Quantification and Valuation of the Effects of Traffic-Induced Air Pollution on Mortality

An Analysis of 14 British Cities

Bernhard F. Walter

(Dipl.-Volkswirt)

Thesis submitted for the degree of

Ph.D.

in Economics at the

University of St. Andrews

February 2000

Abstract

An effective transport system is vital for economic well-being and the quality of life. Yet negative external effects of road traffic play a growing role in the environmental political discussion. The majority of these effects produce so-called external costs as they are not paid by generators but are inflicted on society as a whole. In particular, emissions form motor vehicles are of serious concern as they are widely recognised in causing damage to human health, ranging from breathing discomfort to cardio-respiratory diseases resulting in premature death. The latter are the focus of this study, where the ultimate objective is the valuation of the effects of traffic-generated air pollution on mortality. The valuation procedure is undertaken in a stepwise manner. First, the short-term association between daily mortality and ambient air pollution in 14 major British cities between 1992 and 1997 is assessed, using time-series analysis. Second, based on the estimated exposure-response functions the total numbers of premature deaths attributable to air pollution in general, and to road traffic in particular is quantified. Finally, by applying willingness to pay estimates for the value of a statistical life (employing air pollution related adjustments) the total external costs attributable to air pollution are calculated. These figures are an important element for transport policy decision-making, particular on a decentralised local authority level.

Declarations

I, Bernhard F. Walter, hereby certify that this thesis, which is approximately 107,000 words in length, has been written by me, that it is the record of work carried out by me and that it has not been submitted in any previous application for a higher degree.

April 14, 2000

(Bernhard F. Walter)

I was admitted as a research student in September 1996 and as a candidate of Ph.D. in September 1996; the higher study for which this is a record was carried out in the University of St. Andrews between September 1996 and February 2000.

April 14, 2000

(Bernhard F. Walter)

I hereby certify that the candidate has fulfilled the conditions of the Resolution and Regulations appropriate for the degree of Ph.D. in the University of St. Andrews and that the candidate is qualified to submit this thesis in application for that degree.

April 14, 2000

(Prof. Felix R. FitzRoy)

In submitting this thesis to the University of St. Andrews I understand that I am giving permission for it to be made available for use in accordance with the regulations of the University Library for the time being in force, subject to any copyright vested in the work not being affected thereby. I also understand that the title and abstract will be published, and that a copy of the work may be made and supplied to any *bona fide* library or research worker.

April 14, 2000

(Bernhard F. Walter)

Acknowledgements

I would like to thank my supervisors Prof. Felix R. FitzRoy and Dr. David G. McMillan for their guidance, aide and reassurance throughout the process of writing this thesis.

My thanks also go to Prof. Gavin. C. Reid (Postgraduate Co-ordinator) and the secretarial staff, who supported and helped me whenever required.

I would like to express my gratitude to the European Commission who awarded me a Marie Curie Fellowship (TMR Grant). Without this financial support this project would not have been feasible in its entirety.

The vast amount of data that was required for this research was kindly provided by the Office for National Statistics, the General Register Office for Scotland and The UK Meteorological Office.

Thanks are owed to Dr. C. Spix, Prof. R. Dockery, Prof. A. Pope III, Prof. J. Schwartz, Prof. M. W. Jones-Lee and Prof. A. Evans for their comments and advice at various stages of my research.

I am indebted to my friends Ulrich Wacker, Kris Towson, Dieter Weiss, Christoph Zaborowski, Frank Eisele, Gordian Kemen and Peter Hay for their tireless photocopying, proof-reading and encouragement.

I am grateful to my parents whose emotional and financial support throughout my student life has ultimately made this project possible. They have always been my most enthusiastic supporters.

Finally, words cannot express my thanks to my partner Hilary for putting up with me. With her limitless love and patience, she managed to be my cook, maid, secretary and editor!

TABLE OF CONTENTS

List of Illustrations

List of Tables

1. Introduction

When Henry Ford first introduced his Model T in 1908, the implications of motor vehicles on society were unimaginable, initiating a new era of personal transportation. Little doubt exists that road transportation has helped to guarantee the continuous growth and prosperity western economies have experienced over the last four decades.

The individual need for mobility and accessibility has caused rapid growth within this sector, resulting in an ever increasing number of licensed motor vehicles on roads. Indeed, the number of motor vehicles on British roads has more than doubled in the last three decades, with the class of private cars having increased by even more (122%).

These private vehicles are also used more frequently, accounting for two-thirds of all journeys made per person per year. Conversely, the number of trips by foot or bicycle has declined continuously (by about 25 per cent since 1975) and has now reached an all time low, representing just 2.8% and 0.5% respectively of the total number of trips made per person per year.

Considering the freight sector, more goods than ever before are being moved on British roads and carried longer distances. Using the road for freight transport allows flexible and fast delivery of goods to almost any place in the country, where railways often cannot compete.

However, in addition to these positive impacts, road transport also generates various negative effects. Such adverse effects comprise, for instance, accidents, noise pollution, air pollution, harm to health, crop damage and congestion.

Problems such as road congestion, noise pollution and air pollution affect a growing number of people, and have considerable impacts on health and welfare, as well as causing damage to buildings and the natural environment. These effects can in turn result in substantial material and intangible costs. The associated costs are mainly of an external nature, i.e. those who cause the damage (for example, the motorists contributing to air pollution) do not cover them, but are imposed on entire society instead.

Generally, such external costs are problematic to the economy, as they are not included in the market price, possibly leading to wrong decision-making and a wasting of scarce and vital resources such as clean air.

In July 1998, *A New Deal for Transport: Better for Everyone*, the Government's White Paper on the future of transport, was presented to Parliament by the Deputy Prime Minister. It set out the framework for a sustainable transport system which aims to reduce traffic growth, and to tackle the adverse impacts traffic can have on human health and the environment. This framework intends to provide choice for transport users, integration of transport services, and accountability to users and the nation. The following table illustrates a summary of the main areas where road traffic may give rise to external costs.

Table 1: Impact Areas of Road Traffic Associated with External Costs.

Air Pollution	Noise Pollution	Road Accidents	Land Use
• **Human Health**	• Housing Comfort	• Public Health	• Soil Sealing
• Flora/ Fauna	• Industry/ Production	• Material Damage	• Partition Effect
• Buildings/ Structure	• Tourism/ Recreation	• Soil Pollution	• Landscape
• Water Pollution	• Public Health		
• Soil Pollution			
• Climate			

This overview illustrates the manifold adverse impacts of road traffic, which in turn gives rise to external costs. Specifically human health is endangered through road traffic-induced air pollution, noise pollution and accidents. As marked in the above table, the focus of this study is exclusively on the health costs that can be attributed to air pollution generated by road traffic, an area which has particularly raised increasing awareness amongst individuals, pressure groups and governments over the recent decades.

Since motorists do not have to pay for these external costs of traffic-induced air pollution, they do not take them into account when deciding to use their motor vehicle and behave as if they do not exist. However, by including these external costs, the total benefit of some road trips may indeed be outweighed by the total costs (including external costs). Consequently, if the external costs of using the motor vehicle had to be considered by the motorist, these trips would not have been undertaken.

In attempting to avoid wasting scarce resources the authorities need to take action in order to ensure that these negative impacts of road transport are paid by the polluting motorists, i.e. to ensure the polluter-pays-principle holds. This process is generally known under the terminology of the 'internalisation of externalities', where a price is put on clean air. In doing so, the negative impacts of road transport are paid by the polluter. However, an important condition for such an environmental and transport policy is the knowledge about, first, the negative impacts of road transport (i.e. the adverse effects on human health) and second, their value in monetary terms. The final monetary quantification enables the respective authorities to conduct a thorough cost-benefit analysis.

Trying to quantify road traffic related health costs due to air pollution is not an easy task. The research procedure requires the interdisciplinary analyses and combination of the three domains of air pollution, epidemiology[1] and economics.

The first field includes the evaluation of the exposure of the population to specific air pollutants. Concentrating on the negative impacts of air pollution generated by road traffic, the respective share of total air pollution concentrations attributable to motor vehicles has to be established. Second, the relationship between the exposure to these air pollutants and the response or impact on human health must be assessed. Third, this exposure-response relationship can be combined with the exposure to specific air pollutants, the impact of air pollution on human health can be quantified in terms of additional cases of morbidity or premature deaths. Knowing the road traffic's share of pollution emissions, these number are then adjusted to obtain the respective numbers that are attributable to road traffic. Finally, these figures serve as the basis for the monetarisation of the health effects.

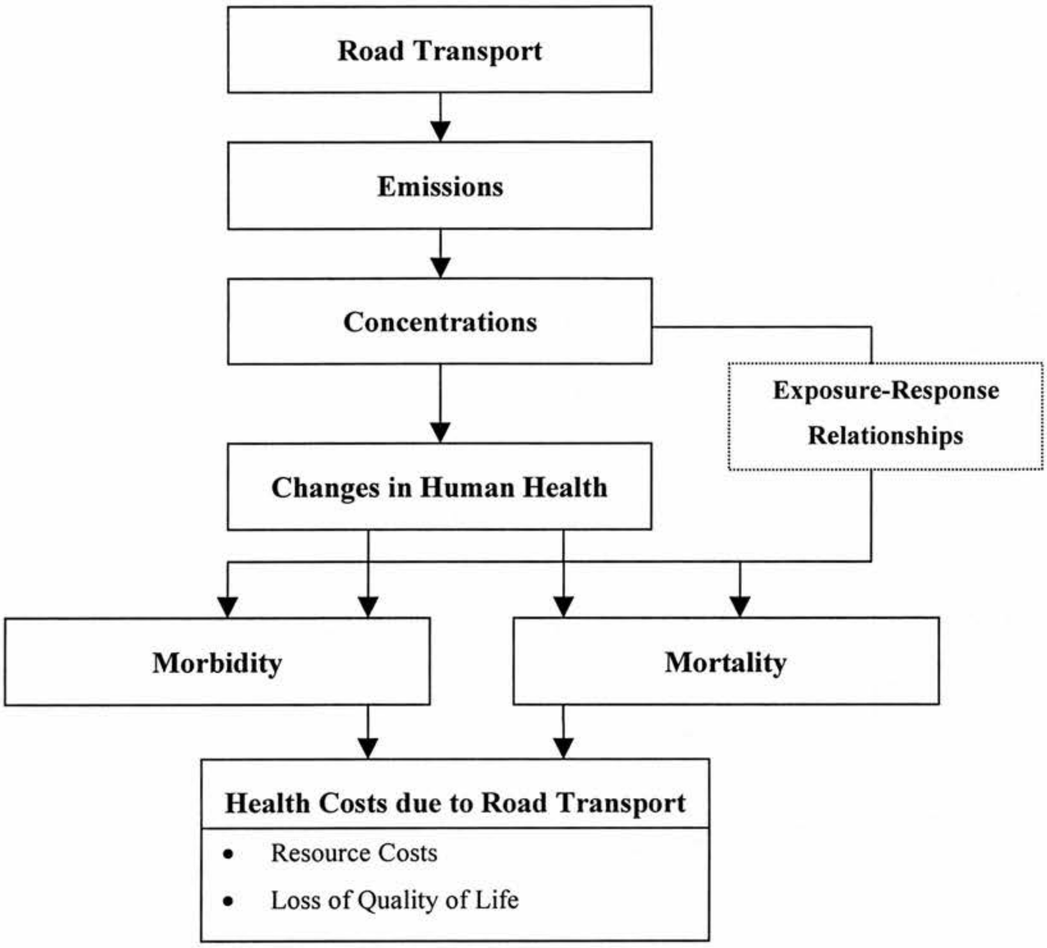
In the methodological framework of this study all three domains as described above, are applied when conducting the analysis of 14 major British cities.

The following figure illustrates the chain of interactions between air pollution and health costs.

---

[1] Following the World Health Organisation, epidemiolgy is the study of the analysis of the distribution of illnesses, physiological variables and social consequences of illnesses in human population groups, as well as factors influencing this distribution. See: WHO (1999), p. 102.

Figure 1: Chain of Interaction between Air Pollution and Health Costs.



Source: Sommer and Neuschwander (1996).

The described interactions indicate that emissions from road traffic result in an increase in the concentration of harmful substances (air pollutants) in ambient air, which in turn may have an adverse effect on human health. Such negative impacts on human health may lead to an increase in morbidity and/or mortality and hence generate additional costs. Such costs may be both in terms of resources (i.e. costs arising from medical treatment and days of work lost) and as deterioration in the quality of life (i.e. the suffering from people directly or indirectly affected by illness and premature death).

Following the described chain of interactions, the analysis for each of the cities included in this study is performed in a stepwise manner, starting with the estimation of exposure-response functions for a specific air pollutant. In general terms, an exposure-response function is the quantified relation between air pollution exposure and health outcome. In this study, these functions are obtained by econometric methods, analysing the day to day variations of counts of deaths and the levels of air pollution concentrations on the same days.

In contrast to similar studies that examined health costs associated with (traffic-related) air pollution, this study emphasises the estimation of the exposure-response functions using actual data from the respective cities included in the analysis. So far, the majority of epidemiological studies that have been undertaken in recent years, providing exposure-response functions, have been undertaken outside the United Kingdom, predominately in the United States of America. This raises the question of how well the results from these non-UK studies can be used to extrapolate to the UK, qualitatively and quantitatively. Consequently, when estimating the number of deaths brought forward due to the exposure of air pollution based on such non-UK exposure-response functions implies that conditions in the study country and the UK are very alike. However, although similarities do exists, one may still raise doubts whether they are similar enough, especially when considering the different climate and environmental conditions as well as differences in social behaviours regarding such issues like the people's overall health status, exercise etc.[2]

---

[2] See, for instance, Kleinman and Mautz (1996).

The structure of this study is as follows:

The following chapter provides a detail description of the road transport sector. Both the developments that occurred over the last four decades, and the current patterns in the road sector are illustrated, differentiating by passenger transport sector and freight transport sector. Further, special focus is made on the changes in both the private and the public transport sector respectively. Additionally, an international comparison, presenting some important statistics regarding the road transport sector will also be given. Finally, some key road traffic forecasts illustrating future developments in the road transport sector in Great Britain are provided.

Chapter 3 begins with a description of the relationship between the transport sector and energy consumption, as well as the association between the transport activities and the emission of various air pollutants, again specifically concentrating on the road sector. These road traffic-related air pollutants, comprising carbon monoxide, oxides of nitrogen, particular matter, sulphur dioxide, and ozone are discussed in detail. The chapter concludes with a brief overview of the current guidelines and air quality legislation, along with future emission targets for future air quality strategies for the UK.

Chapters 4 and 5 provides an in-depth discussion of the phenomenon of externalities. A historical overview is followed by the formal description of externalities, both analytically and graphically. Based on this, the topic of the external effects related to the transport sector is discussed, both from a theoretical and practical point of view. This is followed by the description of the interactions between the road transport sector and other sectors, which then leads to the analysis of the transport-related externalities calling for public intervention. The discussion then focuses on the issue of potential external benefits stemming from the road transport sector, i.e. whether positive external effects indeed exist and may counter-balance the negative external effects.

The quantification of health effects due to road traffic-related air pollution requires the economic valuation of the estimated changes in health status, i.e. in this study the number of premature deaths attributable to air pollution exposure. Chapter 6 discusses the main issues related to the procedure of placing a money value on premature mortality. Special focus lies on the discussion of potential objections against such an idea, as well as on methodological issues related with various conceptions on valuing human lives. From this discussion, it is then concluded that the so-called willingness to pay approach offers the most appropriate and systematic valuation approach at the current state. Results of empirical studies are presented before applying this approach to the specific case of air pollution. Finally, an air pollution related value band of preventing a statistical fatality, including a low, central, and high estimate is presented.

A summary of previous mortality time-series studies providing exposure-response functions for various air pollutants is presented in Chapter 7.

Chapters 8, 9 and 10 are concerned with the estimation of exposure-response functions for the main traffic-related air pollutants for all 14 British cities included in this study. In Chapter 8, descriptive statistics of the employed datasets, comprising mortality, pollution and climate data, are presented for each city. This is followed by the discussion of the methodological issues associated with the estimation of exposure-response functions in Chapter 9. For each air pollutant, a regression model is established and ultimately estimated. This estimation is undertaken for each air pollutant separately, allowing for specific adjustments if necessary. The results of the estimation procedure are then presented and discussed in Chapter 10.

Chapter 11 is concerned with the final calculation of the external costs that arise from premature deaths due to air pollution. Based on the exposure-response coefficients presented in Chapter 10, first, the calculations of the absolute numbers of deaths attributable to the exposure of air pollution per city examined are presented. Second, taking the share of air pollution concentration for which motor vehicles are responsible for, these numbers are then adjusted respectively. Finally, based on these calculations, the willingness to pay estimates for the value of a statistical life, as they have been derived in Chapter 6, are applied, resulting in final monetary values. These values are calculated for both the total numbers of deaths brought forward due to the exposure to air pollution, as well as for those attributable to road transport induced air pollution.

The final Chapter concludes this study, drawing in the many aspects discussed within this study. The key findings are summarised before touching upon possible policy implications and suggestions for future research.

2. The Road Transport Sector

2.1 Introduction

In recent times, it is almost impossible to imagine a society without transportation. Nearly any action in life involves some kind of transportation mode. People's professional life, as well as their private and social life is substantially formed by transport. Travelling by foot, by car, or by any other means in order to work, shop, see friends and spend leisure time is established as an essential part of life. Road transport and, hence, motor vehicles in particular, play a central role in the economy being the primary means of transportation in western countries including the UK. Furthermore, it reflects the conception of personal autonomy and mobility in society. Transport is central and somehow vital for the quality of life, as well as for economic well being, as it provides the necessary accessibility and mobility required in modern life.

However, road transport presents some kind of paradox. Besides the obvious fundamental benefits to commerce, recreation and daily social life, motor vehicles are at the same time a major source of various costs. Therefore, there is a growing concern that the putative benefits of a continuous growth in the transportation sector may actually be outweighed by a number of serious disadvantages. Some of them, such as congestion and the damaging effect to the environment and to human health are more obvious than others, such as the possible interruption to the frictionless functioning of an efficient and effective economy (e.g. caused by overcrowded and congested roads). Specifically, attention focused towards air pollution related to motor vehicle traffic has been growing dramatically over the past decades. Although the issue of environmental pollution has a rather long historical background, the fact that the road transport sector is increasingly becoming the main contributor to air pollution has increased interest into the subject of traffic related air pollution and its consequences. This is very much legitimised by the fact that emissions from exhaust pipes contain several substances, which in turn are said to have an adverse effect on human health if they come in sufficiently high concentrations.[1]

Unsurprisingly, it is generally agreed that the use of motor powered vehicles is causing far and away the largest amount of air pollution than any other single human activity.[2] Specifically, in urbanised areas the contribution of road transportation is most likely the main source of overall emissions, with an increasing tendency. Consequently, its contribution to overall emissions especially in larger cities is continually increasing. This is mainly due to a rather tremendous growth in the road transport sector, as well as the retrogression of other major pollution sources, such as domestic and light industrial coal burning, which previously made a large contribution to the total emission of pollutants. In 1960, for example, the share of energy consumption by final users in the industry sector was about 42 per cent, whereas the transportation sector was accounting for just over 16 per cent. This picture has changed dramatically since then. The transportation sector is now consuming more than one third of total energy used by final users, in comparison to about 20 per cent in the industry sector. The energy consumption of the domestic sector and other final users has been more or less constant over time.[3]

After recovering from two energy shocks in 1974 and 1979, which both significantly affected economic growth in general, and the transport sector in particular, most industrialised countries have seen a sustained economic growth,

---

[1] A comprehensive discussion on this subject is provided in the next chapter.
[2] See: OECD (1995a), p. 9.
[3] See: Department of Trade and Industry (1999), p. 14.

which is impressively reflected in the evolution of the transport sector. Considering the development in the freight transport sector as an indicator of general economic growth, it can easily be seen that both have experienced a similar steady positive trend; this is illustrated in the following table.

Table 2: Trends in GDP and the Freight Sector.

	Figures in 1998	Percentage change from 1988	Percentage change from 1978	Percentage change from 1960
GDP* (million pounds)	773,380	20.7	53.9	147.2
Goods moved (billion tonne km)	246 (160)	16.6 (23.1)	38.2 (60.0)	146.0 (226.5)
Goods lifted (million tones)	2,126 (1,727)	11.4 (12.0)	12.5 (14.9)	39.7 (42.6)

Source: DETR (1999), p. 181 & ONS (1999a), p. 11.

Note: Figures in brackets represent road sector.
(*): Figures are re-valued at 1985 prices.

The figures for the road freight sector, in particular, reflect a more than proportional rise over the last two decades for both the distances of goods moved (in tonne kilometres), as well as the total number of goods lifted (in tonnes). Overall road freight transport has risen by almost 50 per cent in the last 10 years, and over 60 per cent since 1978. Of all goods that have been lifted in 1998, more than 80 per cent were moved on roads. However, since the figures for goods moved has risen in line with the development of the Gross Domestic Product, the figures for goods lifted in million tonnes show a positive but significantly lower rise in the same time period. In other words, although the amount of goods (in weight) has not risen extraordinary, these goods are now moved over much larger distances.

In this chapter, a detailed description of the developments in the road transport sector will be given for both the passenger transport sector and the freight transport sector. Furthermore, a special focus will be made on the changes in both the private and the public transport sector respectively. The characteristics of the growth in traffic in the respective categories over the last three to four decades will be outlined. Further, the overall patterns of movement and the current situation in the road transport sector will also be illustrated together with possible reasons for the described changes in the transportation sector, such as changes in people's lifestyles. The next section will then give an international comparison, presenting some key statistics regarding the road transport sector. Finally, some key national road traffic forecasts for Great Britain will be presented to illustrate the future developments in the road transport sector.

2.2 The Passenger Transport Sector

2.2.1 Developments of Passenger Traffic

All figures regarding the transport sector clearly reflect that there has been a tremendous overall growth in road traffic. However, the distribution among categories of road transportation differs significantly. The following figure illustrates the trend in motor vehicle stocks for the different categories of motor vehicles. Figures for private cars, public transport vehicles (PSV)[4], and motorcycles, as well as the total number of licensed motor vehicles in the UK, are presented for the last four decades. The total number of motor vehicles is made up by the three categories mentioned above, plus the categories of other private vehicles (mainly light goods vehicles[5]), goods vehicles (mainly heavy

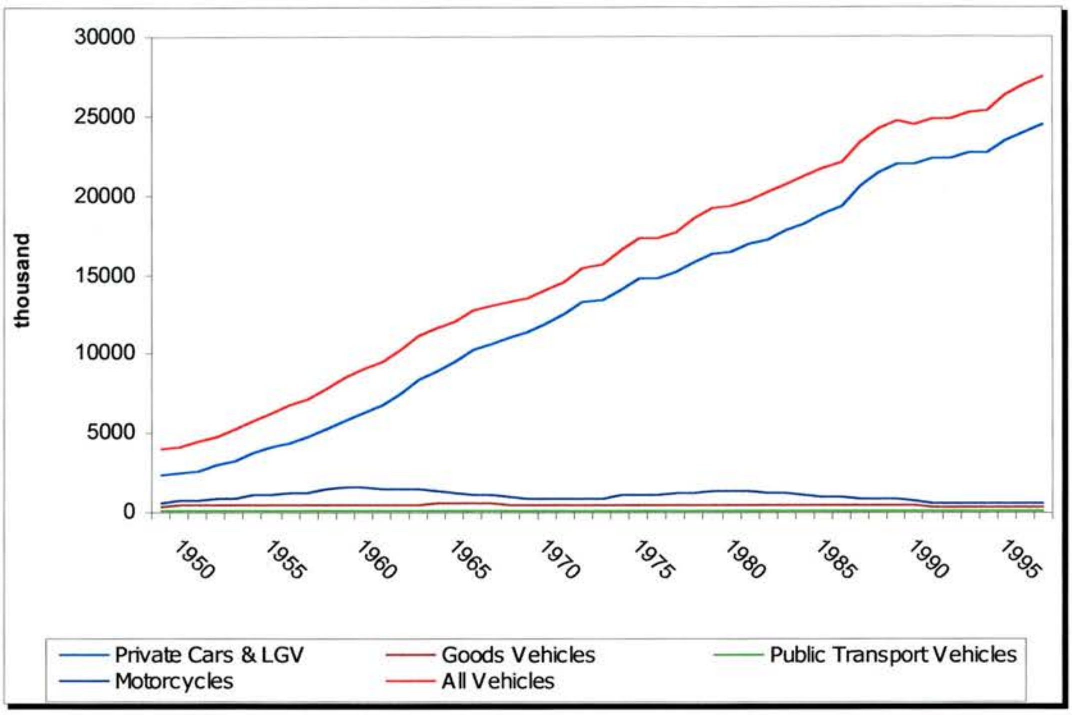
---

[4] Here, public transport vehicles include buses, coaches and taxis.

[5] The category of Light Goods Vehicles (LGV) or light vans includes all goods vehicles up to 3,500 kgs gross vehicle weight. The majority of this category are delivery vans of one type or another including all car delivery vans, transit vans, small pickup vans, three-wheeled goods vehicles, milk floats and pedestrian controlled motor vehicles.

goods vehicles[6]), special machines and special concessionaire, special vehicles, crown and exempt vehicles, and other vehicles. Of these additional categories, only the two categories of light goods vehicles and (heavy) goods vehicles are included for comparative reasons in the figure below. These two categories are also described in greater detail in the next section.

Figure 2: Motor Vehicles Currently Licensed: 1950-1998.



Source: DETR (1999), pp. 76 & 183 and British Road Foundation (1998), p. 2.

The above figure illustrates the tremendous development that has occurred in the road transport sector over the last decades. The total number of licensed motor vehicles in Great Britain at the end of 1998 has now reached an all time high of over 27 million. This is an increase of about 3 per cent on the previous year and represents a rise of almost 20 per cent in the last 10 years.[7] Moreover, since

[6] The category of Heavy Goods Vehicles (HGV) includes all goods vehicles over 3,500 kgs gross vehicle weight. This group can again be subdivided in rigid goods vehicles and articulated goods vehicles. The latter category describes the specification when a goods vehicles is travelling with one or more axles raised from the road (sleeping axles or hobos); here the number of axles on the road determine the classification not the total number of axles.

[7] However, the actual number of all motor vehicles may be underestimated by up to 10 per cent due to untaxed vehicles. See: British Road Foundation (1998), p. 2.

1950, the number of all vehicles currently licensed increased by about 590 per cent.[8]

The above figure also demonstrates quite clearly that this trend has mainly been driven by the development of the private car sector, which has the largest proportion of vehicles on the road. The population of private motor cars has increased by 970 per cent from 2,218,000 in 1950 to 24,477,000 in 1998, and hence accounts for almost 90 per cent of all motor vehicle traffic. In 1998, the number of private cars and light goods vehicles (LGVs) registered for the first time reached 2,367,900, which is considerably higher than in previous years and almost reached the all-time high numbers of 1989.[9]

In contrast to the developments regarding privately owned motor vehicles, the number of public transport vehicles has been relatively constant with some fluctuations, and has ultimately dropped to about 80,000 in 1998.[10] The category of motorcycles has experienced a similar development. Their number reached a maximum in the early 1960s and a second boom in the early 1980s, but has subsequently dropped around 684,000 licensed motorbikes. However, figures indicate an increasing trend from 1995 on.[11]

Examining traffic developments in terms of total distances travelled, a parallel trend can be observed. The following table illustrates the road traffic composition in 1998, and its development in the last 10 and 20 years respectively. The two categories of light goods vehicles and heavy goods vehicles are also included for comparison reasons and are described in more detail in the next section of this chapter.

---

[8] See: Department of the Environment, Transport and the Regions (1999), p. 183.
[9] See: Ibid, p. 184.
[10] See: Ibid, p. 76.
[11] See: Ibid, p. 183.

Table 3: Road Traffic Composition in Great Britain.

Vehicle type	Billion vehicle kilometres	Proportion of total	Percentage change in mileage since 1988	Percentage change in mileage since 1978
Cars & Taxis	375.9	81.82	23.1	85.7
Motorcycles	4.0	0.87	-33.33	-34.45
Buses & Coaches	5.0	1.08	16.2	51.5
Light Goods Vehicles	42.5	9.25	32.8	88.0
Heavy Goods Vehicles	32.1	6.98	15.0	45.2
Total	459.4	100	22.3	79.1

Source: DETR (1999), pp. 96 & 182.

The figures presented in the above table give a clear picture of the development in the road transport sector, and are similar to the overall changes in this sector. Thus, not only has the actual number of motor vehicles increased tremendously, but these vehicles have also been used more intensively.

Drawing from the above table, the given number for body type vehicles once again confirms the overall developments in the road transport sector. With a share of almost 82 per cent of all travelled kilometres on British roads, body type cars are without doubt the most significant mode of road transportation. Not surprisingly, the increase in kilometres travelled per body type car has also been greater than the numbers for total road traffic. However, an even more impressive change has taken place within the category of light goods vehicles. This category has experienced an increase in vehicle kilometres, which is significantly above the average during this last decade, whereas while the category of HGVs has shown a positive trend, it is well below the average. A more detailed discussion of the developments in the freight transport sector will be given in a later section of this chapter. Finally, drawing from the presented figures above, the trend for buses and coaches is positive but far below the overall average changes experienced in the road transport sector.

Looking at the development of the kilometres driven by motorcycles, a distinct cyclical trend can be observed, with highs in the early 1960s and early 1980s, which complies with the changes in the absolute numbers of this category, as described above.[12] The current number of motorcycles, which has fallen remarkably over the last two decades, has almost reached an all-time low.[13] The following table illustrates recent historic traffic growth, measured in per cent per annum for cars, passenger service vehicles (PSV), and for total road traffic.

Table 4: Annual Traffic Growth: 1976-1996.

	Cars	PSVs	Total
1976 - 1986	3.3	1.2	3.0
1986 - 1996	3.2	2.6	3.3

Source: http://www.roads.detr.gov.uk/roadnetwork/nrpd/heta2/nrtf97/nrtf05.htm; table C.

The presented figures confirm the incomparable development with respect to private passenger cars. Between 1976 and 1986, car traffic has grown at an annual rate of 3.3 per cent. Hence, this annual growth rate exceeds even the high figures for total traffic.

Passenger service traffic on the other hand, shows a rather modest annual growth rate of 1.2 per cent, which again reflects the overall trend away from public transport towards private passenger traffic. The figures for 1986 through 1996

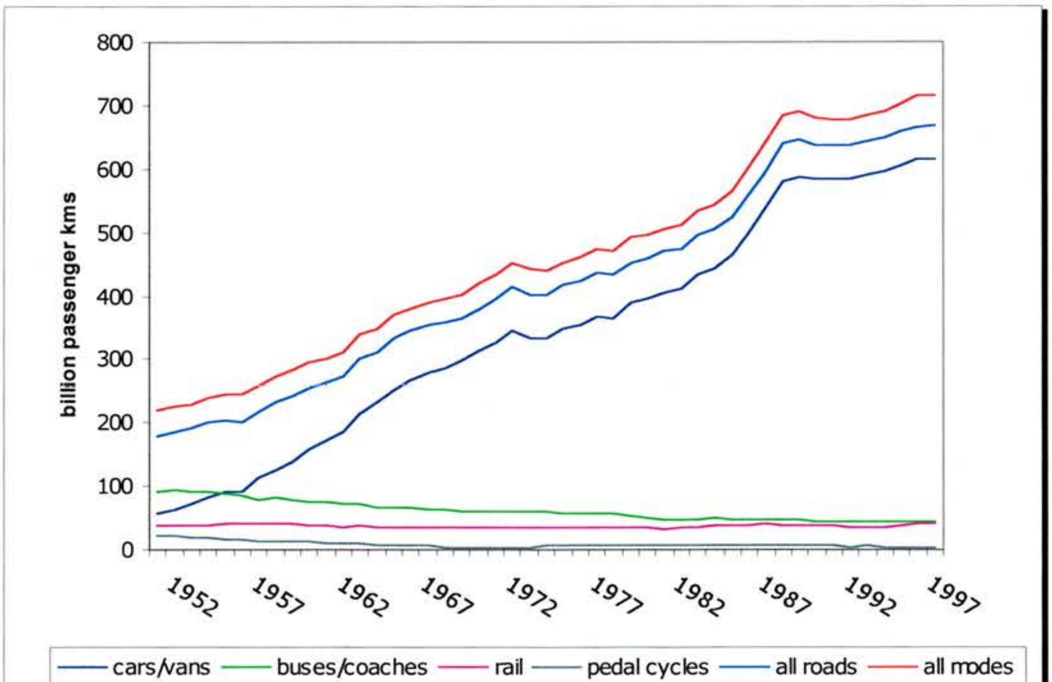
[12] Total passenger kilometres (excluding rail and air travel) has gone up by about 20 per cent in the last 10 years to a total of about 670 billion passenger kilometres per year. More than 85 per cent of this amount was accounted for by cars, vans, and taxis with a share of about 620 billion passenger kilometres. See: Department of the Environment, Transport and the Regions (1999), p. 179.

[13] See: Ibid, p. 39.

show a small decline in annual car traffic growth to 3.2 per cent per year, whereas the respective figures for total traffic increased to 3.3 per cent annually. This may be caused by an increase of annual growth rates of passenger service traffic during these years by up to 2.6 per cent.[14]

The numbers for passenger transport are in line with the development in road traffic in terms of kilometres travelled per vehicle. The numbers for passenger transport, which are measured in passenger kilometres, is derived from the traffic series (vehicle kilometres) and average occupancy rates (persons per vehicle) from the National Travel Surveys.[15] The following figure shows the development over the last 45 years.

Figure 3: Passenger Transport by Mode: 1952-1997.



Source: DETR (1999), p. 179.

[14] An even more significant influence on total annual traffic growth in theses years can be found by looking at the respective developments in annual traffic of light vehicle goods. This will be described in more detail in a later section.

[15] See: Department of the Environment, Transport and the Regions (1998a), p. 27.

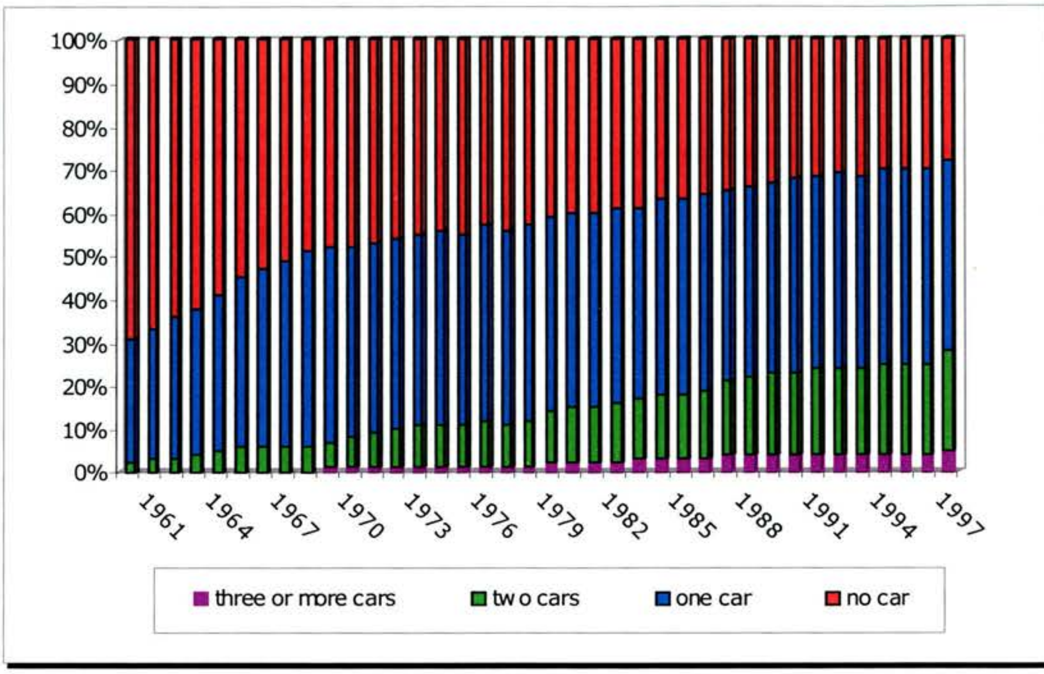
Whereas in 1952 the three main modes of passenger travel, i.e. bus/coach, cars and railways, are almost evenly distributed, the sector of body type cars has experienced a seemingly unstoppable growth, resulting in a share of 86 per cent of total passenger travel in 1998. Thus, private road transport is clearly the most dominant mode of total passenger travel accounting for 93 per cent of all passenger travel in 1998, leaving just 6 per cent for railways as a mode of travel.[16]

Underlining the tremendous developments in the road transport sector over the recent years, there has also been a significant shift in the accessibility to private cars. Whereas in 1960 more than 70 per cent of all British households had no regular use of a motor vehicle, this number has been constantly falling over time. In 1998, 70 per cent of all households in Great Britain had access to at least one car, whereby 25 per cent had a regular use of two cars or more. Since the 1970s, this is an increase of about 35 per cent.

The following figure illustrates the development of the accessibility of households to motor vehicles over the last three decades.

[16] The remainder of one per cent represents air passenger travel. See: Department of the Environment, Transport and the Regions (1999), p. 39.

Figure 4: Households with Regular Access to Motor Vehicles.



Source: DETR (1999), pp. 83 & 179.

The absolute number of households in the United Kingdom has increased by about 45 per cent since 1960 up to 23.6 million in 1998.[17] This has occurred despite the fact that the population of the UK has increased by 'only' 17.8 per cent during the same time period, between 1961 and 1998 to reach 62,244 thousand.[18]

One of the main reasons for this development is a growing tendency in society of a continuous trend for diminishing sizes of households. The average size of households in Great Britain has fallen by about 23 per cent in the last 30 years to 2.4 people living in a household in 1998. The proportion of 'traditional' households comprising a two-parent family with dependent children has fallen from 38 per cent in 1991 to just 23 per cent in 1998. The number of households consisting of a lone parent with dependant children tripled to 7 per cent of all

[17] See: Office for National Statistics (1999a), p. 42.
[18] See: Ibid, p. 30.

British households in 1998. A similarly increase can be observed for one-person households.[19]

The described trends give a clear image of the development, which has taken place in the transportation sector over the last decades. By breaking down the numbers presented in the previous section, it is possible to describe the changes happening at the level of individual's behaviour. The resulting figures for the individual use of private vehicles, which will be derived in the following section, lie predominantly in line with the developments described hitherto.

2.2.2 Current Patterns of Passenger Traffic

As mentioned in the previous section, the current patterns of passenger traffic in Great Britain have changed significantly over the last few decades. Both the total numbers of private motor vehicles as well as the total length of distances travelled by private motor vehicles have seen a tremendous increase.

This section describes the current patterns of private passenger traffic in Great Britain as observed in the year 1998. In particular, overall figures will be broken down according to various subcategories, in order to analyse certain development and behaviour patterns in the transportation sector.

Starting with the distances travelled, individuals covered, on average, 6,728 miles per year within the UK in 1995/98, which is an increase of over 27 per cent since 1985/86. Travelling by car and van (either as driver or passenger) has accounted for 82 per cent of this mileage, which is about 5,517 miles per person

[19] See: Office for National Statistics (1999a), pp. 42-44.

per year.[20] In comparison, the transport mode accounting for the next largest distance travelled, is the combination of buses in London, other local buses, and non-local buses with 344 miles per person per year, followed by surface rail with 290 miles. While the travelled distance by car alone has increased by 37 per cent since 1985/86, the numbers for other local buses and non-local buses have fallen by 22 and 15 per cent respectively. For London-buses and surface rail, a rather small increase of 5 and 1 per cent respectively was experienced over the same time period.[21]

The following table illustrates the distribution of travelling undertaken per person measured in absolute numbers of journeys, as well as journey distance in miles, with the latter is reported in brackets. The relevant figures, which are taken from the 1995/98 National Travel Survey, are also broken down by purpose of the journey.

[20] See: Department of the Environment, Transport and the Regions (1999), p. 39. Interestingly, the distance travelled by taxi or minicab increased by about 5 per cent since 1985/86 and more than 200 per cent since 1975/76.

[21] The overall change for all modes of distance travelled has been 25 per cent since 1985/86.

Table 5: Distribution of Travelling by Mode and Purpose.

	Walk	Bicycle	Car/ Van	London buses	Other local bus	Non-local bus	Surf-ace rail	All modes†
Commuting	20 (14)	6 (14)	116 (1,063)	3 (12)	9 (46)	N/A (2)	8 (126)	166 (1,341)
Business	4 (3)	N/A (1)	29 (581)	N/A (1)	N/A (1)	N/A (1)	1 (39)	37 (681)
Education	55 (28)	1 (1)	47 (159)	3 (9)	9 (40)	N/A (3)	1 (14)	119 (285)
Shopping	69 (38)	2 (3)	126 (710)	5 (13)	15 (58)	N/A (2)	3 (25)	222 (860)
other personal business	40 (20)	1 (2)	133 (751)	12 (5)	5 (19)	N/A (3)	2 (36)	187 (844)
visiting friends	46 (25)	2 (4)	122 (1,233)	2 (7)	7 (26)	N/A (20)	6 (49)	187 (1,400)
social/ entertain-ment	10 (7)	1 (4)	45 (380)	1 (2)	2 (8)	N/A (7)	1 (18)	62 (454)
Holidays/ day trip	2 (2)	2 (8)	22 (632)	N/A (1)	N/A (4)	1 (60)	1 (52)	29 (817)
Other, including just walk	42 (32)	N/A (N/A)	1 (12)	N/A (N/A)	N/A (N/A)	N/A (N/A)	N/A (1)	44 (47)
All purposes	288 (168)	16 (38)	643 (5,524)	15 (51)	47 (202)	2 (98)	16 (370)	1,051 (6,728)

Source: DETR (1999), pp. 40-41.

Note: (†): The figure for all modes includes some additional modes that are not explicitly mentioned here, such as motorcycles, London Transport Underground (LTU) and other private and public transport.

Following the above figures of the 1995/98 National Travel Survey, out of 1,051 journeys made per person per year, 643 (71 per cent) were made by car or van (as driver or passenger). The most popular journeys are those for other personal business (133), closely followed by shopping (126), trips to visit friends (122), and journeys made to commute to and from work (116). In terms of journey length, cars and vans are used to cover a total 5,524 miles per year per person. Here, the longest distances are travelled to visit friends (1,233 miles) and to commute for work reasons (1,063 miles).

In contrast, public transport is being significantly less used to travel. With 47 journeys per person per year, the transport mode of so-called other local buses (excluding London) has been used most frequently. The number of journeys by buses in London and bicycle journeys was 16 journeys per person per year. Non-local buses were used to make 2 journeys per year, and finally surface rail accounted for 17 journeys per person per year. Surface rail is mainly used for commuting purposes and to visit friends, whereas buses are more commonly used for shopping, commuting and education reasons. Interestingly, bicycles are very rarely used and are well below walking, both in terms of absolute numbers of journeys as well as the journey distance per person per year.

The figures show that the number of journeys made per week are more or less evenly divided between the groups of the three major purposes of travel (i.e. business related journeys including commuting, shopping related journeys, and journeys for other private purposes including holidays). However, considering weekly mileage by journey purposes, leisure journeys on average are longer than those for work or education reasons.[22] Consequently, leisure (including personal business) is the most important type of journey measured by total weekly distance. These are also the trips most likely to be made by car. In contrast to this, the number of journeys made by rail is comparably small, with the exception of journeys for work purposes. The average length of a rail trip is

[22] In 1996, about 70 per cent of people working in Great Britain travelled to work by car, whereas just 13 per cent used public transport to get to their work.

about 23.5 miles compared to 7.7 miles, on average, driven on the road and about 3.7 miles as a bus passenger.[23]

An interesting point to be noted in this relation is concerned with the significant changes in people's habits. One such change is the dramatic way people shop, i.e. a shift towards out-of-town shopping can be observed over the recent years. The growth of out-of-town stores is rather impressive, between 1960 and 1981 just under 15 per cent of all new retail floor space opened was at out-of-town sites, this number increased between 1992 and 1996 to more than 50 per cent.

According to The Royal Commission on Environmental Pollution (1994), there has been a significant rise in total retail sales made in out-of-town locations from just 5 per cent in 1980 to a rather impressive figure of over 37 per cent in 1992.[24] While DIY stores are traditionally located outside the city centre, out-of-town locations are becoming an increasing importance for companies selling electrical goods, clothing, sports equipment, footwear and office supplies. The creation of huge shopping centres in the open countryside is attracting more and more people. These extensive shopping malls usually contain several hundreds of shops including grocery stores.

Shopping centres are no longer considered as pure shopping places anymore, but are rather considered as 'multi-centres' which combine a broad range of entertainment including cinemas, restaurants, bars and other amusements. The intention is to create the impression that shopping is supposed to be pleasure rather than a tiresome duty. Being located outside busy city centres, the out-of-town shopping places offer putative advantages in comparisons to shops situated

---

[23] See: Department of the Environment, Transport and the Regions (1998a).
[24] See: The Royal Commission of Environmental Pollution (1994), p. 16.

in inner city areas. However, being located outside city centres it is necessary for people to be mobile, in terms of having direct access to private transportation.

The results of several surveys cited in the Royal Commission on Environmental Pollution (1994) show that up to 98 per cent of all sales of out-of-town shopping centres are accounted for by customers travelling by car. The studies also revealed that the respective customers usually live in a catchment area up to 40 minutes driving time. Assuming that most of the journey time motorways are being used, this can be a distance of 40 miles and more.[25]

The dependency on motor vehicles has increased in a similar manner as the share of out-of-town shopping sites have been increased, since they are too far away from residential neighbourhoods to reach by foot or bike and in such remote areas that is usually not possible to get there by public transportation. Therefore, customers need to have direct access to private motor vehicles in order to take up the putative advantages of out-of-town shopping. As a direct effect of this development, many shops at central inner city locations that may be easily reached by foot or public transportation face the problem that they become less profitable, less able to compete and sometimes forced to close down. Although it seems rather difficult to establish a direct link between the actual number of vehicles possessed by private households and the location of shops, there is certainly a relationship with the number of journeys per person per year, as well as for the average distance travelled per journey per person per year.

---

[25] See: The Royal Commission of Environmental Pollution (1994), p. 16.

Having described the current composition of travel modes, the large disproportion in the expenditure pattern of private households is not surprising. According to the Family Expenditure Survey, British households, on average, spent 328.80 pounds a week. The following table gives an overview on the components of the household expenditure:[26]

Table 6: Components of Weekly Household Expenditure.

Commodity or Service	**Average weekly expenditure per household (£)**	**Percentage of total expenditure**
Food & non-alcoholic drinks	55.90	17.00
Housing (net)	51.50	15.67
Motoring	46.60	14.18
Leisure services	38.80	11.80
Household goods	26.90	8.19
Clothing & Footwear	20.00	6.09
Household services	17.90	5.45
Leisure goods	16.30	4.97
Alcoholic drink	13.30	4.05
Fuel & Power	12.70	3.86
Personal goods & services	12.50	3.80
Fares & other travel costs	8.10	2.47
Tobacco	6.10	1.86
Miscellaneous	2.00	0.61

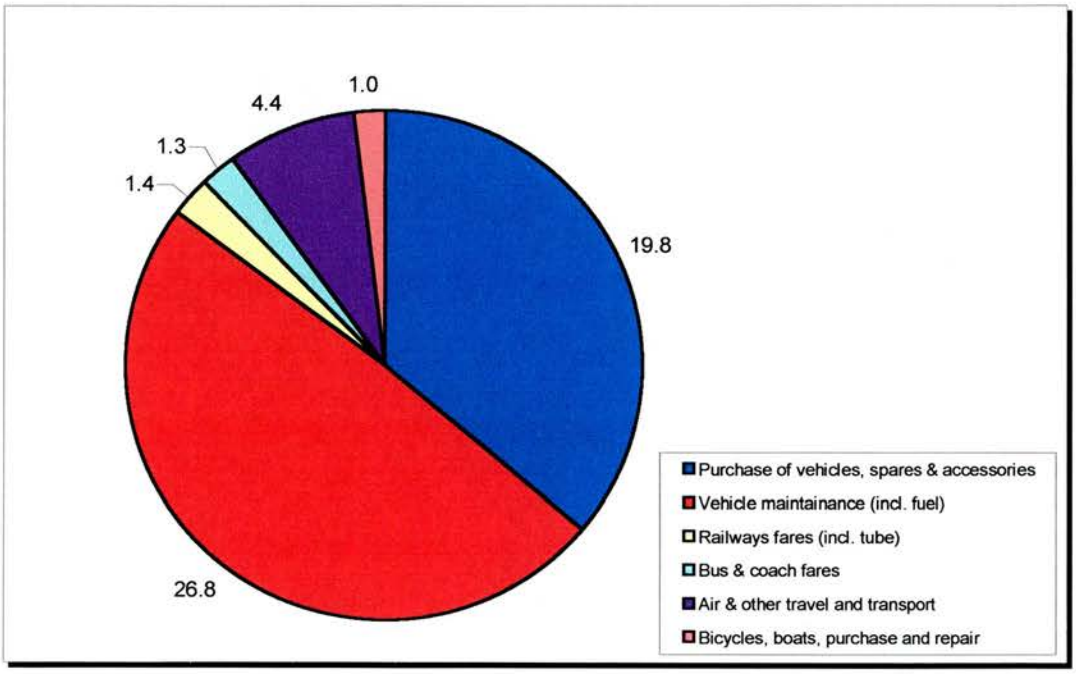
Source: ONS (1998a), pp. 102-112.

As it can be seen, the amount weekly spent on motoring is the third largest component accounting for over 14 per cent of the total expenditure; households only spent more on food and non-alcoholic drinks and housing per week.[27] On the other side, only 8.10 pounds per week are being spent on fares and other travel costs, including air transport. This is about 2.5 per cent of all household expenditures, and only expenditures for tobacco and the miscellaneous

[26] See: Office for National Statistics (1998a), pp. 102-112.
[27] Motoring costs contain all costs for purchasing cars, vans and motorcycles, as well as the money spent on spares and accessories, car and van repairs and servicing, motor vehicle insurance and taxation and petrol, diesel and other motor oils, and other motoring costs.

component are lower. Separating the category of fares and other travel costs, one can deduce that British households spent 2.67 pounds per week on public transportation, including bus, coach, rail, and tube fares.[28] The following figure illustrates a more detailed overview of the various sub-components of the expenditures related to the transport sector.

Figure 5: Components of Weekly Household Expenditure on Transportation (in £).



Source: ONS (1998a), pp. 109-110.

The chart in the figure above clearly demonstrates the disproportionate distribution of the total transport expenditures with the largest portion spent on the maintenance of private motor vehicles (49%), closely followed by expenditures on the purchase of motor vehicles, spares and accessories (36.2%). In contrast to these expenditures relating to the private transport sector, the proportion of money spent on public transport is small. The two categories of fares for rail and tube, and bus and coaches together, represent just under 5 per

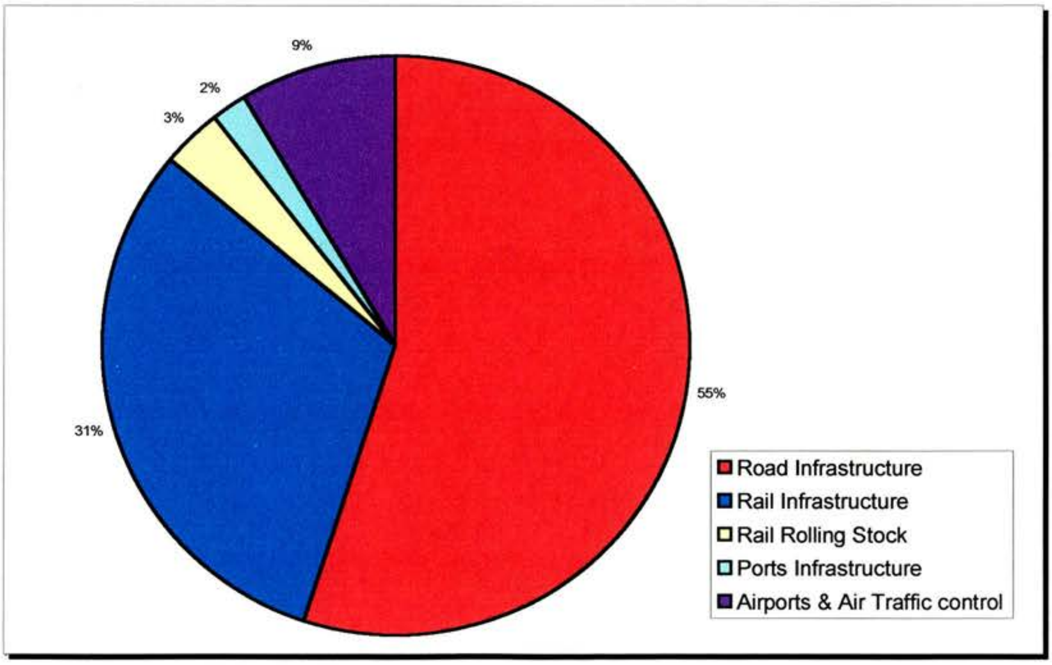
[28] This is just over 6 per cent of the money spent for total motoring and travel and less than one per cent (0.86 %) of the average weekly expenditure per household.

cent of the total amount on transport, and symbolises the relatively minor role that public transport plays in people's live nowadays.

In short, after allowing for inflation, British household expenditures on transport doubled between 1971 and 1997. However, expenditure in real terms on bus and coach travel was 40 per cent lower in 1997 than 1971. This increase in household expenditures corresponds to an increase in real household disposable income per head, which has nearly doubled between 1971 and 1997.[29]

Similar to the expenditures for private transportation, the picture for public investments in various transport modes shows significant differences. The following figure shows the investments made in transport in 1996/97 in Great Britain.

Figure 6: Investment in Transportation: 1996/97.



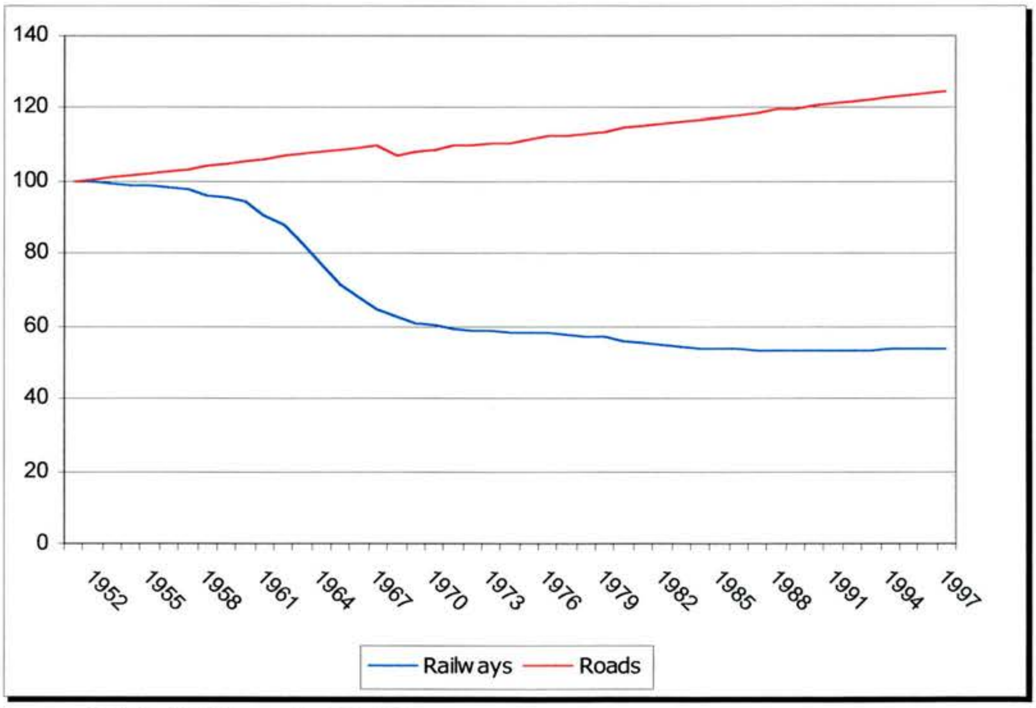
Source: DETR (1999), p. 44.

[29] In the same time period, expenditure on aircraft travel increased eightfold. See: Office for National Statistics (1999a), pp. 87 & 206.

In 1996/1997 £7.429 billion were invested in transport (excluding road vehicles and companies operating at ports). The largest part of total investment is accounted for by investments for road infrastructure with £4.119 billion in 1996/97. This amount is almost twice the investment made in rail infrastructure (£2.303 billion). The remaining amount of £1.037 billion is accounted for by investments in rail rolling stock, ports and airports.[30] This difference of public expenditure between private and public transport modes is also reflected in the developments in the lengths of the respective modes.

The following figure illustrates the changes in the length of roads versus the length of railways.

Figure 7: Change in Roads & Railways Lengths: Index 1952.



Source: DETR (1999), pp. 186 & 190.

[30] See: Department of the Environment, Transport and the Regions (1998b), p. 44.

The total length of National Rail route once covered over 30,000 kilometres in the 1950s and 1960s, has now almost halved to over 16,000 kilometres in 1997/1998. On the other hand, road lengths have risen in the same time period by almost 25 per cent from less than 300,000 kilometres in 1952 to just over 370,000 kilometres in 1998.

Finally, looking at the development of passenger transport prices may also provide a reason why people use public transportation less. While the cost of travel has roughly kept pace with inflation there can be some differences between the various transport modes observed. Whereas total motoring costs have increased slightly less than the general inflation rate, the fares for rail, bus, and coach have in fact risen by more than inflation. The following table shows this tendency since 1981 and also provides the respective retail price index for comparison.

Table 7: Developments in Transportation Prices.

	1981	**1988**	**1991**	**1996**	**1998**
Motoring costs	100	131	163	205	224
Bus & Coach fares	100	139	198	261	278
Rail fares	100	137	201	262	278
All fares	100	135	186	229	244
Retail Price Index	100	137	185	214	227

Source: ONS (1999b), p. 206.

2.3 Description of the Freight Transport Sector

2.3.1 Developments in the Freight Transport Sector

As mentioned briefly in the introductory part of this chapter, a remarkable development has occurred in the sector of freight transport in recent years. One criteria used to measure freight transport is the weight of goods moved. The following table provides some data on the change of various freight transport modes over the last two decades.

Table 8: Freight Volumes by Modes.

Transport Mode	Goods lifted in million tonnes in 1998	Percentage change from 1988	Percentage change from 1978
Road	1,727	12	21
Rail	102	- 27	- 40
Water	149	4	22
Pipeline	148	78	97
Total	2,126	11	19

Source: DETR (1999), p. 181.

The figures for both pipeline and waterborne freight transport are not actually comparable since the changes are largely due to differences in coverage.[31] Hence, the two transport modes of most interest in this study are road and rail freight transport.

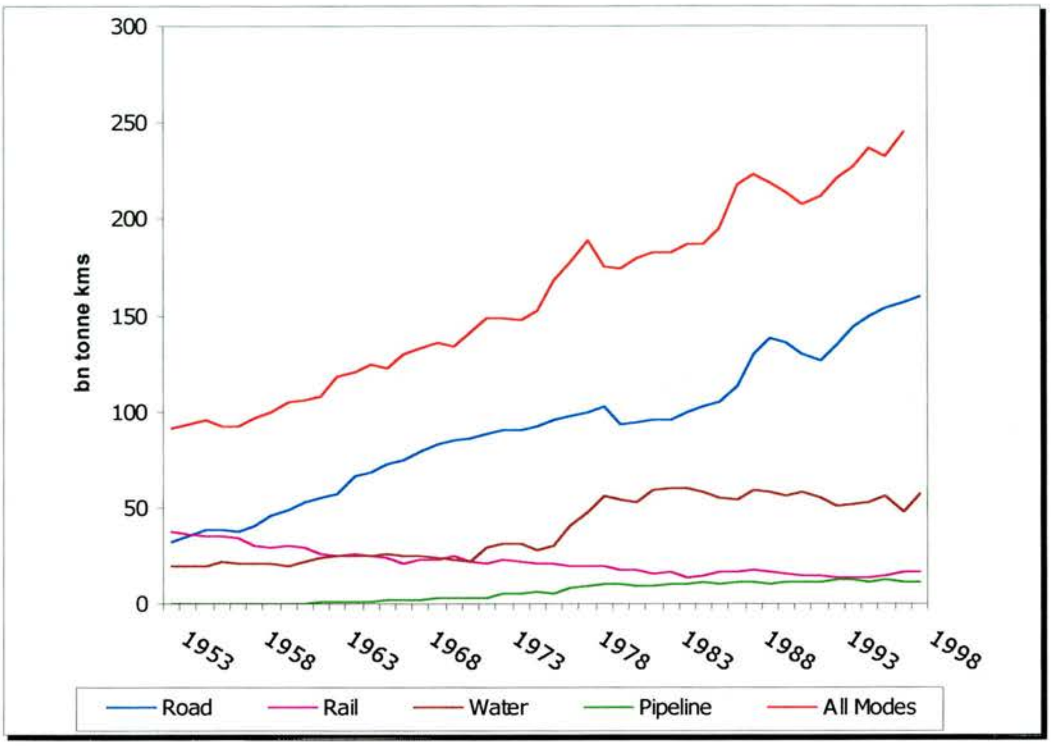
[31] For the latter, which accounts for nearly one quarter of freight moved, there exist often no other practical alternative, such as the traffic with islands and off-shore installations (e.g. petroleum products). See: Department of the Environment, Transport and the Regions (1998b), p. 177.

By comparing the data for these two modes, a clear trend is evident. Whereas the amount of freight lifted on the road has been increasing steadily over the past two decades, the rail freight transport has been significantly reduced over the same time period. In 1998, a total of 2,126 million tonnes of goods were lifted, whereby road freight transport accounted for 1,727 million tonnes or 81 per cent. In comparison, rail transport has only been used to lift 102 million tonnes of goods, which accounts for less than 5 per cent of total goods lifted by all transport modes.

Further, the overall increase regarding all modes has been about 19 per cent between 1978 and 1998. In the same time period, the amount of goods lifted on the road has increased by over 21 per cent whereas the respective figure for rail freight transport fell by 40 per cent.

However, using the weight of goods moved as an indicator for the amount of freight transport may produce a misleading picture since new technology, for instance, may allow a shift from heavy bulky goods to lighter high value manufactured goods. Consequently, it seems appropriate to include also the amount of goods moved, which is achieved by multiplying the weight by the average length of haul. The following figure shows the development in the freight transport sector by mode expressed in billion tonne kilometres.

Figure 8: Domestic Freight Transportation by Mode: 1953-1998.

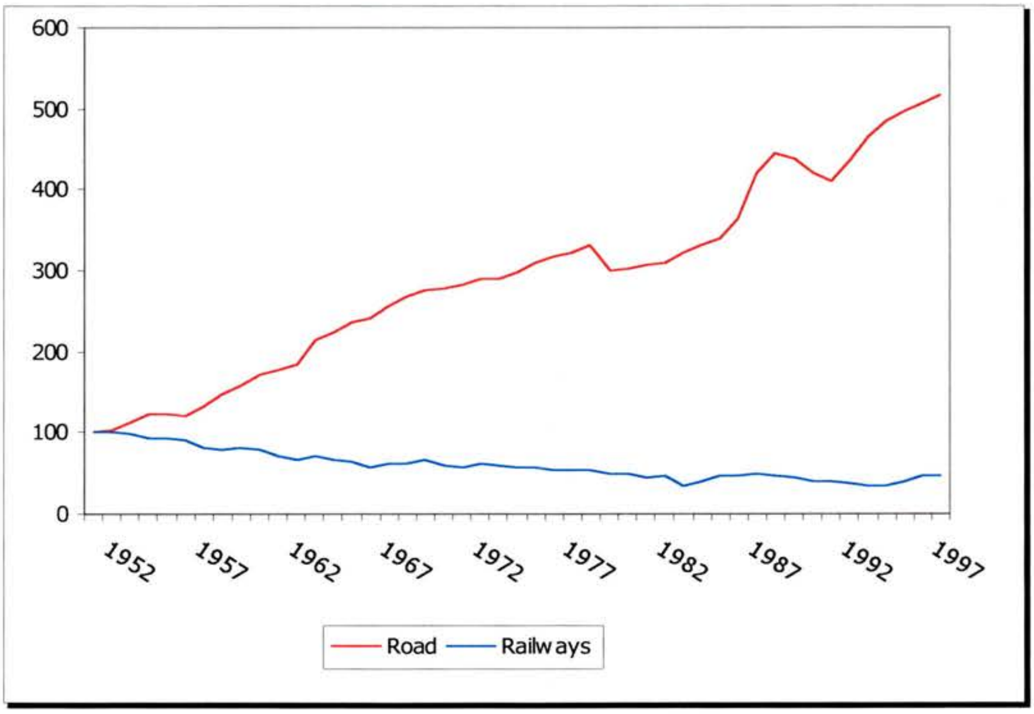


Source: DETR (1999), p. 180.

The figure clearly indicates a significant shift towards road freight transportation over time. Between 1988 and 1998, the number of goods moved on the road increased by almost 40 per cent, and by more than 60 per cent since 1978. As the overall change for all modes has been 26 per cent and 46 per cent for the same time period respectively, it seems obvious that road freight transport has virtually accounted for all the increase in total goods moved over the last two decades. Hence, by looking particularly at the developments of the rail freight sector, a picture of a sustaining decline can be observed. Over the last four decades, the amount of goods moved on rail has more than halved, accounting now for just under 7 per cent.

Concentrating especially on the two modes of road freight and rail freight, the situation is fairly evident. Road freight transport has strengthened and increased its dominant role in freight traffic, whereas rail freight traffic has lost its share on overall freight transport continuously. The following figure shows the percentage change of goods moved by these two modes over time.

Figure 9: Percentage Change of Goods Moved on Road & Railways: Index 1952.



Source: DETR (1999), p. 181.

It is evident that whereas the figures for goods moved on the road, measured in billion tonne kilometres, has increased drastically over the last four decades or so, fewer goods have been moved using railways during the same time. Since 1985, the road freight sector has experienced an increase of goods moved by 55 per cent. In contrast, although there have been some fluctuations over the same time period, the rail freight sectors remained more or less at the same level as it was in 1980.[32]

---

[32] The current figures are presented in the following section.

Examining the amount of goods lifted, measured in million tonnes, a more or less identical picture occurs, with an increase in the road sector and a continuous fall in the railway sector. In fact, in 1998, more goods have been lifted using water (149 million tonnes) and pipelines (148 million tonnes) than have been on railways (102 million tonnes). Freight transport on roads accounts, with 1,727 million tonnes, by far for the largest portion of goods lifted.[33]

2.3.2 Current Patterns of Domestic Freight Transport

The current picture in terms of domestic freight transport is fairly one-sided. Road freight transport is by far the most dominant means of freight transportation in Great Britain.

The following table shows the distribution of two measures of freight transport in 1998. First, current figures of freight activities measured in terms of the weight of goods (tonnes) handled (i.e. goods lifted) is presented. However, since this measures does not take account of the actual distance these goods are carried, a second measure is introduced, i.e. goods moved. This measures is the weight of the load multiplied by the distance it is carried and is seens as a 'better' measure of the work done by goods vehicles, especially heavy goods vehicles.

[33] See: Department of the Environment, Transport and the Regions (1999), pp. 42 & 182.

Table 9: Domestic Freight Transport by Mode.

	ROAD	RAIL	WATER	PIPELINE	TOTAL
Goods lifted (million tonnes)	1,727 (81)	102 (5)	149 (7)	148 (7)	2,126
Goods moved (billion tonne kilometres)	159.5 (65)	17.4 (7)	57.2 (23)	11.2 (5)	245.3

Source: DETR (1999), pp. 48-49.

Note: Figures in brackets are percentages of total.

The figures of most interest are those for road and rail freight transport. For both criteria, road freight transport is responsible for the majority of overall freight transport with a share of about two thirds of all goods moved, and over 81 per cent of all goods lifted.[34] In contrast, rail freight transport plays a minor role; only about 7 per cent of total freight are been moved, and just 5 per cent of total goods have been lifted using railways.

Having identified road freight transportation as the predominate mode of overall freight transportation, an interesting issue is the investigation of the composition of road freight transportation. The main distinction can be made between light vans and lorries and heavy goods vehicles with more than 3.5 tonnes gross vehicle weight.

The following table shows annual recent historic growth for light good vehicles and heavy good vehicles, whereas the latter is distinguished between rigid HGVs and articulated HGVs.

[34] The different proportion for both measures arises because of the fact that the average length of haul for road vehicles is, compared to other modes, significantly shorter.

Table 10: Annual Growth Rates in Goods Vehicles.

	LGVS	Rigid HGVs	Articulated HGVS	TOTAL
1976 - 1986	2.0	0.6	2.3	3.9
1986 - 1996	4.3	1.5	4.8	3.3

Source: http://www.roads.detr.gov.uk/roadnetwork/nrpd/heta2/nrtf97/nrtf05.htm; table C.

The classifications of light goods vehicles and articulated heavy goods vehicles show annual growth rates between 1986 and 1996 of 4.3 per cent and 4.8 per cent respectively. These figures are well above those given for total traffic and clearly illustrate the ever-increasing trend in road freight transport, and hence confirm the ongoing trend of a shift from public freight transportation towards road freight transportation.

Table 11: Comparison of Light and Heavy Goods Vehicles: 1997.

	LGV	HGV
Goods lifted (million tonnes)	N/A	1,643 (1,380)
Distance travelled (billion vehicle kilometres)	40.5 (23.1)	31.8 (22.6)
Number (thousand)	2,317 (1,461)	414 (507)

Source: DETR (1999) and British Road Foundation (1998).

Note: Numbers in brackets are for 1980.

LGVs are predominately used for relatively short distance movements especially in delivering and service industries. Hence, there are two different trends in the road freight sector that can be observed. First, while the overall amount of freight moved on the road has increased, the number of HGVs has decreased over the last years. This means that fewer HGVs lift and move goods for longer distances. The introduction of the 38 tonne vehicle as the maximum capacity allowed has certainly underpinned this trend away from the use of medium weight vehicles with weights between 7.5 t and 16 t. Second, there is also a shift towards LGVs with a weight of 3.5 t or less. This is supported by the fact that no special HGV license is required for driving such vehicles.

Moreover, one main explanation amongst others for the described trend concerning LGVs can be found in recent changes in business processes of companies. The re-organisation of the production planning and the introduction of 'new' instruments, such as the 'just-in-time-production', have an enormous reach. For producers, a main interest is to reduce the holding level in order to minimise inventory-holding costs. Therefore, the main idea is to get the right product (e.g. raw materials and supplies) to the right place at the right time and avoid any sub-stores. Under the premise to guarantee a smooth production flow, it is necessary that there is no interruption in the chain from the supplier to the final destination at the production plant. Here, the supplier has to ensure that the needed product will be at the production line just in time. This is only possible if they can deliver to possibly various different companies at the same time in very short time periods, possibly several times a day. The formerly traditional once a week delivery with all the products needed for the following week using a large HGV has now been at least partially substituted with a fleet of LGVs with a weight of less than 3.5 tonnes, such as delivery vans and smaller lorries.

As a result of this rather dramatic change, not only the absolute number of LGVs has rapidly increased (and will continue to do so in the future), but also both the total number of kilometres driven, as well as the average driven distance per LGV.

Considering this development in the background of traffic related air pollution, it is rather important to stress that more than 90 per cent of all new registrations of goods vehicles, of which a growing majority are LGVs, are diesel engined vehicles.[35] This is particular important since air pollution from diesel exhausts is recognised as potentially dangerous as it will be discussed in Chapter 3.

Additionally, the figures for both the actual numbers of LGVs, as well as the forecasts, are showing a significant and steady trend. Comparing the figures for LGVs and HGVs, the evolution heads exactly in opposite directions. Whereas the number of HGVs in use has declined since 1975, the numbers of LGVs is continually rising.

Looking at the distances travelled each year, a very similar picture is given; a massive rise in kilometres driven by LGVs since 1980 (by more than 70 per cent) and a relatively constant rate of growth for HGVs. The predictions made by the Department of the Environment, Transport and the Regions indicate that LGV traffic will increase by between 31 per cent and about 60 per cent from 1996 to 2010.

---

[35] In 1996, 91 per cent of all new registered goods vehicles were diesel powered compared with 51 per cent in 1985 and 34 per cent in 1965. See: British Road Foundation (1998), p. 3.

2.4 International Comparison

Having described the developments in the UK transport sector, this section provides some figures for the international comparison of transportation. In the last two decades, the number of motor vehicles in the world has more than doubled. In 1993, there were over 615 million motor vehicles in the world fleet consisting of 469 million passenger cars and 148 million freight vehicles, indicating a very rapid growth.[36] In the following section, a number of tables are presented illustrating various aspects regarding the international road sector.

Table 12: World Motor Vehicle Fleet: 1970-1993.

Country/ Region	1970: Number of vehicles (thousands)	1993: Number of vehicles (thousands)	Shares of total (per cent)		Annual growth rate (per cent) 1970-1993
			1970	1993	
OECD	**211,686**	**469,233**	**86**	**76**	**3.5**
USA	108,418	194,063	44	31	2.6
Other OECD	103,268	275,170	42	45	4.4
Non-OECD	**34,692**	**147,854**	**14**	**24**	**6.5**
Total (world)	246,378	617,087	100	100	4.1

Source: OECD (1993), p. 221. & OECD (1995b), p. 215.

Most significantly, it can be noted that between 1970 and 1993, the US share of the world's motor vehicle fleet decreased from 44 per cent to 31 per cent.

[36] See: U.S. Department of Transportation (1997), p. 207.

Similarly, the share for all OECD countries has decreased from 96 per cent in 1970 to 76 per cent in 1993.[37] On the other hand, during the same time period the share of the world's motor vehicle fleet held by non-OECD countries increased from 14 to 24 per cent indicating the growing importance of these countries particularly regarding global air quality.

Although the per capita ownership of passenger cars is far lower in non-OECD countries than in OECD countries, the average annual growth in the number of motor vehicles was nearly twice as high in these countries (6.5%) compared with OECD countries between 1970 and 1993, with some countries, such as Mexico and India, experiencing even greater increases.[38] The following tables illustrate the road traffic volume, divided into passenger cars and freight vehicles, for selected OECD countries.

Table 13: Road Traffic Volume: Passenger Cars.

Country	**Vehicle-km travelled (billions)**			**Annual growth rates (per cent)**	
	1970	1980	1993[a]	Distance travelled 1970-1993	Passenger cars 1970-1993
USA	1,434	1,789	2,652	2.7	2.2
Japan[b]	120	241	429	6.5	6.9
France	165	245	343	3.2	3.0
W-Germany	216	297	425	3.0	3.6
Italy	123	191	356	4.7	4.8
Great Britain	141	197	334	3.8	3.2
Netherlands	38	61	81	3.4	3.8
OECD	2,584	3,604	5,473	3.3	3.4

Source: OECD (1993), p. 221; OECD (1995b), p. 215 & DETR (1999), pp. 166-167.

