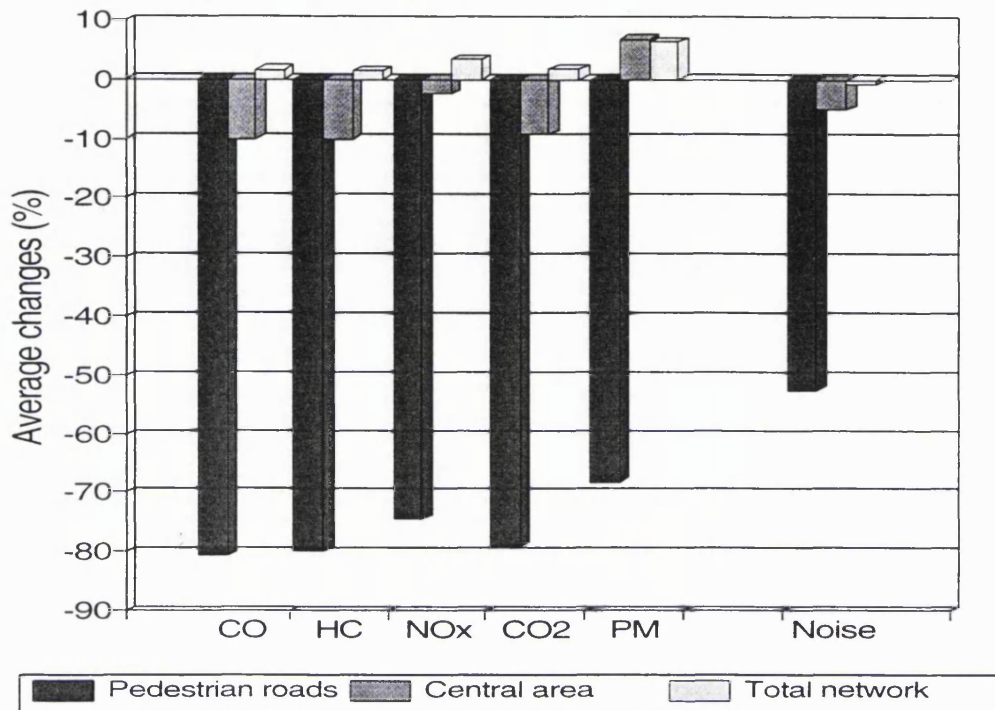
The environmental impacts of traffic at the points 1, 2, 4 and 5 were completely eliminated after the pedestrianised roads were closed. This happens because there is no traffic circulating on these roads and there are no other traffic emission sources around which can contribute to the environmental degradation at these receptor points.

However, pollutant concentrations and noise levels have changed at varying levels at the points 3, 6, 7 and 8. Most changes were in terms of reductions in the environmental degradation levels, due to the reduction in traffic levels. Such improvements did not achieve one hundred per cent because these points are situated near other roads from which traffic emissions influence the impact levels at the estimating locations. At the point 6, particularly, increases in PM, NO<sub>x</sub> and noise levels have been verified. It has been demonstrated that buses produce most prominent levels of PM and NO<sub>x</sub> emissions (Section 6.2.1) and noise (Section 4.3), in comparison to cars. A great deal of the buses deviated from the central area were re-assigned to the major road situated close to point 6. The reduction in PM and NO<sub>x</sub> concentrations and noise level caused by the elimination of traffic on the pedestrianised road containing point 6 was outweighed by the effect of the increase in the traffic, particularly of buses, on the surrounding roads.

Changes in the environmental impacts as a result of the implementation of this scheme have been averaged in order to provide general indicators for other parts of the network. The average changes in pollutant concentrations and noise levels at points in the pedestrianised

roads, in the central area and in the total network are shown in Figure 8.6.

**Figure 8.6: Average percentage changes in environmental impacts after pedestrianisation**



The results shown by Figure 8.6 indicate that the pedestrianisation scheme provides major environmental benefits for people the pedestrian area, in terms of air pollution (concentrations have been reduced on average in the order of 70 to 80%, depending on the pollutant) and noise levels ( $L_{10}$  were reduced on average by about 50%). This picture can be quite relevant for policy decisions, since environmental benefits as such accrue to a large number of people in the pedestrianised busy roads.

Reductions in pollutant concentrations and noise levels have also been found considering the average changes in the central area as a whole, with the exception of PM pollution, which has increased slightly. These reductions were due to the elimination in traffic emissions in the pedestrian roads, which outweigh the environmental effect of some traffic re-routing in the central area. However, the apparently peculiar increase in average levels of PM concentrations was caused by the re-routing of buses from the roads closed to traffic (these vehicles are large producers of PM emissions, and most bus routes were extended after pedestrianisation).



Small increases in the concentration levels of all pollutants have been verified in the network as a whole, which were due to the overall redistribution of traffic. Increases in the order of magnitude as such (up to about 6%) are not very significant for transport policy decisions. The changes in average levels of traffic noise for the total network were not considerable either. Noise levels are far less sensitive to changes in the characteristics of traffic flows than the emission of air pollutants.

Other parameters have been derived in order to provide a better understanding of the changes in the environmental impacts originated from the implementation of the pedestrianisation scheme in Chester. These parameters, namely: mean, standard deviation, maximum reduction and maximum increase, are shown in Table 8.2.

**Table 8.2: Parameters related to changes in environmental impacts after pedestrianisation**

| Location         | Number of roads | Parameter (%)  | Air pollution concentration |        |                 |                 |        | Noise           |
|------------------|-----------------|----------------|-----------------------------|--------|-----------------|-----------------|--------|-----------------|
|                  |                 |                | CO                          | HC     | NO <sub>x</sub> | CO <sub>2</sub> | PM     | L <sub>10</sub> |
| Pedestrian roads | 8               | Mean           | -80.5                       | -79.9  | -74.3           | -78.9           | -68.1  | -52.6           |
|                  |                 | Std. deviation | 33.7                        | 34.2   | 41.8            | 34.9            | 52.8   | 50.8            |
|                  |                 | Max. reduction | -100.0                      | -100.0 | -100.0          | -100.0          | -100.0 | -100.0          |
|                  |                 | Max. increase  | 0.0                         | 0.0    | +1.3            | 0.0             | +38.3  | +0.7            |
| Central area     | 83              | Mean           | -10.0                       | -10.0  | -2.3            | -9.1            | +6.8   | -4.9            |
|                  |                 | Std. deviation | 57.0                        | 54.4   | 40.1            | 50.4            | 45.4   | 21.7            |
|                  |                 | Max. increase  | +198.4                      | +180.0 | +100.8          | +157.9          | +142.5 | +5.3            |
| Total network    | 414             | Mean           | +1.6                        | +1.5   | +3.4            | +1.8            | +6.3   | -0.8            |
|                  |                 | Std. deviation | 35.7                        | 34.5   | 30.2            | 33.7            | 45.7   | 9.9             |
|                  |                 | Max. increase  | +273.5                      | +282.2 | +362.8          | +314.8          | +720.1 | +7.9            |

The mean alone (which was portrayed as the average changes in Figure 8.6) is usually not enough to indicate the variation of a distribution. A measure of the degree of scatter of the differences is needed to assess the reliability of the average figures and to serve as the basis for control of the variability. The most meaningful measure of scatter from the average is standard deviation. It provides considerable information about the absolute dispersion of a given distribution, the higher the standard deviation the greater the amount of scatter. In other words, standard deviation is a measure of the homogeneity of observations in relation to the average value. The mathematical expression for standard deviation is given in Equation 8.1.

$$\text{Standard deviation} = \sqrt{\frac{\sum_{i=1}^n (x_i - \bar{x})^2}{(n - 1)}} \quad \text{Equation 8.1}$$

$x_i$  = observation  
 $\bar{x}$  = mean  
 $n$  = number of observations

The standard deviation has been utilised to measure the spread of the differences around the mean difference for all environmental indicators. Thus,  $x_i$  becomes the percentage difference between 'before' and 'after' pedestrianisation and  $\bar{x}$  is the mean percentage difference amongst all observations.

The standard deviations given by Table 8.2 show that the scatter of the percentage differences of air pollution concentrations in relation to the mean ranges from about 30 to about 57%, as a function of the location and pollutant. This indicates that the changes in air pollution concentration from the 'before' to the 'after' situation are highly variable according to the changes in the characteristics of traffic flows for the various locations analysed.

The standard deviation for the percentage changes in noise levels is high (about 51%) considering the pedestrian roads only. For the central area, the standard deviation is about 22% and it is much lower considering the whole network (about 10%). The reason for this drastic variance in standard deviations is that the changes in noise levels in the pedestrianised roads reach 100% reduction, where no other traffic source contributes to noise levels at the receptor point (see locations 1, 2, 4 and 5 in Table 8.1). In other locations, however, the percentage changes in noise levels are very low (see locations 3, 6, 7 and 8 in Table 8.1), since traffic noise is very little sensitive to changes in flows. Small changes as such depress the value of the standard deviation, particularly considering the whole road network.

The maximum reductions in air pollution concentration levels have occurred, as expected, in the pedestrianised roads, where all environmental indicators are reduced by up to 100%. Increases in the pedestrian area are only significant in relation to the levels of PM pollution which reached up to about 38% in a particular location (receptor point 6 in Table 8.1). The

reason for this increase is that most bus routes which have been removed from the pedestrianised roads pass through the main traffic corridor adjacent to that particular location, contributing to increase the concentrations of PM at that location. Considering the central area of Chester, the maximum increases have been in the order of about 100 to 200%, and accounting for the whole network, maximum increases have varied from about 270 to 720%, depending on the pollutant. The changes in noise have been very modest, up to about 8%.

The application of the modal split model (Section 6.1.2) in Chester has indicated that the pattern of travel choice between the modes available has not suffered appreciable changes, after the pedestrian roads have been closed to traffic. This was with no doubt to be expected. The likely changes that would be expected are to the motorised trips originating or destined on the pedestrian roads. Such trips which used to be made by the modes car and bus would have been split into walking only, or a combination of walking-car and walking-bus.

The EAF (as given in Chapter 7) is applied to the results of the simulation of the implementation of the pedestrianisation scheme, in order to illustrate the magnitude of the economic and environmental costs incurred.

The percentage differences in the traffic indicators for situations 'before' and 'after' pedestrianisation, as given by the traffic assignment model, have been shown in Figure 8.3. However, for the purposes of the EAF, a computer program was created for estimating the traffic characteristics, such as the total distance travelled, total time and fuel spent on travelling, on each road separately for petrol cars, diesel cars, buses and lorries. The results are presented in Table 8.3.

**Table 8.3: Total distance, time and fuel spent before and after pedestrianisation**

| Vehicle category | Before pedestrianisation |              |                   | After pedestrianisation |              |                   |         |
|------------------|--------------------------|--------------|-------------------|-------------------------|--------------|-------------------|---------|
|                  | Distance (pcu.km)        | Time (pcu.h) | Fuel consumed (l) | Distance (pcu.km)       | Time (pcu.h) | Fuel consumed (l) |         |
| Car              | petrol                   | 33,490.3     | 2,564.5           | 2,423.7                 | 33,274.8     | 2,474.5           | 2,347.2 |
|                  | diesel                   | 2,471.6      | 189.2             | 248.2                   | 2,451.8      | 182.2             | 240.0   |
| Lorry            | 5,948.5                  | 455.5        | 597.5             | 5,900.8                 | 438.5        | 577.7             |         |
| Bus              | 2,209.2                  | 183.5        | 221.9             | 2,535.4                 | 206.9        | 248.3             |         |
| Total            | 44,119.6                 | 3,392.7      | 3,491.3           | 44,162.8                | 3,302.1      | 3,413.2           |         |



These figures were multiplied by the unitary environmental costs (per each kilometre travelled) and economic costs (per minute spent on travelling and litre of fuel consumed) which have been described in Chapter 7 (these costs are given by Tables 7.5 and 7.1, respectively). This yields the total economic and environmental costs incurred in the situations 'before' and 'after' the implementation of the pedestrian scheme. For the estimation of the environmental costs produced by road traffic, the units 'pcus' have been transformed in terms of 'vehicles'. Appropriate conversions have been applied, since both buses and lorries are considered to be equivalent to 3 pcus by the traffic assignment model, whereas each light vehicle is equivalent to 1 pcu (see Section 6.1.1). The estimation of the value of time spent by travellers using each mode of transport required information on the vehicle occupancy rates which were derived from the results of the modal split model. The expenses with fuel were obtained directly from the consumption rate of each type of vehicle. Table 8.4 presents the economic and environmental costs under consideration (given in £) which were incurred by each type of vehicle during the simulation period.

**Table 8.4: Economic and environmental costs incurred before and after pedestrianisation**

| Vehicle category | Before pedestrianisation (£) |             |                    |                 | After pedestrianisation (£) |             |                    |                 |        |
|------------------|------------------------------|-------------|--------------------|-----------------|-----------------------------|-------------|--------------------|-----------------|--------|
|                  | Envir. costs                 | Travel time | Expenses with fuel | Total costs (£) | Envir. costs                | Travel time | Expenses with fuel | Total costs (£) |        |
| Car              | petrol                       | 559         | 28,774             | 1,214           | 30,547                      | 556         | 27,764             | 1,176           | 29,496 |
|                  | diesel                       | 36          | 2,123              | 122             | 2,281                       | 36          | 2,044              | 118             | 2,198  |
| Lorry            |                              | 179         | 1,704              | 295             | 2,178                       | 178         | 1,640              | 285             | 2,103  |
| Bus              |                              | 79          | 638                | 109             | 826                         | 90          | 1,033              | 122             | 1,245  |
| Total            |                              | 853         | 33,239             | 1,740           | 35,832                      | 860         | 32,481             | 1,701           | 35,042 |

The costs related to the value of time spent on travelling which were incurred in the Chester road network both 'before' and 'after' the implementation of the pedestrian scheme are far higher than the other costs considered in the EAF. This finding seems to justify the trend in traffic assignment experiments to express travel costs only in terms of travel time. The expenses with fuel consumption only represent a small fraction of the total costs incurred. The environmental costs are the lowest in real terms. These are comparable (in the order of half) to the costs incurred from the consumption of fuel.

The costs incurred from the environmental degradation, time spent in travelling and fuel consumption were higher in the 'before' scenario in relation to all vehicle categories, except for the buses. These results are in line with the outcomes of the assignment model, which predicted that the total travel time (the most important parameter in the EAF) and fuel consumption decrease after the pedestrian roads are closed to traffic (see Figure 8.3 above and text explanation). The increase in the costs produced by buses was due to the augmented kilometres, fuel consumption and journey time driven by buses after the central roads were closed. The increase in the total environmental costs produced by buses after pedestrianisation outweighed the decreases of these costs provided by the other vehicle types, and this contributed to an overall increase in the environmental costs. However, the more elevated cost burden produced by buses did not outweigh the gains from the other categories in terms of travel time costs and expenses with fuel consumption, what resulted in a decrease of the costs with travel time and fuel. The occupancy rate in the buses was actually very low, but had there been more passengers travelling by public transport, the total travel time costs could have been higher in the 'after' case.

The total economic and environmental costs incurred in the 'before' situation are actually higher than in the 'after' scenario. This finding indicates that pedestrianisation will bring an overall economic efficiency to the road network of Chester. Even though the scope for inaccuracy in the economic valuation is vast and the differences in costs between the 'before' and 'after' scenarios may seem small, the relative cost estimates have been made for one hour period only and annual differences might account for much more significant amounts.

In conclusion, the results from the SPITE have shown that the pedestrianisation will provoke distinct variations in the levels of environmental impacts for different parts of the network, particularly in relation to different environmental indicators. In general terms, the implementation of this scheme has proven to be beneficial to the population of the town of Chester. Great reductions to the levels of environmental degradation have been found in the central area, and such benefits accrue to a large number of people who live, work or visit the pedestrianised roads.

The EAF produced a range of economic and environmental indicators which can be very useful for assessment purposes. The environmental costs have proven to represent only a small fraction of the total costs incurred, which should not necessarily dismiss its relevance in decision-making.

## 8.2 The implementation of a traffic calming scheme in Chester

Traffic calming comprises of a range of traffic management and engineering measures designed to restrain through traffic or slow traffic down, in order to encourage driving behaviour that is in keeping with the area. All traffic restraint schemes provide local environmental benefits to a greater or lesser extent (TEST, 1988) and safety to pedestrians, particularly at environmentally sensitive locations such as residential areas. They also tend to encourage the use of public transport, walking and cycling. Traffic calming schemes can be executed in a specific location or as part of a area-wide traffic restraint project.

Traffic calming has been implemented in a variety of forms, scales and creative designs, noticeably in continental Europe (Tolley, 1990). There is a growing appreciation by the local authorities in the U.K. of the attractiveness provided by traffic calming schemes. The most common traffic calming schemes are road humps and '20-mile-per-hour' speed zones. They can consist of speed bumps, cushions, ramps or tables, rumble strips, pinch points, chicanes, width restrictions, mini-roundabouts, speed control islands, pedestrian refuges, footway widening, entry treatment, traffic throttle, loop road, road depression, raised plateau, road closure, signs and road markings, culs-de-sac, pavement peninsula, etc. Traffic calming may be applied in conjunction with other street furniture, design paving, trees, seats and other pedestrian or cycling facilities. There are no rules for the adoption of particular types of traffic calming schemes. However, central governments can provide advice and regulations about where and how traffic calming measures can be used. Special regulations have been made in Britain by the Department of Transport in relation to road humps, which have recently relaxed previous stringent controls, giving more freedom in the siting of their features (Hass-Klau et al., 1992).

The cost-benefit analyses of traffic calming schemes in Britain, using construction cost and financial valuation of accidents for before and after periods, have shown that 14 out of 16 schemes tested offered 'good value for money', justified only on the basis of accident savings (Hass-Klau et al., 1992). The incorporation of further important economic and environmental improvements, such as an increase in property values and a decrease in air pollution and noise levels, would enlarge the overall benefits presented by these schemes and make them even 'better value for money'. A survey on public acceptance of traffic calming schemes in four areas in Britain revealed that the majority of respondents felt well disposed towards the schemes (Windle and Mackie, 1992).

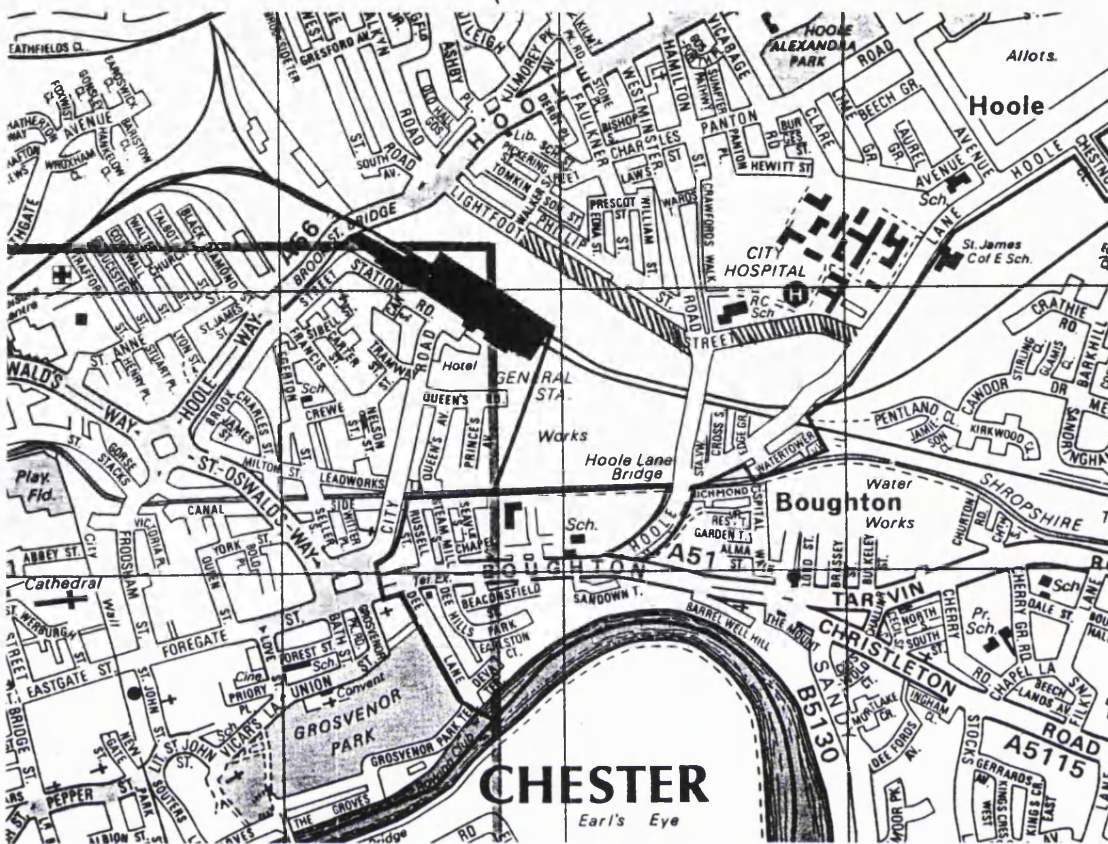
Traffic calming can also result in greater inconvenience and create other negative impacts. By reducing traffic speed, calming schemes can make journeys longer and cause an increase in fuel consumption and emissions of air pollutants and noise. By provoking traffic displacements, alternative routes become more congested. Traffic calming can only produce benefits in terms of air pollution if the effects of environmental improvements from reductions in traffic volumes are more significant than the effects from speed reductions, since the latter provokes an increase in the emissions of most pollutants. Thus, it is not clear whether traffic calming schemes always lead to improved overall environmental quality conditions, but they usually comply with the aim of alleviating local impacts.

Traffic calming schemes can play an important role in relation to the levels of environmental impacts produced locally. The implementation of traffic calming schemes in Germany showed reductions in traffic noise levels by 1 to 3 dB(A) (Hass-Klau et al., 1992). The average traffic speed exerts a logarithmic but important influence on noise levels (this relationship can be verified in Equation 4.9). Speed reductions of around 30%, which are usually achieved with these schemes, would produce average noise reductions in the order of 2.5 to 4 dB(A). Although the implementation of road humps provokes a discouragement of traffic, particularly of heavy goods vehicles, and a decrease in speed, they can actually increase the rolling noise emissions of individual vehicles. The sound of acceleration as vehicles move both over and away from humps can also lead to increased power noise (Kent County Council, 1994).

Air pollution can either decrease as a result of the decline in traffic or increase as vehicles change from constant to varying speeds. Some observations in the U.K. suggest that drivers maintain a constant but lower speed along the treated road. This allied with the reduction in the number of vehicles circulating through the calmed area is likely to improve the overall air quality (Kent County Council, 1994).

The Cheshire County Council has plans to implement a number of traffic calming schemes in diverse locations of the town. Currently, there are over 30 roads on the list for assessment of environmental traffic calming schemes in Chester. One of the locations which has been defined is on Lightfoot Street, by the City Hospital, between Hoole Road and Hoole Lane. This scheme is part of the implementation of a '20-miles-per-hour' zone and consists of a series of road humps along the street in order to reduce traffic speed and deviate through traffic. Figure 8.7 shows the area where the traffic calming scheme will be implemented. The traffic calmed street can be seen in dashed.

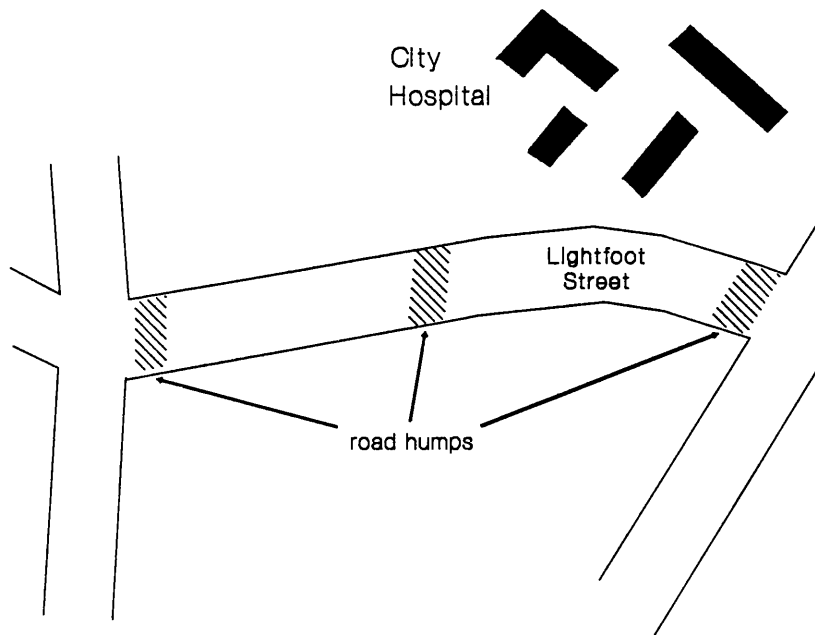
Figure 8.7: Localisation of the traffic calming scheme area



As the County Council has not designed a definite project for the traffic calming scheme, it was assumed for the modelling purposes that there will be three humps along the street, one in each end and one in the middle. Figure 8.8 shows in greater detail the street traffic calmed with road humps.

The implementation of road humps in Lightfoot Street in Chester is expected to produce changes both in the characteristics of traffic flows and in the levels of local environmental impacts.

**Figure 8.8: Traffic calming scheme in Lightfoot Street**



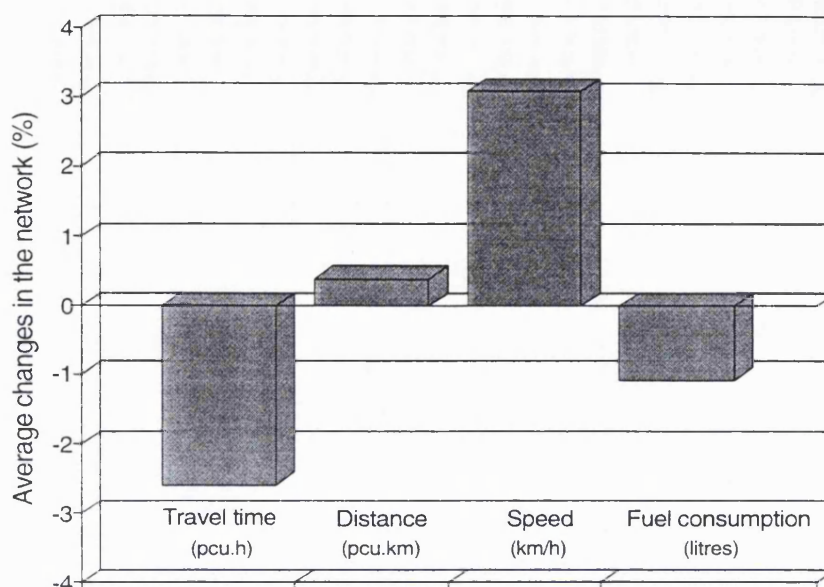
The implementation of road humps was simulated by altering the network coding file, which contains input data to the traffic assignment model. A new node was introduced in the middle length of the street, as if the street had been divided into two segments. The three nodes situated along the street were set to coincide with the locations where humps have been placed. The turning capacity at each node was reduced to represent the greater difficulty to cross the hump, and the travel time on the that particular street was increased in order to simulate the decrease in traffic speed along the street.

The environmental impacts from the deceleration and acceleration of vehicles moving both toward and away from the road humps along the traffic calmed road will be accounted for in terms of emissions (vehicle emissions were modelled as a function of the patterns of driving, as reported in Section 6.2.1), but will not be considered in terms of noise (traffic noise was model with basis on the average traffic speed, as reported in Section 6.3).

The percentage changes in the traffic indicators for the road network in the case study are shown by Figure 8.9. These changes are related to the base-scenario estimates (or 'before' situation).



**Figure 8.9: Percentage changes in the traffic indicators due to traffic calming**

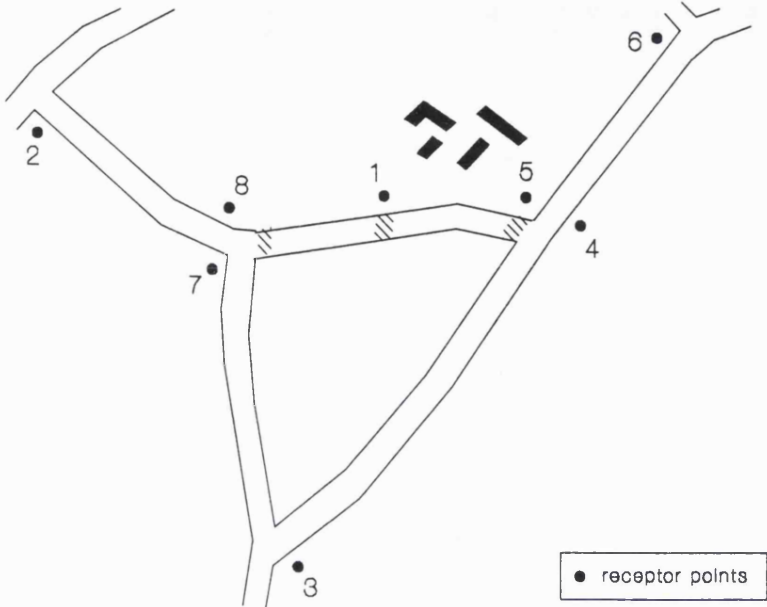


As in the case of pedestrianisation, the total travel time and fuel consumption in the whole network have decreased and the average traffic speed has increased after one particular road was traffic calmed. Once again a traffic restriction has proven to improve the overall circulation conditions (more details about this phenomenon can be found in Section 8.1). Nevertheless, the total distance travelled has increased, as in the pedestrianisation scenario.

The changes in the traffic indicators are small, since only a fraction of the network is affected by changes in the traffic conditions. Nevertheless, the implementation of this particular traffic calming scheme has proved to provide benefits in economic terms, by provoking an overall reduction in the consumption of fuel and time spent in travelling.

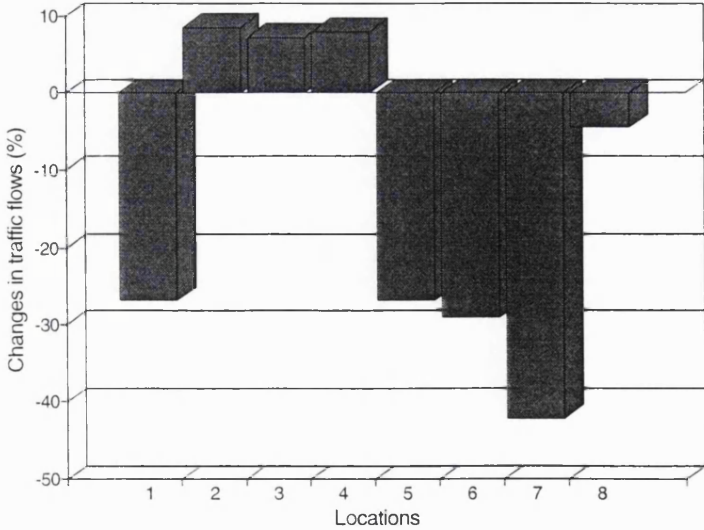
The analysis of the traffic and environmental changes are concentrated in the area around the traffic calming scheme, which comprises the traffic calmed street and surrounding roads. Figure 8.10 illustrates the area most influenced by the implementation of the traffic calming scheme and shows the locations (receptor points) where the environmental impacts are estimated (As reported in Section 6.2.2, each receptor point represents the location where maximum levels of environmental degradation are expected along the road. Two-way roads have two reception locations).

**Figure 8.10: Representation of the area around the traffic calmed street**



The conditions of traffic circulation have varied within the traffic calmed area after the implementation of road humps was simulated. Figure 8.11 shows the percentage changes in traffic flow between the 'before' and 'after' scenarios for the roads corresponding to the receptor points shown in Figure 8.10.

**Figure 8.11: Percentage changes in traffic flows**

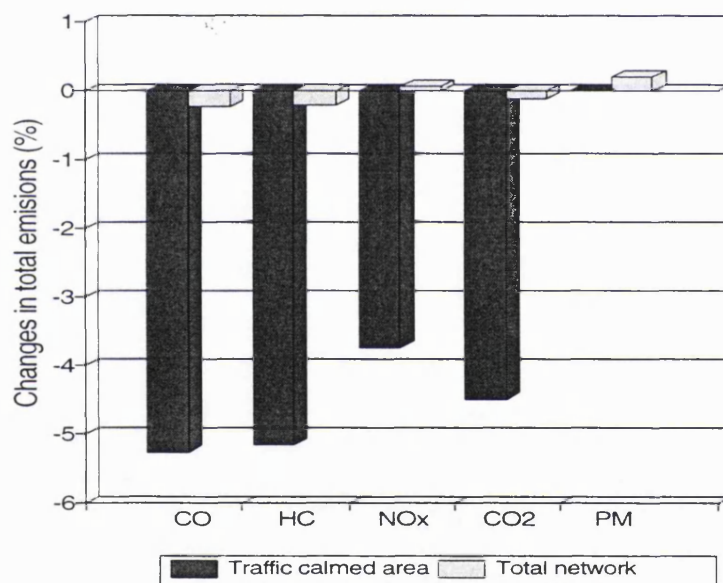




The traffic flows have decreased in most roads at various rates. The traffic flow in the road represented by the location 7 was decreased by over 40% whereas the flow in both segments of the calmed street was reduced by nearly 30%. Such reduction in traffic volumes was indeed intended by the setting up of the scheme. However, the flows in the roads represented by the locations 2, 3 and 4 were increased by less than 10% after the implementation of traffic calming. This was due to the deviation of some traffic from the calmed street.

As a result of changes in traffic conditions, the total vehicle emissions also varied. The percentage changes in the total emissions produced 'before' and 'after' the implementation of road humps in the traffic calmed street are illustrated in Figure 8.12. The changes in total emissions are shown for the traffic calmed area and for the entire road network.

**Figure 8.12: Changes in total emissions due to traffic calming**



The total emissions produced in the traffic calmed area have decreased by about 5%, with the exception of PM emissions which did not change at all. The reduction in the emission of most pollutants was due to the reduction in traffic volumes within the calmed area. The traffic changes, however, did not affect PM emissions. Cars emit a low small rate of this pollutant and buses, which are responsible for a considerable proportion of PM emissions, were not affected by the implementation of the calming scheme because there were no bus routes through the calmed street. The changes in the total emissions in the whole network have been negligible. Like pedestrianisation, traffic calming schemes and any other small-

scale transport plans are very unlikely to produce significant impacts in terms of global emissions.

The local environmental impacts were then estimated by the application of the air pollution dispersion and the noise propagation models in the SPITE. The percentage changes in the environmental impacts for the locations in the traffic calmed area are shown in Table 8.5. These locations correspond to the points indicated in Figure 8.10 above.

**Table 8.5: Percentage changes in environmental impacts in the traffic calmed area**

| Location | Pollutant concentration |       |                 |                 |       | Noise<br>L <sub>10, 1-hour</sub> |
|----------|-------------------------|-------|-----------------|-----------------|-------|----------------------------------|
|          | CO                      | HC    | NO <sub>x</sub> | CO <sub>2</sub> | PM    |                                  |
| 1        | -35.8                   | -36.3 | -23.9           | -28.4           | -15.9 | +0.1                             |
| 2        | +14.2                   | +13.7 | +4.2            | +11.3           | +0.1  | 0.0                              |
| 3        | -3.3                    | -3.0  | -2.8            | -3.0            | -2.0  | -0.2                             |
| 4        | +3.0                    | +1.6  | +2.4            | +2.0            | -2.0  | +0.1                             |
| 5        | +3.8                    | +1.1  | +4.0            | +2.9            | -2.7  | +0.2                             |
| 6        | -1.9                    | -2.2  | -3.4            | -2.0            | -3.4  | -0.7                             |
| 7        | -43.9                   | -37.7 | -34.1           | -37.0           | -16.9 | -1.1                             |
| 8        | -43.1                   | -36.4 | -32.6           | -35.5           | -12.4 | -1.3                             |

The largest reductions in air pollution concentrations were verified at points 1, 7 and 8, which are the closest locations to the calming scheme, where traffic volume has been mostly reduced. These reductions were variable from about 13 to 44%, according to the pollutant and location. However, there have been actual increases in the concentration levels of pollutants in other locations, such as points 2, 4 and 5. These increases were due to the rise in traffic flows in the roads adjacent to them, as a result of traffic diversion. Very small improvements in air quality have been verified at points 3 and 6. Pollutant concentrations do not vary proportionally to changes in traffic flows because they are a result of the contribution of traffic sources emitting different exhaust rates at various distances, from different roads around the receptor point.

The changes in traffic noise levels were not significant, apart from those at locations 6, 7 and

8, where noise was reduced by about 1%. This suggests that transport policies can be little effective in reducing noise levels. The differences found are small in relative terms, and such modest changes are induced by the logarithmic relationship between noise levels and traffic flow and speed. In real terms, however, these differences represent 0.5 to 0.9 dB(A), which can be relevant for transport planning purposes (see Section 4.3).

The application of the modal split model has indicated that the pattern of travel choices between the modes available has not suffered any change after the traffic calming scheme was implemented.

The assignment model has estimated the characteristics of traffic flows for the 'before' and 'after' traffic calming scenarios. The total distance travelled, travel time and fuel consumed by each vehicle category (petrol cars, diesel cars, buses and lorries) are shown in Table 8.6.

**Table 8.6: Total distance, time and fuel spent before and after traffic calming**

| Vehicle category | Before traffic calming |                |                   | After traffic calming |                |                   |         |
|------------------|------------------------|----------------|-------------------|-----------------------|----------------|-------------------|---------|
|                  | Distance (pcu.km)      | Time (pcu.h)   | Fuel consumed (l) | Distance (pcu.km)     | Time (pcu.h)   | Fuel consumed (l) |         |
| Car              | petrol                 | 33,490.3       | 2,564.5           | 2,423.7               | 33,625.3       | 2,493.4           | 2,395.0 |
|                  | diesel                 | 2,471.6        | 189.2             | 248.2                 | 2,480.9        | 184.3             | 245.6   |
| Lorry            | 5,948.5                | 455.5          | 597.5             | 5,971.0               | 443.6          | 591.0             |         |
| Bus              | 2,209.2                | 183.5          | 221.9             | 2,209.2               | 183.5          | 221.9             |         |
| <b>Total</b>     | <b>44,119.6</b>        | <b>3,392.7</b> | <b>3,491.3</b>    | <b>44,286.4</b>       | <b>3,304.8</b> | <b>3,453.5</b>    |         |

These figures were derived, as in the previous section, by the application of a model created for the estimation of the distance, travel time and fuel consumption incurred by each vehicle category. As can be noticed, buses did not suffer changes from the implementation of the traffic calming scheme at the particular location analysed.

Table 8.6 provide the basis for the EAF, which estimates the total economic and environmental costs incurred both in the 'before' and 'after' situations. Table 8.7 presents the costs under consideration (these cost were estimated as reported in Section 8.1) which were incurred by each type of vehicle during the simulation period.

**Table 8.7: Economic and environmental costs incurred before and after traffic calming**

| Vehicle category | Before traffic calming (£) |             |                    |                 | After traffic calming (£) |             |                    |                 |        |
|------------------|----------------------------|-------------|--------------------|-----------------|---------------------------|-------------|--------------------|-----------------|--------|
|                  | Envir. costs               | Travel time | Expenses with fuel | Total costs (£) | Envir. costs              | Travel time | Expenses with fuel | Total costs (£) |        |
| Car              | petrol                     | 559         | 28,774             | 1,214           | 30,547                    | 562         | 27,961             | 1,200           | 29,723 |
|                  | diesel                     | 36          | 2,123              | 122             | 2,281                     | 36          | 2,068              | 121             | 2,225  |
| Lorry            | 179                        | 1,704       | 295                | 2,178           | 180                       | 1,659       | 291                | 2,130           |        |
| Bus              | 79                         | 638         | 109                | 826             | 79                        | 638         | 109                | 826             |        |
| Total            | 853                        | 33,239      | 1,740              | 35,832          | 857                       | 32,326      | 1,721              | 34,904          |        |

As in the case of pedestrianisation, the costs related to the value of time spent on travelling are also far higher in this case than the other costs contained in the EAF. The environmental costs are the lowest in real terms, and represent only a fraction of the total costs incurred.

The costs incurred from the time spent in travelling and fuel consumption were higher in the 'before' situation in relation to all vehicle categories, except for buses, which had their circulation conditions unchanged. However, the environmental degradation costs are higher in the 'after' scenario. Like the results of the simulation of the pedestrianisation scheme (Section 8.1), the total costs incurred in the network are higher in the 'before' situation. This suggests that the implementation of the traffic calming scheme yields overall economic benefits.

In conclusion, the results of the SPITE have shown that the implementation of traffic calming in a particular location in Chester produces significant reductions to the levels of environmental impacts in the vicinity of the scheme. Air quality has improved in five out of eight locations, those which are more sensitive and for which improvements are mostly needed. The air quality has deteriorated slightly in a few locations as a result of the traffic re-distribution.

Although the total environmental costs have increased due to the greater distances driven by some vehicles, the implementation of this scheme has proven to be beneficial in overall economic terms.

### **8.3 Restricting the traffic of heavy goods vehicles in the central area**

The presence of heavy goods vehicle traffic is very unpleasant for local residents, car drivers, pedestrians and cyclists. These vehicles are responsible for a great deal of the environmental degradation in urban centres and generate risk for other road users. Lorries emit considerably more noise than cars (Section 4.3) and massive quantities of PM originate from the combustion of fuel from such powerful diesel engines (Section 3.2). The emission rate data given in Tables 6.1 and 6.2 show that one lorry emits over 75 times more PM than an average petrol car. A lorry also emits larger amounts of NO<sub>x</sub> than a petrol car equipped with a catalytic converter (over 20 times more).

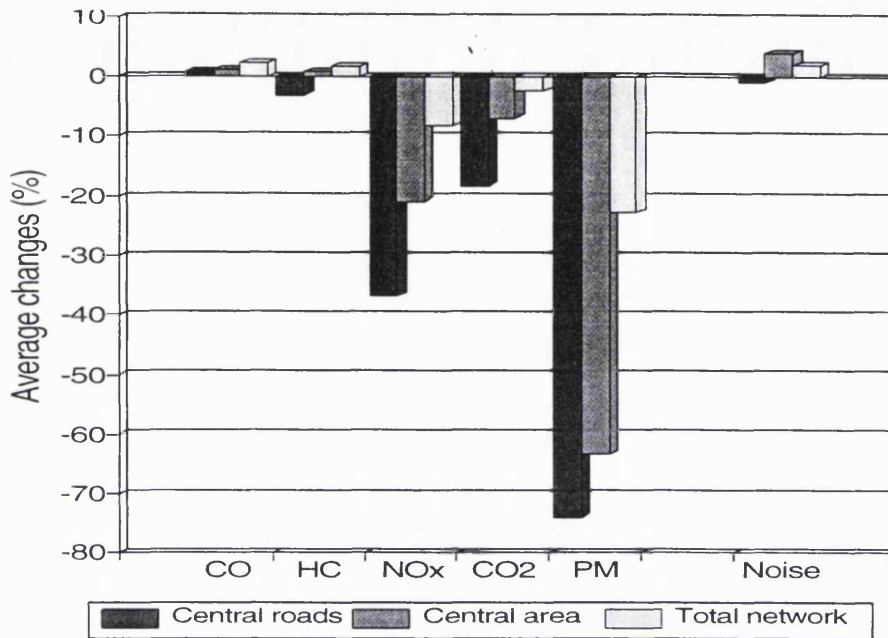
For such reasons, lorries very often cause concern to environmentalists and local authorities in transport planning matters. Measures to reduce the environmental impacts of heavy goods vehicles include deviating or banning their circulation in residential or central areas and restricting their delivery times. Traffic calming schemes (see Section 8.2) can be specifically designed to prevent or discourage heavy vehicles from penetrating certain roads or areas.

Restrictions to the flow of heavy vehicles can affect the quality of the local environment in two distinct ways. First, by directly reducing pollutants (particularly PM) and noise emissions as these vehicles cause more pollution than cars, and second, by improving the traffic flow conditions since a lorry occupies the road space equivalent to about three cars.

In order to quantify the extent to which schemes involving the removal of lorries could improve local environmental quality in central or residential areas, a hypothetical situation is considered in which heavy vehicle traffic is banned from certain roads in the central area of Chester. The selected roads are those to which pedestrianisation has been simulated and are given in Section 8.1. However, the implementation of a traffic planning scheme such as this cannot always be put into practice over a wider area since deviations would not only affect the heavy goods vehicles through traffic but also the essential delivery vehicles which would be denied access, and this could have serious economic implications.

All traffic was re-assigned into the network considering the lorry restriction scheme and the SPITE was applied to all receptor locations. The mean changes to air pollution concentration and noise levels in Chester caused by the restriction of the heavy goods vehicles traffic in the central roads are shown in Figure 8.13 below.

**Figure 8.13: Changes in environmental impacts from restricting heavy vehicles traffic**



The exclusion of heavy goods vehicles from the selected roads in the central area produces major reductions in the average level of PM concentration. The average reduction in the central locations (the roads from which lorries were deviated) was about 74%. Considering all the locations in the central area the average reduction was 63% and in the total network it was nearly 23%. The maximum reduction in PM concentration amongst all the receptor locations was found to be 93%. This means that the flow of lorries, which accounts for less than 5% of the total traffic flow, is responsible for a great deal of the PM pollution in Chester.

The exclusion of heavy goods vehicles also reduced the average level of NO<sub>x</sub> concentrations by about 37% within the locations where restrictions were implemented and by about 21% considering all the locations in the central area. The maximum reduction achieved in the concentration of this pollutant was about 77%. The reductions in average concentration levels of other pollutants were not substantial.

Noise levels were reduced by an average of 1% in the central locations, but a maximum reduction of about 9% was obtained in some areas. In percentage terms, the average improvements do not seem to be meaningful. However, these reductions represent about 1 dB(A) in real terms which is equivalent to about 20% reduction in traffic (Section 4.3) and

thereby can represent an appreciable benefit to residents in the area.

These findings indicate that transport policies which provoke changes to the flow of lorries can have considerable impacts on the quality of the local environment. Therefore, transport planners should consider the effect that such policies have on levels of PM concentrations and traffic noise at early stages in the planning process.

The EAF was not applied to this particular policy, because the most relevant implications from the removal of lorries from the central area (such as delay penalties for late loadings, loading and delivery times, etc.) are not accounted for in the framework. The difference in the travel time (which is the main component of the total costs) for lorry drivers is not such a relevant consideration in the assessment of this policy.

#### **8.4 Other policies**

The objective of this section is to show the potential of the application of the SPITE in conjunction with transport models for generating information about the environmental impacts of traffic. Policies whose implementation tends to affect the characteristics of travel demand and the subsequent impacts on the environment that they produce are briefly presented and some conclusions are drawn. However, a full environmental analysis (as given in preceding sections) has not been carried out for these policies.

##### ***8.4.1 Improvements in the provision of public transport***

One of the most effective means of curbing air pollution in urban areas is to find alternatives to car use such as promoting the use of public transport, cycling and walking. Public transport priority schemes, such as bus lanes, increase the speed of buses and therefore decrease the in-vehicle travel time. Other improvements in the quality of service, reliability, convenience, safety and comfort of public transport would be expected to increase its attractiveness and impact modal choice. The potential for such modal transfer is high in Britain, where about 72% of journeys are under 8 kilometres in length and 29% are under 1.6 kilometres (Department of Transport, 1994c).

Various steps can be taken to improve the attractiveness of public transport, for example, increasing the frequency tends to shorten the waiting time for the passenger at the stop and



increasing the number of routes, dispersing them through the network and diminishing stop spacings tend to reduce walking distance and time. The implementation of park-and-ride facilities and traveller information systems also enhance the experience of travelling by public transport. Measures which give priority to public transport systems provide superior energy efficiency and considerable environmental benefits, since the public transport is capable of carrying more people per unit of energy consumed than private cars. Also, the environmental degradation per passenger-kilometre of public transport vehicles is usually very small in comparison to cars provided that the occupancy rate is sufficiently high. Public transport priority measures have been shown to be able to reduce bus exhaust emissions by up to 60% (European Conference of Ministers of Transport, 1990). Horowitz (1982a) reports that public transport improvements provide environmental benefits mainly in the form of reductions in the use of the car for work trips in urban areas. Improvements in public transport added to carpool incentives were found capable of reducing car emissions from work-trips in high volume corridors by 20-30% during peak hours. Krawack (1993) relates considerable reductions in NO<sub>x</sub> and particulate emissions from implementing bus lanes and bus priority schemes at traffic lights. The Department of the Environment and Department of Transport (1993) suggest that emissions may be reduced by an order of 16% by extending public transport provision and regenerating existing centres whilst limiting the increase in highway capacity.

#### ***8.4.2 Reduction of public transport fare***

Most metropolitan cities in both developed and developing nations subsidize public transport. Therefore transport policies are potentially able to promote incentives to attract travellers to public transport and create a more efficient and environmentally friendly transport system.

The extent to which travellers may find it advantageous to switch from private to public transport depends greatly on the fare level employed. The mean elasticity of bus fare with respect to passenger demand has been found to be -0.41 (Goodwin, 1992), however, bus fare elasticities of about -0.3 are also common in urban areas (European Conference of Ministers of Transport, 1990). This means that low fares will help to reduce car use, but only to a limited extent. Reducing fare levels will likely generate additional trips among public transport users, however, it would be most effective if it can encourage car users to transfer to the public transport.



#### ***8.4.3 Introduction of a new public transport mode***

New modes of transport may be necessary in some cases in order to provide the service required by travellers. For instance, the introduction of a light rail transit system in Chester has been proposed with the aim of providing fast and reliable links into the city centre and a study has been undertaken into the feasibility of implementation of this additional mode of transport (Cheshire County Council, 1990). The introduction of the new tram system would be expected to increase the attractiveness of public transport and reduce car traffic which would considerably affect the patterns of travel in the city. Preliminary simulations carried out on the implementation of a light rail system in Chester have indicated significant environmental benefits (Chiquetto, 1994) since the electrical powered vehicles would emit no fumes and the noise levels produced by trams are far lower than those of buses.

#### ***8.4.4 Restrictions in car ownership***

Car ownership is one of the principal factors determining whether people travel by car or not. A reduction in car ownership is expected to affect not only car use but also to cut down trip rates and lengths. Changes in car ownership can be achieved through increasing taxes on both vehicle purchase price and annual duties, such as road tax and insurance. Taxes could be set according to the level of exhaust and noise emissions so as to discourage the purchase of more polluting vehicles (see economic instruments in Section 5.1). The elasticity of car ownership with respect to car price has been found at -0.87 (Goodwin, 1988). Car ownership measures have been implemented, for instance, in Hong Kong where about 20% reduction was achieved in the number of daily trips (Jones, 1989) and in Bermuda, where a limit of one car per household has been enforced (May, 1986).

In the U.K., many companies have traditionally provided employees with cars as a non-monetary perk whether or not it is utilised for the purposes of the job. The advantage to the employer is that less tax is charged on the value of company cars than on the equivalent sum in payments. This has encouraged drivers to use cars as the main form of transport and has led the U.K. to achieve the highest proportion of company cars in the world (Selwyn, 1995). This indicates that a profound change in the company cars policy ought to be pursued before other policies to control car ownership could be implemented.

#### ***8.4.5 Increases in private vehicle occupancy rate***

The use of higher occupancy modes of transport would improve overall energy efficiency and provide environmental benefits. Theoretically, an increase in car occupancy rate confers many of the solutions for the urban traffic problem. The most effective way of promoting an increase in car occupancy rate is through employer-based travel programmes (more details in Hughes, 1992). In practice, however, its implementation is very complex and therefore it is unlikely to promote successful long-term results. Road pricing or tolls may also be implemented in order to discourage single occupancy car trips amongst the poorer classes.

#### ***8.4.6 Optimising traffic signal times***

Many traffic light systems are operated on fixed time plans and do not account for varying traffic conditions. The effects of congestion could be significantly reduced if signal times are controlled on the basis of current flow information, for instance, as in the SCOOT systems employed by many cities in the U.K.

Improvements in the coordination of traffic signals may provide significant environmental gains in the vicinity of road intersections (Matzoros, 1988). The abatement of noise and pollutant emissions can be achieved since traffic conditions such as queues, speed, driving pattern, travel time, volume, capacity and fuel consumption could be optimised. Horowitz (1982a) reports a decrease of 3% on pollutant emissions plus traffic speed increases after signal synchronization when traffic volume remained constant. The Institution of Highways and Transportation (1992) estimates that vehicle delays could be reduced by 13% if signal timing were optimised. It has been demonstrated that coordinating signal timings can only reduce emissions marginally in an unsaturated network, but decreases could be substantial in congested networks (Bell, 1990). An Urban Traffic Control system has recently been implemented in Chester and the County Council expects to achieve a 15% extra capacity on the existing road network just by optimising signal timings.