Note: (a): Provisional data.
(b): Excludes light vehicles.

[37] OECD = Organization for Economic Cooperation and Development includes currently the following countries: Australia, Austria, Belgium, Canada, Denmark, Finland, France, Germany, Greece, Iceland, Ireland, Italy, Japan, Luxembourg, Mexico, The Netherlands, New Zealand, Norway, Portugal, Spain, Sweden, Switzerland, Turkey, the United Kingdom, the United States. In 1994, Mexico also joined the OECD.

[38] See: U.S. Department of Transportation (1997), pp. 218-220.

During the time period of 1970 to 1993, road traffic volume for passenger vehicles (in kilometres travelled) grew dramatically, and in fact much more rapidly than population in the respective countries. The following two tables present figures for the road networks, the number of vehicles in use and the resulting number of vehicles per kilometre of road in 1996 for some selected OECD countries.

Table 14: Road Networks and Vehicle Use.

Country	Total road networks 1996 (kilometres)	Total number of vehicles in use 1996 (thousands)	Vehicles per kilometre of road	
			1996	Change to 1970 (per cent)
Belgium	143,200	4,768	35	20.7
Denmark	71,660	2,040	29	26.1
France	892,500	30,558	34	54.6
Germany	633,700	43,351	66	57.1
Ireland	92,650	1,109	12	100.0
Italy	316,400	38,586	121	116.1
Netherlands	127,060	6,208	49	14.0
Great Britain	371,870	23,338	63	40.0
Japan	1,152,070	69,244	59	126.9
USA	6,419,400	203,660	31	40.9

Source: British Road Foundation (1998), pp. 35-36.

For all countries presented, a significant increase in the number of motor vehicles per kilometre can be observed. Whereas some countries including Belgium and the Netherlands experienced a more moderate growth, the vehicle population per kilometre in other countries, such as Italy and Japan, has more than doubled in the last two decades. In short, although the road networks have constantly been increasing, these roads are getting increasingly crowded.

Table 15: Road Traffic Volume: Freight Vehicles.

Country	Vehicle-km travelled (billions)			Annual growth rates (per cent)	
	1970	1980	1993[a]	Distance travelled 1970-1993	Freight vehicles 1970-1993
USA	346	619	1,039	4.9	4.1
Japan[b]	100	142	272	4.4	4.1
France[c]	32	49	94	4.8	3.9
W-Germany	37	32	46	3.6	2.0
Italy[d]	23	33	52	3.6	4.9
Great Britain	35	41	64	2.7	2.2
Netherlands	6	8	16	4.4	3.4
OECD	656	1,086	1,807	4.5	4.1

Source: OECD (1993), p. 22 & OECD (1995b), p. 215.

Note: (a): Provisional data.
(b): Excludes light vehicles.
(c): Excludes freight vehicles over 15 years old with capacity greater than or equal to 3 metric tonnes.
(d): Includes three-wheel vehicles.

As described in a previous section, in Great Britain freight road traffic volumes grew faster than GDP for the time period between 1970 through 1993. The following table shows the figures of freight moved on the road and by rail in billion tonne-kilometres in 1985 and 1995.

Table 16: Freight Moved On National Territory: 1985 and 1995.

Country	Road		Railways	
	1985	**1995**	**1985**	**1995**
Belgium	22.1	42.6[a]	8.3	7.3
France	20.1	157.1[b]	8.1	9.6
Germany	83.8	279.0	121.6	68.8
Ireland	4.5	5.4	0.6	0.6
Italy	144.1	194.8	16.9	22.2
Netherlands	18.4	27.0[c]	3.3	3.1
Great Britain	101.1	143.7	16.0	13.3
Japan	205.9	295.0	22.1	25.1
USA	981.0	1,345.0[d]	1,280.4	1,980.0

Source: DETR (1999), p. 169.

Note: (a): Including national and international traffic.
(b): In vehicles above a size threshold which (for EC countries) may not exceed 3.5t net.
(c): National traffic only.
(d): Intercity transport only.

Whereas the amount of freight moved in billion-tonne kilometres has increased in all countries in the time period 1985 to 1995, this is not necessarily given for railways as the means of freight transportation. In some countries, such as Germany and Great Britain, less freight has been moved by rail indicating the general trend away from public transportation towards road freight transportation.

2.5 Forecasts and Future Trends for Passenger Traffic

Since it is rather unlikely that the overall present social and economic conditions will radically change in future, it is expected that the demonstrated trend in both the freight transport as well as the private passenger transport sector will continue in future.[39] The forecasts made by the Department of the Environment, Transport and the Regions (DETR) for the growth in the volume of motor traffic on roads in Great Britain until the year 2031 leave no doubt that private motor vehicles will continue to establish their dominance and further displace any other transport mode.

The following table illustrates these estimates for cars, passenger service vehicles (i.e. buses and coaches), and total overall traffic, representing a low forecast (L), a central estimate that is considered the most likely outcome (C), and a high forecast (H). The presented forecasts are based on current (1996) policies, the best available evidence of behaviour, and the capacity of the current (1996) network. Hence, these are forecasts of outcomes where policies and behaviour do not change significantly.

---

[39] See: Linster (1990), pp. 19-21.

Table 17: Forecasts for Passenger Transport Sector.

	CARS			PSVs			TOTAL		
	L	C	H	L	C	H	L	C	H
1996 traffic (bn kms)	**362.3**			**4.9**			**438.3**		
1996 = 100	**100**			**100**			**100**		
2001	103	109	114	98	103	109	103	109	115
2006	110	118	126	100	107	114	110	119	127
2011	116	127	137	101	111	120	117	128	139
2016	122	136	149	104	115	127	124	138	151
2021	126	143	159	106	120	134	129	146	163
2026	128	148	167	109	126	143	132	153	173
2031	130	153	175	113	133	153	136	160	184

Source: http://www.roads.detr.gov.uk/roadnetwork/nrpd/heta2/nrtf97/nrtf05.htm.

The presented estimates for traffic growth, in terms of all motor vehicles form 1996 to 2016, range between 24 per cent and 51 per cent with a central estimate of 38 per cent. From 1996 to 2031, these estimates range between 36 per cent and 84 per cent with a central estimate of 60 per cent.

Comparing the two categories of cars and passenger service vehicles (PSVs), both show a significant trend, with an increase in car traffic much higher than those of PSVs, which stays well below the total trend.

Additionally, the DETR provides forecasts of annual percentage growth rates for various traffic modes.[40] Here, it is estimated that between 1996 and 2006 the annual growth rate is 1.65 per cent (central estimate). However, this rate will eventually decline to an annual growth rate of 0.67 per cent for the years 2026 to 2031, which reflects some kind of saturation process for cars. The respective figures for total traffic lie above these estimates throughout, and assume annual

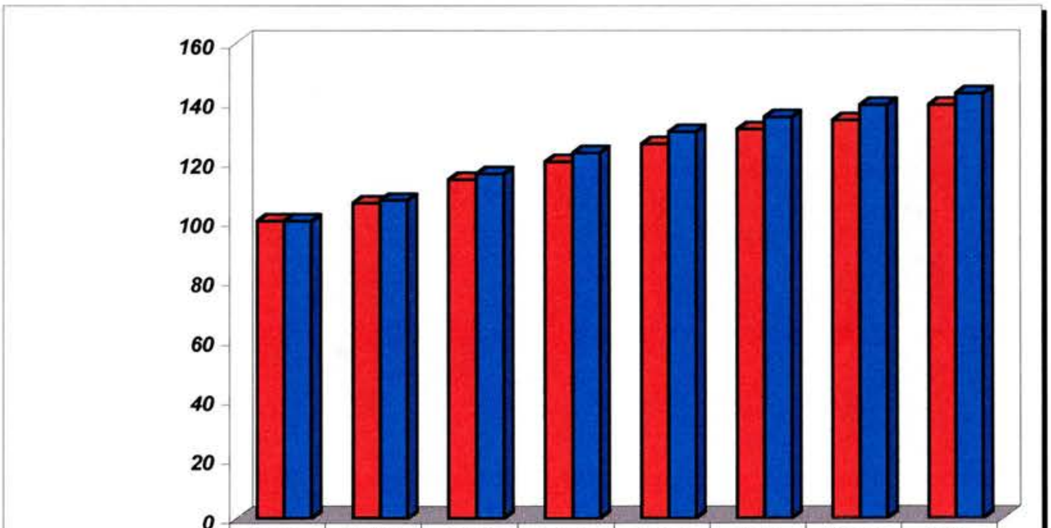
[40] The here presented figures are central most-likely forecasts.

growth rates of 0.89 per cent between 2026 and 2031, after starting with rates of 1.74 between 1996 and 2001.

The estimated trend of annual growth rates for passenger service vehicles, on the other hand, is assumed to go in the opposite direction. Starting with estimated annual growth rates of 0.68 per cent between 1996 and 2001, it is expected that these figures will continually rise. Eventually annual growth rates for buses and coaches of 1.6 per cent between 2026 and 2031, which is not only above the figures for total traffic, but also well above those for cars, are forecasted.[41]

Accompanying these figures for traffic growth, the DETR also provides forecasts for car ownership. The following figure illustrates the central estimates for cars per person, as well as the absolute number of cars up to 2031.

Figure 10: Forecasts of Car Ownership: 1997-2031.



	1997	2001	2006	2011	2016	2021	2026	2031
Cars per person	100	106	114	120	126	131	134	139
Number of cars	100	107	116	123	130	135	139	143

Source: DETR (1999), p. 96.

[41] See: http://www.roads.detr.gov.uk/roadnetwork/nrpd/heta2/nrtf97/nrtf05.htm.

The estimated numbers for cars per person show a steady upward trend, while the estimated proportions of households owning more than one car also support this. It is assumed that in 2031 about 33 per cent of the British households will own more than one car, with a relatively constant share of 47 per cent owning only one car. Consequently, the proportion of households with no access to a car is assumed to drop from 31 per cent in 1996 to just over 20 per cent in 2031.

Additionally, the presented numbers assume an increase in the total number of cars of 43 per cent by the year 2031, although the annual percentage growth rates are declining from 1.65 in 1996 to 2001, to 0.67 in 2026 to 2031. Nevertheless, it will be observed that some household categories will still be far from a saturation level of ownership in 2031.[42]

A contributory influence to the growth in the number of cars is the growth in the number of households by 24% from 1996 to 2031. This happens because (for a given GDP growth) the effect of more households on the vehicle ownership itself, outweighs the effect of the lower income per household. Households closest to saturation are forecasted to have the slowest growth in ownership, and vice versa.[43]

Likewise to the private passenger transport sector, the DETR provides forecasts of future trends in the freight sector. However, only estimates for the road freight sector are given. Nevertheless, a clear future development can be recognised, especially for the developments regarding LGVs and HGVs.

---

[42] Saturation is the maximum ownership level, and differs by type of household. Also, the values used for saturation are those observed in the highest income households of each type in recent years.

[43] Source: http://www.roads.detr.gov.uk/roadnetwork/nrpd/heta2/nrtf97/nrtf05.htm.

The following table shows the estimates in vehicle kilometres travelled given by the DETR for LGVs, rigid HGVs,[44] articulate HGVs,[45] and total road traffic. The estimates are broken down by low, central and high estimates (L, C, H).

Table 18: Forecasts for Freight Transport Sector.

	LGVS			RIGID HGVS			ARTIC. HGVS			TOTAL		
	L	C	H	L	C	H	L	C	H	L	C	H
1996 traffic (bn kms)	**40.4**			**19**			**11.7**			**438,3**		
1996 = 100	**100**			**100**			**100**			**100**		
2001	109	115	121	98	104	109	108	114	120	103	109	115
2006	120	129	138	100	108	115	120	129	138	110	119	127
2011	131	144	156	103	112	122	133	146	158	117	128	139
2016	145	161	177	106	117	129	148	165	181	124	138	151
2021	158	179	200	109	123	137	164	186	207	129	146	163
2026	172	198	225	112	129	146	180	208	235	132	153	173
2031	185	218	251	115	136	156	196	231	265	136	160	184

Source: http://www.roads.detr.gov.uk/roadnetwork/nrpd/heta2/nrtf97/nrtf05.htm.

From the above table, a significant development in the road freight sector can be observed. In comparison to the estimates for total traffic, the forecasts for light goods vehicles reflect an expected growth that is by far above those given for the total road traffic sector. Taking the central estimate, the amount of kilometres travelled per vehicle is estimated to have more than doubled by the year 2031.

Also, not only is the high estimate assuming a traffic growth of more than 150 per cent, but even the lowest forecast for LGV traffic in 2031 is greater than the highest estimate for total road traffic. It is assumed that the annual central

---

[44] Rigid HGVs are goods vehicles of the category 1, i.e. large rigid vehicles, except four axles.

[45] Articulated HGVs are goods vehicles of the category 2, i.e. all articulated vehicles, and four axle rigid vehicles.

growth rates between 1996 and 2001 will be 2.87 per cent, and will slightly fall to an average annual growth rate of 1.87 per cent in 2026 to 2031.

The according figures for total road traffic start with an annual central growth rate of 1.74 and 0.89 receptively.[46] The figures for the expected developments in the heavy goods vehicles sector are similarly significant. Although the traffic forecasts for rigid HGVs are well below those for the total road transport sector, the estimated trend for articulated heavy goods vehicles lies even above that for light goods vehicles.

By looking at the more common categorisation of goods vehicles, forecasted figures for heavy goods vehicles over 3.5 tonnes gross vehicle weight are, with a central estimated growth of 69 per cent up to 2031, very similar and lie between the figures of the two above presented categories.[47] According to the presented figures, the forecasted traffic for both categories of LGVs and HGVs lie above the respective figures for passenger traffic (exclusive two wheelers).

Finally, since this study is predominantly concerned with adverse health affects of road traffic related air pollution in urban areas, it seems appropriate to present some further interesting forecasts particularly concerning these urban areas. Hence, the following three tables illustrate central estimates of future urban traffic for certain times classified in various transport modes.

---

[46] See: http://www.roads.detr.gov.uk/roadnetwork/nrpd/heta2/nrtf97/nrtf05.htm. Particularly, for the annual growth rates, table B, on the stated Internet site.

[47] See: Department of the Environment, Transport and the Regions (1999), p. 96.

Table 19: Forecasts of Urban Traffic in Morning Peak (Mon – Fri, 7 – 10am).

	CARS	LGVS	HGVS	PSVS	ALL MOTOR TRAFFIC
1996 traffic (bn vehicle kms)	21.7	2.8	1.6	0.6	26.9
1996 = 100	100	100	100	100	100
2001	107	114	105	104	108
2011	123	137	114	111	124
2021	136	167	126	122	139
2031	145	199	144	136	150

Source: http://www.roads.detr.gov.uk/roadnetwork/nrpd/heta2/nrtf97/nrtf05.htm; table 5.

Table 20: Forecasts of Urban Traffic in Morning Peak Hour (Mon – Fri, 8 – 9am).

	CARS	LGVS	HGVS	PSVS	ALL MOTOR TRAFFIC
1996 traffic (bn vehicle kms)	9.1	1.1	0.6	0.3	11.1
1996 = 100	100	100	100	100	100
2001	107	113	104	104	107
2011	122	137	112	112	123
2021	135	166	124	123	137
2031	143	197	141	138	148

Source: http://www.roads.detr.gov.uk/roadnetwork/nrpd/heta2/nrtf97/nrtf05.htm; table 6.

Table 21: Forecasts of Urban Traffic Inter-peak Period (Mon - Fri, 10am – 4pm).

	CARS	LGVS	HGVS	PSVS	ALL MOTOR TRAFFIC
1996 traffic (bn vehicle kms)	40.3	5.6	3.4	1.0	50.3
1996 = 100	100	100	100	100	100
2001	108	113	105	104	108
2011	124	136	116	11	125
2021	138	164	128	121	140
2031	147	195	145	135	152

Source: http://www.roads.detr.gov.uk/roadnetwork/nrpd/heta2/nrtf97/nrtf05.htm; table 7.

The figures presented in the three tables above give a clear indication on the future composition of urban traffic. It seems obvious that above all, light goods vehicles will play a very significant role in the development of urban traffic during peak hours, as well as inter peak hours. Light goods vehicle traffic is expected to double by the year 2031, which lies about 50 per cent above the forecasted figures for all motor traffic.

A main reason for this is certainly the trend towards the use of smaller delivery vehicles for short and medium distances, which is already present in current urban traffic patterns. Hence, HGVs are expected to play a less important role for future urban traffic developments. The same applies to buses and coaches (PSVs), indicating that the current trend to use private cars in cities rather than public transport for travelling is not expected to change significantly in future.

2.6 Summary

The purpose of this chapter was to present a detailed description of the transport sector. Both the developments over the last few decades, as well as the current patterns of the transport sector are illustrated. The chapter was basically divided into three main sections. First, the passenger transport sector was described. Here, special emphasis was given to the different developments in the private car sector in comparison to the public passenger transport sector including the bus and railways modes. It was shown that the last few decades have been a period of outstanding growth.

However, a divergence between the developments in the private and public transport sector are fairly evident, where increasingly less importance is placed on the latter. The private car is without any doubt the preferred mode of transport for all main journey purposes. About 95 per cent of passenger travel was made by road, and 5 per cent by rail.

Second, the developments in the freight transport sector were analysed. Again, the two main modes of road and rail were compared. Although the trends in the freight transport sector are somewhat more mixed, the description in this chapter clearly demonstrated the different developments in the figures for freight hauled by road compared to that hauled by rail. The latter experienced a sharp fall in the relative share as well as the absolute volume of total freight hauled in Great Britain. On the other hand, more and more freight has been hauled on roads.

Finally, some figures for road traffic forecasts are presented, indicating a substantial rise in the number of vehicles, as well as distances travelled (vehicle kilometres). Almost all of the estimated 7 to 12 per cent increase in road passenger travel to 2000 is expected to be account for by increased car use. These forecasts have a significant influence on future air quality. Not only the absolute figures for these trends, but also the composition determine traffic-related air pollutants.

For instance, assuming current legislation, emissions of particulates are expected to rise similar to the tremendous growth of LGVs, since these vehicles mainly run on diesel fuel, which is the main traffic related source of particulate pollution. The following chapter presents a detailed discussion of the main air pollutants related to road traffic.

3. The Main Traffic-Related Air Pollutants

3.1 Introduction

The descriptions in the previous chapter impressively reflect the tremendous developments in the road transport sector over the past decades. Accordingly, road traffic is the focus of attention more than ever.[1] This is again strongly supported by the forecasts predicting an ever-growing stock of motor vehicles, as well as a dramatic rise in the usage of these vehicles. Thus, since the vast majority of the vehicles operate by burning fossil fuel, the described continual growth in vehicle stock comes along with a significant increase in energy and fuel consumption.

This chapter will first describe this relationship between the transport sector and energy consumption. This is then followed by the description of the association between the transport activities and the emission of various air pollutants, again specifically concentrating on the road sector. These road traffic-related air pollutants are then described in some detail in the subsequent section. The air pollutants this study mainly focuses upon are carbon monoxide, oxides of nitrogen, particular matter, sulphur dioxide, and ozone. However, two more traffic related pollutants, namely the group of volatile organic compounds and lead will also be described, although they will not be included in further analyses in this study.

---

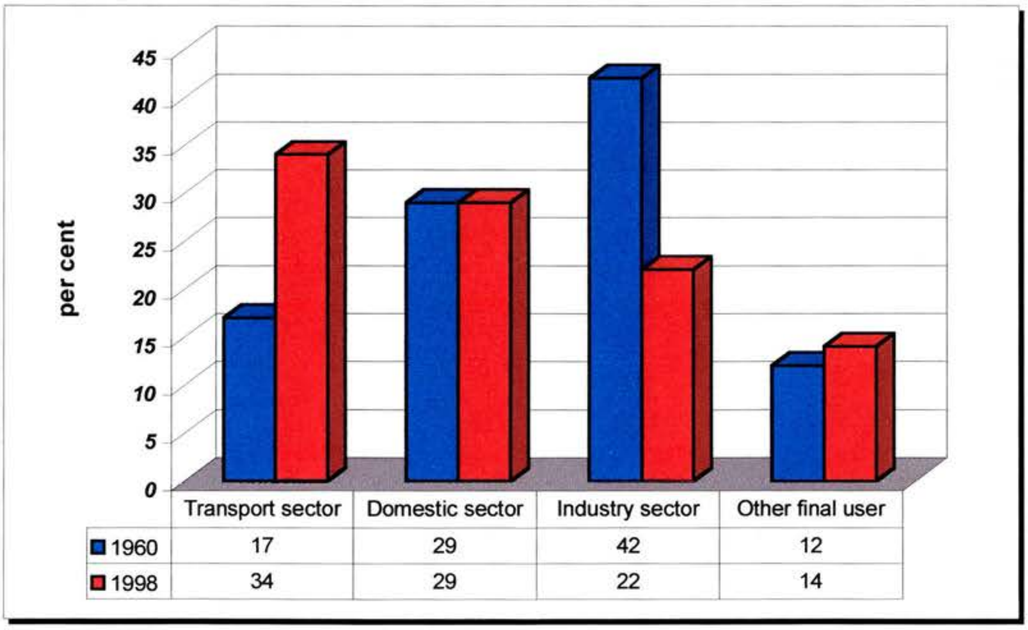
[1] See, for instance, Environment Agency website at: http://www.enviornment-agency.gov.uk.

Finally, a brief overview of the current guidelines and air quality legislation, as well as future emission targets for future air quality strategies for the UK is given, before summing up the chapter.

3.2 The Link between the Energy and Transport Sector

In 1998 the final energy consumption was about 169.4 million tonnes of oil equivalent.[2] As illustrated in the following figure, the transport sector is the main single source of energy consumption, and accounts for some 34 per cent followed by the domestic sector (29 per cent) and the industrial sector (22 per cent).

Figure 11: Energy Consumption by Final User: 1960 vs. 1998.



Source: DETR (1999), p. 17.

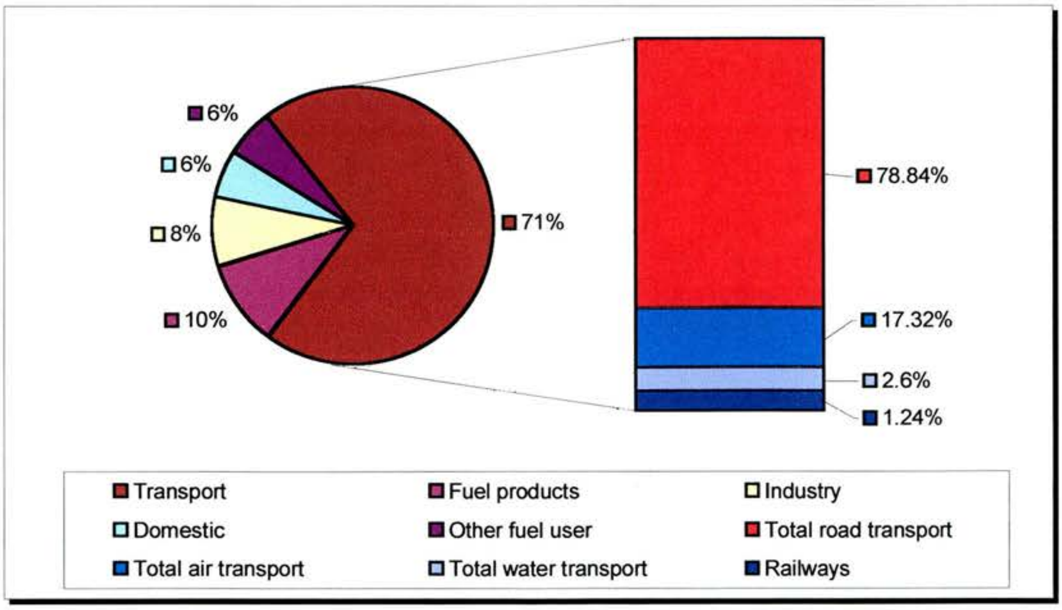
---

[2] The term oil equivalent is a commonly used unit of measurement, which enables different fuels, to be compared and aggregated. It is, however, regarded as a measure of energy content rather than a physical quantity. For conversion factors see: Department of Trade and Industry (1999), p. 18.

Comparing the figures for 1998 with those of 1960, clear trends have developed in two opposite directions. First, the share of the industry sector, which once had the greatest level of energy consumption with up to 44 per cent of total final energy consumption (between 1960-73), has steadily fallen to about 22 per cent, while second, there has been a significant increase in energy consumption in the transport sector from 17 per cent in 1960 to a share of 34 per cent in 1998. The present share held by the domestic sector, with 29 per cent, is at the same level as it was in 1960, although there has been some fluctuations in the intervening years.

By examining the products used for energy, the dominant supremacy of the transport sector in terms of energy use can be once again underpinned. This is illustrated in the following figure.

Figure 12: Petroleum products used for energy in 1998.



Source: DTI (1999), pp. 78-79 & pp. 98-99.

The transport sector's share of total products used for energy has increased from 62 per cent in 1992 to some 71 per cent in 1998, and with 47,320,000 tonnes out of a total of 79,073,000 tonnes is by far the most significant user of petroleum products.[3] Subdividing the transport sector's share into different categories of transportation, total road transport is accounting for 37,230,000 tonnes out of the 47,320,000 tonnes used in the transport sector. This is a share of about 79 per cent of the total transport's share.

Total air transportation is accounting for most of the remainder, with total railways at about 483,000 tonnes being by far the smallest share of the total amount of petroleum products used for energy. Hence, in the UK, as in most other industrialised countries, combustion of fossil fuels by transportation, and in particular road transport, is the major source of emissions polluting the air.

Having roughly described some aspects in the energy sector that are related to the transport sector, attention shall now be placed on the road transport sector. Before subsequently turning to a more comprehensive description of the main pollutants related to road transport, a brief insight into some technical aspects of motor vehicle combustion and the related conditions for air pollutant emissions shall be given.

Generally, there is a wide range of factors that have a definitive influence on both the amount and the proportion of substances found in the exhaust gases released from vehicle engines.

---

[3] See: Department of Trade and Industry (1999), pp. 104-105.

The most significant of these factors are the following:[4]

- the design of the respective engine.
- the size and power of the engine.
- the specific characteristics of the fuel being used.
- the conditions in which the vehicle is being used.
- the way the vehicle is being driven (emission rates of vehicle engines vary considerably with the respective operating mode such as idling, accelerating, cruising, or decelerating).
- weight of vehicle.[5]
- the state of maintenance and the age of the vehicle.

Currently, the reciprocating internal combustion engine is nearly the only power source for automotive applications, where the fuel used is mainly fossil, i.e. petrol or diesel.[6] In simplified terms, a petrol engine works in the following way: in the combustion chamber a mix of air and fuel is drawn into the cylinder where it is compressed and then ignited by a spark. In comparison to a spark ignited combustion in petrol engines, diesel engines work in a rather different way. Here, liquid diesel fuel is injected into air, which is already very hot as a result of compression, and then it evaporates, mixes, and eventually ignites after a delay time.[7] Since the first fuel entering the combustion bowl is forming a premixed flame, a spark is superfluous for ignition.

---

[4] See: Watkins (1991), pp. 27-31.

[5] Generally, the higher the vehicle weights the higher the fuel consumption, and consequently, the higher the amount of pollution produced.

[6] See: Ospelt (1994).

[7] This delay time mainly depends on the cetane number of the fuel as well as on the temperature level of the hot air. See: Ospelt (1994).

Generally, diesel engines operate with a much higher (better) air/ fuel ratio than petrol engines (approximately at a rate of 20-40:1). Consequently, the combustion process of a diesel engine is more efficient than a petrol engine, which is due to the fact that it works with a surplus of air.[8] Moreover, the emission rates as far as the pollutants carbon monoxide (CO), nitrogen oxides (NO_x) and unburned or partially burnt hydrocarbons (HC) produced by petrol engines are concerned, are also significantly lower in diesel engines in comparison to petrol engines.[9]

However, the use of diesel engines does not come without any serious disadvantages. Diesel fuel contains about ten times as much sulphur as petrol does; therefore sulphur dioxide (SO_2) emissions from diesel are much greater than from petrol engines per unit mass of fuel consumed. The substance most often related to diesel engines, which gives most concern, is certainly the group of particulate matter. The use of diesel engines is causing at least 90 per cent of the total amount of airborne particulate matter generated by vehicles.

In 1998, about 60 per cent of the petroleum used in the road transport sector was accounted for by motor spirit (21,848,000T), and diesel-engined road vehicle fuel (DERV) represented the remaining 40 per cent (15,143,000T). Further, there is a clear trend recognisable towards the use of DERV over the previous years, since its share has increased by almost 10 per cent between 1992 and 1998, which again reflects significantly the current trend towards a greater share of diesel vehicles.[10] Breaking down these two categories, a clear picture of the distribution within both of them can be established.

---

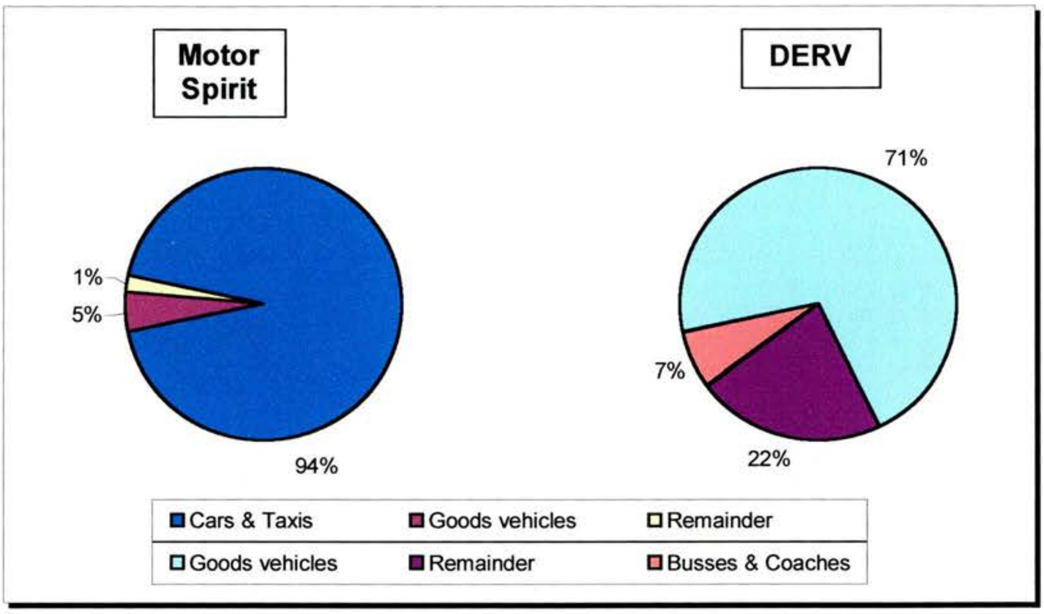
[8] The consumed amount of fuel (diesel) is at least 25 per cent less by volume and about 15 per cent less in terms of mass than in a comparable powered petrol engine. Hence, they have inherently a better fuel consumption than petrol engines.

[9] See: Watkins (1991), pp 30-33.

[10] See: Department of Trade and Industry (1999), p. 104.

By far the greatest share of motor spirit is used by body type cars, whereas goods vehicles make up only a minor share of the total petrol use.[11] On the other hand, figures for diesel fuel show a different situation. Here, goods vehicles account for almost three quarters of the total DERV fuel use. However, the considerably large share, named as the remainder in the figure below, is some 22 per cent and with an increasing tendency. This class represents predominantly diesel-engined cars and taxis.

Figure 13: Consumption of Road Transport Fuels, 1998.



Source: DTI (1999), p. 94.

[11] The remainder includes mainly motor cycles, mopeds, etc.

3.3 Air Pollution and the Transport Sector

Both petrol and diesel fuels are mixtures of hydrocarbons which are compounds containing hydrogen and carbon atoms. Under the optimal condition of a complete combustion and oxidation in the engine, oxygen in the air would convert all the hydrogen in the fuel to water and all the carbon in the fuel to carbon dioxide. The nitrogen present in the air would remain unaffected. However, it is almost impossible to achieve a complete oxidation under realistic conditions. Therefore, a considerable quantity of products of incomplete combustion is formed as well as several other harmful substances, and are emitted into the air either directly (e.g. carbon monoxide) or indirectly as a secondary pollutant (e.g. ozone).[12]

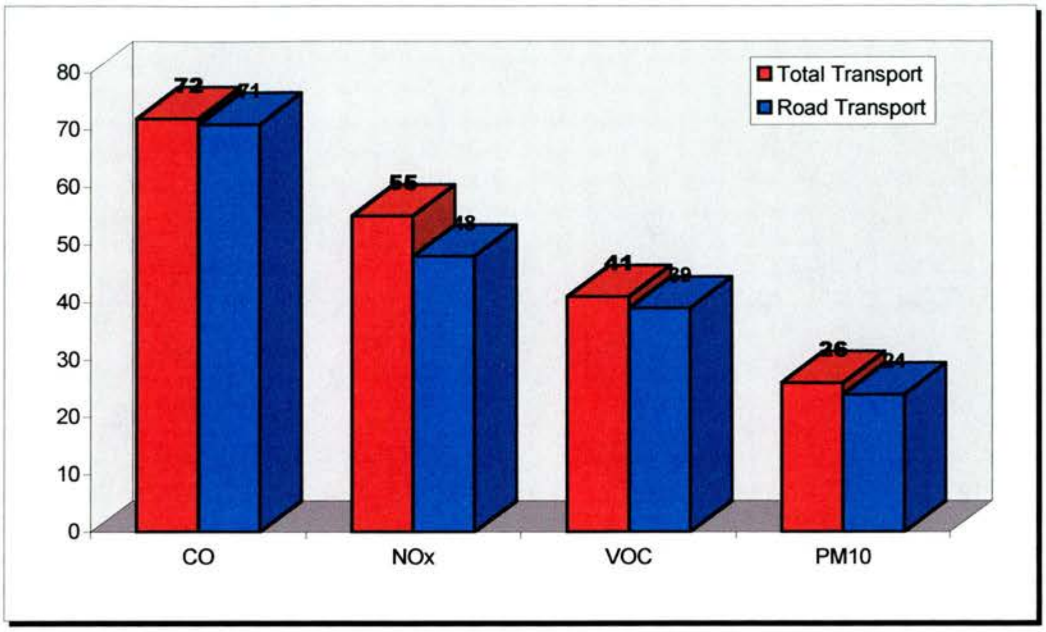
Undoubtedly, the introduction of stricter emission control has helped to reduce the absolute rate of emission. However, with an ongoing growth in traffic density, and a resulting rise in energy consumption, the global emissions of total pollutants produced by motor vehicles have increased. Consequently, the problem of air pollution by exhaust emissions is still of great concern and has led to various activities to control in recent years.

The following figure illustrates the proportion of emissions from total transportation and road transport respectively broken down for the main air pollutants. A more detailed discussion for the respective pollutants will follow in later sections of this chapter.

---

[12] See: Watkins (1991), pp. 5-6.

Figure 14: Pollutant Emission from Total Transport and Road Transport.



Source: DETR (1999), p. 64 & DETR (1998c).

The transport sector, and specifically road traffic, is the dominant emission source for most pollutants. Additionally, the presented figures are indeed national averages. When concentrating on urban inner city situations, road transport is an even more dominant source of pollution emissions, as it will be illustrated in a later section of this chapter.

Emission caused by mobile sources, mainly in the transportation sector, are extremely complicated in their composition, especially since they can consist of several hundreds of different substances that are present as gases, particulates and aerosols.

Additionally, a large amount of these compounds, also known as primary pollutants, are assumed to play an important role in the formation of secondary pollutants such as tropospheric ozone (smog) in the atmosphere. These indirectly emitted pollutants can be even more harmful than their precursors.

The attempt to determine specific impacts of certain pollutants led to a situation where most western countries have established guidelines for these pollutants.[13] To establish these guidelines, usually six so called 'criteria pollutants' are taken into consideration.[14] These criteria pollutants are identified as the main hazards from the exhausts of motor vehicles.[15]

Since the introduction of stricter emission controls as well as the use of technical fixes such as catalytic converters to new petrol cars, some substantial improvements in the amount of pollutants per vehicle have been made. The emissions of the three main locally-acting category of pollutants which are generated by petrol cars, namely carbon monoxide (CO), oxides of nitrogen (NO_x), and volatile organic compounds (VOC) including hydrocarbons (HC) have been dropping significantly since then.

However, although these means seem to contribute substantially to meet pollution problems, and the obvious decline in total emissions support the short term benefits of these measure, a middle to long term view may give a different picture. Given the current developments in traffic figures, namely the increase in vehicle number and vehicle use, estimations predict that such benefits achieved by the introduction of the catalytic converters in Britain in 1992 will be negated by 2004 at the latest.[16]

---

[13] However, these guidelines usually differ from country to country. The respective current legislation will be described in the final section of this chapter.

[14] The US Environmental Protection Agency (EPA) first introduced the term 'criteria pollutants' as an indicator for the US National Ambient Air Quality Standards (NAAQSs) they established. Generally, criteria pollutants are those substances, which are considered to present a general risk to public health. See: USEPA webpage at : http://www.epa.gov.

[15] In comparison, so-called hazardous air pollutants are related with cancer, birth defects, and neurotoxicity. It is assumed that for these pollutants ambient standards are not appropriate and not practical at all, and are more related with 'industrial atmosphere' than with 'urban atmospheres'. See: Lipfert (1993), pp. 13-14.

[16] See: TEST (1991), p. 90. Also, it was shown that the future vehicle growth has a significant impact on overall emissions. A one per cent difference in annual traffic growth can significantly change the emission prospects. See: Mitchell and Hickman (1990), pp. 56-57.

Regarding the sector of diesel-powered vehicles, similar concern can be raised. Due to different characteristics in the way of operating, it is practically impossible to use the same technology of catalyst to clean up diesel exhaust gases. Furthermore, current popularity of diesel powered vehicles allows one to assume that diesel emissions will play a proportionally greater role in urban air pollution in future. The following air pollutants are considered to be the main substances being present in the exhaust gases stemming from motor vehicles:[17]

- Carbon Monoxide (CO)
- Oxides of Nitrogen (NO_x)
- Carbon Particles (PM_{10})
- Sulphur Dioxide (SO_2)
- Volatile Organic Compounds (VOC)
- Lead Components (PB)

In order to give an initial impression of the importance of various pollutants, a summary of estimated emission factors of such pollutants for both petrol vehicles and diesel vehicles is provided below.[18] Both categories are again subdivided into average operating conditions and urban cycle. The figures are taken from a CSERGE working paper, which again compiled the results of several studies.[19]

[17] See: Watkins (1991), p. 6.
[18] The estimated emission rate is the product of the activity rate (taken from traffic surveys, fuel consumption and vehicle kilometres travelled) and emission factors, which are derived from results of measurements. See: http://www.london-research.gov.uk/emission/webhtm.htm.
[19] See: Eyre et al. (1996), pp. 3-7.

Table 22: Comparison of Petrol and Diesel Fuel Emissions.

Pollutant	Average Emissions (g/km)		Urban Cycle (g/km)	
	Petrol	Diesel	Petrol	Diesel
CO	1.540	0.520	3.750	0.840
NO_x	0.370	0.650	0.700	0.590
SO_2	0.053	0.050	0.062	0.069
Particulate Matter	0.000	0.1700	0.000	0.3000
NMVOC				
Exhaust	0.11	0.0850	0.2200	0.2100
Evaporative	0.0944	0.000	0.1100	0.000
Total	0.2044	0.0850	0.3300	0.2100

Source: Eyre et al. (1996), pp. 3-7.

In addition, as a result of chemical atmospheric reactions in which the products emitted by vehicles react with others, some unpleasant secondary products such as ozone can be produced. The following section provides a fairly comprehensive description of these pollutants mainly related to road transport. Starting with the description of the chemical composition, a general characterisation of the respective pollutants will be introduced. In this, the two types of primary and secondary air pollutants are differentiated.

Also, the proportional split into the various sources of the total emissions of the respective pollutants will be given. In order to round up the general description of the main pollutants, a general overview of the potential adverse effects on human health related to the respective pollutant will also be given.[20]

[20] A more comprehensive review of the results of empirical studies concerning health effects of various air pollutants will be the focal point of the following chapter.

3.4 Description of Primary Air Pollutants

Those pollutants emitted directly into the air by the polluting source, namely motor vehicles are defined as primary pollutants.[21]

3.4.1 Carbon Monoxide

Until recently, little attention had focused upon carbon monoxide (CO) as a severe air pollutant. The fact that the potential adverse impacts of CO were not as evident as other pollutants may be an important reason.[22] carbon monoxide is an odourless, colourless and poisonous gas. It is one of the most directly toxic air pollutants, which is largely derived from combustion processes where carbon and its compounds in fuel are only oxidised incompletely, due to a lack of sufficient oxygen.[23] Other possible sources of CO emissions are mainly industrial processes and fuel consumption in sources such as boilers and incinerators.

However, carbon monoxide is not only emitted into the atmosphere as a result of combustion processes, but also formed from the oxidation of hydrocarbons and other organic compounds. Nevertheless, the highest concentrations of carbon monoxide are usually found close to the sources of combustion (e.g. roadside). The atmospheric lifetime of carbon monoxide is approximately one month. The following figure gives an overview of the composition of total CO emissions (in million tonnes).

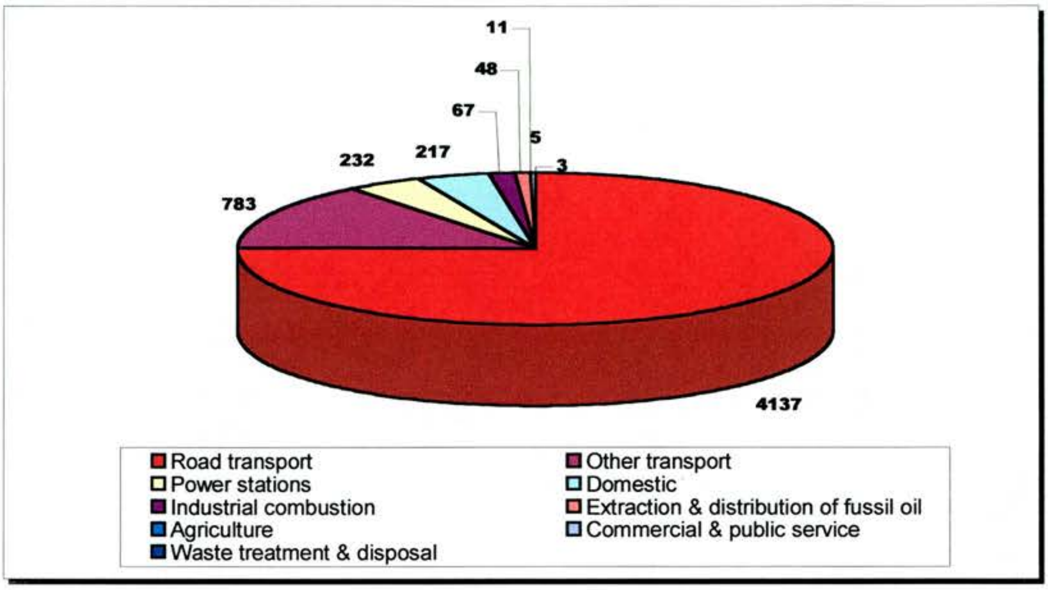
---

[21] For the most part, these primary pollutants are transformed in the atmosphere into other secondary pollutants.

[22] See: McMichael (1997), p. 15.

[23] Hence, only carbon-based fuels are relevant in this respect, but not diesel fuel.

Figure 15: Proportion of CO Emissions (1995).



Source: DTI (1999), pp. 272-273.

By far the largest amount of emissions of carbon monoxide is produced by the transport sector, with power stations and the domestic sector following. Almost all of the emissions generated in the transport sector are accounted for by road transport. Motor vehicles are responsible for around 75 per cent of total carbon monoxide emitted into the atmosphere nation-wide.[24] Typically, however, concentrations of CO are much higher in urban areas with heavy traffic than in more rural environments. Hence, these emissions also lead to exceedingly high concentrations of carbon monoxide in these areas.[25]

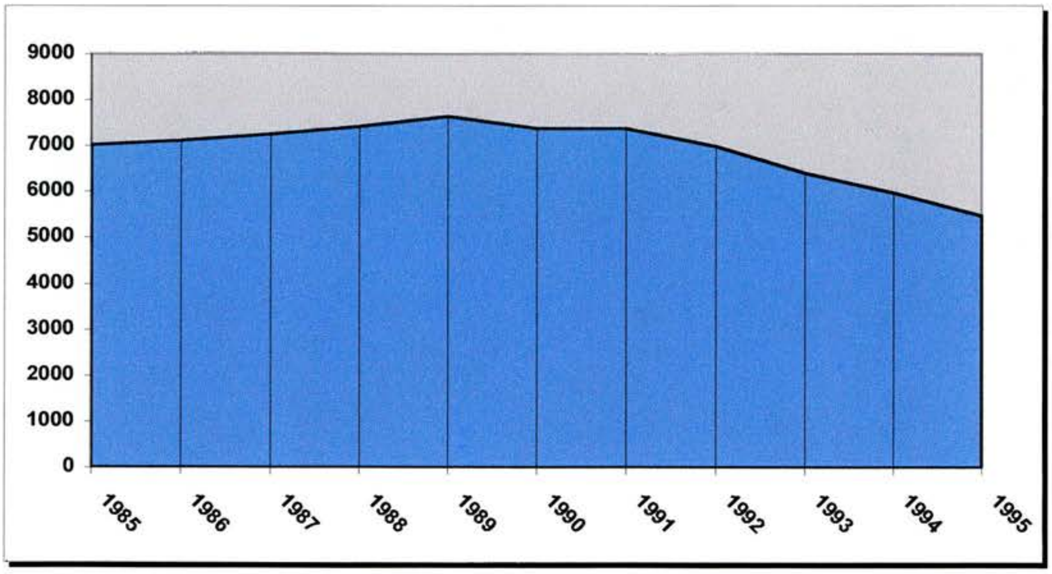
---

[24] The highest carbon monoxide emissions are produces when spark ignition petrol-fuelled vehicle engines are idling or decelerating, and being lowest at very low speeds. Additionally, the emission rate for individual vehicles depends critically whether the engine is cold or poorly maintained. See: http://www.london-research.gov.uk/emission/webhtm.htm, p. 6.

[25] See: Department of the Environment, Transport and the Regions (1999), p. 55. This is especially the case in cases where vehicles lack catalytic converters or the vehicles are not adequately maintained. Generally, the highest concentrations of carbon monoxide occur at the roadside on cold winter days with low wind speeds

Despite the overall downward trend in emissions, and consequently lower concentrations of carbon monoxide, metropolitan areas still experience rather high levels of CO. An increasing level of inner city traffic congestion resulting from the continuous growth in vehicle density is the main cause. Peak levels of carbon monoxide usually occur during the busy times of rush hour, especially at cold winter temperatures when low wind speeds allow the build-up of pollution. In addition, vehicles with cold engines and those idling at intersections cause relatively higher proportions of emissions. The following graph shows the actual trends in total CO emission in the UK since 1986.

Figure 16: CO Emission Trends UK (in thousand tonnes).



Source: DETR (1999c), p. 42.

Since CO emission reached its peak in the late 1980s, the amount of carbon monoxide has fallen below the 1970 level, and has continued to decrease. Similar to the development in the emission of other pollutants, such as NO_x and VOC, this decline is mainly the result of the introduction of tighter emissions standards[26] and the growing penetration of diesel cars, which cause less carbon monoxide emissions than petrol-fuelled cars. Estimated figures show that petrol powered motor vehicles generate some 95 per cent of the total CO emissions attributed to road transport.[27]

In high concentrations of traffic related exhaust emissions, carbon monoxide is certainly one of the most directly acting toxic substance.[28] Consequently, exposure to carbon monoxide may lead to severe adverse health effects. The greatest danger of carbon monoxide is its ability to interfere with the absorption of oxygen by haemoglobin (red blood cells) and consequently restricts the supply of oxygen by the blood to body tissue.

When carbon monoxide is inhaled, it enters the bloodstream and competes with oxygen to bind with haemoglobin. Its affinity for haemoglobin is between 200 and 250 times higher than of oxygen.[29] By combining with haemoglobin, carboxy-haemoglobin (COHb) is formed. As a result of this formation, the oxygen carrying capacity of blood in the human body is lowered, since the

---

[26] The introduction of the obligation in 1992 that all new petrol cars have to be fitted with a 3-way catalytic converter from 1993 onwards is certainly the most significant aspect producing this trend.

[27] It is clearly impossible to actually measure the emissions on a very individual level, i.e. from each motor vehicle, home or factory. Hence, the estimates presented here as well as for other air pollutants are taken from National Atmospheric Emissions Inventory (NAEI) database. Annual emissions for 10 pollutants or pollutant groups resulting from both fuel combustion and from non-combustion sources are calculated. Further, the estimates are produced and categorised within the scope of the UNECE (United Nations Commission for Europe)/ CORINAIR (CORe INventories AIR) SNAP (Selected Nomenclature for Air Pollution) project. This is a programme by the European Environmental Agency (EEA) and supervised by the Commission of the European Communities in 1990. The objective of this project is to produce estimates for European emissions of air pollutants from all sources (stationary, mobile, natural). For the UK, the National Atmospheric Emissions Inventory is the official air emission inventory, funded by the Department of the Environment, Transport and the Regions. The National Environmental Technology Centre maintains the inventory for the Department of the Environment, Transport and the Regions. The database comprises estimates for most pollutants covering the period 1970 to the latest year (currently 1996). The database is accessible via the World Wide Web on: http://www.aeat.co.uk/netcen/airqual/emissions/naeihome.html.

[28] See: Linster (1990), p. 30.

[29] See: http://www.imib.rwth-aachen.de/HYGIENE/mitarb/dott/luftr/1.html, p. 2.

reaction between oxygen and haemoglobin is blocked and consequently not sufficient haemoglobin is available. Those tissues that are very sensitive to reduced oxygenation, such as the brain and the heart are consequently jeopardised, which again may lead to persisting health damage, or even premature death from suffocation.[30]

The health threat from exposure to carbon monoxide is most serious for those who already suffer from cardiovascular disease. However, healthy individuals are also affected, but normally only at correspondingly higher levels of exposure.

Generally, exposure to elevated carbon monoxide levels is associated with the impairment of vision, physical co-ordination, perception, thinking and reflexes. Hence, it leads to a reduced work capacity and manual dexterity, poor learning ability, difficulty in performing complex tasks, drowsiness, fatigue, and stress. It also induces angina and affects the foetal growth and the development of the tissues of young children and may retard the unborn babies growth and mental development.[31] Sickle cell anaemic are also quite susceptible to exposure of low levels of CO. Moreover, the main consideration is its interference with the respiratory chemistry and can also affect the central nervous systems as well as the cardiovascular system.

---

[30] Additionally, some research suggests that an additional mechanism for CO toxicity may involve CO binding to myoglobin in the heart muscle cells called myocytes. Under normal conditions myoglobin binds the oxygen released by haemoglobin and transports it to the site of ATP synthesis. CO, after being released from haemoglobin, could also bind to myoglobin, forming caboxymyoglobin, thereby reducing the amount of oxygen transported by myoglobin, which in turns leads to a reduced amount of synthesised ATP, and ultimately decreases the ability of muscle cells to contract. See: Wittenberg and Wittenberg (1997).

[31] See: Whitelegg, Gatrell and Naumann (1993), pp. 15g-15f.

Additionally, these effects can also be exacerbated in combination with other pollutants[32], with this synergistic action being particular relevant for people with already existing respiratory or circulatory ailments.[33] The following table gives a listed summary of signs and symptoms of toxic effects of CO as measured by percentage of carboxy-haemoglobin in the blood (COHb).

Table 23: Adverse Effects of Carbon Monoxide on Human Health.

Per cent COHb	**Signs and symptoms for an average man**
0 – 10	No signs or symptoms.
10 – 20	Tightness across the forehead, possible slight headache, dilation of the cutaneous blood vessel.
20 – 30	Headache and throbbing in the temples.
30 – 40	Severe headache, weakness, dizziness, dimness of vision, nausea, vomiting and collapse.
40 – 50	Same as above, greater possibility of collapse, syncope and increased pulse and respiratory rates.
50 – 60	Syncope, increased pulse rate, coma, intermittent convulsions, and Cheyne-Stokes respiration.
60 – 70	Coma, intermittent convulsions, depressed heart action and respiratory rate, and possible death.
70 – 80	Weak pulse, slow respiration, respiratory failure and death within a few hours.
80 – 90	Death in less than an hour.
90+	Death within a few minutes.

Source: Watkins (1991), p. 67.

32 See: Holman (1991), p. 10.

33 See: Linster (1990), p. 30.

At levels where more than 10 per cent of the haemoglobin has been transformed into COHb, in average healthy persons first symptoms appear; such as slight headaches and the dilution of the cutaneous blood venal.[34] These symptoms becoming more severe with increasing atmospheric concentrations of CO and consequently COHb levels. Any rate above 50 per cent of transformed haemoglobin will lead sooner or later to premature death.[35] A dose of approximately 50 per cent of COHb is undoubtedly associated with serious adverse cardiovascular effects. Evidence of consistent controlled clinical experiments examining the effects of carbon monoxide exposure on human health show that carbon monoxide exposure may encourage exercise-induced angina in individuals with pre-existing cardiovascular disease.[36]

Additionally, the onset-to-angina time is reduced (at levels of 50-100ppm) as well as the duration of angina increases.[37] Therefore, there are constraints to the activity of persons with ischaemic heart disease from certain exposure levels on. Epidemiological studies could also give positive, but non-specific evidence,[38] whereby empirical epidemiological evidence of increased risks of actual cardiovascular disease events is still rather elusive.[39]

---

[34] For susceptible persons, however, CHOb levels as low as 2 to 3 per cent may already lead to first symptoms.
[35] See: Linster (1990), pp. 30-31.
[36] See: Kleinman et al. (1997).
[37] See: Allred et al. (1989).
[38] See: Stern et al. (1988).
[39] See: McMichael (1997), p. 16.

3.4.2 Oxides of Nitrogen

When atmospheric nitrogen and oxygen atoms react in the air, various oxides are formed. Two of resulting substances are nitrogen monoxide (NO)[40] and nitrogen dioxide (NO_2), which are grouped together in the collective term nitrogen oxides (NO_x).[41] This grouping is due to the fact that most of the NO_2 present in urban environment is actually derived from NO. Practically all oxides of nitrogen derived from combustion processes are discharged as nitrogen monoxide (NO). After nitrogen monoxide has been emitted into the atmosphere it is rather quickly oxidised into nitrogen dioxide (NO_2).[42]

Accordingly, the colourless gas nitrogen monoxide has a very short dwell time in the air and oxidises rapidly to form NO_2, which has a red and brownish colouring.[43] This process is enhanced with an increasing burning temperature, which can actually have some unwanted side effect. Any technical change in order to increase the combustion temperature in vehicle engines to achieve more economical fuel consumption may consequently lead to an increased formation of NO_x.

---

[40] Nitrogen monoxide is also known as nitrous oxide.

[41] Other components formed by the reaction of nitrogen and oxygen atoms include dinitrogen tetroxide (N_2O_3) and dinitrogen pentoxide (N_2O_5). Like all oxides, they are converted in forms of weight into NO_2 equivalents. Also, these components are found in very low concentrations and are seen as being relatively non-toxic. However, they are not included in the definition of NO_x and will hence not be included in the investigation of this study. See: Lynam and Pfeifer (1991), p. 260.

[42] See: Harrisson (1997), pp. 33-34.

[43] Together with ozone, nitrogen dioxide is the dominant component of photochemical smog. In contrast to ozone, it is, however, less biologically reactive which leads that its effects usually do occur with some time lag.

Nitrogen dioxide and nitric oxide, which are the most abundant man-made oxides of nitrogen, are basically formed in two possible ways; first, through the oxidation of the nitrogen contained in fuel (fuel NO_x)[44] and second, through the high-temperature combustion of oxygen and nitrogen in the combustion air, which occurs at temperatures exceeding 1400C° (thermal NO_x).[45] As mentioned before, NO_x is mainly formed by reactions of oxygen and nitrogen in the combustion air during combustion processes.

Nitrogen dioxide (NO_2), however, is only emitted through vehicle emissions into the atmosphere in very small amounts. Rather, the majority of NO_2 is formed through the chemical reaction of nitrogen monoxide (NO) and oxidants, mainly ozone, in the atmosphere. Since NO_2 represents one of the two pollutants summarised in the term nitrogen oxides, this transformation shall be described at this point, rather than in a later section when describing the secondary pollutant ozone. Simplified, there are two different chemical reactions through which NO_2 may be formed. These processes can be written as follows:[46]

$$NO + O_3 \Rightarrow NO_2 + O_2 \qquad (1)$$

or

$$NO + HO_2\bullet \Rightarrow NO_2 + \bullet OH \qquad (2)$$

Accordingly, a nitrogen monoxide atom is reacting with an ozone atom to form nitrogen dioxide and oxygen.[47]

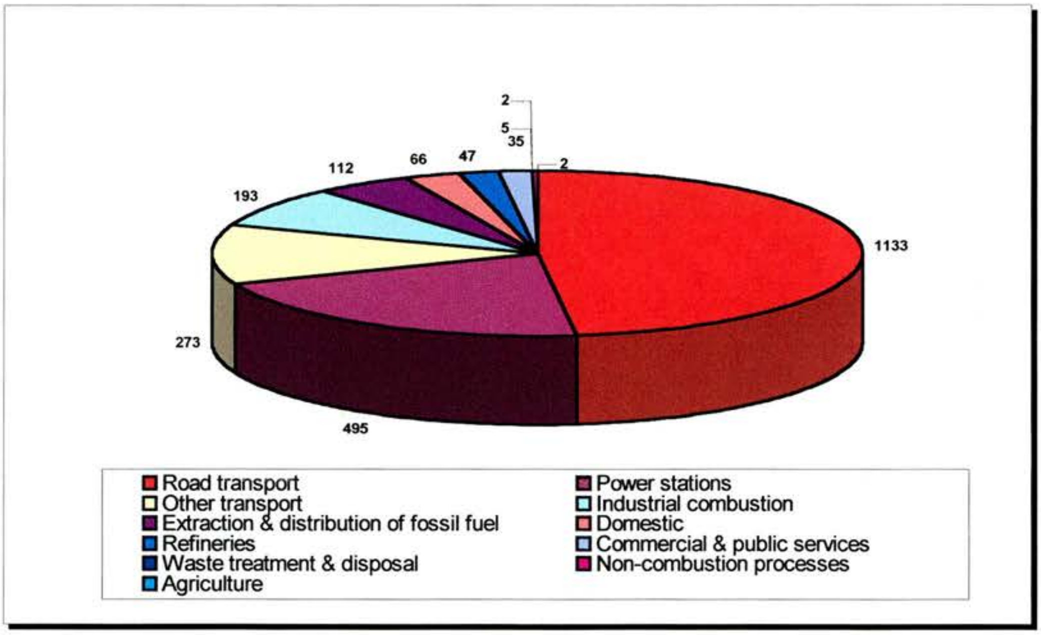
---

44 See: Halkos (1998).
45 See: Complainville and Martins (1994), p. 6.
46 See: Harrison (1997), pp. 33-34 and The Royal Commission on Environmental Pollution (1994), p. 26.
47 Alternatively, NO may also react with oxygen naturally present in the atmosphere to convert it into NO_2. See: http://www.london-research.gov.uk/emission/webhtm.html.

This reaction, however, can also be in reverse order when nitrogen dioxide is acting as a precursor for the formation of ozone. In fact, there is a series of different photochemical reactions concerning the formation of ozone, which will be discussed in more detail in a later section of this chapter.[48]

On the other hand, nitrogen dioxide may also be formed by the oxidisation of NO and the hydroperoxy radical ($HO_2\bullet$), where hydroxyl radicals ($\bullet OH$)[49] are also being formed.[50] The following graph illustrates the composition of the main sources for emissions of nitrogen oxides (in million tonnes).

Figure 17: Proportion of NO_x Emissions.



Source: DTI (1999), p. 271.

[48] Generally, the formation of NO, NO_2, and their reversible reactions under the influence of O_2 or O_3, as well as the concurrence of hydrocarbons and the influence of the daily temperature are all very decisive factors for the ambient concentrations of NO_x. Additionally, NO_2 also changes in the atmosphere to form acidic particles and liquid nitric acid.

[49] The hydroxyl radical is a central species in atmospheric chemistry. It is responsible for the breakdown of many pollutants, which is mainly due to its high reactivity.and consequently for the formation of various so-called secondary pollutants.

[50] Nitrogen monoxide may, in turns, react with the hydroxyl radicals to form nitrous acid (HNO_2). Additionally, especially in polluted air a series of other oxidation reactions may take place.

The transport sector accounts for the majority of NO_x emissions with road transportation, namely the exhausts emitted by motor vehicles, being by far the largest single source of emission.

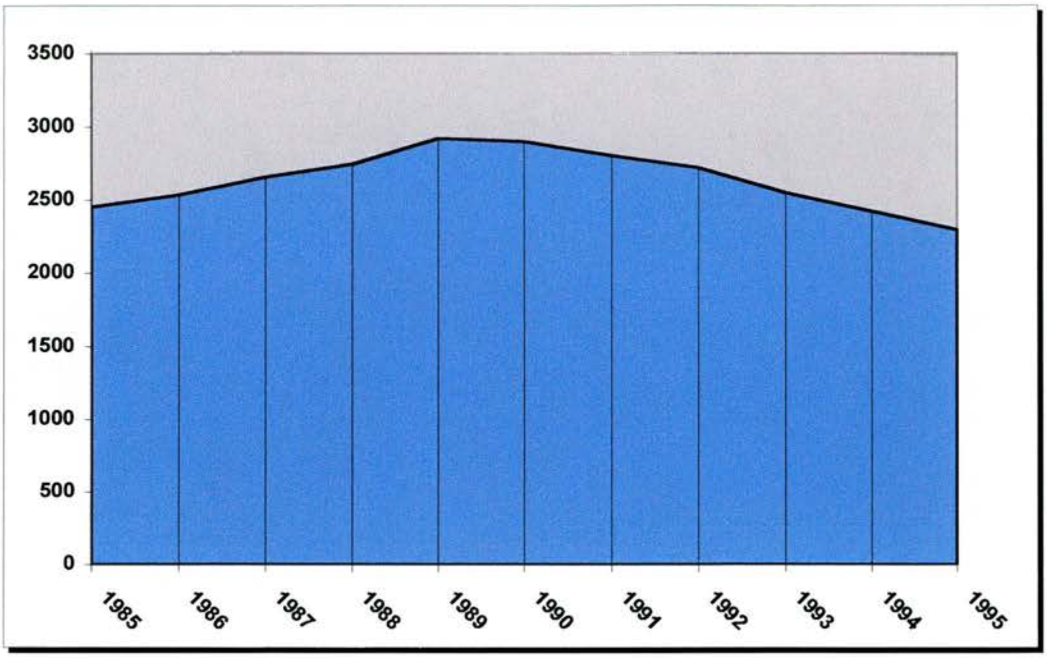
In urban areas, road transport is responsible for nitrogen dioxide concentration levels as high as 75 per cent.[51] Significantly, heavy duty vehicles are responsible for some 50 per cent of total NO_x emissions, which is primarily due to rather poor regulating controls on such vehicles.[52] This is impressive considering the relatively low share of total kilometres travelled by motor vehicle.[53] Caloric power stations and industrial combustion are also important sources, as well as cigarette smoke.[54] Since the introduction of catalytic converters for petrol-engined vehicles, total NO_x emissions have been declining[55]. Eventually, this current rearward trend is, after some lag time, reflected in lower NO_x concentrations especially at urban monitoring sites.[56]

Also, the proportion of emissions generated by petrol-engined vehicles and DERVs has changed in recent years. Estimated figures for 1996 show that diesel fuels account for almost half of the total emission attributed to the road transport sector, while the share of petrol fuels used to be as high as some 70 per cent in previous years.[57] The developments in total emissions of nitrogen oxides are demonstrated in the following figure.

---

[51] See: http://www.scm.tees.ac.uk/local/airnotes.html.
[52] See: OECD (1995a), p. 10.
[53] For a more detailed discussion see explanations in a later section of this chapter.
[54] However, NO and NO_2 are also formed naturally by microbiological processes in the soil as well as during discharges of thunderstorms, giving it the characteristic smell.
[55] Exhaust gas reduction techniques in automobiles with petrol engines, such as the closed-loop three-way catalytic converter, can reduced the proportion of nitrogen oxides contained in a vehicle's exhaust gas by over 90 per cent. See: http://www.umweltbundesamt.de/uba-info-daten-e/dateb-e/nitrogen-oxides.htm.
[56] However, both NO and NO_2 levels are still very much higher in urban areas with roads with heavy flows of traffic.
[57] See: http://www.aeat.co.uk/netcen/airqual/emissions/naeihome.html.

Figure 18: NO_x Emissions Trend UK (in thousand tonnes).



Source: DETR (1998c), p. 38.

As a result of increasing emission predominantly generated by the road transport sector, figures between 1985 and 1990 rose steadily. However, from this peak on, the amounts of emissions have been declining again to a level in 1995 which is 21 per cent lower than the one in 1990. This development is mainly due to substantial falls from both road transport and power stations.[58]

Concentrating on road transport, the figures show, that after reaching a peak in NO_x emissions in 1989 which was 116 per cent higher than their 1970 level, there has been a significant descent since, mainly due to the introduction of tighter emissions standards for both passenger and goods vehicles.

[58] The main reason for the fall from power stations is certainly the increased output from nuclear power station. Additionally, coal-fired plants have been mainly substituted with combined cycle gas turbine stations, with the remainder being equipped with low NO_X burners.

This has also been supported by the new legislation that since 1993 all new vehicles have to be equipped with catalytic converters. As a result, there has been a fall of 22 per cent between 1970 and 1995 of the amount of NO_x emitted per vehicle-kilometre travelled.[59] From time to time, there are episodes of very high NO_2 concentrations, particularly during stable meteorological conditions with cold, still, and foggy winter weather periods, where the airmass is unable to disperse. These episodes usually occur over a period of just a few hours or a few days, and consequently do not result in an exceeding of the current concentration guidelines, since these episodes are too short lived.

The compound class of nitrogen oxides is strongly associated with severe adverse effects on human health. These effects may be of a direct or indirect character. Both nitrogen monoxide (NO) and NO_2 are toxic gases with NO_2 being the more extreme one. Nitrogen dioxide can have some harmful effects on the respiratory tract and the conjunctiva of the eyes.[60] This is even at very low concentrations, where in particular children and the elderly may suffer from breathing failure. The exposure to nitrogen monoxide on the other hand is starting to show this sort of symptoms only at higher concentration levels.

Undoubtedly, nitrogen dioxide is the more serious substance from the group of nitrogen oxides for human health. This is particularly due to the very short dwell time of nitrogen monoxide and hence the fact that it is transformed into NO_2 almost immediately, any extended direct exposure to it is rather unlikely. The effects of NO_2 on human health are twofold, either directly or indirectly.

---

[59] See: Department of Trade and Industry (1999), p. 222.

[60] For a rather comprehensive explanation on the physiological effects of air pollution and the measurement of respiratory functions, see: Lipfert (1993), pp. 92-107.

On the one hand, it is causing respiratory problems and also can reduce the functioning of the lung. As a result, irritations may occur followed by more severe effects such as oedema, or even emphysema. Further, acute exposure to NO_2 may irritate the mucous membrane and favour bronchitis and pneumonia. There is also a higher likelihood of an increased susceptibility to viral infections as well as an increased sensitivity to dust and pollen, especially in people already suffering from respiratory disorders, including asthmatics.[61]

On the other hand, as already noted above, nitrogen dioxide also plays a rather significant role in the formation of ground level ozone. It reacts with hydrocarbons in sunlight to produce photochemical oxidants.[62] These substances are very harmful compounds including so-called peroxyacetyl nitrates (PANs) and ozone.[63] Further, there are also other indirect effects via ammonium nitrate, a component of PM_{10}.[64] Generally, the most serious health effects caused by NO_x, however, are said to arise in combination with other air pollutants such as sulphur dioxide (SO_2) and occur in persons with already existing health problems.

---

[61] See: TEST (1991), pp. 98-100, and Anto and Sụnyer (1995).
[62] See: http://www.mistral.co.uk/nsca/leaf111.htm.
[63] The formation of ozone is subject of a later section in this chapter.
[64] See: Maddison, et al. (1996), p. 71.

3.4.3 Particulate Matter

Particulate Matter is the generic term used for a type of air pollution that embraces complex and varying mixtures of different chemical substances suspended in the atmosphere, which can vary widely in their chemical and physical composition. This mixture can be either of organic or inorganic origin and includes a wide range of solid particles and liquid droplets suspended into the air, such as dirt, soil dust, pollens, moulds, ashes, and soot. Sometimes particulate matter is also referred to as ambient aerosol, which is the suspension of relatively stable solid or liquid particles in the surrounding air.[65] Strictly speaking, however, the term ambient aerosol also includes the surrounding air and reflects particles suspended in gas.[66]

According to its origin, particulate matter is either emitted directly into the atmosphere (primary aerosols) or is the product of a gas-to-particle conversion process (secondary aerosols).[67] While individual particles cannot be seen with the naked eye, collectively they can appear as black root, dust clouds or grey hazes (i.e. ten micrometers are about one-seventh the diameter of a human hair). Generally, they can be distinguished into natural and anthropogenic aerosols, according to their sources.

The following table gives a brief summary of this classification.

---

65 See, for instance, Linster (1990), p. 32.

66 See: Koutrakis and Sioutas (1997), p. 15.

67 Secondary pollutants are usually the formed through chemical reactions involving other pollutants. These reactions usually involve the group of pollutants of NO_x, which explains why most secondary particulate matter occurs as nitrates in the air. Other examples are secondary carbon, which is formed through the emissions of VOC by motor vehicles and particulate matter related to sulphur dioxide. See: Koutrakis and Sioutas (1997), p. 15.

Table 24: Classification of Particulate Matter.

Anthropogenic	Natural
• Direct anthropogenic particle emission (e.g., soot, smoke, road dust, etc.). • Products of the conversion of anthropogenic gases from the combustion of fossil fuels (e.g., sulphate and nitrate formed from oxidisation of SO_2 and NO_2, respectively).	• Sea spray residue and windblown mineral dust. • Volcanic effluvia. • Biogenic materials (mainly particles produced by condensation of VOCs). • Smoke from burning of land biota. • Natural gas-to-particle conversation products (e.g., sulphate generated from reduced sulphur emitted from the ocean surface, marshes, etc.).

Source: Prospero et al. (1983) as referenced in Koutrakis and Sioutas (1997), pp. 15 - 17.

Since the aim of this study is on the examination of traffic-related air pollutants, the main focus will be on anthropogenic particulate matter. Furthermore, airborne particulate matter might have several potentially severe effects, while the main effects of particulates, and hence the main focus of this study is the possible effects of particulate air pollution on human health.[68] However, various terms have occurred in the literature describing particulate matter, which may sometimes lead to some confusion. Hence, for the sake of clarification, it seems appropriate to introduce these different classifications and finally give a definition for the terminology, which is normally employed.[69]

First, the wide range of terminology applied to airborne particulates may reflect the methods used for measuring airborne concentrations. Here, usually Total Suspended Particulates (TSP) and Black Smoke or British Smoke (BS) are differentiated.

[68] Other effects caused by particulate air pollution such as the effects on visibility, the nuisance from soiling and the effects of atmospheric aerosols on climate change are described elsewhere. See: Quality of Urban Air Review Group (1996).

[69] See: Ball, Hamilton, and Harrison (1997), p. 9.

Black Smoke represents the level of particulate matter estimated by the smoke shade technique.[70] This black smoke reflectance technique is used to assess the measurement of the soiling capacity of particular matter, and is a function of both the mass and optical properties of smoke.[71]

However, this technique, which has been used for several decades in the UK, is only most appropriate when coal combustion is the primary source of particulate emissions to the atmosphere, especially since it can be difficult to distinguish amongst shades of black (i.e. saturation of the filters). Therefore, gravimetric particulate measurements have emerged, where the mass of particulates per unit volume of air is measured, i.e. specific physical characteristics such as the measured size of the total suspended particles are considered. Total suspended particulates include all airborne particulate matter measured gravimetrically.[72] The two measurements are, however, not readily comparable, since different properties of the particulate matter, i.e. blackness and mass respectively are being assessed.[73]

However, there is also some kind of general agreement that it is not sufficient enough for the investigation of health related effects using 'total suspended particles' (TSP) or 'suspended particulate matter' for this purpose. This is mainly due to the fact that these measurements depend on the specific measure which is been used for sampling, whereas the important size fraction being sampled are not specific enough.[74] Therefore, there has been a progress in

---

[70] See: Holland et al. (1979), pp. 543-553.

[71] Basically, this method measures the degree of staining of a filter paper.

[72] This method records the gravimetric catch of all particles with less than 50μm diameter on a glass-fibre filter. This measuring method is also known as the US high volume method, developed in The United States in the early 1950s. See, for instance, Holland et al. (1979), pp. 542–543.

[73] However, analysis of BS and PM_{10} measurements show a relationship between the two measurement sets. See: http://www.warwickdceh.demon.co.uk/poll.htm.

[74] See: Quality of Urban Air Review Group (1996), p. 17.

measurements of particulate matter towards size-classified particle mass in order to characterise such pollution. One example for this technology of particle sampling is so-called 'smoke', which has been emitted during a combustion process and includes particulate matter with a mass of less than 15μm.[75]

However, the dominant and now widespread method for the continuous determination of atmospheric particle mass loading are so-called PM_{10}, which includes particles with a median aerodynamic diameter of less than 10μm. Strictly speaking, the PM_{10} fraction is defined as particles that pass through a size-selective inlet with a 50 per cent efficiency cut-off at 10μm aerodynamic diameter.[76]

More recently, so-called particulate matter with a median aerodynamic diameter of less than 2.5μm ($PM_{2.5}$) and ultrafine fractions (< 0.1μm) have become the focus of attention, since it may well be the fact that exposure to these particles may have the most serious adverse effects on human health.[77]

Finally, particulate matter may also be characterised according to the site of deposition in the human body. Hence, inhalable particles are such particles which may be breathed in. Respirable particles, on the other side, are such particles which penetrate deep into the lung. Additionally, sometimes so-called thoracic particles are used to characterise particular matter. These are fine particles in the range which can penetrate the human thorax and are often referred to as PM_{10}.

---

[75] In this context, 'smoke' shall not be confused with the earlier mentioned Black or British Smoke. See: http://www.mistral.co.uk/nsca/fs13.htm.