#### ***8.4.7 Vehicle technological improvements***

Technological improvements have occurred recently in several areas of the automobile industry. Vehicle manufacturers have been spending a great deal of time and resources in developing more technologically advanced products. Cars are becoming faster, safer, more

reliable, more comfortable, more economic and efficient in terms of fuel consumption and more environmentally friendly. Wachs (1993) suggests that some pollutants are reduced by 98% in new cars when compared with new American cars twenty years ago and that nowadays 80% of the air pollution is produced by the 10% of the vehicles which are very old or badly out of tune. This picture is similar in the U.K. and is to a large extent the result of the pressures of governmental and environmental organisations on vehicle manufacturers.

Vehicle emissions have been reduced by technological improvements as much as by the introduction of regulations and emission checks. The main improvements in vehicle efficiency include electronic ignition and fuel injection, improved carburettors, recirculation of the exhaust gas oxides, closed crankcase ventilation, air injection, turbochargers, improved intake manifolds, improved fuel tanks with charcoal canisters, lower compression in the combustion chamber (Gould, 1989) and the introduction of catalytic converters. European regulations have established emission limits, which forces car manufacturers to control pollutant exhausts (see Table 3.4), and emission checks have been introduced into the MoT test, which forces car owners to keep their engines in tune.

The decreases in vehicle exhaust emissions which occurred within the last decade have produced significant environmental benefits (Chiquetto and Mackett, 1994a). The environmental degradation produced by emission rates of in-service cars in the U.K. over ten years ago when the Warren Spring Laboratory carried out two emission survey studies (Potter and Savage, 1982 and 1983) was compared by Chiquetto and Mackett (1994a) to the current emission rates (as given in Section 6.2.1), in order to verify the degree of environmental improvements provided by emission reductions over the last decade. When the characteristics of traffic flows were kept constant, the results of the modelling application in Chester have indicated that average levels of CO, HC and NO<sub>x</sub> concentrations have decreased by about 60%, 50% and 40%, respectively. This suggests that technological improvements in vehicle manufacturing and Government's initiatives during the last years have produced substantial reductions in exhaust emissions.

Further technological improvements are expected in the future. In conventional cars, only about 20% of the fuel is converted into power at the wheel, the rest being lost through normal operations of the vehicle. The maximum theoretical fuel conversion to motive power is around 37% and therefore there would appear to be great scope for improving engine efficiency (Selwyn, 1995) and further reducing vehicle emissions. Furthermore, the European

Union has established new emission limits for the 1996 models and has proposed stricter limits for the 2000 models which will guarantee that new cars produce reduced emission levels.

#### **8.4.8 Other alternatives**

Land use programmes can discourage the need to travel, by promoting population and employment moves or staggered working hours. Land use measures can induce the reduction of the number or length of trips and this would have a significant effect on the traffic levels in urban areas (Department of the Environment and Department of Transport, 1993). Restricting the use of vehicles on certain roads or areas will also encourage the use of non-motorised transport.

Improvements in fuel quality will provide better efficiency in combustion and less emissions. The Royal Commission on Environmental Pollution (1994) recommends that the Government develops specifications for cleaner fuels which will contribute to achieving the national targets for air quality.

Alternative fuels also have an important role to play in improving urban air quality (Holman et al., 1991). In the U.K., electric traction has been used for milk delivery and could be extended for other urban delivery services. Electric propulsion would have important effects on the conservation of fossil resources and would significantly reduce pollutant and noise emissions from transport sources. However, a counter argument is that the pollution is transferred from the cities into the rural areas where the power stations providing electricity are sited.

Considerable reduction in emissions can be achieved if drivers are prepared to accept lower performance vehicles which are more fuel efficient. Improvements in engine and catalyst efficiency as well as reductions of size and weight of vehicles will also produce less emissions.

On-board guidance and information systems will provide up to date information on the road conditions. This will enable the driver to choose short routes or those with least congestion which will result in less emissions. Eventually, tele-commuting will reduce travel needs, energy and transport emissions.

## 9 THE IMPACTS OF TRANSPORT TRENDS ON THE ENVIRONMENT

The need for transport within an area can change significantly according to the demographic growth, employment, housing, education and wealth of the population. Changes in transport needs and mobility conditions will therefore affect the characteristics of traffic flows in the area as well as the environmental impacts generated.

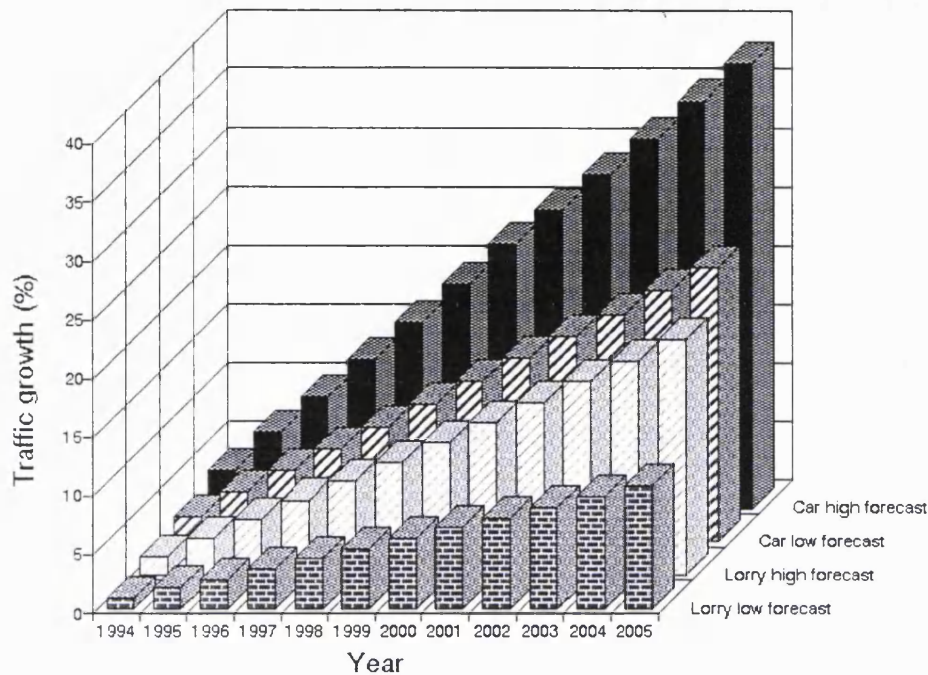
This chapter examines the extent to which long-term transport trends can influence total emissions and local levels of road traffic air pollution concentration and noise within the urban road network of Chester.

Three trends in the transport sector, which are expected to occur in the future, are considered in the analysis. The first trend will affect the traffic circulation conditions and these in turn will influence the vehicle emission patterns. The other two trends will affect directly vehicle emissions. These are:

- **traffic growth** and the consequent increase in congestion,
- **the introduction of the 1996 vehicle emission regulations** and restrictions on future vehicle emissions (see Tables 3.4, 3.5 and 3.6), and
- **changes in the vehicle market**, with the increasing penetration of petrol cars fitted with catalytic converters and diesel cars into the U.K. vehicle fleet.

The first transport trend to be considered in this chapter is the growth of traffic volumes over the next decade. The effects of traffic growth on future traffic circulation and environmental conditions are based on estimates of the National Road Traffic Forecasts (NRTF) (Department of Transport, 1989a). The low and high traffic growth forecasts for cars and goods vehicles, from 1993 to 2005, are shown in percentage terms in Figure 9.1.

**Figure 9.1: Traffic growth forecast (base year: 1993)**



Source: Department of Transport, 1989a.

The figure above shows that car traffic grows at a higher rate in comparison to lorries. The percentage of lorries in the total flow is estimated to decrease slightly from about 4.5% in 1993 to about 4% in 2005. This projection suggests that the effect of future growth in the needs for personal transport is likely to provoke a more distinct increase in traffic volumes than the effect of increasing demands for goods.

The forecasts for the growth in the traffic of cars and lorries look linear. They do not take into consideration many economic and social changes that may take place in the future. In fact, since the forecasts were made traffic activity has fallen below the levels originally predicted. The implications to the use of the NRTF for the estimation of future traffic trends, particularly in urban areas, is reported in the SACTRA report (Department of Transport, 1994d). An important consideration on the use of the NRTF is that traffic growth in urban areas is likely to be lower than the national trend due to congestion and more dynamic modal shifts. Thus, traffic projections applied to urban areas might incorporate an element of overestimation.

However, more updated forecasts have not been produced so far, and the NRTF are the

official forecasts of the Department of Transport in the U.K., who still considers them to be valid in the long term. These traffic forecasts have been consistently used in a variety of transport studies and are also applied in this work.

The second trend to be considered in this chapter is the reduction in emission rates imposed by the introduction of the emission limit regulations on new vehicles from 1996. As reported in Section 3.4, from 1996 all new vehicles must conform with the emission regulations. The enforcement of these regulations will change the patterns of vehicle emissions and should provide important benefits in terms of environmental quality. The suggested standards for the year 2000 are not considered in this analysis, since they are only recommendations and have not been put into a legislation framework as yet.

It is evident that less pollutants will be emitted by new vehicles from 1996 onwards, but it is practically impossible to estimate accurately future vehicle emission rates. Thus, the emission model simulates the effects of the emission regulations by assuming that each new vehicle brought into the market after 1996 will emit the maximum rate allowed by the regulation. The emission rates are a function of the limits for different vehicle categories.

The emission limits prescribed by the 1996 regulations (as shown in Tables 3.4, 3.5 and 3.6) are given as average rates (in grams per kilometre) for the whole driving cycle. In order to estimate the relative emission factors for new petrol cars after 1996 in each driving pattern, these were assumed proportional to the current emission distribution patterns, considering the distances and times spent in each pattern within the driving cycle. Since the legislation gives a single limit for the exhaust emission of HC and NO<sub>x</sub> together, the limits for each of these pollutants were also considered proportional to the current rates. The pollutant emission factors for new petrol cars in the market after 1996, as a function of each pattern of driving, are presented in Table 9.1.

**Table 9.1: Emission factors from new petrol cars after 1996 for each driving pattern**

| Driving pattern | Unit  | Emission factors |      |                 |
|-----------------|-------|------------------|------|-----------------|
|                 |       | CO               | HC   | NO <sub>x</sub> |
| Acceleration    | g/km  | 3.37             | 0.39 | 0.49            |
| Deceleration    | g/km  | 0.86             | 0.14 | 0.06            |
| Cruise          | g/km  | 1.73             | 0.19 | 0.16            |
| Idle            | g/min | 0.23             | 0.05 | 0.005           |

The emission limits imposed upon diesel vehicles will be introduced at the same time as the limits for petrol cars. The limits for diesel vehicles include a restriction on PM emissions, which is not specified for petrol cars. The emission factors for new diesel vehicles after 1996 have been applied in accordance with the emission limits, as indicated in Section 3.4. As there are no specific emission limits for buses, the emission factors after 1996 were assumed the same as the current rates from buses.

The assumption of the emission limit as the average emission rate for all new vehicles is generally acceptable in the prediction of future environmental impacts, because while some of the new vehicles will be badly maintained or out of tune (and therefore will emit over the limit), others which comply with the regulations will emit less than the maximum rate.

The third and last trend to be considered in this chapter is related to future changes in the vehicle fleet in the U.K. Two main changes are expected in the future car market. The first is the increasing penetration of petrol cars fitted with catalytic converters and the second is the increase in the proportion of diesel cars in the light vehicle fleet. Both changes have the potential to affect greatly local and global air pollution levels, since petrol cars fitted with catalytics and diesel cars produce very distinct patterns of emission in comparison to the standard petrol car without catalytic converters.

The effect of the market penetration of cars to future vehicle emissions has been based upon three projections produced by the Department of Transport (1994b). These projections have been applied in previous studies in order to represent future car market scenarios (Chiquetto and Mackett, 1994a and 1994b; and Chiquetto, 1995). However, for the purposes of this work, these projections have been further classified in order to account for other effects



which also influence vehicle emissions, such as the introduction of the 1996 emission regulation limits and the failure of catalytic converters.

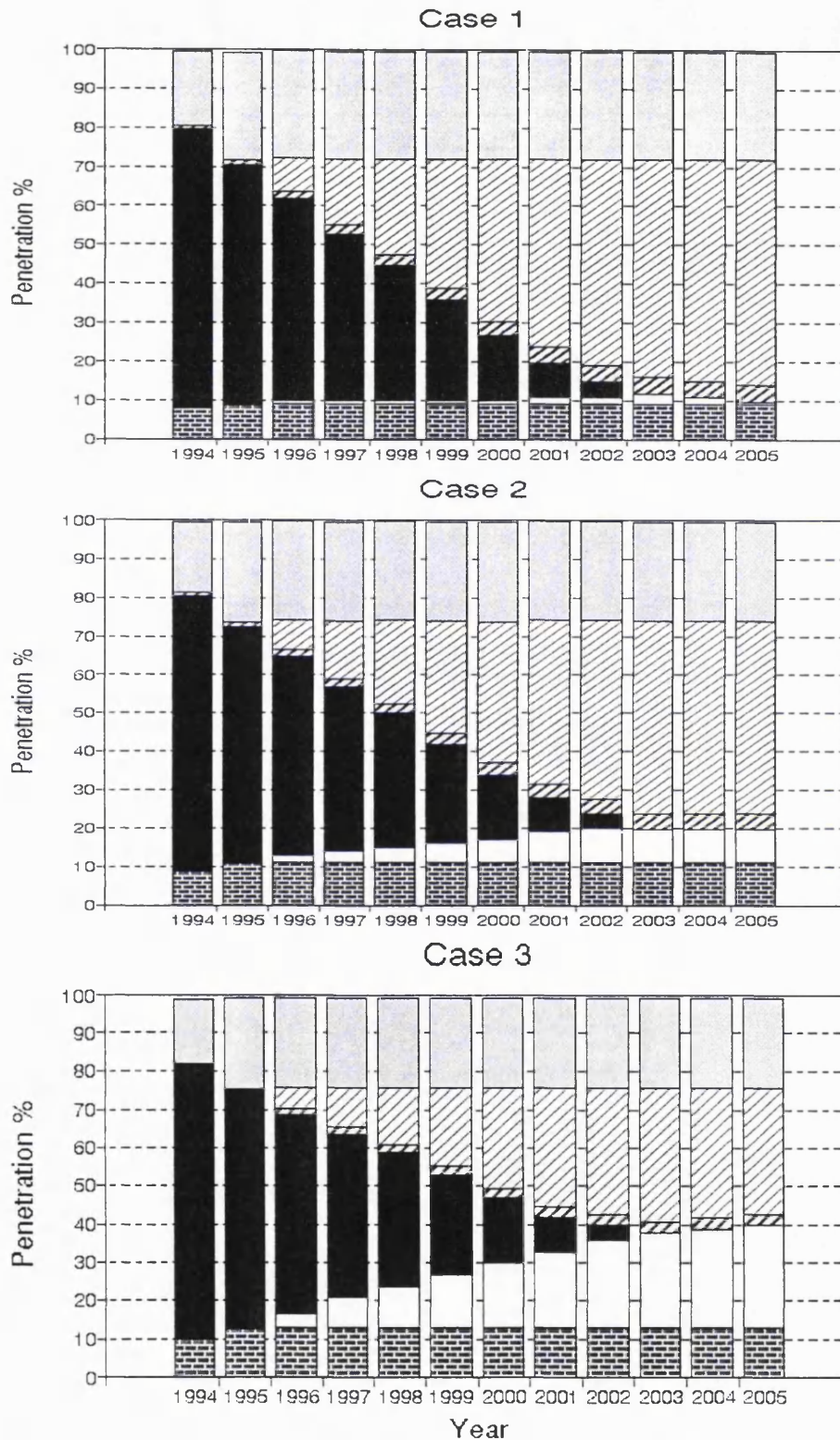
Figure 9.2 illustrates the projections of stock turnover profile to cars in three different cases. The likely proportions of cars in the future, in respect of the typical profiles of emissions produced, are given for various vehicle categories. The categories of vehicles considered are:

- diesel cars which produce current emission rates,
- new diesel cars which comply with the 1996 regulation limits,
- petrol cars without catalytic converters,
- petrol cars with failed catalytic devices,
- new petrol cars which comply with the 1996 regulation limits,
- petrol cars fitted with catalysis and which produce current emission rates.

It is assumed in all cases that the participation of petrol cars without catalytic converters in the fleet will decline constantly until their complete extinction in the year 2003. The proportion of diesel cars and the penetration of cars fitted with catalytic converters are estimated to increase in all cases. Case 1 assumes a lower increase in the participation of diesel cars (up to about 10% in the year 2005, split between diesel cars which emit the current rates and those which will comply with the emission regulation) and a higher proportion of petrol cars with catalytic devices. Case 2 estimates that the proportion of diesel cars will reach about 20% of the market at the end of the projection, whereas Case 3 assumes that the growth in diesel cars will make up to 40% of the fleet.

The simulations of future traffic circulation conditions and changing vehicle emission rates were carried out considering the three trends reported above: traffic growth, the introduction of the 1996 vehicle emission regulations and the changes in the car market. The following sections analyse separately the extent to which particular trends in the transport system can influence environmental quality and total emissions. Section 9.1 reports the effects of traffic growth considering the low and high forecasts. Section 9.2 analyses the effects of changes in the vehicle market, with particular reference to the different car market scenarios. Section 9.3 examines the effects of petrol price increases in the future, as recommended by the Royal Commission on Environmental Pollution (1994). Yearly scenarios were created up to the year 2005 in order to illustrate the sensitivity of each of these transport trends on the environment. The results of these projections will be compared to the base-scenario conditions.

Figure 9.2: Market penetration of cars



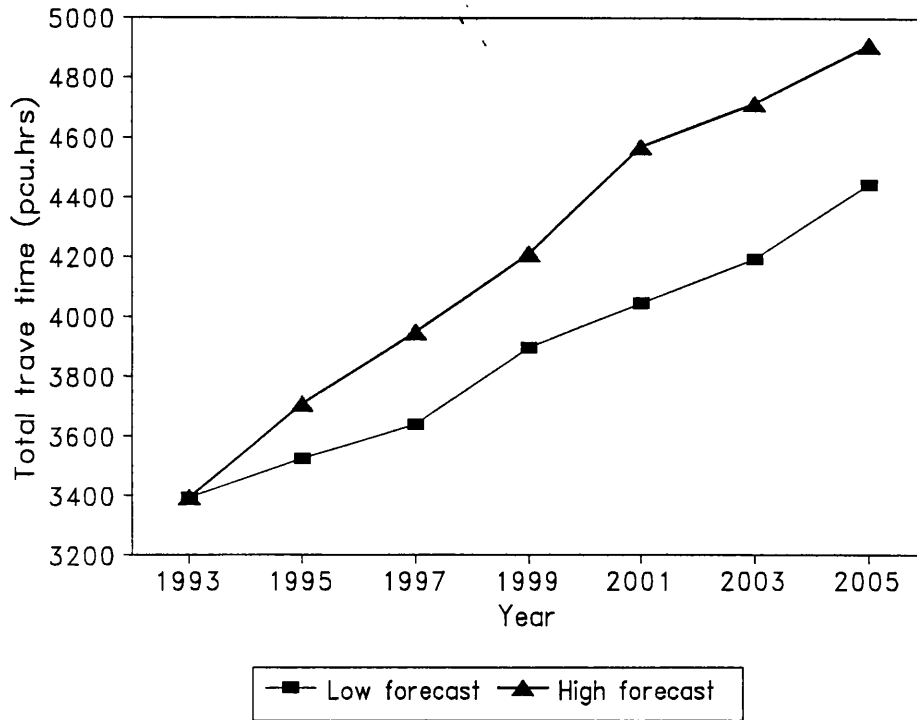
## 9.1 The impacts of the low and high traffic growth

The growth in travel demand in Britain has been a consistent trend for many years and it has involved increases in both the number and length of journeys undertaken (Department of Transport, 1993). Traffic levels are likely to become even higher in the future, considering the real threat of continuous growth in the demand for transport. Car travel plays the dominant role in traffic growth, which will cause more congestion and environmental problems.

This section analyses the impacts of traffic growth on the environment, considering both the low and high forecasts to the year 2005. The movement of stock in the fleet of private cars and heavy goods vehicles has been modelled by applying the factors of traffic growth produced by the NRTF (as shown in Figure 9.1) to traffic flows in all trips made in the case study network. The flow on each road of the network is assumed to have a constant proportion of diesel cars and petrol cars equipped with catalytics in the national fleet. The congestion effects from increasing growth in traffic volumes have been taken into account in the assignment process. The total fleet of buses has been assumed to have no growth, so that emissions from new registrations would return to the average level of previous years. This assumption of no increase in bus vehicle kilometres during the time scale of this study is consistent with the forecasts from the NRTF and Gover et al. (1994).

The application of the traffic model has provided a range of indicators related to future traffic circulation conditions. The changes in the overall characteristics of traffic in the Chester road network are shown in the following illustrations. The projected changes in travel times, distance travelled, average speed and fuel consumption are given in Figures 9.3, 9.4, 9.5 and 9.6, respectively.

**Figure 9.3: Total time spent in travelling**



The total time spent in travelling (Figure 9.3) is expected to increase along the projection, and this increase is more rapid in the forecast of high traffic growth. Such a trend is a result of the combined effect of more trips and congestion generated, which by increasing overall traffic levels in the future will substantially affect the travel times in the road network. This has important economic implications, since travel time constitutes a major component in the assessment of traffic schemes.

Figure 9.4 below shows that the total distance travelled in the network will also increase as a result of increased trip making and congestion levels. Congestion may induce journeys to become longer if congested routes become less attractive to travellers. The increase in travel distances is more distinct in the high growth scenario.

Figure 9.4: Total distance travelled

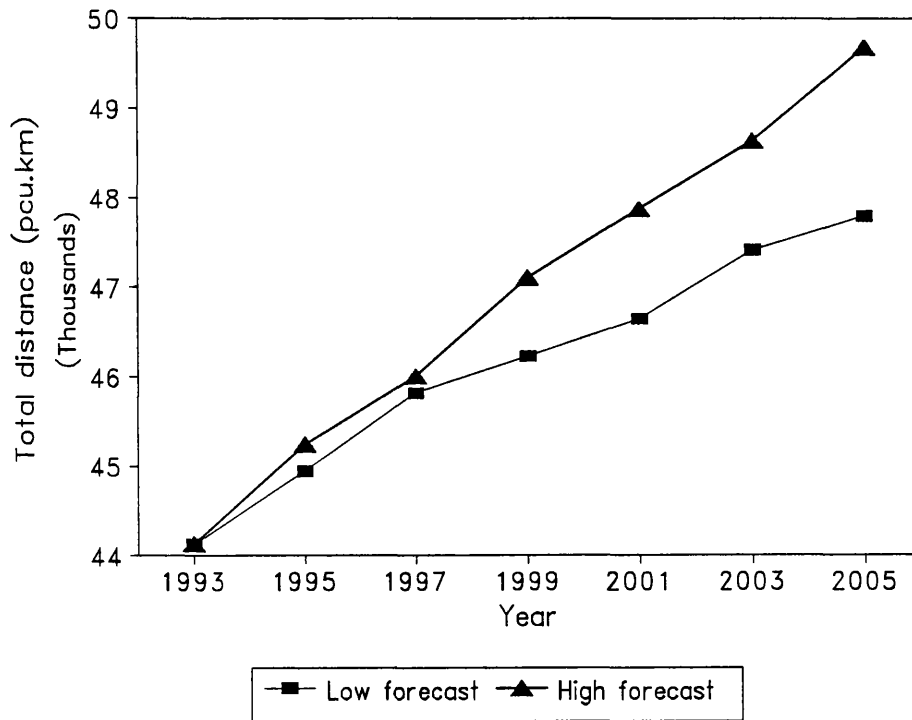
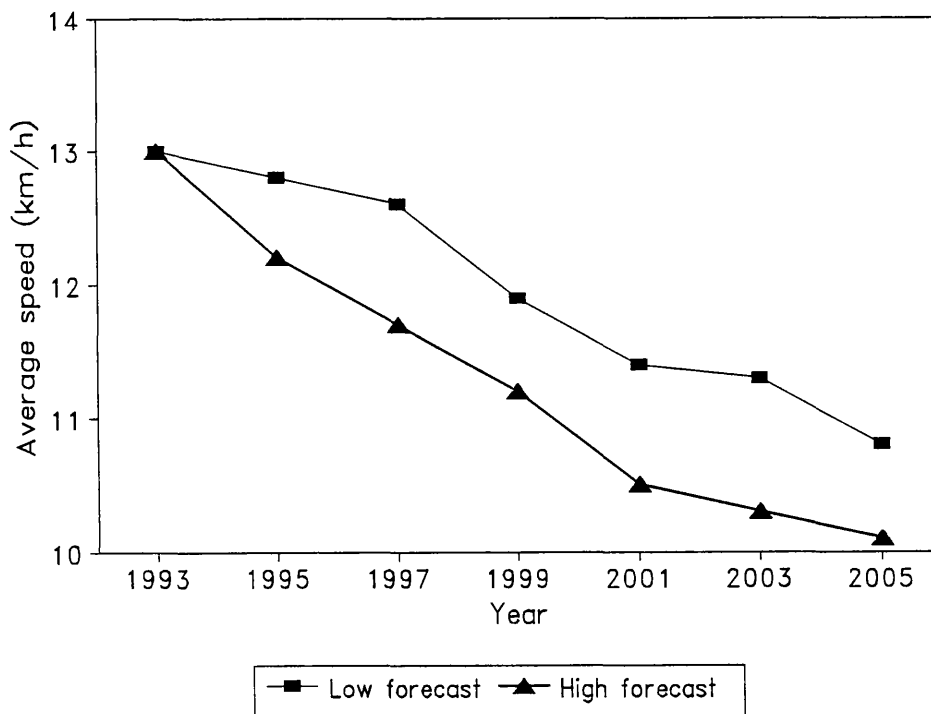
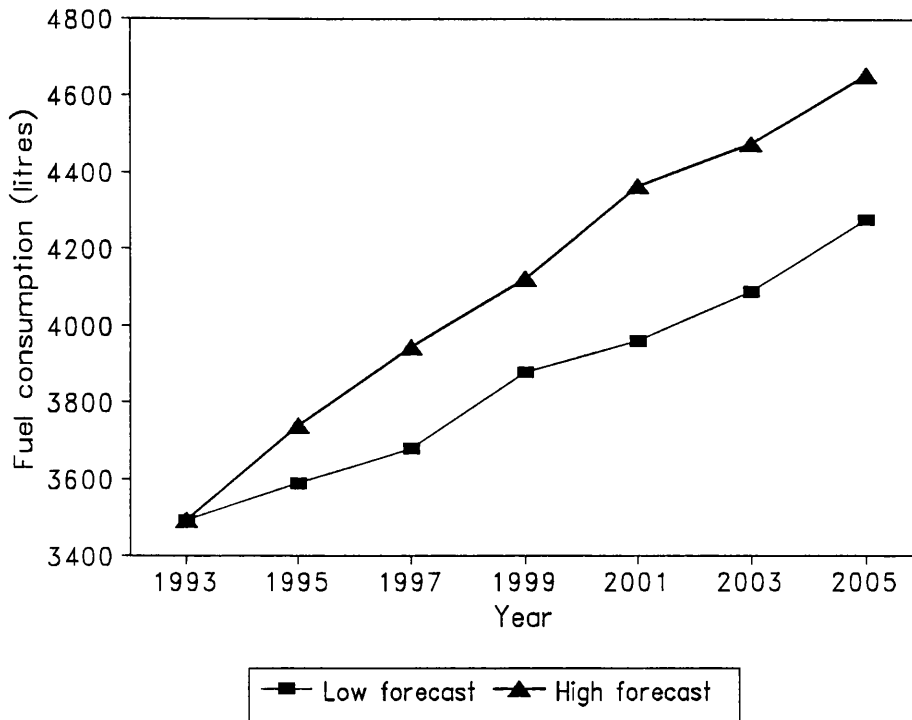


Figure 9.5: Overall average speed



**Figure 9.6: Total fuel consumption**

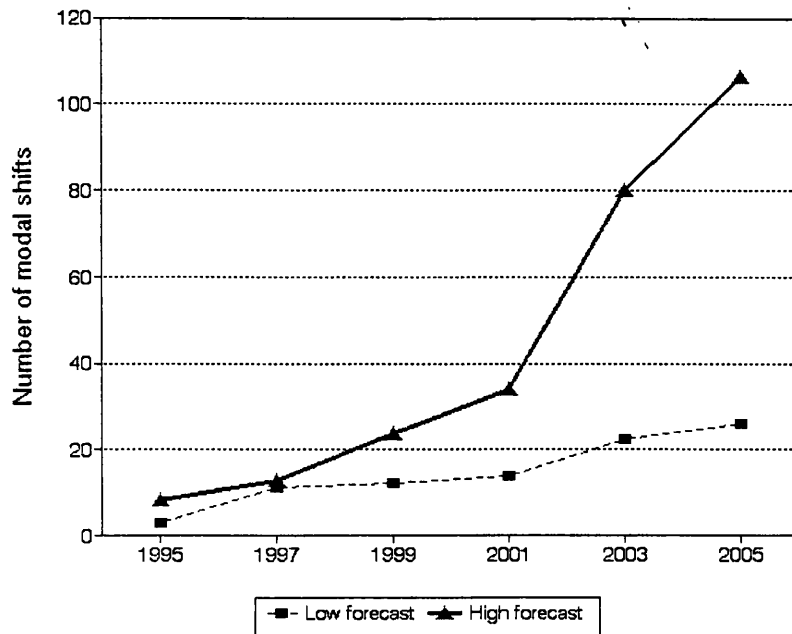


The average traffic speed in the network is projected to decrease from about 13 km/h in the 1993 scenario to about 11 km/h under the low traffic growth conditions or about 10 km/h under the high growth conditions (Figure 9.5). The continuous decrease in traffic speeds also reflects the increasing levels of traffic congestion.

The increasing levels of fuel consumption are also relevant in economic assessments. Figure 9.6 indicates that the total consumption of fuel in the network increases sharply, particularly in the forecast of high traffic growth. It is important to bear in mind, however, that future improvements in energy efficiency have not been accounted for in the analysis and therefore the total fuel consumption is likely to have been overestimated, particularly at the end of the projection.

The application of the modal split model (Section 6.1.2) has indicated the extent to which travellers choose to switch from private to public transport as a result of increasing traffic levels in the study network. Figure 9.7 shows a projection of the number of modal shifts from private to public transport due to traffic growth.

**Figure 9.7: Number of modal shifts to public transport due to traffic growth**



Increases in the demand for transport generate increasing travelling costs. However, the elasticity of modal choice in relation to transport demand is generally low (this issue will be dealt with in more detail in Section 9.3). As expected, the changes in modal split have been more substantial in the high traffic growth scenario (in which up to about 105 out of a total of 19,500 travellers have chosen to change modes at the end of the projection) than under the low growth consideration (in which only up to 23 travellers decided to give up their cars in favour of the public transport).

The predictions of the environmental impacts produced by the SPITE in relation to the low and high traffic growth forecasts have been based on the estimated traffic flow conditions and future vehicle emission trends. As far as the future vehicle market is concerned, a gradual increase in the participation of private vehicles using diesel as fuel and petrol cars fitted with catalytic converters has been considered in the projection. The scenario presented by 'Case 2' (shown in Figure 9.2) is considered in the analysis.

The compliance with the emission regulation limits of new vehicles both petrol and diesel entering the market after 1996 is also taken into consideration. All other vehicles in the fleet which are not under the new regulation are assumed to emit the 1993 representative rates

throughout the projection. As CO<sub>2</sub> emissions are not regulated, it is assumed that there will be no changes in the emission of this pollutant.

The yearly average changes in total emissions, air pollution concentrations and noise levels have been estimated by the SPITE in all the locations in the central area of Chester. The SPITE has been run for each scenario, according to the low and high traffic growth forecasts.

The yearly changes in the total emissions produced in Chester are illustrated in the following figures. Figures 9.8, 9.9, 9.10, 9.11 and 9.12 show the absolute changes in the emissions of CO, HC, NO<sub>x</sub>, CO<sub>2</sub> and PM, respectively.

**Figure 9.8: Changes in CO emissions due to traffic growth**

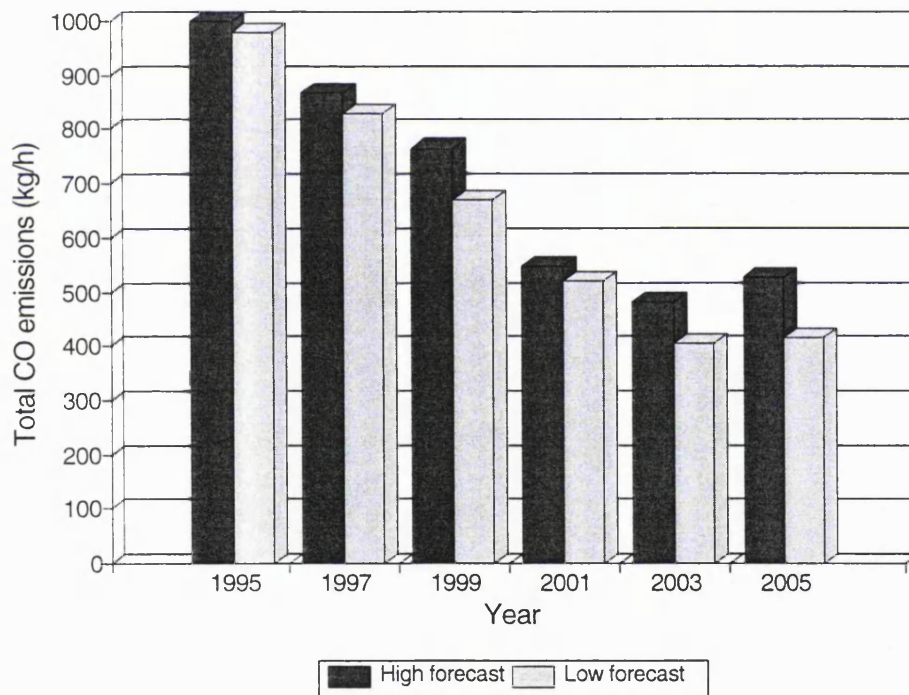




Figure 9.9: Changes in HC emissions due to traffic growth

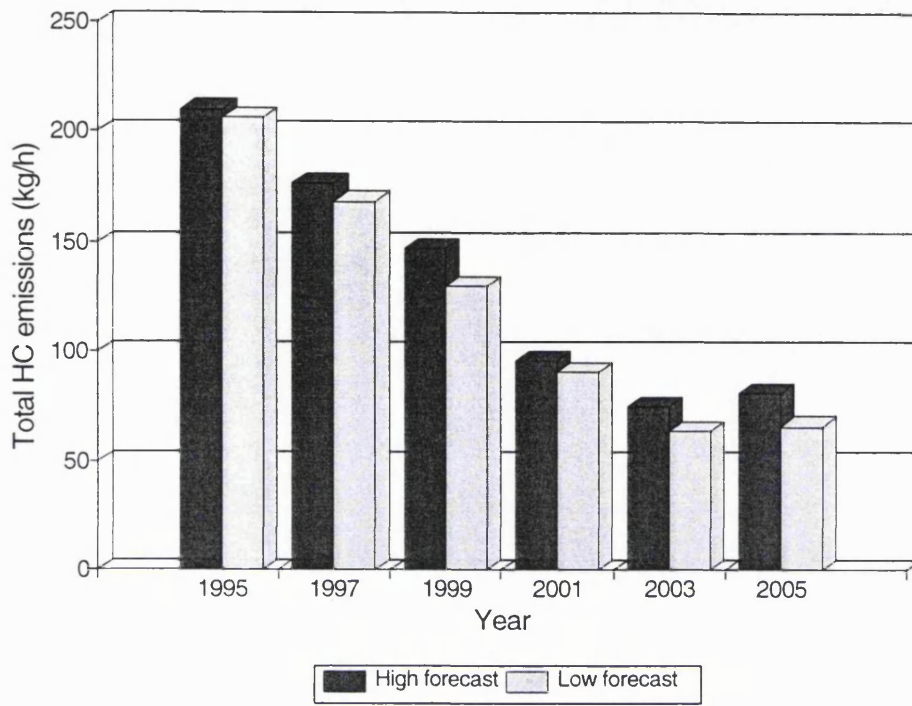


Figure 9.10: Changes in NO<sub>x</sub> emissions due to traffic growth

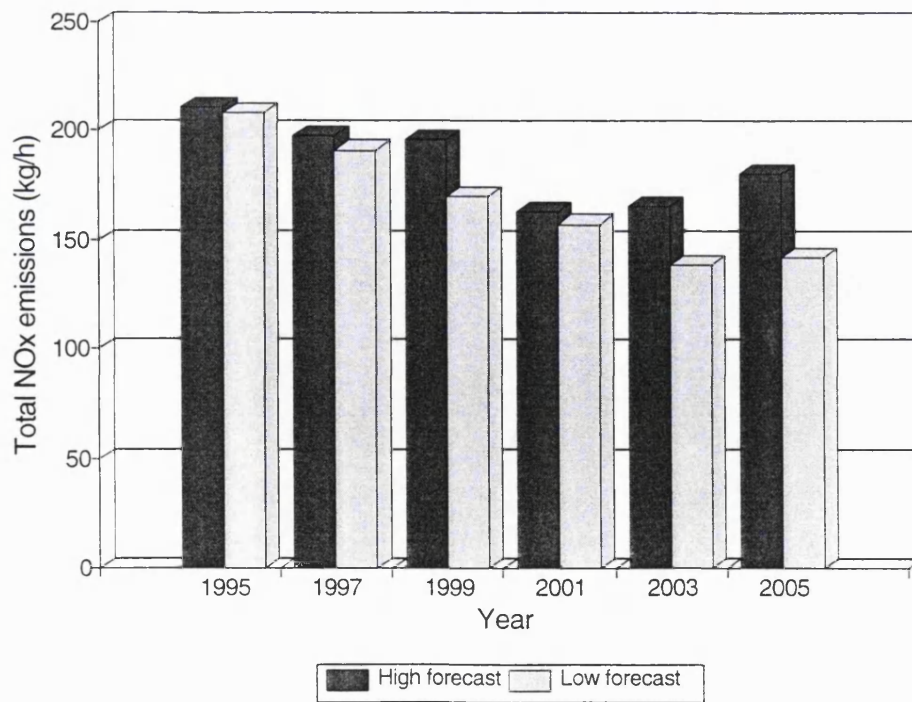


Figure 9.11: Changes in CO<sub>2</sub> emissions due to traffic growth

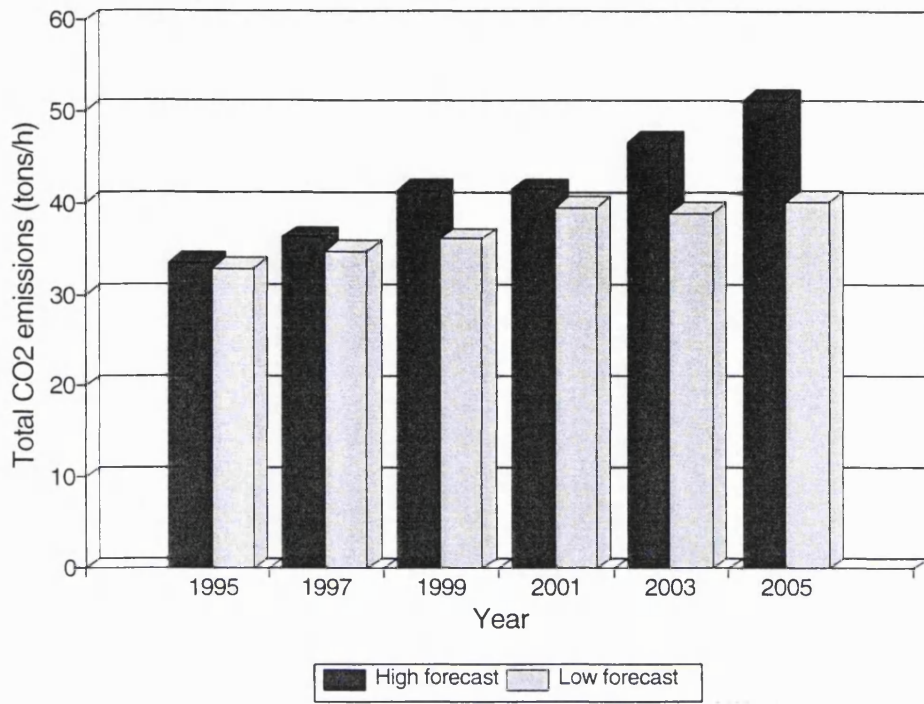
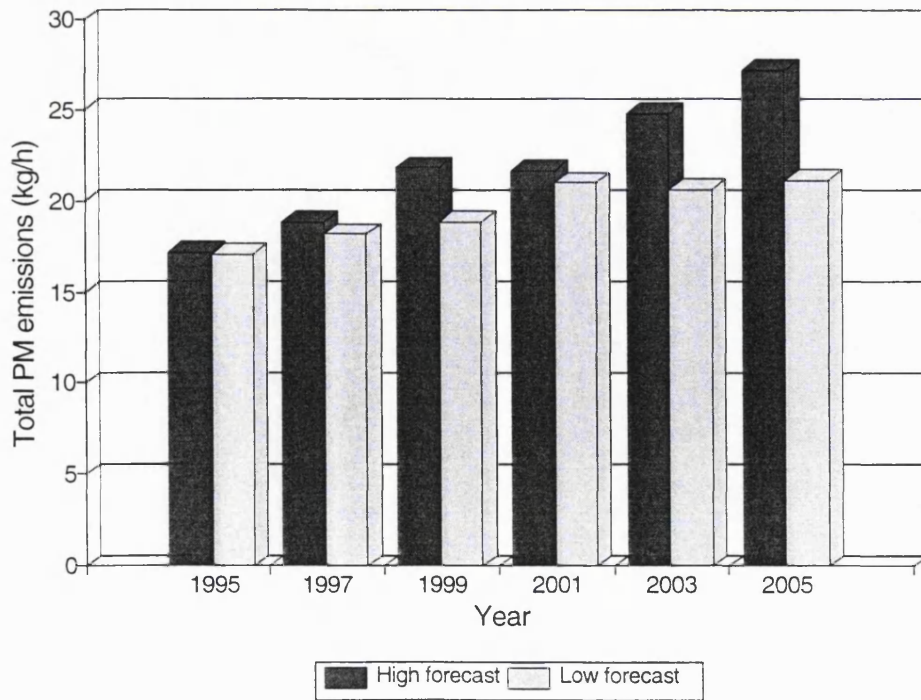


Figure 9.12: Changes in PM emissions due to traffic growth



The total emission levels of CO, HC and NO<sub>x</sub> (Figures 9.8, 9.9 and 9.10) tend to decrease over the projection. This decrease is due to the growing penetration of cars fitted with catalytic converters, which emit far smaller amounts of these pollutants (Section 3.3). Thus, the effect of the introduction of catalytic converters to these vehicle emissions has proven to outweigh the effect of traffic growth. The differences in total pollutant emissions between the low and high forecast scenarios are in general small. The total emission levels of these pollutants tend to stabilise after the year 2003 under the low forecast scenario. However, considering the high traffic growth forecast, the effect of traffic growth after 2003 starts to outweigh the effects of the emission improvements provided by the penetration of petrol cars fitted with catalyst devices. This suggests that further technological improvements will be necessary to stabilise the emissions of these pollutants in the longer term.

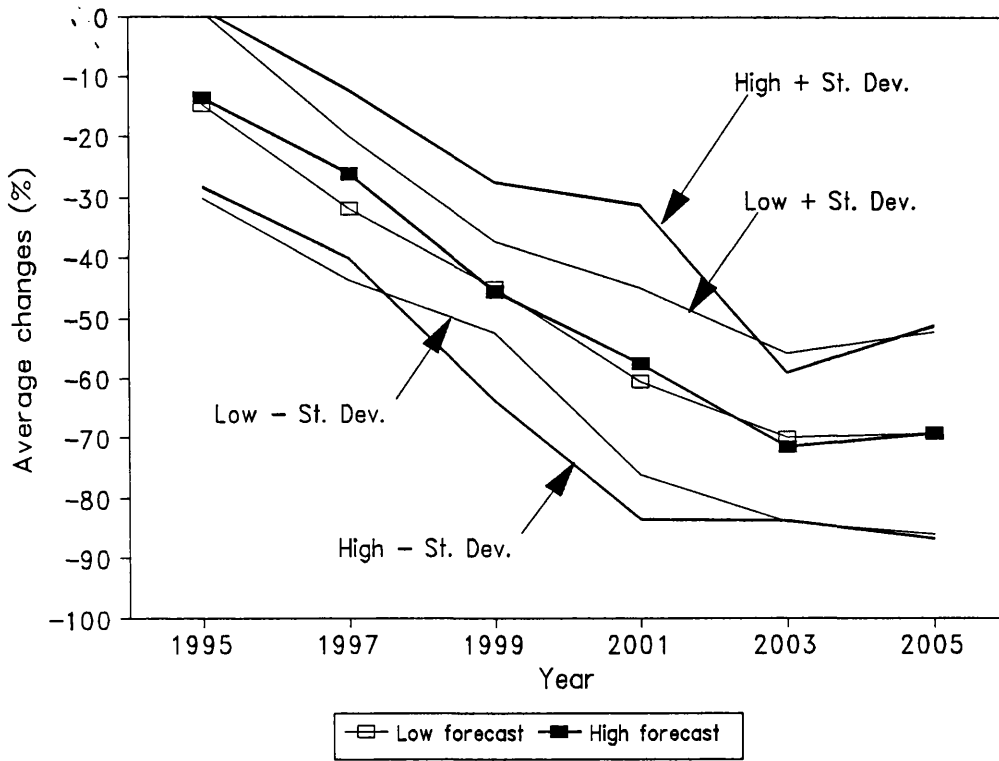
The magnitude of CO<sub>2</sub> emissions is substantially higher than the other emissions (notice in Figure 9.11 that its units are given in tons per hour rather than in kilograms per hour) and the total emission levels of this pollutant rise constantly. Such increase is a result of the combined effect of traffic growth and the introduction of catalysts, which provoke an increase of the emission level of this air pollutant (Section 3.3). The difference of the emission level of this pollutant between the low and high forecasts is significant. At the end of the projection, the total emissions reach almost 40 tons/h under the low forecast scenario and over 50 tons/h in the high forecast consideration. Since future emissions of CO<sub>2</sub> constitute a major threat in changing the world's climate (Department of the Environment, 1992), the results of the SPITE suggest that the extent to which global levels of traffic grow will have important effects on the global environmental degradation.

The total emissions of PM also rise constantly throughout the projection (Figure 9.12). The increase in PM emissions is due to the effect of traffic growth, particularly the growth of heavy goods vehicles, and to the increase in diesel cars in the fleet of light vehicles. According to the results of the emission model in the SPITE, the decline in PM emissions as result of tighter vehicle emission standards in the future is offset by the increasing traffic volumes, as forecast by the Quality of Urban Air Review Group (1993b). Catalytic converters do not affect the emissions of this pollutant. The differences between the low and high forecasts become significant at the end of the projection, when more diesel vehicles (both cars and lorries) will be running on the roads. The increase in future levels of PM emissions can be critical for the local environmental quality.

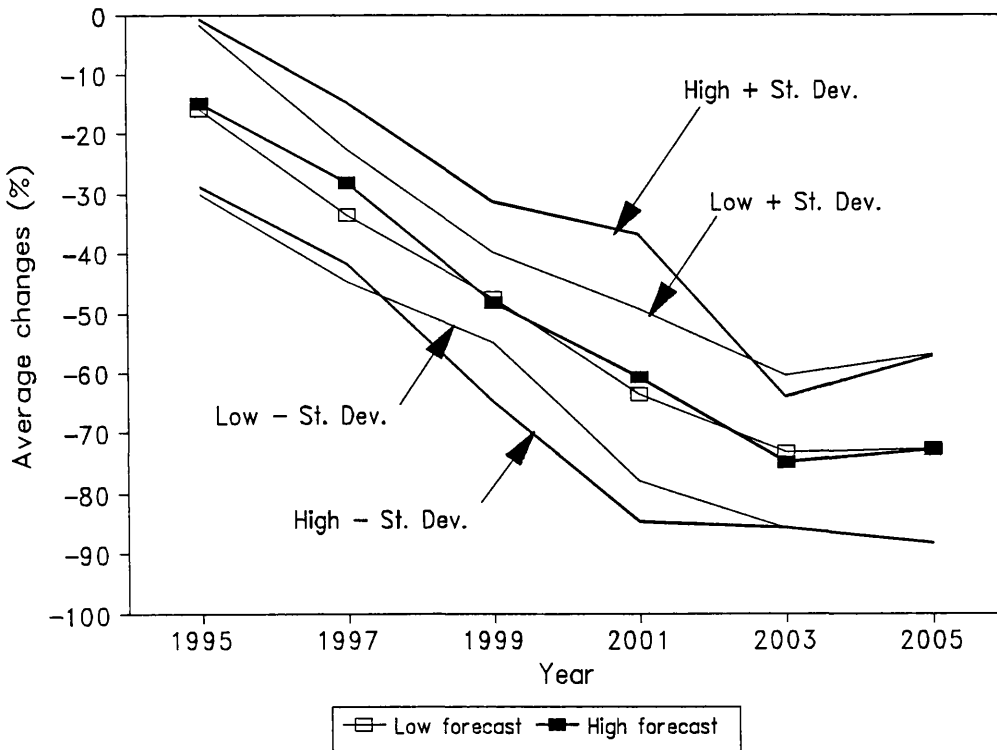
The future emission trends shown above are compatible with the changes in road transport emissions in the U.K., as illustrated in a number of publications, such as: Holman et al. (1989), Fergusson et al. (1989), European Conference of Ministers of Transport (1990), TEST (1991), Quality of Urban Air Review Group (1993a), Whitelegg (1993), Wade et al. (1993), Centre for Exploitation of Science and Technology (1993), Holman et al. (1993), Gover et al. (1994), Department of the Environment (1994), Royal Commission on Environmental Pollution (1994) and Local Transport Today (1995a). According to the Centre for Exploitation of Science and Technology (1993), emissions have been projected to fall significantly before levelling off around 2010 as traffic growth begins to offset improvements in vehicle emissions. The extent to which emissions would resume an upward trend will depend on the magnitude of actual traffic growth and future technological improvements.

The levels of local environmental degradation as a result of traffic growth have been estimated by the application of the SPITE (air pollution concentrations in Section 6.2.2 and noise levels in Section 6.3). The average changes in pollutant concentrations in the central area are given below in Figures 9.13 to 9.17, whereas the changes in noise levels are shown in Figure 9.18. These figures present the average differences in environmental degradation between the low and high forecasts, and also provide a measure of scatter from the averages. The thicker lines along the average trends represent the average value plus and minus the standard deviation for the estimates of the high growth scenario. The thin lines show the average changes plus and minus the standard deviation for the low forecast.