[76] See: http://www.warwickdceh.demon.co.uk/poll.htm.

[77] In the U.S., measurement of these particles has already been started on a quite regular base, whereas in the UK the adequate samplers are still not used in a routine way. Since mid 1997, Stanger Science and Environment is gathering $PM_{2.5}$ data on behalf of the Department of the Environment, Transport and the Regions at four UK monitoring sites. See: http://www.aeat.co.uk/aqarchive/pm25.html.

Obviously, all three described classifications are somehow linked to each other. Recent studies have identified that particularly the size of the particles determines how they will be transported and react in the human respiratory tract and hence have significant effects on human health.[78] Accordingly, the following table presents the relationship between particle size and respiratory penetration.

Table 25: Relationship between Particle Size and Respiratory Penetration.

Particle Size (in μm)	Respiratory Penetration
> 11	no penetration
7 – 11	penetrate nasal passages
4.7 – 7	penetrate pharynx
3.3 – 4.7	penetrate trachea and primary bronchi
2.1 – 3.3	penetrate secondary bronchi
1.1 – 2.1	penetrate terminal bronchi
0.65 – 1.1	penetrate bronchioli
0.45 – 0.65	penetrate alveoli[79]

Source: Spengler and Wilson (1996), pp. 43-45.

While particles with an aerodynamic diameter up to 100μm can enter the human body through the nose and mouth during breathing, they are highly unlikely to reach deep into the respiratory system and lungs. Most of these larger particles actually settle in the mouth and nose. Only so called very small particles, usually with an aerodynamic diameter below 5μm can actually penetrate the respiratory system beyond the larynx (thoracic fraction) or to the unciliated airways of the lung, the so-called alveolar region (respirable fraction).

[78] Generally, the smaller the fraction size, the deeper it may penetrate into the respiratory tract and lungs.

[79] The alveolis are small air sacs, where oxygen enters the bloodstream.

Currently, the focus of health related sampling and research of particulate matter is mainly on PM_{10}, which may consist of inhalable or respirable particles depending on the actual fraction size. These particles penetrate deep into the respiratory system. They are also believed to irritate the lung tissue as well as damage the lining of the lungs, which then can enable inhaled allergens to pass through to deeper tissues and activate the immune defences. Significant health problems have been linked with the exposure to particulate matter, especially fine particles. These include acute respiratory symptoms, notably aggravated coughing and difficult or painful breathing as well as aggravated asthma. These more acute effects are also represented in increased absences from work or school, and in increased respiratory related hospital admissions and emergency room visits.

Furthermore, long-term disorders, i.e. problems of chronic nature, are also connected with PM_{10}, such as chronic bronchitis and decreased lung functions, which usually results in shortness of breath, have also been connected to an increased exposure to particulate matter. There is also evidence of a strong correlation between exposure to these particulate matter and variations in infant mortality, as well as total premature mortality. Those parts of the population that are certainly most at risk include the elderly, children, as well as asthmatics and asthmatic children.[80]

Additionally, another severe health threat of the exposure to particular matter is due to the fact that they may be in themselves toxic or may carry toxic trace substances adsorbed in their surface.[81] This is especially important since airborne particulates are also associated with the group of so-called polycyclic

---

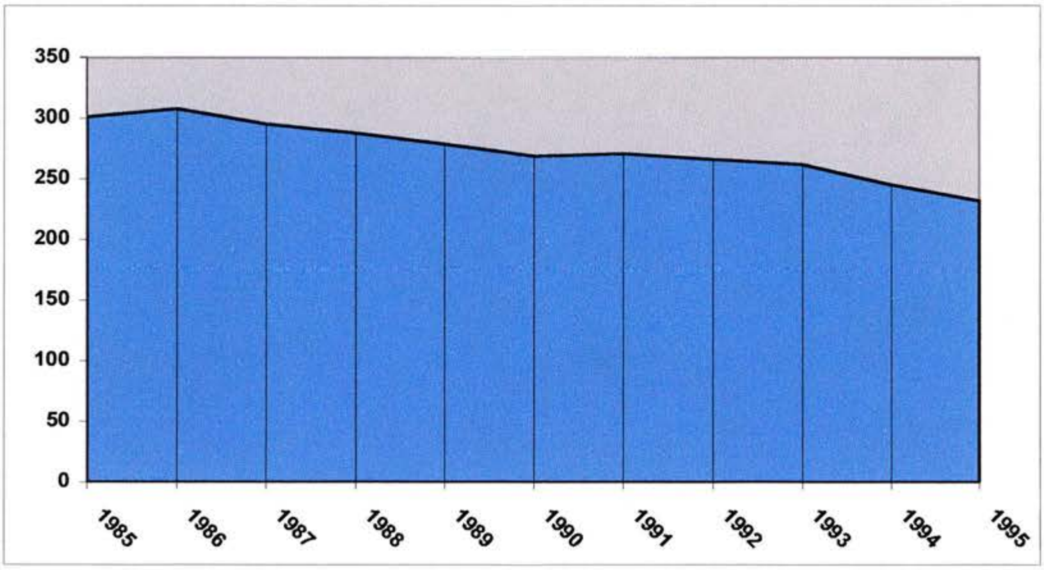
[80] See: U.S. Environmental Protection Agency website at: http://www.epa.gov and the American Lung Association website at http://lungusa.org.

[81] This includes also carcinogenic substances, such as benzo(a)pyrene. See: http://www.env.gov.be.ca/epd/epdpa/ar/fpwtaaht.html.

aromatic hydrocarbons (PAHs), some of which are recognised as carcinogenic. They can be carried deep into the lung by particulates. Furthermore, acid aerosols, which are formed in the atmosphere from nitrogen dioxide as well as sulphur dioxide, are present in the particulate phase and may also cause health problems.[82]

In the UK, concentration data for both Black Smoke and PM_{10} are monitored, while the direct method of measuring particular matter (PM_{10}) has increasingly become the more common measurement. This is due not only because it is the more precise measuring method, but also because most air quality guidelines are based on PM_{10} as a criteria pollutant and hence makes comparison easier. The following figure illustrates the development in PM_{10} emission over the last decade.

Figure 19: PM_{10} Emission Trends UK (in thousand tonnes).



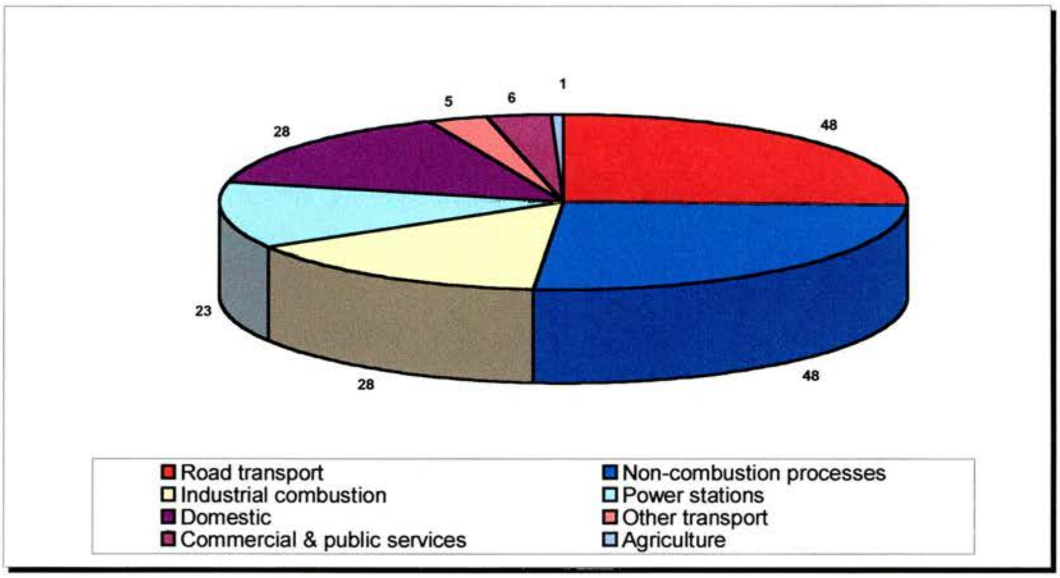
Source: DETR (1998c), p. 36.

[82] See: Holman (1994), p. 5.

The above figure shows a continual downward trend in PM_{10} emissions over time. It is estimated that emissions of PM_{10} have fallen by some 56 per cent since 1970. This is largely due to the fact that the particulate emission from the domestic sector has fallen in the same time period by about 90 per cent.[83] Furthermore, there has been a significant overall change in the proportion of PM_{10} emissions over time.

The following graph illustrates the contribution of various emission sources to the total PM_{10} emissions in 1995 (in million tonnes).

Figure 20: Proportion of PM_{10} Emissions (1997).



Source: DTI (1999), pp. 270-271.

As shown in the figure above, the main sources of PM_{10} emission are non-combustion processes (including construction, mining, quarrying and industrial processes) accounting for 26 per cent. Road transport also accounts for 26 per cent of the total particular matter emission.

---

[83] See: Department of Trade and Industry (1999), p. 221.

However, in the case of urban areas, emissions of particulate matter (particularly PM_{10}) are believed to be predominantly caused by road transportation. The proportion in London, for example, can rise to over 80 per cent, whereas on the average national level, road transport accounts for just under one third of total PM_{10} emissions and stationary combustion sources are much more significant contributors to particulate matter emissions.[84]

With the introduction of unleaded petrol and three-way-catalysts, the emission rates of particulates from petrol-engined road vehicles have experienced a significant reduction.[85] Diesel vehicles are certainly the largest source of such particles/smoke in the UK.[86] The amount of particulates emitted by diesel engines is some 10 to 20 times that of comparable petrol engines running on unleaded fuel.[87]

Particulate matter generated by diesel emission is extremely small, and therefore of special concern, since they are combined with toxic compounds with potential carcinogenicity. A study on potential health effects of diesel fumes could demonstrate an increased incidence of bladder cancer with drivers of diesel vehicles.[88] Additionally, evidence is given that, on a kilometre basis, the mutagenic activity of particulate matter emitted from a diesel engine is about six times greater than from petrol powered vehicles without a catalytic converter. In comparison to petrol engines that are fitted with such a catalyst, it has been found to be up to thirty times higher.[89]

---

[84] Traditionally, domestic coal burning is the major source of particulate matter emissions in the UK, although it is rather restricted nowadays.

[85] However, it has to be mentioned, that these reductions 'only' come as a result of the introduction of emission limits for NO_x, VOC, and CO and not for particulate matter specifically.

[86] See: http://www.doc.mmu.ac.uk/aric/diesel.html.

[87] See: Watkins (1991), p. 21.

[88] See: http://www.doc.mmu.ac.uk/aric/health.html.

[89] See: Holman (1989).

3.4.4 Sulphur Dioxide

The majority of about 90 per cent of sulphur compounds emitted into the atmosphere is due to human activity and is predominantly in the form of sulphur dioxide (SO_2). It is produced during the combustion of fossil fuels containing sulphur and is, together with NO_x, the indicator pollutant for winter smog.[90]

The main sources of SO_2 emissions in the UK are the industry and domestic sector with 36 and 27 per cent of the total emissions respectively, whereas road transport, with just over 6 per cent of total SO_2 emissions plays a relatively minor role.[91]

Diesel fuel contains approximately 10 times more sulphur than petrol, which consequently leads to the result that SO_2 emissions from diesel-powered vehicles are much greater than from petrol engines. The introduction of 'cleaner' fuel, i.e. fuel containing very little or no sulphur has lead to decreasing SO_2 emissions from vehicle exhaust. However, the permitted sulphur content of motor fuels (petrol and diesel) in the UK is still significantly higher than EU averages and the strict Swedish limits.

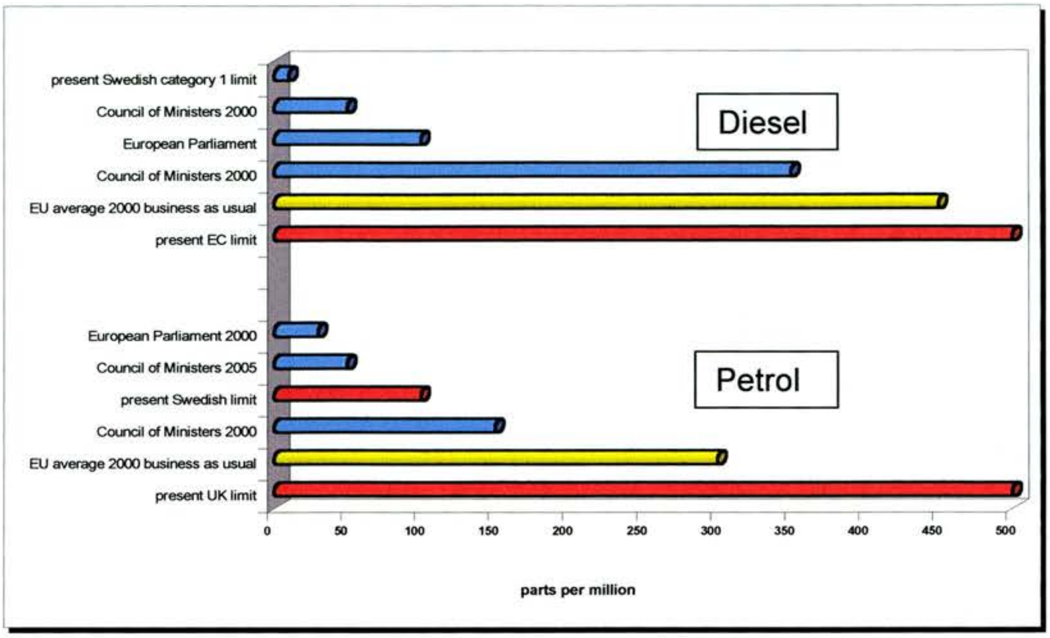
The following chart gives an impression of current and proposed future limits of sulphur content of motor fuels.

---

[90] Winter smog is one type of air pollution episode identified by the Advisory Group on the Medical Aspects of Air Pollution Episodes. See: MAAPE (1995).

[91] See: Department of the Environment, Transport and the Regions (1998c), p. 32.

Figure 21: Sulphur Content in Fuels.



Source: The Royal Commission on Environmental Pollution (1997), p. 25.

The sulphur content of petrol has been reduced from 500 ppm to 100 ppm by weight (from 1 January 1995). Similarly, the maximum allowed amount of sulphur in diesel has been reduced from the original 3000 ppm by weight to 2000 ppm (from 1 October 1994) and finally to 500 ppm (from 1 October 1996).[92] The Council of Ministers as well as the European Parliament have both proposed further reductions for 2000 and 2005. It is, however, remarkable that these proposed limits are still considerably higher than those in Scandinavia, especially with regard to the sulphur levels in diesel fuels.[93]

Also, the Royal Commission on Environmental Pollution noted out that in petrol-engined vehicles fitted with three-way catalytic converters, sulphur increases the time taken for the catalyst to reach its operational temperature after engine ignition, inhibits the reduction of nitrogen oxides, can cause on-board diagnostic systems to malfunction, and can reduce the overall efficiency of the catalytic

---

[92] See: Royal Commission on Environmental Pollution (1997), p. 24.

[93] In Sweden, newly reformulated ultra-low sulphur diesel with a maximum content of 10 ppm is already in use, which is up to 50 times less than the present UK limits.

converter by 10-15%. Further, in continuously regenerating traps fitted to diesel vehicles, sulphur inhibits the formation of nitrogen dioxide and thus its use as an oxidant to burn trapped particulates.[94]

Since SO_2 emission from road transport only comprises about 6 per cent of the national UK sulphur dioxide emissions, they are considered as negligible.[95] However, in terms of potential adverse effects to human health, the situation is a different one. sulphur dioxide is an acid gas, and hence may cause serious harm to people. It can have effects on human health in two different ways, either directly in the form of SO_2 emitted by vehicle exhaust, or SO_2 can have an indirect effect through the formation of particulate matter by its oxidation in the atmosphere. It is assumed that there is a continuous relationship between certain health effects and the dose of SO_2, even at very low concentrations, although to date no specific threshold could be established.[96]

Exposure to SO_2 is related with irritations of the mucous membrane of the respiratory system and the eyes, as well as an increased incidence in bronchitis. Studies in human volunteers suffering from asthma proved that exposure to sulphur dioxide could actually cause a narrowing of the airways of the lung, through the immune response of the airway to the irritant.[97] Due to the fact that sulphur dioxide is a soluble gas, it is eventually removed in the mouth and pharynx and merely affects the upper respiratory tract.

---

[94] See: Royal Commission on Environmental Pollution (1997), p. 24.
[95] See: Department of the Environment, Transport and the Regions (1998c), p. 32.
[96] See: For instance, Holman (1991), p. 11.
[97] See: http://www.med.ed.ac.uk/hew/medical/asthma.html.

As noted above, SO_2 can also attach to the surface of particles, where it forms acidic coatings, and hence be carried deeper into the respiratory tract similar to so called inhalable particles.[98] Moreover, besides the direct impacts of sulphur, it is also a determinant for the amount of particulates produced by diesel or petrol engines as well as other emissions. It is assumed that sulphur dioxide causes most damage when mixed with other smoke particles, such as ozone.

3.4.5 Volatile Organic Compounds

Volatile organic compounds (VOC) is a generic description given to a wide range of substances, including methane, alcohol and various hydrocarbons, which are mainly formed when fuel molecules in the engine do not burn or burn only partially. However, the term volatile organic compounds[99] and hydrocarbons (HC) are frequently used interchangeably.

Strictly speaking, however, the group of hydrocarbons embraces such compounds that are solely formed of carbon and hydrogen, and may be of volatile or involatile nature i.e. not all hydrocarbons are volatile organic compounds and vice versa. Volatile organic compounds, on the other side, are solely volatile species and give car exhaust its characteristic smell.[100] It may, however, also include compounds of carbon involving elements such as oxygen (termed oxygenates), nitrogen, chlorine and sulphur, as well as hydrocarbons.[101]

---

98 See: Lipfert (1993), p. 14.

99 Volatile organic compounds are also known as Reactive Organic Gases (ROG). See: Small and Kazimi (1995), p. 8.

100 Due to the respective diversity of their structure and differences in their reactivity, the rates and products of hydrocarbon oxidation are highly varied.

101 However, hydrocarbons are the main ingredients of VOCs.

Particularly when examining emissions stemming from vehicle exhaust, methane (CH_4) is often excluded, with the remainder termed as non-methane volatile organic compounds (NMVOC).[102]

The two NMVOCs of particular concern are benzene and 1.3–butadiene, with both compounds also falling into the category of hydrocarbons. Benzene (C_6H_6) is a colourless liquid which has a characteristic odour and burning taste. Mixed with some toluene and other compounds, it is added to petrol.[103] Benzene is emitted in vehicle exhaust as unburned fuel as well as a product of the decomposition of other aromatic compounds. Principally, 1.3–butadiene is emitted into the atmosphere from fuel combustion of petrol and diesel vehicles. In contrast to benzene, however, it is not a constituent of the fuel. Rather, it is produced by the combustion of olefins.

The Quality of Urban Air Review Group state in their second report on 'Diesel Vehicle Emissions and Urban Air Quality', that "[t]his group of organic pollutants is perhaps the most difficult to characterise briefly because it contains so many different compounds whose abundance and environmental significance vary widely."[104] Some 400 different organic compounds have been identified in vehicle exhaust so far.[105]

---

[102] One reason why methane is often excluded is that it is less reactive than other VOC components and is hence regularly excluded from regulation. Another reason is that methane is not directly connected with motor vehicle emissions. Rather, it is generated when organic matter is broken down in the absence of oxygen. Examples are the enteric fermentation in cattle and sheep (> 940,00 ktonnes), spreading of animal manure and landfill (> 1720,00 ktonnes) sites, coal mines (> 330,00 ktonnes), oil extraction and gas distribution activities (> 359,00 ktonnes). See: http://www.aeat.co.uk/netcen/airqual/emissions/naeihome.html. Nevertheless, methane is a very important greenhouse gas. There are several other possibilities of distinction, such as VOC versus polycyclic aromatic hydrocarbons (PAHs) which are again a subgroup of toxic organic micropollutants (TOMPs).

[103] The constituent is about 2 % by volume. See Health Effects Instiute website: http://www.healtheffects.org.

[104] Quality of Urban Air Review Group (1993), p. 32.

[105] See: Graedel, Hawkins and Claxon (1986) as quoted in: Ball, Brimblecombe and Nicholas (1991).

Besides many more, the list represents more or less all major classes of organic compounds, including saturated and unsaturated aliphatic hydrocarbons, aromatic hydrocarbons including polycyclic compounds, alcohols, ethers, acids and esters and a number of nitrogen, sulphur and organometallic compounds.

The proportion of VOCs emissions generated by petrol-engined motor vehicles is significantly greater than those from DERVs. The latest estimates for 1996 imply that some 86 per cent of the VOC emissions from road transport are generated by petrol run motor vehicles and the remainder of about 14 per cent is accounted for by DERVs.[106]

The largest source of emissions of VOCs is the sector of road transport with some 35 per cent of total emissions, followed by the use of organic solvents, which accounts for about 28 per cent. Both categories of non-combustion processes and fossil fuel extraction (oil refining) and distribution are, with a share of just over 26 per cent, substantial non-combustion sources.

Similar to the development in emissions of other pollutants, the trend for VOC emission shows a significant decline. Although there has been a substantial fall in VOC emissions from the domestic sector throughout the 1970s and 1980s, the total number was still rising since the mentioned reduction was more than offset by enormous increases mainly in the road transport sector.[107] Similar to other air pollutants, the total emission of VOCs has been declining since the end of 1980s. This is also predominantly due to reduced emissions stemming from road transport arising from tighter emission legislation. In 1995 the total amount emitted has been about 14 per cent lower than in 1990.[108]

---

[106] See: http://www.aeat.co.uk/cgi-bin/w3-msql2/corinair/naeil.html.

[107] Emissions from offshore oil and gas activities were also contributing significantly to the described trend.

[108] This figure, however, is still about 5 per cent higher than the level in 1970.

Besides the direct effect VOCs may have on human health, they play, together with nitrogen oxides, a significant role in the formation of ground level ozone, which in turn is related to serious adverse health impacts. Although the typically observed concentration levels of most volatile organic compounds normally do not present health risks, there are also a number of hydrocarbons, which are assumed to be toxic with the potential to cause cancer. Especially benzene and 1,3 butadiene are proven to be carcinogens.[109] Other VOCs are assumed to have less severe health effects, such as eye irritations and coughing.[110]

3.4.6 Lead

Lead has been introduced as an anti-knocking agent in fuel in order to improve the octane rating. The aim was to achieve higher compression ratios in the engine and hence improved fuel economy and performance. Additionally, being a lubricant, lead helps to improve the durability of exhaust valves and other motor parts. The main sources for lead emissions are the lead from petrol and coal combustion, and metal works. With the recognition of adverse health effects stemming from lead pollution, especially the association with the reduction in development of young children's intelligence in the 1970s, the awareness of traffic related problems has been growing ever since.[111] Hence, the concerns following the scientific knowledge led to the taking of an initiative for specific legislation to reduce the overall emissions and the level of lead in gasoline in particular.

---

[109] See, for instance, Gilli et al. (1996).

[110] Also, VOCs may be oxidised to form oxygenated organic products, which again could appear to be harmful.

[111] See, for instance, Dubourg (1996).

A first step was to reduce the amount permitted in petrol from 0.45 gram per litre to 0.40 gram per litre in 1981. However, this was mainly introduced to comply with the EC directive 78/611. A further step was taken by reducing this level to 0.15 gram per litre in 1985.

Introducing unleaded petrol in the UK in 1985 and the obligation that all new car models have to be capable of running on unleaded petrol has completed the effort to reduce lead emitted by petrol-engined cars. In addition, with the introduction of catalytic converters, it became necessary to ensure that petrol is absolutely lead free, since otherwise the precious metals in the converter would be contaminated.[112]

Hence, the number of motor vehicles that can use unleaded petrol has been rising from just 14 per cent in 1988 to over 71 per cent in 1996.[113] Accordingly, about two thirds of the motorists now actually buy unleaded petrol whereas in 1988 just 5 per cent did so.[114] As a result of this and additional efforts to reduce lead emissions from other sources as well, lead levels in most urban areas have fallen considerably to levels of less than 0.3 $\mu m/m^3$. Lead emissions produced by motor vehicles have decreased by about 87 per cent since 1975. This trend occurs despite an increase in overall petrol consumption of 40 per cent in the same time period.[115]

---

[112] See: Hamilton and Harrison (1991), p. 16.

[113] In 1996, super premium unleaded motor spirit account for just 5 per cent of total unleaded deliveries. See: Department of Trade and Industry (1999), p. 273.

[114] See: Department of the Environment, Transport and the Regions (1998b), p. 34.

[115] See: Department of Trade and Industry (1999), p. 225.

3.5 Secondary Pollutants

Although several pollutants emitted by motor vehicles into the atmosphere present a major health risk in their own right, a significant additional impact results from their oxidation in the atmosphere. This results in the formation of a range of so called secondary pollutants. It is understood that some of these secondary pollutants may have a more harmful potential to human health than their precursors.

Emissions generated by transport and in particular road transport are a major contributor to the production of derived or secondary pollutants in the atmosphere. Above all, the production of photochemical oxidants and smog is the result of the transformation of primary pollutants. The most prevalent of these photochemical oxidants is ozone, which will be described in detail below.[116] The other secondary pollutant, which is principally attributable to transport is nitrogen dioxide, which has already been discussed in a previous section of this chapter.[117]

---

[116] The list of other photochemical oxidants includes such substances as hydrogen peroxyde, peroxyacetyl nitrate (PAN), peroxybenzoyl nitrate (PBzN), as well as some other peroxyacetyl nitrates. However, in this respect the focus of this study will be on ozone especially since the presence of elevated levels of ozone is usually used as an indicator of photochemical smog. Hence these substances are not subject of this study and will not be considered further.

[117] See: The Royal Commission on Environmental Pollution (1994), p. 26. However, there is certainly a wide range of other secondary pollutants. Especially particulate matter has also a secondary component, which is formed within the atmosphere, mainly during summer months, as a result of the oxidisation of gaseous sulphur and nitrogen oxides. However, there will be no further focus on this component, since the 'primary pollutant 'particulate matter' has already been described in detail in an earlier section.

Ozone (trioxygen, O_3) is a pale blue, highly poisonous gas with a very strong odour, i.e. a sweetish-pungent smell. It is a highly reactive allotropic form of oxygen, a so-called triatomic oxygen molecule. Since it 'wants to' give up one extra oxygen atom as well as its extra energy to transfer back to the more stable diatomic oxygen (O_2) and hence can react with virtually any type of biological substance.[118]

Generally, one can distinguish between two different 'types' of ozone, 'high' level and 'low' level ozone. Naturally, there is an omnipresent background ozone concentration in the atmosphere. This natural ozone is produced mainly through the effects of an electrical discharge or high-energy ultraviolet radiation on oxygen in the air of the ozonosphere or (lower) stratosphere, which is some 20 to 30 kilometres above the Earth's surface.[119]

Due to various transportation processes, mainly diffusion, traces of this ozone can also reach the lower atmosphere, the so-called troposphere, whereby the concentration very much depends on the latitude and time of the year.

Alongside this process, however, ozone is also formed in the lower strata of the atmosphere through certain photochemical reactions in polluted air. Consequently, in this lower area, the background level is exceeded by the formation of ozone primarily resulting from a complex series of actions involving oxygen, nitrogen oxides (NO_x), and photochemical reactive hydrocarbon precursors in the presence of strong solar radiation, i.e. sunlight.[120]

---

118 Ozone is certainly the most complex, difficult to contrast and pervasive of the six principal air pollutants included in this description.

119 This stratospheric ozone has an important protective function, as it filters out dangerous low-energy ultraviolet radiation from the sun, which has not been absorbed by O_2 and hence forms a natural protection shield.

120 As described in earlier sections, the most significant emission sources of theses primary pollutants NO_X and VOCs are road transport and stationary sources such as industrial plants.

Particularly, during the summer period with temperate climate conditions and anticyclonic weather conditions, the photochemical activation of becalmed air pollution is enhanced, which in turn causes the typical brown-yellow photochemical smog that frequently occurs particularly in traffic-filled cities.[121] These stable summer anticyclones[122] provide the sunshine needed to drive the chemical reactions and the low wind of around 0.5 meter per second to inhibit any possible dispersion of the emissions. There are also restricted boundary layer depths, which allow the built-up of precursors.

Finally, high temperatures of above 20° C also enhance evaporative emissions of the necessary hydrocarbon precursors and additionally promote certain chemical reactions for the formation of ozone. In the plume of such sources as it moves downwind, ozone formation can occur over a time scale of a few hours to several days.

As a result, ozone concentrations are de-coupled temporally and spatially from precursor sources and ambient concentrations are strongly dependent on meteorological conditions, together with scavenging and deposition rates. In urban areas chemical scavenging by NO_x emissions will result in highly variable ozone concentrations over small spatial scales. Hence, concentrations of ozone will probably be lowest where corresponding levels of other pollutants such as NO are highest, since these urban pollutants have the ability of removing ozone.[123]

---

[121] Since the level of ozone concentrations depends very much on temperature, concentrations in ground level ozone increase in particular during hot weather periods in summer. During these periods, natural background concentrations can be exceeded many times.

[122] Stable summer anticyclones are high weather systems.

[123] As described in a previous section, during the formation of nitrogen dioxide (NO_2), ozone reacts with nitrogen monoxide (NO), which again leads to a reduction in ozone concentrations.

Consequently, ozone concentrations in cities tend to be lower in central areas and increase in the suburbs and more rural areas.[124] In open spaces in urban areas, however, levels of ozone may also approach similar levels to those found in nearby rural areas. Additionally, since the main precursor pollutants nitrogen oxide and hydrocarbons can be carried away from their original source of emission for rather long distances, significantly high concentrations of ground level ozone can be found over large areas. The chemical reaction is roughly described with the following equations.[125]

$$RO_2 + NO \Rightarrow NO_2 + RO \tag{3}$$

$$NO_2 + hv \Rightarrow O\bullet + NO \tag{4}$$

$$O\bullet + O_2 \xrightarrow{\mathbf{M}} O_3 \tag{5}$$

where:

RO_2: HO_2 or an organic peroxy radical, which is formed through the oxidation of hydrocarbons.

hv: radiation ($280 < \lambda < 430$nm)[126], where **h** is Planck's constant and **v** is the frequency.

O•: excited state oxygen atom. When reacting with water vapour, hydroxyl is formed.

M: any molecule such as N_2 or O_2, or other naturally occurring gas. This molecule acts to dissipate the excess energy, which is released during the described reaction, in order to ensure that the reaction is not immediately reversed and hence to prevent the decomposition of ozone.[127]

---

124 The spatial variation is, of course, of very complex nature.

125 The explanations follow mainly European Commission (1995a), pp. 46-47.

126 Usually, λ is at around 315nm. See: Hamilton and Harrison (1991), p. 23.

127 This happens basically by the absorption of any excess energy from the newly formed ozone molecule.

Basically, during the chemical process, NO is converted to NO_2 which in turn is subsequently photolysed to form NO and an oxygen atom. This oxygen atom then combines again with oxygen, which results in the formation of ozone. On the other side, as already mentioned in a previous section, nitrogen monoxide (NO), the other product of this photolysis, reacts almost instantly in a similar photochemical process with ozone to reform nitrogen dioxide (NO_2) and oxygen (O_2).

For the case, where the last described relationship is the dominant loss process of ozone, an equilibrium relationship of O_3, NO and NO_2 can be formulated as follows.

$$[O_3] = \frac{[NO_2]}{[NO]} \times \frac{k_2}{k_4} \tag{6}$$

where:

k_2:	rate constant of photolysion of NO_2.
k_4:	rate constant of the reaction between NO and O_3.
[..]:	concentration of respective gas.

A low NO_2/ NO ratio, for example, implies that the concentration of ozone remains low.[128] According to the reaction shown above it can be said that there is a tendency that high concentration levels of NO_2 coincide with low concentration levels of O_3, and vice versa. Additionally, in stable atmospheric conditions in highly polluted (urban) areas, emission from motor vehicles are the main source for the formation of ground level ozone, since vehicle exhausts contain large amounts of both NO_x and VOCs.

---

[128] In the presence of VOCs, the net amount of ozone produced by the chemical reaction is much greater. VOCs are also broken down into a complex mixture of various substances by the influence of sunlight.

Consequently, the net amount of such ground level ozone formed, is greater in the presence of these pollutants, especially volatile organic compounds, which again are generally broken down under the influence of sunlight into a quite complex mixture of substances.

Ozone can not only damage crops and building materials, but has also a significant impact on human health. Exposure to ozone is related to a number of adverse health effects especially amongst sensitive people. Ozone is not only a severe irritant for the human eyes, but it is also a respiratory irritant to the upper tract and can cause impaired breathing and reduced athletic performance. Due to the fact that O_3 is not soluble in water, it can penetrate to the deeper reaches of the lung and cause increased upper and lower respiratory symptoms such as coughs and chest discomfort.[129] It also supports the exacerbation of asthma and leads to a reduced pulmonary function.[130]

3.6 Current and Future Legislation on Air Quality

3.6.1 Air Quality Bands

The Department of the Environment, Transport and the Regions (DETR) introduced on 19 November 1997 a new public air pollution information system for the UK in order to make air quality more meaningful and easier to survey.[131] The system includes a set of criteria to classify air pollution into so-called bands.

---

[129] The absorption of a gas up in the airways is increasing with the level of solubility of the gas in water. The very water-soluble pollutant SO_2 for example is absorbed mainly in the nose and the upper airways. The less water soluble nitrogen dioxide (NO_2) on the other hand is taken all the way up at all levels of the respiratory system for the most part. The even less soluble ozone is mainly absorbed in the finest airways and the alvedi.

[130] See: Watkins (1991), pp. 68-69, and Koenig (1995).

[131] See: http://www.aeat.co.uk/netcen/airqual/pressrel/press1.html.

The key features of the new revised information system are as follows:[132]

- The respective reports are on 'air pollution' rather than on 'air quality'.
- A change in the way the air quality bands are categorised.
- Five criteria pollutants are covered, including airborne particles (PM_{10}) and carbon monoxide.
- The air quality bands are now based on health and are consistent with the national Air Quality Strategy and the advice from the Committee on Medical Effects of Air Pollutants, which is led by Professor Holgate on behalf of the Department of Health.

A brief description of each band is also provided. Basically, there are four different air quality bands, with each band covering specific concentrations for the respective pollutants. These concentrations are set with reference to what is known about health effects of each pollutant.

The following table illustrates the new and revised set of bands for each of the covered pollutants.

[132] The previous system included hourly information and daily forecasts on the three pollutants sulphur dioxide, nitrogen dioxide and ozone. They were described in four bands, which reflected a 'very good', 'good' 'poor' or 'very poor' air quality.

Table 26: UK Air Quality Bands for Various Air Pollutants.

Air Pollutant	LOW (Standard threshold)	MODERATE (Information threshold)	HIGH (Alert threshold)	VERY HIGH
Sulphur Dioxide (ppb, 15 minute average)	< 100	100 - 199	200-399	> 400
Ozone (ppb)	< 50 (8 hour running average)	50 – 89 (hourly average)	90 – 179 (hourly average)	180 (hourly average)
Carbon Monoxide (ppb, 8 hour running average)	< 10	10 – 14	15 – 19	> 20
Nitrogen Dioxide (ppb, hourly average)	< 150	150 – 299	300 – 399	> 400
Fine Particles (micrograms per cubic metre, 24 hour running average)	< 50	50 – 74	75 – 99	> 100

Source: http://www.aeat.co.uk/netcen/airqual/pressrel/oldband.html.

Starting with the first band, any concentration below the first threshold level, the so-called standard threshold is described as a low level of air pollution. A concentration level between the standard threshold and the so-called information level is described as moderate, and between the information level and the alert threshold as high. Finally, if concentration levels were above the alert threshold, the level of air pollution would be described as very high.

The first breakpoint between the low and moderate band covers levels for each pollutant, which are in line with the air quality standards set in the National Air Quality Strategy.[133] The Department of Health's Committee on the Medical Effects of Air Pollution (COMEAP) adopted the breakpoints between the 'moderate', 'high' and 'very high' air quality bands from the recommendation. These information and alert levels are also in line with the EC Directives on Air

---

[133] The National Air Quality Strategy standards are again based on the recommendation of the UK Expert Panel on Air Quality Standards.

Quality. The proposed banding system is designed as a broad guide to likely health effects and should be seen as such, whereby special emphasis has been made on the gradually increasing risk of effects as concentrations of pollution rise.

The effects associated with the four air pollutant bands are as follows:

Table 27: Potential Health Effects Associated with Air Quality Bands.

Air Quality Band	Health Effect
Low	Effects are unlikely to be noticed even by individuals who know they are sensitive to air pollutants.
Moderate	Mild effects unlikely to require action may be noticed amongst sensitive individuals.
High	Significant effects may be noticed by sensitive individuals and action to avoid or reduce these effects may be needed, e.g. reducing exposure by spending less time in polluted areas outdoors. Asthmatics will find that their "reliever" inhaler is likely to reverse the effects on the lung.
Very High	The effects on sensitive individuals described above for "high" levels of pollution may be worsen.

Source: DETR website at: www.environment.detr.gov.uk.

In their report, the COMEAP emphasise that the risk to individuals enjoying good health is assumed to be very low at all levels of air pollution.[134] However, due to no upper limit in the 'very high' band, it is actually possible that individuals not usually sensitive to air pollution may notice effects when concentrations of air pollutants move well into this band.[135] Also, individuals with already existing health problems, such as chronic chest diseases (e.g.

---

[134] The results, and hence the guidelines given, are limited to the UK.

[135] This includes such effects like eye irritation, coughing and pain on breathing deeply.

chronic bronchitis or emphysema), and heart problems, may be more sensitive to changes in levels of air pollution. Hence, the described general descriptions may actually underestimate the possible health effects if such conditions are present.

For the two bands of 'high' and very high' air pollution, the COMEAP provides some special advice, depending upon whether winter air pollution episodes or summer air pollution episodes are present. During winter air pollution episodes, the levels of nitrogen dioxide, carbon monoxide and particles are likely to be raised.[136]

Even though most people will not experience any adverse health effects at all, those already suffering from lung disease, may experience a worsening of their symptoms.[137] Hence, it is advised to spend less time outdoors and keep warm, additionally they should avoid busy roads during such episodes. During summer air pollution episodes, levels of nitrogen dioxide, ozone and fine partials may increase. The advice provided is almost identical to that for winter air pollution episodes. Further, those who are, in particular, at risk such as the elderly and children should avoid any kind of strenuous outdoor activity.[138]

3.6.2 Main Principles on Air Quality

In order to establish any strategy to improve air quality in specific areas (locally or nation-wide), and hence to reduce possible significant risks to human health, and finally to achieve the wider objectives of sustainable development in relation to air quality, the DETR has published concrete requirements for the

[136] Sometimes sulphur dioxide levels may increase as well.
[137] Older people in particular may be affected.
[138] However, the COMEAP argues, that there is no reason for children with asthma to stay away from school. See: http://www.aeat.co.uk/airqual/pressrel/press1.html.

development of such a strategy. The final version of the National Air Quality Strategy was published on 12 March 1997 and provides a wide range of improved and new regulatory powers for the improvement of air quality. For the sake of completeness, the main principles on which the governmental strategy is based will be reproduced as described by Williams (1997) on behalf of the DETR.[139]

- A statement of the Government's general aims for improving air quality.
- Clear measurable targets. These are based, as close as possible, on an understanding of the health effects of the pollutants concerned, and cost-feasible abatement methods. They have identified timescales for their achievement.
- A balance between national and local action to ensure a flexible and cost-effective approach to air quality management.
- A transparent framework to allow all parties - business, local government and the wider community - to identify the contribution they can make to better air quality.
- The need to ensure consistency with international commitments.
- Regular review of the National Air Quality Strategy and all the elements contained therein.

<u>The key elements of the National Air Quality Strategy are:</u>

- Health-based air quality standards and objectives, to act as reference points by which policies are to be directed.
- A target of 2005 for achievement of the objectives.

---

[139] See: Williams (1997), pp. 86-88.

- Policies for meeting those objectives, including an assessment of the improvements already expected under current policies, and where and how those policies might need to be supplemented.
- The contribution key sectors - in particular industry, transport and local government - can make towards the cost-effective achievement of those objectives.
- A commitment to review the National Air Quality Strategy every three years.

The National Air Quality Strategy provides a specific framework for air quality standards as well as air quality objectives. As described in the previous section, the air quality standards are recommendations made by the Expert Panel on Air Quality Standards for concentrations levels for each atmospheric pollutant at which the risk to human life would be minimised, i.e. extremely small or zero.[140]

Additionally, the Government published air quality objectives shall provide the framework for determining the extent to which certain policies should aim to improve the respective air quality. Currently, they encompass a time period up to 2005. Whereas the general objectives give a fairly rough guideline of what is the ideal achievement in this time, the given specific objectives are measurable and hence an assessment of the progress made can be carried out. Both the current air quality standards for each pollutant (i.e. air quality band) as well as the respective specific air quality objectives, which have to be achieved by 2005, are summarised in the following table together with the outcome of the 1999 review.[141]

---

[140] For the case where such recommendations are still missing, the respective air quality standards are based on the guidelines published by the World Health Organization (WHO).

[141] A report produced for the DETR predicting PM_{10} concentrations in the UK for 2005 and 2010 concludes that NAQS and as well as well as EU limit values particularly at urban background and busy roadside monitoring sites in larger cities are likely to be exceeded. See: Stedman et al. (1998).

Table 28: Summary of UK Air Quality Objectives.

Pollutant	Standard Concentration	measured as	Specific objectives to be achieved by 2005	Outcome of 1999 Review
CO	10 ppm	running 8-hour mean	10 ppm	10 ppm (11.65 mg/m^3) by 31.12.2003.
NO_2	150 ppb	1 hour mean	150 ppb, measured as the 99.9th percentile, to be achieved by 2005*	104.6 ppb (200μg/m^3) by 31.12.2005 (maximum of 18 exceedences)
	21 ppb	annual mean		21 ppb (40 μg/m^3) retained as provisional objective for 31.12.2005. New annual national objective for the protection of vegetation of 15.7 ppb (30 μg/m^3) for 31.12.2000.
PM_{10}	50 μg/ m^3	running 24-hour mean	50 μg/m^3, measured as the 99th percentile, to be achieved by 2005*	New annual objective of 40μg/m3 and 24 hour objective of 50 μg/m^3 (max of 35 exceedences for 31.12.2004). New indicative annual level of 20 μg/m^3 and 24 hour level of 50 μg/m^3 (max of 7 exceedences) for 31.12.2009.
SO_2	100 ppb	15-minute mean	100 ppb, measured as the 99.9th percentile, to be achieved by 2005*	100 ppb (267 μg/m^3) as 99th percentile of 15 minute mean. New 1 hour objective of 131 ppb (350 μg/m^3), not to be exceeded more than 24 times per year, and 24 hour objective of 46.8 ppb (125 μg/m^3), not to be exceeded more than 3 times per year, for 31.12.2004. New annual winter objectives for the protection of eco-systems of 7 ppb (20μg/m^3) for 31.12.2000.
Ozone	50 ppb	running 8-hour mean	50 ppb, measured as the 97th percentile, to be achieved by 2005*	50 ppb (100 g/m^3) retained as indicative level for 31.12.2005.
Lead	0.5 μg/ m^3	annual mean	the air quality standard	0.5 μg/ m^3 by 31/12/2004 0.25 μg/m^3 by 31.12.2008.

Source: The Royal Commission on Environmental Pollution (1997), DETR (1999), p.18 & DETR at website: http://www.environment.detr.gov.uk.

Note: ppm = parts per million; ppb = parts per billion; μg/ m^3 = micrograms per cubic metre: * these objectives are to be treated as provisional, as described above.

In 1999 the introduced standards and objectives on air quality have been reviewed and changed if necessary, especially when new compelling evidence was available which calls for tighter guidelines.[142]

3.6.3 Description of Different Air Quality Standards and Guidelines

As described in the previous section, the UK guidelines and standards of the major urban air pollutants are set according to the recommendations of either the Expert Panel on Air Quality Strategy (EPAQS) or the World Health Organisation (WHO). Since other international organisations also provide such criteria, it seems reasonable to present the main recommendations of them at this stage, in order to integrate the UK levels into an international context and open the possibility for comparison and assessment.[143] For each of the air pollutants discussed in previous sections, a table will be presented where the respective guidelines and standards proposed by different institutions are summarised. In order to present a complete picture, not only the proposed health guidelines are covered but also the guidelines for vegetation where extant. Before describing the guidelines brought forward for the individual air pollutant, the various standards are briefly characterised in the following table.

---

[142] This includes also the consideration of other factors, which may be relevant, such as consumer behaviour (e.g. driving patterns), weather factors, improved emission forecasts, and new scientific and medical evidence on the effects of air pollution.

[143] The guidelines proposed by the WHO are generally not mandatory. However, they serve often as the basis for European Union, as well as national limit values and guidelines.

Table 29: Characterisation of Various Air Quality Standards.

Standard	Characteristics
UK National Air Quality Guidelines (i.e. the UK Government's National Air Quality Strategy; NAQS)	• adopted in 1997 • standards and objectives relating to air quality (see previous section) • based on recommendations made by the Expert Panel on Air Quality Standards • based on public health assessments
UK Air Pollution Bands (set by the DETR)	• air quality information system on levels of pollution for the public
European Community Directives Limit and Guide Values	• compliance of European Union countries on EU framework directive on ambient air quality
World Health Organisation Guidelines	• generally more stringent than EC Directive Limits and Guidelines
United Nations Economic Commission for Europe Critical Levels (UNEC)	• guidelines for individual pollutants • relate to pollutant levels below which vegetation should not be

Source: DETR website: www.roads.detr.gov.uk.

3.6.3.1 Carbon Monoxide

Table 30: Current Guidelines for Carbon Monoxide.

Guidelines set by	Description	Criteria based on (timescale)	Value ppb ($\mu g/ m^3$)
NAQS	Health guideline	running 8-hour mean	10ppm (11.6)
DETR	Low Moderate High Very High	running 8-hour mean	< 10 (11) 10-14 (11-16) 15-10 (17-22) ≥ 20 (23)
WHO	Health Guidelines	15 minute mean 30 minute mean 1 hour mean 8 hour mean	87 (100) 50 (60) 25 (30) 10 (10)

Source: http://www.eea.dk/Document/Topicrep/air/AirHealth/taking%20actions.htm; http://www.doc.mmu.ac.uk/aric/st&gd2.html; http://www3.mistral.co.uk/cleanair/fs1-2.htm; http://www3.mistral.co.uk/cleanair/fs1-3.htm; http://www3.mistral.co.uk/cleanair/fs1-6.htm; http://www3.mistral.co.uk/cleanair/fs1-5.htm; http://www.umweltbundesamt.de/uba-info-daten-e/daten-e.htm.

3.6.3.2 Ozone

Table 31: Current Guidelines for Ozone.

Guidelines set by	Description	Criteria based on (timescale)	Value ppb ($\mu g/m^3$)
NAQS	Health Guideline	running 8 hour mean	50 (100)
DETR	Low Moderate High Very high	running 8 hour mean hourly mean hourly mean hourly mean	< 50 (100) 50–89 (100–179) 90–179 (180–359) ≥ 180 (360)
European Council EC-guideline 92/72/EEC	Population information threshold	1 year (98th percentile of hourly means	100 (200)
	Population warning value	1 hour mean	90 (180)
	Health protection threshold	1 hour mean	180 (360)
	Vegetation protection threshold	fixed 8 hour means (hours 1-9; 9-16; 17-0; 13-20)	55 (110)
	Vegetation protection threshold	1 hour mean 24 hour mean	100 (200) 32 (65)
WHO	Health guideline	1 hour mean	76-100 (150-200)
	Health guideline	running 8 hour mean	50-60 (100-120)
	Vegetation guideline	1 hour mean	100 (200)
	Vegetation guideline	daily mean	33 (65)
	Vegetation guideline	growing season mean*	30 (60)
UNEC	Vegetation guideline	growing season mean*	25 (50)
	Vegetation guideline	1 hour mean	75 (150)
	Vegetation guideline	running 8 hour mean	30 (60)

Source: http://www.eea.dk/Document/Topicrep/air/AirHealth/taking%20actions.htm; http://www.doc.mmu.ac.uk/aric/st&gd2.html; http://www3.mistral.co.uk/cleanair/fs1-2.htm; http://www3.mistral.co.uk/cleanair/fs1-3.htm; http://www3.mistral.co.uk/cleanair/fs1-6.htm; http://www3.mistral.co.uk/cleanair/fs1-5.htm; http://www.umweltbundesamt.de/uba-info-daten-e/daten-e.htm; http://www.wcceh.gov.uk/polldata/newair.htm.

Note: (*) = Growing season is defined as April to September for WHO guidelines, but as daytime (0900-1500) April to September for UNECE guidelines.

3.6.3.3 Nitrogen Dioxide

Table 32: Current Guidelines for Nitrogen Dioxide.

Guidelines set by	Description	Criteria based on (timescale)	Value ppb ($\mu g/m^3$)
DETR	Low Moderate High Very high	hourly average	< 150 (282) 150–299 (282-563) 300–399 (564-751) ≥ 400 (≥752)
NAQS	Health guideline	1 hour mean annual mean	150 ppb 21 ppb
		<u>Calendar year of data:</u>	
European Council	Limit Value	98th percentile of total frequency of hourly means	104,6 ppb (200)
EC-Guideline 85/203/EEC Appendix I	Guide Value	98th percentile of total frequency of hourly means	70,6 ppb (135)
EC-Guideline 85/203/EEC Appendix II	Guide value	median value of daily averages	26,2 ppb (50)
WHO	Health guideline	1 hour mean	210 ppb
	Health guideline	daily mean	80 ppb (200)
	Vegetation guideline	running 4 hour mean	50 ppb
	Vegetation guideline	annual mean	16 ppb
Draft WHO recommended guideline	Health guideline Health guideline	1 hour mean annual mean	110 ppb 21–26 ppb (40)
UNEC	Vegetation guideline	annual mean	15 ppb

Source: http://www.eea.dk/Document/Topicrep/air/AirHealth/taking%20actions.htm; http://www.doc.mmu.ac.uk/aric/st&gd2.html; http://www3.mistral.co.uk/cleanair/fs1-2.htm; http://www3.mistral.co.uk/cleanair/fs1-3.htm, http://www3.mistral.co.uk/cleanair/fs1-6.htm; http://www3.mistral.co.uk/cleanair/fs1-5.htm; http://www.umweltbundesamt.de/uba-info-daten-e/daten-e.htm.

3.6.3.4 Sulphur Dioxide

Table 33: Current Guidelines for Sulphur Dioxide.

Guidelines set by	Description	Criteria based on (timescale)	Value ppb ($\mu g/m^3$)
DETR	Low		< 100 (266)
	Moderate	15 minute mean	100–199 (266–530)
	High		200–399 (532-1061)
	Very high		≥ 400 (1064)
NAQS	Health Guideline	15 minute mean	100 (266)
European Council	Limit value	pollution year (median of daily values)	30 (80) if smoke * > 34 45 (120) if smoke ≤ 34
	Limit value	winter (median of daily values Oct. – Mar.)	49 (130) if smoke > 51 68 (180) if smoke ≤ 51
	Limit value**	pollution year (98th percentile of daily values)	94 (250) if smoke > 128 131 (35) if smoke ≥ 128
	Guide value	pollution year (mean of daily values)	15-23 (40–60)
	Guide value	24 hours (daily mean value)	38–56 (100–150)
WHO	Health guideline	10 minute mean	175 (500)
	Health guideline	1 hour mean	122 (350)
	Vegetation guideline	daily mean	38 (100)
	Vegetation guideline	annual mean	10.4 (30)

Source: http://www.eea.dk/Document/Topicrep/air/AirHealth/taking%20actions.htm; http://www.doc.mmu.ac.uk/aric/st&gd2.html; http://www3.mistral.co.uk/cleanair/fs1-2.htm; http://www3.mistral.co.uk/cleanair/fs1-3.htm; http://www3.mistral.co.uk/cleanair/fs1-6.htm; http://www3.mistral.co.uk/cleanair/fs1-5.htm; http://www.umweltbundesamt.de/uba-info-daten-e/daten-e.htm.

Note: (*) = Limits for Black Smoke (BS) are given in ug/m^3 for the BSI method as used in the UK. The limits stated in the EC Directive relate to the OECD method, where OECD = BSI/ 0.85.

(**) = Member states must take all appropriate steps to ensure that three consecutive days do not exceed this limit value.

3.6.3.5 Thoracic Particulates

Thoracic particulates is a term which relates to the site of deposition in the lower respiratory tract. Smaller particulates can penetrate further into the lungs, usually particles of 2.5 microns (um) in diameter or less. The EPAQS guideline refers to PM_{10}. Some of the larger particles measured by this sampling process would not be able to penetrate deep into the lungs.

Table 34: Current Guidelines for Particular Matter.

Guidelines set by	Description	Criteria based on (timescale)	Value $\mu g/m^3$
DETR	Health guideline (PM_{10})	running 24 hour mean	50 (provisional measured as 99[th] percentile)
NAQS	Low (PM_{10}) Moderate (PM_{10}) High (PM_{10}) Very High (PM_{10})	running 24 hour mean	< 50 50– 74 75– 99 ≥ 100
WHO	Health guideline	running 24 hour mean	70 (if $SO_2 \geq 125$ (48ppb))
EC Air Quality Standards	Health guideline (PM_{10}) Health guideline (PM_{10})	running 24 hour mean yearly average	50 (not to be exceeded more than 7 times a year) 30 (20 by January 2010)

Source: http://www.eea.dk/Document/Topicrep/air/AirHealth/taking%20actions.htm; http://www.doc.mmu.ac.uk/aric/st&gd2.html; http://www3.mistral.co.uk/cleanair/fs1-2.htm; http://www3.mistral.co.uk/cleanair/fs1-3.htm; http://www3.mistral.co.uk/cleanair/fs1-6.htm; http://www3.mistral.co.uk/cleanair/fs1-5.htm; http://www.umweltbundesamt.de/uba-info-daten-e/daten-e.htm.

3.7 Summary

The main aim of the discussion in this chapter was to characterise the main road traffic related air pollutants. For this purpose, first a brief general description of the respective pollutants has been given, followed by the exposition of the proportional split of the respective pollutants amongst the various sectors, with the main emphasis on the connection to the transport sector, and road transport in particular. In this, past and current trends have also been presented. Second, an initial, rather brief and general description of the potential health effects of each air pollutant has been given, basing on results of empirical studies. Finally, a short overview on the current legislation in terms of air quality as well as the proposed future developments in this area is presented.

This section will summarise some of the main points discussed previously, which will also serve as further analysis in later chapters of this study. Since the primary aim of this study is to valuate traffic-related health costs, it is essential to take the relative contribution of different sectors to the total national emissions of the relevant pollutants into consideration. The following table presents the percentage contribution from road transport of various air pollutants of the national UK emission level in 1997, as well as the percentage contribution of the pollutants in London.[144]

---

[144] Since ozone is not emitted directly into the atmosphere, but is rather a so-called secondary pollutant, which is formed by nitrogen oxide and VOC, it is not directly covered by the National Air Quality Strategy.

Table 35: Contribution of Road Transport to UK Air Pollutant Emissions (1997).

Pollutant	% of national emissions	% of emissions in London
CO	75	97
NO_x	48	75
PM_{10}	26	78
SO_2	2	23
Lead	61	N/A
VOC	30	60

Source: http://www.roads.detr.gov.uk/cvtf/impact/3.htm.

Road transport is one of the major sources of air pollution not only on a national level, but also especially in urban areas, where the contribution from road transport to national air pollution emissions is expected to be significantly higher.[145] Accordingly, the extent to which road transport is a significant source of specific pollutants at any place and time varies depending very much on the respective traffic and weather situation as well as emissions from other local sources.

As seen in the above table, in urban areas the contribution of road transport to overall emissions can be considerably higher than the national figures, especially for NO_x and PM_{10}. For large conurbations, it assumed that about three-quarter of all PM_{10} and NO_x emissions are in fact from road transport, while in some areas this share may even be significantly higher.[146]

[145] An exception is sulphur dioxide, which is almost exclusively caused by industrial combustion.

[146] See Department of the Environment, Transport and the Regions website at http://www.detr.gov.uk.

Additionally, the total amount of traffic related emission could again be subdivided into emissions generated by petrol cars and those from diesel cars. Diesel cars have very different emission characteristics which could mean that an increase in the number of licensed diesel cars at the expense of petrol cars, e.g. through the promotion that they are cleaner than petrol cars by the motor industry, could have important implications on urban air quality.[147]

However, due to the lack of measurements of emissions from both petrol and diesel fuels, it is rather difficult to conclude which engine type is cleaner. Although diesel cars produce less CO and VOCs, they not only have the higher emissions of nitrous oxides than petrol cars with catalytic converters and lambda sensors, but also produce all the particulate matter generated from road transport.[148]

The following table presents a comparison of the emissions of air pollutants from various types of motor vehicles.

---

[147] One main reason why the private vehicle manufacturing industry favours the direct injection diesel engine is the fact that due to an increasing weight given to carbon monoxide and hence to an overall reduction in fuel consumption, it seems to be the suitable engine.

[148] Furthermore, the lack of an adequate maintenance of a diesel car can increase the amount of emission of particulate matter by 5 to 10 times. See: http://www.umweltbundesamt.de/uba-info-daten-e/daten-e/diesel-engine.htm.

Table 36: Types of Motor Vehicles Contributing to Air Pollution.

Pollutant	Petrol car without a catalyst	Petrol car with a catalyst	Diesel car without a catalyst	Diesel car with a catalyst
Carbon monoxide	****	***	**	*
Nitrogen Oxides	****	*	**	***
Particulate Matter	**	*	****	***
VOCs	****	**	***	*
Sulphur Dioxide	*	*	***	****

Source: QUARG (1993), p. 15.

Key: Asterisks indicates which type of cars has typically the highest emissions: *Lowest emissions: **/*** Intermediate: **** Highest emissions.

Note: The table only indicates the relative order of emissions between the different types of vehicle. No attempt has been made to quantify the emissions. The difference in emissions between, say, **** and *** may be an order of magnitude, or much smaller.

The benefits of diesel cars relative to petrol cars fitted with a three-way catalyst are that they typically generate less carbon monoxide and give rise only to little evaporative and exhaust emissions of hydrocarbons.[149] They also intrinsically operate with higher fuel efficiency. However, diesel cars produce higher emissions of oxides of nitrogen and particular matter relative to three-way catalyst cars.[150] Especially in terms of particulate matters, the Quality of Urban Air Review Group (QUARG) states that "the net result is that in urban areas where road transport is the principal source of airborne particulates concentrations are likely to remain similar to the levels found today [1993], despite an improvement in abatement technology."[151]

[149] This is due to the fact that diesel fuel is much less volatile than spark ignition petrol engines. See: Quality of Urban Air Review Group (1993), p. 13.

[150] However, a large proportion of new-car diesels is now fitted with oxidation catalysts, which actually help to reduce particulates. Additionally, more sophisticated electronic engine management systems and gas re-circulation systems as well as so-called after treatment particulate traps to filter these emissions further are being developed. The latter are already available as bigger versions for buses and lorries.

[151] Quality of Urban Air Review Group (1993), p. 13.

Additionally, although forecasts for urban road transport emissions of the main air pollutants indicate an initial reduction until 2015, respective figures show an increase thereafter, mainly due to continuous and rapid increase in the number of motor vehicles and the number of kilometres driven, offsetting the reduction in emission.[152]

Since a main emphasis of this study lies in the analysis of the relationship between traffic related air pollution and potential adverse health effects, a brief overview of such effects, which have been already mentioned in previous sections of this chapter, shall be given in the following. A review of the results of empirical studies that have been analysing this relationship will be presented in a later chapter of this study. Although in 1979, the then Ministry of Transport still told the Clean Air Council that "the effects of pollution from motor vehicles can be summarised: There is no evidence that this sort of pollution has any adverse effects on health",[153] the situation has changed dramatically since. A number of studies have shown a significant relationship between traffic fumes and various health effects.

As Walters (1994) states, the main traffic-related air pollutants, i.e. particulate matters, nitrogen dioxide and ozone are all strongly associated with "a fall in lung function and an increase in respiratory problems in healthy people as well as those with asthma".[154] Additionally, increased death rates from heart and lung disease have been recognised.[155] The following table presents a summary of the

---

[152] See: Department of the Environment, Transport and the Regions website at http://www.enviornment.detr.gov.uk/airq/consukt/naqs.htm. See also traffic forecasts as presented in the previous chapter.

[153] Green (1994), p. 3.

[154] Walters (1994), p. 11.

[155] Cohen and Pope (1995), for instance, show that air pollution may be responsible for increases in lung cancer.

main adverse health effects resulting from the traffic-related air pollutants analysed in this study.[156]

Table 37: Summary of Health Effects Associated with Air Pollutants.

Pollutant	Health Effect
CO	Lethal at high doses (asphyxiation). At low doses can impair concentration and neuro-behavioural function. Causes headaches, nausea, fatigue followed by unconsciousness. Levels above 2000 ppm may prove fatal in less than 30 minutes. Increases the likelihood of exercise related heart pain in people with coronary heart disease. May cause heart disease and present a risk to the foetus. Chronic exposure may accelerate atherosclerosis or precipitate coronary vessel spasm and may cause gastrointestinal symptoms.
NO_x	Attacks membranes of the respiratory organs and increase the likelihood of respiratory disease. May exacerbate asthma and possibly increase susceptibility to infections. Is important substance in the formation of ozone and other photooxidants (e.g. photochemical smog).
PM_{10}, TSP, BS	Associated with a wide range of respiratory symptoms. Linked to increased hospital admissions and emergency room visits for respiratory problems. Can cause wheezing and other symptoms in people with asthma or sensitive airways. Long-term exposure is associated with an increased risk of death from heart and lung disease. Particulates can carry carcinogenic materials (PAHs) into the lungs. Increase in total mortality as well as in mortality from respiratory or cardiac disease. Increases in respiratory, and cardiac hospitalisation. Increases in the daily prevalence of respiratory symptoms. Increases in ER visits for asthma and other respiratory conditions. Increases in functional limitation as reflected by restricted activity days and, in children, by school absenteeism.
SO_2	It constricts air passages, which is in particular a problem for people with asthma and for young children whose small lungs need to work harder than adult lungs. May provoke wheezing and exacerbate asthma, even at a brief exposure to relatively low levels. It is also associated with chronic bronchitis.
O_3	Irritates the eye and air passages. Reacts with lung tissue and can inflame and cause harmful changes in breathing passages. Decreases the lungs' working ability and cause both coughing and chest pain. Increases the sensitivity of the airways to allergic triggers in people with asthma. May increase susceptibility to infection.

Source: Gee (1997), p. 39 & Ewetz and Camner (1983), pp. 111-124.

[156] See also Green (1994), pp. 2-4; and California Environmental Protection Agency at website http://arbis.arb.ca.gov/health/health.htm.

These adverse effects on human health due to the exposure to air pollution may result in economic costs that arise from the admission to hospital, medical treatments, or the loss of life. However, although the sources of these emissions are generally known, such as the road users, the respective costs associated with this activity are not entirely borne by the individuals involved.

In other words, the individual road users do not consider these costs when deciding whether to make a journey and how often, because they are either unaware of these costs or are not willing to take them into account when making their decision. These costs are not (entirely) included in the market prices the road users have to pay, and hence are assumed to be so-called external costs to the society or economy. These external effects are the main focus of discussion in the following chapter.

4. Analysis and Discussion of the Notion of Externality

4.1 Introduction

As mentioned in the previous chapter, the exposure to air pollution may result in severe adverse effects to human health. These effects often result in costs. However, as mentioned before, the source of pollution, i.e. the road users, often have no obligation to cover these costs and hence do not take them into account when making decisions on when and how often to use their motor vehicles; the polluter pays principle does not hold. In other words, these costs associated with the emission of air pollutants are a burden for society as a whole rather than privately covered, and defined as external effects or externalities.

The issue of externalities has a long tradition in economic literature. In 1890, Alfred Marshall first introduced the concept of external economies and diseconomies.[1] Since then the concept has been widely discussed and various definitions and interpretations have emerged in the economic literature. Probably the main reason for the vigorous debate has been the issue of the reality of external effects.[2] Surprisingly, however, there still seems to be no generally accepted approach to the meaning attributable to the notion of externality.

[1] See: Marshall (1898).

[2] Clapham (1922), for instance, argues that the original idea proposed by Marshall is nothing else than an 'empty box'. His argument was, that, what Marshall called external economies, is a category, which could not be associated with any real world phenomena and was therefore empty. In the long and rather controversial discussion over the significance of what came to be known as externality, economists tried to fill this empty box with applications of real world phenomena, such as pollution, congestion, etc. Furthermore, attempts have been made to find some sort of rationale for what the actual content of the box should be. Two examples of several additional publications concerning this matter are Pigou (1922), pp. 485-465; & Robertson (1924), 34, pp. 16-31.

Indeed, there exists fundamentally different attempts to characterise the notion of external effects, which have occurred in the literature over the years. This is remarkable, especially concerning the amount of work that has been undertaken on theoretical aspects of external effects and hence the extensive use of the notion of externality. Cornes and Sandler's (1996) comment sufficiently represents the situation:

> There is a strong temptation to avoid giving an explicit definition of *externality*, since even the first step has been a fertile source for controversy, and instead to approach the matter rather obliquely by putting to work various models in each of which an externality is obviously present.[3]

Thus, since there seems to be no single clear-cut definition for the term externality, it seems appropriate to give an overview of some of the main approaches made on this topic in economic literature over time. Based on this, a more formal definition of the notion of externality will then be elaborated upon. This also includes a more detailed look on several classifications and specifications of externalities. The results of this formal discussion will then serve as a basic background for the final characterisation of external effects arising in the transportation sector as presented in the following chapter. Thus, the choice of certain definitions and interpretations concerning the notion of externality applied to the transport sector can be elucidated more decisively from the preceded detailed discussion. The outcome of an analysis of possible external effects in the transport sector depends very much on the underlying definition and interpretation of the notion of externality. This is true, especially in the case of alleged external benefits of transportation.

---

[3] Cornes and Sandler (1996), p. 39.

4.2 Historical Overview of the Notion of Externality

As mentioned above, in his "Principle of Economics" Alfred Marshall (1890) introduced the notion of external effects for the first time to the economic literature. His main ambition was to explain how the impact of a production with increasing returns to scale could still exist in a model of competitive efficiency of the market. Although these increasing returns to scale are typically not compatible with a market structure, which is highly competitive and atomised, they still can be observed in the real world.

Marshall points out:

> We may divide the economics arising from an increase in the scale of production of any kind of goods, into two classes - firstly, those dependent on the general development of the industry; and, secondly, those dependent on the resources of the individual houses of business engaged in it, on their organization and the efficiency of their management. We may call the former *external economies*, and the latter *internal economies*.[4]

In other words, by distinguishing between internal and external economies, he tried to illustrate that a competitive industrial structure could coincide with the existence of increasing returns to scale.[5] He determined that external economies represent an alternative source of inefficiency.[6] According to his statements, this is given if they are seen as external (the size of the industry) and not in the contract of a single producer. The industry would then not be exposed to single firms' monopolistic strategies.

---

[4] This quote is taken from the 4th edition of "Principles of Economics" published in 1898, pp. 344-345. However, it has been traced back to the very first edition in 1890. See: Papandreou (1994), p. 14.

[5] Marshall pointed out that increasing returns are mainly caused by the 'general development of the industry' which is producing external economies.

[6] Marshall did not investigate inefficiencies associated with monopolies, even though he discussed theoretical aspects of monopolies. See: Marshall (1898), pp. 537-553.

Marshall's idea was then picked up by Pigou (1917)[7], who refined and developed it further.[8] Basically, his main contribution was that externalities are given if the marginal social product and the marginal private product differ. Such a divergence would arise if the costs or benefits resulting from the agents' activities were actually not completely incorporated into their way of action; whether or not the agents are aware of the total amount of the total cost or benefits. Using Pigou's words, the problem has been described as follows:

> It might happen, for example, [. . .] that costs are thrown upon people not directly concerned, through, say, uncompensated damage done to surrounding woods by sparks from railway engines. All such effects must be included - some of them will be positive, others negative elements - in reckoning up the social net product of the marginal increment of any volume of resources turned into any use of place. Everything of this kind must be counted in.[9]

In line with Marshall, he could demonstrate that external benefits (which are equivalent to downward-sloping supply curves in the industry) will lead to an output level which is below the socially optimal output level. Likewise, in the case of external diseconomies (which are equivalent to upward-sloping supply curves or increasing costs in the industry), the output levels are greater than the socially optimal one. In both cases, a social optimum is not given.

Thus, in looking to a remedy of the problem, Pigou pointed out, that "it is, however, possible for the State, if it so chooses, to remove the divergence in any

---

[7] The first edition of Pigou's "The Economics of Welfare" was published in 1917. However, the references made in this study are derived from the 4th edition published in 1932.

[8] Astonishingly, however, the concept developed by Pigou has been done without any reference to Marshall's work.

[9] Pigou (1932), p. 134. In this context Pigou pointed out, that the fact that a single company in a certain industry experiences an increase in the quantity of resources employed, may lead to a rise of external economies in the concerned industry as a whole. Due to this development, the real product costs of a given output might be lowered.

field by 'extraordinary encouragements' or 'extraordinary restraints.'"[10] Consequently, in order to restore a social equilibrium where social marginal costs and private marginal benefits equal, the placing of a subsidy (in the case of external economies) or levy of a tax (in the case of external diseconomies) respectively, would guarantee such equilibrium. At this point social welfare would be maximised. In doing so, the tax or subsidy respectively should equal the difference between the social and private marginal costs.

However, the analysis brought forward by Marshall and Pigou suffered from some rather serious weaknesses. There has been a vigorous and, for many years, ongoing discussion concerning their analysis, which ultimately revealed flaws and finally refuted some of the original results. This discussion also made an important contribution to clarify previous misunderstandings and supposed absurdities entailed by the sometimes very vague and rather imprecise use of terminology and formulations. Cheung (1973) expressed this by remarking that "[t]he popularity and acceptance of Pigou's thesis is rather remarkable since much of his original analysis makes difficult and confusing reading"[11]. Likewise, Coase states:

> It is strange that a doctrine as faulty as that developed by Pigou should have been so influential, although part of its success has probably been due to the lack of clarity in the exposition. Not being clear, it was never clearly wrong. Curiously enough, this obscurity in the source has not prevented the emergence of a fairly well-defined oral tradition [. . .]. I propose to show the inadequacy of this Pigovian tradition by demonstrating that both the analysis and the policy conclusions which it supports are incorrect.[12]

[10] Pigou (1932), p. 192. However, as Pigou mentioned, this is only for the case, where the divergence cannot be raised through a modification of the contractual relation between the (two) parties involved.

[11] Cheung (1973), p. 24.

[12] Coase (1960), p. 39. The approach suggested by Coase will be described in more detail in a later part of this section.

Several people including Clapham (1922), Robertson (1924), Knight (1924), Young (1913), Viner (1931), Robinson (1941), and later Coase (1960) led the debate. In particular, it could be refuted by Young (1913) and Knight (1924) that treating all upward sloping supply curves as an indication of inefficiency would be the right approach.

In other words, whether or not a competitive equilibrium and social optimum diverge is not actually caused by inclined supply curves. Following Knight (1924), the reason for a divergence between the private net product and the social net product may be rather the fact that a natural resource, which is of scarce nature, has been exploited in a rather wasteful way.[13] Also, considering the matter who is actually causing an externality, Pigou simply assumed that externalities are unilateral and hence making only one party liable for the externality, the necessary and sufficient conditions to restore an optimal outcome are assumed to be fulfilled with the imposing of a negative tax.

Coase, however, argued that there is no reason why externalities should not be reciprocal rather than just blaming one side as opposed to the other. The party which could change its behaviour cheapest, and hence whose modified situation is also cheapest for society to bear should be the responsible one.[14] Vatn and Bromley (1997) also claim that it is incorrect to say that a Pigovian tax is a least cost measure, since it is not always most efficient when the party which caused the externality is made liable and hence taxed.[15]

---

[13] Knight (1924) disproved Pigou's example in which external diseconomies arise from the congestion of the better of two roads, as drivers decide on the margin which road to use. According to him it is essentially a case where an open access and scarce resource is used and in fact not a problem of decreasing returns. See: Knight (1924), p. 386.

[14] See: Coase (1960), pp. 34-35. Interestingly, Coase also pointed out that even Pigou did actually not apply the 'Pigovian rule' consistently in his work.

[15] From a moral perspective, however, this might well be recommended.

Additionally, they argue that "the Coasian allegation that the Pigovian tax may impose costs on the 'wrong' party - that the net social dividend sometimes would be enhanced if the 'victim' would have to take action - bears substantial merits"[16].

In line with Coase, they could show that making the emitter of pollution for example exclusively responsible and hence put a tax on this side of the conflict, may induce that 'too many' victims will move into that area.[17] With the publication of Viner's seminal paper 'Cost Curves and Supply Curves', the long ongoing, rather confusing and sometimes very controversial debate on the issue of increasing returns to scale could be clarified. He basically identified the involved effects related to increasing returns as pecuniary externalities and could consequently show that the analysis of what is usually called externality is decisively different from the discussion concerning increasing returns to scale. Papandreou pointed out, that "it had become quite clear that the inefficiency had little to do with the slope of the supply curve as such, but with the divergence of the 'perceived' or 'effective' supply curve from the actual or correct one."[18] Hence, the idea of externalities (external economies and diseconomies) should be separated from the concept of increasing or decreasing returns to scale as it was originally proposed by Marshall.

In order to solve this 'problem', Viner (1931) suggested the separation of the notion of externalities into the two categories of technological and pecuniary externalities.[19] He states that it was "the primary purpose of [his] article to develop a graphical exposition of the manner in which supply curves are

[16] Vatn and Bromley (1997), p. 141.
[17] See: Vatn and Bromley (1997), p.144 and Coase (1960), p. 42.
[18] Papandreou (1994), p. 23.
[19] See: Viner (1931).

dependent upon the different possible types of technological and pecuniary cost situation"[20].