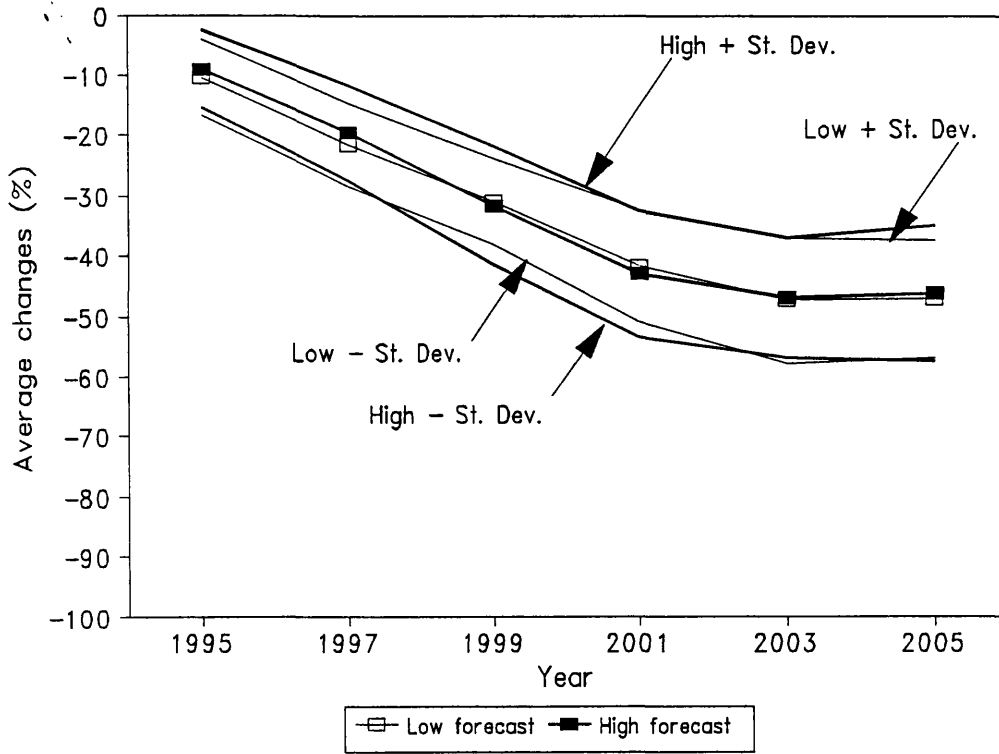
**Figure 9.13: Average changes in CO concentrations due to traffic growth**



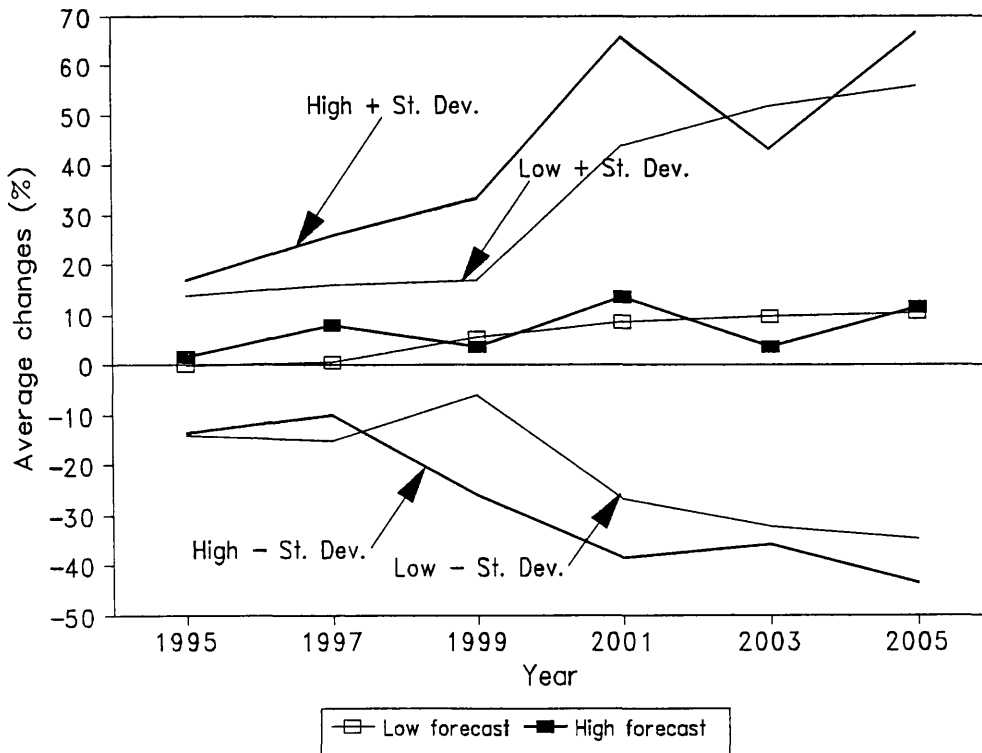
**Figure 9.14: Average changes in HC concentrations due to traffic growth**



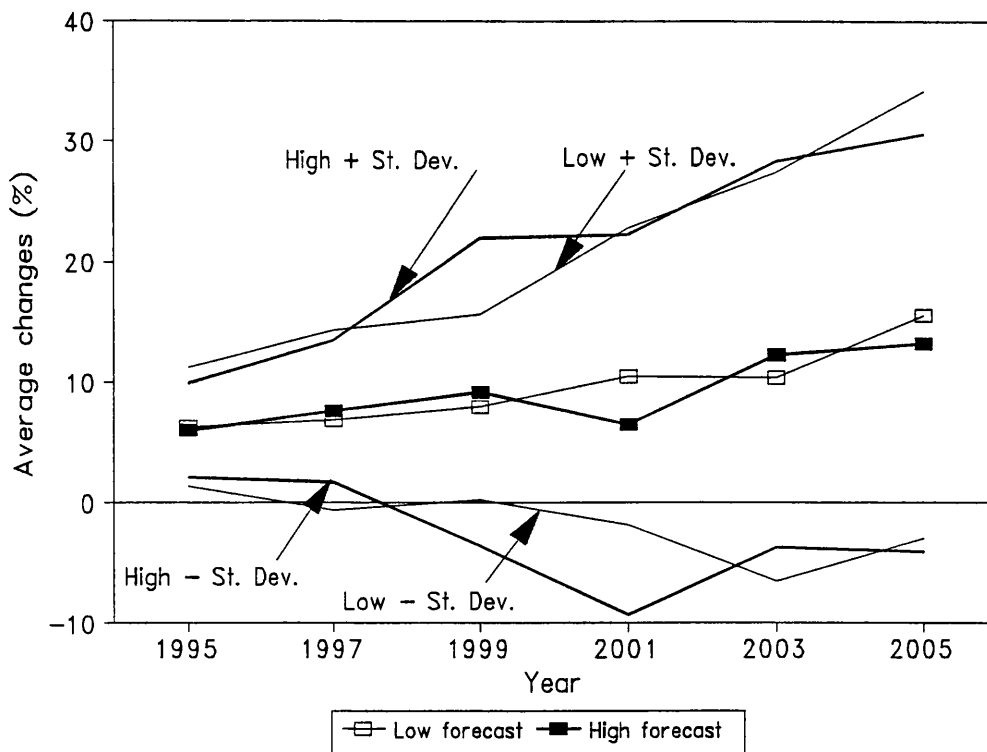
**Figure 9.15: Average changes in NO<sub>x</sub> concentrations due to traffic growth**



**Figure 9.16: Average changes in CO<sub>2</sub> concentrations due to traffic growth**



**Figure 9.17: Average changes in PM concentrations due to traffic growth**



Figures 9.13, 9.14 and 9.15 indicate that the air quality in relation to the pollutants CO, HC and NO<sub>x</sub> is already improving and tends to get gradually better in the future, due to the reduction in the emissions of these pollutants promoted by the compulsory introduction of catalytic converters in petrol cars. Reductions in the average levels of CO and HC concentrations reach about 70% at the end of the projection, whereas decreases of NO<sub>x</sub> levels reach up to about 45%. Such reductions outweigh the effect of traffic growth in the medium term, but in the longer term this tendency is likely to be reversed. This is shown by the slight inclination of the curves in the end of the projections and is compatible with estimations of the reduction in the emission levels of these pollutants, as reported earlier in this section. Thus, improvements in air quality are likely to cease around the year 2003, when the effect of traffic growth starts to offset the benefits from the new technology. There is only a small difference in the projections of average changes in air pollution concentrations from the low to the high traffic growth estimates. In most circumstances, however, the average reductions are lower in the high forecast scenario due to the worse traffic circulation conditions. It can also be noticed in these figures that the lines representing the 'average plus and minus standard deviation' in relation to the high forecast (thicker lines) are further from the average

than those representing the 'average plus and minus standard deviation' in relation to the low forecast (thin lines). Since the standard deviation represents a measure of scatter, this indicates that there are some higher individual changes in the high forecast scenario. In some locations the air quality has become much worse than in the low growth scenario, due to isolated effects of traffic congestion. These effects, however, are dissipated when the average change is derived for all the locations in the central area.

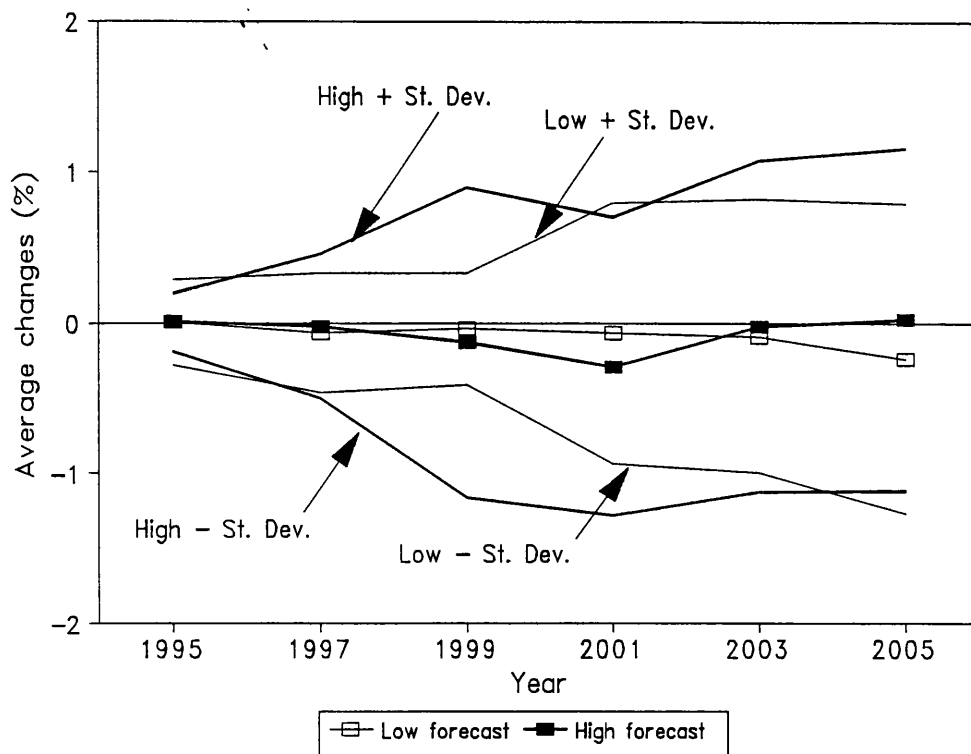
The average concentrations of CO<sub>2</sub> tend to increase up to about 10% of the original levels (Figure 9.16), due to the continuous increase in the emission of this pollutant. As reported earlier in this section, this increase is due to the combined effect of traffic growth and introduction of catalysts. It is important to note that increases in CO<sub>2</sub> emissions have accounted exclusively for the contribution from traffic sources, which produce only a fifth of the total emissions. Any significant changes in the emission of this pollutant from other non-transport sources have not been considered and could change the magnitude of the projections. The high growth scenario presents, in general, higher increases and higher standard deviations, which indicate the worse air quality under such conditions. In fact, the maximum increase in CO<sub>2</sub> concentration within the locations in the central area under the high forecast conditions exceeds 210%.

The air quality in respect to ambient levels of PM also tends to deteriorate along the projection (Figure 9.17), which is a consequence of the definite trend of continuous growth on the sales of diesel vehicles. Average concentration levels of PM increase up to about 15%, but the maximum increase within the locations in the central area exceeds 100%. The 1996 emission limit regulations for PM emissions are very strict. Had these limits not been considered in the simulation of future air quality conditions, the concentration levels of PM would have increased by a much larger amount (Chiquetto and Mackett, 1994a and 1994b, Chiquetto, 1995). The high forecast scenario indicate generally higher increases in pollution concentration levels as well as higher patterns of scatter, which also reflects worse environmental quality. Such increases in PM levels are only likely to be realised if diesel penetration of the passenger car fleet does increase at the projected rates, and PM emissions are not reduced drastically by the development of new technologies.

Next, the changes in noise levels as a result of traffic growth are analysed. Figure 9.18 shows the average and standard deviation of expected changes in noise levels for the locations in the central area.



**Figure 9.18: Changes in noise levels due to traffic growth**



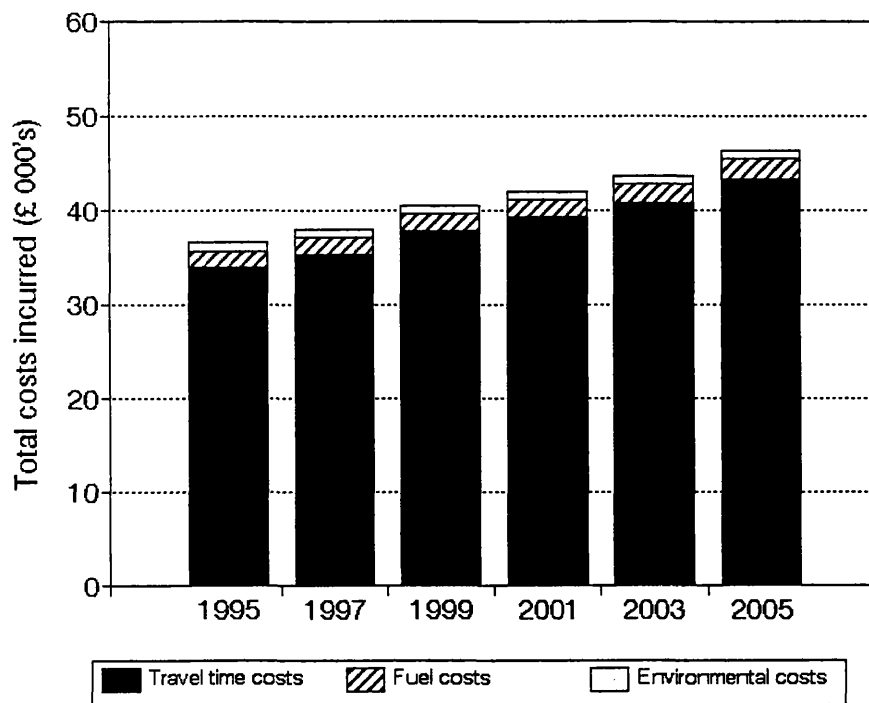
It has been reported that  $L_{10}$  noise levels are little sensitive to changes in traffic flows, and it has been suggested that transport policies are, in general, not very effective in reducing noise levels. Hence, the trends in Figure 9.18 indicate that changes in noise levels in relation to the magnitude of traffic growth are very small. Increases in traffic flows would affect noise levels only marginally, in a logarithmic scale (Section 4.3). The differences between the low and high traffic growth forecasts are virtually negligible.

The maximum increase found in noise levels among the various locations in the central area was about 4%. In absolute terms, this percentage increase represents about 3 dB(A), which is indeed relevant in the light of the nature of the weak relationship between traffic flows and noise. Since some people may notice when noise changes are as small as 1 dB(A), as reported in Section 4.3, modest changes in traffic flows can have important consequences for noise appraisal, particularly where thresholds, such as the noise level above which compensation may be requested, are crossed (Department of Transport, 1994d).

The application of the EAF has provided estimates of the total costs incurred in the network

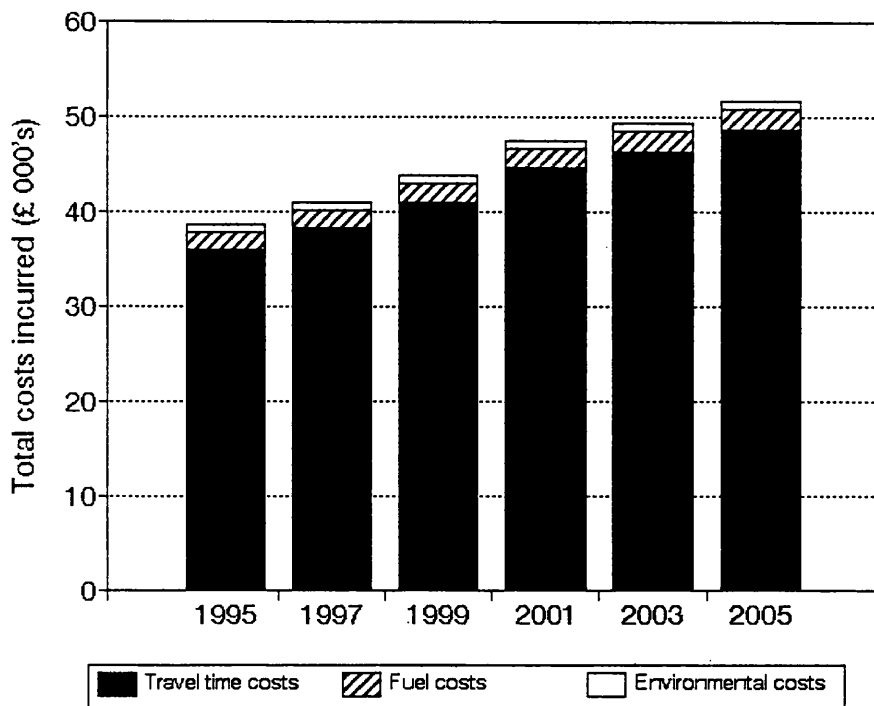
along the projection, according to the low and high traffic growth forecasts. As reported in Chapter 7, the factors considered on the estimation of these costs are: travel time, fuel consumption and environmental damage. Figures 9.19 and 9.20 show the estimates of the total costs incurred in Chester for the low and high scenario, respectively. These costs have been based on 1993 prices.

**Figure 9.19: Total costs incurred in the low traffic growth scenario**



As concluded in the monetary evaluations carried out in Chapter 8, the costs related to the value of time spent on travelling are much higher than the other costs comprised in the EAF. The expenses with fuel consumption and the costs from the environmental degradation represent only a portion of the total costs incurred. The percentage increase of travel time costs along the projection is also higher than the percentage increase of the other costs, what suggests that the importance of the costs associated with travel time in the overall economic assessment grows as traffic becomes more congested.

Figure 9.20: Total costs incurred in the high traffic growth scenario



The costs incurred in each category considered in the EAF increase constantly along the projection, as a result of the higher number of trips being made, and therefore more time spent in travelling, longer distances travelled and more environmental degradation produced. This is in agreement with the results of the traffic model, which indicate that travel times, distances and fuel consumption increase with traffic growth (Figures 9.3, 9.4 and 9.6, respectively).

The differences between the total costs produced in the low and high scenarios are substantial and such contrast is emphasised through the projection. According to the estimates shown above, the total costs incurred in Chester within the simulation period (one hour) increase from about £ 36,500 to about £ 46,300 from 1995 to 2005, under the low forecast scenario (Figure 9.19), and from about £ 38,500 to about £ 51,800 (Figure 9.20), considering the high traffic growth. This finding suggests that the magnitude of costs produced varies considerably with traffic levels.

In conclusion, the effect of low and high traffic growth is unlikely to affect significantly the overall pollution concentration levels, but it will probably cause major impacts at particular

locations. The effect of traffic growth is not as important as the effect of the changes in car market trends (this will be analysed in Section 9.2), which will alter quite substantially the patterns of exhaust emissions. In terms of the total emissions of CO<sub>2</sub>, the effect of the magnitude of traffic growth can play an important role, particularly at the end of the projection. Noise levels are very little affected by the effect of traffic growth.

Congestion itself induces the spread of traffic and, as a consequence, the dissemination of the environmental impacts produced. Thus, authorities should concentrate their efforts in coercing vehicle manufacturers for further developing the existing technologies, creating new devices for controlling the emission levels of noise and air pollutants, or as suggested by Holman et al. (1991), considering alternative transport fuels. Initiatives such as those are most likely to be effective in reducing the environmental degradation produced by road traffic in the years to come.

However, as far as the overall costs incurred in the future are concerned, the magnitude of future traffic growth levels will have important economic implications for the community not only in Chester, but also in other urban areas where similar levels of traffic growth are expected.

## 9.2 Changes in the vehicle market

It has been shown that the type of fuel used by cars and the fitting of catalytic devices are crucial for the amount of emission produced (Sections 3.3 and 6.2.1). Thus, the changing composition of the vehicle fleet is of great importance to future trends in road transport emissions. In particular, the growth in the sale of diesel-engined passenger cars has important implications for the emissions of PM, and the increasing number of petrol-engined cars which are fitted with catalytic devices has a significant influence on the emissions of the other pollutants considered.

This section analyses the environmental consequences in the study area from changes in emission rates produced by vehicles in the future. Changes in the market penetration of cars are assumed not to affect the levels of traffic noise.

Air pollution can be reduced by the installation of prevention devices at the source, which is considered an ideal environmental policy. Catalytic converters are the most efficient devices for reducing vehicle emissions. Vehicle exhaust emissions are subject to regulations and catalytic converters play an important role on meeting them.

A catalytic converter looks like a cylindrical bulge which is enclosed in stainless steel and installed in the conventional exhaust pipe. The internal structure is honeycombed which increases the surface area to facilitate the conversion process. It is coated with precious metals, such as platinum, rhodium and palladium, the catalysts. The so-called three-way devices catalyse three of the main traffic pollutants, which are CO, HC and NO<sub>x</sub>, into CO<sub>2</sub>, water vapour and nitrogen. Modern converters do not reduce engine performance and last longer than older ones (Gould, 1989).

Since January 1993, all new passenger cars have been compulsorily fitted with closed loop three-way catalytic converters (Eggleston, 1992). The level of efficiency of new catalysts is quite remarkable. New cars fully warmed up produce so little pollution that reducing miles of travel seems ineffective in reducing air pollution (Wachs, 1993). Under specific conditions, a controlled three-way catalyst can reduce emissions of CO, HC and NO<sub>x</sub> by 80-95% (London Planning Advisory Committee, 1990; Jost et al., 1992; Des Courtils et al., 1993), but their efficiency for CO<sub>2</sub> reductions is either negligible or negative. More detailed research by the TRL reveals the average efficiencies during hot start tests in petrol vehicles after about

80,000 kilometres driven on public roads in Britain (Pearce and Davies, 1990). The three-way catalyst have presented a 69% of efficiency for NO<sub>x</sub> and 84% for HC and CO.

The average efficiencies of catalytic converters fitted in petrol engines are given in Table 9.2, for different pollutants and driving patterns. Such figures were taken from data produced from bench test surveys on in-service petrol vehicles, by Tickford Limited (1994) (see Table 6.1), in accordance with the European driving cycle (Figure 6.2).

**Table 9.2: The efficiency of catalytic converters**

| Driving pattern | Efficiency (%) |      |                 |                 |
|-----------------|----------------|------|-----------------|-----------------|
|                 | CO             | HC   | NO <sub>x</sub> | CO <sub>2</sub> |
| Acceleration    | 63.3           | 78.8 | 68.0            | - 7.2           |
| Deceleration    | 76.1           | 87.1 | 82.2            | -15.1           |
| Cruise          | 61.7           | 79.9 | 77.6            | - 6.2           |
| Idle            | 74.6           | 75.0 | 83.3            | - 5.8           |

In the above table, efficiency is defined as the percentage reduction in emission rates from the implementation of catalysts. The negative signs imply an increase in emissions. Considering that these figures were taken from in-service conditions, the overall level of device efficiency is very significant in relation to CO, HC and NO<sub>x</sub> emissions. However, the emission rates of CO<sub>2</sub> are increased by the implementation of catalytic devices by about 6 to 15%.

The introduction of catalytic converters is a slow business, as yearly substitution of vehicles covers only a small percentage of the total fleet. Traffic emissions will substantially decrease only when a great deal of pre-1993 cars have been replaced by new ones. The Society of Motor Manufacturers and Traders (1994) has estimated that by the end of the century the proportion of petrol passenger cars in the national fleet equipped with catalytic converters is likely to rise to 70-80%.

The other significant change expected in the future car market, as far as exhaust emissions are concerned, is the expansion of diesel-powered cars. In recent years, diesel cars have become increasingly popular and the gap with petrol cars has narrowed in terms of

performance, refinement, economy, noise levels and purchase price (Quality of Urban Air Review Group, 1993b). In fact, diesel engines present lower fuel costs and are more efficient than petrol engines (Gover et al., 1994). Diesel cars consume about 25% less fuel by volume than equivalent petrol cars. Diesel cars have also been promoted as cleaner than petrol by the motor industry, since they emit less CO<sub>2</sub>, CO and HC than equivalent petrol cars with catalysts. The concern over the contribution of road transport to global warming has encouraged the motor industry to present diesel cars as a more benign choice for environmentally conscious drivers. The total global warming potential of diesel cars is 20% lower than the equivalent catalytic converter equipped petrol car when all greenhouse gases are considered (Wade et al., 1993). Diesel engines are also generally more durable and are likely to be less prone to deterioration (Gover et al., 1994). The sales of diesel cars have been rising considerably in the U.K. over the last 5 years and they have come to account for over 20% of the new cars sold in the U.K. by the end of 1993 (Quality of Urban Air Review Group, 1993b and Department of Transport, 1994a). In France, nearly half of car sales are diesel and in Britain there is a tendency for a further increase in the participation of diesel cars in the fleet of light vehicles. In fact, their participation has already almost tripled (increased from 2.2 to 5.9%) between 1988 and 1993 (Department of Transport, 1989b; Department of Transport, 1994a).

The Quality of Urban Air Review Group (1993b) estimates improvements in CO and HC and deterioration in respect to NO<sub>x</sub> and PM with increased diesel market share. Thus, it is not simple to decide which car type is more polluting, since petrol and diesel cars emit different quantities of each pollutant. One would have to judge whether a certain amount of one pollutant is environmentally better than a different amount of another pollutant. Besides, it is difficult to match petrol and diesel vehicles in terms of engine capacity, power and other performance parameters.

Yearly scenarios from 1993 to 2005 were created by the application of the emission model in the SPITE, in order to illustrate the trends in total emissions from the expected changes in the patterns of vehicle emissions as a result of future market car trends. The effect of traffic growth has been considered in this analysis. The low forecast has been assumed in the projection, since this is the most likely considering the current statistics and tendencies. Like in the projections of traffic growth (Section 9.1), all new diesel cars and new petrol cars fitted with catalytic converters in the market after 1996 are assumed to comply with the emission regulation. The following figures illustrate the extent to which the penetration of diesels and

catalytic converters changes future levels of total emissions. Figures 9.21, 9.22, 9.23, 9.24 and 9.25 show the changes in CO, HC, NO<sub>x</sub>, CO<sub>2</sub> and PM, respectively, as a function of the three cases related to future market penetration of cars, as presented in Figure 9.2.