Whereas technological externalities represent a source of inefficiency and hence market failure, pecuniary externalities in no way indicate inefficiency, as it will be illustrated in a later section of this chapter. They are rather a sign that the market mechanism is working efficiently and are basically the result of [factor] price and cost changes and are transmitted via the market.[21] However, criticism has arisen based upon the idea proposed by Viner. Mishan (1972), for example, claimed that the distinction between technological and pecuniary externalities is superfluous and possibly confusing.[22] As described above, (negative) pecuniary externalities cover the case where the change in relative factor prices as output expands results in an increase in supply price. Mishan, however, showed that in the absence of any external effects, "a rising supply price is an implication of any interdependent economic model".[23]

Mishan (1972), therefore, argued that when looking at the problem from this perspective, there seems to be absolutely nothing special about rising supply curves whatsoever. Hence, no correction needs to be made in order to achieve an optimal equilibrium output, given the conditions of perfect competitions are fulfilled. Furthermore, he pointed out that the introduction of the term pecuniary diseconomies is ultimately nothing else than a verbal extravagance to explain supply curves that are in fact already explained by the described interdependent economic system in order to distinguish them from external diseconomies

---

[20] Viner (1931), p. 23. The proposed analysis has been made under the assumption of perfect competition and rational economic behaviour.

[21] See: Kapp (1969), p. 338.

[22] Mishan (1972), p. 6.

[23] Ibid. This holds, of course, only in the case where the familiar assumptions such as production functions homogenous of degree one, imperfectly elastic factor supply, and factor properties differing from one product to another are given.

proper. Likewise, he responded upon the approach regarding positive pecuniary externalities with the following argument:

> Moreover, the use of pecuniary external economy to refer to a reduction in the average cost of industry as it expands its purchases of materials or services from a falling cost industry B, will surely confuse most readers because this phenomena is neither more nor less than the original Marshallian conception of external economies that are internal to the competitive industry A, and attributable to economies of scale in the B industry.[24]

With a growing trend on environmental aspects as a field of interest, Meade (1952) suggested "to distinguish between certain types of external economies and diseconomies which are connected with marginal adjustments in purely competitive situations".[25] Generally, he defined as external economies and diseconomies those cases where the activities in one industry react upon the activities of production in another industry, "in some way other than through the possible effect upon the prices of the product or of the factors in that other industry".[26]

Starting from this definition, he proposed to divide external economies and diseconomies according to two characteristics: unpaid factors of production and the creation of atmosphere. For the type of unpaid factors, Meade's argument was that there might be a situation where no market for relevant input factors is established, and therefore no money paid for them. This is given although the mutual relationship between them is well known by the involved parties. Hence, inefficiencies may occur resulting from this situation.

---

[24] Mishan (1972), p. 6.
[25] Meade (1952), p. 54.
[26] Ibid, p. 56.

To demonstrate this argument he described the relationship between apple farmers and nearby beekeepers. According to well-established welfare economics, total individual welfare will be maximised if both apple and honey production are carried to the point where marginal social cost of each product is equal to its marginal social benefit. This is only guaranteed through the normal market mechanism and in the absence of externalities, where production functions are independent. In this case marginal social cost in terms of the value of resources foregone equal the marginal private cost of employing these resources. In Meade's example, however, this is not given, since the production function is not independent. Here, the input factor is the nectar from the apple blossoms for honey production. The apple production will be undertaken to the point where marginal private cost equal price (marginal social benefit). Any point beyond this optimal output reflects a point where the additional marginal cost will exceed the additional revenue generated, and is therefore not desirable. However, with a growing apple production, the production of honey may also be improved, since nectar from the apple blossoms is also an input factor for the honey production.

The increased availability of nectar represents a positive input factor for the honey production, and hence should be paid for by the honey producer.[27] However, the apple farmer cannot charge the beekeeper for the bees' 'food' and therefore he basically provides some of his factors (nectar) free of charge.[28] Therefore, the apple farmer is paid less than the value of his marginal social net product, whereas the beekeeper receives more than the value of his marginal social net product.[29]

---

[27] A positive market price for this input factor (nectar) would stimulate the apple production to the socially optimal amount.

[28] According to Meade, the reason for this is that the social accounting institutions have deficiencies or do not exist at all.

[29] This is an example for (positive) external economies, whereas if the bees would have some bad effects upon the apples, external diseconomies are given.

Consequently, less than the optimal amount of apples is produced and on the other side, a more than optimal quantity of honey is produced by the beekeeper.

According to Meade (1952), a socially optimal point could be achieved by subsidising the apple producer and taxing the beekeeper in order to ensure that marginal social costs and marginal social benefits in both industries are equal.[30] He also considered the reciprocal case where the apple production not only provides an input factor in form of nectar but also the keeping of the bees would lead to an increase in the productivity of the apple production due to additional pollination. Thus, ensuring that each factor is rewarded equal to its marginal social net product, a tax has to be levied on some factors, whereas others will be paid a subsidy.[31] The revenue of appropriate taxes just covers the appropriate subsidies, i.e. the budget remains balanced.[32]

Turning to the second category of the creation of a physical or social atmosphere affecting production, the main difference to an unpaid factor is, that although both affect the output of a certain industry, "the atmosphere is a fixed condition of production which remains unchanged for all producers in the industry in question without anyone else doing anything about it, however large or small – within limits - is the scale of operations of the industry."[33]

[30] However, looking at the society as a whole, there is no adding-up problem. This can be explained as follows; the combination of returns to scale across the industries is constant, although each industry taken individually produces under decreasing returns to scale. This means that a 10 per cent increase in all factors in both industries would cause a 10 per cent increase in the output of both products. However, this only holds if all factors (including the 'unpaid factors of production') are variable. See: Meade (1952), p. 60.

[31] This is for the cause where the 'unpaid factor' is of reciprocal nature.

[32] See Meade (1952), p. 198. However, this is not given in the case where the combined or social production function is non-convex. For further discussion on this issue, see: Papandreou (1994), p. 24.

[33] Ibid, p. 61. Meade assumed that rainforest is somehow not controllable by any industry.

More generally speaking, the activities of one group of producers do provide or create an atmosphere which is favourable (external economy) or unfavourable (external diseconomy) to the activities of another group of producers. The actual distinction is based on certain characteristics of production functions.

Meade describes this by using the example of the rainforest deforestation forced by the lumber industry, which again may have an impact on the regional rainfall. However, the affected farmers in the region can see this 'only' as a change in the atmosphere. Neither negotiations nor trading would solve the problem, because the affected party (farmer) does not notice the cause of the changes.[34] There are taxes (subsidies) required to discourage (promote) the creation of unfavourable (favourable) atmosphere. These are net subtractions from (net additions to) society's general fiscal burden.[35]

However, several points of critique have been raised about Meade's approach to analyse external effects by introducing a distinction between 'unpaid factors of production' and the 'creation of atmosphere' and hence a different remedy to 'internalise' these externalities.[36] Johnson (1973), for instance, claims that he could not find any theoretical or institutional reason why mutually beneficial contracts between bee-keepers and apple farmers could not be established a priori because of nonappropriability or difficulties of keeping tabs, as originally argued by Meade.[37] He stated that "a causal observation of the real world not only confirm [this point] but illuminates the workings of the market is alleviating many problems which Meade's example ignored".[38]

---

[34] See: Meade (1952), pp. 61-61.
[35] See: Ibid, p. 67.
[36] For a more detailed discussion on this see, for instance, Johnson (1973), Bator (1958), and Papandreou (1994).
[37] See: Johnson (1973), pp. 38-43.
[38] Ibid, p. 43.

Generally, Johnson (1973) argues that the flaws in Meade's analysis are threefold:[39]

- lack of exploration of the institutional causes of the marginal product (cost) divergence; he assumed what should have been a conclusion and hence did not produce a useful classification of externalities.
- the examples used are not only wrong but trivial.
- the conclusions drawn are based upon an asymmetrical methodology.

Cheung (1973) also conducted a field investigation on apple farmers and beekeepers in the US State of Washington. Using the results of this study, he could show that Meade's approach of 'unpaid factors' could basically be refuted because of the simple fact that some contractual arrangements between the apple farmers and beekeepers have been become a rather common routine.[40] Hence, he concluded his article:

> I have no grounds for criticizing Meade and other economists who follow the Pigovian tradition for their use of the bee example to illustrate a theoretical point: certainly, resource allocation would in general differ from what is observed if the factors were "unpaid". My main criticism, rather, concerns their approach to economic inquiry in failing to investigate the real-world situation and in arriving at policy implications out of sheer imaginations. As a result, their work contributes little to our understanding of the actual economic system.[41]

---

[39] Johnson (1973), pp. 47-52.
[40] See: Cheung (1973).
[41] Ibid, p. 33.

Meade also admitted that, first "in fact, of course, external economies or diseconomies may not fall into either of these precise divisions and may contain features of both of them".[42] Second, "[i]t is not claimed that this division of external; economies and diseconomies into unpaid factors and the creation of atmosphere is logically complete".[43] Also, it must be said, however, that the emphasis of Meade's study certainly was on the development of a mathematical analysis of the stated problem. Thus, the notational representation of externalities he proposed has become rather commonly accepted. According to Meade, external effects exist whenever production functions of the following form are given:

$$x_1 = F_1 (l_1, c_1, l_2, c_2, x_2) \tag{7}$$

$$x_2 = F_2 (l_2, c_2, c_1, l_2, x_1) \tag{8}$$

where:

x_i: output of industry i.

l_i: labour employed in industry i.

c_i: capital employed in industry i.

In other words, the output of industry 1 is determined by capital and labour used in both industries, as well as the output of industry 2, and vice versa.

Probably one of most influential contributions to the discussion on externalities was made by Coase with his famous article "The Problem of Social Costs."[44] However, it is rather interesting that Coase, in fact, made no explicit connection between what he was discussing in the paper and the concept of externality per

---

[42] Meade (1952), p. 67.
[43] Ibid.
[44] See: Coase (1960).

se. It might well be, as Papandreou (1994) assumes, that Coase's main intention was to discuss what would be an appropriate policy in the situation where "action of business firms have harmful effects on others"[45] rather than to find another definition for the notion of externality.[46] He questioned the analytical approach suggested by Pigou that it would be either desirable to make the generator of (negative) externalities liable for the damage or to place an output related tax equivalent to the marginal damage it would cause. Alternatively, the externality generating activity may be stopped.[47]

In contrary, Coase contended that "the suggested courses of action are inappropriate, in that they lead to results, which are not necessarily, or even desirable."[48] In other words, an external effect does not necessarily give rise to a misallocation of resources; given no barriers to trade between producer and consumer exist. Given that property rights are well-defined and enforceable, and transaction cost do not exist, there is the familiar market incentive for both concerned parties to negotiate a trade which is mutually beneficial between themselves. In doing so, the externality will be removed ('internalised'). Since the resulting equilibrium from such a trade will be maximised at the very point where marginal social benefits equal the marginal social costs, and there will be no misallocation of resources. Hence, Coase argued that the main reason for the divergence between social and private costs is the fact that the property rights for jointly used resources are not defined properly.

---

45 See: Coase (1960), p. 1.

46 However, in order to make this rather brief summary of his idea less complicated, the term externality will be used exactly for the case where private and social costs differ.

47 Coase also questioned the applicability of the examples used by Pigou to verify his approach.

48 Ibid, p. 2.

Generally speaking, in respect of possible policy implications in order to solve the problem, Coase distinguished between two different cases. First, the case where no transaction costs exists and second, the case where transaction costs are present. Regarding the first case, the intervention policy on the part of the respective government in order to restore a socially optimal equilibrium should not go beyond the establishment of property rights. As Coase formulated it:

> It is necessary to know whether the damaging business is liable or not for damage caused since without the establishment of this initial delimitation of rights there can be no market transactions to transfer and recombine them. But the ultimate result (which maximises the value of production) is independent of the legal position if the pricing system is assumed to work without cost.[49]

In other words, regardless of the allocation of such rights, i.e. which of the interacting parties is in fact receiving these rights, the resulting equilibrium will be a Pareto-efficient one.[50] This theoretical approach is now commonly known as the *Coase Theorem*.[51] Whereas in the case of costless market transactions, a rearrangement of legal rights through the market "would be made whenever this would lead to an increase in the value of production"[52], this does not hold once the costs of carrying out market transactions are taken into account. In this case, such a social arrangement should be carried out, at which the resulting social product is maximised.[53]

---

[49] Coase (1960), p. 8.

[50] According to Coase, a policy which would unilaterally tax (subsidise in the case of positive external effect as proposed by Pigou, the generator of the externality, would this move the economy away from the Pareto-optimal allocation and would therefore be the wrong measure to guarantee the desired outcome. This holds under the assumption that the demand pattern will not be influenced by changes in the distribution of wealth.

[51] Papandreou (1994) points out that there had been some question and controversy on this. He stated that the above mentioned version of the Coase Theorem is the commonly accepted version today, since it implies that "the correct proposition is that the allocation [of output] will be Pareto-optimal irrespective of legal rule, although this does not mean that the output will be necessarily be the same [under different legal rules]'. See: Papandreou (1994) p. 39.

[52] Coase (1960), p. 15.

[53] See, for instance, Turvey (1963) and Dahlman (1979).

In other words:

> [S]uch a rearrangement of rights will only be undertaken when the increase in the value of production consequent upon the rearrangement is greater than the costs, which would be involved in bringing it about.[54]

This means that the possibility could occur where no correction should be made at all if the costs of such an intervention outweigh the gain resulting from it. It is important to consider this when a decision has to be made about a certain tax-subsidy scheme. Hence, Coase's idea does not necessarily imply an unilateral tax or subsidy in order to modify the behaviour of the generator of an externality. Since externalities may be reciprocal, the elimination of a damaging activity via a tax, for example, could suddenly lead to a reduction of the social product, especially after taking the cost of market transaction into account. Johnson (1973) formulated it as follows:

> The major contributions of Coase were his proof that economic analysis should not concentrate upon the divergences between marginal private and marginal social products, his demonstration of the reciprocal nature of social cost and his argument that economist must examine the relative efficiency of alternative institutional arrangements.[55]

Almost unavoidable, the publication of his paper was followed by a large and ongoing debate. Coase's article has certainly been highly influential and pioneering for the development of the entire discussion on externalities and social costs. With his work, he focused attention on the fact that economic discussions have to be more cautious in analysing social costs and above all drawing conclusions for policy there from.

---

[54] Coase (1960), pp. 5-6.
[55] Johnson (1973), p. 36.

Furthermore, in order to evaluate the efficiency of an economic system, it is of great importance to take into account the costs of organising economic activities. However, there have been, in particular, discussions on how to interpret what Coase meant both with the statements and on the proposition he made. Vatn and Bromley (1997), for instance, claim:

> By focusing exclusively on negotiating agents - thus claiming that everything could become a market - Coase (1960) preserved an *apparent* consideration with standard economic analysis, but at the steep price of lost coherence. By ignoring transaction costs, the dropped-up post-Coasean externality model became incapable of generating any claim of Pareto-relevant externalities [. . .].[56]

This rather brief synopsis on the development of the notion of externality is certainly in no means complete and comprehensive and is intended only to show a limited section of the development of the notion over time. Generally, the discussion has not come to an end with the publication of Coase's article in 1960. However, the way the discussion has been led has certainly changed. As Papandreou (1994) describes it:

> Coase's article (1960) represents a kind of junction or turning point, reflecting a growing awareness of the importance of institutions in issues of resource allocation. More and more, externality was considered with that hazy borderline between the world of institutions and the beyond.[57]

Also, it became more and more conventional to discuss economic issues in specific contexts, and consequently the very vocabulary has been differentiated across certain spheres of enquiry in economics. Following Papandreou (1994), the further development of the notion of externality can be distinguished into the

---

[56] Vatn and Bromeley (1997), p. 137.

[57] Papandreou (1994), p. 44.

following three categories which, however, also have substantial overlapping features in common:

- Phenomenological approach: This category is especially related to the rising concern over environmental issues and what classes of events or phenomena are entailed.[58]
- General-equilibrium approach: The notion of externality in missing markets is a main issue in this category where externality is seen as a reason of failure, a subset of market failure.[59]
- Institutional approach: In this category the main focus is on the formation of institutions, i.e. rather than assuming that the institutional framework is given, it is seen as at least partly endogenous.[60]

However, since the aim of the study is not primarily to analyse and discuss the notion of externality with all its different facets and characteristics in great detail, no further statement will be made at this point concerning the notion.[61] Nevertheless, whenever it seems necessary and appropriate for the progress of this study, the relevant arguments and interpretations attributed to externality will be taken into consideration.

---

[58] See, for instance, Baumol and Oates (1990) and Meade (1973).

[59] See, for instance, Arrow (1970), Heller and Starret (1976) and Bator (1958).

[60] See, for instance, Demsetz (1967), Buchanan and Tullock (1962), Samuels (1972), Zerbe (1976) and Bromley (1989).

[61] For a more detailed and rather extensive discussion see, for instance, Papandreou (1994) and Cornes and Sandler (1996).

4.3 Formal Characterisation of External Economies & Diseconomies

As described in the previous section, it is evident that the concept of externalities and external effects has been discussed in the economic literature rather extensively. The interpretation and characterisation of external economies and diseconomies, however, usually vary depending mainly upon the range of effects to be involved, and as a result "ultimate definitions are a matter of taste and convenience."[62] Before analysing the theoretical aspects of externalities and giving a formal and graphical analysis of thc notion, various examples of definitions that occur in the economic literature shall be presented at this point.

Beginning with some textbook examples, the general outline of what is currently attributed with the term of externality will be introduced. Rosen (1995) offers the following definition:

> When the activity of one entity (a person or a firm) directly affects the welfare of another in a way that is not transmitted by market prices, that effect is called *externality* (because one entity directly affects the welfare of another entity that is "external" to it). Unlike effects that are transmitted through market prices, externalities adversely affect economic efficiency.[63]

Pindyick and Rubinfeld (1989) offer the following rather broad definition. In addition, they emphasise the distinction between the possible positive and negative character of externalities.

> Externalities can arise between producers, between consumers, or between consumers and producers. Externalities can be *negative* - when the action of one party imposes costs on another party - or *positive* - when the action of one party benefits another party.[64]

---

[62] Baumol and Oates (1990), p. 15.
[63] Rosen (1995), p. 91.
[64] Pindyck and Rubinfeld (1989), p. 617.

Hanley, Shogren and White (1997) give a similar definition, which establishes the fact that no exchange institution exists (i.e. market mechanism) where the person pays for the external benefits or pays a price for imposing the external costs.

> If the consumption or production activities of one individual or firm affect another person's utility or firm's production function so that the conditions of a Pareto optimal resource allocation are violated, an externality exists. Note that this external effect does not work through a market price, but rather through its impact on the production of utility or profit.[65]

Likewise, due to an incomplete set of markets, externalities do not act through market prices. The presented examples for definitions are comparable to most which have occurred elsewhere in the literature.[66] They commonly turn on partially or wholly unpriced welfare or production effects as well as the dependence of a production function or consumption function on the activities of a third party.[67] In a report by the Netherlands Research Foundation (ECN), points regarding the different characterisation of external effects raised above have been summarised. Accordingly, externalities exist if at least one of the following characteristics is presented:[68]

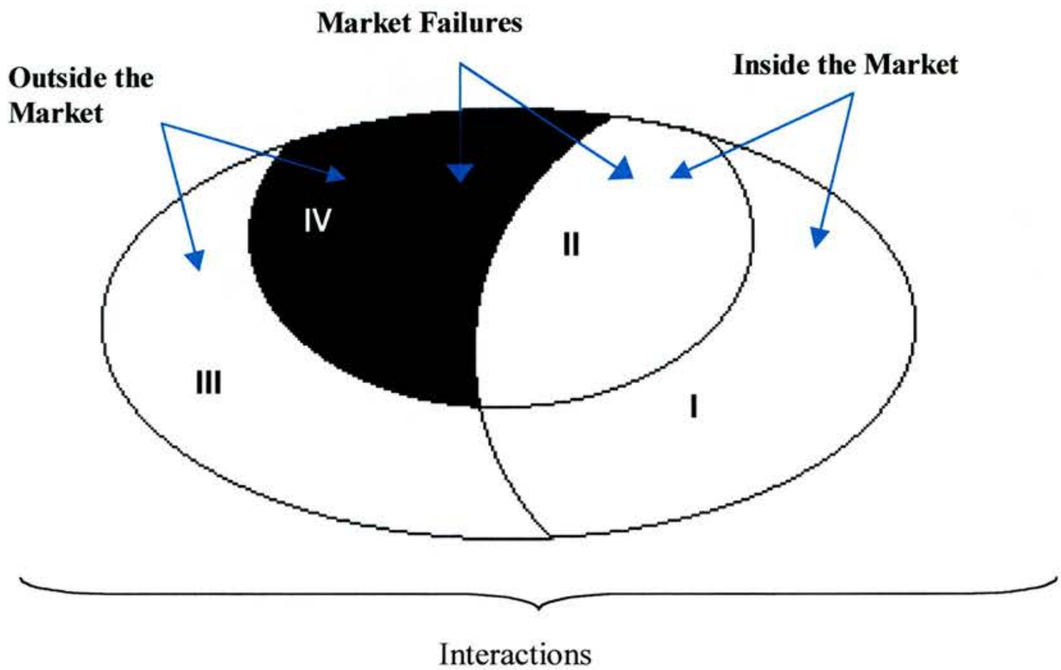
- interactions among individuals or groups.
- inefficiencies through market failures.
- non-existence of markets or property rights.

---

[65] Hanley, Shogren and White (1997), p. 29.
[66] See, for instance, Button (1994), p. 9 and European Commision (1995b), p. 4.
[67] Occasionally, a factor, which a firm is unable to control for, is also said to be an external factor of production. See for example.
[68] The following statements are mainly based on Kram et al. (1996), p. 19.

Using the following diagram, the different extensions of the notion of external effects can be illustrated.

Figure 22: Different Extensions of Externalities.



Source: Kram et al. (1996), p. 19.

In the above graph, four different areas can be defined which may be connected with the notion of externality, depending upon how wide or narrow the notion is extended. If external effects are seen as the outcome of any kind of interactions between different agents where one side actually has not been actively participating in the activity which caused the external effect, the broadest definition of the notion is given and includes all four areas (I, II, III, and IV). Area II and IV are those at which the notion of externality is narrowed down to the fact that inefficiencies in the economy are caused by external effects. These inefficiencies actually can arise inside the market (area II)[69] or outside the market

[69] Market failures inside the market occur, albeit property rights are well defined and institutions necessary for market trade are also established. This is given in the case of incomplete information, or when the respective agents do not behave in a sovereign way.

(area IV).[70] A further narrowing of the scope of externalities is achieved by concentrating only on interactions that are performed outside the market. This is represented by the areas III and IV, with area III representing the case where interactions are indeed performed outside the market. However, they do not result in inefficiencies to the economy.[71]

Finally, by excluding all these broader approaches to characterise the notion of externality, area IV of the above diagram reflects the most narrow definition, where only those interactions are considered that are performed outside the market mechanism and additionally result in market failure. To complete the definition for the notion of externality, which will finally serve as an underlying definition in this study, one further characteristic shall be discussed, and finally be taken into consideration. In the definitions given above, no specifications have been made whether the external effects are intentionally imposed. In this respect, Mishan (1971a) gives the following statement:

> What the notation alone does not succeed in conveying, however, is that the essential feature of the concept of an external effect is that the effects produced is not a deliberate creation but an *unintended* or *incidental* by-product of some otherwise legitimate activity.[72]

In other words, an external effect does not necessarily exist even when a firm's production function or a person's consumption function is influenced by another agent's activity.

---

[70] In the case of market failures outside the market, a market is not existent, i.e. institutional arrangements for performing trade are missing.

[71] Possible examples for this case (area III), are implicit contracts (because of missing institutions) or "multilateral interactions by distributing knowledge from research and development". See: Kram et al. (1996), p. 19. High transaction costs and the hope of multilateral benefits for all agents involved, prevent that even after property rights are defined (i.e. through patens or licenses), all outputs made by R & D are not transmitted through the market.

[72] Mishan (1971a), p. 2.

This is only given for the case where any form of criminal or altruistic behaviour is excluded. Using the following examples, Mishan (1969) describes this statement rather vividly.

> If I [Mishan] deliberately, and with malice aforethought, pour hydrochloric acid into the pure waters of a stream used by a whiskey distillery, or if I gradually poison my mother-in-law, I certainly affect the production function of the former and the consumption function of the latter. But neither activity accords with the popular notion of an external or spillover effect. Nor will the deliberate sabotage of a works by a gang of neo-Luddites do so, notwithstanding that their activities are outside the control of the firm.[73]

Hence, the final definition for the notion of externality, which will be applied in this study, is the one given by Verhoef (1994) which again is based on Mishan (1971a), and Baumol and Oates (1990).

> An external effect exists when an actor's (the receptor's) utility (or profit) function contains a real variable whose actual value depends on the behaviour of another actor (the supplier), who does not take these effects of his behaviour into account in his decision making process.[74]

It is also assumed that there are no compensation payments made, which, of course, would affect the utility function or the production function of the concerned parties.[75] Formally, this can be expressed as follows:

$$F^i = f^i\left(x_1^i, x_2^i, \dots x_m^i; x_n^j\right) \tag{9}$$

Proceeding from the above general functional relationship, one can now distinguish between the two cases of consumption and production externalities.

---

[73] Mishan (1969), p. 343.
[74] Verhoef (1994), p. 274.
[75] See: Baumol and Oates (1990), pp. 17-18.

a) Consumption externality:

where:

F^i : utility level of person i.

x^i_s: amounts of goods, $x_1, x_2, \ldots x_m$ utilised by person i.

x^j_n: amount of good x_n utilised by person j or produced by industry j (x^j_n could, of course, be $x^1_j, x^2_j, \ldots x^m_j$).

b) Production externality:

where:

F^i: output of firm or industry i.

x^i_s: amounts of its input.

x^j_n: amount of input or output of some other firm or industry.

Alternatively, F^1 can be seen as the total cost of all goods $x^i_1 \ldots x^i_m$, produced by firm 1. Here, the total cost not only depend on the amounts produced of these goods, but also on x_{jn}, the amount of good X_n produced by firm or industry j.

Generally, an externality is generated by entity j on entity i, if

$$\frac{\partial F^i}{\partial x^j_n} \neq 0 \tag{10}$$

with:

$\frac{\partial F^i}{\partial x^j_n} > 0$: positive external effect (external economy); and

$\frac{\partial F^i}{\partial x^j_n} < 0$: negative external effect (external diseconomy).

In the alternative case where F^i is regarded as the total cost, the signs are, naturally, reversed.

4.4 Categories of Externalities

4.4.1 Technological versus Pecuniary Externalities

As described above in some detail, Viner's classic article contributed enormously to clarifying issues relating to the whole discussion on returns to scale and externalities.[76] He showed that there exists a situation where no misallocation of resources is generated even though the underlying relationship seems to involve externalities. He therefore suggested that it is important to look at the nature of externalities more closely and introduced the distinction between technological and pecuniary externalities. Generally speaking, this means that externalities may consist either in changes of the technological coefficients of production (technological externalities) or in changes in the prices paid for the factors as a result of an increase in the amounts purchased.[77] Baumol and Oates (1990) described the situation as follows:

> There is a category of pseudo-externalities, the *pecuniary externalities*, in which one individual's activity level affects the financial circumstances of another, but which need not produce a misallocation of resources in a world of pure competition.[78]

and

> For they do not contribute any change in the real efficiency of the productive process viewed as a means to transform inputs into utility levels of the members of the economy.[79]

---

[76] See: Viner (1931).
[77] Ibid, p. 35.
[78] Baumol and Oates (1990), p. 29.
[79] Ibid, p. 30.

Externalities as they have been discussed and defined in the previous section represent what is known as technological externalities or externalities proper. As mentioned, in the presence of these externalities, there is a direct change of the allocation pattern, which is caused by influencing production technologies or consumer preference scales. Thus, it can be said that

> [T]echnological external economies [and diseconomies] are the only external economies [and diseconomies] that can arise because of direct interdependence among producers [and consumers] and within the framework of general equilibrium theory.[80]

In the case of pecuniary externalities, however, there is no misallocation of resources given, even though the underlying relationship might involve some sort of externality.[81] In contrast to technological externalities, in the presence of pecuniary externalities the interdependencies among producers is regulated through the market mechanism.[82] In other words, any effects on welfare are transmitted via prices since these effects are actually part of the normal functioning of a market and hence, they rather result from the change of financial circumstances of the agents involved. As Viner (1931) stated:

> Pecuniary diseconomies of this kind will always tend to result from expansion of output of an industry because of increased purchases of primary factors and materials which this entails to raise their unit prices.[83]

---

[80] Scitovsky (1954), p. 145.
[81] See: Viner (1931), p. 39.
[82] See: Scitovsky (1954), p. 146.
[83] Viner (1931), pp. 40-41.

In other words, pecuniary externalities can be seen as a result of price changes of input or output factors, which again result from changes in demand.[84]

> [C]omparing two otherwise identical states in which there is a technological externality in one, but not in the other . . . a given vector of real inputs allocated identically in both cases will not leave all members of the economy indifferent between the two states.[85]

However, it can be shown that regardless of whether an externality is of pecuniary or technological nature, there will be most likely changes in both prices and the values of the relevant variables. Thus, it cannot generally be said that the essential characteristic of pecuniary externalities, which distinguish them from technological externalities, is the fact that they affect only the values of monetary, rather than real, variables.[86] Rather, the distinctive difference is that changes are transmitted via the market. Hence, the result of such relations does not actually result in a shift of the production function or consumption function, but merely in a movement along these functions.

The introduction of a pecuniary externality, however, allows all members of the economy to stay at their original level of utility, if the following conditions are fulfilled.

- there is no change in the original usage of all input factors.
- in order to compensate for the income effects caused by the change of prices, an appropriate redistribution of income must be guaranteed.

---

[84] Correspondingly, pecuniary external economies result from a reduction in prices of services and materials. These changes arise because of an increased demand for these services and materials by the industry as a whole. See: Viner (1931), p. 39.

[85] Baumol and Oates (1990), p. 30.

[86] Baumol and Oates (1990) illustrate this using an example where a rise in prices for leather resulting from an increased demand for shoes is influencing the manufacturers of (leather) handbags. As a consequence of this change, the manufactures of (leather) handbags may have to modify their production process used so far. This could be done by the substitution of labour for leather through a more careful cutting of the raw material. However, there is absolutely no reason why the production of a handbag would now require a larger amount of leather, after the demand for shoes has increased. Certainly the welfare of the purchasers of handbags is affected. However, the resulting increase in price, is a sign of the redistribution of income from the purchasers or manufacturer of handbags to the suppliers of leather. See: Ibid, pp. 29-31.

Looking at it from another perspective, the introduction of pecuniary externalities will result in a movement along the production-possibility frontier rather than a shift of it. Hence, it will not produce any divergence between the social marginal rate of transformation and the private marginal rate of transformation at any point of the function. Likewise, there will be no divergence between any social marginal rate of substitution and private marginal rate of substitution, due to the fact that pecuniary externalities do not enter the utility function.[87]

The conclusion that can be drawn from the analysis of these two types of externalities (technological and pecuniary), mainly concerns the question whether the social optimum will be disturbed in the presence of such externalities and hence whether the existence of these externalities call for public intervention. As shown, the presence of pecuniary externalities does not call for any kind of public intervention. This is basically because pecuniary externalities are irrelevant for an optimal outcome of the market equilibrium of the competitive system, i.e. despite the presence of pecuniary externalities, an optimal resource allocation is guaranteed. They do not produce any misallocation of resources since they do not produce any divergence between private and social marginal rates of substitution or transformation.

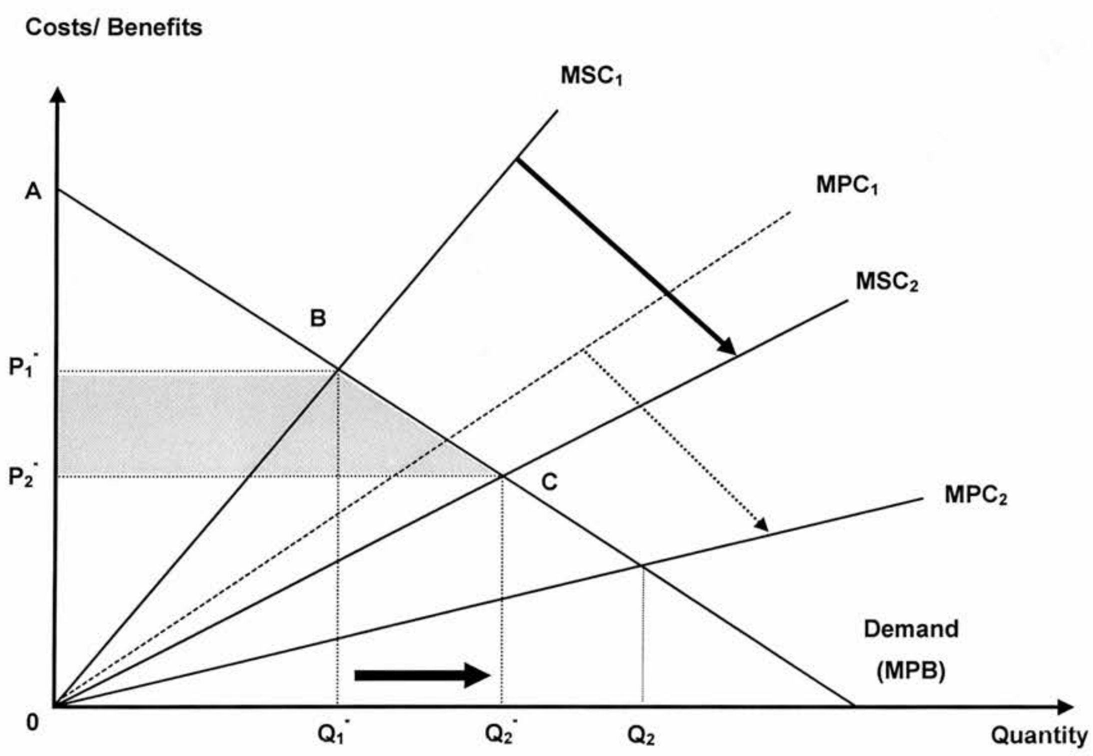
This is certainly different for the case of technological externalities. Therefore, this study will only focus on technological externalities, which disturb the optimal allocation of resources, and therefore may call for a public intervention in order to realise a (social) optimal equilibrium.

---

[87] See: Baumol and Oates (1990), p. 31.

The results of the more detailed discussion of technological and pecuniary externalities are rather important when the case of (potential) external benefits of road transport will be examined in the following chapter. The described theoretical aspects regarding pecuniary externalities can be illustrated using the following diagram.

Figure 23: Pecuniary Externalities.



Source: Verhoef (1994), p. 274.

Starting from the assumption of a given an activity, which is causing external costs, i.e. MSC_1 exceeds MPC_1, the original private cost curve (MPC_1) of the activity may shift downward as a result of, for example, technological change or lower prices for input factors. Hence, the marginal social cost curve will also shift downward leading to a move from Q_1* to Q_2*. Given ceteris paribus, i. e. unchanged external costs, the new socially optimal equilibrium is at point Q_2*, where MSC_2 and MPB intersect. This new equilibrium reflects increased overall social welfare.[88] The resulting lower market price p_2* with an increased output Q_2* will again lead to an increased consumer surplus, which is represented by the shaded area p_2*p_1*BC.

However, since this benefit is resulting from a movement along the marginal benefit curve (MPB) rather than a shift, it is not external but in fact pecuniary. So, the new socially optimal market equilibrium (move Q_1* to Q_2*) is achieved by the market mechanism solely and no public intervention is necessary, in terms of encouragement of stimulation. At this point, however, it has to be mentioned that pecuniary benefits do not compensate for the original external costs, hence, "social welfare maximization still requires a restriction from the new market outcome Q_2 to the new social optimum Q_2*.[89]

4.4.2 Relevant versus Irrelevant Externalities

For the analysis of the notion of externality and for the examination of possible internalisation policies, Buchanan and Stubblebine (1962) felt that since "rigorous definitions of the concept itself are not readily available in the literature", there is a need for a definition that is "more operational and useable".[90] To begin with, they introduced a definition of externality, which is one of the broadest definitions presented in economic literature and seems to be very similar to the one given by Meade (1953) described earlier in this section.[91]

More important for this study, however, is the fact that they subsequently give a comprehensive analysis of the notion of externalities in terms of relevance, i.e.

---

[88] The triangle area AC0 (Q_2*) is larger than the original triangle area AB0 (Q_1*).
[89] Verhoef (1994), p. 275.
[90] Buchanan and Stubblebine (1962), p. 371.
[91] See: Ibid, p. 372.

they believe that it is rather important to separate relevant from irrelevant external effects. They argue, "whether or not a relevant externality exists depends upon the extent to which the activity involving the externality is carried out by the person empowered to take action, to make decisions".[92]

Introducing the notion of potentially relevant externalities narrows their rather broad original definition. Externalities are defined as potentially relevant when the underlying activity causes external damages or benefits for one party and hence this very party has some desire to change the behaviour of the party whose action is causing the externality. This could be reached through several different measurements, such as compromise, persuasion, agreements, convention, collective action, etc.[93] If no such desire is given, the externality is defined as potentially irrelevant. In this case the affected party has no incentive to alter the behaviour of the party generating the external effect. In other words the affected party has achieved an equilibrium in which it is saturated, i.e. any other possible values of the external activity would not guarantee the same level of utility, and the party's utility is maximised.[94]

Finally, they subdivide the notion into Pareto relevant and Pareto irrelevant externalities. Generally, potentially relevant externalities require a desire to modify the behaviour of another party. However, this does not necessarily imply that there actually exists the required measures or abilities to implement this desire.

---

92 See: Buchanan and Stubblebine (1962), p. 373.
93 See: Ibid, pp. 373-374.
94 See: Ibid, p. 374.

Hence, according to the authors, a Pareto-relevant externality only exists:

> When the extent of the activity may be modified in such a way that the externality affected party, A, can be made better off without the acting party, B, being made worse off. That is to say, 'gains from trade' characterise the Pareto-relevant externality, trade that takes the form of some change in the activity of B as his part of the bargain.[95]

This means that the desire to modify another's behaviour provides, only then, a good rationale for actual modification, when the ensuring change can be done in such a way that the affected party will gain without the acting party being made worse off at the same time.[96] In the opposite case, however, where it is not possible to make the affected party better off without the other party loosing, the externality is Pareto-irrelevant. Hence, the activity level in question may be Pareto-optimal.[97] However, it is of great importance to bear in mind that in the latter case the externality still exists.

In other words, it can be said, "a position may be classified as Pareto-optimal or efficient despite the fact that, at the marginal, the activity of one individual externality affects the activity of another individual."[98] Hence, the policy implications, which may be drawn from this analysis, are rather significant. State intervention is only necessary in the case where Pareto-relevant externalities still remain, even though all possible negotiations have taken place.

---

[95] See: Buchanan and Stubblebine (1962), p. 374.
[96] This is similar to the extended approach of Pareto's criterion of welfare-improving actions. However, Buchanan and Stubblebine (1962) show that if the empowered party is in an utility-maximising equilibrium, a potentially relevant (marginal) externality is also Pareto-relevant. See: Ibid, p. 374.
[97] See: Ibid, pp. 375-376.
[98] See: Ibid, pp. 380-381.

Following Buchanan and Stubblebine:

> [T]he observation of external effects, taken close, cannot provide a basis for judgement concerning the desirability of some modifications in an existing state of affairs. there is no *prima facie* case for intervention in all cases where an externality is observed to exist.[99]

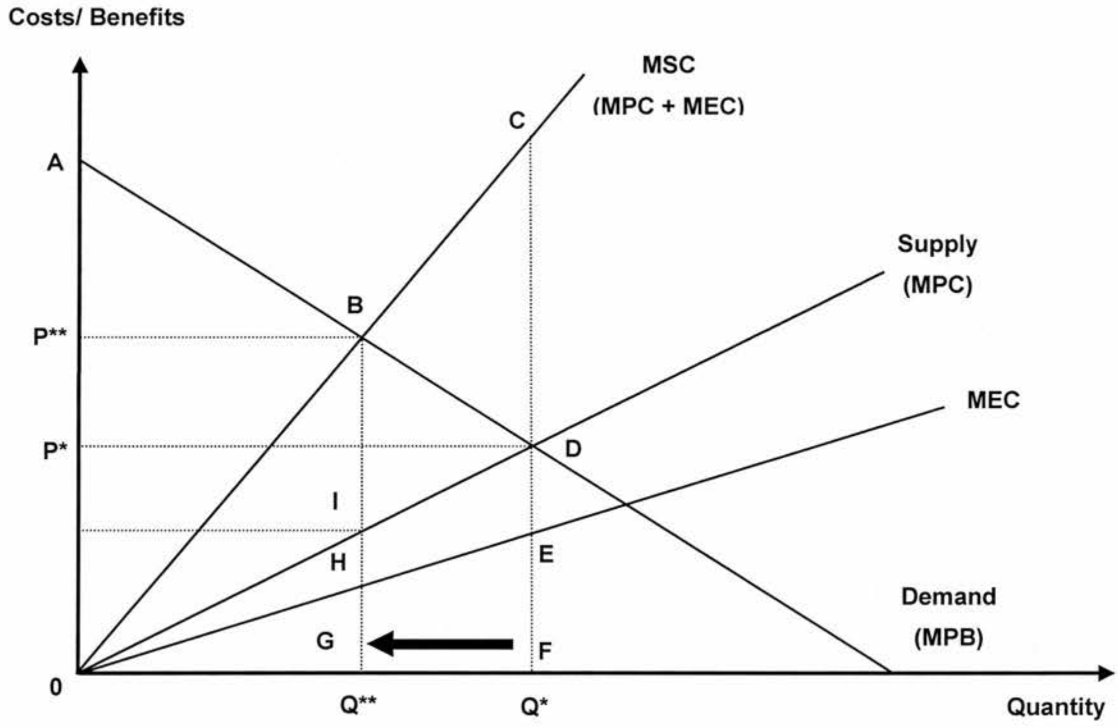
4.5 Graphical Analysis of Externalities

The described relations can also be illustrated graphically. Referring to the figure below, in the absence of any external effect, the market mechanism is working optimally. Adam Smith's 'invisible hand', the resulting market price p*, is securing the market equilibrium with an optimal output at Q*. At this very point, the marginal private benefit curve (MPB)[100] equals the marginal cost curve (MPC); demand equals supply. Hence, social welfare is maximised, which is represented by the triangle AB0 in the figure presented below. In the presence of externalities, however, the situation is different.

---

[99] Buchanan and Stubblebine (1962), p. 381.
[100] Benefit of each level of output.

Figure 24: Negative External Effects.



Source: Verhoef (1994), p. 274.

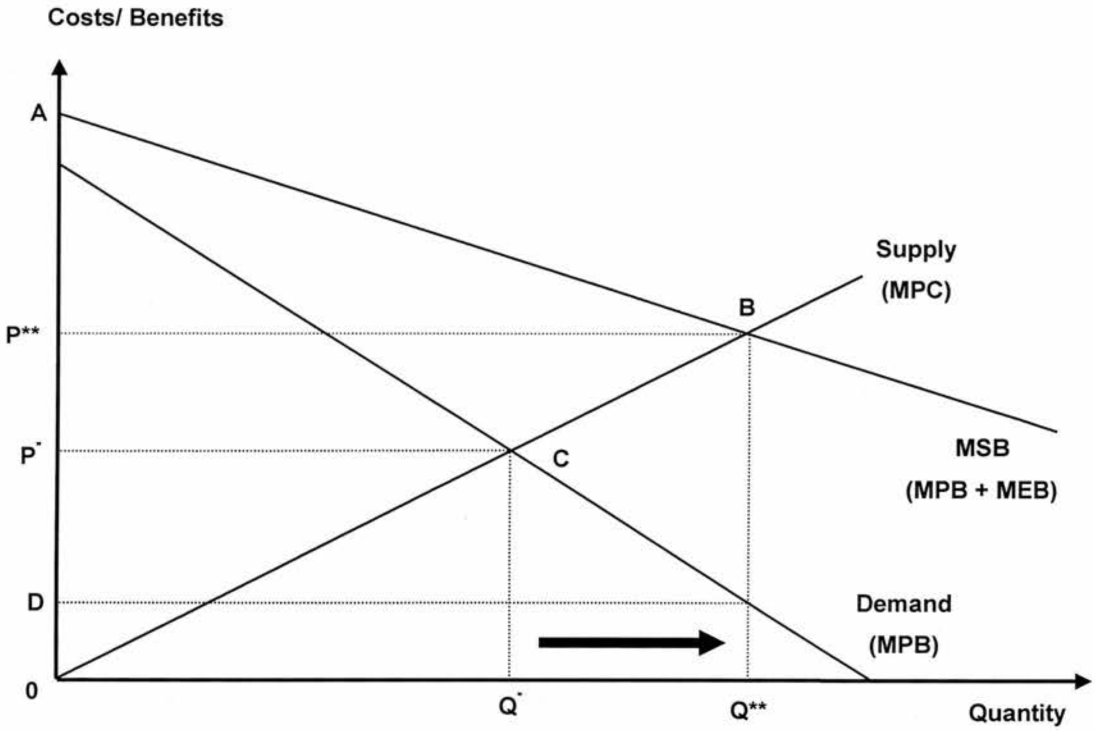
Now a situation emerges where, starting from the 'optimal' situation described above there is now a situation where marginal damage caused by one party is inflicted on another party at each level of output. The curve of these marginal external costs (MEC) is an upward sloping one, since it is assumed that the affected party is getting worse off at an increasing rate with any additional output. The marginal social cost curve (MSC) is formed by adding up the heights of both the MPC and MEC curve at each level of output. Hence, a wedge is driven between the marginal social costs and the marginal private costs.

As a result, the optimal output from a private and social point of view differs. Private welfare is maximised at the market outcome Q*, whereas from a social point of view production should only be carried to Q**, the very point where MPB and MSC intersect. Since the triangle area 0CD, representing the resulting level of externalities, is excessively large, efficiency could be enhanced if output is restricted to a level of Q**. In this case, there will be a net gain to society, which is equal to the difference between BCDI and BDI, which is in fact BCD, and hence the dead weight welfare loss BCD will be avoided.

Several implications can be drawn from this analysis. First, it seems obvious that the 'normal' private market mechanism is not generating the socially optimal market result in the presence of negative externalities; too much of the externality generating good is produced (Q*) relative to the socially desired and efficient output (Q**). Therefore, public intervention is required to restrict the activity to a socially optimal level Q**. Possible measurements could be either to restrict the output to a quantity Q**, or via the levy of a monetary measure such as a so-called Pigouvian tax on the externality generating activity. Second, however, it is also obvious that a restriction of this activity to a quantity of zero, e. g. a ban, is neither socially desirable. The trade off between the costs and potential benefits generally results in a positive level of output of the externality generating good.

The following figure illustrates the case where a positive externality (marginal external benefit) exists.

Figure 25: Positive External Effects.



Source: Verhoef (1994), p. 274.

Here, the market mechanism is leading to an equilibrium (MPC = MPB) at an output level Q*. This is the optimal outcome from a private point of view. However, since external marginal benefits are given, marginal social benefits (MSB) exceed marginal private benefits (MPB). While the market mechanism leads to an equilibrium at point Q*. From a social point of view this does not represent an optimal point. In order to maximise social welfare, the desired market equilibrium would be at point B, which leads to the transport volume (quantity) Q**. Again, since this equilibrium cannot be achieved by the 'normal' market mechanism via prices, a policy of government intervention is required. The underprovision of the activity may be corrected by Pigouvian subsidies of the size $\overline{D\,p^{**}}$ which would encourage the expanding of the production of the external benefit generating good (in this case for instance the number of road trips) to the optimal level Q**.

4.6 Summary

The aim of this chapter was to introduce and discuss the notion of externality. As seen from the short review of some of the main literature on externalities, the notion of externality has obviously experienced some change over the time. However, both the original, and further attempts to widen, to narrow or even to 'clarify' the notion of externalities have contributed to the fuzziness of the notion.

Papandreou (1994) concludes his rather comprehensive work "Externality and Institutions" with the following remark:

> [T]here cannot be a unique good characteristics of externality. Externality has come to denote many things, none of which separately, or in combination, seems to justify the appellation 'externality', and more important, none of which do full justice to the important issues underlying this notion.[101]

Thus, it seems to be reasonable and perfectly legitimate to discuss whether a certain concept in terms of externalities is the more appropriate one or not, rather than to consider whether one or another approach regarding externalities is the right one.

Following the general outline of the historical development of the notion of externality, a brief outline of the formal and graphical illustration of external effects was provided and discussed. Special emphasis was made on the characterisation of so-called pecuniary externalities, a phenomenon that in fact does not fall into the category of external effects. Second, an analysis of potential external benefits was given, which will be an important issue in the

---

[101] Papandreou (1994), p. 281.

following chapter when discussing the specific case of external effects related to the road transport sector.

5. Externalities and the Road Transport Sector

5.1 Introduction

As illustrated in the previous chapter, the phenomenon of externality has been discussed extensively in the economic literature. However, the exact characterisation and interpretation can vary depending mainly upon the range of effects to be included. The notion of externality is ambiguous, and therefore difficult to ultimately define. Rather, it appears to be the case that it is necessary to discuss whether the underlying concept for a specific analysis in respect to external effects is an appropriate one or not. Besides the obvious benefits of road transport regarding the development and the wealth of an economy as well as the satisfaction of people's need for mobility, concern over the negative effects of road transport has mounted over recent years.

Following the discussion in previous chapters, the road transport sector is one of the most significant sources of airborne pollution. The exposure to high pollution concentrations can lead to negative effects on human health, with premature death being the most severe one. These damages impose costs on society that are commonly known as external costs or externalities. The road user covers only a portion of the costs, whereas a significant amount is external in the sense that the producer (road-user) has no payment obligations for the cost of these adverse side effects.[1]

---

[1] See, for instance, Button (1993), Chapter 5.

Generally speaking, the polluter-pays-principle is not fulfilled. Since external costs are not specifically included in market prices, but rather imposed on society and the environment, there is a growing awareness in the necessity to comprise the described damages into the respective transport policy. Thus, it is important for any sort of transport scheme appraisal, cost-benefit analysis or internalisation strategies, such as levying a tax on the activity equal to the marginal external cost (Pigouvian tax), to quantify these costs, e.g. the costs of air pollution. Consequently, the exact amount of the external costs has to be known in order to implement any policy to internalise these external costs.

This chapter provides a detailed discussion of the external effects related to the transport sector. First, the notion of externality associated to the road transport sector is discussed and established. Based on this, the interactions between the transport sector and other sectors will be discussed, before identifying and analysing the transport-related externalities calling for public intervention. This discussion then leads to the issue of potential external benefits stemming from the transport sector, i.e. whether positive external effects indeed exist and may counter-balance the negative external effects. Before summing up the discussion results of this chapter, sources of external effects of road transport are examined.

5.2 Social versus External Effects of Transportation

For any analysis on the 'total' or 'full' costs of road transportation, it is essential to have a clear and unequivocal definition of the notion of cost as an underlying working base. However, it seems that confusion still exist about this. Consequently it is not surprising that a variety of different definitions have been developed in different studies on the 'total' cost of transportation to society.

There seems to be no consensus on either the notion, nor on the actual content-wise composition of the notion. In this study, the following broad definition of social costs will be used as a working basis.

> Social costs include all costs to society, direct or indirect, monetized or-kind, incurred by private individuals and firms or by collective entities up to and including the planet.[2]

To specify this broad definition, it is assumed that the total costs of road transport, which are a burden to society as a whole, is the sum of internal and external costs.[3] Using this definition, further subdivisions of motor vehicle transportation costs categories can be made.

Litman (1997) offers the following distribution of transportation costs where the emphasis mainly lies on how costs affect private or public decisions. He differentiates between internal (user) and external, variable and fixed, and market and non-market costs, the latter being underlined in the following table.

---

[2] Lee (1997), p. 113.

[3] See, for instance, Kågeson (1994), p. 74 and Mauch and Rothengatter (1995), p. 5. Sometimes 'internal' and 'private' costs are used exchangeable. However, this is not perfectly correct when analysing the transport sector. Here, 'private' costs are merely a component (though the main one) of 'internal' costs as it will be described later in this section.

Table 38: General Classification of Transport Externalities.

	variable	fixed
internal (user)	• fuel[†] • short term parking • vehicle maintenance (part) • user time & stress[‡] • user accident risk	• vehicle purchase • vehicle registration • insurance payments • long-term parking facilities • vehicle maintenance (part)
external	• road maintenance • traffic law enforcement • insurance disbursements • congestion delays • environmental impacts • uncompensated accident risk	• road construction • 'free' or subsidised parking • traffic planning • street lightening • land use impacts • social inequity •

Source: Litman (1997), pp. 1-5.

Note: (†) = Components not underlined represent market costs.
(‡) = Underlined components represent non-market costs.

Whereas vehicle owners tend to make their decision whether to use their vehicle based upon perceived internal short-term variable costs, public agencies base their decisions primarily on perceived costs to their institutions, i.e. market costs. Litman (1997) also points out that on the other side, decisions on transport planning and investment are mainly based on short- and medium term direct market costs and hence important costs tend to be ignored in their decision making process. However, analysis and decisions regarding transport planning and investments require that all occurring costs be taken into consideration, i. e. it is also important to include long-term, non-market and indirect costs.

Delucchi (1997) has presented a comprehensive classification of the full costs of motor vehicle transportation, which is reproduced in the following table.

Table 39: Classification of External Effects Stemming from Road Transport.

Personal non-monetary costs of using motor vehicles (unpriced)	**Explicitly priced private sector motor vehicles goods and services, net of producer surplus and taxes and fee (monetary costs)**	**"Bundled" private-sector goods (implicitly priced)**
• Uncompensated personal (non-work) travel time, excluding travel delay imposed by others • Accidental pain and suffering and death inflicted on oneself • Personal time spent working on motor vehicles and garages, refuelling motor vehicles, and buying and disposing of motor vehicles and parts • Noise inflicted on oneself • Air pollution inflicted on oneself	**Usually included in GNP-type accounts:** • Annualised cost of the entire car and truck fleet, excluding taxes • Cost of transactions on used cars • Fuel, lube oil, except costs due to travel delay • Maintenance, repair, washing, renting, storage, and towing; excluding external repair costs, costs attributable to travel delay • Parts, tires, tubes and accessories • Automobile insurance: administrative and management costs and profit • Accident costs paid for by automobile insurance of responsible party: lost productivity, medical and legal services, victim restitution • Parking away from residence, excluding parking tax **Usually not included in GNP-type account:** • Compensated (work) time of business, government, and commercial travellers, excluding travel delay imposed by others • Overhead expenses of business, commercial, and government fleets • Accident costs paid for by responsible party, but not by automobile insurance: lost productivity, medical services, legal services, damage to non-vehicular property, victim restitution • Vehicle inspection by private garages • Legal services, security devices due to motor-vehicle related crime.	• Annualised cost of non-residential off-street parking included in the price of goods and services or offered as an employee benefit • Annualised cost of home garages and other residential parking included in the price of housing (including interest on home loans) • Annualised cost of roads provided or paid for by the private sector and recovered in the price of structures, goods, or services

continued:

Government services charged partly to motor vehicle users (monetary costs)	Monetary externalities (unpriced)	Non-monetary externalities (unpriced)
• Annualised cost of public highways and highway maintenance (including on street parking) • Annualised cost of municipal off-street parking • Highway patrol • Environmental regulation, protection, and clean up, including landfills and sewage treatment plants • Energy and technology R & D	• Cost of travel delay imposed by others, including accident delay: extra fuel, oil, maintenance and compensated (work) travel time • Probabilistic loss of GNP due to sudden changes in the price of oil • Accident costs not paid for by responsible party: lost productivity, medical, legal, property damage (including cost paid by government) • Price effect of using petroleum fuels for motor vehicles: increased payments to foreign countries for oil used in other sectors (not an external cost globally) • Losses from robbery in parking lots or theft from vehicles (net of dollar gain to criminals)	• Air pollution (including toxics) inflicted on others: effects on human health, crops, materials, and visibility • Accidents: pain and suffering, and death not paid by the responsible party: fear of accidents • Extra uncompensated (non-work) travel time due to travel delay imposed by others, including accidents delay • Global warming due to fuel-cycle emissions of greenhouse gases (count US damages only) • Noise inflicted on others • Water pollution: health and environmental effects of leaking storage & waste sites, oil spills, road runoff • Pain, suffering, inconvenience, and other non-monetary costs due to motor-vehicle related crimes • Land-use damage: habitat and species loss due to highway and MV infrastructure • The socially divisive effect of roads as physical barriers in communities • Vibration damages • Aesthetics of highways, vehicles, and service establishments

Source: Delucchi (1997), pp. 28-32 & pp. 41-49.

Kram et al. (1996) and Quinet (1997) have suggested another slightly different categorisation, where the degree of differentiation of the 'full' costs of transportation is certainly not as detailed as the one previously described by Delucchie (1997).[4] However, even though it is not as extensive the different components of total costs of transportation can clearly be identified. It becomes particularly clear that private costs are not identical to internal costs, since this

[4] See: Kram et al. (1996), pp. 21-22 and Quinet (1997), pp. 77-78.

only holds true for the case where infrastructure costs are totally and correctly covered by fares and toll. This also applies to the construction, maintenance, and operation of infrastructure. Their categorisations are shown in the table below.[5]

[5] See also: Komanoff (1994), Gastaldi et al. (1996) and Forman and Alexander (1998).

Table 40: Categories of Transport Externalities.

<table>
<tr><td rowspan="7">Full Costs

of

Transportation</td><td>External Costs of User</td><td colspan="2">Congestion</td></tr>
<tr><td rowspan="4">External Costs of Society

(inclusive future generations)</td><td colspan="2">Accidents</td></tr>
<tr><td colspan="2">Use of Space</td></tr>
<tr><td>Ecology Costs

(Costs of depletion of non-renewable resources)</td><td>Effects on Fauna & Flora
Energy
Air, Water, Land Pollution
Noise
Landscape Effects
Vibration</td></tr>
<tr><td colspan="2" rowspan="2">Infrastructure Costs and other Material Resource Costs</td></tr>
<tr><td rowspan="2">Internal Costs</td></tr>
<tr><td>Private Costs

(commercial & individual costs of operation, personnel, fines & insurance)</td><td>Fuel
Maintenance
Repairs
Insurance & Tax
Vehicle-Amortisation</td></tr>
</table>

Source: Kram et al. (1996) and Quinet (1997), pp. 69-113.

As shown above, the total costs of transportation can be classified in varies ways. Depending upon the aim of the analysis, some of the categories could, for example, be merged or split leading to a different classification. It is even possible for an entirely different general principle of organising to be imagined. Since the purpose of this study is primarily to give an evaluation of the external health costs stemming from motor vehicle use, the classification by Kram et al. (1996) and Quinet (1997) as discussed last, seems to be the most appropriate one for this purpose.

5.3 Interaction Levels between the Transport Sector and other Sectors

Bonnafous (1994) proposed an approach to differentiate between different levels of interactions, which are focused upon the classification of different 'spheres'.[6] He correctly claims that in the case where externalities have to be taken into consideration, appropriate definitions are necessary as a suitable working basis in order to solve the question of how to measure them. Obviously, a variety of either broader or sometimes rather narrow definitions have occurred in economic literature in general and in particular with respect to the transport sector. Bonnafous claims that

> [i]t is quite clear that the diversity of external effects calls for a new type of vocabulary - or at any rate, a more precise one - and that the concepts of social costs, external cost, or even externalities can be confusing, if they are not given more rigorous definition.[7]

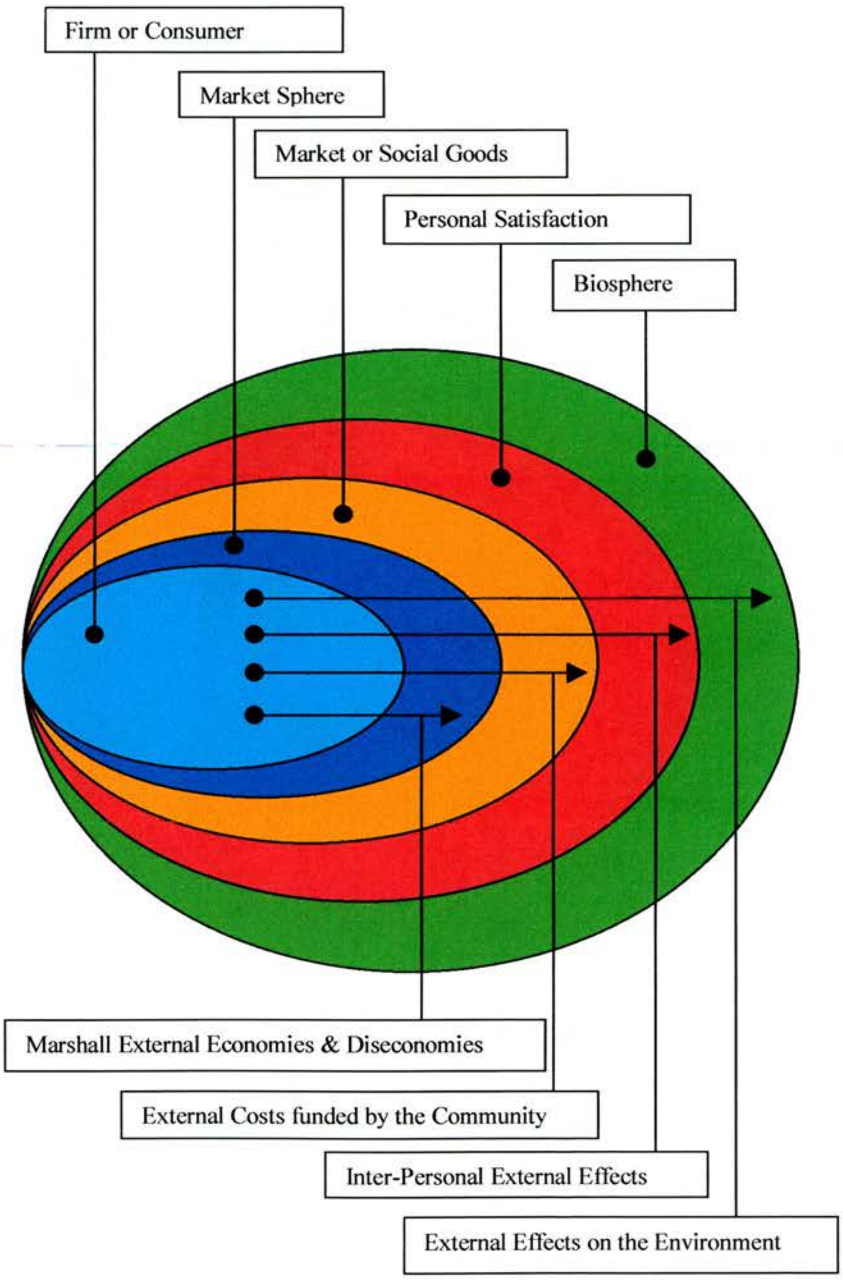
Moreover, he argues that when defining the concept of external effects, it is necessary "to specify 'external relations to what?'"[8]

---

[6] See: Bonnafous (1994), pp. 181-186.
[7] Ibid, p. 183.
[8] Ibid, p. 185.

Hence, a typology of such effects is offered which sets out the different categories of external effects with reference to the corresponding 'spheres' that are used for determining the externalities. This typology is illustrated by the following figure.

Figure 26: Typology of External Effects.



Source: Bonnafous (1994) p. 184.

Starting with the sphere that represents any agent, e.g., consumer, user or firm, it is assumed that this agent is generating certain effects which do not affect his or her personal budget. In other words external effects are generated which again may differ depending on the respective sphere in which they occur.

- ***Marshall external economies and diseconomies***

  These effects given from the sphere of the firm or consumer towards the market sphere are external in the sense of Marshall's concept of external economies and diseconomies as described in more detail in the previous chapter. Concerning the transport sector, the example where an additional commercial lorry enters an already congested road will contribute to further slow traffic down. As a result the production costs of other hauliers will increase since more time has to be spent on the road. However, this phenomenon reflects what is called pecuniary external effect and is not operated externally, i.e. outside the market.

- ***External costs funded by the community***

  These external costs arise by going from the sphere of the firm or consumer to the sphere which includes market goods and services, as well as social goods and services. An example would be the expenses which have to be made on maintaining the road network damaged by wear and tear from road traffic.

- ***Inter-personal external effects***

  Here, the personal satisfaction factor is introduced. The respective external effects may comprise factors such as the uneasiness caused by traffic noise or the lack of safety as well as the time wasted by individuals due to congestion.

- ***External effects on the environment***

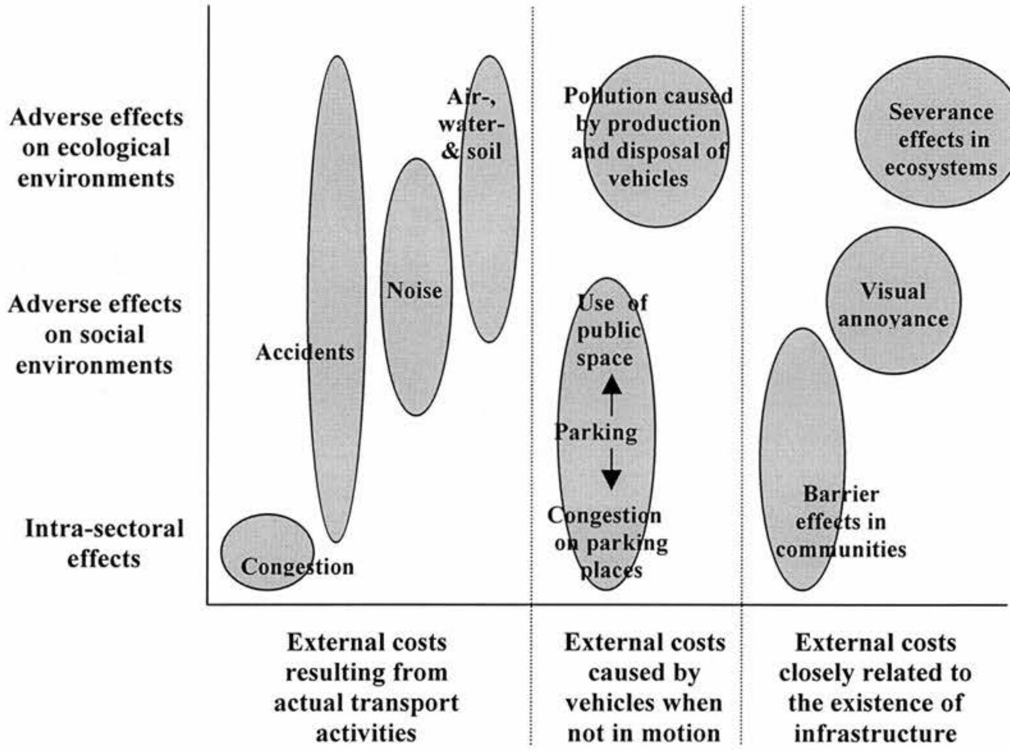
  This category encompasses the spheres of the firm or consumer that includes in the broadest sense the biosphere and in a narrow sense the quality of the environment. The main example for the transport sector is certainly the emission caused by motor vehicle use and the subsequent adverse effects.

Furthermore, by moving from the first to the fourth category, "the effects are less and less closely bound up with 'pecuniary' counterparts".[9] Hence, since the main focus of this study is on the valuation of external health costs caused by motor vehicles pollution, the focus will be on the fourth category. Bonnafous (1994) argues that "the problem of external costs calls for a means by which they can be internalised - in the sense that their production implies pecuniary counterparts - and which integrates them directly in the market sphere."[10] Verhoef (1996) offers the following typology of external costs regarding road transport.

---

[9] Bonnafous (1994), p. 188.
[10] Ibid, p. 186.

Figure 27: Typology of Road Transport Externalities.



Source: Verhoef (1996), p. 203.

Verhoef (1996) offers a classification of external costs of road transport along two different dimensions. The first dimension, which represents the vertical axis, describes the possible levels of interactions. By moving along this axis, the distinction starts with the narrowest level of intra-sectoral effects where road users impose external costs on each other, and tends toward to the fairly broad category of inter-sectoral or environmental externalities. The category of adverse effects on social environments represents the situation where road users impose externalities on people outside the population of car users.

On the horizontal axis, which represents the second dimension, Verhoef (1996) separates external costs resulting from the actual activities in the transport sector from those which are caused by motor vehicles when they are not in motion and finally from those effects arise in such areas which are closely related to the existence of the transport infrastructure.[11] The shaded areas represent the first-order incidences of the externalities.[12] The scope of effectiveness clearly varies between the mentioned externalities. The scale of the external costs may be limited to one type along the vertical axis, such as congestion, which causes exclusively intra-sectoral effects.

Other externalities, however, may be extended to more than one type, such as air, water, and soil pollution, where both the ecological environments as well as the social environments are adversely affected. In the case of accidents all three described categories are to some extent affected.[13]

The described typology as introduced by Verhoef (1996) is of great relevance for the analysis of externalities stemming from road transport, especially when it is important to separate the externalities related to the actual transport activities and those externalities arising from the mere presence of the transport infrastructure. A similar approach was proposed by Kram et al. (1996), Mauch and Rothengatter (1995) and Rothengatter (1993). Following this work, the different agents affected by transport activities outside the market can be subdivided into three different regimes or sections. These sections are:

- Sector of private production and consumption.
- Sector of public production and consumption.
- Sector of non-renewable resources, i.e. the environment and non-producible human capital.

---

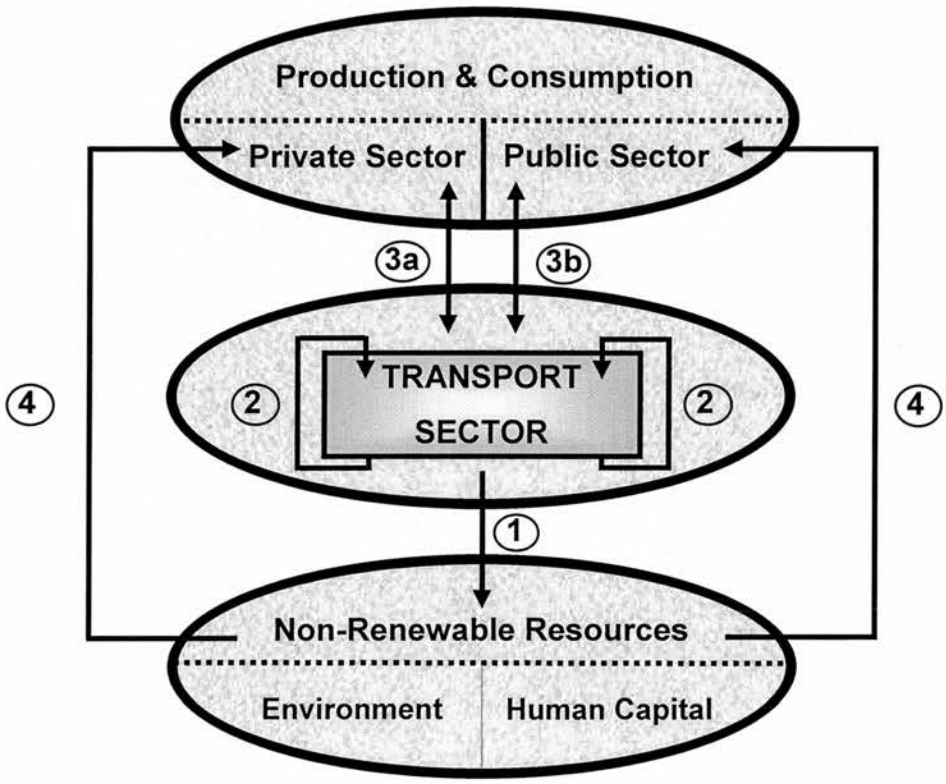
[11] These external costs, however, are normally not solely attributable.

[12] Second-order incidences of the externalities may occur on different scale level. Whereas the first order incidence of congestion are 'only' intra-sectoral effects, this is different for the second-order one. Here, it is obvious that congestion also increases the emissions per vehicle kilometre driven and hence is also an environmental externality in the ecological sense.

[13] See: Verhoef (1996), pp. 203-204.

The following figure illustrates these different relations.

Figure 28: Transport Sector and Regimes of Interactions.



Source: Mauch and Rothengatter (1995), p.14.

Between the transport sector and these regimes, certain interactions may occur which again can be subdivided. By doing so, four levels of potential externalities in which the transport sector is involved can be identified. These levels are characterised by the different numbers in the previous figure.

- **Externalities on level 1:**

  These externalities occur in the presence of an interaction between the transport sector and the stock of non-renewable resources, such as the environment or non-producible capital. The production of services in the transport sector can basically be done without special regards for the use of these resources, i.e. no compensation is made for using these resources. Therefore, in the case of the environment, a conflict emerges between those who use it to live and use it to increase their quality of live, and the group of agents who use the environment to dispose waste generated by either their production or consumption.

- **Externalities on level 2:**

  These are externalities arising because of interactions within the transport sector. Since different agents use the transportation network at the same time, and hence may interact, a conflict may develop between them. A rather concise, and probably most often cited example, is certainly the case of traffic congestion. First mentioned by Pigou, the example is used to explain how a sup-optimal user pattern of the transport network occurs when the user of this network does not take into account the potential (negative) effects of their transport activities on other road users. In other words, the decentralised decision making process leads to a total sum of user costs, which exceed by far the aggregate cost the users would face in the case of a central optimal control.

- **Externalities on level 3:**

  These externalities can be divided further into two subgroups.

  3a) The externalities in this category caused from the interaction between the transport sector and either private production or consumption. Basically, one can define three different sources of externalities resulting from the interaction between the transport and the private sector.

- Damages to material, such as the soiling of buildings due to air pollution. Transportation may also be responsible for production losses in the form of a loss of reproducible resources.

- Since the public sector is usually the provider of the transportation infrastructure, commercial users receive extra benefits from the fact that the total costs stemming from the provision and operation of the transport network are not charged fully by the public.[14]

- Several interactions between activities in the transport sector and economic activities interactions may induce new forms of operations and distributions logistics and contribute to technical progress.[15] Also, transport may have significant effects on economic growth insofar as an improved and extended transport infrastructure may also improve accessibility to more remote regions which do not lie at a main arterial route. As a result inter regional exchange activities are being improved.[16]

However, as stated in Mauch and Rothengatter (1995), "it is important to notice that it is the *supply* of [transport] infrastructure that is generating theses externalities, not the *use*"[17]. This crucial aspect will be discussed in more detail in a later section.

---

[14] Correspondingly, there are losses for these users in the case where the public charges more than the full costs of providing and operating the transport infrastructure. This would be done by either particular taxes or special user charges in the commercial sector.

[15] See: Rothengatter (1994), p. 324.

[16] This includes the improved link of regions to the non-produced natural transport infrastructure, such as waterways. Mauch and Rothengatter (1995) also mention the charge in land use brought about by the supply of transport infrastructure as a (positive) externality. See: Mauch and Rothengatter (1995), p. 15.

[17] See: Ibid.

3b) Externalities derived from the interaction between the transport sector and the regime of public production or consumption. This category of externalities can be explained by assuming that the transport infrastructure can be characterised as a public good. The regime of any kind of public power as well as the general supply of certain public services, such as national defence, emergency services, and a minimum level of communication, is based on the existence of a working transportation network.

- **Externalities on level 4:**

The externalities in this category are a result of interactions between the stock of non-renewable resources, such as environmental and non-producible resources, and the public and private regimes. The subsidisation of transport activities could also stimulate activities in other economic sectors, which again could deteriorate environmental conditions. For example, the effect of making the mountain regions more accessible due to improved road networks could have devastating effects on the regional and local ecology. As a result of improved access, there might be an increase in leisure activities such as skiing or hiking, which again could have detrimental effects on natural resources, i.e. the erosion of the respective mountain region. No financial compensation is made for these commercial or public activities.[18]

---

[18] See, for instance, Maibach et al. (1996).

5.4 Externalities in the Transport Sector Relevant for Public Intervention

After investigating the nature and implications of the proposed levels of externalities, it is possible to assume that only the previously introduced categories of externalities related to the transport sector on **level 1** are actually calling for public intervention in order to reduce their impact. Rothengatter (1994) points out the following reasons for this statement.[19]

- Because there is no existing market, the use of non-renewable resources is not co-ordinated through a price mechanism. Additionally, since no property rights for these resources are defined, they are free of charge. As a result, an overexploitation by underpricing is given. In other words, the private agents extract more than the socially optimal level of these inputs for production or consumption.

- One of the assumptions made in the Coase theorem is that the transaction costs of contracting are so small that they are neglectible. This may be given in the case where just two parties are involved. However, in the case of interactions between the transport sector and non-renewable resources, the number of parties involved is usually rather large. Hence, the transaction costs of contracting on these resources are not neglectable. Given these costs of transaction, it is actually rather unlikely to achieve an arrangement between the parties for settling specific property rights.

- The economic power of the agents on both sides involved in the interactions is unbalanced. The group of agents on the side of private consumption and production is usually in a position of large economic power. They form some

---

[19] Rothengatter (1994), pp. 324-325.

sort syndicate which may not only consists of common users of the transport network, but also might enjoy the begging of powerful organisations such as the automotive industry and automotive clubs (e.g. RAC, AA). The protectors of the environmental resources on the other side suffer from lack of such economic power. Hence, there are very unequal presuppositions in any kind of negotiation between the two groups. The economic syndicates of the users of the environmental resources are in such a dominant position that it is almost impossible for the protectors of the environment to achieve any contractual arrangements, which can be considered as fair in the end.

- It is doubtful that any kind of bargaining scheme would lead to a stage where a fair as well as efficient solution between the users and protectors of the environment is generated. The same is true for a spot market mechanism. Above all, this is due to the fact that the activities in the transport sector contain some risk to health and/ or life. Examples are accidents or the health impacts of exhaust emissions.

Based on these arguments, it seems plausible to claim that only those externalities on **level 1**, the interactions between the transport sector and the non-renewable resources such as the environment and human capital, are relevant for public intervention. Also, it seems obvious that these externalities cannot be compensated by the externalities (external benefits) arising between the transport sector and the private sector (level 3 externalities). The signals from such compensation would be completely misleading in the sense that 'suddenly' the 'wrong' activities would be supported. Hence, the resulting incentive scheme would support the economic system to shift into a distribution that is certainly not intended from a social point of view. The most significant example of externalities on level 2 is the problem of congestion.

Theoretically, however, compensation payments in order to 'solve' or 'internalise' these externalities are unnecessary. Rothengatter (1994) argues that there is no need for public intervention to control for these congestion externalities in the case where private management is involved, since then sufficient incentives to reduce them are given. In other words, the respective private road network operation "would differentiate user charges according to the congestion level inverse demand elasticies and therefore tend to reduce congestion externalities. Consequently, the institutional solution would be the natural way to cope with congestion"[20].

5.5 External Benefits and the Road Transport Sector

So far the main thrust of the discussion on externalities has been on external costs. However, it is certainly legitimate to raise the question whether there are actually any external benefits resulting from road transportation. Furthermore, if such external benefits in fact do exist, the question of deducting them from the amount of external costs of transport also has to be discussed.

Particularly, representatives and lobbyists of the automobile industry and its related sectors have brought this issue forward. Additionally, a number of studies have come to the conclusion that there are a number of external benefits produced by road transport.

Based on this, it has been claimed that the external benefits which result from transport, by far exceed its external costs.[21] In such a case, its implication would

---

[20] Rothengatter (1994), pp. 324-325.

[21] The Allgemeine Deutscher Automobil Club, the German club of automobilits for example claims, that the benefits from road transport exceed its external costs by a factor of 5 to 10. See: ADAC (1992), p. 39.

be that from the perspective of Pareto efficiency the use of cars or any other motor vehicles should indeed be promoted. Road user should be encouraged to increase their mobility, i.e. to make more use of their motor vehicles. The creation of Coasian markets or Pigouvian subsidies, for example, would enhance such a policy. Subsequently, the non-paying profiteers would compensate the road user.

Based on Rothengatter (1994), the following list of potential external benefits of road transport has occurred in various studies. Examples considered as external benefits from road transport include:

- Extension of consumption patterns and improvements in living standards.
- Development of new structures or spatial patterns.
- Decentralisation of production locations.
- Separation of the locations of housing and employment.
- Specialisation of land use.
- Extension of labour markets.
- Induction of growth and structural effects, individualisation and flexibility of freight transport logistics to create new paths for industrial labour division and interaction, setting new quality standards such as 'just-in time' transport.
- Innovations generated by using road vehicles, synergetic processes between industry, spatial economy and transport.
- Remarkable increases of flexibility and innovations, which creates a new quality of service in transport and strengthens the economy for international competition.
- Cost reductions for packing, processing and logistics.
- High quality of regional distribution of consumption goods.
- Improvements of location quality, which seems to be extremely important for a country with high production quality and costs.
- Positive employment effects in remote regions, which have no access to rail.

The list mainly consists of those points raised in the studies conducted by Willeke (1991, 1992, 1996), Aberle and Engel (1992) and the Deutsche Strassenliga (1992). Based on the estimates made in these studies, the respective authors suggest that the external costs from road transport are at least equalled, if not exceeded by the potential external benefits resulting from it. Hence, it is claimed that there is no justification for any public intervention, such as additional charges, in order to internalise the external costs.

In order to clear up some of the confusion connected with this issue, it is important to analyse in some detail which of the effects mentioned above are in fact external benefits, in the sense that they are exchanged outside the market and enhance positive effects from the road transport. Concerning these effects, Rothengatter (1994) makes the following rather critical remark:

> The lists of [alleged] external benefits of the road lobby consists of consumer/producer surpluses, direct and indirect cost savings, multiplier effects, input-output effects, innovation/ technological/ structural effects, and direct and indirect involvement of transport activities in production and trade of goods and services.[22]

Thus, it is claimed that a majority of the effects mentioned are actually not external at all. The alleged external benefits such as lower consumer prices and production costs, as well as a greater variability of choice for consumer and faster good delivery, achieved, for instance, through just-in-time transportation, can in fact be described as pecuniary benefits. Having discussed the phenomena of pecuniary benefits previously, it can be shown that these effects are essentially due to lower transportation costs of road transport compared to other transport modes such as railways.[23]

---

[22] Rothengatter (1994), p. 326.

[23] Alternatively, these benefits could also result from a greater efficiency of road transport in comparison to alternative transport modes.

Assuming that there will be no market distortions in other parts of the economy, these benefits will in due course accumulate to the final consumers of the transported good. This consumer surplus is in fact the difference between the consumer's willingness to pay and the actual price paid. From a technological perspective this consumer surplus is an internal outcome of the market activities and no exceptional effect at all. Likewise, producers can increase their individual profit (producer surplus) by using road transport as long as they improve their production technology as a result of lower input costs.

With respect to the consumers, these effects are nothing less than the realisation of the increased consumer surplus. Hence, there is absolutely no need for public intervention. This is because there is no possibility whatsoever to increase social welfare by any governmental activity, which aims to incite the respective activity. Hence, is seems obvious that these effects are not suitable for compensating external costs stemming from road transport.

Analogously, with respect to producer's surplus, the described individual producer's advantage require no government intervention in terms of a compensation for the transport sector to make it possible for the producer to realise an improved efficiency in the respective production.

As an intermediate result, it can be concluded that a large number of the alleged external benefits of road transport actually can be excluded as such.

Furthermore, Verhoef (1994) presents the following argument:

> [*S*]*pin-off effects* of road transport in terms of value added or employment in related economic sectors (vehicle manufacturing or maintenance, oil industry, etc.) do not provide a sound economic base for not restricting road transport to its optimal level.[24]

As another group of alleged external benefits, Verhoef (1994) identifies such external relations associated with transport that are actually not attached with a product price. However, these effects do not seem to fulfil what is defined as external benefits and can also be excluded from the list of purported external benefits of road transport. An example would be that people derive benefits from visiting guests without actually compensating the associated costs of travelling. Since the travelling person usually expects a counter visit at some point, an implicit arrangement has been made between both involved parties in not paying each other's travel expenses.

As an alternative to this barter behaviour, one can also imagine some kind of altruistic behaviour. Here, "the visitor takes the utility of the receiving person into account."[25] It is expected that the receiving party is revealing its willingness to pay for more, directly to the visitor, since he feels not being paid enough visits. Both cases are consistent with Pareto optimality, which implies that social welfare cannot be increased by measures such as encouraging the described behaviour.

So far the analysis has clearly demonstrated that most of the effects included in the list of alleged external benefits of road transport can be identified as not being external, but rather as consumer or producer surpluses. From a market

[24] Verhoef (1994), p. 277.
[25] Ibid.

perspective, these effects are intrinsically internal. The remaining examples listed above may also be ruled out as being external benefits by looking at the various possible sources of external effects of road transport more closely, which will be done in the following section.

5.6 Sources of External Effects of Road Transport

Unlike most production processes generating external effects, analysing the transport sector requires a separation between the provider and owner of the transportation sector from those who actually use it in order to get useful results. Therefore, especially in term of responsibility for these external effects in the transport sector, it is of great importance to identify the exact sources of them. Hence, the externalities arising in the transport sector can generally be subdivided into the following two sections:

- Externalities stemming from the provision of transport infrastructure.
- Externalities stemming from the actual use of the provided infrastructure, i.e. externalities of road transport itself.

Obviously, these two sections have often been confused and led to sometimes rather surprising and obscure results, especially regarding the discussion on alleged external benefits of road transport. In order to clarify some of the statements made in several studies, the two types of externalities will be characterised and the main differences will now be discussed.

5.6.1 Externalities Stemming from the Provision of Road Infrastructure

Upon closer examination of the argument that road transport would have for example positive employment effects in remote and less developed regions and hence would increase the regional product, it can be argued that the externality is actually already incorporated in the transport infrastructure. Both the production of factories as well as the consumption of households are influenced by any improvement of the transport infrastructure and certainly may lead to a reduction of transportation costs as well as time costs for travelling. As a result of these effects, there may be "substantial redistribution effects among economic groups and also among regions."[26] The public using taxpayer money in order to subsidise at least a portion of the arising costs, usually provides the transport infrastructure. The use of this transport infrastructure is not generating any additional positive effects.

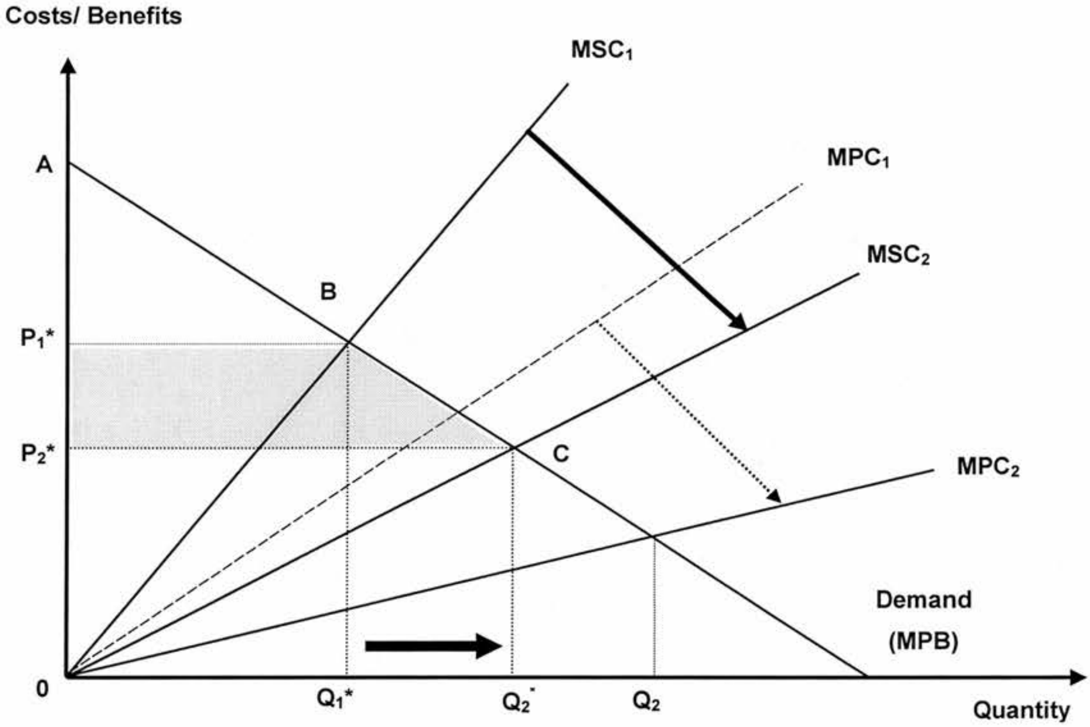
Rothengatter (1994) also mentions that "as long as the road haulage industry does not cross-subsidize the less developed countryside to assist the public there is no reason for an additional compensation for their services beyond the market transport prices."[27] The described effects concerning investments in road infrastructure can be illustrated by using the following figure.

---

[26] Rietfield (1989), p. 256.

[27] Rothengatter (1994), p. 316.

Figure 29: Externalities Stemming from the Provision of Road Infrastructure.



Source: Verhoef (1994), p. 274.

In the figure above, the shaded area of increased consumer surplus represents the extra benefits, which can be enjoyed by the users of road infrastructure resulting from infrastructure improvements. An example would be the gain of travel time resulting from a better-developed road network. Investment in additional infrastructure is desirable from a social point of view, as long as the resulting increase in the social surpluses (i.e. the triangle OCA) at least covers the social costs of the respective investment in road infrastructure.[28] The following table summarises the most common externalities arising from the provision of road infrastructure:

[28] Obviously, this is only valid for the case where the assumption is fulfilled that infrastructure is actually used at the socially optimal point, since only at this point the maximum benefit of infrastructure is been reached. Any point beyond this level of transport would lead to lower benefit from a social point of view.

Table 41: Externalities Stemming from the Provision of Road Infrastructure.

External costs	External benefits
• effects from sealing off the surface • cutting off neighbourhood communications and interactions along the alignments • influence on biotops • detrimental effects on landscape • negative impacts from construction; e.g. water pollution • uncovered infrastructure costs	• improving access to remote regions • improving the spatial structure of economic activities, e.g. new land use patterns • improving productivity • providing a publicly-desired level of basic communications and mobility • synergetic effects for the whole transportation system by choosing second-best solutions at the sectoral level to provide first-best solutions at the system level (e.g. building underground tracks for public transport and allocation of the less expensive surface areas to commercial transport) • positive effects on (regional) economic development and employment levels

Source: Mauch and Rothengatter (1995), p.16 and Verhoef (1994), p. 277.

The items stated in the list above of external effects generated by the supply of transport infrastructure are important effects which have to be included in the consideration when making decision on investments into transport infrastructure, i.e. cost-benefit-analysis. In such an analysis, they are key figures in the decision making process on whether certain elements of the transport infrastructure should be built or not, and if an already existing infrastructure should be extended or not.

According to Rietveld (1989), the decision to build a new transport infrastructure may have different economic effects, depending on the already existing situation at the time the decision is being made. In the case where such infrastructure already exists, the effects of the building of new (additional) infrastructure would be neutral. The economic effect would be (moderately) positive in the case where there was no such infrastructure before.[29] On the other hand, negative effects would arise if the investment in new transport infrastructure were leading to redistributional effects between different regions.[30]

5.6.2 Externalities Stemming from the Use of Transport Infrastructure

As mentioned earlier, in order to analyse the external costs of road transport, it is essential to separate the external effects from transport infrastructure clearly from those of the road transport activities, i.e. the actual use of the transport infrastructure. The following table gives a summery of both external costs and external benefits stemming from the use of transport infrastructure.[31]

Table 42: External Effects Stemming from the Use of Transport Infrastructure.

External costs	External benefits
• environmental effects related to the transport activities, such as noise, air pollution, climate change, separation of communication between neighbourhoods, water and soil pollution, as well as disamenition and detrimental effects of operations on the infrastructure • traffic congestion in the sense of additional time and operation costs caused by user interactions • traffic accidents which deplete the stock of human resources; the costs are relevant inasmuch as they are not covered by insurance	• services of public social organisations • services of emergency systems producing societal benefits in excess of the individual benefits (this is in the case where they are defined as public good) • services from production of other public goods, such as school bus shuttles, military transport, etc. • increased security resulting from better accessibility for fire engines, police cars, ambulances, etc.

Source: Mauch and Rothengatter (1995), p.17, Kram et al. (1996), p. 27 and Verhoef (1994), pp. 277-278.

[29] See: Rietvield (1989), pp. 262-264.
[30] See: Ibid, pp. 270-272.
[31] See also: European Conference of Ministers of Transport (1992).

Concerning the external costs, two major types of externalities can be distinguished, environmental effects (especially pollution) and congestion. The discussion in the previous section leads to the reduction of the original fairly extensive list of alleged external benefits offered in various studies.

Eliminating those effects which are either not external or no such effects stemming from the actual use of the road infrastructure, the effects that can actually identified to be external benefits of road transport comprise the following:[32]

- benefits resulting from watching vehicles (car spotting).
- benefits resulting from generating information for the communication industry.
- benefits resulting from road emergency services.

However, it is also controversial whether these points are actually external benefits, and according to Rothengatter (1994) and Verhoef (1994), even these points can be ruled out as well. The external benefits enjoyed through watching vehicles could be excluded as they are not really relevant in this consideration.

As defined in Mauch and Rothengatter (1995), "an externality is relevant if it significantly affects the adaptive efficiency of the market economy".[33] Consequently, although there might be some externality from watching cars, there is absolutely no need to internalise them.

---

[32] See: ECOPLAN (1992, 1993).
[33] Mauch and Rothengatter (1995), p. 10.

Likewise, the alleged benefits resulting from the 'generation' of information for the industry sector can be excluded from the list for two reasons. First, they are not really unusual market phenomena and second, it seems very hard if not impossible to find an argument which would support the statement that these effects would not arise on a completely voluntary base at all.

Probably the category most often claimed to generate external benefits of road transport is the increased security stemming from an improved accessibility for emergency services, such as police cars, ambulances, fire engines, etc. These effects can actually be seen as being relevant if they are treated as public goods.[34]

However, an argument exists that increased security resulting from the improved accessibility for emergency services may actually not be seen as falling into the category of external benefits from road transport. According to Rothengatter (1994), these benefits are already partly included, if the road truck costs of these users are subtracted from the total bill. He further claims, "that part of social benefits of emergency services would have to be cancelled which is supplied for road accidents".[35]

In addition, Verhoef (1994) argues that the improved accessibility under consideration is in fact just a result of the provided transport infrastructure and is not generated by the actual transport activity as such.[36] In this respect, he even claims that the actual external costs of road transport could be increased, and the emergency services could suffer from congestion caused by increased accessibility, which again is the result of the improved infrastructure.

---

[34] See: Mauch and Rothengatter (1995), p. 18.
[35] Rothengatter (1994), p. 326.
[36] See: Verhoef (1994), p. 278.

Hence, it seems rather unlikely that there are any external benefits in terms of an improved emergency system. As Rothengatter (1994) concludes:

> If there is an uncompensated [positive] externality left, it gives rise to subsidize the emergency system. But it is not justified to deduce the external benefits of emergency systems from the external cost bill of car travel or trucking.[37]

Obviously, after a careful analysis of the fairly impressive list of putative external benefits presented in several studies, it seems to be that there are actually no significant positive external effects stemming from road transport. However, it also seems legitimate to make a short remark on the theoretical situation where external benefits of road transport actually do exist. These benefits are external effects which would occur on **level 3** in the model described earlier, i.e. interactions between the production or consumption sector (private or public) and the transport sector. Hence, if the benefits from road transport enjoyed by the private production or consumption sector would be greater than the benefits resulting if all transactions are actually operated through the normal market mechanism, social welfare would be improved by compensating the transport sector by the private sector.[38]

Since it would be rather unlikely that a private solution of this situation would be found due to the sheer amount of individual agents involved, it would be for the public to intervene via an appropriate tax/ subsidise scheme. The obvious policy would be to tax the producer/ consumer, and subsidise the transport sector.

Rothengatter (1994), however, argues that simply offsetting the external costs against the external benefits would not be the appropriate policy, since it would

[37] Rothengatter (1994), p. 326.
[38] The same applies, of course, for the public sector.

leave the sector which benefits from the subsidy of the transport sector completely unchanged. Rather, such a policy is proposed for solving the externality problem occurring on different decision levels which would "let the transport sector pay fully for the external costs of using non-renewable resources and establish a transfer payment from the private production/ consumption sector to the transport sector for compensating for external benefits".[39]

Moreover, assuming that a market economy is given where decisions are taken on a decentralised level, the costs and benefits should be allocated on an individual base, taking the involved agents at the respective location directly into account. In other words, it would be the wrong approach simply to allocate net benefits and net costs respectively which are obtained by a global macroeconomic benefit/ cost calculation.

5.7 Summary

The analysis in this chapter primarily concentrated on the issue of external effects occurring in the transport sector, focusing intently on the identification and specification of the external costs resulting from them. Since these external costs are a main part of the 'total' cost of road transportation it is of great importance to value them in order to calculate the total costs due to road transport. It could be demonstrated that those externalities in the transport sector which call for public intervention are those which are associated with the free of charge consumption of non-renewable resources (level 1 externalities). In other words, the transport sector exhausts the environment as well as the human capital mainly by air pollution and road accidents.

---

[39] Rothengatter (1994), pp. 326-327.

Furthermore, the discussion in this chapter clearly showed that it is essential for the analysis of externalities in the road sector, that one distinguishes between the externalities arising from the provision of transport infrastructure (supply of road network) and those resulting from the actual use of this infrastructure. This distinction helps to clarify certain related issues, particularly when referring to the, at times, controversial discussion of external benefits and the question of whether such external benefits, if they in fact exist, compensate for external costs of transport.

Generally, one of the main conclusions that can be drawn from the analysis in this chapter is that it is actually not possible to offset external benefits of road transport against respective external costs. The main reason for this is that after having tested the long list of putative external benefits presented in various studies for various criteria, it can be shown that it is possible to reduce the number of effects included in the list significantly.

By eliminating those affects which are not external at all but are rather confused with consumer's and producer's surpluses and those which are related to normal market interactions, the remaining potential external benefits of road transport are fairly insignificant, such as the utility resulting from car spotting. Hence, the only item left would be the benefits of improved emergency services through better accessibility, when treated as a public good, although there are also several arguments given why even this effect can be excluded. Consequently, gathering from the discussion, there seems to be no ultimate justification for any lump-sum transfer between external benefits of road transport and respective external costs.

On the contrary, since road transport is causing a significant amount of external costs, which do not operate through the normal market mechanism, a motivation for appropriate public policy intervention schemes is given. Such schemes, however, require the knowledge of the absolute amount of external costs associated with road traffic related air pollution. This study is concerned with one of the main categories of such costs; the premature deaths that can be related to the exposure to various air pollutants. Policy actions, such as the internalisation, where the knowledge of the monetary value of this important component of the full costs of air pollution is required, leads ultimately to the issue of the valuation of human life, which will be discussed in detail in the following chapter.

6. The Economic Valuation of Life and Health

6.1 Introduction

As mentioned in previous chapters, air pollution is the cause of a number of different kinds of damage. Some of this damage results in financial expense, such as the corrosion of buildings and vehicles, the loss of income due to reduced harvests, as well as medical care for people suffering from air pollution related diseases. Principally, such direct expenses can be computed at market prices. However, following the discussion in the previous chapter, air pollution also causes substantial damages which cannot be valued in market prices.

One such damage, which is the focus of this present study, is the adverse effect of air pollution on human health and in particular premature death caused by the exposure to traffic related-air pollution.[1] The loss of human life can be seen as a 'non-market good' where it is impossible to assign a price to these goods on the basis of the market mechanism alone.[2] Nevertheless, regardless of the fact that they do not have a specific market price, one cannot necessarily conclude that they do not have a value representing the contribution these goods make to people's well being. However, an approach to value these non-market goods is not without any difficulties, regardless of the valuation technique actually used.

Possibly the most obvious problem generally related with the analysis of potential external health costs stemming from road transport is assigning a monetary value to changes in health status or even length of life. The assigning of a value measure in monetary units to human life carries with it great potential

---

[1] Other non-market damage due to air pollution is mainly damage to environmental resources, such as flora and fauna, including the loss of biodiversity. See: Zweifel and Breyer (1997), p. 24. In the discussion on 'dying in dignity', this distinction plays a major role.

[2] See discussion in Chapters 4 and 5.

to generate controversy. Obviously, the first putative objection of such an idea is the fact that it is morally and ethically reprehensible to compare human life and money. Second, even when this first objective may have been accepted on a more pragmatic level, the objection still may be raised that such a comparison between human life and monetary values would only be acceptable if the result is an infinite value of life.

This chapter is structured as follows, initially, the two frequently raised issues noted above are discussed. The subsequent section will then introduce and evaluate the basic conceptions on valuing human lives. From this discussion it is then concluded that the so-called willingness to pay approach offers the most appropriate and systematic valuation approach at the current state. Results of empirical studies are presented before applying this approach to the specific case of air pollution. Finally, an air pollution related value-band of preventing a statistical fatality is presented, adjusting for various factors, such as age, life expectancy, and quality of life.

6.2 Potential Objections against the Valuation of Human Life

6.2.1 Objections against the Weighing of Life against Money Values

As mentioned above, it seems unavoidable that there will be an objection raised against the fact that human life is expressed in any kind of value term e. g. putting a monetary value on human life. In this context, the following statement is put forward:

> The weighing of life and freedom of bodily harm on one side and the money on the other is considered profane by moral rigorists, whether inspired by Christian belief, the oath of Hippocrates, or humanistic philosophy of life. Sometimes economic approaches to these valuations are even put on a par with the euthanasia programs of the Third Reich. Does such a valuation not imply that it is acceptable to kill those, whose "value" does not cover the cost of living, such as food and medical treatment?[3]

Following Breyer and Zweifel (1996), this statement can, however, be weakened by considering various arguments which lead to the point that the fact that the value of a human life is expressed in monetary terms is in fact not identical with its actual market or financial value. Rather, it "amounts to describing the preferences of the individual concerned or of society at large among mutual exclusive alternatives".[4]

Breyer and Zweifel (1996) suggest that the argument quoted above, generally does not take the morally relevant difference between actively intervening and letting nature run its course into account.[5]

Perhaps the main justification for elaborating and applying an explicit economic evaluation approach is the fact that certain political decisions by parliaments and/or public authorities have to be made on a fairly regular basis. Hence, in order to achieve sound and consistent decisions, they have to be based on such valuations, which imply the weighing of the preservation and lengthening of human life against the input of scarce resources having alternative uses. A decision to prolong human lives requires sacrificing the consumption of goods and services, which could be produced instead using these resources.

---

[3] Breyer and Zweifel (1996), p. 22.
[4] Ibid, p. 24.
[5] In the discussion on 'dying in dignity', this distinction plays a major role, where it is important to recognise the (morally) difference between the act of killing and making do without life-support devices.

Hence, by undertaking action or not, an implicit decision on a weighing of prolonged (statistical) lives against money values has been made. Accordingly, the economic calculus facilitates some kind of awareness on these decisions and finally helps them to make more consistent political decisions. Such public sector decisions do not only occur in relation to health care, but also in various other sectors.

A prime example is certainly the transportation sector, where communities regularly have to decide whether to take any action concerning notorious scenes of accidents, such as eliminating narrow blind road curves by investing in the widening and straightening of the particular road section. Similarly, in residential areas, the implementing of speed-reducing measurements, such as the installing of speed bumps and the planting of additional trees, may reduce the risk of playing children being involved in traffic accidents. Further examples can be found considering environmental policy decisions, such as any kind of safety considerations in nuclear power plants as well as measurements to reduce the emissions of harmful substances from various stationary and mobile sources.

All these considerations usually require some additional expenditure from the public purse. In a situation where economic valuation of human life for whatever reason is foregone, there may be the risk that comparable high cost measures are taken to avoid premature death and failing to take into account possible alternative cheaper measures. As a consequence, society as a whole fails to reach a longer life expectation as well as higher consumption.[6] Thus, a rational decision by the respective authorities can only be made, and consequently be accepted by the public, in the case where a comprehensive and precise valuation of all involved aspects are taken into consideration.

---

[6] This only holds for the case where the life years won or premature deaths avoided do not depend on who is actually obtaining them.

In other words, all future advantages and possibly disadvantages arising from a certain project have to be compared with the present value of the cost stream related to this measure. For such cost-benefit analysis it is of great advantage if the measures considered have the same unit. Consequently, since the costs associated are usually in monetary units, it is certainly desirable that all advantages are also expressed in this measure.

Obviously, this of course includes any measure to prolong human life or to improve the state of health.[7] Furthermore, such a decision-making process in a democracy should not only be consistent, as mentioned above, it should also reflect the preferences of the people involved, regarding life span and quality of life. As Breyer and Zweifel (1996) conclude:

> Since many public decisions inevitably imply a weighing of prolonging statistical lives against other goods, it is advantageous for society to make such valuation explicitly. However, the preferences of the citizens should be reflected in this valuation.[8]

Consequently, it seems important to emphasise that expressing the value of human life in monetary units has to be strictly differentiated from estimating its market or financial value. Rather, it reflects somehow the individual's or society's preferences among mutually exclusive alternatives.[9]

Breyer and Zweifel (1996) also argue that it is not so much the issue of weighing human life against monetary values but rather the length of the life span against money. Medical treatments of public security measures actually prolong life rather than save it since everybody ultimately dies. The moral argument of an

---

[7] See: Breyer and Zweifel (1996), pp. 19-20.
[8] Ibid, p. 24.
[9] Also, the value of human life is not meant to be equivalent to what one would pay in ransom money to a kidnapper in order to save somebody's life.

inadmissible valuation of life in terms of money loses its convincing power, since "a weighing of an extended life span against a better quality of life is what this is about, since 'more money' means more of consumption possibilities, permitting a higher quality of life".[10]

Finally, to conclude the discussion, it is of great importance to emphasise that most public decisions do affect the so-called 'statistical life' and not the actual 'identified' life, i.e. one is in no way concerned with placing a monetary value upon the life of a particular individual. This leads inevitably to the issue of ex ante and ex post values, which will be discussed in more detail in the following section.

6.2.2 Arguments against a Finite Value of Life

From the discussion above and particularly from the point of taking individual's preferences into account, another important point of potential criticism arises. This objection rejects the idea of a specific finite economic valuation of human life and assumes that only an infinite value can be an acceptable result. Accordingly, there are principally only two possible approaches which may be used to determine the amount of money reflecting the value of an individual's own life. Consequently, it is required to know the outcome of either of the following formulations:

---

[10] Breyer and Zweifel (1996), p. 23.

- The amount an individual would be willing to pay to avoid certain and immediate death; or

- The compensation that would have to be paid to the individual to make them accept immediate death.

The first formulation is not particular informative, since probably most people are willing to give up their entire wealth including their future income stream in order to avert the risk of death. Hence, this amount reflects more about the individual's wealth and ability to run up debts than about their actual preferences. The amount which is determinated by the second formulation, however, is most likely not a finite one, since there seems to be no use for money after one's death.[11] Both alternative approaches to define the willingness to pay differ in their allocation of property rights in the sense that only the second formulation is based on the assumption that an individual has the right to live, which, however, can be given up voluntarily. Based on this argument the value of a 'personalised' life may indeed be infinite.[12] This was subject of a rather vivid and controversial discussion over years mainly between Prof. Broome and the advocates of the valuation approach.[13]

One of the main issues of this discussion was the explicit distinction between an ex ante valuation approach and an ex post valuation approach. The ex ante valuation approach assumes that the economic value is derived before the uncertainty about the individual's death during a specified period of time is actually resolved, i.e. before it is known which individual died. In contrast, when

---

[11] Here, it is, of course, assumed that there is no bequest motive.

[12] A personalised life is meant here as opposed to a statistical life.

[13] See: Broome (1978a, 1978b, 1979, 1982, 1985, 1987), Buchanan and Faith (1979), Jones-Lee (1979, 1982, 1987), Williams (1979), Mishan (1971b, 1981, 1985), Ulph (1982), Thaler (1982) and Linnerooth (1979, 1982).

assuming an ex post perspective, each individual will actually know if he/she is to die now or live a while longer. Therefore, it seems obvious that those who would die would be willing to pay an infinite amount of money to change the outcome.

When criticising the economic approach to value human life, the main argument is that this difference in perspective, i.e. ex ante versus ex post, "can have no ethical or moral significance and that therefore the willingness to pay or compensation measures based on the ex ante perspective are morally unacceptable."[14] In other words, critics of the valuation approach argue that a distinction between a personalised and statistical life is not legitimate, since the underlying concept for the formulation of statistical lives is based on incomplete information about the fact of who is actually going to lose their life.

The following example has been used to support this argument. It is assumed that according to statistics, in the course of some public building activities, one worker will be killed who, however, is not yet identified. Consequently, this 'statistical' life is in fact infinitely precious. This is, since according to the second formulation mentioned above, there will be a call for an infinite compensation for the loss of this specific personalised life as soon as the incident has happened and the victim is known. However, it has been shown that this argument as brought forward by Broome is flawed. Following Breyer and Zweifel (1996), it can be argued that it seems rather difficult to think of a risk where not only the actual number of victims is exactly known in advance, but also with certainty.

---

[14] Freeman (1993), p. 322. See also Broome (1978a).

On the contrary, it seems that for most cases it is almost impossible to announce with certainty that there will be any victims at all. The example they use is as follows. It is assumed that due to a dangerous road curve on average one person per year has died in an accident at this particular part of the road in the past. However, this does not imply that one can state with certainty that every year exactly one person will actually die. Rather, it is unlikely to observe exactly one death during a given year, which can be illustrated using the following additional information to the example described above. If it is assumed that, for example, there are 100,000 road users per year, then each of them will face a risk of 1 to 100,000 of being killed in an accident at this particular part of the road. Adding up these individual probabilities over time would presume a perfectly negative correlation of the individual risks.

However, it seems more plausible to assume a stochastic independence or even a positive correlation, which will lead to the result that anybody will be killed with a certain strictly positive probability or that all 100,000 road users will be killed with a positive but very small probability. Hence, the total number of individuals being killed is actually not known. Following from this example, it can be concluded that using the concept of a 'statistical life' is the relevant concept for an economic valuation. Additionally, it is argued that once it becomes the responsibility of each individual to reduce or accept small risks to their own life, it seems legitimate that limited amounts of money are sufficient to compensate an individual for taking such a risk. They give several examples, which support this argument, where individuals are voluntarily willing to take risk to their lives for the sake of thrill, pleasure, or comfort, such as driving without a seat-belt, child restraints, wearing no motorcycle helmets, smoking or travelling by car or plane rather than by train.[15]

---

[15] See, for instance, Blomquist, Miller and Levy (1996).

Focusing on the transport sector, on which the majority of studies trying to estimate the value of life are based on, Hauer (1994) uses the following example against the objection against a finite value of life. "Also, with the turn of the ignition key all drivers indicate that the benefit of the journey is greater than the finite probability of dying in its course." He further claims that "[t]herefore, a person proclaiming that life's value is infinite, has a source of value different from that guiding the choices that most people make."[16] Fraser (1984) states that apart from the "iconoclastic pessimism of Broome (1978a, 1979), the contributors to the outlined debate "seem agreed that assigning a monetary value to life is certainly defensible for the purpose of evaluating projects which change risks to life and limb."[17] Usher (1985) opposes the objections that life is too precious to weigh against a monetary rod because such an objections cannot be translated into rules for any authority (e.g. the Department of the Environment, Transport and the Regions). He states that "[t]o refuse to compare lives and money is not to treat life as infinitely precious. It is to treat life as worthless."[18] Roads, for instance would then be built regardless of lives saved or lost; no attempt can be made to minimise the number of lives lost in road traffic accidents, and hence "the entire national income is not large enough to finance a policy of reducing fatalities in every possible way, regardless of the cost."[19]

Ultimately, the argument can be supported that avoiding small risks is actually not infinitely valuable to people and consequently it not only seems legitimate to weigh (statistical) life against money but also to assume a finite value of life.

---

[16] Hauer (1994), p. 110.
[17] Fraser (1984), p. 307.
[18] Usher (1985), p. 169.
[19] Ibid.

6.3 Basic Conceptions of Valuing Human Lives

Having discussed the main objections against the approach of valuing human life and concluding that it is legitimate to provide a value of a *statistical* life, the remainder of this chapter is mainly concerned with first the procedure on how to obtain such estimates, and second with the discussion of current empirical evidence. Before proceeding with the discussion, it seems appropriate to note that the expression 'value of a statistical life' (hereafter VOSL) can somewhat be the grounds for misinterpretation, since "it is not about valuing 1 life, but about the aggregate value that a large group of people places on typically very small reductions in the risk faced by each individual member of the group."[20] Hence, it would be more appropriately named 'value of risking a life' or 'value of prevention of a statistical fatality' (hereafter VPF), as used by the DETR, and other recent studies.[21] However, in the remainder of this study, both expressions of 'value of a statistical life' and 'value of preventing a statistical fatality' will be used interchangeably, particularly since the relevant literature predominately used the phrase VOSL.

Over the last two decades, a large body of literature on the economic valuation of human life has been published. Hence, a vast amount of literature can be found where not only the advantages and limitations of various valuation methods are discussed, but also estimates of the value of a statistical life are provided.

---

[20] Department of Health (1999), p. 36.

[21] See: Department of the Environment, Transport and the Regions (1999), Department of Health (1999) and World Health Organization (1999).

Whilst the early literature was mainly concerned with the understanding of the general value of life issue and developing a fundamental and meaningful economic framework, research throughout the 1970s and 1980s was mainly concerned with the establishment of sound empirical results to support the theoretical analysis. These empirical estimates were then gradually applied in the policy making process. Viscusi (1992) mentions that "[o]nce an unmentionable issue, value-of-life tradeoffs are now recognized as quite amenable to economic analysis".[22] During this period where the value-of-life literature has gone through a number of stages, several different approaches to value the (welfare) costs arising from a shortened life expectancy have appeared.

Naturally, the resulting economic value is certainly influenced by the respective valuation method used, which in turn may explain the now and then significant differences in official estimates that have occurred in the literature. Although the various concepts may mainly be discussed in a more general context, the actual application of the different approaches is most often made in respect to the transport sector, where one is particularly concerned with the economic valuation of traffic accident fatalities.

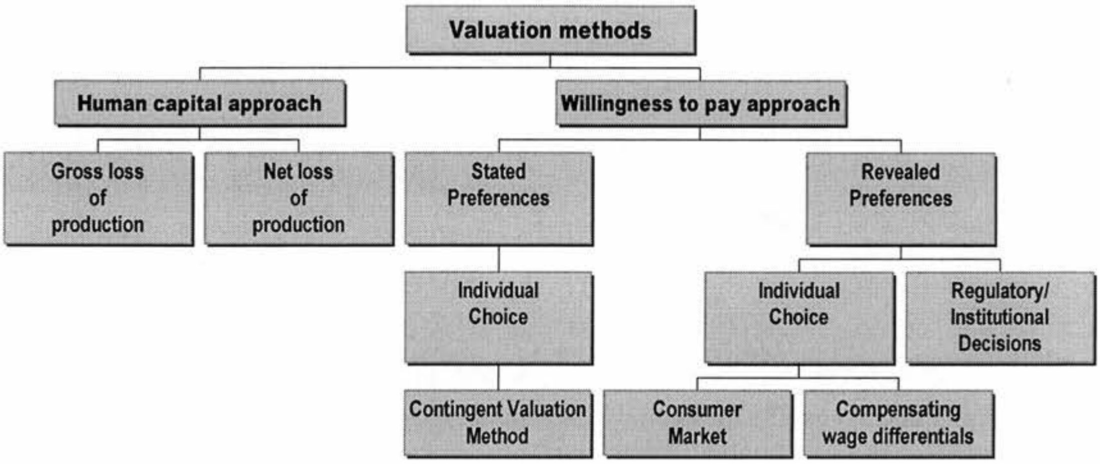
In the following section, the main relevant valuation methods will be introduced and subsequently discussed. However, the primary aim of the discussion in this section lies in providing a broad introduction on the various issues relating to the economic valuation of human life. Therefore, no claim of a complete and comprehensive coverage of all possible valuation issues shall be made here, neither for the various methodologies that have been discussed in the literature nor for the numerous associated empirical estimates.

---

[22] Viscusi (1992), pp. 31-32.

The diagram below provides a broad overview of the main valuation methods that been applied in studies aiming to value a statistical life.[23]

Figure 30: Main Methods for Valuing Human Life.



Source: Adapted from Alfaro, Chapuis and Fabre (1994); Zweifel and Breyer (1997), pp. 9-41 and Soby and Ball (1991).

Basically, two completely different conceptions on how to attribute a monetary value to human life have developed in the literature; the human capital approach and the willingness to pay approach, including various subcategories. Both methods have advantages and drawbacks. The following section provides a general outline of both methods including a critical discussion of their applicability.

---

[23] Earlier methods of estimating the value of life and injury are based on methods including life insurance coverage and legal compensations. Both concepts, however, are seen as not appropriate as a basis for establishing precise and consistent values. For the case of the economic valuation of road accident fatalities, the so-called cost of restitution method has also been developed. This valuation method is basically based on the resources necessary to correct the effects of, for example, a road accident. This valuation method tries to measure the costs for society to restore the victims (in the case of non-fatality) or his/her relatives and friends to the situation in which they were before the accident. Based on the effective value of the production function involved (work/ capital) or values derived from expert opinions or from court judgements, the resulting estimates are often used as the basis for sums paid in compensations by insurance companies. However, the effectiveness of this approach depends very much on the availability of relevant statistical sources and their quality and will not further be considered in this discussion. See: Alfaro, Chapuis, and Fabre (1994) and Soby and Ball (1991).

6.3.1 The Human Capital Approach

The economic valuation of life based on the human capital approach has a long history, which can be dated back to the 17^{th} and 18^{th} century.[24] Subsequent studies improved and developed the theoretical and practical aspects of this approach further, which finally lead to the groundbreaking paper of Rice and Cooper (1967), in which they established the empirical application of this valuation technique for the first time.[25]

The underlying assumption of the human capital approach is the fact that the value to society of an individual's human life can be measured by the present and future production potential.[26] The future production potential is usually measured as the present discounted value of expected earnings from labour. In other words, the value of human life can be expressed as the economic value, which is equal to the discounted sum of the individual's future marginal contributions to the total social product. This contribution again corresponds to the person's future labour income, while the assumption that the respective wage is equal to the value marginal product has to be fulfilled.

---

[24] See: Petty (1699) and Farr (1876), as referenced in Landefeld and Seskin (1982), p. 555 and Williamson (1984), p. 157. The original intent of application of the human capital approach during the twentieth century was to estimate optimal life insurance, before it has been used later to assess the income losses from accidents and disease. See: Linnerooth (1979).

[25] See: Rice and Cooper (1967).

[26] See: Elvik (1995), p. 237.

In other words, the value of preventing the premature death of an individual who is presently of age t can be computed as follows:[27]

$$Value\left(HC\right) = \sum_{i=1}^{T-t} \frac{\pi_{t+i} \times E_{t+i}}{\left(1+r\right)^{i}} \qquad (11)$$

where:

π_{t+1}:	probability of the individual surviving from age t to age t + i.
E_{t+1}:	expected earnings of the individual at age t+i.[28]
r :	social discount rate.[29]
T :	age at retirement from labour force.
T - t :	remaining lifetime.

Basically, two variations of the human capital method have been applied in respective studies. The illustrations so far are related to the method of the so-called gross loss of production, i.e. a person's human capital value is simply the discounted value of future income. In contrast to this method, the so-called net loss of production approach evaluates the reduction in productive potential, excluding the expenditure on consumption. Thus, the cost of premature death is defined as the gross output minus the present value of future consumption, which the respective person would have been able to enjoy had he or she not prematurely died.[30] In other words, the net human capital approach, i.e. expected future earnings net of consumption, assumes that with the death of an individual, not only the respective productive contribution to society will be lost, but also the claims on future consumption.[31]

---

[27] See: Freeman (1993), pp. 322-325.

[28] This may, for example, also include such elements as the imputed value of non-market time spent on housekeeping activities.

[29] This discount rate may be interpreted as opportunity cost of society investing in life-saving programmes instead of some 'next-best' alternative investments.

[30] See: Dalvi (1988), pp. 3-4.

[31] See: Landefeld and Seskin (1982), p. 556.

Generally, the human capital approach is based on two implicit postulates, which are now reproduced following Breyer and Zweifel (1996):[32]

- An individual's value depends on the contribution that he or she makes to the welfare of its fellow citizens.

- The appropriate measure of society's welfare is the Gross National Product.

Both these postulates, however, face serious criticism and are the basis of the shortcomings related to the human capital approach. Some of these main issues addressed concerning the implementation of the human capital approach will be discussed in the following.

With regard to the first postulate, it seems more appropriate to a slaveholder society than to a modern liberal democracy. This statement can be based on the fact that it suggests that there is no difference whether a machine or a human being makes the contribution. This rather drastic view can even be strengthened when the net human capital approach is applied. In this case, it is even assumed that any single individual does not count as a member of society, which is due to the fact that his or her own loss of future consumption is not included in the calculation to value his or her own life.

More ethical objections against the use of the human capital approach may be raised when considering that according to the assumptions mentioned above, the value of life for members of the society who are unable to work, such as pensioners or children, is always zero; for the case of the net human capital

[32] See: Breyer and Zweifel (1996), p. 26.

approach, this value may even be negative.[33] Freeman (1993) claims that this measure is in fact "the antithesis of the individualistic premise of conventional welfare economics".[34]

Referring to the second postulate, basing the valuation method solely on GNP results in values which totally ignore such factors as the pleasure of living, for example.

Further, when discounting future earnings, an important and critical issue that must be resolved involves the choice of an appropriate 'social' discount rate to obtain present values. Landefeld and Seskin (1982) state:

> The problem amounts to determining what society forgoes when it invests in life-saving programs. The choice is made difficult because of the effects of taxes and risk aversion which cause the rate of return to society's investments to diverge from the rates of return to private investments."[35]

In other words, taxation leads to the result that the before-tax rate of return to private investment, i.e. the marginal productivity of capital, actually exceeds the after-tax rate of return to the individual investor, i.e. the individual's rate of time preference. This divergence may also result from a risk premium due to risk aversion.

---

[33] Additionally, there is also the problem how to estimate the contribution to the GNP made by people who work but do not actually receive wages, such as housewives and househusband; although for such services now typically imputations are made. See: Jones-Lee (1990), p. 40. Further, considering imperfect labour markets may raise the question regarding possible discrepancies between the wage and the marginal productivity of labour.

[34] Freeman (1993), p. 323.

[35] Landefeld and Seskin (1982), p. 556.

Further, the estimated present value significantly varies with the choice of the real discount rate; the larger the real rate chosen, the smaller the resulting present value. This choice may also affect the relative valuations placed on persons in specific age groups. So, it can be shown that at a certain real discount rate somebody in the age group of 20 to 24 is valued higher than those in the age group 40 to 44, whereas at a higher real discount rate the reverse is true.[36] In other words, the higher the chosen discount rate, the less the value accorded to costs spread out over time. The human capital value of children and young adults in particular, is sensitive to the choice of the discount rate. Freeman (1993) states that "because of discounting and the time lag before children become productive participants in the economy, the human capital approach places a much lower value on saving children's lives compared with saving the lives of adults in their peak earnings years."[37]

Further, there are a number of general uncertainties that may cause errors in respective human capital approach calculations. Neither are the future earnings throughout the rest of an individual's lifetime known with certainty, nor are future rates of inflation and interest.[38]

Consequently, it seems clearly understandable that the human capital approach is ethically misguided, since it assumes that a value of a person is solely based on the amount of goods and services he or she may produce, i.e. the value of an individual's life is solely based in terms of economic output. Rather, the criteria to evaluate the lives of human beings should be other than productive capacity and hence the desires and values about people's own lives should form the basis for the estimation of the value of life.

---

[36] See: Landefeld and Seskin (1982), p. 556.
[37] Freeman (1993), p. 324.
[38] See: Viscusi (1990), pp. 61-62.

Although both the gross and net method of the human capital approach take other 'direct' economic effects of death (and injury) into account,[39] they actually ignore some other dimensions of illness and death as well as non-market activities, which may be of greater important to a person than the economic loss as such.[40] This shortcoming is mainly founded in the fact that the main emphasis of the human capital approach is on economic product.

Summing up the discussion, it can be stated, that "[a]ccording to the human capital approach, the value of life is determined by the contribution the individual could make to the social product. Its relatively easy application is outweighed by serious economic and ethical shortcomings."[41] The human capital approach discriminates against the young, the old, women, and minorities, providing very low values for all these groups. Hodgson (1983) comes to the conclusion that the human capital approach in fact does not measure the value of life,[42] whereas others called this approach 'the value of livelihood' approach'.[43]

Finally, although more subtle versions of the human capital approach add some estimate of the individual's expected non-market contribution, including their affective value for other people,[44] it still "reduces the value of a person's life to what the others get out of it and hence ignores altogether the value of a person's life to the person herself".[45] The human capital method is at odds with both the principles of welfare economics and the theory of value.[46]

---

[39] Such 'direct' costs include medical and police costs as well as material damage.
[40] These activities include factors such as pain, grief and suffering, aversion to risk, as well as the loss of leisure time, which may have a value itself for the respective individual and perhaps for others as well. See: Dalvi (1988), pp. 2-4.
[41] Breyer and Zweifel (1996), p. 26.
[42] See: Hodgson (1983), p.142.
[43] See: Soby and Ball (1991), p. 28.
[44] Here, often more or less arbitrary allowances for the pain, grief and suffering of the dependants, relatives, and friends is incorporated into the human capital measure. See, for instance, Jones-Lee (1990), p.40.
[45] Van Parijs (1992), p. 122.
[46] See: Viscusi (1986, 1994, 1998).

Until 1971, the then UK Department of Transport (now Department of the Environment, Transport and the Regions (DETR)), based its procedure for estimating road accident costs upon the net output approach. However, because estimated figures for females, older age groups, and for the very young were in fact negative, "an allowance was made for 'subjective' costs (pain, grief and suffering), loosely based on the consumption of a non-productive person."[47] The rationale being that this was a minimum measure of society's valuation of such person's continued existence. The change to the gross output approach was then made "on the grounds that the cost figures were used to measure the benefits of accident reduction and so it was right to include the consumption of those who avoided becoming casualties."[48]

Realising that the (gross) human capital approach is principally not only inconsistent with the main principles of welfare economics, but also with the theoretical principles of cost benefit analysis, this general approach to estimate the value of lost productivity also soon became the subject of heavy criticism.[49] Consequently, the DETR has now changed to the willingness to pay approach for reductions in risk as the base for its estimations. The HM Treasury gave the following recommendation:

> The appropriate starting point for valuing the avoidance of death or injury, as for savings in travelling time, is a measure of the individual's willingness to pay. . . . [T]he values generally in use are based on estimates of individual's willingness to pay for small incremental changes in their own risk of loss or injury. . . . . Thus it is the value to unnamed individuals of these marginal changes in risk which are valued rather than the 'lives' of named individuals. However, the term 'value of life' is a convenient shorthand used in much of the literature.[50]

---

[47] Dalvi (1988), p. 4.
[48] Ibid.
[49] See, for instance, Schelling (1968) and Mishan (1971b).
[50] HM Treasury (1991), Annex B.

Since 1993, the valuation of fatal casualties (as well as non-fatal casualties) has been based on a consistent willingness to pay approach by the DETR. This willingness to pay approach, which has now been also adopted by a large number of other western countries as the basis of their official economic valuation of traffic accidents, will be discussed in some detail in the following sections.[51]

6.3.2 The Willingness to Pay Approach

As discussed in the previous section, according to both theoretical reasoning and empirical evidence, the human capital measures are seen as "a poor proxy for the desired willingness to pay measure of value for small changes in the risk of death."[52]

Following one of the earliest theoretical discussions on the valuation of human life by Mishan (1971b), it is suggested that the basis for the valuation of human life in safety decisions should be consistent with the basic rationale of the economic calculus used in cost-benefit analysis, i.e. the potential Pareto improvement.[53] A potential Pareto improvement is an improvement such that the net gains can be re-distributed such that at least one person is made better off and nobody is made worse off.

---

[51] This includes Great Britain, the United States, New Zealand, Sweden and Switzerland. However, other countries do rely on the willingness to pay approach, because of remaining doubts regarding the validity and reliability of such estimates. See: Elvik (1995).

[52] Freeman (1993), p. 324.

[53] Mishan (1981).

So, the question to ask is what are individuals willing to pay (or accept as a compensation) for a change that will affect loss of human life.[54] In other words, an estimate for the value of a statistical life should measure the value that people place on risking their own lives.

Hence, a more appropriate alternative has been developed, which is based on the preferences of individuals and uses, as a measure of value, an indictor of their actual willingness to pay to reduce the risk of adverse effects; the so-called willingness to pay approach (WTP). Following Mishan (1985), "[e]conomists are generally agreed - rather as a canon of faith, as a political tenet, or as an act of expediency - to accept the dictum that each person knows his own interest best."[55]

Generally, the willingness to pay approach provides estimates, which are used to value the change in well being that would result from changing the risk of premature death. Usually, this estimate is measured by how much of other goods and services a specific individual is actually willing to give up to get that reduction in the risk of death. When these estimates are aggregated across all people, an estimated value of a statistical life is provided, which reflects what the whole group of people is willing to pay for reducing each member's risk of

---

[54] In this present study, no explicit distinction between the willingness-to pay and the willingness-to accept will be made, since the relevant empirical studies are predominantly based on the willingness to pay approach. However, from empirical evidence it is known that there may be a large disparity between the values obtained from both methodologies. In other words, it seems that individuals commonly demand far higher sums to give up entitlement than they are willing to pay to acquire or to maintain it. Gerking, De Haan and Schulze (1988), for instance, find that the values for willingness to pay may be up to 2.5 times greater than respective willingness-to-accept values. One reason could be that individuals may underperceive the value of gains or overperceive the value of losses. More generally, one could assume that individuals' preferences over combinations of risk, wealth, and safety are imprecise, which in turn may result in the observed disparity between willingness to pay and willingness-to-accept measures of value. However, this issues will not be discussed any further here. For detailed discussions see, for instance, Coursey, Hovis, and Schulze (1987), Knetsch and Sinden (1984), Cummings, Brookshire, and Schulze (1986), Gerking, De Haan and Schulze (1988), Loomis et al (1998) and Dubourg, Jones-Lee and Loomes (1994).

[55] Mishan (1985), p. 160.

premature death by a small amount. However, since risk is not directly traded in markets, the determination of people's willingness to pay for reductions in risks of death are difficult and not without problems.

The following are the main premises for the willingness to pay approach:[56]

- social decisions should reflect, as far as possible, the interest, preferences and attitudes to risk of those who are most likely to be affected by respective decisions.

- particularly in the case of safety, these interests, preferences and attitudes are most effectively summarised in terms of the amounts that individuals would be willing to pay or would require in compensation for typically small changes in the probability of premature death during forthcoming time period.

Mathematically, an individual's willingness to pay to avoid premature death can be simply expressed as follows:[57]

$$WTP = \left[\sum_{t=0}^{t-T} \frac{B_t}{(1+r)^t} \right] \times A \qquad (12)$$

where:

T:	remaining lifetime.
B_t:	utility or benefit enjoyed by living.[58]
r:	individual rate of time preference.
A:	risk aversion factor.

---

[56] See: Jones-Lee, Hammerton and Philips (1985), p. 49.
[57] See: Dalvi (1988), pp. 6-7, and Landefeld and Seskin (1982), pp. 560-561.
[58] This includes labour income, non-labour income, non-market activities and leisure, and a premium for pain, grief and suffering.

The future benefits, which are reflected by the summation over time of the benefits of living (B_t), include, in contrast to the human capital approach, besides labour income, other implicit components, such as non-labour income, non-market activities and leisure, and a premium for pain, grief and suffering. The discount rate, which is implicit in the determination of the present value of the individual's future benefits, is based on the rate of time preference of the respective individual and therefore avoids the important issue of selecting an appropriate discount rate.[59]

This is contrary to the human capital approach, where the social opportunity cost is used to derive estimates. In order to take account of the fact that it is very likely that individuals are at least as risk averse with respect to loss of life in revealing willingness to pay as they are with respect to financial loss, a risk-aversion factor (A) is also included.

Probably the most important advantage of the willingness to pay approach lies in the fact that it can be assumed that the willingness to pay, for instance, for safety improvement of an individual actually reflects this person's aversion to all of the adverse effects of death or injury. In contrast to the human capital approach, all the humanitarian effects mentioned above, are captured in this approach. The following sections will discuss the various procedures developed to measure individuals' willingness to pay.

---

[59] See, for instance, Cropper, Aydede and Portney (1991), Johannesson and Johansson (1997a), Dreyfus and Viscusi (1995) and Horowitz and Carson (1990).

6.3.3 Measuring Willingness to Pay

Generally, studies to estimate willingness to pay can be divided into two classifications, the direct approach of stated preferences and the indirect approach of revealed preferences.[60] The technique associated with the direct method of stated preferences is the so-called contingent valuation method. The category of the revealed preferences approach can first be divided whether the estimation is based on the preferences shown by individuals (individual choice) or society (regulatory/ institutional decision).

The latter approach is also known as the tutelary approach and appeals to the willingness of society. Here, "[t]he state or society as a whole assigns, often implicitly, a value to human life and health via their decisions in this area."[61] Hence, certain decisions on safety measures, such as accident prevention measures, will only be taken in the case where the putative benefits resulting from these measures indeed exceed the connected costs. These decisions are influenced by the government, by the legal, executive or judicial authorities, by the people,[62] or by companies. However, this approach has been under heavy criticism, including the fact that amounts spent to reduce risks to health imply values of life that may range from £200,000 up to £400 million (in 1986 values), which is indeed a very wide range, and will not be discussed any further in this present study.[63]

---

[60] See, for instance, Ebert (1998).
[61] Alfaro, Chapuis and Fabre (1994), p. 15.
[62] This can be either through their votes or by means of a referendum.
[63] For a more detailed discussion of this valuation approach, including a detailed discussion of the related criticism, see Soby and Hall (1991), pp. 34-37.

The two methods mainly used in relation with transport and health are so-called consumer market studies and compensating wage differentials. Both methods are based on individual choice where people make money-risk tradeoffs.

6.3.3.1 The Method of Revealed Preferences

Generally, the procedure of revealed preferences investigates the actual trade-offs, which are made by individuals in the case where environmental costs are involved. Following Chilton et al. (1998), "revealed preference methods attempt to infer the *implicit* trade-off that individuals make when faced with real economic decisions involving different levels of safety."[64] Two main procedures can be distinguished, the evaluation of consumer data and so-called compensating wage differentials or wage risk studies.

a) Consumer Market Data Evaluation

The technique most commonly used in evaluating consumer market data is the hedonic price approach, where estimates for people's willingness to pay are derived from the variations observed in the housing market. It is assumed that variations in the level of environmental intrusion are reflected in differentials in property values or rents.

---

[64] Chilton et al. (1998), p. 6.

Following Smith and Huang (1995):

> Under competitive conditions, a hedonic equilibrium requires that the change in price of a house in response to a change in any attribute (at given levels of the other attributes) exactly equal the marginal bid and marginal offer of the buyers and sellers for the characteristic. Thus the hedonic price function P(z) "defines" the market equilibrium, and its slope with respect to each characteristic provides an estimate of buyers' MWTP [marginal willingness to pay] for changes in that attribute.[65]

In such a hedonic or implicit house-price function, P = f(z), the vector z includes house-specific characteristics, which may be summarised as follows:[66]

Table 43: House-specific Characteristics included in a Hedonic Price Function.

Variable type	Examples of this category
Accommodation (i.e. attributes of the accommodation itself)	• number of rooms • presence of garage • size of garden • presence of central heating; etc.
Indicators of accessibility	distance to: • bus stop • town centre • school • shopping centre, etc.
Neighbourhood	• average age • race distribution; etc.
Indicators of environmental quality	• air pollution levels • noise levels • presence of scenic view; etc.

Source: Tinch (1995), p. 16.

Based on this function, willingness to pay estimates can be derived, since for each of these housing characteristics, each household is assumed to simultaneously equate marginal willingness to pay with the marginal implicit price of the respective characteristic. Although the property value effects are relatively easy to observe, this advantage is outweighed by a large number of problems and limitations of which this valuation method is subject to. So, it can

[65] Smith and Huang (1995), p. 211.
[66] See also: Hughes and Sirmans (1992).

only cope with existing levels of environmental intrusion such as air and noise pollution, substantial changes in these levels are consequently impossible to value.

Further, this method is not only limited to the housing market and relies upon the personal perception of the environmental intrusion (air pollution or noise), but also assumes that first, individuals understand the mortality risk from e.g. air pollution and second, that the only aspect of air pollution that has value to them is the mortality risk.[67] An extensive review, and further discussions of the problems involved can be found elsewhere.[68]

Other so-called consumer studies are directly related to the estimation of the value of a statistical life. Here, the observable trade-offs people make between risk and wealth in their everyday consumption decisions are examined. Examples include the analysis of data on the purchase price of smoke detectors and their effectiveness in reducing the probability of death resulting in an estimate for the value of a statistical life.[69] However, such consumer market studies have not been repeated many times using different data set, which significantly limits such studies to provide credible estimates of a statistical life. Additionally, such studies of consumer behaviour are subject to some serious criticism including the following:[70]

- Difficulty to separate what proportion of the purchase price is paid to reduce the risk of death and what proportion to reduce risk of injury.

---

[67] In other words, possible morbidity effects of air pollution are excluded.
[68] See, for instance, Freeman (1993), Johansson (1993), Viscusi (1992) and Tinch (1995).
[69] See: Dardis (1980). Other examples include choice on safety and fuel efficiency when purchasing an automobile. See: Dreyfus and Viscusi (1995).
[70] For a more detailed discussion of potential problems with consumer behaviour studies see Soby and Ball (1991), pp. 32-34.

- Potential inaccuracy of consumer perceptions of the level of risk associated with a hazard.

- Individual's motive behind the purchase and use of a safety device is unknown; is there a conscious weighing of 'cost' and 'benefit'?

- Estimates derived from market data reflect at best a lower bound of the value of life; it only reflects what the consumer has spent rather what she would be willing to spend for risk reduction. Also, possible altruistic concerns are not valued on the market and are not included. Hence, is it difficult to estimate the total monetary value people place on changes in mortality risks, i.e. the estimated value may only reflect the value placed on the individual's own risk.

Viscusi (1992) concludes that the described studies "are valuable in their own right but not for the purpose of establishing a value of life from the standpoint of government policy."[71]

b) Compensating Wage Differentials

Probably the most prominent and best-established procedure for the revealed preference approach is the method of compensating wage differentials. Developed more than 200 years ago by Adam Smith, this approach, which is also known as a so-called wage-risk study or the method of 'hedonic wages', is a frequently used method based on decisions made in the labour market, where the

---

[71] Viscusi (1992), p. 67.

implicit premium paid to workers for accepting a job that contains a higher risk of death is estimated.[72]

In other words, it is assumed that individuals reveal the value they place on their own lives when they accept a 'riskier' job in exchange for a certain wage premium. This is based on the assumption that when a type of work has a higher probability of death (or injury), the level of wages should be, ceteris paribus, higher than for work involving less risk.[73]

The wage risk approach is based on the following two main assumptions:

- Workers are actually aware of differing risks across jobs. For the case where workers do not perceive risk accurately, the significance of the estimated coefficients on the risk variable are underestimated.

- Workers are actually in a position where they can move freely between jobs, i.e. the labour market operates freely and is in equilibrium. However, evidence is mixed, with studies reporting that union members receive greater compensation for risk than non-union workers and vice versa.[74]

If these assumptions are fulfilled, the value of life is revealed in the interaction of wage premium and increased probability of death. The following example taken from Kahn (1986) illustrates the respective procedure. If an average premium of

---

[72] See: Gilbert (1995, 1998). Concentrating on such labour market decisions is usually motivated by the availability of sufficient quality data, which is proved to be extremely difficult to obtain in most other areas.

[73] See, for instance, Miller (1999), Leigh (1991, 1995), Viscusi and Moore (1989) and Moore and Viscusi (1988, 1990).

[74] This assumption, which in short means that there is effective competition and full information given, contains several problems, such as the issue concerning how institutional constraints on labour market (e.g. union versus non-union workers) affect the estimate obtained under this assumption. See: Fisher, Chestnut and Violette (1989), pp. 89-90.

£200 is paid to workers as compensation for an increased probability of dying that year by 1/10,000, the value of life is the product of both and is £2,000,000. In other words, one person in 10,000 will die and each of these 10,000 workers is paid £200 to accept the risk. Hence the total amount people are paid to accept the given probability of one death among them is £2,000,000.[75]

Obviously, it is necessary to isolate the influence of the risk in wages from other characteristics of the respective occupations. This is usually undertaken by applying different econometric techniques, including multiple regression analysis. A model of the following simplified form is typically estimated:[76]

$$\ln w_i = \alpha + \beta_M M_i + \beta_K K_i + \beta_R R_i + \varepsilon_i \tag{13}$$

where:

$\ln w_i$:	natural logarithm of the ith individual worker's wage rate;
M_i:	characteristics of ith individual worker' s residential location;
K_i:	ith individual worker's 'personal' characteristics, such as age, experience, etc.;
R_i:	ith individual worker's job characteristics (including measurement of on-the-job risk);
β_M, β_K, β_R:	coefficients to be estimated;
α:	constant;
ε:	random error term.

75 See: Kahn (1986), p. 27.
76 See: Liu, Hammitt and Liu (1997), p. 354.

However, this method of measuring revealed preferences is subject to many criticisms, including the following major problems.[77]

- *Representativeness of persons having risky jobs*: It may be that individual's who take risky jobs are likely to be less risk averse than the population at large and may have a 'special ' preference for risky situations and may even 'enjoy' the thrill that comes with them (e.g. stuntmen).[78]

- *Risk awareness of individuals:* It is questionable whether workers are actually aware of the level of risk to which they are exposed at a specific job. There is indication that workers overestimate the risk in comparisons to the risk level they actually face, i.e. they overestimate the true probability of on-the-job death. Hence, the value of life using the actual risk workers face would be lower than the value which would be revealed by examining the workers' perceived risk.[79]

- *Existence of free labour market:* The wage-risk trade-off of workers may be constrained by limited choice, i.e. individuals usually do not have unlimited choice as to what job to accept.

- *Relevance to different age and socio-economic classes:* Wage risk studies are usually restricted to working class, wage earning adults. Transferability to different age and socio-economic classes cannot generally be assumed.[80]

---

[77] See, for instance, Fisher, Chestnut, and Violette (1989), Soby and Ball (1991), Viscusi (1990, 1992), and Zweifel and Breyer (1997).
[78] It is also debatable, if these individuals in fact feel the same way about a less spectacular risk of equal magnitude such as poisoning due to the exposure to harmful substances.
[79] See: Staller, Sullivan, and Friedman (1994).
[80] See: Fisher, Chestnut Violette (1989), p. 92-93.

➢ *Difficulties to separate the risk of death from risk of injury:* It seems very difficult or even impossible to determine the proportion of the wage premium that is subject to the increased risk of death and the proportion that is subject to the increased risk of injury.

Further problems causing potentially biased estimates of the value of a statistical life include the following points:

- All deaths are not similar.
- Wage-risk estimates include only people's evaluation of risk to their own life.
- Wage-risk estimates only give the value of risking a life in the present, but not in the future.

Finally, the approach of wage differential compensation is rather limited to the area of occupational health. As Chilton et al. (1998) state "in most areas of safety, however, and also in the case of *non*-fatal occupational injury and illness, it is extremely difficult - if not impossible - to obtain sufficient data to disentangle the various factors besides safety which may affect behaviour."[81]

Further, "a value derived from wage premiums paid for a certain array of risks to death [. . .] associated with occupational exposures cannot necessarily be considered a reliable estimate of what the individual might be willing to pay to avoid an entirely different array of injuries and death from another situation, where the benefits, alternatives and other characteristics of the activity may differ significantly."[82] However, some authors claim that despite the many limitations,

---

[81] Chitlton et al. (1998), p. 6.
[82] Soby and Ball (1991), p. 32.

wage-risk studies, when well conducted, may still be a useful method for estimating the value of a statistical life.[83]

Nevertheless, evidence from a recent study of three (US) national samples support the hypothesis that in fact inter-industry differentials, not compensating wages for risk of dying on-the-job, explain the strong positive partial correlation between wages and death rates. In other words, the positive and statistically significant compensating wage differentials shown in previous studies were actually largely caused by the correlation between inter-industry differentials and death rates. It is consequently concluded that the study results "should raise doubts in the minds of environmental economists who rely on labor market estimates of the value of a statistical life."[84]

6.3.3.2 The Method of Stated Preferences

Probably the main disadvantage of the indirect method of revealed preferences as discussed in the previous section is the fact that the basis of the actual consumer behaviour is not known.[85] In other words, it is usually not known what factors influence and finally determine the decision making process. Questions like how accurate their perceptions of risk are and if they would make the same decision if they have accurate and complete information, are further points that may weaken the reliability of the estimated values of the revealed preference methodologies.

---

[83] Soby and Ball (1991), p. 32.

[84] Leigh (1995), p. 94.

[85] Ford et al. (1995) provide a study on the discrepancy between the value of human life calculated on the basis of WTP and on compensating wage differentials.

Hence, in order to find possible answers to these questions and ultimately obtain more accurate and consistent estimates for the value of a statistical life the approach of stated or expressed preferences has been developed. This approach requires individuals to state or express explicitly the amount they are willing to pay for a specified amount of risk reduction.

Basically, there are two types of sample-survey based empirical procedures for eliciting preference-based values of safety, namely direct and indirect procedures. For the first case, direct estimates of absolute monetary values of safety are obtained. In the second procedure, it is aimed to obtain the monetary values of safety in one particular context relative to the value in another context. However, the present study will not be dealing with this latter procedure any further.[86] The methodology most commonly applied for value of statistical life estimates in recent years is the direct value elicitation procedure of the so-called contingent valuation approach, which will be described in some detail below.[87]

Essentially, there is no standard approach to the design of such contingent valuation surveys. Nevertheless, there are still various well-defined elements on which the approach is based upon.[88]

- Scenario or description of the policy or programme the respondent is being asked to value or vote. This scenario may be hypothetical (mostly) or real.

---

[86] An overview of the several different types of questions that have been employed in such a relative valuation approach can be found in Jones-Lee and Loomes (1996).

[87] The contingent valuation approach is actually the modern name for the method that originally used to be called the 'survey method'. Studying the benefits of outdoor recreation by using specially designed questionnaires, Davis (1964) was the first to use this concept See: Johansson (1993), p. 46. Extensive discussions on the contingent valuation methodology and guidance on contingent valuation survey design and analysis is provided in Hausman (1993), Mitchell and Carson (1989) and Department of Commerce (1993).

[88] See: Portney (1994), pp. 5-6 and Hanemann (1994), pp. 24-26.

- Procedures for eliciting value or choice from the respondents. Examples are *open-ended questions* ("What is the maximum amount you would be willing to pay for...?"), *closed-end questions* ("If it would cost £X, would you be willing to pay this amount?"), and *bidding games* ("Would you pay £X for...?" YES "Would you pay £2X for it?" etc.).[89]

- Gathering of information on socio-economic characteristics (e.g. sex, age, race, income, education, etc.). Possibly follow-up question to 'double-check' answers (are questions understood and taken seriously).

The term contingent valuation method is derived from the fact that the resulting values for the revealed willingness to pay or willingness to accept are contingent on the alternatives presented in the used questionnaire. By using this survey method, information on people's preferences are collected by presenting respondents with a questionnaire where they are asked to directly or indirectly place monetary values on changes in risks of premature death (or injury). In other words, in the course of a survey individuals are presented with a hypothetical market. On this market the good that is to be valued, such as a reduction in the risk of being killed in a road accident, is bought and sold. The individuals are then directly questioned about their willingness to pay to obtain this risk reduction.

The creation of such a hypothetical market is based on two assumptions. First, it is assumed that people have true preferences, which, however, are hidden. Second, it is also assumed that people are actually capable to transform these preferences into monetary units.[90] Both, the actual contents of the presented alternatives as well as the context in which they are presented are included in this

---

[89] See: Soguel (1995), pp. 4-5.
[90] See: Hoevenagel (1994), p. 195.

dependency. As "a policy evaluation tool with results invariant to important changes in these conditions would [. . .] be misleading and uninformative"[91], it is of great importance to include such a dependency. The contingent valuation approach offers several strengths that are briefly outlined below.

- The concept of contingent valuation makes a very large sample feasible. Thus, the descriptive and explanatory powers of the analysis is much enhanced.

- Since there can be a wide range of questions asked on the specific topic, the concept demonstrates a great flexibility. Furthermore, extra flexibility is given since the respective questions may be asked to the most relevant target group. In other words, the concept can be applied to the general population, but also to a sub-sample depending on the main aim of the underlying study.[92]

- The concept also assumes that the information given or data collected from all respondents are standardised.

The contingent valuation method allows one to 'tailor' very specific policy question and address them to the respective respondents for which no actual market data exists or exists only in poor quality. Obviously, the range of possible applications of this concept is rather extensive and may include such aspects as the investigation of some change in the provision of a public good as well as the environmental consequences resulting from building a new power plant or highway.

---

[91] Randall (1986), p. 145.
[92] See: Fisher; Chestnut and Violette (1989), pp. 94-95.

In comparison to wage-risk studies, the contingent valuation method offers the advantage that it provides the potential that it can be applied to policy questions which involve risks other than so-called on the job risks. Moreover, the contingent valuation method also has the ability to account for people's willingness to pay for reducing the risk to others who also may be exposed to such risk.[93]

However, the contingent valuation approach does not come without any limitations, and fundamental concerns have been raised in connection with the use of this concept. Some of the main potential problems that may occur in connection with the contingent valuation method are now described. However, the aim is to give a brief overview rather than a detailed discussion, which can be found elsewhere.[94]

- *Dealing with small probabilities:* Creating a realistic scenario requires the consideration of very small probabilities and probability differences. However, such small probabilities usually do have very little meaning to most people. Kahneman and Tversky (1979), for instance report that people may fail to distinguish between probabilities that differ by the power of up to ten.[95] Thus, respective answers are likely to contradict the axiom of transitivity of preferences. In other words, the form this approach has taken to date requires complex judgements by individuals on unfamiliar topics.