**Figure 9.21: Changes in CO emissions due to future car market trends**

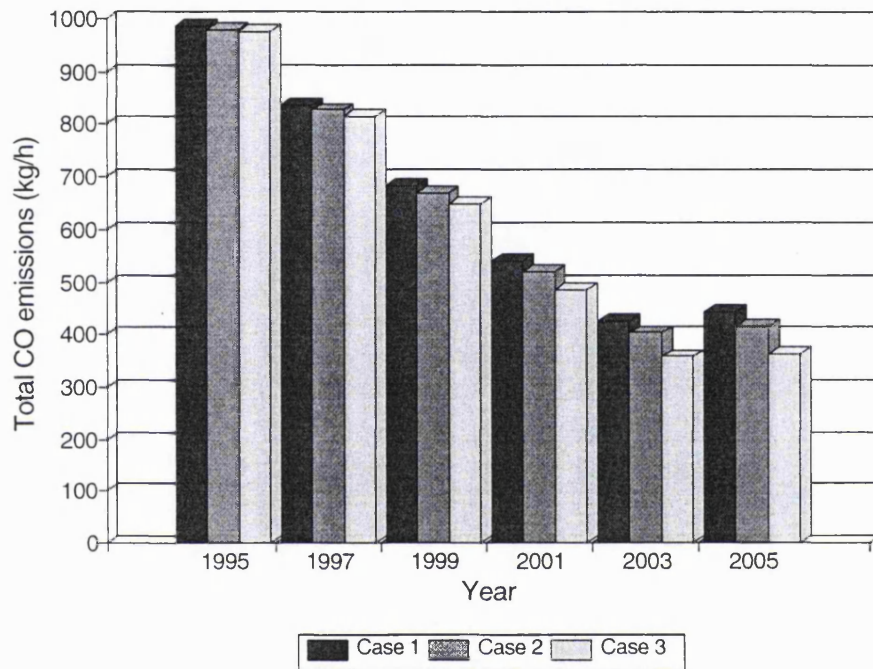




Figure 9.22: Changes in HC emissions due to future car market trends

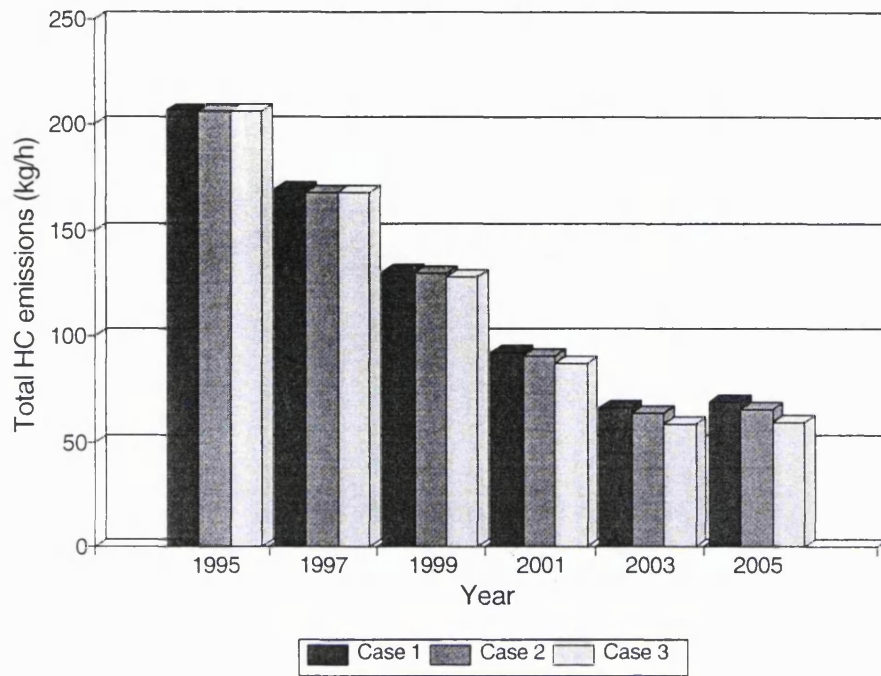


Figure 9.23: Changes in NO<sub>x</sub> emissions due to future car market trends

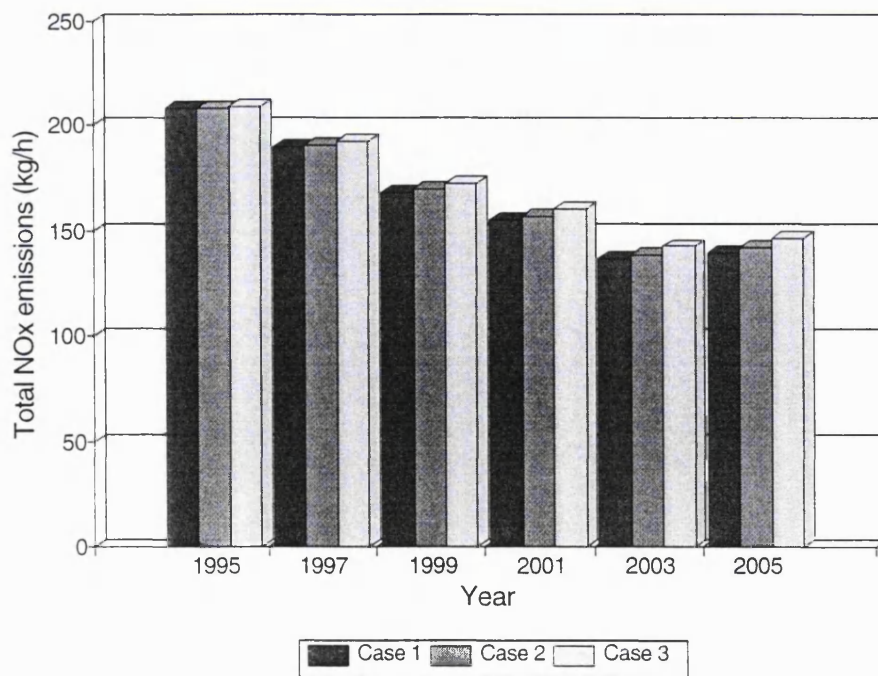


Figure 9.24: Changes in CO<sub>2</sub> emissions due to future car market trends

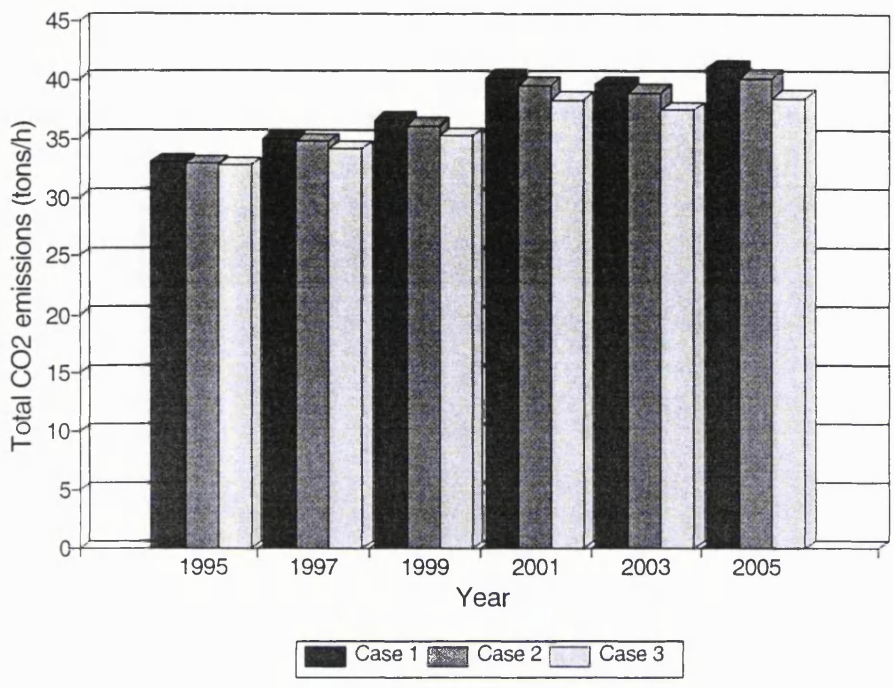
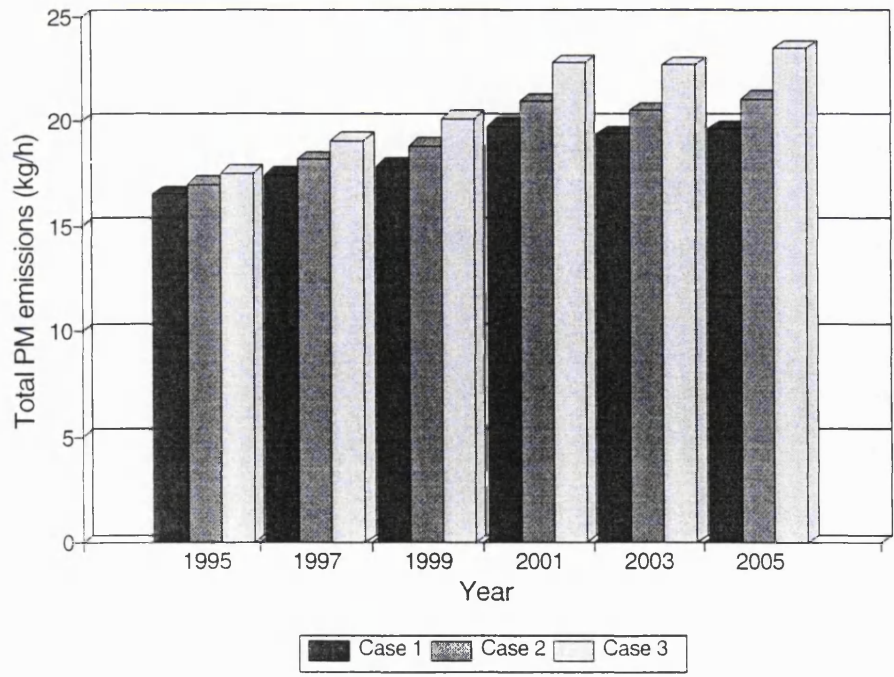


Figure 9.25: Changes in PM emissions due to future car market trends



As in the case of traffic growth (Section 9.1), Figures 9.21, 9.22 and 9.23 have shown that the total emissions of CO, HC and NO<sub>x</sub> decrease constantly along the projection for all cases analysed. Such decreases are a result of the penetration of petrol cars fitted with catalytic converters, which emit far less amounts of these pollutants. The reductions tend to stabilise after 2003, when the effect of traffic growth starts to outweigh the benefits from the new technology. The differences in total pollutant emissions produced amongst all the cases are in general small. This is due to the introduction of the 1996 regulations which tends to level off the emission from various vehicle categories. Previous forecasts of future emissions which did not take into account future emission constraints found much higher differences among these cases (Chiquetto and Mackett, 1994a and 1994b, Chiquetto, 1995).

The total emissions of CO have decreased from about 1100 kg/h in 1993 to only around 400 kg/h at the end of the projection. HC emissions were reduced from about 240 to 60 kg/h. The emissions of these two pollutants are higher in Case 1 and lower in Case 3 along the projection. This is due to the larger number of petrol cars in Case 1 and lower in Case 3 (the future emission limits of these pollutants are higher from petrol than from diesel cars). The reductions in NO<sub>x</sub> emissions were not as substantial as the reductions in the other two pollutants. The emissions of NO<sub>x</sub> decreased from about 240 to 140 kg/h within the period of analysis from 1993 to 2005. The emission levels of this pollutant is higher in Case 3 and lower in Case 1, contrary to the trends of the other two pollutants, since diesel cars emit more NO<sub>x</sub> than petrol cars and Case 3 assumes a higher proportion of diesels.

As shown in Figure 9.24, the total emissions of CO<sub>2</sub> rise constantly due to the combined effect of the introduction of catalysts and traffic growth. CO<sub>2</sub> emissions increase from about 31 tons discharged in the one-hour simulation period in 1993 to around 40 tons in 2005, which can be very relevant in terms of meeting the targets of future greenhouse emissions. The difference of the emission level among the different cases is not very significant. The emissions produced in Case 3 are lower than in the other cases, which is in line with the lower participation of catalytic-equipped cars.

The total emissions of PM (Figure 9.25) also rise constantly throughout the projection, which is in a great extent due to the increase in the participation of diesel cars. The differences among the cases become significant at the end of the projection, when a larger proportion of diesel vehicles is portrayed in Case 3. The increased participation of diesel cars in the future can be critical for the levels of PM emissions produced.

It has been shown that the differences in total emissions produced amongst the cases analysed are in general small due to the consideration of the compliance with the 1996 emission regulations, which tends to level off the emission factors from the various vehicle categories.

Changes in the patterns of emissions will influence directly the air quality in the locations in the central area of Chester. Yearly scenarios from 1993 to 2005 were also conceived by the application of the SPITE in order to illustrate the likely long term environmental consequences from the expected changes in the patterns of emissions due to future market car trends.

Figures 9.26 to 9.30 illustrate the average changes in air pollution concentration levels as a result of future car market trends. The projections indicate the extent to which the penetration of diesels and catalytic converters will affect the air quality within the study area after the turn of the century, considering the three cases shown in Figure 9.2.

**Figure 9.26: Average changes in CO concentrations due to future car market trends**

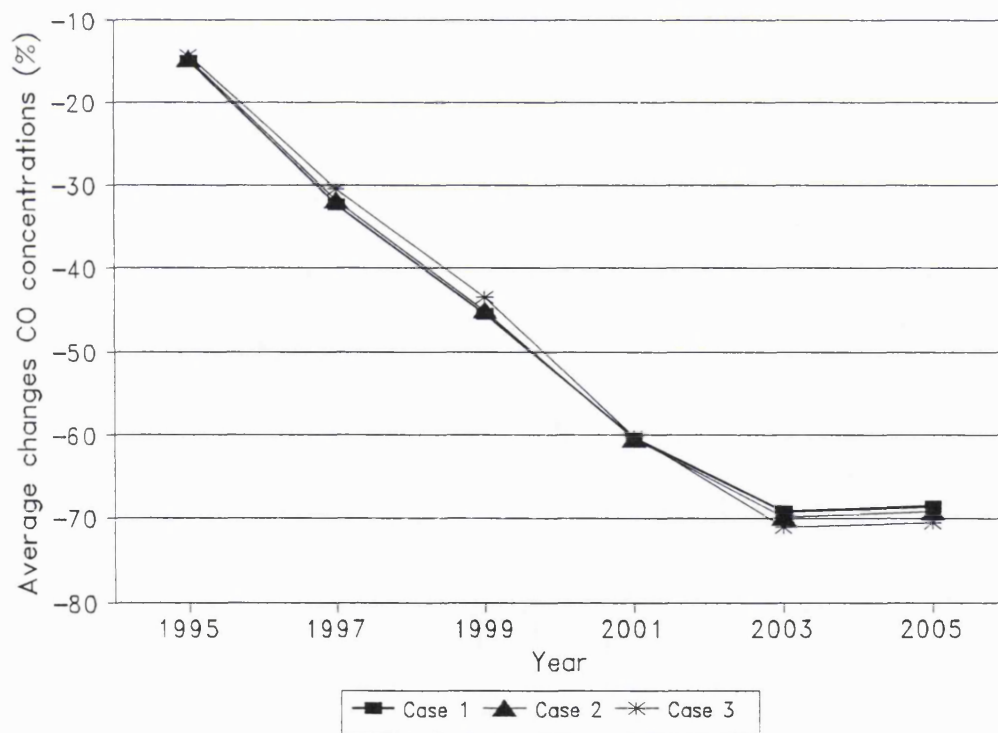


Figure 9.27: Average changes in HC concentrations due to future car market trends

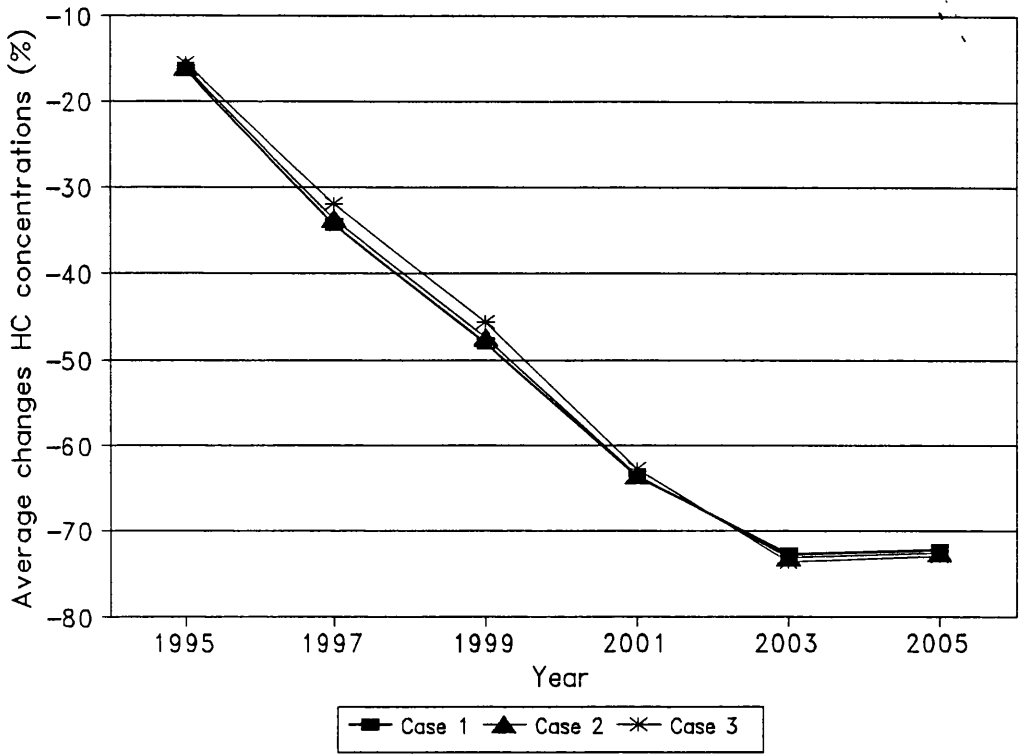


Figure 9.28: Average changes in NO<sub>x</sub> concentrations due to future car market trends

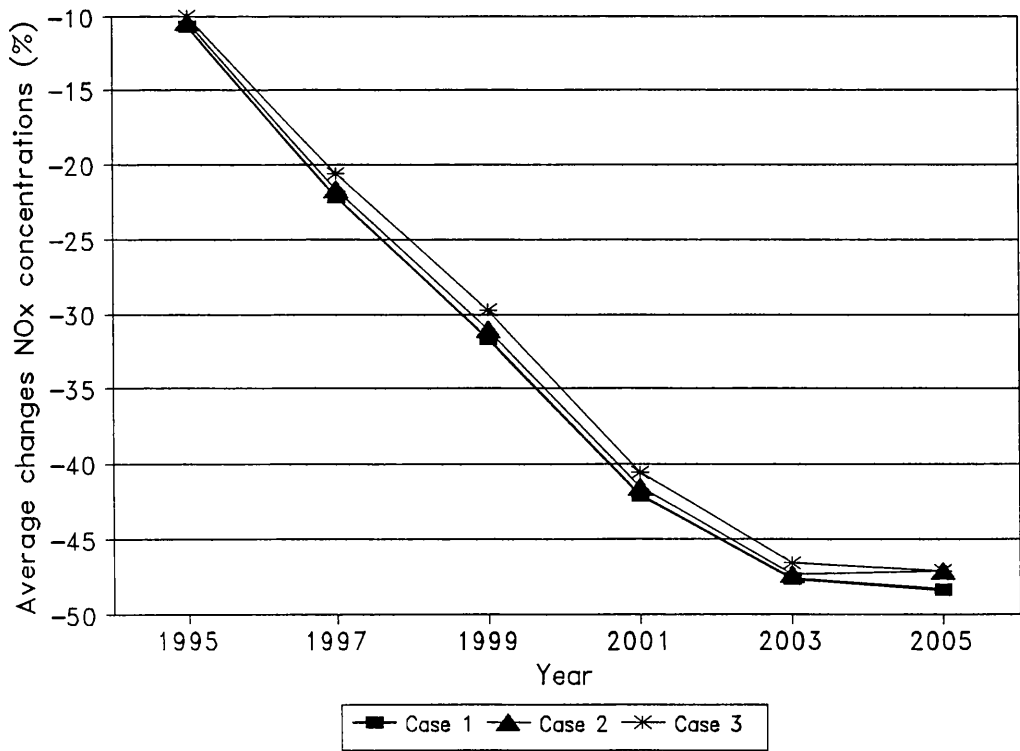


Figure 9.29: Average changes in CO<sub>2</sub> concentrations due to future car market trends

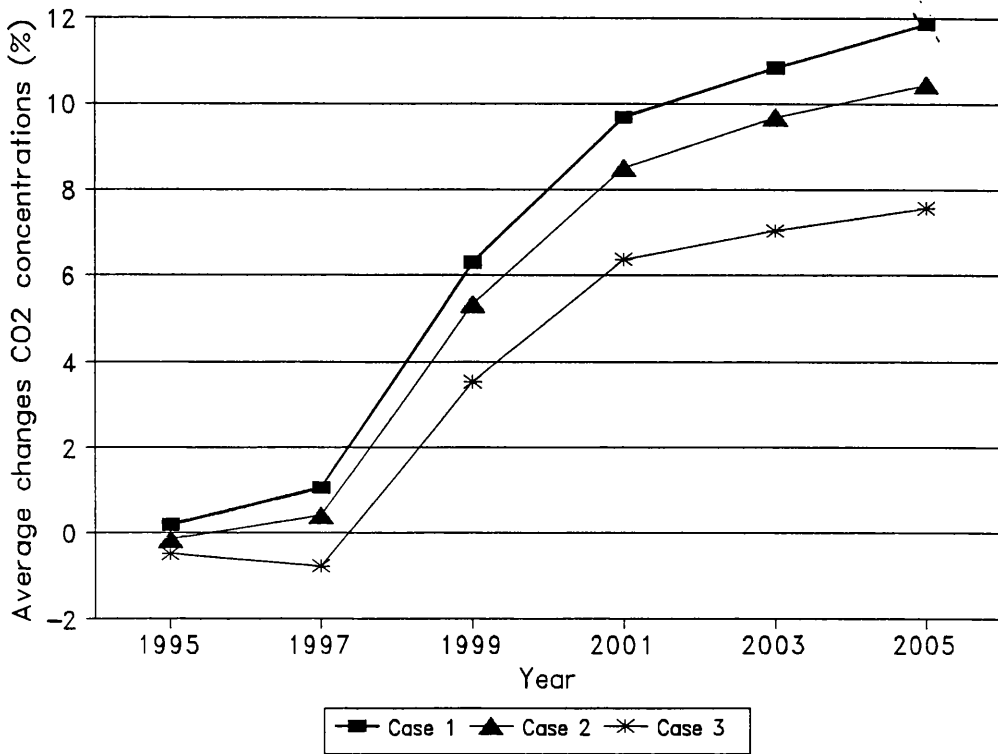
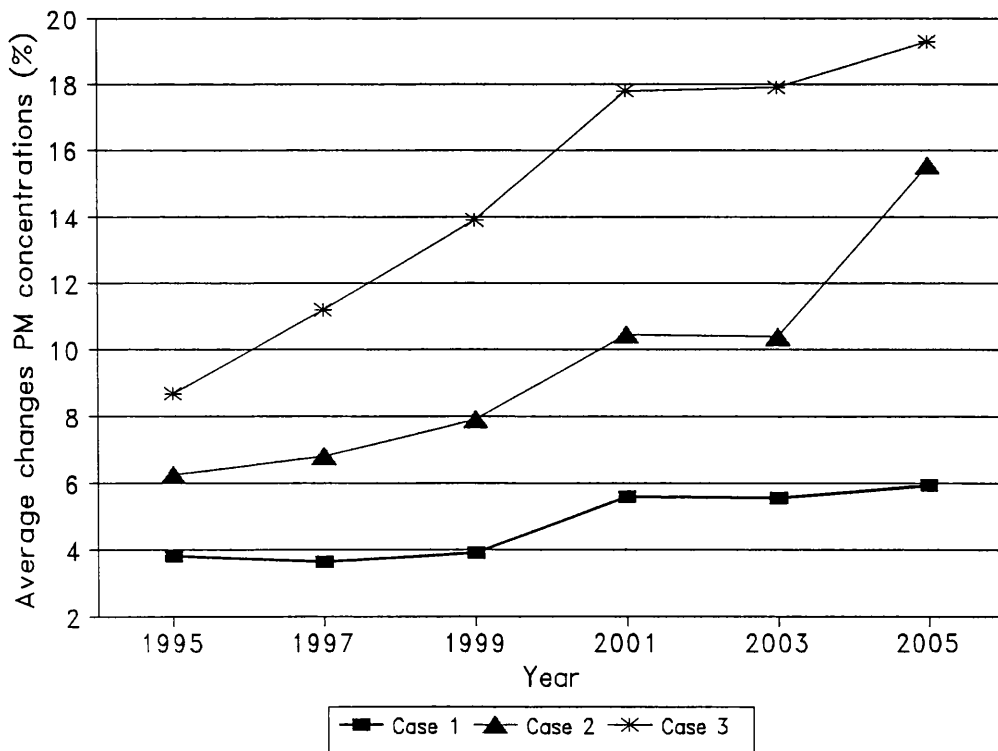


Figure 9.30: Average changes in PM concentrations due to future car market trends



The air quality trends in relation to all the pollutants analysed are compatible with the changes in road transport emissions in the U.K., as illustrated earlier in this chapter.

Like in the analysis of traffic growth (Section 9.1), Figures 9.26, 9.27 and 9.28 show that the average changes in the concentration levels of CO, HC and NO<sub>x</sub> tend to decrease and this tendency is gradually emphasised in the future. This effect is due to the great efficiency of catalytic devices and their growing penetration in the vehicle market. The environmental gains compared to the base scenario (1993) are very significant for CO, HC (in the order of 70%) and NO<sub>x</sub> (nearly 50%).

The simulations carried out indicate continuous environmental gains in relation to these pollutants in the short and medium runs. Reduced emissions are expected until around the year 2003, when such benefits begin to be offset by the effect of growth in the demand for transport. Some authors (e.g. Hamer, 1993) have suggested that the effect of reductions in emissions from the introduction of catalyst devices on the petrol driven vehicles will outweigh by far the effect of traffic growth. However, this is likely to happen only in the long run.

There is no significant difference in the projections of future air quality in relation to these three pollutants for the three cases under consideration. As reported previously, this is due to the compliance with the emission regulations, which impose a reduction on the emission rates of different vehicle categories to similarly low levels.

The average concentration levels of CO<sub>2</sub> (Figure 9.29) tend to rise gradually over the entire period. Case 1 is associated with the highest increases and Case 3 with the lowest. This is a result of three factors: first, the proportion of petrol cars in Case 1 is higher than in the other cases; second, catalysts are fitted in all petrol cars and they increase the emissions of CO<sub>2</sub>; and third, the regulations do not impose a limit for the emission levels of CO<sub>2</sub>.