---

[93] Several contingent valuation studies found substantial altruistic values for reducing other's risk. See, for instance, Viscusi, Magat and Forrest (1988).

[94] See, for instance, Breyer and Zweifel (1996), p. 39, Soby and Ball (1991), pp. 38-39, Davis (1988), pp. 19-20, Tolley and Fabian (1998), Gregory, Lichtenstein, and Slovic (1993), Hanley (1989), Hanley and Spash (1993) and Hoevenagel (1994), pp.209-221 & 251-270.

[95] See: Kahneman and Tversky (1979).

- *Emotional rejection of asked questions:* It may be problematic to get people to answer rather delicate questions regarding the trade-off between life and death. This may result in a conscious or unconscious distortion of revealed preferences. If such questions are predominately met with a refusal to answer by individuals with a high esteem for the value of life, the results are very likely to be biased. Hence, it is important that only those zero bids are included in the calculations that actually represent zero valuations rather than being 'protest votes'. Consequently, it is vital to probe why people bid zero.[96] On the other hand, very high bids that are so large they defy rationality may also be a form of protest and have to be handled with caution and may be discarded.

- *Lack of motivation of the interviewed:* When dealing with a hypothetical situation, there is a 'danger' of a lack of motivation to think seriously about one's preferences as well as image their reaction in a real rather than hypothetical situation. Also, there may be the temptation by the interviewed person to give the 'right' socially desirable answer, i.e. to express what they think the interviewer wants to hear or what they feel will help their personal image.[97]

- *Hypothetical nature of approach:* The individual's response is based on a hypothetical rather than an actual situation. Thus, it can be criticised that an individual's response to a hypothetical situation may be different to the individual's actual behaviour in that situation.[98] This is also known as the hypothetical market bias. In this connection, it has been suggested that in order to achieve unbiased estimates it is important to construct contingent

---

[96] See, for instance, Hanley (1989), pp. 238-241, and Diamond (1996).
[97] See, for instance, Schkade and Payne (1994).
[98] See: Lanoie, Pedro, and Latour (1995), and Viscusi, Hakes, and Carlin (1997).

valuation surveys that indicate to the respondents that policy will actually be influenced by their vote. However, this also may imply that the respondents are given significantly more information and details than has typically been the case.[99] A related problem is the fact that for the outcome of the contingent valuation approach is determined by what 'people say', rather than on what they actually do.

- *Strategic behaviour of respondent:* In the case where the respondent believes that the bids are actually collected, he or she may state a willingness to pay that is below the 'true' value, since they know that, in the case of a public good, the good will be provided for all once it is provided for one person (free-rider problem). On the other hand, the true values may be overstated when it is believed that the bids will not be collected.

- *Phrasing of questions:* The answer of an interviewed individual may significantly be influenced by the way the questions asked are worded.[100] Schuman and Presser (1981) report that small changes in the wording or order of the questions may cause significant changes in the survey responses.[101] Hence, rotating question order reduces the potential for bias. Consistency checks are necessary to make sure that the values respondents attach to risk reductions do in fact not vary with the type of question asked.[102]

---

[99] See: Cummings and Taylor (1998).

[100] Römer, Pommerehne, and Feld (1998), for instance, state that they used 'non-economic language' when providing the respective information to the respondents in their contingent valuation survey.

[101] See: Mitchell and Carson (1989) and Schuman and Presser (1981).

[102] See: Miller (1997), pp. 286-289.

- *Information provided:* The estimated value may be influenced by the nature of the information is provided to respondents. It seems that individuals constantly adjust their values with the arrival of new information (e.g. the knowledge on other respondents willingness to pay).[103] Krupnick and Cropper (1992), for instance, find that individuals who have a relative with chronic lung disease have a larger willingness to pay to reduce their risk of chronic bronchitis than individuals with no such (first-hand) knowledge of the disease.[104]

- *Price of contingent valuation surveys:* In order to obtain meaningful statistical results and to be able to extrapolate the results to the population at large, it is necessary to have a large sample to work with.[105] Furthermore, the context specific questions usually require a separate survey for each sector (e.g. transport, consumer safety, etc.).

- *Dependency on socio-economic factors:* Empirical evidence shows that the size of the bids given by respondents may well be influenced by the personal income and wealth of the respondents. Other factors that may bias the estimated values include, age, gender, and health. Fischer et al. (1991), for instance, find that females and people in the 'student-age' are generally more concerned about the environment, whereas males and older people are more concerned about health and safety risk. Further, they report that the willingness to pay for future risk reductions are greater for risks that present a

---

[103] See, for instance, Hanley (1989) p. 240 and Ippolito and Ippolito (1984).
[104] See: Krupnick and Cropper (1992), pp. 37-43.
[105] Also, the WTP estimates may significantly vary with the statistical model employed. See, for instance, Kanninen (1995), Cooper and Loomie (1992, 1993) and McFadden and Leonard (1992).

direct personal threat (e.g. personal health) than for those risks that pose a more diffuse risk to the environment or to people in general.[106]

Nevertheless, despite the potential shortcomings, it is generally agreed that the attributes of this methodology indicate that the contingent valuation approach is the prime avenue for future development. For the specific case of the economic valuation of human life relevant in this study, it is claimed that the willingness to pay approach and with it the contingent valuation approach, appear to be the most effective means currently available for taking account of variations in safety in public sector allocative and legislative decisions.[107] Being based on welfare-theoretical foundations, the contingent valuation method is, in the framework of the willingness to pay approach, currently seen as the prime methodology to obtain a value for the statistical life.

Soguel (1995) asks if the contingent valuation method should be abandoned for other methods of monetisation because of the problems this method may cause. The answer he gives is a clear no. He argues that rival methods including the human capital approach are not able to offer the same possibility to capture the total costs relating to changes in health, in particular the human costs.[108]

For the case of road accidents, which are the base for the majority of contingent valuation studies, human costs cover "the money spent, to reduce the risk of loss

---

[106] See: Fischer et al. (1991). Additionally, there is also a discussion on the issue of altruism and the value of a statistical life, i.e. it may be claimed that people are not only concerned about their own health (welfare) but also about the health (welfare) of others. The question is whether such altruism components should be included in social cost-benefit analysis, or if one should concentrate on people's own willingness to pay for when using such values (e.g. changes in own safety). Recent work by Jones-Lee (1991, 1992) and Johansson (1994), suggests that one should only take full account of people's willingness to pay for the safety of others if and only if altruism is exclusively safety-focused. For the case where altruism is pure, the willingness to pay for improvements in the safety of others can be ignored, i.e. one should concentrate on the value of statistical life. See: Johannesson, Johannson and O'Connor (1996).

[107] Jones-Lee (1989), p. 36.

[108] See: Soguel (1995), pp. 12-13.

of life expectancy, physical and mental; suffering of the victim, diminution of quality of life, permanent cosmetic damage, and the mental suffering of the friends and relatives of the victims."[109] This is in ln line with Soby and Ball (1991) who claim that the contingent valuation method "remains the prime avenue for future development."[110] Summing up, Schwab-Christe (1995) states:

> Finally, it must be concluded that while the contingent valuation method offers many potential advantages, it also involves numerous deficiencies, like any other method. From this viewpoint, our disappointment with the results stems less from their imperfection than from the perhaps excessively high expectations we had placed on the application of the method.[111]

The following table presents a summary of main publications on the valuation of benefits of health and safety control resulting in an estimate for the value of a statistical life. The summary is adapted from a research report funded by the Health and Safety Executive (HSE).[112] The presented estimates are taken from selected publications on mainly contingent valuation including other direct preference-elicitation methods in the fields of health and safety.

The first figure represents the estimate given in the original study adjusted only for inflation. The figures in parentheses, however, represent the same estimates including further updating for increase in real income per capita, where it is assumed that the income elasticities of the value of statistical life is 0.3 and 1.0 respectively. Also, since in some studies a variety of different estimates for the value of statistical life have been given, such as those for different age groups or gender, a single 'representative' figure is given. This has been done by either taking the particular author's 'preferred' estimate or by averaging.

---

[109] Alfaro, Chapuis, and Fabre (1994), pp. 10-11. Here, the loss of life expectancy is associated with the loss of well-being of the victim due to the loss of consumption and of leisure time.
[110] Soby and Ball (1991), p. 38.
[111] Schwab-Christe (1995), p. 41.
[112] See: Chilton et al. (1998).

However, the authors of the underlying study from which most of the above estimates are taken, emphasise that in some cases estimates have been produced which differ by more than an order of magnitude. In these cases, the complete estimates are given, such as, for instance, Frankel (1979) and Desaigues and Rabl (1995). Additionally, often the estimates for the median responses are given, because due to some extreme responses at the high end, the mean values are pushed upward and no 'trimming' from the sample has been done.

Table 44: Review of Results on the Value of Human Life.

Study	Context	Location	Risk Reduction	Value of Statistical Life (£1994) Sterling	
				Based on Means	Based on Medians
Action (1973)	Heart attack	U.S.A.	2×10^{-3} p.a.	69,800 (80,500) (112,000)	N/A
Melinek, Wooley & Baldwin (1973)	Domestic fire	U.K.	1×10^{-5} p.a.	365,000 (423,000) (593,000)	132,900 (153,900) (215,800)
Jones-Lee (1976)	Aviation	U.K.	$X \times 2 \times 10^{-6}$ per flight	10,100,000 (11,400,000) (15,100,000)	8,590,000 (9,700,000) (12,800,000)
Maclean (1979)	Domestic fire	U.K.	N/A	3,560,000 (3,945,000) (5,007,000)	N/A
Frankel (1979)	Aviation	U.S.A.	1.5×10^{-6} per flight	19,000,000 (20,500,000) (24,500,000)	4,800,000 (5,180,000) (6,180,000)
Frankel (1979)	Aviation	U.S.A.	1×10^{-3} per flight	N/A	79,800 (86,100) (102,800)
Jones-Lee, Hammerton & Philips (1988)	Roads	U.K.	$X \times 10^{-5}$ p.a.	2,660,000 (2,890,000) (3,490,000)	1,330,000 (1,440,000) (1,750,0000
Gerking, De Haan & Schulze (1988)	Workplace	U.S.A.	2.5×10^{-4} p.a.	2,480,000 (2,610,000) (2,940,000)	N/A
Maier Gerking & Weiss (1989)	Roads	Austria	$X \times 10^{-5}$ p.a.	2,830,000 (2,930,000) (3,190,000)	N/A
Persson et al. (1991)	Roads	Sweden	$X \times 10^{-4}$ p.a.	2,120,000 (2,190,000) (2,350,000)	800,000 (825,000) (887,000)
Viscusi & Cedervall (1991)	Roads	U.S.A.	$X \times 10^{-5}$ p.a.	6,060,000 (6,180,000) (6,480,000)	1,690,000 (1,720,000) (1,810,0000

Miller & Guria (1991)	Roads	New Zealand	$X \times 10^{-4}$ p.a.	800,000 (808,000) (828,000)	N/A
Kidholm (1995)	Roads	Denmark	$X \times 10^{-4}$ p.a.	7,400,000 (7,440,000) (7,550,000)	2,020,000 (2,032,000) (2,060,000)
Schwab-Christe (1995)	Roads	Switzer-land	$X \times 6.25 \times 10^{-6}$ p.a.	10,280,000 (10,280,000) (10,280,000)	4,770,000 (4,770,000) (4,770,000)
Lancoie, Petro & Latour (1995)	Roads	Canada	$X \times 10^{-4}$ p.a.	980,000 (1,013,000) (1,095,000)	N/A
Lancoie et al. (1995)	Workplace	Canada	N/A	14,700,000 (15,200,000) (16,400,000)	N/A
Desaigues & Rabl (1995)	Roads	France	50 deaths p.a.	15,300,000 (15,300,000) (15,300,000)	5,060,000 (5,060,000) (5,060,000)
Desaigues & Rabl (1995)	Roads	France	500 deaths p.a.	3,590,000 (3,590,000) (3,590,000)	1,010,000 (1,010,000) (1,010,000)
Desaigues & Rabl (1995)	Roads	France	5000 deaths p.a.	660,000 (660,000) (660,000)	200,000 (200,000) (200,000)
Persson, Lugner Lurinder & Svensson (1995)	Roads	Sweden	4×10^{-5} p.a.	2,950,000 (2,960,000) (2,990,000)	1,090,000 (1,094,000) (1,100,000)
Jones-Lee, Loomes & Philips (1995)	Roads	U.K.	4×10^{-5} p.a.	4,560,000 (4,680,000) (4,880,000)	2,700,000 (2,770,000) (2,890,000)
Covey, Jones-Lee, Loomes & Robinson (1995)	Foods	U.K.	1.67×10^{-7} p.a.	48,840,000 (48,840,000) (48,840,000)	0-25,740,000 (0-25,740,000) (0-25,740,000)

Source: Chilton et al. (1998).

Clearly, the values presented in the above table reflect the rather large discrepancy between the various empirical results. Obviously, the study design has a large determination of the final values. Some of the early studies certainly were largely of exploratory nature. The Acton study (1973) and the Jones-Lee study (1976), for instance, were both rather limited with a sample size of thirty-six and thirty, respectively. Furthermore, it is often not explicitly specified what the reported VOSL in fact comprised, i.e. if the willingness to pay 'just' covers the human costs, or, what is usually assumed, whether other cost-elements are also included, such as the lost productivity capacity.

Concentrating on those estimates derived from road transport sector the following table illustrates estimates for the VOSL based on both means and medians.

However, before computing these values two putative 'outliers' were excluded. First, the study by Schwab-Christe (1995) was excluded based on the fact that this study is somewhat fundamentally different to the other studies. This study especially tries to include not only one's own risk but also mainly focused on the willingness to pay to reduce the risk of grief and bereavement a person would experience in the event of a relative's death. When including this certainly important aspect, the resulting VOSL is systematically higher than those from other studies (up to 100%). Since this study is more or less the only of its kind so far, the results cannot be compared with other studies.[113]

The second study that was not considered is that provided by Desaigues and Rabl (1995). This study actually provides several VOSL estimates depending on the number of lives saved. The mean based estimates provided differ by some

[113] A similar study is provided by Schwab-Christe and Soguel (1996).

significant magnitude ranging from £15,310,000 for 50 lives saved to £660,000 for 500 lives saved (23-times difference) and the median based values range between £5,060,000 and £200,000 (25-times difference). Hence, it seems impossible to compute a single representative value. Also, the underlying contingent scenario of this study is very different to most other contingent valuation studies.[114]

Table 45: Summary of Presented Value of Human Life Estimates.

Average VOSL based on means (11 studies)	Average VOSL based on medians (8 studies)
£3,146,364 (£3,245,091) (£3,501,818)	**£1,355,000** (£1,386,375) (£1,463,375)

Note: The values in brackets represent the adjustments made for an increase in real income per capita as described previously.

Depending on whether the estimates are based on means or medians, the value of preventing a statistical fatality may either be around between £3m to £3.5m or around between £1.3m to £1.5m. These values are in line with those applied in previous studies.[115] Also, in 1992 the Royal Society recommended to apply a VOSL of £2m to £3m.[116]

The Department of the Environment, Transport and the Regions (DETR) also provide estimates for prevention of road causalities. These values are updated every year in order to reflect inflation and the assumption that willingness to pay rises with real income.

---

[114] Basically, in contrast to most contingent valuation studies, any reference to probabilistic risk was excluded in the survey. Instead, the household's annual willingness to pay for various numbers of lives saved (ranging from 50 to 5000) was asked.

[115] See, for instance, Maddison et al. (1996).

[116] See: Royal Society (1992).

The latest published average value of a statistical life is £902,500 for 1997 at 1997 prices.[117] Applying the suggested adjustment factor of 1.0415 to obtain 1998 values leads to a value of a statistical life in 1998 prices of around £940,000. This value, however, does not represent actual costs incurred as a result of road accidents. Rather, it is the cost-benefit value and represents the benefits that would be obtained by prevention of road accidents. Although the empirical evidence does give a wide range of values, as described above, recent work in the UK has actually endorsed the current DETR values for road accidents.[118]

In line with the more conservative approach applied throughout this study, it has been decided to use the value provided by the DETR, though it seems considerably smaller than some of the values provided in the empirical literature. Consequently, after adjusting for inflation and economic growth, a value for a statistical life of £940,000 in 1998 prices will serve as a baseline value for further calculations made in this present study.

6.4 Value of a Statistical Life for the Air Pollution-Related Mortality

As described in the previous sections, there is a large body of social science research on the valuation of benefits of health and safety control. In these studies the estimated values of a statistical life is predominantly based on the prevention of a statistical fatality caused by road accidents or accidents at the workplace.

---

[117] See: Highways Economic Note No.1 1997 on DETR website at: http://www.roads.detr.gov.uk/roadsafety/hen197/I.htm. It is to be noted that this value of prevention of a statistical fatality is smaller than the prevention of a fatal accident. This is due to the fact that a fatal average involves more than one fatality, i.e. for example in 1997, a fatal accident involved 1.1 fatalities, 0.43 serious casualties and 0.58 slight casualties; the average value of prevention per fatal accident is £1,085,670 in 1998 prices.

[118] See, for instance, Beattie et al. (1998) and Carthy et al. (1998).

However, so far no suitable study has been undertaken that has investigated people's willingness to pay for reductions in air pollution mortality risk. It is, however, very questionable whether it is legitimate to apply the current empirical results to the air pollution context without any adjustment. In a recent report by the Ad-Hoc Group on the Economic Appraisal of Health Effects of Air Pollution (hereafter AHG), which was set up by the Department of Health, it is argued that the nature of the risks and the characteristics are too different in order to be transferred from the road sector to the case of air pollution. Being closely based on the study carried out by the AHG mentioned above, the following discussion will take this approach into account.

Thus, it seems absolute necessary to adjust the VPF as provided by the DETR to the specific context of air pollution, rather than simply applying the values, which were exclusively derived from other contexts, mainly road safety. Following the AHG, "it is highly likely that WTP-based valuations of safety measures will be affected by a range of factors relevant to the air pollution context."[119] The following table is taken from the AHG study and summarises the most important of such potential factors:

[119] Department of Health (1999), p. 63.

Table 46: Factors that may Influence the WTP for Avoiding a Particular Risk.

Factor	Impact
• Type of health effect (acute; chronic; latent)	e.g., people may dread a lingering death more than a sudden death.
• Factors related to risk context (e.g. voluntariness; control; responsibility; uncertainty; etc.)	e.g., people seem to regard involuntary risks over which they have no control, risks which are someone else's responsibility, and vague risks, as worse than others.
• Futurity of health effect and discounting	e.g., effects which happen sooner are expected to be regarded as worse than those which happen later.
• Age	e.g., people may attach particular value to life and health at certain ages.
• Remaining life expectancy	e.g., WTP is expected to be positively related to the number of years of life expectancy at risk (although not necessarily in direct proportion).
• Attitudes to risk	e.g., risk aversion is expected to affect willingness to trade wealth for risk; younger people may be less averse to risk.
• State of health-related quality of life	e.g., people are expected to be keener to extend life in good health than life in poor health.
• Level of exposure to risk	e.g., people may be keener to reduce a high risk by a set amount than a low risk by the same amount.
• Wealth/income/socio-economic status	e.g., people with more wealth are likely to have a higher WTP to reduce a given risk than those with less, and there may be other differences between social groups.

Source: Department of Health (1999), p. 63 & Rowe et al. (1995), pp. 427-440.

Whereas the first three of the presented factors are related to the nature of the risk itself, the remaining factors are related to the people that are exposed to the risk. Generally, it has been shown that various factors play an important role in the willingness to pay for a reduction in risk, including, voluntariness, control, responsibility, and direct personal benefit while taking the risk.[120] Several studies show that the acceptance is higher for voluntary risk and risks relying to a large extent on someone's own responsibility and control than for involuntary

[120] See: Jones-Lee and Loomes (1995).

risks that are beyond someone's control.[121] This is important when looking at the air pollution related mortality risk, since risks of road traffic accidents are usually used as a basis. People generally perceive risks of road accidents as largely voluntary, well-understood and easy to control. In contrast, air pollution risks are often perceived as involuntary, poorly understood and not controllable.[122] Therefore, it is assumed that it is most likely that the aversion against risks related to air pollution are considerably higher than the aversion against the risk of fatal road accidents.[123] According to these empirical results it is suggested that the willingness to pay for reductions in such involuntary risks may be twice to three times higher than that of voluntary risks.[124]

Applying the 'air pollution' factor, as derived in the previous section, to the value of (preventing) a statistical fatality for road accident deaths of £940,000 as provided by the DETR, the value of a statistical life for an involuntary risk from air pollution would be around £2.35 million.[125] This value is seen as the air pollution baseline VPF and will serve as a base for further possible adjustments as described in the following section.

---

[121] Additional, so-called cognitive dimensions by which people asses risk include whether death is immediate or delayed, and whether it is a dread risk or one can be thought of in a calm and reasonable way. For a discussion on the effects of psychological perceptions on the willingness to pay to reduce risk see, for instance, Savage (1993) and McDaniels et al. (1992). Jones-Lee and Loomes (1994), for instance, found that safety on the London Underground commanded a premium in terms of 51 per cent.

[122] Additionally, incurring the risk associated with air pollution is less often combined with a direct personal benefit, which also may indicate a lower risk acceptance inducing in general a higher willingness to pay. One study has suggested that the difference between in willingness to pay associated with voluntary and involuntary risk could be as much as 100 times. See: Litai (1980) as referenced in European Commission (1995a).

[123] The reasons for this are undoubtedly related with perceived natural rights and freedom of choice. See: European Commission (1995a), pp. 439-440.

[124] The exact size of this aversion factor is, however, not determined definitely at the moment. See: Jones-Lee and Loomes (1995).

[125] Empirical evidence shows that the average value of preventing the loss of a statistical life for people of average age (ca. 40) and average health is also around £2 million.

6.4.1 Adjusting the Value of Preventing a Statistical Fatality

The air pollution baseline VPF derived in the previous section has the implicit assumption of an average person, i.e. a person of average age, average health, average UK wealth, etc. However, as already mentioned before, this assumption is most likely not fulfilled in the case of air pollution where the people most at risk are older than average, experiencing poor health and a short life expectancy. Consequently, it seems necessary to make further adjustments to the air pollution VPF taking the factors as described in Table 46 into account. The AHG suggests the following adjustments.

a) *The impact of age to risk:* Air pollution predominantly affects people in the age group over 65. The willingness to pay in this age group is seen as about 70% of that at the average age of about 40.

However, it is currently not possible, based on epidemiological evidence, to make direct conclusions about the age structure of the air pollution related premature deaths. There is, however, theoretical as well as empirical evidence that the willingness to pay for risk reduction of mortality is in fact decreasing with increasing age as well as with a reduced remaining life expectancy and with reduced quality of life. It can be shown, that depending on specific assumptions regarding lending and borrowing opportunities, that the VPF will either continuously decline with age or will show an inverted U-shape.[126]

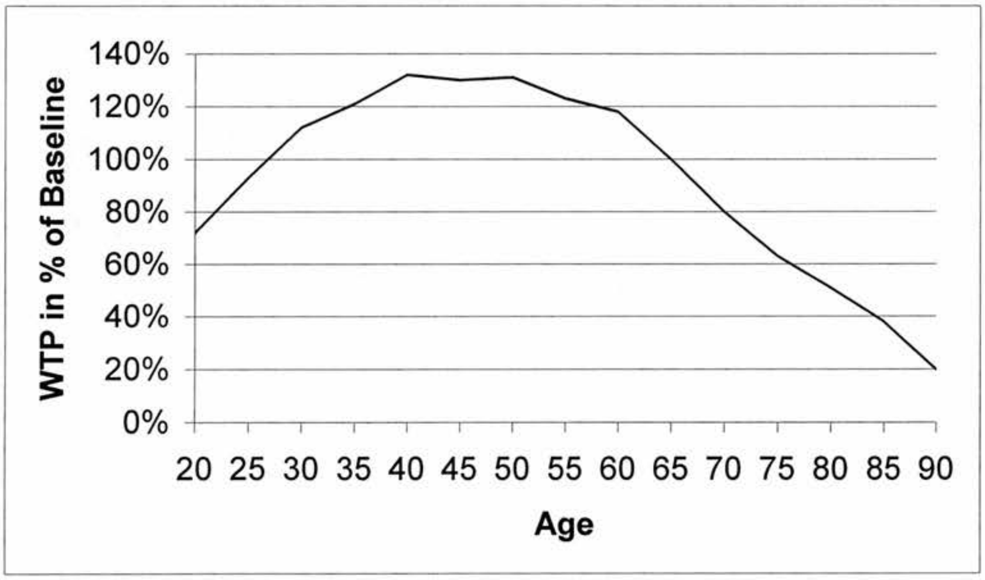
Theoretical predictions by Shephard and Zeckhauser (1982, 1984) could be confirmed in a number of empirical studies. These studies found that the

[126] Assuming restricted lending and borrowing opportunities the VPF will rise as earnings (and therefore the ability to pay) increases in the younger ages, peak at middle age and then declines. No constraints and hence a constant consumption over the life cycle, will lead to a monotonically declining VPF.

willingness to pay and hence the implied value of a statistical life increase with age, reaching a peak at the age of around 40, and then decline with age. Cropper, Aydede and Portney (1994) found that saving eleven 60-year olds was judged equivalent to saving one 30-year old. Johannesson, Johansson, and Löfgren (1997), in comparison, find a much smaller difference between the value of saving young and old lives where saving 1.5 individuals aged 70 is equivalent to saving one 30-year old.[127]

The following graph illustrates the relationship between age and the willingness to pay indicating a reversed U-shaped relationship as described above.[128]

Figure 31: Relationship Between Age and WTP for a Mortality Risk Reduction.



Source: Department of Health (1999), p. 67.

This inverted U life cycle for the value of preventing fatality may be explained by the fact that it reflects the impact of two opposing effects.

---

[127] See: Cropper, Aydede and Portney (1994), Johannesson and Johansson (1997b) and Johannesson, Johansson and Löfgren (1997).

[128] See, Jones-Lee, Hammerton and Philips (1985), Jones-Lee (1989), Cropper, Aydede and Portney (1994) and Johannesson, Johansson and Löfgren (1997).

First, life expectancy decreases with age, which in turn tends to depress the willingness to pay for safety, and second, the tendency of an increased aversion with age, which in turn increases willingness to pay. The Ad-Hoc Group provides the following adjustment factors for different ages, starting with age 65 being the baseline.

Table 47: Adjustment Factors for the Combined Effects of Age and Increasing Aversion to Physical Risk.

Age	Multiplier
65	100% of the baseline VPF
70	80% of the baseline VPF
75	65% of the baseline VPF
80	50% of the baseline VPF
85	35% of the baseline VPF

Source: Department of Health (1999), p. 67.

b) *The impact of life expectancy*: Those who are most at risk of dying from air pollution belong predominantly to the group of over 65s, and hence may already have a much shorter life expectancy than that for an 'average person' because of reasons other than air pollution. Their exact life expectancy, however, is not known with certainty, but may be assumed to be in the range of one month (or less) and one year. Obviously, life expectancy is not the only factor that may influence the individual's willingness to pay, but it is assumed that willingness to pay does not drop more than in proportion to life expectancy. Hence, for a reduction of life expectancy from 12 years to one year, the respective willingness to pay would be reduced by one twelfth, which is equivalent of a factor of 6% of the baseline. A further reduction to one month would lead to a reduction of the willingness to pay by a further factor of twelve, which is equivalent of a factor of around 0.5% of the baseline.[129]

c) *The impact of quality of life*: The discussion on the factors determining adjustments to the VPF discussed so far was based on the assumption that the individual concerned is in a state of health that is not significantly different than the average person in his/her age group. However, as known from the epidemiological literature, those most at risk to die from air pollution are predominantly elderly or, and, are already in a health status that is lower than the average one in their age group. So it may be assumed that they enjoy a lower quality of life than the average elderly population. Since people generally value further time in full health more highly than further time in poor health, the respective willingness to pay is expected to be reduced further. The quality of life is also being reduced from an average factor for their age of 0.76 down to 0.2 to 0.7, the quality of life factors for patients with severe COPD.[130] The Ad-Hoc Group suggests a reduction in willingness to pay in proportion to quality of life, which is analogue to a reduction by 0.2/0.76 or 0.7/0.76 for a quality of life of 0.2 and 0.7 respectively.[131]

In addition to the adjustments discussed above, there are a number of further potential factors that may have some influence on people's willingness to pay for avoiding the risk of dying from air pollution. These factors have already been mentioned in a section above. However, the Ad-Hoc Group came to the

---

[129] For a more detailed discussion on this see: Department of Health (1999), p. 68.

[130] COPD = chronic obstructive pulmonary disease.

[131] A recent study on the behalf of the WHO, however, does not make such adjustments for reduced life expectancy and for lower quality of life. However, this study also concentrated on the evaluation of long term effects where it may be appropriate that the costs of air pollution should not be adopted. The current empirical evidence is not yet sufficient to be in a position to clearly decide which approach should be adopted.

conclusion that no further adjustments to the baseline VPF, other than the ones discussed are required. This has been decided because there is currently not enough information on the exact impact (e.g. the impact of level of exposure to risk), or possible adjustments are not seen as politically acceptable or ethically appropriate (e.g. adjustments for wealth, income and socio-economic status). Adjustments for futurity and type of effect are not necessary on the background of this study, since it is only concerned with acute effects of air pollution (short and very short term effects).

In the case of chronic or latent effects, further adjustments to the VPF would be necessary since the effected people who die from air pollution would do so at some considerable time after they have been exposed to air pollution the first time. Hence, a further discounting would be necessary.

6.4.2 Estimate for Air Pollution Adjusted Value of a Statistical Life

As described above, when trying to produce an estimate for VPP associated with air pollution, several adjustments appear to be necessary in order to obtain reasonable results. Consequently, following the AHG study, in order to obtain an estimate for an air pollution VPF, adjustments for age, impaired health status (reduced life expectancy and reduced quality of life) are made. Taking the proposed adjustments into account and applying them to the baseline value of £2.35 million, values of preventing a statistical fatality related to air pollution can be derived as follows.

Figure 32: Calculation of Air Pollution Related Value of a Statistical Life.

Baseline

£2.35 million

⇓

Affected group are elderly (over 65s)

downwards adjustment by 70%

⇓

£2.35m x 70% = £1.645m

⇓

Adjustment for lower life expectancy (1 month to 1 year) than average for elderly population (12 years)†

⇓	⇓	⇓
Upper bound No adjustment	1 year (= divide by 12)	Lower bound (1 month) (= divide twice by 12)
⇓	⇓	⇓
£1.645m	**£137,000**	**£11,400**
⇓	⇓	⇓

Lower quality of life (0.2 to 0.7) than average for elderly population (0.76) (proportional downward adjustment)

⇓	⇓	⇓
Upper bound No adjustment	1 year (= multiply by 0.7/0.76)	Lower bound (1 month) (= multiply by 0.2/0.76)
⇓	⇓	⇓
£1.645m	**£126,000**	**£3,000**

Source: Department of Health (1999), p. 77.

†: The lower life expectancy may possibly be at least partly caused by the continuous exposure air pollution over a longer time period (chronic effects).

It is important to note that the adjustments made do in fact have a large potential impact on the final estimated value, although is it debatable that the relatively high baseline level of risk on willingness to pay of £2.35 million might actually lessen the strength of the impacts of these adjustments.

Consequently, it seems easily justifiable to suggest the relatively wide range of estimates ranging from £1.645 million, where no adjustments for impaired health are made down to £3,000 for a loss of life expectancy of 1-month adjusting for a quality of life of 0.2. Notably, there is a substantially low value of £3,000 at the lower bound in comparison with the values provided for road accidents (£940,000). However, the AHG Group believes that this is reasonable since there are smaller losses in quality of life and life expectancy involved.

Nevertheless, an important note on the estimated VPF is that the adjustments suggested by the AHG are predominantly related to deaths due to respiratory causes. However, in the case of the adverse impact of air pollution it is known that the effects on respiratory deaths are much greater than on cardiovascular deaths. Thus "[i]t seems unlikely that the deaths brought forward by air pollution are dominated by cardiovascular deaths."[132] However, "if adjustments are made for the reduced life expectancy and quality of life expected in those with respiratory disease, the resultant value is likely to be an underestimate because some of the deaths will be cardiovascular deaths."[133]

Finally, it is important to remark that there are not only arguments in favour of these adjustments but also against them. From an ethical point of view it may be problematic to implement such a valuation system, which varies by age and

[132] Department of Health (1999), p. 78.

[133] Ibid, pp. 78-79. However, since this study is in fact concentrating on respiratory and cardiovascular deaths only, rather than on all-cause deaths, the size of the potential underestimation is assumed to be less severe.

consequently implicitly favours the saving of younger peoples' lives compared to older peoples' lives, although there is also empirical evidence of a decline in willingness to pay in the elderly.[134]

Also, one could argue that the age related adjustment actually causes a considerable reduction in health costs. Consequently, it seems appropriate to apply an approach that allows further calculations according to different scenarios, which finally lead to a range of estimates rather than one single value.

Generally, the AHG study, on which this discussion is heavily based, stressed that the main aim is to find a value for potential benefits of reducing air pollution from the perspective of those who benefit from the reduction. They state that "in other words we try to predict the values that those affected would themselves use, if they were making the relevant choices and trade-offs, to reflect their strengths of preferences." Further it is claimed that "people's preferences for reducing the risks that could shorten their life will be stronger if they would have the chance of living, say, an extra year in good health than if they only stood a chance of living an extra year in poor health."[135]

6.5 Summary

As discussed in detail in this chapter, the subject of valuing a human life in monetary terms certainly raises a number of problematic questions. First of all the issue of potential ethical objections against the valuing of human life as well

---

[134] There are, however, much more uncertainties about the other previously discussed adjustments.

[135] Department of Health (1999), p. 66. Furthermore, as discussed before, one can also include the 'effects' on close relatives, e.g. children, into the WTP estimate, which in fact would in fact (partly) question the above described adjustments.

as a finite value for a human life were discussed in this chapter, concluding that these objections indeed can be rejected. It is argued that when taking an ex ante approach and looking at the issue of avoiding small risks, life is not infinitely valuable to people and consequently it not only seems legitimate to weigh (statistical) life against money but also to assume a finite value of life. The emphasis certainly lies on the fact that an individual and identified human life is not the focus of this matter but rather a statistical one.

The second part of this chapter introduced and discussed in some detail various approaches to value human lives and their respective advantages and limitations. The two main approaches that have been applied in the literature are the human capital approach and the willingness to pay approach. However, the former approach is now considered as not appropriate, reducing a person's life solely to his or her economic contribution to society by future income, neglecting any other non-monetary contribution and reducing the value of those not currently in the work force to zero. Following Carthy et al. (1998):

> There is now fairly widespread agreement amongst economists that monetary values of safety for use in public sector cost-benefit analysis should be defined in such a way as to reflect the preferences of members of the affected population in as expressed their collective willingness to pay (WTP) for safety improvement or their collective willingness to accept compensation (WTA) for increased risk.[136]

In other words, when tackling the problem of putting a monetary value on human lives, it is believed that people's preferences for reducing the risks that could shorten their life are the main factor determining this value. Therefore, this study only concentrates on the welfare economics based willingness to pay approach. As the majority of the empirical analysis applying the willingness to pay

[136] Carthy et al. (1998), pp. 187-188.

approach concentrate on road traffic fatalities, it has been suggested that various adjustments are necessary for the case of air pollution.

Using a baseline value of £2.35m, further adjustments have been made, accounting for various important factors that are specific for the case of air pollution as analysed in this study. This procedure includes adjusting for age, life expectancy, and the quality of life, which has been undertaken in order to account for the fact that those who are actually dying due to the exposure of air pollution belong to the age group of 60 and above, have a reduced life expectancy and possibly are already suffering from a reduced quality of life due to existing ill-health. Also, empirical research produced evidence of an inverted-U shaped life cycle for the value of a statistical life, as it has been predicted by theoretical work of Shepard and Zeckhauser (1982).[137]

However, as previously discussed, there are still various uncertainties on this matter. Consequently, rather than producing one single value, it seems much more justifiable to produce a range of estimates including a lower more conservative value, a central value, and a high value with no adjustments in order to account for the range of different circumstances the population at risk may be confronted with. Hence, this study also produced a range of values illustrated in the following table.

---

[137] See: Carthy et al. (1998), Beattie et al. (1998), Jones-Lee, Hammerton and Phillips (1985) and Shepard and Zeckhauser (1982).

Table 48: Summary of Air Pollution Related VOSL.

Lower bound	Central value	Upper bound
£3,000	£126,000	£1.645m

These values will serve as a basis for further calculations being undertaken in a later chapter, when calculating the value of the traffic-related premature deaths due to air pollution.

7. Literature Review of Time-Series Mortality Studies

7.1 Introduction

As mentioned in previous chapters of this study, the exposure to air pollution can result in adverse effects to human health, ranging from minor irritations of the eyes and restrictions of the respiratory system, to premature deaths due to cardio-respiratory failure. The focus of this study is on the analysis of the latter, trying to establish a statistically significant relationship between the exposure of various air pollutants and premature death due to respiratory and cardiovascular disease. Before discussing and applying a statistical model to analyse respective data for 14 British cities, a brief review of the results of similar studies previously undertaken is provided.

Over a long time period, people have been concerned about air pollution and human health. Early studies were mainly concerned with the analysis of air pollution disasters. These disasters often occurred during certain meteorological conditions where atmospheric dispersion was severely limited, combined with fog, cool or cold weather, which in turn lead to a significant increase of ground-level air pollution concentrations of up to 10 times. Examples of such high air pollution episodes took place in the Meuse Valley, Belgium, December 1930; London, December 1952 and After; New York City, 1950s and 1960s; Paris, December 1972-January 1973; Central and Western Europe, January 1985. All these episodes resulted in an increase in mortality rates in the respective areas. A more detailed description of these incidences is given elsewhere.[1] Most notably in the last two decades or so, the number of studies investigating the relationship between mortality and various morbidity outcomes and air pollution has

[1] See: Lipfert (1993), pp. 111-141 and Ministry of Health (1954).

significantly increased. This is largely due to the vastly improved air pollution measurement procedures, and particularly the accessibility of electronically archived data.

The early studies, noted above, analysing extreme air pollution episodes did not need to apply very 'sophisticated' research methods, since the recorded levels of air pollution were so high, effects were evident even with the crudest study design. However, various control measures have led to much lower air pollution levels, and for sometime, these levels were assumed to be not harmful to human health.

However, results from recent studies published from the early 1980s suggest that these lower levels of air pollution, which have previously been considered as safe, may in fact have serious adverse health effects. The health effects of current air pollution levels are usually subtle, i.e. associations are weak, and can only be observed in large populations, for which consideration of individual cases is clearly impractical. Since, those individuals most at risk from an increased level of air pollution cannot be identified a priori, a large number of people have to be monitored in order to determine the individual exposures of the decedent. As Slzklo (1987) emphasises, there are fundamental problems of extrapolation from experimental panel studies to large populations; thus he warns that fundamentally weak relationships are difficult to detect in all kinds of study design.[2]

Generally, epidemiological study design may be classified according to different criteria, for instance: the conditions of observation (experimental vs.

---

[2] See: Szklo (1987).

observational); unit of data collection and analysis (group vs. individual); and time dimension (cross-sectional vs. longitudinal or time-series).

This study is solely concerned with so-called observational epidemiological studies. In such studies existing information on health effects is matched retrospectively with corresponding environmental data, i.e. routinely collected air pollution data. Such epidemiological studies differ from clinical medicine or biomedical research by virtue of its study of population rather than individual cases or specimens. Often, this emphasis stems from the fundamental objective of epidemiology to improve public health. Following Katsouyanni (1997), "[a]ir pollution epidemiology is the study of the occurrence and distribution of health outcomes in association with community exposure to air pollution."[3]

Further, regarding the time dimension of the study design, only the results of longitudinal or time-series studies are considered in this study and hence summarised below. Such temporal studies using aggregated data allow investigating the relationship between an exposure (air pollutant) and an outcome variable (mortality), where each variable is measured over the same time unit (e.g. days, months) during a specified time period. Thus, the measurements of each variable represent a time-series. The ecological hypothesis in time-series studies is that the same set of air monitoring locations faithfully represents the actual exposure of different subgroups, for all intervals throughout the period studied. This is important, since for each day, or month, a different subgroup is likely to respond (i.e. die, be admitted to hospital, etc.).[4]

This study design has been extensively used to analyse and assess acute or short-term air pollution effects on human health, i.e. such effects that do occur soon

---

[3] Katsouyanni (1997), p. 51.
[4] See, for instance, Ayres (1997).

after exposure, basically within a few days. In contrast, in order to analyse long-term or chronic effects, so-called cross-sectional studies are used.[5] Typically, in such a study design, the air pollution exposure and health outcome in a sample of individuals is studied. However, due to the lack of individual exposure assessment and other potential confounding, such studies are relatively rare in air pollution epidemiology, and will not be discussed any further in this study.

Longitudinal or time-series studies are attractive for various reasons. First, such studies are relatively cheap as they usually employ data that is routinely collected. Second, this design eliminates various potential confounding effects compared with cross-sectional studies, because the population studied serves as the exposed population as well as their own control population.[6] Thus, the possible confounding effects are reduced to those variables that vary with the time units of aggregation (e.g. days), such as meteorological or chronological variables. This problem is discussed in detail in the following chapter.

As previously mentioned, there is an increasing number of empirical studies analysing the relationship between air pollution and acute health effects. These studies have analysed the main criteria air pollutants, employing various methodological approaches to control for confounding effects. The following sections provide a selection of previous time-series mortality studies summarised in tabular form. However, no claim of completeness is made, and detailed reviews have been published elsewhere.[7]

---

[5] See, for instance, Lipfert (1980), Lipfert (1984), Evans, Tosteson and Kinney (1994). A review of further cross-section studies is given in Lipfert (1993).

[6] See: Schwartz et al. (1991), p. 1.

[7] See, for instance, Lipfert (1993), Department of Health (1998), Ostro (1994), European Commission (1995a, 1995b), Pope, Dockery and Schwartz (1995) and Dockery and Pope (1996).

The studies are classified according to author of study, location and time period studied, the relative risk factor (RR) as well as the percentage change of the outcome variable examined. The procedure to calculate the relative risk factor will be presented in detail in Chapter 10 of this study. In short, a relative risk factor of, for example, 1.05 per 100 $\mu g/m^3$ implies that the mortality rate increases by 5 per cent each time the concentration of the respective air pollutant increases by 100. Additionally, the respective confidence intervals (CI) for one standard error are presented in brackets.[8] Such tables are produced for each of the criteria pollutants analysed. Further, since empirical studies analysed total mortality (excluding external effects), but also the cause specific sub-groups of cardiovascular and respiratory mortality, the results are presented in separate tables respectively.

7.2 Summary of Selected Time-series Studies for Particular Matter

➢ Total Mortality

Table 49: Percentage Change in Total Mortality associated with 100$\mu g/m^3$ increase in Particulate Matter.

Reference	**Location**	**Time Period**	**RR per 100$\mu g/m^3$** (95% CI)	**Per Cent Change per 100$\mu g/m^3$** (95% CI)
Fairly (1990)	Santa Clara (c)	1980-82; 1984-86	1.06 (1.02, 1.09)	6.0 (2.0, 9.0)
Schwartz & Dockery (1992b)	Philadelphia (b)	1973-1980	1.13 (1.10 1.15)	13.0 (10.0, 15.1)
Pope et al. (1992)	Provo, Utah Valley	1985-89	1.15 (1.09, 1.23)	15.0 (9.0, 21.0)
Schwartz (1993a)	Birmingham, AL	1985-88	1.11 (1.02, 1.20)	10.0 (2.0, 19.0)
Schwartz (1994a)	Cincinnati (b)	1977-82	1.1 (1.05, 1.17)	10.0 (5.0, 17.0)
Dockery et al. (1992)	St. Louis	1985-86	1.16 (1.01, 1.34)	16.0 (1.0, 29.0)

[8] For the 95% significance level, the confidence interval is computed as 1.96 x standard error; for the 90% significance level, the confidence interval is computed as 1.64 x standard error.

Dockery et al. (1992)	Kingston, TN	1985-86	1.16 (0.88 1.57)	16.0 (-13.0, 46.0)
Schwartz (1994b)	Minneapolis (b)	1973-1982	1.09 (1.04 1.15)	9.0 (4.0, 15.0)
Schwartz (1991)	Detroit (b)	1973-1982	1.12 (1.05 1.16)	10.0 (5.0, 16.0)
Schwartz & Dockery (1992b)	Steubenville (b)	1974-1984	1.08 (1.04 1.10)	8.0 4.0, 10.0)
Kinney et al. (1995)	Los Angeles	N/A	1.05 (1.00 1.10)	5.0 (0.0, 11.0)
Ito et al. (1995)	Chicago	N/A	1.08 (1.01, 1.10)	7.1 (1.0, 10.0)
Toulomi et al. (1994)	Athens, Greece	smoke	1.08 (1.06, 1.10)	8.0 (7.0, 12.0)
Verhoef et al. (1996)	Amsterdam, NL	N/A	1.08 (0.99, 1.16)	6.0 (-1.0, 14.0)
Spix et al. (1993/1994)	Erfurt, Germany (b)	1988-89	1.12 (1.04, 1.16)	12.0 (4.0, 16.0)
Saldiva et al. (1995)	Sao Paulo, Brazil	1990-91	1.14 (1.07, 1.21)	14.0 (7.0, 21.0)
Katsouyanni et al. (1997a)	Europe, Meta analysis	All cities ‡	1.04 (1.026, 1.06)	4.0 (2.6, 6.0)
Katsouyanni et al. (1997a)	Europe, Meta analysis	Western cities (5-9) ‡	1.067 (1.04, 1.08)	6.7 (4.0, 8.0)
Katsouyanni et al. (1997a)	Europe, Meta analysis	Eastern cities (1-4) ‡	1.10 (1.00, 1.22)	10.0 (0.00, 22.0)
Schwartz & Marcus (1990)	London (a)	1958-1972	1.03 (1.02, 1.04)	3.1 (2.9, 3.3)
Plagiannakos & Parker (1988)	Ontario, Canada	1976-1982	1.10 (1.05, 1.15)	9.8 (5.0, 14.9)
Zmirou et al. (1996)	Lyon	1985-1990	1.02 (0.94, 1.10)	2.0 (0.94, 10.0)
Spix & Wichmann (1996)	Cologne (b)	1975-1985	1.04 (1.01, 1.08)	4.0 (1.0, 8.0)
Touloumi et al. (1994)	Athens (a)	1975-1987	1.045 (1.02, 1.07)	4.5 (2.0, 7.0)
Sunyer et al. (1996)	Barcelona (a)	1985-1991	1.06 (1.02, 1.10)	6.0 (2.0, 10.0)
Ponce de Leon et al. (1996)	London (a)	1987/88; 1991/92	1.107 (1.05, 1.17)	10.7 (5.0, 17)
Wordley et al. (1997)	Birmingham, UK	1992-1994	1.10 (1.01, 1.21)	10.0 1.0, 21.0)
Prescott et al. (1998)	Edinburgh, UK (a)	1992-1995	1.14 (1.04, 1.24)	14.0 (4.0, 28.0)

a; Data based on BS, converted to PM_{10} using the formula PM_{10} = BS x 0.9.

b: Data based on TSP, converted to PM_{10} using the formula PM_{10} = TSP x 0.55.

c: Data based on Coefficient of Haze (CoH), converted to PM_{10} using the formula PM_{10} = CoH/x 0.55.

‡: City codes that appear in parentheses, are as follows: 1 = Poznan; 2 = Lodz; 3 = Wroclaw; 4 = Krakow; 5 = London; 6= Paris; 7 = Lyon; 8 = Barcelona; 9 = Milan.

➢Respiratory Mortality

Table 50: Percentage Change in Respiratory Mortality associated with $100\mu g/m^3$ increase in Particulate Matter.

Reference	Location	Time Period	RR per $100\mu g/m^3$ (95% CI)	Per Cent Change per $100\mu g/m^3$ (95% CI)
Fairly (1990)	Santa Clara (c)	1980-82; 1984-86	N/A	35.0 (15.0, 56.0)
Schwartz & Dockery (1992b)	Philadelphia (b)	1973-1980	1.33 (1.10, 1.66)	33 (10.0, 66.0)
Pope et al. (1992)	Utah Valley	1985-89	N/A	37.0 (7.0, 67.0)
Schwartz (1993a)	Birmingham, AL	1985-88	1.16 (0.77, 1.75)	16.0 (-23.0, 75.0)
Schwartz (1994a)	Cincinnati (b)	1977-82	N/A	27.0
Zmirou et al. (1996)	Lyon	1985-1990	1.08 (1.00, 1.19)	8.0 (0.00, 19.0)
Dab et al. (1996)	Paris	1987-1992	1.17 (1.04, 1.31)	17.0 (4.0, 31.0)
Bremner et al. (1999)	London	1992-1994	1.13 (1.03, 1.23)	13.0 (3.0, 23.0)
Vigotti et al. (1996)	Milan	1980-1989	1.12 (1.02, 1.23)	12.0 (2.0, 23.0)
Ponce de Leon et al. (1996)	London (a)	1987/88; 1991/92	1.04 (0.90, 1.22)	4.0 (-10.0-22.0)
Sunyer et al. (1996)	Barcelona (a)	1985-1991	1.10 (0.99, 1.22)	10.0 (0, 22.0)
Zmirou et al. (1998)	Meta-Analysis APHEA	Western Europe (5-8) ‡	1.08 (1.00, 1.16)	8.0 (0.00, 16.0)
Zmirou et al. (1998)	Meta-Analysis APHEA (b)	Western Europe (8-9) ‡	1.05 (0.95, 1.07)	5.0
Prescott et al. (1998)	Edinburgh, UK (a)	1992-1995	1.41 (1.09, 1.81)	41.0 (9.0, 81.0)

a: Data based on BS, converted to PM_{10} using the formula PM_{10} = BS x 0.9.

b: Data based on TSP, converted to PM_{10} using the formula PM_{10} = TSP x 0.55.

c: Data based on Coefficient of Haze (CoH), converted to PM_{10} using the formula PM_{10} = CoH/x 0.55.

‡: City codes that appear in parentheses, are as follows: 1 = Poznan; 2 = Lodz; 3 = Wroclaw; 4 = Krakow; 5 = London; 6= Paris; 7 = Lyon; 8 = Barcelona; 9 = Milan.

➢Cardiovascular Mortality

Table 51: Percentage Change in Cardiovascular Mortality associated with $100\mu g/m^3$ increase in Particulate Matter.

Reference	Location	Time Period	RR per $100\mu g/m^3$ (95% CI)	Per Cent Change per $100\mu g/m^3$ (95% CI)
Fairly (1990)	Santa Clara (c)	1980-82; 1984-86	N/A	8.0 (10.0, 16.0)
Schwartz & Dockery (1992b)	Philadelphia (b)	1973-1980	1.18 (1.14, 1.22)	18.0 (14.0, 22.0)
Pope et al. (1992)	Utah Valley	1985-89	N/A	18.0 (4.0, 33.0)
Schwartz (1993a)	Birmingham, AL	1985-88	1.17 (1.04-1.31)	17.0 (4.0, 31.0)
Schwartz (1994a)	Cincinnati (b)	1977-82	N/A	14.0 (5.0, 24.0)
Bremner et al. (1999)	London (a)	1992-1994	1.106 (1.011, 1.20)	10.6 (1.1, 20.0)
Zmirou et al. (1996)	Lyon	1985-1990	1.08 (0.98, 1.2.1)	8.0 (0.0, 12.1)
Ponce de Leon et al. (1996)	London (a)	1987/88; 1991/92	1.04 (0.95, 1.07)	4.0 (0.0, 7.0)
Sunyer et al. (1996)	Barcelona (a)	1985-1991	1.09 (1.04, 1.15)	9.0 (4.0, 15.0)
Zmirou et al. (1998)	Meta-Analysis APHEA (a)	Western Europe (5-8)	1.08 (1.02, 1.12)	8.0 (2.0, 12.0)
Zmirou et al. (1998)	Meta-Analysis APHEA (a)	Central Europe (1-4) ‡	1.00 (0.99, 1.04)	1.0 (-0.1, 4..0)
Zmirou et al. (1998)	Meta-Analysis APHEA (b)	Western Europe (8-9) ‡	1.01 (0.99, 1.03)	1.0 (-0.1, 3.0)

a; Data based on BS, converted to PM_{10} using the formula PM_{10} = BS x 0.9.
b: Data based on TSP, converted to PM_{10} using the formula PM_{10} = TSP x 0.55.
c: Data based on Coefficient of Haze (CoH), converted to PM_{10} using the formula PM_{10} = CoH/x 0.55.
‡: City codes that appear in parentheses, are as follows: 1 = Poznan; 2 = Lodz; 3 = Wroclaw; 4 = Krakow; 5 = London; 6= Paris; 7 = Lyon; 8 = Barcelona; 9 = Milan.

7.3 Summary of Selected Time Series Mortality Studies for Nitrogen Dioxide

➢ Total Mortality

Table 52: Percentage Change in Total Mortality associated with 100μg/m^3 increase in Nitrogen Dioxide.

Reference	Location	Time Period	RR per 100μg/m^3 (95% CI)	Per Cent Change per 100μg/m^3 (95% CI)
Zmirou et al. (1996)	Lyon	1985-1990	1.02 (0.98, 1.04)	2.0 (-0.1, 4.0)
Spix & Wichmann (1996)	Cologne	1975-1985	1.01 (0.99, 1.02)	1.0 (-0.1, 2.0)
Sunyer et al. (1996)	Barcelona	1985-1991	1.03 (1.01, 1.06)	3.0 (1.0, 2.0)
Ponce de Leon et al. (1996)	London	1987/88; 1991/92	1.01 (0.99, 1.01)	1.0 (-0.1, 1.0)
Touloumie et al. (1997)	Meta-Analysis APHEA	N/A	1.04 (1.034, 1.52)	4.0 (3.4, 5.2)

➢Respiratory Mortality

Table 53: Percentage Change in Respiratory Mortality associated with 100μg/m^3 increase in Nitrogen Dioxide.

Reference	Location	Time Period	RR per 100μg/m^3 (95% CI)	Per Cent Change per 100μg/m^3 (95% CI)
Zmirou et al. (1996)	Lyon	1985-1990	0.98 (0.96, 1.02)	
Dab et al. (1996)	Paris	1987-1992	1.02 (0.94, 1.1)	2.0 (-0.2, 10.0)
Ponce de Leon et al. (1996)	London	1987/88; 1991/92	0.97 (0.93, 1.01)	
Sunyer et al. (1996)	Barcelona	1985-1991	1.03 (0.97, 1.09)	3.0 (-0.2, 9.0)
Zmirou et al. (1998)	Meta-Analysis APHEA	Western Europe (5-8) ‡	1.00 (0.98, 1.01)	0.0 (-0.2, 1.0)

‡: City codes that appear in parentheses, are as follows: 1 = Poznan; 2 = Lodz; 3 = Wroclaw; 4 = Krakow; 5 = London; 6= Paris; 7 = Lyon; 8 = Barcelona; 9 = Milan.

➢Cardiovascular Mortality

Table 54: Percentage Change in Cardiovascular Mortality associated with 100µg/m^3 increase in Nitrogen Dioxide.

Reference	**Location**	**Time Period**	**RR per 100µg/m^3 (95% CI)**	**Per Cent Change per 100µg/m^3 (95% CI)**
Zmirou et al. (1996)	Lyon	1985-1990	1.02 (0.98, 1.06)	2.0 (-0.2, 6.0)
Bremner et al. (1999)	London	1992-1994	1.097 (1.013, 1.186)	9.7 (1.3, 18.6)
Ponce de Leon et al. (1996)	London	1987/88; 1991/92	1.01 (0.99, 1.04)	1.0 (0.0, 4.0)
Sunyer et al. (1996)	Barcelona	1985-1991	1.04 (1.01, 1.07)	4.0 (1.0, 7.0)
Zmirou et al. (1998)	Meta-Analysis APHEA	Western Europe (5-8) ‡	1.01 (1.00, 1.02)	1.0 (.0, 2.0)

‡: City codes that appear in parentheses, are as follows: 1 = Poznan; 2 = Lodz; 3 = Wroclaw; 4 = Krakow; 5 = London; 6= Paris; 7 = Lyon; 8 = Barcelona; 9 = Milan.

7.4 Summary of Selected Time Series Mortality Studies for Ozone

➢ Total Mortality

Table 55: Percentage Change in Total Mortality associated with 100µg/m^3 increase in Ozone.

Reference	**Location**	**Time Period**	**RR per 100µg/m^3 (95% CI)**	**Per Cent Change per 100µg/m^3 (95% CI)**
Zmirou et al. (1996)	Lyon	1985-1990	1.08 (0.88, 1.35)	8.0
Sunyer et al. (1996)	Barcelona	1985-1991	1.05 (1.01, 1.09)	5.0 (1.0, 9.0)
Ponce de Leon et al. (1996)	London	1987/88; 1991/92	1.09 (1.05, 1.14)	9.0 (5.0, 14.0)
Touloumi et al. (1997)	Meta Analysis APHEA	N/A	1.05 (1.02, 1.1)	5.0 (2.0, 10.0)

➢Respiratory Mortality

Table 56: Percentage Change in Respiratory Mortality associated with 100µg/m³ increase Ozone.

Reference	Location	Time Period	RR per 100µg/m³ (95% CI)	Per Cent Change per 100µg/m³ (95% CI)
Zmirou et al. (1996)	Lyon	1985-1990	1.02 (0.81, 1.28)	2.0
Dab et al. (1996)	Paris	1987-1992	1.04 (0.93, 1.16)	4.0
Ponce de Leon et al. (1996)	London	1987/88; 1991/92	1.24 (1.09, 1.40)	24.0 (9.0, 40.)
Sunyer et al. (1996)	Barcelona	1985-1991	1.07 (0.96, 1.19)	7.0
Zmirou et al. (1998)	Meta-Analysis APHEA	Western Europe (5-8) ‡	1.10 (1.04, 1.16)	10.0 (4.0, 16.0)

‡: City codes that appear in parentheses, are as follows: 1 = Poznan; 2 = Lodz; 3 = Wroclaw; 4 = Krakow; 5 = London; 6= Paris; 7 = Lyon; 8 = Barcelona; 9 = Milan.

➢Cardiovascular Mortality

Table 57: Percentage Change in Cardiovascular Mortality associated with 100µg/m³ increase in Ozone.

Reference	Location	Time Period	RR per 100µg/m³ (95% CI)	Per Cent Change per 100µg/m³ (95% CI)
Ponce de Leon et al. (1996)	London	1987/88; 1991/92	1.08 (1.01, 1.15)	8.0 (1.0, 15.0)
Bremner et al. (1999)	London	1992-1994	1.11 (1.016, 1.219)	11.4 (1.6, 21.9)
Sunyer et al. (1996)	Barcelona	1985-1991	1.06 (1.01, 1.11)	6.0 (1.0, 11.0)
Zmirou et al. (1996)	Lyon	1985-1990	0.88 (0.67, 1.19)	- 11.0 (-30.0, 19.0)
Zmirou et al. (1998)	Meta-Analysis APHEA	Western Europe (5-8) ‡	1.04 (1.01, 1.06)	4.0 (1.0, 6.0)

‡: City codes that appear in parentheses, are as follows: 1 = Poznan; 2 = Lodz; 3 = Wroclaw; 4 = Krakow; 5 = London; 6= Paris; 7 = Lyon; 8 = Barcelona; 9 = Milan.

7.5 Summary of Selected Time Series Mortality Studies for Sulphur Dioxide

➢ Total Mortality

Table 58: Percentage Change in Total Mortality associated with 100μg/m^3 increase in Sulphur Dioxide.

Reference	Location	Time Period	RR per 100μg/m^3 (95% CI)	Per Cent Change per 100μg/m^3 (95% CI)
Hatzakis et al. (1986)	Athens	1975-1982	N/A	4.8 (2.0, 12.0
Zmirou et al. (1996)	Lyon	1985-1990	1.12 (1.04, 1.09)	12.0 (4.0, 9.0)
Touloumi et al. (1996)	Athens	1987-1991	1.12 (1.07, 1.16)	12.0 (7.0, 16.0)
Spix & Wichmann (1996)	Cologne	1975-1985	1.03 (1.02 1.04)	3.0 (2.0, 4.0)
Sunyer et al. (1996)	Barcelona	1985-1991	1.13 (1.07, 1.19)	13.0 (7.0, 19.0)
Ponce de Leon et al. (1996)	London	1987/88; 1991/92	1.03 (1.00, 1.07)	3.0 (0.0, 7.0)
Katsouyanni et al. (1997a)	Europe, APHEA Meta analysis	All cities ‡	1.047 (1.034, 1.056)	4.7 (3.4, 5.6)
Katsouyanni et al. (1997)	Europe, APHEA Meta analysis	Western cities (5-9) ‡	1.065 (1.048, 1.08)	6.5 (4.8, 8.0)
Katsouyanni et al. (1997a)	Europe, APHEA Meta analysis	Eastern cities (1-4) ‡	1.02 (1.004, 1.038)	2.0 (0.4, 3.8)

‡: City codes that appear in parentheses, are as follows: 1 = Poznan; 2 = Lodz; 3 = Wroclaw; 4 = Krakow; 5 = London; 6= Paris; 7 = Lyon; 8 = Barcelona; 9 = Milan.

➢Respiratory Mortality

Table 59: Percentage Change in Respiratory Mortality associated with 100μg/m^3 increase in Sulphur Dioxide.

Reference	Location	Time Period	RR per 100μg/m^3 (95% CI)	Per Cent Change per 100μg/m^3 (95% CI)
Zmirou et al. (1996)	Lyon	1985-1990	1.10 (1.04, 1.18)	10.0 (4.0, 18.0)
Dab et al. (1996)	Paris	1987-1992	1.08 (0.97, 1.21)	8.0
Ponce de Leon et al. (1996)	London	1987/88; 1991/92	1.06 (0.95, 1.18)	6.0
Sunyer et al. (1996)	Barcelona	1985-1991	1.13 (0.99, 1.28)	13.0 (-0.1, 28.0)
Vigotti et al. (1996)	Milan	1980-1989	1.12 (1.03, 1.23)	12.0 (3.0, 23.0)
Zmirou et al. (1998)	Meta-Analysis APHEA	Western Europe (5-9) ‡	1.12 (1.06, 1.16)	12.0 (6.0, 16.0)
Zmirou et al. (1998)	Meta-Analysis APHEA	Central Europe (1-4) ‡	1.04 (0.98, 1.12)	4.0 (-0.01, 12.0)

‡: City codes that appear in parentheses, are as follows: 1 = Poznan; 2 = Lodz; 3 = Wroclaw; 4 = Krakow; 5 = London; 6= Paris; 7 = Lyon; 8 = Barcelona; 9 = Milan.

➢Cardiovascular Mortality

Table 60: Percentage Change in Cardiovascular Mortality associated with 100μg/m³ increase in Sulphur Dioxide.

Reference	Location	Time Period	RR per 100μg/m³ (95% CI)	Per Cent Change per 100μg/m³ (95% CI)
Zmirou et al. (1996)	Lyon	1985-1990	1.17 (1.06, 1.25)	17.0 (6.0, 25.0)
Ponce de Leon et al. (1996)	London	1987/88; 1991/92	1.01 (0.95, 1.07)	1.0
Sunyer et al. (1996)	Barcelona	1985-1991	1.15 (1.06, 1.23)	15.0 (6.0, 23.0)
Zmirou et al. (1998)	Meta-Analysis APHEA	Western Europe (5-9) ‡	1.10 (1.04, 1.18)	10.0 (4.0, 18.0)
Zmirou et al. (1998)	Meta-Analysis APHEA	Central Europe (1-4) ‡	1.02 (1.00, 1.04)	2.0 (1.0, 4.0)

‡: City codes that appear in parentheses, are as follows: 1 = Poznan; 2 = Lodz; 3 = Wroclaw; 4 = Krakow; 5 = London; 6= Paris; 7 = Lyon; 8 = Barcelona; 9 = Milan.

7.6 Summary of Time Series Mortality Studies for Carbon Monoxide

Weaker evidence can be found in the epidemiological for the association between fluctuations of carbon monoxide levels and mortality. However, some recent studies report a significant positive relationship. Touloumi et al. (1996), for instance report for Athens an average rise in all-cause mortality of 10 per cent associated with an increase in CO concentrations of 10 mg/m³ (RR = 1.10 (CI 1.05-1.15).[9] Three studies examining data for Los Angeles report an increase in all-cause mortality of four, five and seven per cent respectively, associated with an increase in carbon monoxide concentration of 10 ppm (12.05 mg/m³).[10] The

[9] See. Touloumi, Samoli and Katsouyanni (1996).

[10] See: Shumway, Azari and Pawitan (1988), Hexter and Goldsmith (1971) and Kinney, Ito and Thurston (19950.

Department of Health concludes that a rise of 10 ppm carbon monoxide can be associated with increases in all-cause mortality of about 10 per cent.[11]

7.7 Summary

The aim of this chapter was to summarise the results of recent studies on the short-term effects of air pollution on human health, i.e. premature mortality. Severe episodes of air pollution in North America and Europe before the 1960s provided clear evidence of the relationship between high levels of air pollution and acute effects on human health, including a significant increase in mortality. Most national and international air quality guidelines and standards have in fact been based on the results of these early studies. Hence, following these episodes, measures have been taken resulting in decreasing air pollution levels. Additionally, a change in emission sources, as described in Chapter 3, was followed by changes in the air pollution mixtures. The now moderate or low air pollution levels have been assumed to be 'safe' for human health. However, recently published work, mainly from the US, indicates that these low air pollution levels still have short-term effects on mortality and morbidity.

Furthermore, these effects may even occur at air pollution levels that are below the set safety limits. Accordingly, small elevations of air pollution could be associated with a small but statistically significant increase in mortality, although the methodological approaches used vary substantially. Additionally, concern had been raised regarding the question of whether these results could be quantitatively extrapolated to Europe. Differences in emission sources, pollution mixes, climate, lifestyle, and the underlying health of the population may influence the health effects of air pollution. Hence, the multi-centre APHEA

---

[11] See: Department of Health (1998).

project (Air Pollution Health Effects - A European Approach) was initiated, where a standardised methodological approach had been applied across all groups involved in the study.

The results of European data replicated those of previous studies, although the pollution effects appear to be lower than those found in the earlier non-European studies. For the two pollutants PM_{10} and SO_2, a positive and statistically significant association was found. The results indicate an increase in mortality ranging from 2 to 12% for PM_{10} and 3 to 12% for SO_2 each time the respective concentration increases by $100\mu g/m^3$.

Examining the same effect parameters for cardiovascular mortality, a similar pattern to total mortality can be seen. The increase of respiratory mortality, however, tended to be generally higher. While positive and, mostly, statistically significant effects between 5 and 9% are shown for ozone, the effects of NO_2 are somewhat weaker and less consistent throughout the various studies. Also Western European cities seem to show greater effects than Central-Eastern European cities included in the APHEA project. However, the consistency of the results across the European cities, with wide differences in climate and environmental conditions, suggest that the observed associations may indeed be causal. Following Katsouyanni et al. (1997): "The observed relative risks are small but they may constitute an important public health problem if the ubiquity of air pollution exposure is taken into account."[12]

[12] Katsouyanni et al. (1997a), p.16.

Having presented the results of previous time-series studies analysing the acute mortality effects of air pollution, the following chapters of this study are concerned with a similar analysis of the respective data from 14 British cities, in order to establish comparable results for Great Britain; particularly since the majority of the studies illustrated above have been undertaken outside Great Britain. Before introducing and discussing an appropriate estimation model, and the consequent estimation results, the following chapter will present and discuss the various data sets necessary for such an analysis.

8. Descriptive Statistics of the Data

8.1 Introduction

Estimating the adverse effects of air pollution on human mortality requires several data sets. Since in this study such health effects are investigated for various British cities, these data sets have to be obtained for each of the cities included in the study. For this study, three main sets of data were obtained, comprising air pollution data, meteorological data and health data. This chapter gives a detailed description of the data obtained and is divided in two main parts. The first part gives a general overview of the data sets, including a description of the respective variables and the motivation for the choice of variables. The second part is further divided into sections each representing one of the cities included in the study.

Due to the rather extensive size of the data involved in this study, the data is summarised in a single table for each city. This is intended to give a summary overview of the main variables analysed. Selected centile points, as well as maximum, minimum and mean values of the respective variables are presented. The tables illustrate the distribution of the three main data sets of air pollution, weather and mortality for the study period. Additionally, histograms are provided for the mortality data in order to illustrate the underlying distribution of this variable. The methodological implications of the expected right skewness of the mortality variables will be discussed in the following chapter.

Since people are most exposed to traffic-related air pollutants in urban areas, the aim was to include as many cities as possible in this study in order to obtain a picture that covers as much of the urban areas of Britain as possible. Ultimately, the limiting factor of whether a specific city could be included in the study was the availability of sufficient data on air pollution, as will be discussed in the following section. The British cities that were finally considered in this study are represented on the following graphical map, and include Birmingham, Bristol, Cardiff, Edinburgh, Glasgow, Kingston upon Hull, Leeds, Leicester, Liverpool, London, Newcastle upon Tyne, Sheffield, Southampton and Swansea. The time horizon for this study covers the calendar years 1992 to 1997.

Figure 33: Map of all 14 British Cities Analysed.

Source: Own graphic.

8.2. Air Pollution Data

Pollution variables were chosen to cover a range of criteria pollutants of both primary and secondary nature, i.e. pollutants that are emitted directly or formed via reactions in the atmosphere. Definitions and characteristics of the respective air pollutants have already been discussed extensively in Chapter 3. The pollutant variables chosen comprise particulate matter (PM_{10}), sulphur dioxide (SO_2), nitrogen dioxide (NO_2), carbon monoxide (CO), and ozone (O_3).

The Department of the Environment, Transport and the Regions funds a number of national-scale air quality monitoring programmes throughout the UK. By September 1998, there were 108 automatic monitoring sites in operation in the UK. These stations are situated in a variety of locations. The majority (89) are so-called 'urban background' stations, which are designed to represent air pollution levels to which large numbers of individuals may be exposed. This category of monitoring sites includes locations such as kerbside, roadside, city centre, background or suburban and industrial sites.[1] The remaining 19 monitoring stations are so-called rural monitoring sites.

Investigating potential health effects associated with air pollution in cities, this present study exclusively employs pollution data from the urban automatic network. Using fixed-point real-time continuous analysis to monitor ambient levels of the five criteria air pollutants included in this study, hourly values are recorded. From these hourly measurements 24-hour daily averages have been computed. The instruments currently deployed in the automatic monitoring networks use the following measurement techniques: Tapered Element Oscillating Microbalance for PM_{10}; UV fluorescence for SO_2;

[1] For a description of the various sites see Broughton et al. (1997), pp. 437-440.

Chemiluminescence for oxides of nitrogen (NO_x)[2]; IR absorption for carbon monoxide (CO); and UV absorption for ozone (O_3).[3]

Information from the described air pollution monitoring network is made readily available through the internet, where the respective data is located on the website of the National Air Quality Information Archive, maintained by AEA Technology plc.[4] Consequently, the air pollution data for all included cities was retrieved from the National Air Quality Information Archive. In instances where pollution data is recorded at more than one urban station, the average of the respective pollution data for all sites has been computed and used as the representative data for the respective city.[5] As noted above, the availability of the pollution data turned out to be the restrictive factor in deciding which cities were to be included in this study.

Only recently has the number of sites that monitor air quality automatically on a regular base in the UK been expanded. Before 1990, the number of such sites was just over 20 for the UK. Subsequently, Local Authority sites have been affiliated into the monitoring network and new centrally funded sites have been established.

The continuous monitoring of particulate matter (PM_{10}), which is regarded as one of the most hazardous air pollutants for human health, only started on a regular basis in 1992. As a consequence, it was decided to include only those British cities where sufficient daily data on all of the five 'criteria' pollutants was

---

[2] Basically, the total oxides of nitrogen (NO_x) is the sum of nitric oxide (NO) and nitrogen dioxide (NO_2). The monitor measures NO in the air stream and also catalytically converts NO_2 to NO and measures that. Thus, the output of the monitor is NO and total oxides of nitrogen (NO_x). NO_2 is then determined by subtraction.

[3] For a detailed description of the measurement techniques, see Broughton et al. (1997).

[4] The National Air Quality Information Archive's Internet address is: http:/www.aeat.co.uk/netcen/aqarchive/archome.html.

[5] See, for instance, Ponce de Leon et al. (1996).

available for the time period 1992 through 1997, which represents a time-series of 2192 days. However, since the expansion of the monitoring network has only recently been undertaken, sufficient air pollution data of all air pollutants was still not available for every city considered in this study. Nevertheless, even these time-series were sufficiently long in order to be used in the statistical analysis (minimum of two years of daily data). For days with missing values for the air pollutants, some studies estimated these missing values on the basis of the remaining ones using linear interpolation models.[6] In this present study, however, no such attempt has been made, and hence the days with missing pollution values were excluded from further analysis automatically by the econometric software package used (EViews 3.1).

Further, in line with the majority of relevant studies, it was decided to use daily mean values for the respective pollution variables, rather than maximum hourly average values.[7] The rationale being that daily averages of air pollution tend to be smoother spatially and thus, more representative of average community level, since peak value of air pollution may only apply to a localised area around the monitoring station and this may not be representative of the entire community.[8] Lipfert (1993) also points out that:

> [I]f it is desired to compare regression or correlation results among various pollutants, it is essential that data for all pollutant species be based on the same averaging time in order to compare on the basis of equivalent random noise levels and spatial representation. Differences in the "noise" in the independent variables can bias the comparison of regression statistics.[9]

---

[6] See, for instance, Hatzakis et al. (1986).
[7] The maximum hourly average is the highest hourly reading of air pollution obtained during the time period under study, usually 24 hours.
[8] See: Lipfert (1993), pp. 231-232.
[9] Ibid, p. 232.

The following table summarises the time periods for which the various air pollutants were available, which in turn determines the time-series analysed in this study.

Table 61: Summary of Periods of Pollution Data Availability.

	PM_{10}	NO_2	SO_2	CO	Ozone
Birmingham	1992-1997	1992-1997	1992-1997	1992-1997	1992-1997
Bristol	1993-1997	1993-1997	1993-1997	1993-1997	1993-1997
Cardiff	1992-1997	1992-1997	1992-1997	1992-1997	1992-1997
Edinburgh	1992-1997	1992-1997	1992-1997	1992-1997	1992-1997
Glasgow*	1992-1997	1992-1997	1996-1997	1992-1997	1996-1997
Hull	1994-1997	1994-1997	1994-1997	1994-1997	1994-1997
Leeds	1993-1997	1993-1997	1993-1997	1993-1997	1993-1997
Leicester	1994-1997	1994-1997	1994-1997	1994-1997	1994-1997
Liverpool	1993-1997	1993-1997	1993-1997	1993-1997	1993-1997
London	1992-1997	1992-1997	1992-1997	1992-1997	1992-1997
Newcastle	1992-1997	1992-1997	1992-1997	1992-1997	1992-1997
Sheffield	1996-1997	1992-1997	1996-1997	1992-1997	1996-1997
Southampton	1994-1997	1994-1997	1994-1997	1994-1997	1994-1997
Swansea	1994-1997	1994-1997	1994-1997	1994-1997	1994-1997

Note: (*): For Glasgow, data on Black Smoke has been analysed rather than PM_{10}, due to availability (See later chapters).

Finally, the concentrations of the various air pollutants are generally expressed in two different ways. First, they may be expressed as the ratio of the volume of the gaseous pollutants (expressed as if pure) to the volume of air in which the respective pollutant is contained. This is usually expressed as a volume mixing ratio or parts per million (ppm) or parts per billion (ppb). Hence, concentrations of 1ppm (ppb) mean that for every million (billion) units of air, there is one unit of the respective air pollutant present.

The second possibility is to express the concentration of air pollutants as the mass of pollutant in a given volume of air. This is usually expressed as micrograms per cubic metre ($\mu g/m^3$). In other words, concentrations of $1 \mu g/m^3$ mean that one cubic metre of air contains one microgram (millionth of a gram) of the respective air pollutant.

The unit employed for expressing pollutant concentrations that is used in the majority of relevant studies is micrograms per cubic meter ($\mu g/m^3$). This is also the unit, which has now been adopted for international use by the OECD.[10] Hence, in order to compare estimation results of this study with other studies, it has been decided to convert the concentrations of all included pollutants and express them as micrograms per cubic metre ($\mu g/m^3$). One exception is carbon monoxide (CO), where the more commonly concentration unit is still parts per billion (ppb). Hence, for this pollutant no conversion has been made. Further, particulate matter (PM_{10}) is already measured and expressed in micrograms per cubic metre, therefore no conversion was necessary. The conversion factors applied for the remaining gaseous pollutants studied here are given in the table below.[11]

Table 62: Conversion Factors for Specific Air Pollutants.

Pollutant	Conversion factor ppb to $\mu g/m^3$
NO_2	1.98
SO_2	2.76
O_3	2.07

Source: Department of Health (1998), p.73.

---

[10] See: Broughton, Bower, Willis and Clark (1997), p. 26.

[11] Strictly, since the mass concentration $\mu g/m^3$ will depend on the ambient temperature as well as atmospheric pressure, it should ideally be specified each time a concentration is measure. However, the variation is normally not large and may only be of significance where large variations in both temperature and pressure occur. See: Broughton, Bower, Willis and Clark (1997) and Department of Health (1998).

8.3 Meteorological Data

When examining the association between short-term fluctuations in air pollution and short-term fluctuations in the number of daily deaths, meteorological data plays a very significant role, as will be discussed in detail in the following chapter. The meteorological data that is to be considered in this study includes daily data on temperature, daily data on relative humidity, and daily data on dew point temperature for each of the cities included in this study.

The UK Meteorological Office provided all the meteorological data used in this study. For the case where the Meteorological Office operates no weather station in the respective city itself (Birmingham, Edinburgh, Glasgow, Kingston upon Hull, Leicester, Liverpool and Swansea), the weather data was obtained from the nearest weather station, usually at the nearest airport. The data covered the study time period between 1992 and 1997 inclusive.

Mean temperature is recorded by adding together the maximum and minimum temperature on the respective day, and dividing by two. The resulting values are noted in degrees and tenths Celsius. The relative humidity is a measure of how close the air is to being saturated with water vapour. Respective values are expressed as a percentage.[12] The dew point temperature is the temperature to which the air, if cooled at constant pressure, would become saturated with respect to water. In other words, it can be considered the temperature at which dew would generally form.

[12] Values of above 95% are associated with mist and fog.

For all three weather variables mean values were considered rather than minimum values (these are sometimes used for temperature variables in order to emphasise the impact of cold weather on human health in the case of temperature), or maximum values. Although some studies report that the extreme values may be a better predictor for various health effects,[13] this study follows the approach used by the majority of the relevant studies and considers mean values to summarise meteorological conditions.

8.4 Mortality Data

The time-series data for the outcome (dependent) variable in this study are daily counts of all deaths as well as daily counts of deaths from specific causes. Hence, the Office for National Statistics provided anonymous death records for the study period between 1992-1997 for all cities except Edinburgh and Glasgow, which was provided by the General Register Office for Scotland. Causes of death were coded following the International Classification of Diseases diagnostic codes Ninth revision (ICD-9), as defined by the World Health Organization (WHO). These are standardised codes used to classify morbidity and mortality information for statistical purposes. Originally, the data were taken from death certificates. Using the ICD-9 codes, two main groups of causes of deaths were considered and classified according to:

---

[13] One study, for instance, found that minimum temperature was a better predictor of asthma symptoms than respective mean or maximum values. See: Schwartz et al. (1993), p. 827.

- Total daily number of deaths from diseases of the circulatory system (ICD-9 390-459).[14]

- Total daily number of deaths from diseases of the respiratory system (ICD-9 460-519).[15]

This classification of the outcome variable has been chosen in favour of total daily death counts, although a large number of relevant studies have restricted their analysis to the examination of the association between air pollution and total daily death counts (excluding external causes).[16] The reason for this choice is that many studies show that air pollution actually has the largest and most direct effect on the human respiratory and cardiovascular system.

Given that the lung is the primary target organ for the health effects of air pollution, this is to be expected. Further, a number of studies have identified impaired lung function as an independent risk factor for heart disease morbidity and mortality, i.e. cardiovascular disease (CVD), as well as for all-cause mortality.[17] Following a report by the Department of Health: "It is known that the increases in all-cause mortality are largely due to increases in respiratory and cardiovascular mortality."[18]

---

[14] This category includes: acute rheumatic fever (390-392), chronic rheumatic heart disease (393-398), hypertensive disease (401-405), ischemic heart disease (410-414), disease of pulmonary circulation (415-417), other forms of heart disease (420-429), cerebrovascular disease (430-438), diseases of arteries, arterioles and capillaries (440-448), diseases of veins and lymphatic other diseases of circulatory system (451-459).

[15] This category includes: acute respiratory infections (460-466),other diseases of the upper respiratory tract (470-478), pneumonia and influenza (480-487), chronic obstructive pulmonary disease and allied conditions (490-496), pneumoconioses and other lung diseases due to external agents (500-508) and other diseases of respiratory system (510-519).

[16] Nevertheless, the data on all-cause daily death, excluding external causes (ICD-9 < 800) has also been obtained.

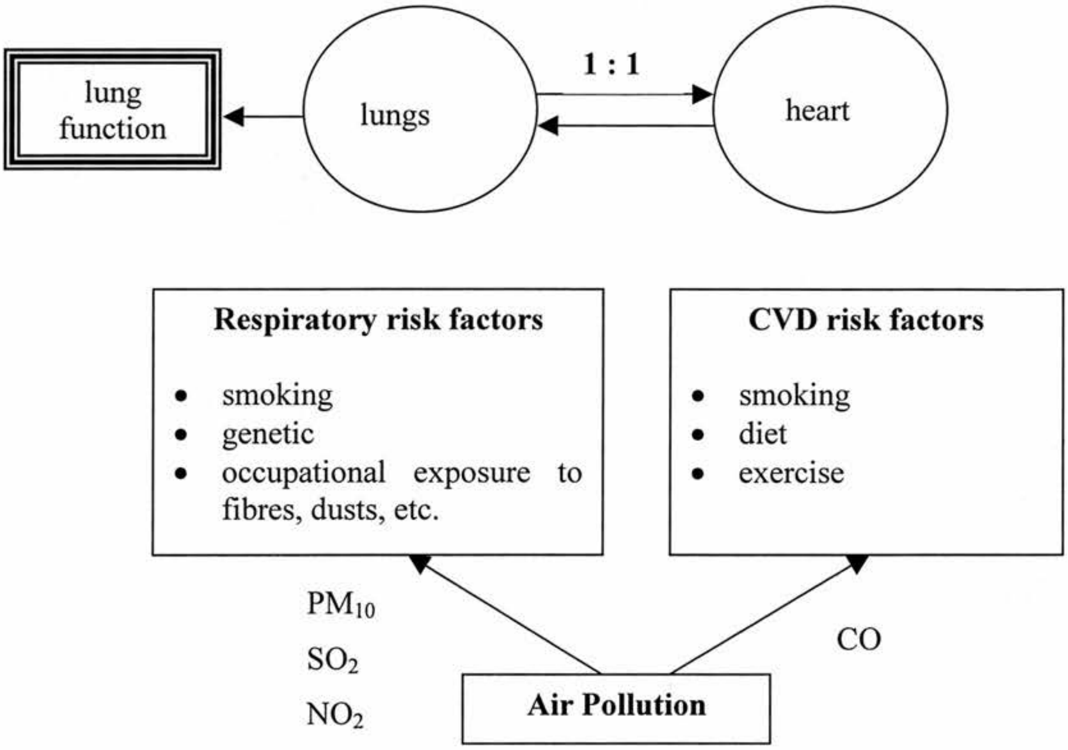
[17] See, for instance, Peters et al. (1997).

[18] Department of Health (1999), p. 14. This has also been shown in various other studies. See, for instance, the review by Dockery and Pope (1996).

Basically, the human cardio-respiratory system functions as a coupled system in order to guarantee the exchange of oxygen and carbon dioxide across the alveolar membranes (i.e. small air sacs at the end of the bronchiole).

The following figure illustrates in a simplified way the basic interrelationships in the cardio-respiratory system.

Figure 34: Interrelationships in the Cardio-Respiratory System.



Source: Lipfert (1993), p. 95.

As illustrated in the figure above and already discussed in Chapter 3, air pollution acts predominantly on both systems, either through the deterioration of the lung performance or through toxic agents in the blood (CO). The 1:1 notation indicates that a given loss in respiratory performance (in per cent) is actually reflected as the same percentage of excess risk due to heart disease.[19] Consequently, it has been decided in this study to concentrate on the analysis of the relationship between air pollution and both respiratory and cardiovascular mortality. Both outcome variables are summed and investigated as one outcome variable.

---

[19] For a detailed discussion, see, for instance, Lipfert (1993), pp. 435-515.

Besides the cause of death and date of death, the data was also broken down by a number of other criteria, including age, sex, postcode of usual residence, social class of deceased or mother, and occupation of deceased or mother. Deaths occurring outside the city of usual residence were excluded from the analysis, which had been controlled for by analysing the respective postcodes.

Due to the way mortality data is reported and finally provided by the respective authority, it could not be confirmed with certainty whether dates with missing counts corresponded with a count of zero or a missing value. Consequently, it has been decided to exclude these data from any further analysis. Finally, these daily death counts have been matched with the daily data on the specific air pollutants examined.

8.5 Distribution of the Data

The following section gives an overview of the distribution of the three main data sets described in the previous sections. These data sets are separately presented in a summarised table for each city. For each variable of the main categories, the dispersion of the data around the central point using the commonly applied percentiles of 5%, 10%, 25% 50%, 75%, 90% and 95% are given. An example of how these percentiles are calculated is illustrated using air pollution.

First, the measured daily concentrations of the respective air pollutant are sorted into ascending order of concentration value, $C_1, C_2, C_3, \ldots, C_N$. Second, the associated percentile value for each concentration value is then calculated using the following formula:

$$P_i = \left(\frac{i}{N+1}\right) \times 100 \tag{14}$$

where:

P_i: percentile for the ith concentration in the sorted set, that is, P_i% of the concentrations will *be equal to or less than* C_i.

N: number of result (daily concentrations) available.

Additionally, maximum, minimum and mean values are also presented. In particular, the distribution characteristics of the mortality data are of great importance and are decisive criteria in terms of the methodological application, as described in detail in the following chapter. Hence, histograms are also provided in order to illustrate the distribution of the outcome variables. As expected the distributions show a skewness to the right with mean > median > mode. Since these graphs, however, do not always provide a clear-cut answer on how the data is distributed, an additional measure is computed. This skewness statistic is a measure of asymmetry of the distribution of the series around its mean and is computed as follows:

$$S = \frac{1}{N}\sum_{i=1}^{N}\left(\frac{y_i - \overline{y}}{\hat{\sigma}}\right)^3 \tag{15}$$

where: N: number of observation in sample.

$\overline{y}$: series mean.

$\hat{\sigma}$: based on the biased estimator of the variance.

For a symmetric distribution, this skewness parameter is zero, whereas a positive skewness means that the respective distribution has a long right tail, as in a Poisson distribution. All series investigated show a positive skewness, which confirms the expected non-normal distribution of the data. This skewness factor is reported together with the kurtosis statistic.

The kurtosis measures the peakness or flatness of the distribution of a specific series.[20] Values for the kurtosis statistic exceeding 3, are said to be peaked (leptokurtic) relative to the normal distribution, where the kurtosis is exactly 3. Likewise, a kurtosis statistic less than 3 is said to be flat (platykurtic) relative to the normal. All mortality series examined here show a peaked distribution with a kurtois exceeding 3, which also confirms the non-normal distribution assumption. Finally, the Jarque-Bera test statistic, which tests whether a specific series is normally distributed, is also calculated. Here, the difference of the skewness and kurtosis of the respective series with those from the normal distribution is measured. The statistic is computed as follows:[21]

$$J - B = \frac{N-k}{6}\left(S^2 \frac{1}{4}(K-3)^2\right) \qquad (16)$$

where:

S: Skewness.

K: Kurtosis.

k: number of estimated coefficients used to create the series.

N; sample size.

Under the null hypothesis of a normal distribution N (0,1), the Jarque-Bera statistic is distributed as χ^2 with 2 degrees of freedom. For all examined mortality data series the statistic lead to a rejection of the null hypothesis of a normal distribution at both the 5% significance level (critical value 5.99) and the 1% significance level (critical value 9.21).

---

[20] The kurtosis is computed like the skewness factor, substituting the power of 3 by 4.

[21] See: Stewart and Gill (1998), pp. 161-163.

The following city-specific sub-sections additionally provide a short description of the air pollution monitoring sites.[22] This is seen as appropriate, since the availability of quality air pollution data recorded by these monitoring systems mainly determined the size of this study. Additionally, for each city some key characteristics, representing the year 1996, are summarised in tabular form.[23]

8.5.1 Birmingham

In Birmingham data from two monitoring sites was obtained. The first monitoring site is located at the Centenary Square within a pedestrianised area of the city centre. The nearest road is approximately 10 metres distance and is used for access to the adjacent car park. The nearest heavily trafficked urban road is approximately 60 metres from the station. The manifold inlet is 3.5 metres above ground level. The surrounding area is generally open and comprises urban retail and business outlets. Trees occur within 2 metres distance of the monitoring station.

The second monitoring site at Ward End School, Ingleton Road is located within the grounds of a local school in an urban residential area. The station is more than 10 metres from the nearest residential road and some 250 metres from the nearest heavily trafficked urban road. The manifold inlet is approximately 3 metres above ground level. The surrounding area is generally open and comprises residential dwellings. A large industrial complex occurs to the Northeast of the site approximately 400 metres from the monitoring station.

---

[22] The site description have been adapted from the information provided by the DETR Automatic Urban and Rural Netwwork Site Information Archive on http://www.seiph.umds.ac.uk.detr/site.htm.

[23] These figures are all taken from ONS (1998b), pp. 166-205.

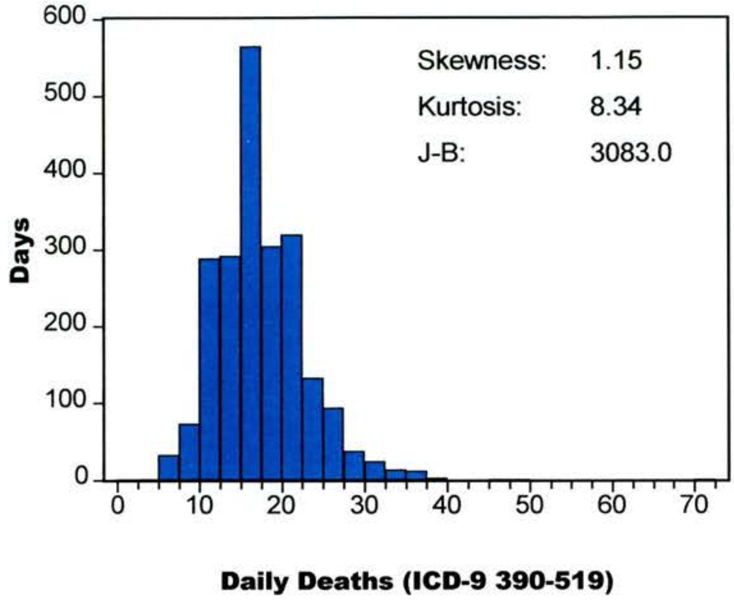
Table 63: Key Figures for Birmingham.

Area (km^2)	255
Persons per km^2	3,844
Male Population (thousands)	504
Female Population (thousands)	518
Total Population (thousands)	1,021

Table 64: Distribution of Mortality, Weather and Air Pollution in Birmingham.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All causes	20	22	25	29	33	38	41	103	3	30
Respiratory	1	2	3	4	6	8	9	30	1	5
Cardiovascular	7	8	10	13	15	18	20	48	1	13
Respiratory & Cardiovascular	10	11	14	17	20	24	27	70	1	17
Weather										
Daily mean temperature (°C)	3.8	5.9	9.1	13.2	18.21	22.19	24.35	32.2	-3.2	13.68
Daily mean rel. humidity (%)	61.28	64.3	69.7	78.0	65.7	91.5	94.12	100.0	46.6	77.24
Daily dew point temperature (°C)	-2.3	-0.64	2.1	5.8	9.3	11.74	12.9	17.6	-7.7	5.64
Air pollutants										
PM_{10} (μm^3)	10	12	15	20	29	41	51	131	0	24.2
NO_2 (μm^3)	17.82	21.78	33.66	45.54	61.38	73.26	81.18	192.1	7.92	47.9
SO_2 (μm^3)	5.52	5.52	9.66	16.56	27.6	44.16	60.72	212.5	0	21.9
CO (ppm)	0.3	0.3	0.4	0.5	0.7	1.0	1.15	6.1	0.1	0.62
O_3 (μm^3)	6.21	9.32	18.63	34.16	46.58	57.96	66.24	120.1	0	34.09

Figure 35: Distribution of Counts of Daily Mortality in Birmingham.



8.5.2 Bristol

The Bristol monitoring station is located in an urban thoroughfare near a city centre shopping area. The nearest road is a busy urban dual carriageway. The manifold inlet is approximately 3 metres above ground level. The surrounding area is generally open and comprises retail business premises.

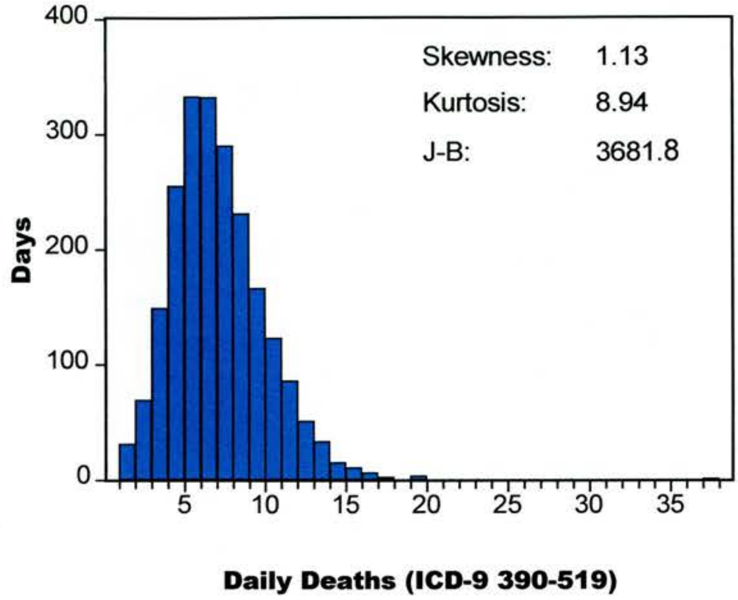
Table 65: Key Figures for Bristol.

Area (km^2)	110
Persons per km^2	3,622
Male Population (thousands)	198
Female Population (thousands)	201
Total Population (thousands)	400

Table 66: Distribution of Mortality, Weather and Air Pollution in Bristol.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All causes	6	7	9	11	14	16	18	48	1	12
Respiratory	1	1	1	2	3	4	4	17	1	2
Cardiovascular	2	2	3	5	7	8	9	20	1	5
Respiratory & Cardiovascular	3	3	5	6	8	10	12	37	1	7
Weather										
Daily mean temperature (°C)	3.055	4.9	7.5	11.1	15.5	18.2	19.7	27	-3.1	11.37
Daily mean rel. humidity (%)	57.4	62	69.8	77.15	84	88.89	91.45	97.5	40.9	76.23
Daily dew point temperature (°C)	-1.2	0.6	3.5	7.1	10.4	12.7	13.9	18.6	-10.8	6.84
Air pollutants										
PM_{10} (μm^3)	11	13	15	21	30	42	50	96	5	24.17
NO_2 (μm^3)	19.8	23.76	31.68	45.54	61.38	77.22	85.14	128.7	9.9	46.96
SO_2 (μm^3)	5.52	5.52	8.28	13.8	22.08	30.36	35.88	96.6	0	15.36
CO (ppm)	0.2	0.3	0.4	0.6	0.8	1.2	1.5	3.2	0.1	0.65
O_3 (μm^3)	8.28	10.35	21.74	35.19	49.68	60.03	64.17	109.7	2.07	34.23

Figure 36: Distribution of Counts of Daily Deaths in Bristol.



8.5.3 Cardiff

The monitoring station in Cardiff is located within a pedestrianised area in Frederick Street, City Centre. The nearest road is some 200 metres distance with a flow of 27,000 vehicles per day and is subject to periodic congestion during peak periods. The manifold inlet is located approximately 3 metres above ground level. The surrounding area comprises retail and business premises.

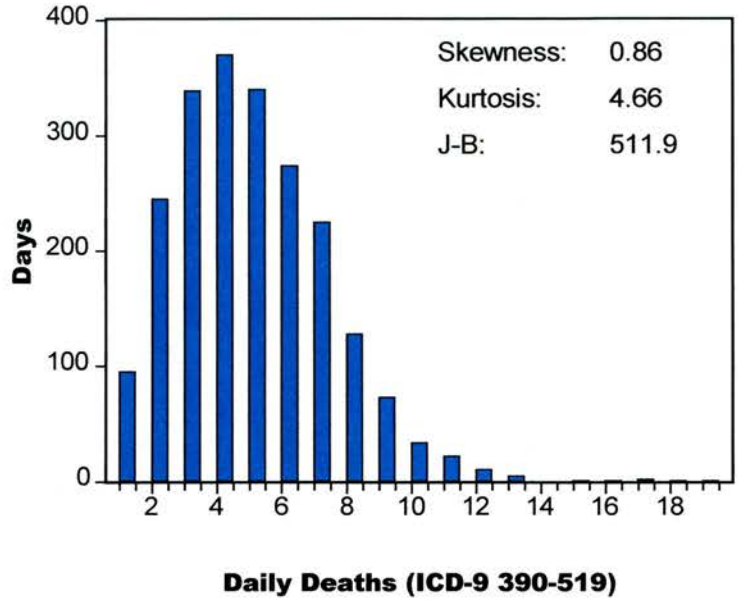
Table 67: Key Figures for Cardiff.

Area (km^2)	140
Persons per km^2	2,225
Male Population (thousands)	155
Female Population (thousands)	160
Total Population (thousands)	315

Table 68: Distribution of Mortality, Weather and Air Pollution in Cardiff.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All Causes	4	5	6	8	10	12	14	27	1	9
Respiratory	1	1	1	2	2	3	4	10	1	2
Cardiovascular	1	2	2	3	5	6	7	14	1	4
Respiratory & Cardiovascular	2	2	3	5	6	8	9	19	1	5
Weather										
Daily mean temperature (°C)	3.335	5.3	7.9	11.3	15.53	18.3	19.9	27	-2.1	10.73
Daily mean rel. humidity (%)	56.9	62.0	69.7	76.8	84.1	89.8	92.16	99.8	37.5	76.06
Daily dew point temperature (°C)	-1.0	0.5	3.5	7.1	10.5	12.8	14.0	18.5	-10.0	6.89
Air pollutants										
PM_{10} (μm^3)	13	15	18	23	32	41	49	91	7	25.49
NO_2 (μm^3)	21.78	25.74	33.66	43.56	51.48	61.38	67.32	103.0	5.94	42.54
SO_2 (μm^3)	5.52	5.52	8.28	11.04	16.56	27.6	33.12	129.7	0	14.03
CO (ppm)	0.3	0.3	0.4	0.6	0.7	1.0	1.3	3.9	0.0	0.63
O_3 (μm^3)	6.21	10.35	18.63	31.05	43.47	55.89	66.24	132.5	2.07	32.45

Figure 37: Distribution of Counts of Daily Mortality in Cardiff.



8.5.4 Edinburgh

The monitoring station in Edinburgh is located in Princes Street Gardens East, Princes Street. The nearest road is approximately 35 metres from the monitoring station. The manifold inlet is 3 metres above ground level. The park is mainly grassed with many mature trees, the nearest of which is within 5 metres of the monitoring station.

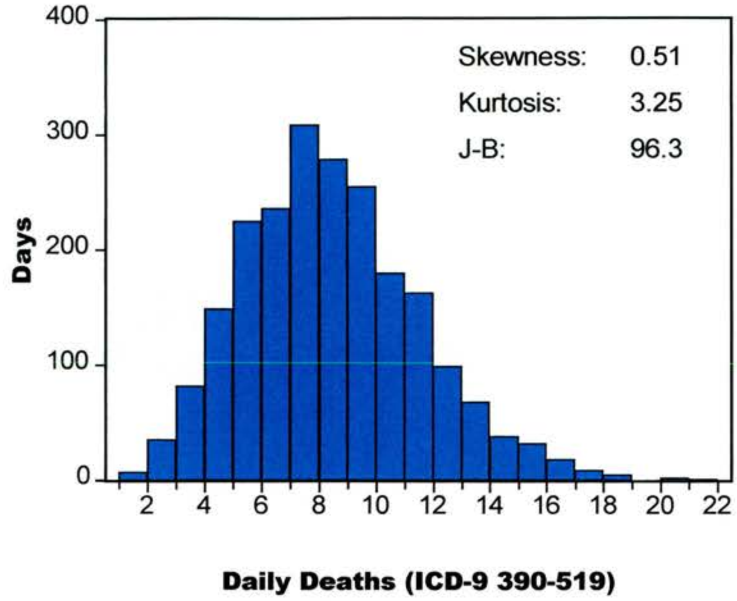
Table 69: Key Figures for Edinburgh.

Area (km^2)	262
Persons per km^2	1,711
Male Population (thousands)	217
Female Population (thousands)	232
Total Population (thousands)	449

Table 70: Distribution of Mortality, Weather and Air Pollution in Edinburgh.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All causes	8	9	11	14	17	20	22	34	3	15
Respiratory	1	1	1	2	3	4	4	9	1	2
Cardiovascular	2	3	4	6	8	10	11	18	1	6
Respiratory & Cardiovascular	3	4	6	8	10	12	13	21	1	8
Weather										
Daily mean temperature (°C)	1.1	2.5	4.895	8.7	12.9	15.1	16.1	21.1	-7.7	8.77
Daily mean rel. humidity (%)	65.26	68.32	74.1	79.6	85.0	90.49	92.6	99.6	52.8	79.5
Daily dew point temperature (°C)	-2.1	-0.8	1.8	5.5	9	11.3	12.4	17.1	-13.8	5.34
Air pollutants										
PM_{10} (μm^3)	10	12	14	18	23	31	37	72	5	20.05
NO_2 (μm^3)	29.7	33.66	41.58	54.48	59.4	69.3	77.22	134.6	11.88	49.92
SO_2 (μm^3)	2.76	5.52	8.28	13.8	22.08	33.12	44.16	138	0	17.64
CO (ppm)	0.3	0.3	0.4	0.5	0.7	0.9	1.0	2.5	0.0	0.57
O_3 (μm^3)	8.28	12.42	18.63	28.98	39.33	49.68	53.82	84.87	2.07	29.52

Figure 38: Distribution of Counts of Daily Mortality in Edinburgh.



8.5.5 Glasgow

Glasgow has two monitoring stations. One is on the second floor, to the rear, of the City Chambers building at the junction of Montrose Street and Cochrane Street. Both streets are subject to frequent congestion. The surrounding area is urban and comprises street canyons with retail and business outlets. The second site is located in a pedestrianised area in the city centre. The nearest road is situated 10 metres from the site with a busy commercial thoroughfare approximately 20 metres distance from the monitoring station. Traffic flow on the thoroughfare is approximately 20,000 vehicles per day. The manifold inlet is approximately 3 metres high. The surrounding area is open with city centre business and retail premises surrounding the site on three sides.

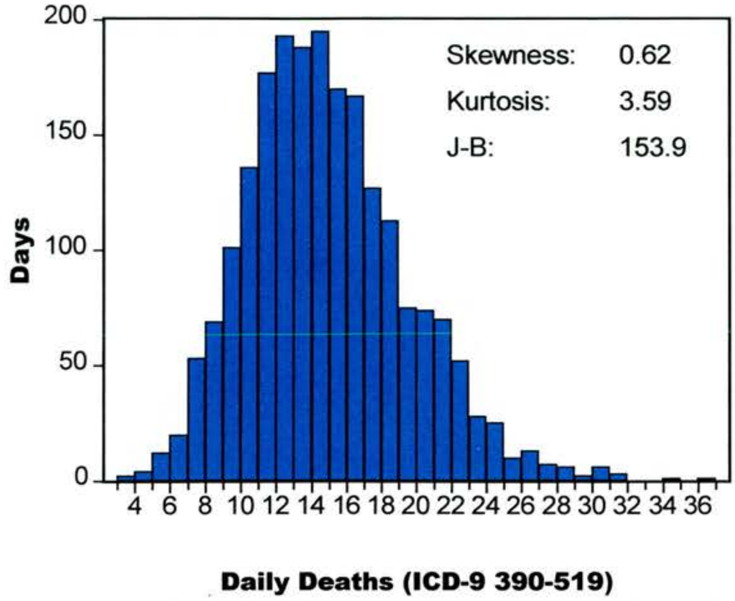
Table 71: Key Figures for Glasgow.

Area (km^2)	175
Persons per km^2	3,522
Male Population (thousands)	294
Female Population (thousands)	322
Total Population (thousands)	616

Table 72: Distribution of Mortality, Weather and Air Pollution in Glasgow.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All causes	17	19	22	26	30	34	37	53	4	26
Respiratory	1	1	2	3	4	6	7	14	1	4
Cardiovascular	6	7	9	11	14	16	18	28	2	11
Respiratory & Cardiovascular	8	9	11	14	17	21	22	36	3	15
Weather										
Daily mean temperature (°C)	0.73	2.5	4.95	8.9	12.9	15.1	16.37	22.3	-14.3	7.36
Daily mean rel. humidity (%)	65.5	69.3	75.6	81.7	86.6	90.49	92.5	99.5	46.2	80.61
Daily dew point temperature (°C)	-1.9	-0.5	2.1	5.8	9.2	11.3	12.65	17.4	-19.6	5.58
Air pollutants										
PM_{10} (μm^3)	10	12	15	19	25	33	38.7	85	7	14.64
NO_2 (μm^3)	23.76	29.7	39.6	51.48	63.36	73.26	81.18	196.1	0	51.06
SO_2 (μm^3)	2.76	5.52	5.52	8.28	11.04	19.32	24.84	49.68	2.76	7.19
CO (ppm)	0.3	0.4	0.5	0.7	1.0	1.5	1.8	4.5	0.1	0.83
O_3 (μm^3)	4.14	6.21	14.49	26.91	37.26	47.61	55.89	76.59	2.07	18.99

Figure 39: Distribution of Counts of Daily Mortality in Glasgow.



8.5.6 Kingston upon Hull

The Kingston upon Hull monitoring station is located in a pedestrianised area within the city centre at Vernon Street. The nearest urban road is approximately 6 metres from the monitoring station with a 12-hour traffic flow of approximately 7000 vehicles. The manifold inlet is approximately 3 metres above ground level. The surrounding area is generally open and comprises gardens laid to lawn.

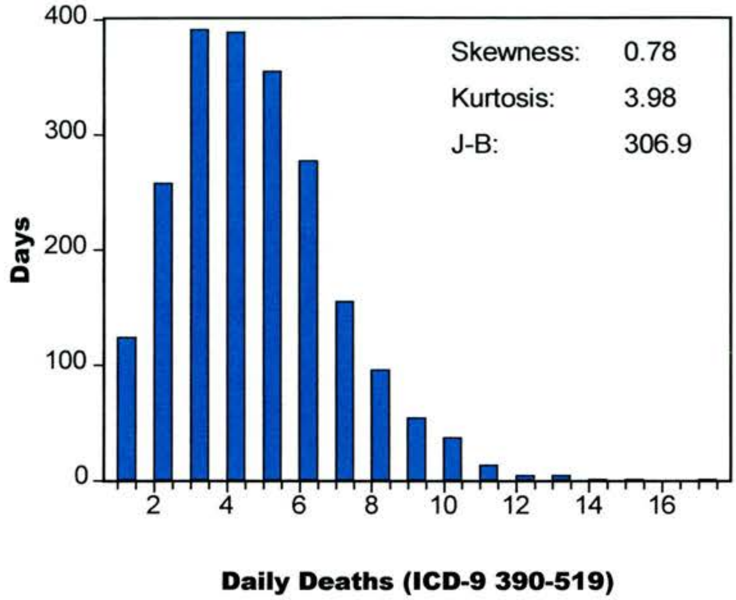
Table 73: Key Figures for Kingston upon Hull.

Area (km^2)	71
Persons per km^2	3,740
Male Population (thousands)	132
Female Population (thousands)	134
Total Population (thousands)	267

Table 74: Distribution of Mortality, Weather and Air Pollution in Kingston upon Hull.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All causes	3	4	6	8	10	12	13	19	1	8
Respiratory	1	1	1	2	2	3	4	7	1	2
Cardiovascular	1	1	2	3	4	6	6	13	1	3
Respiratory & Cardiovascular	1	2	3	4	6	7	9	17	1	5
Weather										
Daily mean temperature (°C)	1.5	2.7	5.5	9.6	13.7	16.3	17.99	22.4	-7.7	8.94
Daily mean rel. humidity (%)	67.67	70.2	75.0	81.0	88.3	93.2	95.7	100.0	53.3	80.23
Daily dew point temperature (°C)	-0.9	0.2	2.6	6.2	9.8	12.2	13.6	17.9	-10.1	6.15
Air pollutants										
PM_{10} (μm^3)	12	14	17	22	29	40	48	169	7	24.24
NO_2 (μm^3)	21.76	27.12	33.66	43.56	53.46	63.36	71.28	101.0	9.9	43.28
SO_2 (μm^3)	5.52	8.28	11.04	19.32	27.6	38.64	46.92	110.4	2.76	21.09
CO (ppm)	0.3	0.3	0.4	0.5	0.7	1.0	1.2	2.8	0.1	0.58
O_3 (μm^3)	8.28	10.35	22.26	31.05	43.47	55.89	62.1	113.9	2.07	32.48

Figure 40: Distribution of Counts of Daily Mortality in Kingston upon Hull.



8.5.7 Leeds

The monitoring station in Leeds is located approximately 30 metres from a busy 4-lane inner-city road at Queen Square Court. Traffic flow is approximately 21,500 vehicles per day and is subject to periodic congestion during peak periods. The location is approximately 150 metres from an urban motorway with traffic flow of approximately 93,500 vehicles per day. The manifold inlet is approximately 4 metres high. The surrounding area is generally open and comprises a busy urban setting.

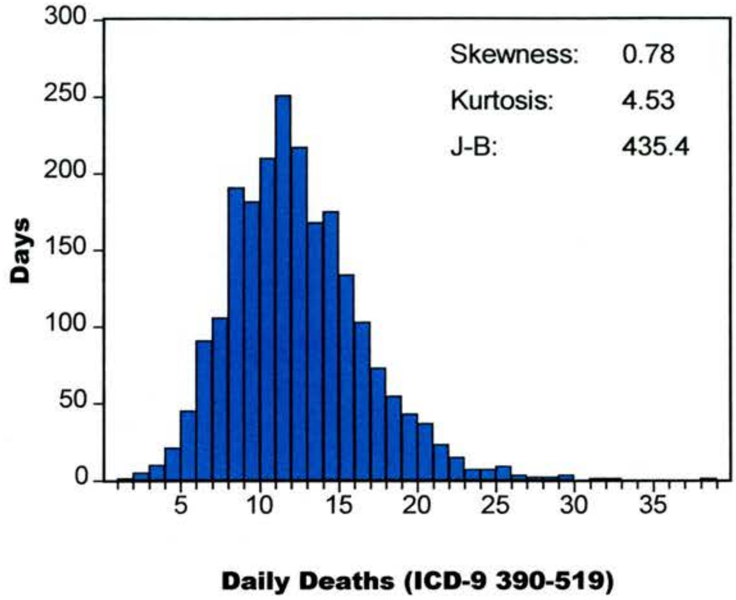
Table 75: Key Figures for Leeds.

Area (km^2)	562
Persons per km^2	1,294
Male Population (thousands)	359
Female Population (thousands)	368
Total Population (thousands)	727

Table 76: Distribution of Mortality, Weather and Air Pollution in Leeds.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All causes	13	14	17	20	24	28	30	64	4	21
Respiratory	1	1	2	3	4	6	7	16	1	4
Cardiovascular	4	5	6	8	11	13	14	28	1	9
Respiratory & Cardiovascular	6	7	9	11	14	17	20	38	1	12
Weather										
Daily mean temperature (°C)	1.8	2.9	5.85	10.1	14.45	17.4	19.25	24.9	-1.1	10.21
Daily mean rel. humidity (%)	61.3	63.9	69.0	76.3	83.9	90.2	93.35	100.0	46.2	76.56
Daily dew point temperature (°C)	-1.7	-0.4	2.2	5.7	9.3	11.6	12.8	17.3	-6.9	5.71
Air pollutants										
PM_{10} (μm^3)	12	13	17	23	33	47	58	151	3	25.84
NO_2 (μm^3)	29.7	33.66	41.58	51.48	63.36	75.24	83.16	144.5	11.88	52.82
SO_2 (μm^3)	2.76	5.52	8.28	13.8	27.6	46.92	66.24	303.6	0	21.66
CO (ppm)	0.3	0.3	0.5	0.6	0.9	1.1	1.3	4.1	0.2	0.71
O_3 (μm^3)	6.21	8.28	16.56	26.91	39.33	47.61	55.89	99.36	2.07	28.01

Figure 41: Distribution of Counts of Daily Mortality in Leeds.



8.5.8 Leicester

The Leicester monitoring station at the New Walk Centre, Welford Place, is located in a pedestrian piazza between eight and eleven-storey council offices. It is situated approximately 30 metres from a three lane one-way road which is subject to congestion at peak times. Traffic flow is approximately 14,500 vehicles per day. The manifold is approximately 3 metres above ground level. The immediate area in the vicinity of the manifold inlet is open.

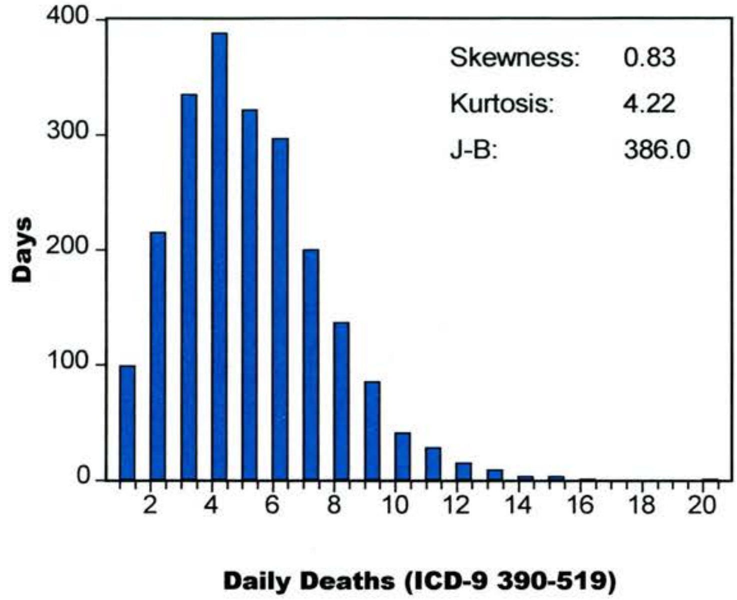
Table 77: Key Figures for Leicester.

Area (km^2)	73
Persons per km^2	4,021
Male Population (thousands)	146
Female Population (thousands)	148
Total Population (thousands)	295

Table 78: Distribution of Mortality, Weather and Air Pollution in Leicester.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All causes	4	5	6	8	10	13	14	23	1	9
Respiratory	1	1	1	2	2	3	4	9	1	2
Cardiovascular	1	1	2	3	5	6	7	13	1	4
Respiratory & Cardiovascular	2	2	3	5	6	8	9	20	1	5
Weather										
Daily mean temperature (°C)	0.4	2.3	5.0	9.1	13.6	16.5	18.3	24.2	-7.9	9.27
Daily mean rel. humidity (%)	64.6	68.3	74.35	82.0	89.3	94.3	96.3	100.0	42.3	78.11
Daily dew point temperature (°C)	-1.8	-0.5	2.3	6.0	9.5	12.0	13.2	18.1	-8.2	5.61
Air pollutants										
PM_{10} (μm^3)	9.4	11	14	18	25	34	41	84	2	21.48
NO_2 (μm^3)	21.78	23.76	33.17	43.56	53.46	63.36	73.26	116.8	7.92	43.38
SO_2 (μm^3)	5.76	5.76	8.52	14.04	22.32	33.36	41.64	121.7	3	17.98
CO (ppm)	0.2	0.3	0.3	0.5	0.7	0.9	1.2	3.5	0.1	0.54
O_3 (μm^3)	6.21	8.28	18.63	33.12	45.54	60.03	68.31	118.0	0	33.13

Figure 42: Distribution of Counts of Daily Mortality in Leicester.



8.5.9 Liverpool

Liverpool's monitoring station at St John's Gardens, St John's Lane is located within a park in central Liverpool. The park is encircled by a busy 2-4 lane urban road some 20 metres from the site. The manifold inlet is approximately 3.5 metres high. The surrounding area is urban, comprising retail and business outlets.

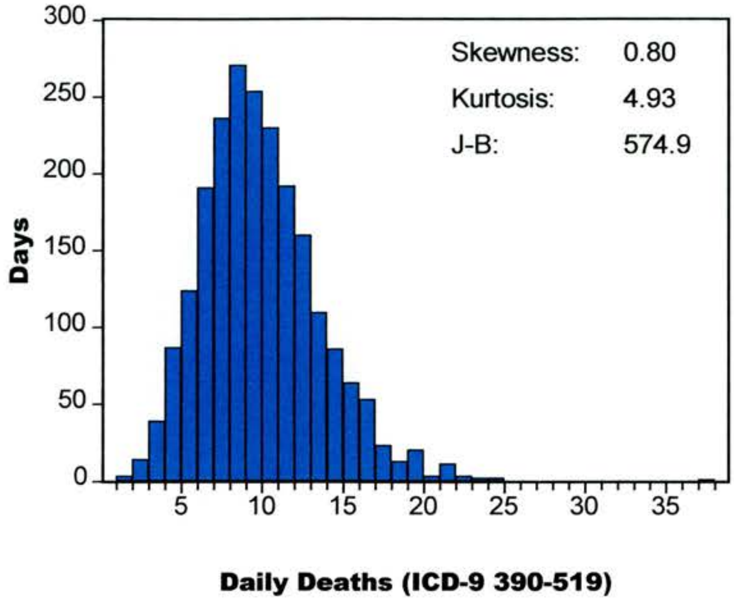
Table 79: Key Figures for Liverpool.

Area (km^2)	113
Persons per km^2	4,148
Male Population (thousands)	228
Female Population (thousands)	240
Total Population (thousands)	468

Table 80: Distribution of Mortality, Weather and Air Pollution in Liverpool.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All causes	9	10	13	16	19	22	24	57	4	16
Respiratory	1	1	2	3	4	6	7	13	1	3
Cardiovascular	3	3	5	6	8	10	11	24	1	7
Respiratory & Cardiovascular	4	5	7	9	12	14	16	37	1	10
Weather										
Daily mean temperature (°C)	1.75	3.6	6.3	9.9	14.1	16.3	17.7	25.0	-6.3	9.8
Daily mean rel. humidity (%)	68.4	71.1	76.6	82.4	88.1	92.6	95.25	99.9	50.9	81.12
Daily dew point temperature (°C)	-0.8	0.8	3.8	7.0	10.6	12.9	14.0	18.4	-6.4	6.93
Air pollutants										
PM_{10} (μm^3)	11	13	17	22	32	43	52	163	1	25.41
NO_2 (μm^3)	17.82	21.78	31.68	47.52	63.36	77.22	86.33	140.6	7.92	48.54
SO_2 (μm^3)	2.76	5.52	8.28	19.32	38.64	66.24	82.8	218.1	0	27.96
CO (ppm)	0.2	0.2	0.3	0.4	0.6	0.8	1.0	1.9	0.0	0.46
O_3 (μm^3)	6.21	8.28	18.63	35.19	47.61	57.96	66.24	95.22	0	33.6

Figure 43: Distribution of Counts of Daily Mortality in Liverpool.



8.5.10 London

The pollution data for London has been compiled by combining the respective data of various monitoring sites. The monitoring sites from which pollution data was obtained include:

- West London
- London Bridge Place
- London Cromwell Road
- London Bloomsbury
- London Bexley
- London Marylebone Road
- London Southwark Roadside
- London Hounslow
- London Brent
- London Hillingdon
- London Teddington
- Camden Kerbside
- London Kensington
- London Hackney
- London Haringey
- London Lewisham
- London Haringey Roadside
- London Greenwich
- London Sutton Roadside
- London Sutton

At any one time, pollution data from at least four monitoring stations was available. Obviously, with the expansion of the air pollution-monitoring network, the number of monitoring sites and hence the availability of pollution data increased rapidly year to year with 11 in 1995, 19 in 1996 and 22 in 1997. The stations include all types of monitoring site ranging from suburban, urban background urban, centre, roadside, to kerbside.[24] As an example, the urban centre monitoring site London Bloomsbury shall be described at this point. The London Bloomsbury monitoring station is located within the south-east corner of Russell Square Gardens, a central London gardens. All four sides of the gardens are surrounded by a busy 2/4-lane one-way road system with some 35,000 vehicles per day, which is subject to frequent congestion. The nearest road lies at a distance of ca. 35 metres from the monitoring station.

Table 81: Key Figures for London.

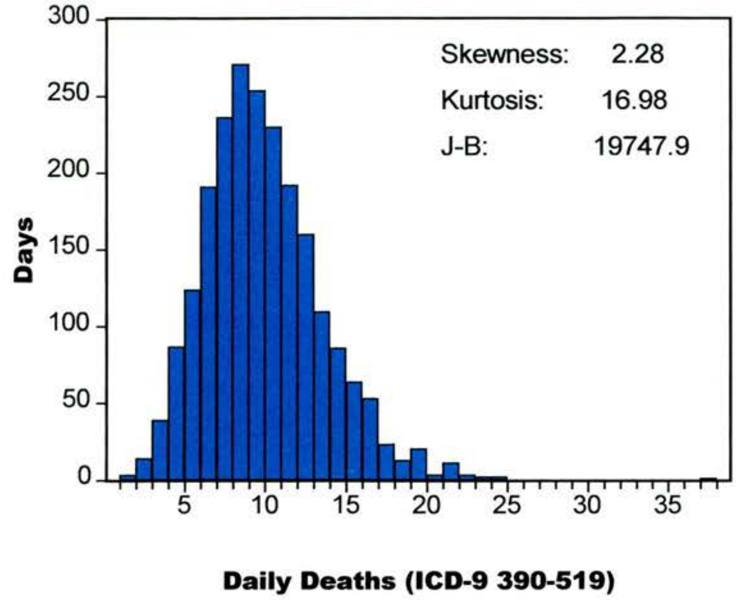
Area (km^2)	1,578
Persons per km^2	4,448.2
Male Population (thousands)	3,475
Female Population (thousands)	3,599
Total Population (thousands)	7,074

[24] For a detailed overview of the definitions of the various monitoring site classes see: Broughton et al. (1997), pp. 437-440 and Department of the Environment, Transport and the Regions website at http://www.environment.detr.gov.uk.

Table 82: Distribution of Mortality, Weather and Air Pollution in London.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All causes	143	149	161	176	195	216	234	572	42	181
Respiratory	15	18	23	28	35	45	55	188	3	31
Cardiovascular	53	57	63	72	81	91	97	190	17	73
Respiratory & Cardiovascular	75	80	88	99	114	132	146	378	20	104
Weather										
Daily mean temperature (°C)	3.7	5.2	7.9	11.9	16.03	19.3	20.7	26.5	-2.3	12.01
Daily mean rel. humidity (%)	52.4	56.5	62.7	70.1	77.4	82.6	85.2	95.1	32.8	69.76
Daily dew point temperature (°C)	-1.8	0.1	2.68	6.1	9.7	12.1	13.3	17.7	-7.9	6.09
Air pollutants										
PM_{10} (μm^3)	13.8	15.5	18.5	24	33	46	55	100	5.5	27.79
NO_2 (μm^3)	38.6	42.9	51.49	63.36	77.72	93.7	106.9	243.5	19.16	66.72
SO_2 (μm^3)	5.91	7.37	11.04	17.94	29.67	46.21	60.11	213.4	2.02	23.74
CO (ppm)	0.48	0.54	0.7	0.93	1.33	1.73	2.08	5.65	0.23	1.08
O_3 (μm^3)	5.4	7.07	13.46	23.81	35.71	46.58	53.59	97.98	2.07	25.79

Figure 44: Distribution of Counts of Daily Mortality in London.



8.5.11 Newcastle upon Tyne

The monitoring station in Newcastle is situated at Newcastle Civic Centre, St Mary's Place. It is located approximately 20 metres away from a major through road of the city. The manifold inlet is approximately at road level due to the downhill nature of the site. The surrounding area is generally open with a private car park and a grassed area in the immediate vicinity of the monitoring station.

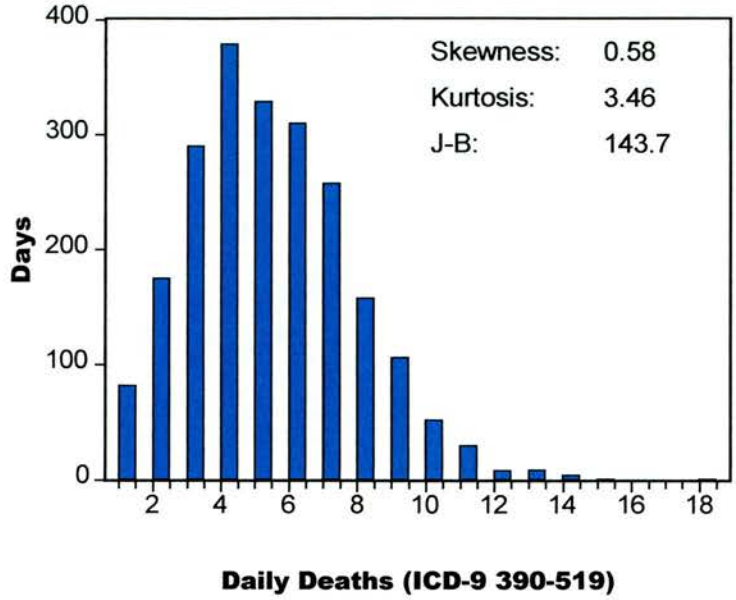
Table 83: Key Figures for Newcastle upon Tyne.

Area (km^2)	112
Persons per km^2	2,526
Male Population (thousands)	139
Female Population (thousands)	143
Total Population (thousands)	282

Table 84: Distribution of Mortality, Weather and Air Pollution in Newcastle upon Tyne.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All causes	4	5	7	9	11	13	15	30	1	9
Respiratory	1	1	1	2	2	3	4	8	1	2
Cardiovascular	1	2	2	4	5	7	7	12	1	4
Respiratory & Cardiovascular	2	2	4	5	7	8	9	18	1	5
Weather										
Daily mean temperature (°C)	2.1	3.3	5.8	9.5	13.6	16.1	17.7	23.1	-3.5	9.66
Daily mean rel. humidity (%)	60.5	63.3	68.3	74.7	82.4	88.8	91.8	98.3	47.2	75.4
Daily dew point temperature (°C)	-2.2	-0.9	1.5	5.0	8.63	11.0	12.3	16.6	-8.2	5.06
Air pollutants										
PM_{10} (μm^3)	11	13	16	21	30	41.2	49.6	141	6	24.48
NO_2 (μm^3)	21.78	25.74	33.66	45.54	57.42	69.3	75.24	106.9	7.92	46.38
SO_2 (μm^3)	5.52	5.52	8.28	13.8	22.08	35.88	46.92	173.9	0	17.57
CO (ppm)	0.3	0.3	0.4	0.6	0.8	1.0	1.2	3.0	0.1	0.63
O_3 (μm^3)	6.21	12.42	20.7	33.12	43.47	55.89	62.1	97.29	2.07	32.52

Figure 45: Distribution of Counts of Daily Mortality in Newcastle upon Tyne.



8.5.12 Sheffield

In Sheffield, two monitoring sites have been considered. The monitoring station at Charter Square is located approximately 20 metres from a busy urban road. Traffic flow is approximately 20,000 vehicles per day. The manifold inlet is approximately 20 metres from kerbside and at a height of 3 metres. The other site is located at Tinsley Community Centre, Inman Lane, Tinsley. This station is about 200 metres from the M1 motorway. The manifold is approximately 3 metres in height. The surrounding area is generally open, and comprises of residential/light industrial premises.

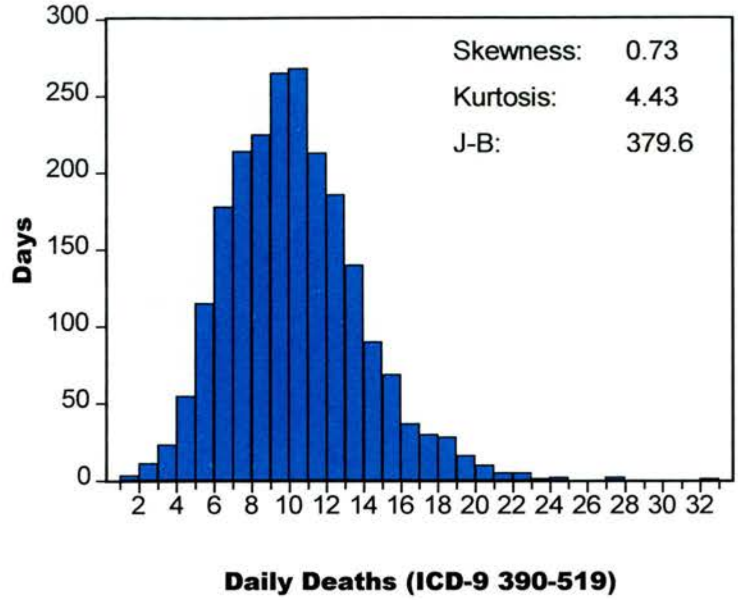
Table 85: Key Figures for Sheffield.

Area (km^2)	367
Persons per km^2	1,444
Male Population (thousands)	264
Female Population (thousands)	267
Total Population (thousands)	530

Table 86: Distribution of Mortality, Weather and Air Pollution in Sheffield.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All causes	10	12	14	17	20	23	25	52	3	17
Respiratory	1	1	1	2	4	5	6	16	1	3
Cardiovascular	3	4	5	7	9	11	12	18	1	8
Respiratory & Cardiovascular	5	6	7	10	12	14	16	32	1	10
Weather										
Daily mean temperature (°C)	1.3	2.9	5.8	9.7	14.0	16.9	18.6	24.1	-4.3	9.82
Daily mean rel. humidity (%)	0	5	59	76	89	97	100	100	40.9	81.08
Daily dew point temperature (°C)	-1.1	0.18	2.65	6.7	10.25	12.6	14.01	18.2	-7.6	4.33
Air pollutants										
PM_{10} (μm^3)	11	13	16.75	23	32	44	54	102	6	24.79
NO_2 (μm^3)	26.73	31.68	41.58	55.44	67.32	79.2	87.12	158.4	0	55.91
SO_2 (μm^3)	5.52	8.28	11.04	19.32	33.12	47.2	57.96	132.5	0	23.51
CO (ppm)	0.2	0.2	0.3	0.5	0.7	1.0	1.4	6.0	0.1	0.58
O_3 (μm^3)	8.28	10.35	18.63	33.12	43.47	53.82	60.03	101.4	2.07	30.8

Figure 46: Distribution of Counts of Daily Mortality in Sheffield.



8.5.13 Southampton

The air monitoring station in Southampton is located at the junction of a residential road and a six-lane dual carriageway. Traffic flow is approximately 25,000 vehicles per day and is subject to periodic congestion during peak traffic flow periods. The manifold inlet is approximately 10 metres from the residential road kerbside and approximately 3 metres high. The surrounding area is mainly urban and comprises urban residential premises.

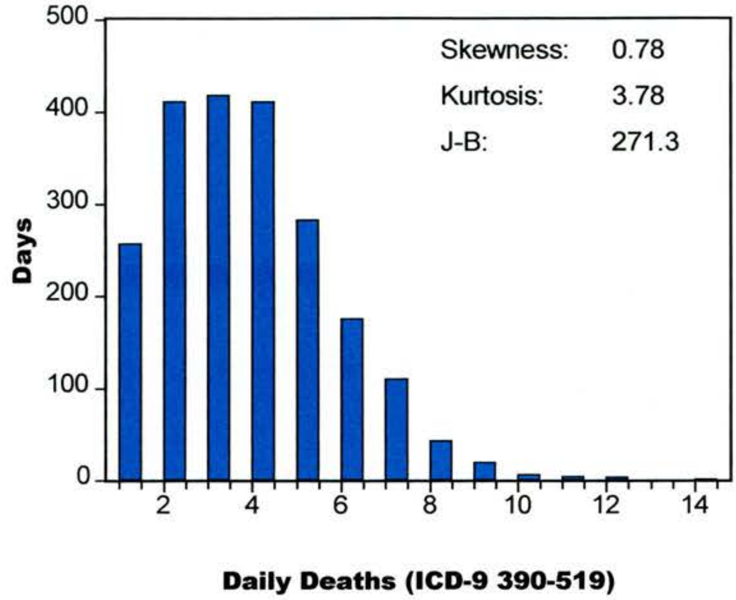
Table 87: Key Figures for Southampton.

Area (km²)	50
Persons per km²	4,311
Male Population (thousands)	108
Female Population (thousands)	107
Total Population (thousands)	215

Table 88: Distribution of Mortality, Weather and Air Pollution in Southampton.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All causes	2	3	4	6	8	9	11	24	1	6
Respiratory	1	1	1	1	2	3	3	8	1	2
Cardiovascular	1	1	2	3	4	5	6	10	1	3
Respiratory & Cardiovascular	1	1	2	3	5	6	7	14	1	4
Weather										
Daily mean temperature (°C)	3.1	4.7	7.5	11.3	15.7	18.5	20.0	25.9	-2.9	11.46
Daily mean rel. humidity (%)	58.06	62.01	69.9	77.9	85.4	90.8	93.16	99.9	38.8	77.2
Daily dew point temperature (°C)	-1.45	0.6	3.6	7.3	10.9	13.3	14.45	18.5	-7.6	7.1
Air pollutants										
PM_{10} (μm^3)	11.55	13	16	20.5	28	37	44	82	5	22.55
NO_2 (μm^3)	23.76	27.72	33.66	43.56	55.44	67.9	77.22	116.8	11.88	45.2
SO_2 (μm^3)	2.76	5.52	5.52	8.28	13.8	19.32	24.84	57.96	0	10.71
CO (ppm)	0.3	0.3	0.5	0.6	0.9	1.3	1.6	4.3	0.1	0.73
O_3 (μm^3)	6.21	10.35	18.63	33.12	45.54	55.89	62.1	93.15	2.07	31.66

Figure 47: Distribution of Counts of Daily Mortality in Southampton.



8.5.14 Swansea

The Swansea monitoring station at Princess Way is located in a pedestrianised area. The site is approximately 30 - 40 metres from a busy dual carriageway. The manifold inlet is approximately 3 metres high. The surrounding area comprises urban business and office premises in a pedestrian area.

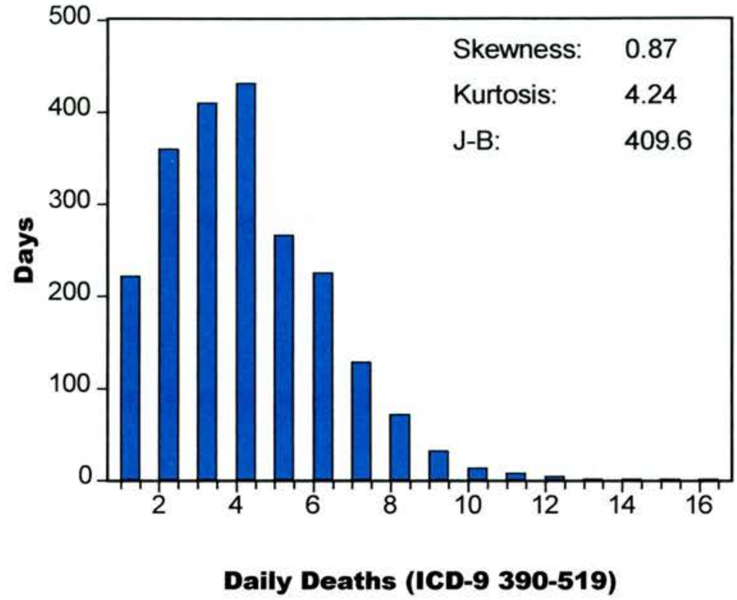
Table 89: Key Figures for Swansea.

Area (km^2)	378
Persons per km^2	609
Male Population (thousands)	113
Female Population (thousands)	117
Total Population (thousands)	230

Table 90: Distribution of Mortality, Weather and Air Pollution in Swansea.

Variable	Percentile (%)							Max.	Min.	Mean
	5	10	25	50	75	90	95			
Mortality (deaths/day)										
All causes	3	3	5	6	8	10	12	27	1	7
Respiratory	1	1	1	1	2	3	3	8	1	2
Cardiovascular	1	1	2	3	4	5	6	14	1	3
Respiratory & Cardiovascular	1	1	2	4	5	7	8	16	1	4
Weather										
Daily mean temperature (°C)	2.5	4.1	6.7	10.1	14.5	16.9	18.1	25.2	-3.6	10.36
Daily mean rel. humidity (%)	63.9	68.41	75.4	81.6	88.2	93	95.2	100	40.9	81.08
Daily dew point temperature (°C)	-1.2	0.6	3.7	7.2	10.4	12.8	13.9	18.5	-8.6	6.93
Air pollutants										
PM_{10} (μm^3)	11	13	17	22	30	39	46	83	4	24.1
NO_2 (μm^3)	15.84	21.78	31.68	41.58	53.46	61.38	69.3	95.04	5.94	41.4
SO_2 (μm^3)	2.76	2.76	5.52	11.04	24.84	38.64	44.18	102.1	0	16.43
CO (ppm)	0.2	0.2	0.3	0.4	0.6	0.9	1.18	2.1	0	0.49
O_3 (μm^3)	10.35	14.49	24.84	39.33	51.75	66.24	74.52	134.6	2.07	38.79

Figure 48: Distribution of Counts of Daily Mortality in Swansea.



8.6 Summary

Examining the distribution data for daily deaths, it appears that the different cities can be divided into three groups. Group one would include Bristol, Cardiff, Edinburgh, Kingston upon Hull, Newcastle upon Tyne, Leicester, Southampton and Swansea, representing the lower range of daily deaths for total mortality as well as cardiovascular and respiratory causes. The second group of 'moderate' daily counts of deaths would include Liverpool and Sheffield. Finally, the upper range of daily deaths would include the cities of Birmingham, Glasgow, Leeds and, with by far the highest numbers, London. However, these differences may just be population related, rather than being linked by other factors. Consequently, the daily means of the mortality counts are adjusted to the respective city population size and summarised in the table below. Since, this study is concentrating on the cause specific mortality data of respiratory and cardiovascular diseases the figures presented in the following table represent the population adjusted figures (per 100,000) for the combination of these variables as described earlier.[25]

Table 91: Population Adjusted Mean of Daily Deaths (Respiratory & Cardiovascular Diseases).

	Mean daily deaths (Respiratory & Cardiovascular Diseases)
Birmingham	1.7
Bristol	1.7
Cardiff	1.6
Edinburgh	1.7
Glasgow	2.4
Kingston upon Hull	1.8
Leeds	1.7
Leicester	1.7
Liverpool	2.1
London	1.5
Newcastle upon Tyne	1.8
Sheffield	1.9
Southampton	1.9
Swansea	1.7

After adjusting the mortality data for population size, the originally apparent pattern cannot be confirmed any longer. It appears that the average number of people dying due to respiratory/ cardiovascular diseases per day lies at 1.8 for the included cities. This relatively low number already indicates that air pollution in fact may only be responsible for a small fraction of daily deaths.

Although there is no significant variation for the majority of the cities, the figures for Glasgow and Liverpool are clearly above the average figures. The high figure for Glasgow is not surprising since Scotland shows the highest age-

[25] Obviously, mortality rates may also vary with age and should ideally also be adjusted for the respective age structure. However, due to the lack of adequate data, no such adjustment could be done.

adjusted mortality rates for circulatory diseases as well as the highest incidence of lung cancer.[26] This is also confirmed when looking at Glasgow's standardised mortality ratio (SMR) for 1996, which with 137 is one of the highest in the United Kingdom, the baseline being 100.[27] A similar high SMR can be observed for Liverpool, which ranks with 125 well above the SMR of the remaining cities included in this study. Finally, it has to be noted that the figure for London is below the average number. However, this is well in line with the SMR calculated for London with 97. Also, the proportion of the age group 16 - 44 in London is, with 46.1 percent, by far the largest in the UK with an average of 41 percent. In contrast, the age groups of 65-79 and 80 or over, which may be most susceptible for both respiratory and cardiovascular disease, are significantly under-represented in London.[28]

Like the health data, the data for the different air pollutants included in this study are also summarised. The figures in the table below show the averages for the respective pollutants for each city for the time period from 1992 to 1997 inclusive.

---

[26] See: ONS (1998b), pp. 99-100. This is also confirmed when looking at Glasgow City's standardised mortality ratio.

[27] The SMR compares the overall mortality in a region (city) with that for the United Kingdom. The resulting ratio expresses the number of deaths in a region (city) as a percentage of the hypothetical number that would have occurred if the region's (city's) population had experienced the sex/age-specific rates of the United Kingdom in that year (here 1996). See: ONS (1998b), p. 227.

[28] See: ONS (1998b), p. 45.

Table 92: Summary of Air Pollution Data.

	PM_{10}	NO_2	SO_2	CO	O_3
Birmingham	24.20	47.9	21.9	0.62	34.09
Bristol	24.17	46.96	15.36	0.65	34.23
Cardiff	25.49	42.65	14.03	0.63	32.45
Edinburgh	20.05	49.62	17.64	0.57	29.52
Glasgow	14.64	51.06	7.19	0.83	18.99
Kingston upon Hull	24.24	43.28	21.09	0.58	32.48
Leeds	25.84	52.82	21.66	0.71	28.01
Leicester	21.48	43.38	17.98	0.54	33.13
Liverpool	25.41	48.54	27.96	0.46	33.60
London	27.79	66.72	23.74	1.08	25.79
Newcastle upon Tyne	24.48	46.38	17.57	0.63	32.52
Sheffield	24.79	55.91	23.51	0.58	30.80
Southampton	22.55	45.2	10.71	0.73	31.66
Swansea	24.10	41.4	16.43	0.49	38.79
UK Average 1995	24.8	50.2	22.54	0.65	31.15

Source: The Meteorological Office & DETR (1998c), pp. 30-55.

For most cities, the average concentrations for the different air pollutants are relatively close to the annual UK average. London shows for all pollutants, except for ozone, an average concentration that is well above the national average. This may in fact be a significant indication of the dramatic traffic situation and the resulting air quality problems in London.

Having presented the different data sets required for the statistical analysis of the association between various air pollutants and acute mortality effects, the following chapter presents a detailed discussion on the methodological issues involved in such an analysis. Based on this discussion, a core model is presented that will be applied in the regression estimation process.

9. Methodological Issues and the Statistical Model

9.1 Introduction

As reported in Chapter 7, a growing number of studies from various locations report the findings of significant adverse short term effects of current levels of air pollutants on human health. These epidemiological studies show relationships, which quantitatively link increases in air pollution and a number of health endpoints especially connected with cardio-respiratory diseases, including premature mortality, hospital admissions, emergency room visits, restricted activity days, the exacerbation of asthma, various respiratory symptoms, and the loss of lung function.

However, such analyses of daily counts of health events and daily air pollution raise a number of important methodological issues, which may have a significant influence on the results. Consequently, a variety of methods have been applied in these studies in order to design adequate statistical models.

The present chapter gives an overview of the main methodological issues, which have occurred and which have been extensively discussed in the literature. The chapter is divided into two main sections. First, the general methodological issues of such studies are discussed. Analysing the short term effects of air pollution on human health endpoints, i.e. in this study premature mortality generally raises two kinds of issues; distributional and modelling issues. The following section will discuss these two points in some detail, although no claim of completeness is made, since both issues have been very extensively discussed elsewhere.[1] Second, applying the discussion from the first section, the specific statistical model used for this study will be then described. Here, a general baseline or 'core' model will be established, which will then be applied to the respective data for each of the cities included in this study. The estimated results will be described in detail in the following chapter.

9.2 Distributional Issues

As a first main step in analysing the associations between common levels of air pollutants and small increases in daily mortality, decisions on the process of the underlying mechanism, which is intended to be modelled, have to be made. These distributional assumptions may well influence risk estimates, since the statistical regression model being used will depend on this. Generally, "statistical regression modelling seeks to explain the average or expected value of an outcome variable by explicitly representing its dependence on one or more explanatory variables".[2] In other words, this means that in the specific case examined here, the expected number of daily deaths in the population of a specified city on any given day in terms of dependence may be represented by a number of characteristics, including season, weather, day of the week, and air pollution.

Further, the number of deaths on any given day is relatively small, i.e. only a small portion of the population dies per day. In other words, the number of daily deaths, the dependent variable, is a rare event of discrete nature, a so-called event

---

[1] For the discussion of distributional issues see, for instance, Cameron and Trivedi (1998) and Winkelmann (1997). For modelling issues, see, for instance, Lipfert (1993), Schwartz (1994c) and Schwartz et al. (1996), with the latter discussing both issues closely to the topic of air pollution and health.

[2] European Commission (1995a), p. 72.

count. An event count is the realisation of a non-negative integer-valued random variable and refers to the number of times an event actually occurs.[3] Formally, the probabilistic system that is aimed to identify can be described as follows:

$$N = f(X) \qquad \text{where:} \quad X \in \mathbb{R}^k$$
$$\mathbb{N} \in \{0, 1, 2, \ldots\}.$$

Like the normal linear regression model, models for count data show a dual structure with a 'systematic' component, and a 'random' component.[4] Winkelmann (1997) points out that "[i]t is the coexistence of systematic and random effects that is addressed by statistical regression models"[5]. The estimation of the 'systematic' or 'explained' part of the regression model may also depend on the real and assumed pattern of 'random' or 'unexplained' variation in the examined data. This random or unexplained component captures the departures of the observed realisations from their expectations.

Making assumptions on the distribution of this random component have to take into consideration that the chosen distribution has to account for both main criteria of the data; the non-negativity of the data as well as their integer valuedness. Especially for count data, where event histories can be interpreted as the outcome of an underlying stochastic process,[6] this random component is of

---

[3] The assumption of non-negativity implies that the underlying distribution has a lower bound, but no explicit upper bound. The principal of randomness implies that the events are scattered by chance alone. See: Cameron and Trivedi (1998), p. 1.

[4] The systematic effect is the theoretical effect of X on N. See: McCullagh and Nelder (1989), p. 3- 4.

[5] Winkelmann (1997), p. 2.

[6] A stochastic process can be defined as a collection of random variables from an exponential distribution indexed by time. Such a stochastic process is said to be a count process, if N(t) represents the total number of events that have occurred before t. For a more detailed discussion on the univariate Poisson process, see, for instance, Winkelmann (1997), pp. 9-18, Cameron and Trivedi (1998), pp. 3-10, Isham (1991), pp. 177-203 and Diggle (1990), pp. 99-102.

major importance and may give insights about the way the data have been generated.[7]

The two most prominent probability distributions that account for these described characteristics are the Poisson and the negative binomial distribution.[8] The most basic model generally serving as a benchmark model is the Poisson distribution, which has been derived as a limiting case of the binomial by Poisson (1837).[9] The underlying process resulting in the Poisson distribution for the number of counts during a fixed time interval, is a stochastic process in continuous time, the so-called univariate Poisson process. The main assumptions concerning this Poisson process are independent and constant probabilities for the occurrence of successive events.

The following basic Poisson model is taken from Cameron and Trivedi (1986).[10] First, it is assumed that Y_i is the number of occurrences, for the ith of N individuals, of an event in a given interval of time (t, t+dt), and Y_i is also a random variable with a discrete distribution being defined on $N \cup \{0,1,2\ldots\}$. If

$$\Pr[y(t, t + dt) = 0] = 1 - \lambda dt + 0(dt) \tag{17}$$

and

$$\Pr[y(t, t + dt) = 1] = \lambda dt + 0(dt) \tag{18}$$

so that

$$\Pr[y(t, t + dt) \geq 2] = 0(dt), \text{ as } dt \rightarrow 0 \tag{19}$$

---

[7] See: Winkelmann (1997), p. 2.

[8] There is now a large body of literature on a variety of possible stochastic models and their implications. See, for instance, Cameron and Trivedi (1998), Winkelmann (1997) and Harvey and Fernandes (1989).