Figure 9.30 illustrates that the concentration levels of PM also tend to increase in relation to all cases, despite the compliance with the strict limits for PM emissions imposed by the new regulations. The increase in PM pollution is a result of the effects of traffic growth allied to the increased exhaust emissions produced by the growing numbers of diesel cars into the fleet. The magnitude of the increase in PM concentration levels depends highly on the proportion of diesel vehicles on the road. The predictions from Cases 1, 2 and 3 indicate that PM concentrations would increase by about 6, 15 and 20%, respectively, at the end of the

projection. Had the emission regulations not been considered, increases would have varied from about 50% to over 200% within the cases analysed (Chiquetto and Mackett, 1994a and 1994b, Chiquetto, 1995). The participation of diesels will have a strong effect on the level of PM pollution, since they emit many times more PM than petrol cars. This effect will be particularly damaging if the limits in the new regulations are not met or if the PM emission rates from new diesel cars from 1996 deteriorate with use.

In relation to the penetration of diesel cars in the market, the Quality of Urban Air Review Group (1993b) states that: "*... an increase in diesel usage in our urban areas at the expense of the three-way catalyst petrol vehicles is likely to have on balance a deleterious effect on air quality*", and "*... there must be considerable concern over any increase in the proportion of diesel vehicles on our urban streets as their impact of urban air quality is undoubtedly quite serious*".

It is important to bear in mind that although catalytic converters seem to represent a very promising alternative for reducing vehicle emissions, they present limited efficiency when engines are cold or operating at high speeds. The former has important implications for air quality in urban areas in the light of the increasing number of journeys by car which are typically short in length and start cold. In hot weather, a petrol car may have to be driven for about ten kilometres in an urban area before the engine is fully warmed up and operating efficiently. As in the U.K. almost 60% of the journeys undertaken by car are under eight kilometres in length (Department of Transport, 1994c), catalysts often do not reach efficient temperature. Wachs (1993) reports that about 75% of the daily exhausts produced by a modern and controlled American vehicle is emitted in the first few miles of driving. Thus, new cars fully warmed up produce so little pollution that reducing miles of travel seems ineffective at reducing air pollution.

The in-service life expectancy of catalysts is uncertain. There have been reports of devices failing after relatively low kilometreage, but theoretically, a catalyst used with unleaded petrol should have a lifetime of 80,000 to 160,000 kilometres. The failure of catalysts is likely to result in worse emissions than those from a well tuned non-catalyst car (Selwyn, 1995). Such devices are powerless to reduce PM and CO<sub>2</sub> emissions from petrol vehicles. There is no catalyst technology available to be applied to diesel engines, although oxidation catalysts can be applied in diesel cars to remove some of the CO and HC produced (Quality of Urban Air Review Group, 1993b).



In conclusion, the Government's initiative to impose compulsory fitting of catalytic devices into new vehicles will provide great environmental benefits in terms of emissions of CO, HC and NO<sub>x</sub>, whereas the incorporation of emission tests in the annual MoT will ensure that catalytics are working well and pollutant exhausts are kept below the restrictive emission limits. However, further measures are needed to be implemented now in order to control the emissions of CO<sub>2</sub> and PM from transport. Other technological progress in the production of automobiles, such as lean burn engines, pre-heated catalytic converters, the repositioning of catalysts nearer to the engine and the introduction of electric vehicles are expected in the future. Changes such as these will continue to improve engine efficiency and effectiveness with additional reductions in vehicle emissions. Programmes to encourage voluntary retirement of cars which produce high emissions or which are not fitted with catalysts will help the reduction of traffic emissions.

The assessment framework (EAF) was not applied to this particular trend, because the total costs incurred (as shown in Chapter 7) will be very similar for the different cases analysed: vehicle travel times do not vary among the three cases; the costs from fuel consumption vary very little, since the difference in the price between petrol and diesel is offset by the difference in energy performance of petrol and diesel engines; and the difference in the environmental costs produced by petrol and diesel vehicles is almost negligible (see Table 7.5).

### **9.3 Increases in private transport costs**

A change in the generalised costs for travelling by private transport should influence travel patterns and the characteristics of traffic flows. This evidence is acknowledged from the very conception of transport demand modelling and it is evident through the definitions of cost function. The sensitivity of the cost function variables indicates the extent to which marginal travel costs provoke alterations in travel patterns.

In fact, the price mechanism is the most flexible and effective way of influencing travel behaviour. In the short run, travellers can respond to travel price increases by cutting out unnecessary journeys (trip generation), changing travel habits and patterns in order to accomplish less expensive journeys (through changes in the choice for mode, route and destination), or decreasing non-work trip frequencies and lengths or linking them together into multi-destination tours (Horowitz, 1982b). At the long run, travellers can respond price

fluctuations by changing the number and type of vehicles they own or even by changing the location of their residence and work. Long run demand tends to be more elastic than short-term runs because in the former case consumers are better able to adjust to price signals. Transport cost measures are likely to provide regional rather than local environmental benefits, and these would be proportional to the decline in trip making or the reduction in trip lengths.

It is normally difficult to discourage car use, since subsidies to car are very high. Hence, car usage is priced far below its full social costs, which includes an array of externalities such as the costs of congestion and environmental impacts. Nevertheless, the introduction of economic instruments has proven in many occasions to be able to control traffic flows. If traffic flows can be controlled in such manner, so can the environmental degradation produced. The potential for reducing emissions from cars is greater than from other modes, considering that their share in the U.K. transport sector has reached 86% in terms of passenger kilometres (Department of Transport, 1994a).

This section analyses the influence of private transport costs on the environmental impacts generated. Economic instruments may include the taxation of new cars, petrol taxes, parking charges or the creation of tolls such as road pricing. Some considerations will be made next in relation to parking prices, road pricing and the taxation of new cars, but the analysis will concentrate on the environmental implications from changes in the petrol price policy.

Car parking policy is one of the most important instruments for controlling traffic circulation and distribution of space in cities. Planners may choose to make parking easier, through the creation of parking facilities, or more costly and difficult, through charging and controlling parking permissions and available spaces. Parking policy can well serve as an instrument to reduce traffic in busy areas. However, it is ineffective for controlling through traffic or trip length. Besides, the search for spaces may add to congestion and distance travelled. Horowitz (1982a) presents a range of results from modelling studies on the reduction of traffic emissions through parking pricing policies. Hall (1995) reports that parking price policies in the San Francisco Bay Area have changed the emission level of HC and NO<sub>x</sub> by only 1% and CO levels by 3%.

As far as tolls are concerned, electronic road pricing can improve traffic congestion conditions, by eliminating trips from congested areas or reallocating a portion of the demand

on the peak to non-peak periods. Indeed, road pricing is a means to implement the marginal cost pricing principle, and as such, it seems a very effective solution for the mitigation of the environmental deterioration caused by traffic (May, 1986). Road pricing can be implemented through supplementary vehicle licensing fees, manual charges via tollgates or cordon lines, automatic vehicle identification, smart card technologies (Hau, 1992) and area licensing. Hence, it may differ among geographical areas, times of the day or even trip purpose. The elasticity of traffic levels with respect to tolls has been found at -0.45 (Goodwin, 1988). Road pricing was responsible for up to 17% reduction in vehicle emissions in Hong Kong (Jones, 1989), but although technically successful, it has failed both politically and in terms of public acceptance.

In relation to taxation of new cars, the removal of subsidies to company cars seems potentially beneficial to reduce car usage, since about one seventh of the cars are owned by companies (TEST, 1991; Hughes, 1992). Company car usage is very often encouraged by the provision of free fuel and parking.

Fuel taxation is amongst the economic instruments available to curb car usage. Taxation is known to exert influence on fuel consumption and promote fuel efficiency (Royal Commission on Environmental Pollution, 1994). It can influence the demand for travel by adjusting the market price of different forms of travel. In Germany, for instance, a tax allowance is given for trips to work and the costs of car travel for business trips are tax deductible. Diverse environmental effects from petrol tax relief schemes in Germany has been analysed by Blum and Rottengatter (1990). In the Netherlands, taxation on diesel is on average 2.8 times lower than on petrol (Vleuget, Van Gent and Nijkamp, 1990). In some countries, the largest share of fuel selling prices is fuel tax (Wachs, 1993), which has even been portrayed as an alternative to the carbon tax on fossil fuels.

The characteristics of traffic can be affected in different ways by petrol price increases. The variation of travel time, speed and driving modes along each road will in turn affect the patterns of vehicle emissions with implications to both local and global environmental quality. In the United States, the most cost-effective measure for reducing greenhouse gas emissions has been considered to be a fuel tax raise over a period of 5 years (Mills et al., 1991).

There has been a large number of studies on demand-price elasticity. Some works suggest that traffic flows may be little sensitive to travel costs. The argument is that travel cost

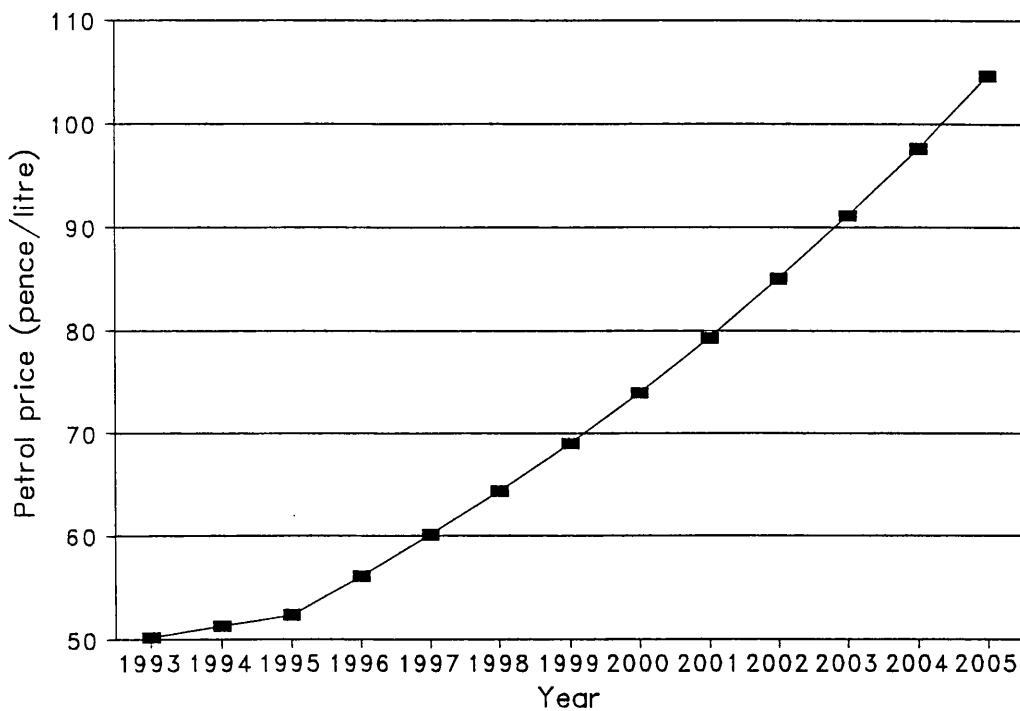
elasticities are usually low, particularly in the short run, and quite often below the level at which revenue would be lost through price increases. According to Virley (1993), drivers tend to maintain existing trip patterns even when faced with large real increases in motoring costs. Other evidences, however, support that fuel pricing could be successful in reducing consumption and affect the pattern of travel demand (Horowitz, 1982a; Goodwin, 1988; Sterner, Dahl and Franzén, 1992; Goodwin, 1992; Oum, Wates and Young, 1992; Virley, 1993). Goodwin (1992) estimates that a 10% increase in fuel price would produce a decrease in traffic of about 1.5% in the short run and from 3 to 5% in the long run, split between a reduction in car ownership and a decrease in car use. Very similar figures have been previously established by Goodwin (1988), who concluded that the mean value of traffic elasticity with respect to petrol price was -0.39. Elasticities of this order may affect the traffic impacts produced. Hall (1995) reports that a price increase of one dollar per gallon (or about 80% increase) would reduce HC and NO<sub>x</sub> emissions by 2% and CO emissions by 4%. Recent estimates from the Cambridge Econometrics indicate that fuel taxation measures will exert an influence on both consumers, by inducing a reduction on the growth in road transport fuel demand, and car manufacturers, who will adjust the level of their supply. This effect is expected to be felt mainly in the longer term, but as suggested by Selwyn (1995), it is unlikely to result in an actual decrease in traffic emissions.

The retail price of motor fuel is currently lower in real terms than it has been for much of the last 40 years (Royal Commission on Environmental Pollution, 1994), despite the fact that taxation accounts for 70% of the forecourt price of fuel (Selwyn, 1995). The proportion of expenditure on fuel out of the average household transport and travel expenses in the U.K. has declined from a third in 1982 to about a quarter in 1992 (Department of Transport, 1994a). One of the strategies for the U.K. Government to reduce future emissions from transport is portrayed by increased taxes on motor fuels. The Royal Commission on Environmental Pollution (1994) proposed that fuel duty be increased year by year so as to double the real price of fuel by 2005. The proposed annual increase in fuel tax required to achieve this doubling in price is 5% in real terms, on top of a further one-off tax rise of 10% (Hughes, 1994), or about 7.2% increase per year over 10 years. This strategy follows the recommendation to reduce traffic, reflect wider costs of transport and help to meet the U.K. target for CO<sub>2</sub> emissions, which is part of a wider sustainable transport policy. The main environmental target is to limit the surface transport emissions of CO<sub>2</sub> in the year 2000 to the 1990 levels (U.K. Government, 1994; Royal Commission on Environmental Pollution, 1994). This proposition was announced in the November 1993 budget, and the Government is

already pursuing steps to increase fuel duty.

Figure 9.31 shows the actual petrol price increases since 1993 and a projection of the proposed petrol prices in the future, considering annual increases which double the 1995 price level to the year 2005.

**Figure 9.31: Projection of future petrol price levels**



The inclination of the projected line changes after 1995, which reflects the extra increases in petrol prices expected within the next ten years.

It has been reported earlier in this section that motor vehicle fuel is generally considered to have fairly inelastic demand, which means that any increase in fuel prices are likely to result in a less than proportionate reduction in fuel consumption. Thus, large increases in petrol prices are believed to be necessary before people cut down substantially on their driving. Howard (1990) also suggests that, if an increase in fuel price were to achieve significant environmental gain, it would have to be very large, which would cause further social and distributional implications as well as political problems. This is indeed a relevant

disadvantage of policies which consider fuel duty increases. As low income households spend a greater proportion of their earning on petrol than higher income households, the former group is likely to be more affected by such taxation than other better-off groups.

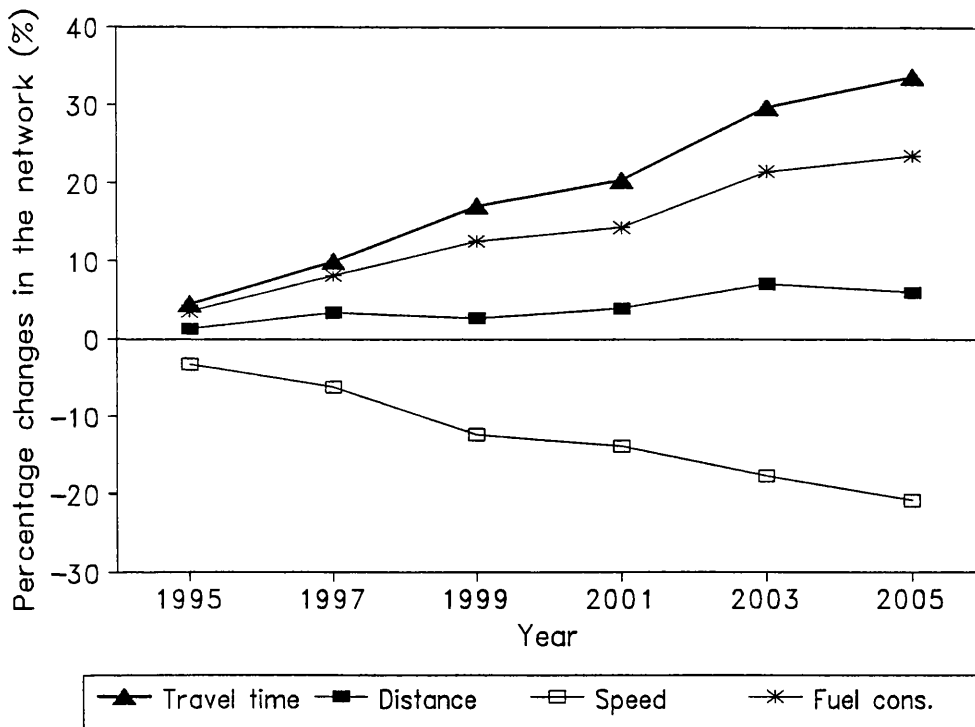
There remains uncertainties in predicting the reaction of motorists and manufacturers to price rises of the nature that was proposed by the Royal Commission. A survey commissioned by a leading motor insurance company found that an immediate rise in petrol price as to almost double the current level, would have no impacts on the driving habits of 52% of Britain's motorists (Nicholson-Lord, 1994). Thus, the Government's strategy of increasing the cost of petrol may not have much of a deterrent effect in itself. The Royal Commission acknowledges that the impact of fuel price increases will not be substantial, since the cost of fuel represents only about a seventh of the total transport costs. Another reason why increases in fuel duty may not be cost-effective for achieving the CO<sub>2</sub> reduction targets is that transport contributes only a small portion of the total emissions and other CO<sub>2</sub> emitting sectors do not have the same tax burden. Other sectors are notably more responsive to changes in price and therefore emission reductions should have been stimulated in these sectors in the first instance (Selwyn, 1995).

A report recently published by the TRL has concluded that it will not be possible to return passenger car greenhouse emissions to 1990 levels by the year 2000 and therefore other sectors will have to contribute for further reductions if the targets are to be met (Selwyn, 1995). Similar conclusions have been drawn in this work. Considering that spending on fuel is likely to increase by 1-2% per annum as real incomes rise (Barker et al., 1994), real transport fuel prices would have to rise very sharply in the shorter term or as under the Government's plan in the long term, in order to reduce greenhouse emissions sufficiently as to achieve stabilisation targets.

The effects of petrol price changes are analysed in this work through the creation of projected scenarios. These scenarios simulate yearly increases in petrol prices, as recommended by the Royal Commission on Environmental Pollution (1994). The projections consider the effect of low traffic growth, which is based on the forecasts by the NRTF (more details in Section 9.1). Traffic conditions and travel costs will change as a result of the combination of the gradual effect of traffic growth and increase in fuel prices. For each year, traffic is re-assigned onto the road network under different cost function parameters. As petrol prices increase, the parameter CK in the cost function (Equation 6.1) becomes more significant, and

as traffic grows, travel times ('T' in Equation 6.1) increase. Thus, in each scenario travellers are faced to new sets of cost parameters and are forced to re-assess their travel choices in relation to the modes and routes available. Figure 9.32 shows the results from the various applications of the transport model (Section 6.1) to future scenarios, in terms of the percentage changes in the traffic indicators for the Chester road network, as a consequence of the effect of traffic growth and petrol price increases.

**Figure 9.32: Percentage changes in the traffic indicators due to petrol price increases**

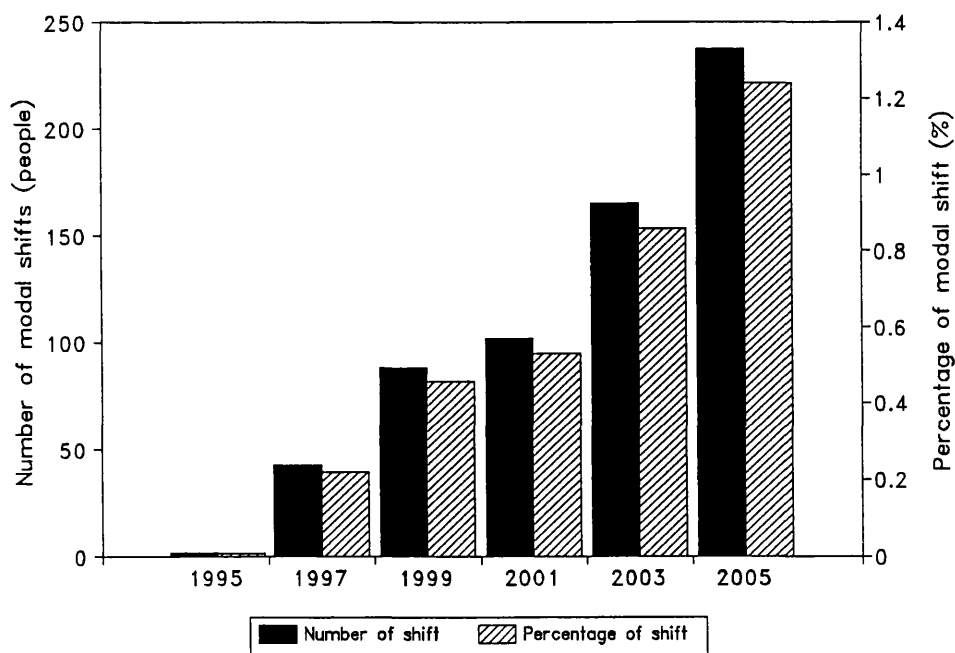


These trends are very similar to those indicated in Figures 9.3, 9.4, 9.5 and 9.6. In percentage terms, travel time is the indicator which rises most sharply within the projection. At the end of the period analysed, it is estimated that travellers will spend almost a further 35% of time in travelling. Fuel consumption is also expected to increase constantly, up to about 25% of what is utilised in the base-scenario. The total distance travelled in the network increases very slightly and reach around 5% at the end of the projection. As a result of increasing congestion in the network, the average traffic speed is estimated to decrease gradually down to about 20% of the speed at the beginning of the projection.

It is very important to bear in mind, however, that because the trip matrix was assumed fixed in the modelling of transport demand (Section 6.1), any changes in trip generation and distribution, particularly as a result of petrol price increases, were not taken into consideration in the analysis. Thus, this assumption is likely to have overestimated the effects in all traffic indicators described above.

The changes in modal choice have been derived through estimates produced by the modal split model (Section 6.1.2). Figure 9.33 shows the absolute (dark bars on the left) and percentage (dashed bars on the right) shifts from private to public transport in the study network, as a result of petrol prices increases and considering the effects of traffic growth.

**Figure 9.33: Shift from private to public transport due to fuel price rises**



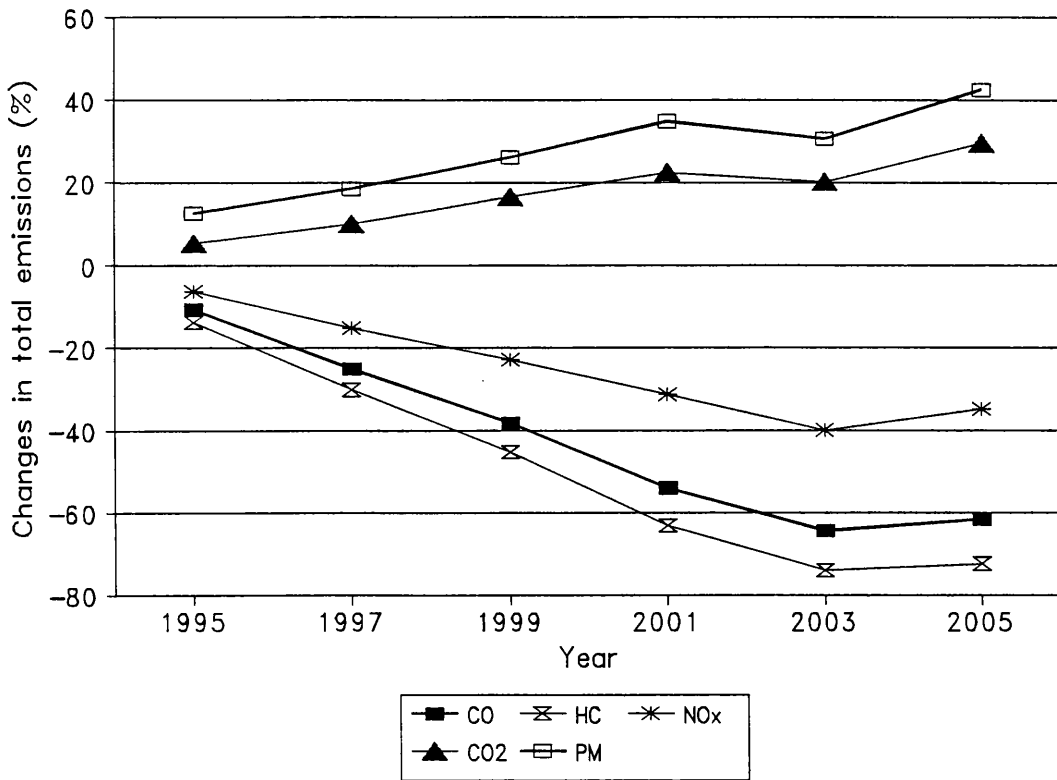
As expected, the changes in the choice for alternative modes of transport have been very modest, since the elasticity of modal choice in relation to fuel price is very low. According to the modal split simulation, by doubling the price of petrol from 1995 to the year 2005, only about 1.24% of car travellers would have chosen to shift to public transport. This figure is considerably lower than the elasticity of petrol prices in respect to public transport demand of +0.34, as reported by Goodwin (1988). In absolute terms, however, the number of



travellers which have changed modes as a result of fuel price rises at the end of the projection is about ten times higher than in the low traffic growth scenario with no changes in prices (see Figure 9.7).

The total emissions produced in the Chester network have also been calculated by the application of the emission model in the SPITE, considering the combined effects of traffic growth and the proposed increase in petrol prices. Figure 9.34 below shows the projections of the percentage changes in the total emissions of CO, HC, NO<sub>x</sub>, CO<sub>2</sub> and PM.

**Figure 9.34: Percentage changes in total emissions**



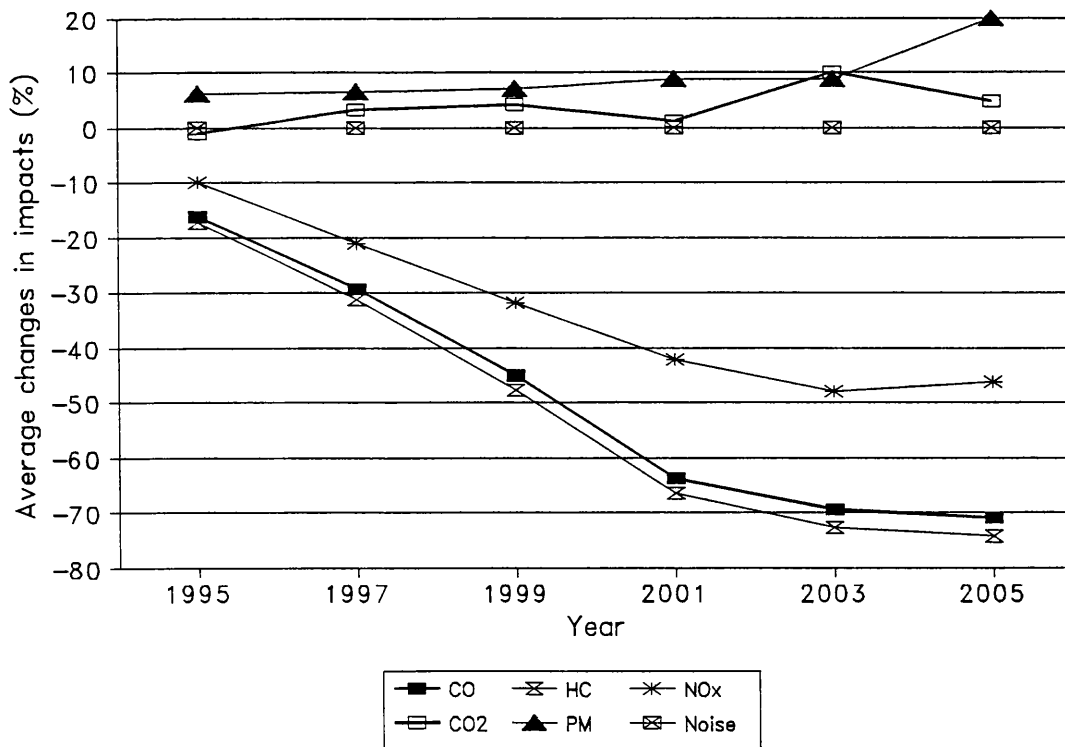
In accordance with previous projections, Figure 9.34 shows that the total exhaust emissions of CO, HC and NO<sub>x</sub> from traffic sources in Chester tend to decrease continuously until the year 2003, when they start again to rise. The magnitude of decreases reach up to about 60, 75 and 40% of the base-scenario emissions in relation to the above mentioned pollutants, respectively. The effects of petrol price increases are not much significant along these projections. Thus, the reductions in traffic growth due to the increase in fuel duties must have fallen within the bounds of the NRTF scenarios.

The emissions of CO<sub>2</sub> increase up to 30% in relation to the base-scenario figures. The effect of petrol price increases does not seem to be noticeable on these projections, which would suggest that such Government's strategy will not be very effective if not accompanied by further devices to discourage car usage. The increase in the CO<sub>2</sub> emissions in Chester indicate that an overall increase is expected on the total emissions produced nationwide and this might be a problem to governmental attempts to reduce greenhouse emissions in the future. This means that either new strategies to reduce these emissions will have to be put into practice or other sectors will have to reduce their emissions further in order to compensate the very large increase from the transport sector.

The total emissions of PM also increase gradually up to almost 40% in relation to the levels in the base-scenario. Such increase will have important implications in future air quality.

Next, the SPITE (as described in Sections 6.2 and 6.3) was applied in order to verify the likely changes in future environmental quality as a result of the effect of gradual increases in the price of petrol combined with the effect of traffic growth. The projected average changes in air pollution concentration and noise levels are shown in Figure 9.35.

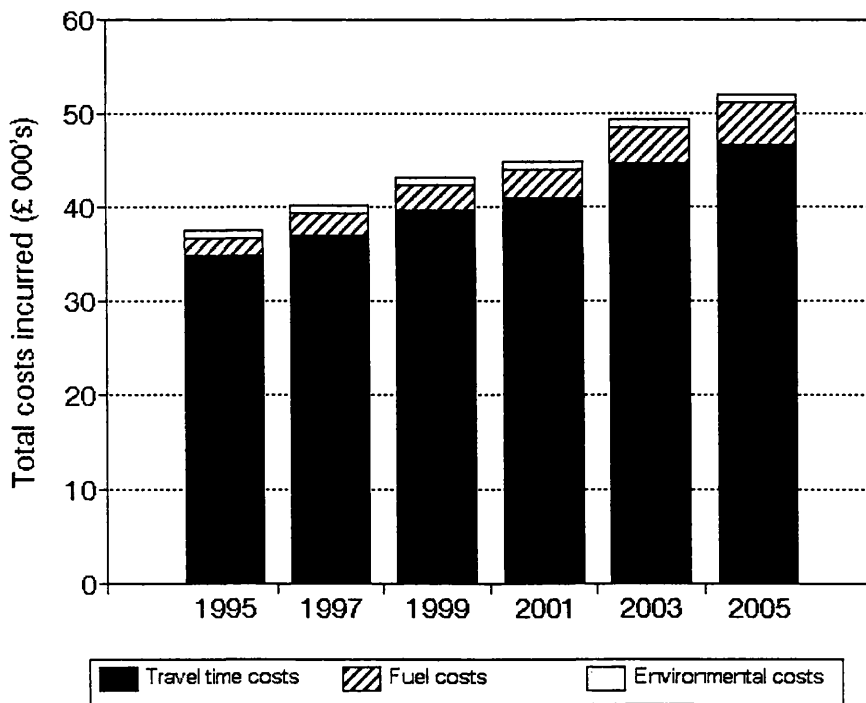
**Figure 9.35: Average changes in the environmental impacts**



This illustration indicates that the average changes in the concentration levels of CO, HC and NO<sub>x</sub> tend to decrease up to about 70, 75 and 45%, respectively, in relation to the base-scenario. The average concentration levels of CO<sub>2</sub> and PM tend to rise over the projection. The gradual increase in PM concentration (up to about 20% of the initial levels at the end of the projection) indicates that PM pollution will have serious consequences for future air quality. The changes in the average levels of traffic noise are negligible and these cannot be noticed in the above illustration, even the effects of traffic growth being accounted for.

The application of the EAF has provided estimates of the costs incurred in the network along the projection, in terms of travel time, fuel consumption and environmental damage (Chapter 7). These costs are assumed not to increase in real terms over time. Figure 9.36 shows the estimates of the total costs incurred in Chester, considering the gradual increase in petrol prices and the effect of traffic growth.

**Figure 9.36: Total costs incurred due to traffic growth and petrol price increases**



In accordance with previous analyses, the costs related to the value of time spent in travelling are much higher than the other costs included in the EAF. The costs incurred in each category considered in the EAF increase constantly along the projection. However, as petrol

prices increase, the expenses with fuel consumption represent a higher proportion of the total costs incurred.

The increase in petrol prices tends to influence travellers in their route choices and this effect is accounted for in the assignment procedure by a higher value of the parameter CK in the cost function (Equations 6.1 and 6.2). Thus, as petrol price increases, the factor 'distance' becomes more important in the travel decisions and travellers start to trade 'time' for 'distance', in the sense that they become more prepared to choose journeys which take longer but are shorter in length. As a result, the costs incurred from the total time spent in travelling are much higher in comparison with those in the low traffic growth scenario (Figure 9.19), whereas the costs incurred from the environmental degradation, which are accounted in the EAF only as a function of the distance travelled, are lower than in the low forecast scenario.

The total costs incurred in the road network under the petrol price increase scenario (Figure 9.36) were estimated to become higher in the longer term than the total costs in the high traffic growth scenario (Figure 9.20). This emphasises the economic implications of fuel price rises to society in the future.

In conclusion, according to the simulations carried out in this work, neither future levels of the total emissions produced in Chester nor local levels of environmental degradation will be affected considerably by the individual effect from the gradual increase in petrol prices. Overall emission reductions from traffic are more likely to be achieved by improvements in vehicle fuel economy rather than reduced kilometreage. Pricing policies on private vehicles will be way more effective if their introduction is accompanied of further strategies designed to promote public transport, walking and cycling, or to discourage the use of the car.

It must be borne in mind, however, that the transport demand model used in this work has assumed a fixed origin-destination trip matrix. The changes in trip generation and distribution, which would have probably resulted from petrol price increases, were not taken into account, which has indeed oversimplified the environmental effects of petrol price increases in the future.

Nevertheless, as far as the overall costs incurred in the system are concerned, the cost estimates for Chester indicate that the magnitude of fuel price rises will have major economic implications for the nation in the years to come.

## 10 SUMMARY AND CONCLUSIONS

Three key issues have been examined in this research on the impacts of road transport on the environment. The first issue was the relevance of the environmental concern in transport. The review chapters (Chapters 2, 3 and 4) have demonstrated that road traffic air pollution can cause a wide range of problems to human health, and that traffic noise can also cause health hazards. It has been emphasised that the consideration of the environmental externalities produced by road traffic in transport planning policies has become important for the protection of the urban environment. As shown in Chapter 8, transport policies are now regarded as practical and effective means to resolve environmental problems, particularly in urban centres. Sustainable planning policies have started to be implemented in order to reduce both traffic volumes and the amount of emissions produced at the source.

The second and most important issue dealt with in this research was the investigation of the extent to which the implementation of transport policies and the realisation of future trends can affect local and global environmental conditions. The analysis of environmental impacts was based on the application of widely accepted transport models (Section 6.1) in conjunction with environmental predictive models (Sections 6.2 and 6.3) to the Chester road network. The level of accuracy which can be expected from the application of the environmental predictive models has been carefully examined and was reported at the end of each section in Chapter 6. The transport and environmental models together form the Suite for the Prediction of the Impacts of Transport on the Environment (SPITE). The range of considerations incorporated in the SPITE constitutes important advances in relation to existing predictive models. In particular, the emission model in the SPITE has been redeveloped in this research to account to a great extent for the pattern of vehicle emissions under conditions which are typical in urban driving. Also, the noise propagation and pollutant dispersion models have been adapted as to predict the level of environmental degradation at a large grid of receptor points located at road junctions. The transport policies simulated (Chapter 8) were: the implementation of a pedestrianisation scheme, the implementation of a traffic calming scheme and a restriction of the traffic of heavy goods vehicles in the central area. The future transport trends investigated (Chapter 9) were: the impacts of the low and high traffic growth, the likely changes in the vehicle market and the gradual increase in petrol prices.

The third issue analysed in this research was the magnitude of the costs associated with the

local environmental impacts produced by road traffic. The literature review has demonstrated that monetary valuation of the environmental impacts is possible and that numerous attempts have been made to derive approximate economic values to environmental deterioration (Chapter 5). It has been suggested that, although still not fully developed, the techniques for environmental valuation are now quite sophisticated to be set beside financial measures of other costs and benefits. Chapter 7 has presented the Environmental Assessment Framework (EAF), which was specifically developed in this research to incorporate the environmental dimension into the economic evaluation of transport policies. This framework constitutes progress in relation to usual assessment procedures.

The conclusions from the various topics investigated in this research are addressed below.

- **Accuracy in environmental modelling**

It has been demonstrated that a high level of accuracy in air pollution modelling cannot realistically be expected, due to the complex nature of the relationships represented by models as such and to the difficulty of including further relevant variables in them. The validation procedure of the air pollution dispersion model (Section 6.2.3) has indicated that, considering the difficulties and uncertainties involved with predictions, it has performed well in most of the time periods analysed. The analysis of the comparisons between measurements and estimates in the study area has shown that the percentage of the estimates which lie within  $\pm 50\%$  of the measurements of CO, HC and NO<sub>x</sub> concentrations were 51, 37 and 52, respectively, and the percentage of estimates within  $\pm 100\%$  were 83, 100 and 96, respectively. The level of accuracy provided (Table 6.4) is comparable to the best results from other models available not only in this country but also abroad. It has been shown in Section 6.3.1 that noise propagation models are able to produce results which are, in general, more reliable in comparison with the outcomes from air pollution dispersion models. The results from the validation procedure of the traffic noise model have indicated that the noise predictions in this work can be subject up to 3% of inaccuracy in comparison to the observations from field surveys. The main reasons for the better capability of the prediction of traffic noise are, first, traffic flows and speed exert only a logarithmic influence in noise levels (whereas traffic influences the pollutant emission levels directly), second, the estimation of L<sub>10</sub> exclude the noise peaks in the 10% of the time where the highest levels occur (whereas all the peaks in pollutant emissions are accounted for in the hourly average predictions), and third, meteorological factors influence noise far less than air pollution.

Despite the constraints in the accuracy of predictive models, they are generally more reliable when used to compare the impacts of different policies. Thus, when relative rather than absolute results are derived from comparative applications of the predictive models, many of the inaccuracies in the modelling assumptions and simplifications will cancel out. This was the case in this work, where predictions have been made for the percentage changes in environmental quality between the various policies and the base-scenario. Thus, because the same assumptions were made for both situations, the results from the environmental models are expected to be more reliable than if they had been applied to derive absolute values in isolated situations. It is worth recalling that both the air pollution and noise models are currently recommended in the Design Manual for Roads and Bridges for the prediction of the environmental impacts from road traffic. This suggests that both methodologies are widely accepted and expected to provide adequate results.

- **The application of the Suite for the Prediction of the Impacts of Transport on the Environment (SPITE)**

The SPITE has been shown to have a great potential for application to a wide range of transport planning policies and traffic management schemes and their application can produce a great deal of information about environmental conditions throughout a road network.

The application of the modal split model to the various policies and trends analysed in this work has shown that the patterns of travel choice between the transport modes available have not suffered appreciable changes. As a consequence, the low elasticity of modal switches in the simulations carried out in Chester has produced little environmental impacts. However, the results of the assignment model have indicated that the characteristics of traffic flows change considerably when various changes are introduced into the transport system (as shown by the scenarios analysed in Chapters 8 and 9). It has been shown that the results from transport models can greatly influence the outcomes of air pollution models, but noise levels were found to have little sensitivity to changes in the characteristics of traffic flows.

The emission model in the SPITE has shown the magnitude of the total emissions produced within the network under the various scenarios analysed in this work, and this is particularly relevant in terms of the contribution of the study area to the greenhouse effect. The predictive models have indicated the levels of air pollution concentration and noise in various locations of the Chester network, which were very high at points around certain busy junctions (Section

6.4). It has been shown that the locations where the highest concentration levels have been found are not necessarily the same for all pollutants.

The results from the application of the SPITE in Chester (Chapters 8 and 9) have indicated that transport trends, planning policies and traffic management schemes can have significant impacts on the environment. However, it has been demonstrated that the implementation of an isolated scheme is unlikely to produce absolute environmental benefits, whereas the implementation of a package of schemes and policies combined as part of a comprehensive plan is much more likely to produce successful results. There is not a simple answer to the changes in the environmental impacts from changes in the transport system. For a particular change in the system, the impacts produced can vary largely according to different environmental indicators and the location in the network. The results of the application of the predictive models can indicate the locations where the environmental standards are likely to be exceeded. This can have important implications to human well-being and health and can be potentially helpful to aid decision-making in environmental policies as well as in the resource allocations for transport schemes.

Therefore, the application of the SPITE can be useful in transport planning as a tool to furnish politicians and planners with scientifically based information about specific environmental consequences from changes in the transport system. The results can be used in policy decisions as an indication of the environmental consequences, together with other conventional transport indicators such as accidents, congestion, travel time and energy savings. Although the results from the particular policies simulated in Chapter 8 are specific to the study area, some general trends may be found in similar urban areas. The findings from the long-term transport trends analysed in Chapter 9 (such as the future total greenhouse emissions) have indicated the magnitude of the impacts expected in the study area, but some of these findings could be expanded for the nation as a whole.

- **The application of the Environmental Assessment Framework (EAF)**

The results from the EAF have shown that the magnitude of the figures derived for the environmental costs in Chester have been found to be very low in comparison to other economic costs (Chapter 7). The travel time savings are far the greatest component of the total costs incurred from road transport. However, it is important to bear in mind that the consideration of such costs has probably been overestimated (since it is probable that not



many people actually perceive their travel times as money expenditures, or perhaps not as highly as suggested in the Traffic Manual Appraisal), while the incorporation of the environmental costs would have been probably underestimated (many of the various effects from environmental damage, such as the global warming, have not been accounted for in the framework). According to the analysis undertaken, the differences in costs between the base-case and the various simulated scenarios are also relatively small. However, the cost estimates have been made only for one-hour periods, so the annual valuations may account for significant amounts in absolute terms.

Thus, the monetary outputs from the environmental valuation performed by the EAF could help to furnish politicians and planners with further economic grounds for supporting decision-making. Even though the estimates of the environmental costs may be relatively small, this should not discourage planners from incorporating them into the assessment of transport schemes. On the contrary, this modest attempt to show that these costs exist, that they can be comparable with other costs such as fuel consumption costs and that they are probably more substantial than those found in this study, should inspire decision-makers to continue making efforts in order to incorporate the environmental dimension into the economic assessment of transport schemes in a more comprehensive manner.

- **Interpretation of results**

It is important to bear in mind that, considering the difficulty in obtaining reliable input data together with the vast scope for inaccuracy due to a range of simplifying assumptions made at the various stages of the modelling process in this research, model results can only provide mere indicators for supporting decision-making rather than definitive answers to transport problems.

- **The implementation of the pedestrianisation scheme**

The results from the simulation of the implementation of the pedestrianisation scheme in Chester, as reported in Section 8.1, have indicated that major environmental benefits in terms of air quality and noise levels can be achieved in the central area. These benefits accrue to a large number of people who live, work or visit the pedestrianised streets. However, the changes in the concentration of air pollutants and traffic noise levels in the network as a whole, which were due to the overall redistribution of traffic, have been found to be small.

The results from the EAF have indicated that pedestrianisation is likely to bring an overall economic benefit to the Chester network, since the total costs incurred in the situation 'after' pedestrianisation were lower than in the 'before' scenario, which was the consequence of the traffic assignment particularities in both situations.

- **The implementation of the traffic calming scheme**

As reported in Section 8.2, the implementation of a particular traffic calming scheme in Chester has produced a range of different changes in air pollution concentrations and noise levels around the traffic calmed area. Considerable environmental improvements were achieved in the locations closest to the calming scheme, which are environmentally sensitive areas in which some of the traffic volume has been reduced. The environmental impacts in other locations have increased or decreased very slightly as a result of the small changes in traffic conditions outside the calmed area.

- **The exclusion of heavy goods vehicles from the central area**

The results from the simulation of the exclusion of heavy goods vehicles from the selected roads in the central area (as reported in Section 6.3) have indicated that the flow of lorries can be largely responsible for the impacts on the quality of the local environment in Chester. This scheme has shown to be able to produce major reductions in the average levels of PM concentration and considerable benefits in terms of NO<sub>x</sub> and noise pollution in the central area. Therefore, policies which cause appreciable changes to the flow of lorries should take into consideration the environmental implications, particularly in relation to the emissions of PM, NO<sub>x</sub> and noise from traffic, at early stages in the planning process.

- **Road traffic growth**

The results from the predictions reported in Section 9.1 have suggested that road traffic growth is unlikely to affect significantly the average levels of both air pollution concentration and noise. Because of the assumptions used in the traffic assignment model, the congestion effect in the scenarios analysed has induced the spread of traffic and the dissipation of the environmental impacts produced. However, it has been demonstrated that the congestion originating from traffic growth will probably cause significant impacts at particular locations, which can be relevant for traffic management purposes. The results from the application of

the EAF in Chester have indicated that the magnitude of future traffic growth levels will have major economic implications on the overall costs incurred by the local community. By extrapolating the results of this research to the entire country, it can be surmised that future traffic growth may have major implications for the nation's economy.

- **Changes in car market**

According to the analysis carried out in Section 9.2, future changes in car market trends will have important implications for both local air quality and global emissions, because these changes will alter quite substantially the patterns of vehicle exhaust emissions. In particular, the effect of the compulsory fitting of catalytic converters into new vehicles has been predicted to outweigh the effect of traffic growth in relation to the emission levels of CO, HC and NO<sub>x</sub>, which are expected to decrease continuously in the short and medium terms. On the other hand, the findings from this research have indicated that the fitting of catalytic devices (which contribute towards the increase in CO<sub>2</sub> exhaust emissions from traffic sources) together with the effect of traffic growth, can play an important role in terms of the total greenhouse emissions in the long run. It has been estimated that a high increase in PM levels will take place due to the effect of the increasing penetration of diesel cars into the market, which emit many times more of this pollutant than petrol cars. Therefore, specific measures may be necessary to control future emissions of CO<sub>2</sub> and PM from road transport.

- **Increases in petrol price**

The results from the predictions carried out in Section 9.3 have shown that the gradual increase in petrol prices, as proposed by the Royal Commission on Environmental Pollution (1994), will not produce considerable environmental benefits. The modelling predictions suggest that much larger increases in petrol prices would be necessary to achieve any significant environmental gain. This is because fuel price has, in general, fairly inelastic demand in relation to fuel consumption and it represents only a small portion of the total transport costs. Therefore, the Government's strategy for achieving the targets of greenhouse emission reduction by increasing the cost of petrol is unlikely to be entirely successful on its own. Pricing policies on fuel will be likely more effective if their introduction is accompanied by further strategies to promote public transport, walking and cycling and to discourage the use of the car. Besides, as was shown by the results of the EAF, this strategy can have serious economic and social implications arising from the overall increase in transport costs.

## **11 RECOMMENDATIONS**

### **11.1 Further academic research**

This section presents recommendations for further work to be carried out, which could greatly improve the quality of the results of research in the field of environmental modelling and assessment.

#### ***11.1.1 Setting more appropriate air quality standards and guidelines***

The establishment of air quality standards and guidelines has been, in many cases, politically based rather than scientifically founded (Sections 3.7 and 4.5). Academic initiatives should now be taken in seeking more appropriate settings. It is recommended that efforts are made to define a comprehensive set of guidelines which consider the need for environmental protection, based on long term epidemiological work on the degree of damage that various air pollutants cause to health. The setting of environmental standards and guidelines should seek a proper balance between environmental protection and economic implications, such as the enforcement of further technological requirements and restrictions in mobility. Appropriate settings will enable decision makers to establish their policies in accordance with environmental objectives over specified time scales.

#### ***11.1.2 Improving transport modelling***

The assumption of fixed origin-destination trip patterns implies that any changes in the transport system are considered not to affect the attraction and suppression of trips (trip generation) nor the travel destination patterns (trip distribution). This has been a major cause of criticism in transport modelling. Such common simplification may produce deceptive results, principally when considering future trends in transport (Chapter 9) from which travel costs increase and trip making tends to become less attractive. It is recommended that more comprehensive transport models which incorporate the effects of trip generation and distribution are applied to provide better representation of the actual changes in the characteristics of traffic conditions.

Improvements in transport modelling with the consideration of more dynamic changes in

transport demand would enable further considerations to be incorporated in the environmental modelling process. In particular, a more realistic simulation of the interaction between buses and cars (which would influence the flow and frequency of buses) would provide a better representation of the actual traffic circulation conditions, especially in areas where the participation of public transport is substantial, and therefore a more precise estimate of the environmental impacts produced.

#### ***11.1.3 Improving emission models***

The improvement of emission models based on actual vehicle emission rates, such as the one developed in this research (Section 6.2.1), can be achieved by deriving more comprehensive vehicle emission rate data. It is recommended that more vehicle categories are created in order to disaggregate emissions into various homogeneous groups and local rather than national figures are used for the estimates of typical traffic emissions. Emission rates can be modelled with better precision if the trajectories of individual vehicles are analysed instantaneously, according to space-time variations, speed, driving pattern, acceleration rate, and not averaged over the whole cycle as is currently the case. Since traffic emissions are likely to change within a few years, emission rate inventories should be permanently reviewed in order to provide updated information to be used in emission modelling.

#### ***11.1.4 Improving dispersion models***

Extensive data from continuous air quality monitoring under various traffic and meteorological conditions are necessary for the establishment of more precise calibration parameters for predictive dispersion models. It is recommended that other meteorological factors such as temperature (and the incidence of temperature inversions), humidity (and the incidence of rain), pressure and atmosphere stability are empirically incorporated into dispersion modelling in order to improve results. Furthermore, a greater knowledge of vehicle-induced turbulence is required in order to improve model predictions.

#### ***11.1.5 Improving the monetary valuation of environmental assets***

Although the costs from the environmental damage are subject to various uncertainties inherent in the nature of the valuation, some effects which two decades ago were classified as intangible, are now being estimated monetarily. Thus, if development continues, other

remaining uncertainties are expected to be resolved within the near future. Progress in improving results of the monetary valuation of environmental assets is possible and should be pursued continually. Considerable research expertise ought to be built up until environmental valuation is better accomplished. More profound research design should be pursued with stronger links between empirical work and economic theory. More information, time and resources should be made available for encouraging researchers to improve the performance of existing techniques or develop new, more reliable and more cost-effective methods. More direct and accurate evidence must be developed on the dose-response relationships between changes in environmental quality and the various consequences in health, damage, productivity, nuisance and well-being. As research in environmental valuation progresses and improved knowledge of environmental costs is revealed, current valuations may be updated to more realistic levels. Finally, efforts should be made to better inform society, so people can be more prepared to accept the devices conceived to attribute values to environmental degradation.

#### ***11.1.6 Analysing the environmental impacts of other transport policies and trends***

It would be useful to verify the extent to which other policies and schemes can affect the impacts of traffic on the living environment. In particular, it would be interesting to analyse the impacts from the implementation of road congestion pricing and more comprehensive traffic calming schemes, such as wide-area street treatment and '20 miles per hour' zones, which would have produced much more significant changes in relation to the traffic calming measures analysed in this work, both in terms of traffic circulation and environmental benefits. Other possibilities include schemes such as changes in road capacity, bus routes and frequencies, public transport fares, parking charges, park-and-ride, bus priority strategies and new transport systems.

It would also be worthwhile to analyse the extent to which factors that are likely to be implemented in the future, such as technological improvements in vehicle manufacturing (as suggested in Section 8.4.7), the further development of the existing technologies (for instance, lean burn engines), the creation of new devices for controlling the emission levels and the use of alternative transport fuels, will contribute to reduce the emissions produced by road traffic in the years to come.

## **11.2 Political initiatives**

This section presents some recommendations for political initiatives which could reduce the environmental impacts of traffic and improve the quality of living of present and future generations.

### ***11.2.1 Enforcing emission limits to new and in-service vehicles***

Inspections should be more rigorous in the MoT test so that the implementation of emission limits for in-service vehicles are more effective. Stricter limits for in-service vehicles after 1996 should be established and enforced in the MoT test, in order to assure that emissions from new vehicles do not increase in subsequent years and that the environmental objectives of the regulations imposed on the emissions from new vehicles do not fail to be met over time.

### ***11.2.2 Implementing automatic continuous monitoring***

It is important that further automatic continuous systems to monitor air quality be implemented in the U.K. The availability of more comprehensive information on actual levels of air pollution concentrations and traffic noise would provide better grounds for environmental assessments, would make it easier for decision-makers to set new policies, would enable initiatives to be taken based actual environment data and would facilitate the implementation of quick control measures. The results of continuous monitoring would also be very useful for the calibration of environmental predictive models.

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