[9] All three distributions mentioned display a similar structure, and are in fact related with each other through various limiting forms. Besides others, one feature all three have in common is their skewness to the right.

[10] See: Cameron and Trivedi (1986), p. 31.

then the number of events in an interval of a given length is Poisson distributed with the following probability density:[11]

$$\Pr(Y_i = y_i) = f(y_i) = \frac{e^{-\lambda_i} \lambda_i^{\,y_i}}{y_i!} \tag{20}$$

where: $y_i = 0, 1, 2, \ldots$; (realised value of random variable).

$i = 1, 2, \ldots, N$.

$\lambda_i \in \mathbb{R}^+$.

The Poisson distribution has, in contrast to the normal distribution only one parameter, namely, the mean λ.[12] This single parameter, which must be positive, is the constant rate of occurrence of the event of interest and simultaneously determines mean and variance. In contrast to the normal distribution, where two adjustable parameters are given, namely the mean and the variance, with the latter being independent of its mean, the Poisson distribution is entirely determined by the mean alone. However, many actual count processes show the attribute that the variance usually increases with the expected value. In other words, the observations on those days with higher expected value are more variable than observations with lower counts, which means that proportionality between the variance and the expected value is given.[13]

Applying the described features to the case of daily deaths, these counts are derived from a Poisson process and are indeed directly observed counts.[14] The

---

[11] The following descriptions are mainly based on Winkelmann (1997), Cameron and Trivedi (1998), Greene (1997) and McCullagh and Nelder (1989).

[12] Consequently, the Poisson distribution may also be written as $Y \sim Po(\lambda)$.

[13] The issue of the mean-variance relationship will be discussed in more detail later in this section.

[14] Although the normal distribution is not admissible, it may provide a reasonable approximation if the number of count events in each time period is relatively large. If the distribution has a large enough mean, the Poisson distribution is fairly symmetric. For daily mortality due to air pollution this might have been given during high pollution episodes during the 1950s and 1960s, whereas present data does not permit a Gaussian approximation. See: Schwarz et al. (1996), p. S4.

Poisson process assumes that there is a homogeneous risk to the underlying population, and consequently, the expected number of deaths on any day is given by the parameter λ.[15] In other words, equation (20) can be interpreted as the probability of y deaths that occur on a given day during a fixed time period.

Based on the described issues of the Poisson distribution, the Poisson regression model, which is the standard model for count data can be derived. The regression model is usually described in the form of a log-linear regression model. In order to incorporate exogenous variables x_{ij} (j = 1, ..., K), the intensity parameter λ is allowed to depend on the regressors or covariates. By setting[16]

$$\lambda_i = \exp(x_i \beta) \qquad i = 1, 2, \ldots, n. \tag{21}$$

observed heterogeneity is introduced.

Combining both equations result in the classic Poisson regression model, where the explanatory variables are introduced in the following log-linear form: [17]

$$E\left(Y_i \mid x_i\right) = \exp\left(x_i\ \beta\right) \qquad i = 1, 2, \ldots, n. \tag{22}$$

where:

x_i: (1 x k) vector of non-stochastic covariates (explanatory or exogenous variables).

β: (k x 1) vector of coefficients (parameter vector).

y_i: observed values, which are drawings from a Poisson distribution of the form that has been described above (explained or endogenous variable).

---

[15] In other words, the underlying process is assumed to be stationary and homogeneous with iid arrival times for events (i.e. daily deaths).

[16] Using an exponential function ensures the non-negativity of y_i. See: Cameron and Trivedi (1986), p. 31.

[17] See, for instance, Winkelmann and Zimmermann (1995), p. 2.

The above model is called mean function or mean regression, since it specifies the conditional mean of y as a log-linear function of x and β. Using an exponential form implies that the level change in $x\beta$ required for a given percentage change in $E(Y|x)$ is kept constant. In this log-linear model version, the mean parameter λ_i is parameterised as described in equation (21).[18] Also, in order to account for the non-negativity of the Poisson distributed dependent variable, it is assumed that the conditional mean has a multiplicative (or log-linear) form.[19]

Applying this regression model to the analysis of air pollution and health, most studies have reformulated the above equation (22) to:

$$Log(E(Y)) = c + \beta_1 X_1 + ... + \beta_p X_p \qquad (23)$$

where:

- Y: count of deaths on a given day.
- E(Y): expected value of Y on that day (corresponds to λ).
- $X_1, \ldots, X_p$: predictors of daily death counts.
- $\beta_1, \ldots, \beta_p$: regression coefficients for the predictors ($X_i, \ldots, X_p$).

This standard Poisson regression assumes that the dependence is parametrically correct and involves exogenous covariates. In other words, the value of the dependent variates or responses is believed to be affected by the explanatory variables or covariates, which are sometimes also called stimulus.[20] These

---

[18] See: Cameron and Trivedi (1998), pp. 9-10.

[19] Generally, in the classical linear model, the systematic effects are described using an additive model and the normal distribution is assumed as the nominal error distribution (constancy of variance is, of course, also assumed). In contrast, in the analysis of discrete data applying log-linear models for counts, these two components are replaced by a multiplicative model for the systematic effects and by the assumption that the errors are approximated by the Poisson distribution. For a more detailed analysis see, for instance, McCullagh and Nelder (1989), pp. 14-32.

[20] See: Tukey (1962).

covariates may be quantitative or qualitative; quantitative variates take on numerical values, qualitative variates take on non-numerical values or levels from finite set of values or labels.[21]

However, it does not allow for any other source of stochastic variation. If the latter is given, i.e. the function relating λ and the covariates is stochastic, possibly due to unobserved random variables, a so-called mixed Poisson regression is given.[22]

Using the described parametric Poisson regression model, estimation with maximum likelihood is straightforward. Maximising the following log-likelihood functions provides the maximum likelihood estimator (MLE) of the parameter β:

$$l(\beta) = \sum_{i=1}^{N} \left(y_i \left(x_i^{'} \beta \right) - \exp \left(x_i^{'} \beta \right) - \log \left(y_i ! \right) \right) \tag{24}$$

The MLE estimator of β, $\hat{\beta}$, is obtained by differentiating this log-linear function with respect to β and is the solution of the following first-order conditions:[23]

$$\sum_{i=1}^{N} \left(y_i - \exp \left(x_i^{'} \beta \right) \right) x_i = 0 \tag{25}$$

The second-order derivative matrix (Hessian) is given by:[24]

---

[21] See: McCullagh and Nelder (1989), pp. 8-9.

[22] For this study, the main focus lies on the problem where neglected or unobserved heterogeneity is inadequately captured by the covariates in the conditional mean function. This issue of so-called overdispersion will be described later in this section.

[23] See, for instance, Lee (1986), pp. 690-691 and Lawless (1987), pp. 808-809.

[24] The inverse of this matrix with the negative sign gives the asymptotic covariance matrix of the ML estimates. See, for instance, Maddala (1983), pp. 51-52.

$$H(\beta; y, x) = \frac{\partial^2 l(\beta; y, x)}{\partial\beta\partial\beta'} = -\sum_{i=1}^{n} \exp(x_i \beta) x_i' x_i \tag{26}$$

The Hessian is negative definite and thereby demonstrates that the log-likelihood is globally concave as long as X is full column rank and $\exp(x_i \beta)$ is non-negative for all x_i. Hence, the second order conditions for a maximum at $\hat{\beta}$ are fulfilled.[25] Due to the fact that the log-likelihood function is globally concave, standard numerical algorithms will converge rapidly to a unique maximum of the log-likelihood function. The non-linearity of the first order condition in β shown in equation (26), indicates that the equation system can be solved using an iterative algorithm.[26] The resulting unique MLE $\hat{\beta}$ is consistent, efficient and asymptotically normally distributed if the conditional mean function is correctly specified, i.e. the conditional mean equals the conditional variance.[27]

The corresponding variance matrix is consistently estimated by:

$$\text{var}(\hat{\beta}) = \left(\sum_{i=1}^{N} \frac{\partial \hat{m}_i}{\partial\beta} \frac{\partial \hat{m}}{\partial\beta'} \Big/ \hat{m}_i \right)^{-1} \tag{27}$$

where: $\hat{m}_i = \exp\left(x_i', \hat{\beta} \right)$.

25 See: Hausman, Hall and Griliches (1984), pp. 911-912.

26 For count data, gradient methods, such as second derivative methods, including the Newton-Raphson and quadratic hill-climbing (Goldfeld-Quandt) methods, or alternatively the first derivative algorithm referred to as Berndt-Hall-Hall-Hausman method, are commonly used non-linear optimisation algorithms for the estimation of such models. For a more detailed description, see: EViews3 User's Guide (1998), pp. 619-623 and Winkelmann (1997), pp. 60-61.

27 For count data models, consistency requires a correct specification of the mean and efficient estimation requires a correct specification of the mean *and* variance. Also, it is of course assumed that the conditional distribution of y is Poisson.

As already mentioned above, the main condition for this Poisson model is the mean-variance equality, which can be shown as follows. Deriving from the probability generating function P(s)[28], the expected value of the Poisson distribution is given by $E(X) = P'(1) = \lambda$, and the respective variance by $Var(X) = P''(1) + P'(1)-[P'(1)]^2$, which is equal to the expected value λ.[29] Hence, the single parameter λ postulates equality of the (conditional) mean and the (conditional) variance, which is a main characteristic of the Poisson distribution and is referred to as equidispersion.[30]

However, in most empirical applications, this restrictive variance assumption does not hold due, for instance, to omitted explanatory variables independent from the exogenous variables x. For the case where the conditional variance is smaller than the conditional mean, underdispersion is present.

For the more common opposite case where the conditional variance is greater than the conditional mean, the count processed is said to be overdispersed, i.e. 'extra Poisson-variation' is present. Here, the (conditional) variance is proportional to λ, the conditional mean. Contrary to other distributions, particularly the normal distribution, the violation of the equidispersion assumption is a sufficient condition for a violation of the Poisson assumption.[31]

Generally, in both cases, the variance is modelled as a function of the mean, whereby it is crucial, which assumption is being made in respect of the specific

---

[28] The probability generating function (pgf) can be derived as follows:

$P(s) = E(s^x) = \sum_{k=1}^{\infty} e^{-\lambda} \frac{(\lambda s)^k}{k!} = e^{-\lambda + \lambda_s}$. See: Winkelmann (1997), p. 20.

[29] Winkelmann (1997) gives an explicit derivation of this, see: pp. 19-21 and pp. 171-173.

[30] In other words, the conditional expectations of the endogenous variable given the exogenous variables and the corresponding conditional variance cannot vary independently.

[31] Basically, in the case of overdispersion, the Poisson standard errors underestimate the true ones and vice versa.

form of the variance function. This issue has been discussed rather extensively in the literature with particular focus on how to test for and how to model violations of equidispersion using parametric and semi-paramateric alternatives.[32] Following Winkelmann and Zimmermann (1992):

> [I]n many ways the Poisson model is as relevant for count data as ordinary least squares is for continuous data: Optimality properties analogous to the Gauss Markov theorem can be derived for the Poisson model. In the same way as for the homoscedastic linear model, violations of the variance assumption cause regression estimates to be inefficient and the estimated standard errors to be potentially biased.[33]

In other words the parameter estimators are inefficient, although they are still estimated consistently. The resulting variance covariance matrix of the Poisson model is biased, which causes spurious inference. A number of different approaches have been introduced in the literature in order to achieve robust Poisson regression and particularly robust standard errors.[34]

The present study estimates the extra-Poisson variability using the technique suggested by McCullagh and Nelder (1989) using GLM (generalised linear models) standard errors.[35] Generally, the class of GLM is a class of statistical models which generalises classical linear models to include other models, such as log-linear models and multinomial response models for counts, probit and logit

---

[32] For a good overview on this issue see, for instance, King (1989), Winkelmann and Zimmermann (1992), Winkelmann (1997) and Cameron and Trivedi (1998).

[33] Winkelmann and Zimmermann (1992), p. 1.

[34] Robustness in this context is meant in the sense of being robust against deviations from a distributional or variance-mean specification rather than in the sense of resistance to outliers. See: Winkelmann and Zimmermann (1995), pp. 12-13.

[35] Another common method is to allow the parameter λ of the Poisson distribution to vary randomly across the population according to a certain probability distribution. The form of the resulting compound Poisson depends on the specific distribution of λ; with the gamma distribution and the negative binomial distribution for the λ parameter. If the specific form of the compounding distribution is not known, methods such as quasi-likelihood as proposed by McCullagh and Nelder (1989) may be employed. For a more detailed discussion of these methods see, for instance, Cameron and Trivedi (1998), Wedel et al. (1993), Gourieroux, Monfort and Trognon (1984a, 1984b) and Hausman, Hall and Griliches (1984).

models for data in form of proportions (ratios of counts), and models used for survival data. McCullagh and Nelder (1989) state that:

> [t]he above models share a number of properties, such as linearity, that can be exploited to good effect, and that there is a common method for computing parameter estimates. These common properties enable us to study generalized linear models as a single class, rather than as an unrelated collection of special topics.[36]

Since the Poisson model belongs to this class of GLM, a Poisson pseudo-MLE or quasi-MLE can be obtained. In other words, estimators are obtained by maximising a likelihood function associated with a family of probability distributions, which does not necessarily contain the true distribution.[37] The assumption of a correctly specified Poisson mean is still made, but the Poisson restriction on equidispersion is relaxed, i.e. the distribution generating probability (dgp) used to obtain the distribution of the estimator need not to be the Poisson. The basic assumptions of GLM are that the distribution of the dependent variable (y) belongs to the exponential family and that the conditional mean of this variable (Y) is a smooth non-linear transformation of the linear part x'β or λ, which gives the following relationship:

$$E\left(y \mid x, y\right) = h\left(x_i^{'} \beta\right) \tag{28}$$

Knowing that the standard QML covariance does not possess any efficiency properties, even though it is robust to general misspecification of the conditional distribution of y, as described above, a consistent estimate of the covariance can be obtained by taking the GLM conditions into consideration. Specifically, it is assumed that the true variance of the dependent variable y is proportional to the

---

36 McCullagh and Nelder (1989), p. 1.

37 For an extensive discussion on pseudo maximum likelihood methods, see the two seminal papers in Econometrica by Gourieroux, Montfort and Trognon (1984a) and Gourieroux, Monfort and Trognon (1984b).

variance of the distribution used to specify the log-likelihood. Further, it is also assumed that the (true) variance of y is proportional to the variance of the distribution. Hence, the following estimated variance function is given:[38]

$$\operatorname{var}\left(y \mid x\right) = \sigma^2 \operatorname{var}_{ML}\left(y \mid x\right) \tag{29}$$

So, the ratio of the (conditional) variance to the mean is assumed to be σ^2, the dispersion parameter, which is assumed to be constant over the data. The three possible cases are: $\sigma^2 > 1$ is equivalent to overdispersion (empirically relevant case); $\sigma^2 = 1$ is equivalent to equidispersion; $\sigma^2 < 1$ is equivalent to underdispersion. If the proportional variance condition as described above holds, a consistent estimate of the general linear model covariance is given by

$$\operatorname{var}_{GLM}\left(\hat{\beta}\right) = \hat{\sigma}^2 \operatorname{var}_{ML}\left(\hat{\beta}\right)^{39} \tag{30}$$

The constant dispersion parameter σ^2 that is independent of x can be estimated by a certain moment estimator given by:

$$\hat{\sigma}^2 = \frac{1}{N-K} \sum_{i=1}^{N} \frac{\left(y_i - \hat{y}_i\right)^2}{\sqrt{Var\left(\hat{\varepsilon}_i\right)}} = \frac{1}{N-K} \sum_{i=1}^{N} \frac{\hat{u}_i}{\hat{v}_i} \tag{31}$$

An extensive discussion on both generalized linear models and the phenomenon of overdispersion is given in McCullagh and Nelder (1989).[40]

---

[38] See: McCullagh and Nelder (1989), pp. 193-200 and Winkelmann (1997), pp. 118-123.

[39] This is equivalent to: $var\left(y \mid x_i; \hat{\beta}\right) = \hat{\sigma}^2 \hat{\lambda}_i$

[40] See also Cameron and Trivedi (1990), Gurmu (1991) and Dean and Lawless (1989).

9.3 Modelling Issues

After discussing the main distributional issues relating to the statistical analysis of the association between air pollution and daily counts of deaths, this section will deal with the issues relating to the specifications of an appropriate regression model. The analysis in this study is based on time-series data, examining changes in mortality data within a specific area (i.e. city) as air pollution levels fluctuate over time. In comparison, cross-sectional analysis compares the rate or prevalence of a given health outcome across various locations for a given point in time. In both cases the most important concern to be addressed in specifying the regression model is the issue of controlling accurately for potentially confounding variables.

However, as Schwartz et al. (1996) describe, using time-series techniques for the analysis of daily deaths due to air pollution contains a key advantage over others, particularly cross-section analysis.[41] Many of the potential variables that may confound the relationship between air pollution and chronic (long-term) health effects cannot confound short-term temporal (acute) relationships between air pollution and health.

The list of such variables include, gender, age, smoking habits and diet, activity patterns, socio-economic status, as well as occupational exposures and indoor exposure.[42] This is due to the fact that they do not have any short-term variation in time, i.e. they do not vary with daily fluctuations in air pollution sufficiently to

---

[41] See: Schwartz et al. (1996), pp. S3-S4.

[42] See, for instance, Ostro (1994), p. 7, Jones (1995), Donnelly (1994) and World Resources Institute website at http://www.igc.apc.org. For socio-economic and cultural factors in air pollution epidemiology see, for instance, Portney (1983), Korc (1996), Sexton et al. (1993), Edwards, Walters and Griffiths (1994)and Jantunen (1997). A study by Drever, Whitehead and Roden (1996), for instance, found that mortality in the lowest social class is over three times that of the highest social class. See also Charlton (1996).

drive the observed observation. In other words, even though the underlying risk to the population may vary with such factors, the Poisson parameter λ will not be influenced by these factors, since, for instance, the age distribution and smoking and diet history of the population will not vary from day to day.[43]

Further, it is pointed out that "[o]ne feature of the Poisson process is that even if all the covariates predictive of λ were known and measured without error, there would still be considerable unexplained variability in daily mortality. That is because the explanatory variables can at best predict λ."[44] Furthermore, as described in equation (22), the Poisson process ensures stochastic variability around an expected count, even though the number of deaths on any day (λ) is known with certainty.

Nevertheless, there are still other mortality determining factors that do vary over time, and may therefore confound the regression analysis. Consequently, there is a need for adequate controlling for these confounders in order to obtain unbiased estimation results. As Brunekreef et al. (1995) point out "[f]ailure to adjust for confounders that vary in time may lead to either under- or overestimation of the effect or air pollution".[45] The two main categories of potential confounding factors that have to be considered are so-called systematic long wavelength and systematic short wavelength patterns. The following sections will describe the features of both patterns and discuss possible methods to control for them.

---

[43] This means that the underlying risk λ varies with time varying predictor variables, i.e. the Poisson process is not stationary over time.

[44] Schwartz et al. (1996).

[45] Brunekreef et al. (1995), p. 9.

9.3.1 Systematic Long-Wavelength Patterns

Substantial seasonal and other long-term temporal patterns are often found in daily mortality data. While air pollution might be the cause for this pattern, other factors that may be causally unrelated to air pollution changes clearly play a dominant role. Hence, it is believed that the likelihood of confounding air pollution factors with other variables is less severe when comparing short-term fluctuations in mortality and air pollution than when comparing longer-term fluctuations. Consequently, a strategy that focuses on those shorter-term fluctuations, for instance, by filtering out the longer-term patterns, will reduce the potential for confounding. The variables ascribed to long wavelength pattern include mainly two components; season and trend.[46]

Various methods have been applied in studies to control for these long wavelength patterns. Smoothing the relevant data is possibly the most straightforward method.[47] Here, types of moving average procedures are applied to the data, where the same length on either side of the day whose expected number of deaths is estimated (centred moving average).

However, applying a 'simple' moving average is commonly agreed to be unsatisfactory since it is more realistic to assume that the impact of air pollution will be less severe the further one is away from the day of exposure.[48]

A more adequate filter should ideally incorporate different (lower) weights for points further apart in time when estimating the deaths for today.[49] This also

---

[46] These components are described as long wavelength patterns, since the patterns describing interval (in days) is long.
[47] See, for instance, Kinney and Özkaynak (1991).
[48] See: Schwartz et al. (1996), p. S5.
[49] This is also known as kernel smoothing.

avoids the distortion of creating a sudden decrease of the weight from one to zero between the last day included into the moving average and the subsequent one. Also, although there appears to be a tendency towards a length of 15 days, there seems to be no rationale for a generally accepted length for such a filter.[50] Schwartz et al. (1996) claim that "[t]he best choice may in fact vary from study to study depending on other local characteristics such as meteorology, etc, and is likely to be larger than 15 days".[51]

An important fact when smoothing the mortality data is that, while filtered Gaussian data remains Gaussian, Poisson data will no longer be Poisson after filtering.

Consequently, since daily death counts are Poisson, pre-filtering is not possible, and a different filtering technique is required. One such approach is the incorporation of a filter, such as the weighted moving average, directly into the Poisson regression as a regressor on the right-hand side of the regression equation.[52] Again the aim is to achieve a substantial reduction in the variance contributed by annual and other long-period cycles, while leaving intact short-term, day to day variations.

However, when applying a moving average filter in order to smooth the data, one has to be careful that not only the seasonal cycles are removed, but also anything which is correlated with them, and with particular reference here, mortality due to air pollution. Since the effect one is trying to investigate is very small (the pollution-mortality relationship), de-trending and de-seasonalising all elements

[50] The choice of 15 days may be favoured, because it leaves exactly one week on either side of the day whose expectation was actually being computed.

[51] Schwartz et al. (1996), p. S6.

[52] See: Burnett et al. (1994), p. 179.

prior to the analysis is highly likely to lead to 'overcorrection'.[53] Consequently, a number of studies felt that instead of pre-filtering the data, it would be more appropriate to allow the cyclical features to 'compete' with all the other components, mainly air pollution variables, within the regression.

A seasonal variation is apparent in many natural time-series, such as temperature. Many of these time-series also display a regular daily variation that has appearance of rough 'cycles'. Approximating such periodic behaviour, a parametric approach is often applied, where trigonometric functions are included as additional explanatory variables in the estimating regression. Thus, in order to fit the long wavelength pattern (season) in the data, sinusoidal terms are used. Such sinusoidal terms are a combination of sine and cosine waves of various frequencies that are included in the model.[54] Schwartz et al. (1996) point out that "[o]ne concern with such a model is that it assumes that the seasonal peak is the same height and occurs at the same time each year"[55]. Rather, seasonal cycles of an intensity of more than one year can be observed. Consequently, sinusoidal terms with a two-year period or more are also usually introduced in the model. The aim is to produce a predicted curve that shows different shaped peaks for the different years included in the study, which also includes major differences in the shoulders. If the observed daily death counts do not show such a pattern, this is, of course, less of a concern.

---

[53] Two alternative approaches have also emerged in the literature. First, as a 'generalised' form of the moving average filter, the generalised additive model has been used. This is a more flexible approach, where a regression model is to be fitted, which controls for a smooth function of time, as well as the other covariates, such as temperature and other continuous variables. Alternatively to this, a semi-parametric approach using regression spline functions may be applied. Here a cubic polynomial is fitted to each interval, in which the variables are divided. Problems may arise when deciding in how many intervals the data should be divided. For a more detailed description, see: Schwartz et al. (1996), p. S5 and particularly Hastie and Tibshirani (1990). Also, these filtering methods seem to be problematic for Poisson data, since they were developed for Gaussian data.

[54] Frequency is defined as the number of times the function repeats itself in a period of length 2π. Assuming that January 1st is not the start for all sinusoidal terms requires that when fitting a sine wave with a given frequency to the model, a cosine wave with the same frequency has to be fitted as well.

[55] Schwartz et al. (1996), p. S6.

The following function gives an example of such an annual harmonic sinusoidal wave.

$$\sin(2\pi t/365) + \cos(2\pi t/365) \qquad (32)$$

where: t = 1 to T (day of study)

The frequency or period of the harmonic wave is determined by f = $2\pi*t/365$, which assumes a symmetric pattern. However, it is certainly more realistic to assume some kind of asymmetric pattern, where steep increases and flat declines, additional peaks and dips, or long summer periods and short winter periods are present.

Hence, it may be more realistic to substitute the frequency term f by a term, which takes this pattern into account. Such a term might be described by f* = $k*2\pi*t/365$, with k taking the values 1, 2, 3, etc. Including such terms, for instance, up to the 6^{th} order (k<7) picks up two month periods (or events of one month's length). In order to account for possible differences in seasonality between years, a bi-annual cycle (k = 0.5) and/or interaction terms between years and cyclical terms may be included.[56]

Furthermore, in order to control for possible differential effects of a specific air pollutant on the health outcome in summer and winter, additional dummy variables for season, i.e. summer and winter, as well as interaction terms between season and the relevant air pollution terms can be included.[57]

---

[56] See: Katsouyanni et al. (1996), S16.
[57] See, for instance, Katsouyanni et al. (1996), p. S17.

In addition to these trigonometric functions, trend terms are often included in order to account for the long wavelength pattern in the data. These trends are also known as secular trends. Longer-term air quality trends may arise due to community growth (i.e. changes in the size of the underlying population) or air pollution abatement efforts. Controlling for these trends is of great importance in time-series analysis, "[s]ince any two variables that show a long term trend must be correlated, searches for correlation that are more likely to be causal must exclude these trends."[58]

The form of trend included into the model may vary, depending on the data and time period studied. The longer the time period included in the analysis, the more likely is it that these trends are non-linear. Consequently, in addition to trend variables (linear and non-linear), most relevant studies have also included annual dummy variables in the model, in order to capture any additional temporal trend over the studied time period.

9.3.2 Systematic Short-Wavelength Patterns

Short-term systematic components that might also confound time-series regressions are so-called calendar specific effects. This category includes variables for day of the week and holiday effects, as well as infectious disease epidemics, such as influenza.[59] Weekly cycles, for example, may be expected especially in traffic-related air pollutants or in those pollutants related to local industrial sources that operate with reduced emissions on weekends. Hence, a number of studies have shown that weekends are better for people's health than weekdays, particularly Mondays. The same may be observed for holidays.

---

[58] Schwartz et al. (1996), p. S4.
[59] Influenza epidemics may, however, also be seen as a mid-term systematic pattern.

Consequently, it is generally agreed that these variables should be considered in time-series studies on air pollution and acute health. Also, adequate control is required if infectious disease epidemics have occurred during the period of the study. This may either be done by including actual data for these epidemics or by including indicator variables into the regression model for days such an epidemic has been identified. However, various studies indicate that the inclusion of these variables may actually not have any major impact on the estimation outcome of the air pollution coefficients, provided that the applied model will adequately control for season.

9.3.3 Unsystematic Meteorological Patterns

In addition to the systematic variables described above, less systematic variables, namely meteorological terms, have to be included in the model. These terms carry information about the effects of short-term variations in weather on mortality independent of pollution. It is believed that these variables are typically much more important than air pollution in terms of explaining day-to-day variations in mortality and hence form the most serious potential confounder in analysing the short-term fluctuations in air pollution with short-term fluctuations in mortality.

The two main weather variables included in pollution studies are daily temperature and relative humidity. With regard to temperature, the debate as to whether minimum, maximum, or mean temperatures provide the best predictor is not entirely resolved, and whether there is any additional information in one measure after control for the other.[60] Hence, most relevant studies include mean

[60] See: Schwartz et al. (1996), p. S7, Katsouyanni (1995) and Khaw (1995).

temperature into the model, whilst particular heat waves for example are separately controlled for by the inclusion of respective indicator variables.

The second important meteorological term is relative humidity, which is a measure of how close the air is being saturated with water vapour. Alternatively, dew point temperature is sometimes used and seems to be also suitable.[61] Alternatively, some studies have included data for dew point temperature, mainly for the reason that data on relative humidity was unavailable or fragmentary.[62] Additionally, dummy variables to indicate hot and humid days respectively may be included into the regression model, as well as interaction terms indicating hot and humid days.

Special concern should be taken about the dependency of these meteorological variables on mortality, i.e. whether they should be treated as linear or non-linear predictor of daily death counts. In most cases a non-linear shape can be assumed, with higher numbers of daily deaths on very hot and very cold days. Hence, a U-shape or V-shape relationship seems to be a straightforward and often adequate assumption.

However, this may be too simplistic an approach, and other, more sophisticated and flexible methods, such as the generalised additive model, as described before, or a 'synoptic' approach may better fit a weather pattern. The latter allows for the simultaneous evaluation of numerous weather elements by combining them into groups or categories that are representative of the variety of micro-climates in a given location (city); and examining mortality difference in

---

[61] See: Spix and Wichmann (1996), p. S55.
[62] The definitions for both relative humidity and dew point temperature are already given in the previous chapter.

terms of these synoptic categories or micro-climates. Consequently, this approach may help to distinguish better between weather and pollution effects.

A recent study examined the effectiveness and consequences of using such alternative approaches to control for weather.[63] They also looked at how weather may modify air pollution effects, as weather patterns could plausibly alter air pollution's effect on human health. They compared four approaches developed for controlling for weather on mortality and investigated how well these approaches performed in controlling potential confounding by weather.

The four approaches included in the mentioned study are two synoptic categories, namely the Temporal Synoptic Index (TSI) and the Spatial Synoptic Classification (SSC),[64] as well as a non-parametric approach (LOESS)[65]. Finally, the authors compared these relatively sophisticated methods with a more straightforward approach using a descriptive model, where weather terms are more or less directly included into the regression model, as described above.

The authors found that although some variation could be noted among the various models, no systematic pattern of variation, by choice of approach used to control for weather, could be found.[66] Further, they claimed that they could not

---

[63] See: Samet et al. (1998).

[64] For both synoptic categories, weather data is classified according to the synoptic categorisation proposed by Kalkstein et al. (1987), where factor analysis is applied to routinely collected meteorological data (air temperature, dew point temperature, visibility, total cloud cover, sea-level air pressure, wind speed and direction), in order to identify the independent components of the data. Scores for these components are then calculated for each day, before these days are then grouped into clusters with similar meteorological characteristics based on the scores. The TSI and SSC calculations differ in the initial selection and subsequent inclusion of a set of representative 'seed days' in the discriminant analysis that produces the categorisation of the days. However, both synoptic categories are chosen to cluster similar weather days, and not to optimally predict mortality.

[65] A detailed definition of this non-parametric approach is given in a later section of this chapter.

[66] As assumed, they found that the descriptive model and the LOESS weather models were much better predictors of total mortality than both the synoptic categorisations of TSI and SSC, since the latter two are chosen to optimally create categories with days similar to one another in certain weather variables and different from days in other categories, but not on the basis on how good mortality can be predicted.

establish any significant evidence that weather conditions actually modified the effect of pollution on mortality, regardless of the respective approach used to represent weather in the model.[67]

Generally, Samet et al. (1998) concluded that using synoptic weather categories in regression models appeared not to change the pollution-mortality relationship with comparison to other approaches such as the descriptive model introduced above.[68] The estimated pollution effects are indeed not highly sensitive to the variables selected for weather conditions.[69]

This result may be used to strengthen the finding of other studies which show that the inclusion of weather variables has no significant impact on the estimated coefficient of the air pollution variables. Schwartz and Dockery (1992), for instance, report that the "[e]xclusion of all weather and season variables did not substantially change the total suspended particulate association",[70] the analysed air pollution index. Hence, since the nature of the dependence of mortality on temperature and relative humidity is still unresolved, no uniform method of controlling for such potential confounding has yet been developed. Consequently, most investigators decide to apply a more 'conservative' approach, where these weather variables will remain in the respective statistical model, regardless of their significance.

Nevertheless, a remarkable variety in the number of weather variables that have been taken into account can be found in various relevant studies. Mackenbach et al. (1993), for example, included 15 such weather variables of different time lags

---

[67] See: Samet et al. (1998), p. 15.
[68] See: Ibid, p. 18.
[69] This has also been confirmed by, for instance, Kalkstein (1993) and Pope and Kalkstein (1996).
[70] Schwartz and Dockery (1992b), p. 15.

in their statistical model.[71] Most studies, however, go for a more balanced approach in sorting out the independent effects of weather and pollution and apply a more parsimonious strategy, i.e. the avoidance of any unnecessarily large numbers of variables.

Generally, the principal population-based studies have adopted a fundamentally conservative strategy in attributing mortality to air pollution in the presence of weather and other potential confounding variables. Since the complex relationships and interactions between all factors involved is by no means entirely understood such a strategy is certainly justified.[72] As Zmirou et al. (1996), for instance, emphasise: "Some core variables (sinusoidal terms for long wave seasonally, week patterns, temperature, and relative humidity) were forced into the model in all cities irrespective of their statistical significance."[73]

9.3.4 Structure of Time Lags

All explanatory variables discussed above may have either immediate effects or delayed effects, i.e. occur with some time lag. Additionally, this time lag structure may also be disturbed.[74] That is, a certain temperature situation (i.e. cold weather) may effect mortality not only on the concurrent day but also on the next day and possibly days after. In other words, the effect of very low temperature on any one day's mortality may be the sum of the effect on that very day and the previous days. However, it can be assumed that the magnitude of impact between these days will be different. Therefore, some studies tested a lag structure with several simultaneously included lags of the explanatory variables.

---

71 See: Mackenbach et al. (1993).
72 See, for instance, European Commission (1995a), p. 75.
73 Zmirou et. al. (1996), p. S32.
74 One of the first air pollution related studies considering time lags is Wyzga (1978).

This approach, however, raises the question of serial correlation and hence unstable estimates, which will be discussed in greater detail in the following sub-section.

Hence, other methods, such as the moving average of the explanatory variables for varying numbers of days have been used. Although this approach accounts for the contribution of multiple days, it is based on the rather unrealistic assumption that the effect of every days included is identical. To overcome this drawback, more realistic methods have been introduced allowing the influence of the explanatory variables to decline with time. Two types of lag models that deal with this issue have been considered in previous studies; geometrically distributed lag models and polynomial distributed lag models.[75] In the geometrically distributed lag model it is assumed that the weights of the lagged explanatory variables are both positive and decline geometrically with time.

Applying this to the problem discussed in this present study, the following regression model can be derived:

$$\log\left(E(Y_t)\right) = \mathrm{cov}\,ariates + \beta\left(X_t + \omega X_{t-1} + \omega^2 X_{t-2} + \omega^3 X_{t-2} + \ldots\right)$$

$$= \mathrm{cov}\,ariates + \beta \sum_{s=0}^{\infty} \omega^s X_{t-s} \tag{33}$$

While the weights ω may take values between 0 and 1, they never become zero and diminish, resulting in a negligible effect of the explanatory variables beyond a reasonable time. Such a model can be iteratively fitted, but has the limitation that it hypothesises a declining set of lag weights.

---

[75] The following descriptions are derived from Pindyck and Rubinfeld (1991), pp. 203-219 and Greene (1997), pp. 511-538.

The polynomial distributed lag model or Almon lag is a more general formulation. Generally, "[t]he polynomial lag model assumes that the lag weights can be specified by a continuous function, which in turn can be approximated by evaluating a polynomial function at the appropriate discrete point in time"[76]. Assuming, for example, lag weights that follow a second-order polynomial with a two period lag, the specific log linear regression model applied in this study, can be described as follows:

$$\log\left(E\left(Y_t\right)\right) = \operatorname{cov}ariates + \beta\left(\omega_0 X_t + \omega_1 X_{t-1}\right) \tag{34}$$

where the weights are defined as:

$$\omega_i = c_0 + c_1 i + c_2 i^2 \quad \text{for: } i = 0, 1, 2. \tag{35}$$

Hence, in this model where a second order polynomial is centred around lag 1, it is assumed that the largest impact of the respective explanatory variable (e.g. temperature) is from the previous day. The impact from the concurrent day and from two day's before is actually the same, but reduced from the lag 1 day by a factor defined by the parabolic function being fitted to the model. Obviously, determining the right order for the polynomial as well as the appropriate lag length has to be made with special care.[77]

In conclusion, no model for an ultimate lag structure can be presented. Previous studies considered lags from the same day up to 10 days previously for the effect of air pollution as well as weather variables. With cumulative combinations, up to 20 different lags may be generated. A single choice of lag, however, reduces the multiplicity of tests and therefore makes p-values more meaningful.[78] Most

---

[76] Pindyck and Rubinfeld (1991), p. 210.
[77] See: Greene (1997), p. 525.
[78] See: Poloniecki et al. (1997), p. 535.

relevant studies found that a simple one-day lag actually gave the best model fit.[79] The choice on how to account finally for potential delayed effects cannot be made beforehand. Rather, various models have to be considered and tested taking the specific structure of the available data into consideration and should be based upon prior understanding and clinical and biological judgement.

9.3.5 Serial Correlation

When using time-series regression it is important to control for the possible similarity of outcome on adjacent or proximate days. In other words, one has to investigate whether two observations closer together in time are more alike than two randomly chosen ones. This problem is known as serial correlation and is a potential issue when studying health effects over time. Schwartz et al. (1996) state that

> [m]easurements connected in time and/or space, such as repeated measurements of the same population on consecutive days or measurements of persons from nearby geographical areas, are likely to be correlated and not independent.[80]

Such serial correlation may be to some extent explained by similarities in explanatory variables such as season, weather and pollution. However, unexplained or residual serial correlation may remain even after adjustment for these factors, and according to its severity, may distort risk estimates (specifically, may inflate estimates of statistical significance) if unaccounted for in the analysis.

---

[79] See, for instance, Schwartz (1991).

[80] Schwartz et al. (1996), p. S8.

Generally, for the case where serial correlation is present, the estimated regression coefficients will not be biased, while the estimated standard errors, however, will be biased. In other words, the Poisson (pseudo)-maximum likelihood estimator will still be consistent in the presence of serial correlation, while the estimator of the variance-covariance matrix may not be consistent, which is, however, necessary for statistical inference. For OLS regression models, a number of methods handling such serial correlation have been well established over the years.

Regarding Poisson regression analysis, Cameron and Trivedi (1998) point out that, although a number of models have been developed in order to control for potential bias through such serial correlation, "times series models for count data are still in their infancy".[81] They claim that the proposed models are generally restrictive and that "[a]t this stage it is not clear which, if any, of the current models will become the dominant model for time-series count data".[82] One straightforward method to model dependencies across time intervals in count data, is the introduction of lagged dependent variables and trend terms into the regression, which is also referred to as an observation-driven process.[83]

While the more sophisticated classes of count time-series models that have been applied in various studies include integer-valued ARMA models, autoregressive models, state-space models, hidden Markov models, discrete ARMA models, and serially correlated error models.[84] In recent studies, Schwartz and colleagues applied an estimation procedure using the generalised estimating equations (GEE) proposed by Liang and Zeger.[85] As with the classic Poisson regression

---

81 Cameron and Trivedi (1998), p. 221.
82 Cameron and Trivedi (1998), p. 221.
83 See: Firth (1991).
84 For a detailed description of all models mentioned, see: Cameron and Trivedi (1998), pp. 234-250. See also Al-Osh and Alzaid (1987) and Alzaid and Al-Oosh (1988).
85 See: Liang and Zeger (1986), Zeger and Liang (1986) and Diggle, Liang and Zeger (1994).

model as described above, the following model structure is assumed: $\log[E(Y_i)] = X_i \beta$, where E denotes expected value; X_i is the matrix of covariates of day I, and Y_i represents the mortality counts on day i. Additionally, in order to model the serial correlation an autoregressive covariance structure is assumed.

This covariance matrix is assumed to have the following form.[86]

$$\text{cov}(Y_{ik}, Y_{jj}) = \alpha A^{\frac{1}{2}} R A^{\frac{1}{2}} \tag{36}$$

where:

A_{ij} = the classic Poisson covariance $[E(Y_i)\delta_{ij}]$[87].

α = the overdispersion parameter.

R = autoregressive matrix.

The overdispersion parameter α is estimated using the GLM method proposed by McCullagh and Nelder (1989) as described above. While the order of the autoregressive matrix R is estimated empirically from the data. This approach actually incorporates the covariance structure in the estimation of the regression coefficients as well as their variances, which is believed to result in more efficient estimates of the respective parameters.[88] It also allows the calculation of robust variance estimates providing unbiased hypothesis tests even if the covariance is actually misspecified.[89]

However, some recent studies have questioned the methodology of generalised estimating equations as an extension of the conventional Poisson regression and

---

86 See: Liang and Zeger (1986), pp. 15-16.

87 $\delta_{ij} = 1$ when $i = j$ and 0 otherwise. See: Schwartz (1993a), p. 1139.

88 See, for instance, Schwartz and Dockery (1992a), p. 601 and Schwartz and Dockery (1992b), p. 14.

89 Liang and Zeger (1986) show that asymptotically unbiased estimates of the standard errors can be obtained using the sandwich estimator. See: Liang and Zeger (1986), pp. 18-22.

other generalised linear modelling to take account for residual (unexplained) serial correlation between days. Moolgavkar et al. (1995), for example, point out that the GEE approach developed by Liang and Zeger may actually be inappropriate in the population-based time-series analyses, due to the relatively small sample sizes. They claim:

> The issue may best be considered as open at present. Fortunately, however, it seems not to be of great practical importance because residual autocorrelation in most studies is low and results may usually be insensitive to adjustment for autocorrelation or not. In practice, therefore, conclusions are based on generalised linear models without the use of the more recent GEE methodology.[90]

This view is supported by a number of studies. In fact, the vast majority of the relevant studies do not actually find significant evidence of serial correlation in the number of daily deaths.[91]

Generally, the results of such studies show that, after controlling adequately for possible confounding variables, such as season, trend and epidemics, the actual remaining magnitude of serial correlation is, if it is present at all, very low.

Finding little or no serial correlation in regressions analysis of time-series of counts is unsurprising, given that a pure Poisson process generates a time-series of independent counts. "The order of magnitude of remaining serial correlation is reported to be as low as 0.50 to 0.20."[92] This finding is perfectly justifiable, since the serial correlation here is not due to the fact that a health event such as a death on one day is causally related to a similar or the same health events on the next day. It is rather the underlying causes, such as weather, epidemic infectious disease, or air pollution that tend to be highly correlated day to day. As a

---

[90] European Commission (1995a), pp, 73-74.
[91] See, for instance, Pope, Schwartz and Ransom (1992).
[92] Schwartz et al. (1996), p. S9.

conclusion, it is suggested that no significant improvement to the estimates will be achieved by incorporating terms for serial correlation.

As a final remark regarding the modelling issues discussed above, it can be stated that various studies have carefully analysed the methods applied and have come to the conclusion that the statistical methods used in the principle papers on the association between short term-health effects and air pollution are reliable and appropriate. No substantial basis can be found for considering that the positive associations reported are an artefact of the relatively sophisticated methodology often used.[93]

Based on the methodological issues outlined and discussed in this section, the following section will deal with the development of a statistical model with particular relation to the underlying data of this present study. This baseline model will then serve for the estimation of the association between daily counts of deaths and daily air pollution concentration in all British cities included in this present study. The results of these estimations are presented and discussed in the following chapter.

9.4 The Core Model

In order to investigate the association of the daily number of deaths and air pollution, this study will follow the protocol for the APHEA project. In the scope of the APHEA project, a standardised procedure for testing and quantifying the short-term acute health effects of various air pollutants has been

[93] See: European Commission (1995a, 1995b).

developed and successfully applied in a number of studies.[94] Generally, the broad strategy adopted to the modelling of confounders within Poisson regression is first to construct a statistical baseline or core model.

In this core model the relationship between the potential confounding variables, described in the previous section, and the health outcome variable is analysed, with no air pollution variable included. The aim is to explain as much as possible of the daily numbers of deaths in terms of those non-pollution confounders. Second, once the best fitting core model is obtained, it is then tested whether this baseline model could be improved by the inclusion of air pollution variables. Finally, it is examined whether the estimated effects of these air pollutants are sensitive to the exact representation of the non-pollution confounders, i.e. whether changing the baseline model leads to changes in the estimated effects of pollution.

Also, sensitivity analysis within studies generally show that the estimated effects of air pollution are insensitive to the specific representation of non-pollution confounders. Hence, this supports the view that the relationships with air pollution are not artefacts of unadjusted confounding factors.

Nevertheless, there still remains the issue of unmeasured confounders and how they are to be dealt with, and in particular, whether weather effects have been taken sufficiently into account. The baseline model is built up by including the controlling variables sequentially following the same order of confounders as they have been described in the previous section. Hence, systematic long-wavelength and short-wavelength patterns, as well as unsystematic short-wavelength patterns have been gradually added to the model.

---

[94] See: Various epidemiological studies summarised in the Journal of Epidemiology and Community Health under the framework of the APHEA Project (1996).

Further, various structures of time lags are examined in order to account optimally for possible delayed effects between stimulus and response, as described in the previous section. The resulting core model is subsequently amended from city to city when a better model fit could be achieved by, for example, including or excluding certain variables, or applying a different time lag structure. These variations are stated in the following chapter when the results for the individual city estimations are presented.

Assuming systematic variations over time, i.e. the underlying populations of each city studied was not necessarily constant during the study period, a continuous time trend variable was introduced into the model. When it improved the model fit, a quadratic time trend variable was also included. Additionally, an indicator variable for each year of the study period (1992 through 1997) was also included into the model.

It was decided to control for seasonal patterns by introducing trigonometric cycles in a form as described above.[95] These smooth combinations of sine and cosine waves included time periods between two years and two month.[96] In other words, up to 6th order sinusoidal terms were included into the model. It was believed that higher order terms, i.e. shorter time intervals, would remove too much of the short-term fluctuations with a potential of overcorrecting the model. Additionally, the model was expanded to include sinusoidal terms with a two year period (order of 0.5) in order to capture changes exceeding one year. When all these trigonometric terms were included, a predicted curve was produced that showed different shaped peaks for the different years, including

---

[95] These cyclical components included into the model were independent of the meteorological as well as air pollution data.

[96] The estimation results using an alternative filtering model are presenetd in Appendix 4.

major differences in the shoulders, which again seemed to be realistic when compared with the plots of the original data.

Although these sinusoid corrections control for the fact that daily mortality may be significantly different between seasons, possible remaining residual confounding was controlled for by including indicator variables for seasons.

After controlling for these systematic long-wavelength patters, systematic short-wavelength patterns were investigated. Here so-called calendar specific effects were examined. It is known that mortality varies with the day of the week. Consequently, six dummy variables were introduced indicating the weekdays from Tuesday through Sunday (Monday formed the representation level). Furthermore, binary variables were also generated and introduced to control for differential mortality during holidays.[97]

Examining whether infectious disease epidemics had occurred during the study period completed controlling for short-term wavelength patterns. As described above, infectious diseases epidemics are a main concern, especially since they may occur in the same season of the year as air pollution. Generally, mortality in the UK is consistently higher during wintertime than during the rest of the calendar year.[98] During the winter period of 1996/1997, a particularly sharp rise in the total number of deaths registered could be observed.[99] The peak in the number of deaths from all causes also coincided with a similarly sharp rise in the number of deaths from influenza.

---

[97] Indicator variables for 7 national holidays have been included. The 'region-specific' holidays in Scotland and Wales have also been considered.

[98] The following descriptions follow closely Christophersen (1997), pp. 11-17.

[99] Winter is here defined as the four-month period from December to March.

During the winter 1996/97 mortality was some 29 per cent higher in relation to the preceding and subsequent four-month periods.[100] Over 70 per cent of the total number of deaths during the winter period were accounted for by the four cause categories of pneumonia, ischaemic heart disease, cerebrovascular disease and bronchitis. Respiratory diseases accounted for nearly half of the winter deaths, with deaths from influenza showed the largest percentage increase in mortality during the winter 1996/97. Most of these deaths occurred during the relatively short period between 24th December 1996 and 8th January 1997.

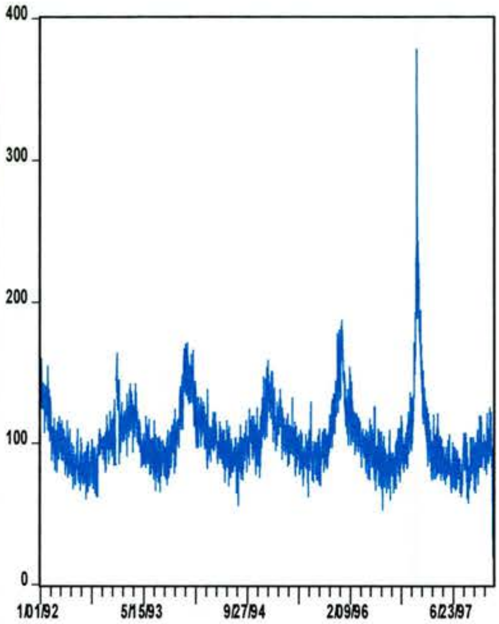
The following graphs representing the total daily number of deaths during the study period between 1992 and 1997 for London and Birmingham, clearly show the described difference between the winter period 1996/97 with a significant spike in the data and all other winter periods. The shaded areas indicated the described influenza epidemic. The same pattern could be observed for all other cities examined in this study; these graphs are reproduced in Appendix 2.

---

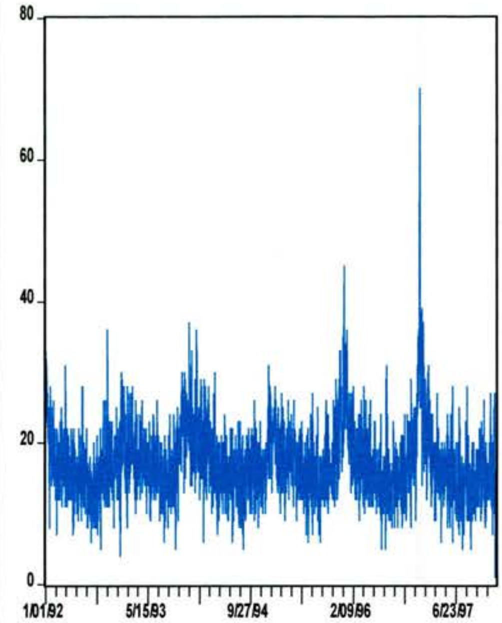
[100] In comparison, the winter mortality was about 17 per cent higher in each of the three previous winters relative to the non-winter periods.

Table 93: Counts of Daily Deaths (ICD-9 390-519) for London & Birmingham 1992-1997.

London



Birmingham



Since no actual data for this influenza epidemic could be obtained, indicator variables have been generated taking the value 1 on each day of the 20 day epidemic period between 20th December 1996 and 8th January 1997, and the value of zero on all days outside this period, controlling for this event.[101]

Obviously the most serious potential confounder in analysing the short-term fluctuations in air pollution with short-term fluctuations in mortality is weather. The impact of weather on mortality has already been discussed very extensively in the previous section. The results of this discussion have been taken into account when establishing the core model for this present study.

Since the weather impact on daily mortality may be linear or non-linear, both linear and non-linear terms for temperature and relative humidity were

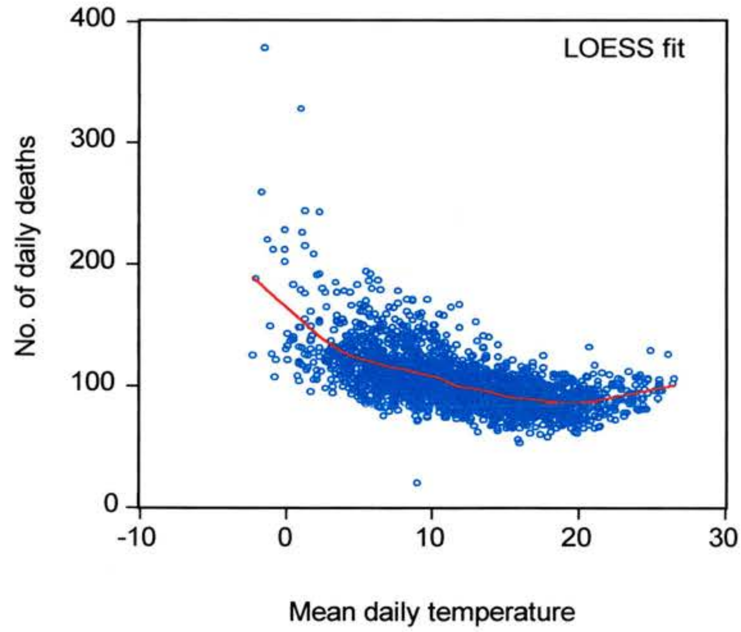
---

[101] Also, random variations, such as the intensity of influenza epidemics that may also induce year to year variations in mortality, has already been controlled for by the inclusion of dummy variables for year of study into the model.

considered in the core model. However, the results of graphical analysis were taken into account when making the decision which terms finally were included. In particular, it was examined whether a U-shape was present in the data, i.e. whether the number of daily deaths increased with both cold daily mean temperatures and hot daily mean temperatures.

Such a flat U-shape can be seen in the following scatter graph, where the number of daily death counts is plotted against daily mean temperature during the study period for London. A non-parametric regression line (LOESS) has been included to indicate the shape of the relationship.

Figure 49: Scatter Plot Daily Deaths against Mean Daily Temperature in London.



As suggested by a number of previous studies, mean temperature and relative humidity were included into the model as un-transformed terms generally lagged by one day in order to account for realistic delayed response effects. Nevertheless, a variety of combinations of time lags have also been examined, such as polynomial distributed lag structures and cumulative effects of various time lags, and if a better overall model fit could be achieved, the one day lag was substituted by a different lag structure. For most cities, however, the one-day lag turns out to be most explanatory. The specific lag structure chosen for the individual cities are presented in the scope of the overall estimation results in the following chapter.

The conservative approach followed in this present study required that all variables discussed so far remained in the final model, even for the case where individual terms were not significant, i.e. they were forced into the model regardless of their significance. For the variables discussed below, however, this was not necessarily the case, and decisions were made on an individual basis.

Additionally, indicator variables for hot days, cold days, humid days and hot and humid days were also considered. Following various other studies, the definitions for the dummy variables for hot days, cold days, humid days, and hot and humid days were set at the 95th percentile (5th percentile for cold days).[102] When generating the indicator variable for humid days, the daily data for dew point temperature was taken as a basis, rather than the data for relative humidity.

For most cities, it turned out that the indicator variables for hot days, cold days and humid days were not significant, which indicated that the respective temperature and humidity effects have already been covered by the temperature and relative humidity terms as described above. If these variables were insignificant, they were not included in the final model.

In contrast, however, the interaction term indicating hot and humid days, turned out to be very significant with a relatively large coefficient in most cases. This

---

[102] See, for instance, Schwartz et al. (1996).

could suggest that there is an increase in mortality on hot and humid days, when the high dew point temperature keeps the temperature high even at night. In other words, oppressive night-time conditions, indicated by high dew point temperatures, following a very hot day may be more stressful than the maximum temperature on a very hot day. This further supports the decision to include mean daily temperature rather than daily maximum temperature in the statistical model. This result is consistent with the results of previous analyses.[103]

Finally, it is conceivable that the effects of secular and weekly trends in mortality as well as the weather effects differ with season. Thus, to control for these differential effects, interaction between the seasons and the above mentioned weather factors were also tested in the model. These interaction terms were mainly generated in order to test for a different relationship between temperature and mortality in different seasons, e.g. the different temperature-mortality relationship in winter and summer. These variables have not been considered in the final model, when they failed to make a significant improvement of the overall model fit.

As mentioned before, the build up of the core model has been performed in a gradual manner, where variables are included into the statistical model sequentially, starting with terms for systematic long-wavelength patterns, systematic short-wavelength patterns to unsystematic short-wavelength patterns. In order to assess and compare the relative quality of the mortality predictions after the inclusion of a further variable into the model, specific information criteria have been considered.

[103] See, for instance, Schwartz and Dockery (1992a), p. 15.

Generally,

> The Kullback-Leibler quantity of information contained in a model is the distance from the "true" model and is measured by the log likelihood function. The notion of an information criterion is to provide a measure of information that strikes a balance between this measure of goodness of fit and parsimonious specification of the model.[104]

The number of different information criteria consequently differ in how to strike this balance. In this present study the Akaike Information Criterion (AIC) is used as a measure of fit of each respective model to the data.[105] The AIC is one of several information criteria that have been widely applied in time-series studies as a guide in model selection.

The general definition of the AIC is:

$$AIC = -2l / n + 2 \times k / n \tag{37}$$

where:

k = number of estimated parameters.

n = number of observations.[106]

l = value of the log likelihood function using the k estimated parameters.

Hence, the AIC is based on minus 2 times the average log-likelihood function, adjusted by a penalty function.[107]

---

[104] Quantitative Micro Software (1998), p. 627. See also Judge et al. (1985) and Kullback (1968).

[105] See: Akaike (1970).

[106] Sometimes, AIC is not divided by the number of observations n.

[107] Generally, this information criterion balances the ability of a particular statistical model to explain the observed data against its complexity. In the present study where a log-linear (Poisson) regression is used, the penalty function is scaled by the degree of overdispersion. In other words, the penalty increases with the number of degrees of freedom when there is more overdispersion. The penalty function for two other common information criteria is k*(log n)/n for the Schwarz criterion (SC) and 2*k*[log (log n)]/ n for the Hannan-Quinn criterion (HQ).

Following Samet et al. (1998), the AIC definition can then be reformulated as:

$$AIC = \text{deviance} + 2*\delta*p \tag{38}$$

where:

δ = estimate of overdispersion.

p = number of estimable parameters in the model.

Technically, the AIC is a first-order approximation to the expected deviance between the model predictions and a new, independent set of data from the same underlying process. Although more complex models will generally outperform simpler models, they also require more degrees of freedom. This is controlled for in the AIC by incorporating the deviance that measures the fidelity of the model predictions to the observed data. Hereby, a penalty is given for adding more parameters into the model.

Generally, the AIC can be taken as model selection guide in so far as smaller values indicate a closer fit of the respective model to the data. However, in the build up of the core model used in this study, the AIC has been used as a general guide for model selection, but in no way as a rigid optimisation criterion. Rather, the combination of various methods, particularly the use of graphical scatter plots, lead to the choice of the final model for each city.

These graphical analyses are now described in more detail. In order to assess the ability of the final core model, with weather and other seasonal patterns, performed in removing seasonal and other systematic long-wavelength patterns from the mortality data, a plot over time of the adjusted data, controlling for all covariates except the air pollution data was generated. This plot was then compared with a plot over time of the raw mortality data. To determine how the expected value of daily mortality varied with time in each case (city), non-parametric smoothed curves have been used.[108] Generally,

> [a] smoother is a tool for summarising the trend of a response measurement Y as a function of one or more predictor measurements $X_1 \ldots X_p$. It produces an estimate of the trend that is less variable than Y itself.[109]

This is especially useful for the case when there is no or little prior information given on which to base a decision that the correct trend is linear. The majority of such smoothers represent a generalised form of weighted moving averages, which predict the expected value of Y at X_i as the weighted mean of the actual values of the Y's corresponding to all X's in a symmetric neighbourhood around X_i. The weights decline with distance from the centre of the neighbourhood.[110]

The nonparamteric smoothing method used was a scatter diagram with a nearest neighbour fit. In this method, for each data point, a locally weighted polynomial regression is fitted.[111] The particular technique used in this study, which is included in this class of regressions, is the so-called LOESS or Lowess (locally weighted scatterplot smoother) technique described by Clevelend.[112] LOESS, a robust,[113] local non-parametric smoother is a special case of the nearest

---

[108] The non-parametric nature of the smoother lies in the fact that that no rigid form for the dependence of Y on $X_1, \ldots, X_p$. is assumed.

[109] Hastie and Tibshirani (1990), p. 9.

[110] A good overview of a number of useful smoothers, including running-mean and running-line smoothers, bin smoothers, kernel smoothers, various spline smoothers, and LOESS is given in Hastie and Tibshirani (1990), chapter 2 and Delgado and Robinson (1992).

[111] It is a local regression since only a subset of the total set of observations, which lie in a certain neighbourhood of the specific data point to fit the regression model, are used. Specific weights are put at the regression in such a way that those observations further away from the given data point are attributed with less weight, which may be seen as a 'penalty'.

[112] Cleveland has published a considerable number of papers on the graphical analyis of data in general, and in particular on LOESS. See, for instance, Cleveland (1979), Cleveland (1984), Cleveland (1985), Cleveland (1993) and Chambers et al. (1983).

[113] Loess has a robustness feature in which, after a first smoothing, outliers are identified and downweighted in a second smoothing. This procedure of identification, downweighting, and resmoothing can be done any number of times in order to prevent outliers from distorting the smoothed values. See: Cleveland (1985), pp. 178-179.

neighbour fit, where the degree of the polynomial is set to 1,[114] and tricube weighing.[115] The bandwidths of these local polynomial regressions based on nearest neighbours vary, adapting to the observed distribution of the regressor.

The LOESS line $s(x_0)$ using k nearest neighbours is computed in four steps:[116]

(i) The k nearest neighbours of x_0 are identified, denoted by $N(x_0)$.

(ii) $\Delta(x_0) = \max_{N(x0)} |x_0 - x_i|$ is computed, the distance of the furthest near-neighbour from x_0.

(iii) Weights w_i are assigned to each point in $N(x_0)$, using the following tri-cube weight function:

$$W\left(\frac{|x_0 - x_i|}{\Delta(x_0)}\right) \tag{39}$$

where:

$$W(u) = \begin{cases} (1 - u^3)^3, & \text{for } 0 \leq u \leq 1 \\ 0 & \text{otherwise} \end{cases}$$

(iv) $s(x_0)$ is the fitted value at x_0 from the weighted least-squares fit of y to x confined to $N(x_0)$ using weights computed in (iii).

The following figure illustrates the daily mortality in London plotted versus time, where each dot is the count of deaths on one of the 2192 days in the study for

---

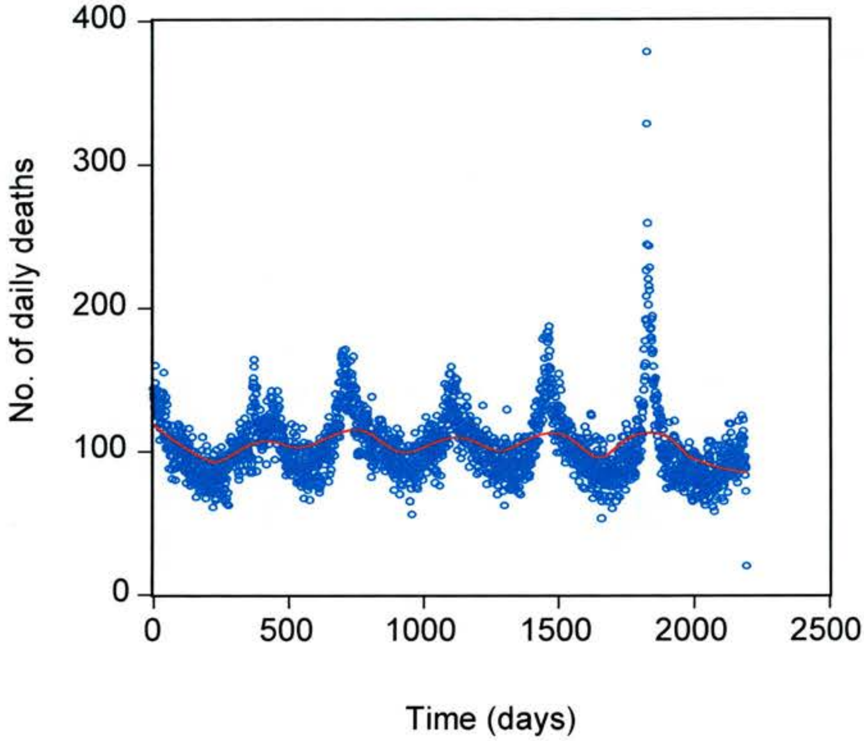
114 The polynomial degree specifies the degree of polynomial to fit in each local regression.

115 Tricube or local weighing, weights the observations of each local regression, whereby the weighted regression minimises the weighted sum of squared residuals $\sum_{i=1}^{N} \omega_i \left(y_i - a - b_i x_i - b_2 x_i^2 - \ldots - b_k x_i^k\right)^2$, where the tricube weights ϖ are given by $\omega_i = \left(1 - |d_i / d([\alpha N])|^3\right)^3$ for $|d_i / d([\alpha N])| < 1$ and 0 otherwise, with $d_I = |x_i - x|$ and d $([\alpha N])$ is the $[\alpha N]$-the smallest such distance. In other words, those observations that are relatively far from the data point being evaluated get small weights in the sum of squared residuals.

116 See: Hastie and Tibshirani (1990), pp. 29-30.

London. The smoothed curve using non-parametric smoothing regressions indicates a substantial seasonal pattern in the mortality data, which was also found in the data for all other cities included in this study.[117]

Figure 50: Daily Death Counts 1992-1997 in London.



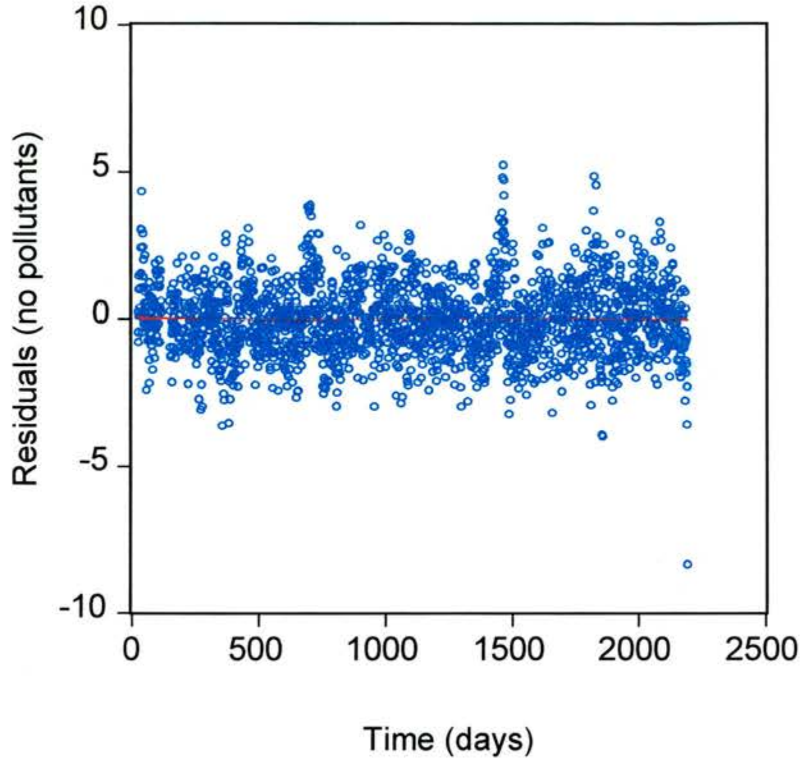
A further scatter plot was produced after adjusting the number of daily deaths in London for each of the 2192 days in the study. The adjustment was done by Poisson regression with GLM correction, using all of the covariates as described above, except air pollution. The same non-parametric smoother (LOESS) was fitted to this adjusted data as to the unadjusted data described before. Looking at the following figure, the remaining variability in the mortality data no longer has a seasonal component.[118]

---

117 The respective figures for all other cities are produced in Appendix 2.

118 The same scatter plots were generated for all 14 cities examined in this study and are presented in Appendix 2.

Figure 51: Residuals of Core Model for London.



Further scatter plots were generated where the mortality residuals of the core model (without pollutants) are plotted against mean temperature. These plots served as a further check whether there was any range of temperature (e.g. cold or hot temperatures) where the core model does not fit well to the data. The same procedure was undertaken for relative humidity. Both scatter plots for each city examined are presented in Appendix 3.

After the core model that best fitted the data, controlling for all confounders described above, had been established, the daily counts of deaths were then matched to the air pollution data. Here, single pollutant regressions have been applied; the reason being the potential interactions between pollutants that may bias the estimation results. The correlation matrices presented in Appendix 1 indicate strong multi-collinearity between the pollutants comprised in this study.

Following Schwartz et al. (1996):

> One occasionally sees studies that have fitted regression models using four or even more collinear pollutants in the same regression, and sought to draw inferences about which variables were causal from that model. Other studies have used stepwise procedures, again with many pollutants in the candidate list. Given the non-trivial correlation of the pollutant variables, and the relatively low explanatory power of air pollution for mortality or hospital admission, such procedures risk letting the noise in the data choose the pollutant.[119]

Thus, following this argumentation, the present study concentrates on the analysis of the effects of individual pollutants by applying single pollutant regression models. The results of these Poisson regression estimates are presented for each individual city in this study, in the following chapter.

9.5 Summary

The purpose of this chapter was twofold; first, to discuss the methodological issues that arise with the analysis of acute mortality effects associated with daily concentrations of air pollution, and second, to establish a sound regression model for estimating these effects. The time-series of the type analysed in this study are approximately Poisson distributed, overdispersed, i.e. the variance is greater than the mean, and positively autocorrelated. However, the latter two characteristics are usually a result of extraneous factors, such as weather and seasonally, and less an intrinsic feature of the daily death counts. In other words, after adequately controlling for these confounding factors, these features usually do not cause any severe problems.

---

119 Schwartz et al. (1996), p. S8.

Special considerations were made regarding the control for seasonally and other cyclical patterns. It was decided, rather than pre-filtering the data, to employ a parametric approach by including sinusoidal terms in the regression model. Such trigonometric filtering leaves the count data 'unchanged', still allowing the application of multiple Poisson regressions. Further, controlling for potential confounding variables has been made by including terms for meteorological factors (that is temperature, humidity, interactions), long term trends, calendar effects (that is season, day of week, holidays), and influenza epidemics.

Based on these discussions, a baseline or core model was constructed, analysing the association of the daily number of deaths with all potential confounding variables as described above. The modelling procedure followed closely the one suggested by the APHEA project. This procedure included modelling for all potential confounding factors, including seasonal and long term patters, meteorological factors, day of the week, holidays, and other unusual events such as influenza epidemics. The model that best fitted the data was identified by fitting non-parametric LOESS functions of the fully adjusted residuals in order to investigate whether any cyclical patterns were still present. After identifying the best fitting 'core' model, air pollution indicators were included into the final model. The results of the final multiple Poisson regression analyses are provided in the following chapter.

10. Presentation and Discussion of the Estimation Results

10.1 Introduction

In this chapter, the parameter estimates of the multiple Poisson model regressions, asdescribed in the previous chapter are presented and discussed. This chapter is divided in three main sections. The first section is further separated into city specific sub-sections. In each of these sub-sections, first, the parameter estimates for the core model, and second, the estimation results of the final multiple Poisson regression model including air pollution are presented. The second section will provide a summary of the main results, which are then be discussed in the final section.

Following the detailed description presented in the previous chapter, core models were specified and tested. For each city these core models were tested separately and amended if a better fit to the data could be achieved. The parameter estimates for the respective core model variables from a Poisson model without air pollution are presented in a tabular form. To illustrate how well the chosen core model fits to the data, time plots of residuals of the examined outcome variable including all potential confounding variables are also presented for each city. As described in the previous chapter, additional non-parametric smooth lines (LOESS) have been drawn on the plot, in order investigate whether there was any evidence of a temporal pattern remaining in the data.[1]

Having established the core model, air pollution was included into the model and tested to examine whether its inclusion improved the overall fit of the model.

[1] See: Appendix 2.

Presenting the results of the multiple regression estimations can be performed in various ways. The statistic chosen in this study to summarise the findings, is the relative risk function (RR).[2] Such a relative risk function at level x of the independent variable can be defined as follows:[3]

$$RR(x) = \frac{E(Y|\, x, z)}{E(Y|\, 0, z)} \tag{40}$$

This is the ratio of the expected number of the end points at level x of the independent variable to the expected number of end points if the independent variable were 0. Since the Poisson regression model is a relative risk model, these relative risk factor are simply obtained as:

$$\mathrm{RR}(\Delta) = \mathrm{e}^{(\beta * \Delta)} \tag{41}$$

where: β = estimated coefficient.

Δ = risk increase of interest.

In air pollution studies, the risk increase of interest (Δ) is the increment in pollution exposure. Various such deltas have been applied in the relevant literature, including a fixed amount of pollution (e.g. 100 $\mu g/m^3$, 50 $\mu g/m^3$, 10 $\mu g/m^3$, etc.),[4] the inter-quartile range of the respective pollutant,[5] the 10^{th} to 90^{th} percentile range,[6] or the range 5^{th} to 95^{th} percentile[7].

---

[2] Alternative statistics, besides the estimated regression coefficient that have been used in various studies include the slope of the fitted regression line at a specific point, the elasticity of the fitted regression line, and the estimated odds ratio. For a detailed discussion see: Baxter et al. (1997), Calthrop and Maddison (1996), Aunan (1996) and Lipfert (1993).

[3] See: Baxter et al. (1997), pp. 274-275.

[4] See, for instance, Dab et al. (1996) and Zmirou et al. (1996).

[5] See, for instance, Vigotti et al. (1996).

[6] See, for instance, Ponce de Leon et al. (1996).

[7] See, for instance, Spix and Wichmann (1996).

Whilst each of these procedures are acceptable, it should be noted that deltas which go beyond the range of air pollution that is actually observed are seen as being problematic.

In this study, the relative risk is calculated in two ways. First, the regression results are expressed using the relative risk factors for the range of the 5^{th} to the 95^{th} percentile. In other words, the relative risk is given comparing the 95^{th} percentile of the respective pollution data with the 5^{th} percentile of the data. The resulting relative risk factors express the percentage increase in mortality, assuming an increase in pollution concentration from the 5^{th} percentile to the 95^{th} percentile. Regarding the current air pollution levels, increments of 100 $\mu g/m^3$ are rather large and may even lie outside the maximum concentration level measured. Hence, the illustrated way of expressing the estimation results may be more pertinent, since concentration levels of air pollution actually measured are taken into account leaving out extreme values.

Second, as an alternative way to express the estimation results, the relative risk for a 100 $\mu g/m^3$ increase in the respective pollutant is also calculated. This has been chosen in order to make the results easier to compare with previous studies. For the review given in Chapter 7, estimation results of previous studies had been re-calculated for a delta of 100 $\mu g/m^3$ if they had been expressed differently, in order to make the results more comparable across the different locations analysed. Similarly, the changes associated with a rise in CO concentration of 10 ppm are presented. These results are presented in Appendix 5.

10.2 Results of the Poisson Estimations

In this section, the estimated results for each city are presented. First, the parameter estimates of the core model, without air pollution variables are summarised. For the case where air quality pollution data was not available for the entire study period (1992-1997), the core model was amended in so far, that the number of annual indicator variables had been reduced to match the length of the respective air pollution data.

Second, the results of the final multiple Poisson regression models, controlling for all potential confounders are produced. In order to ensure that the model was robust, days with extreme values for the respective pollutants were excluded, with all days exceeding the 95^{th} percentile, or below the 5^{th} percentile for the pollution concentrations mean temperature omitted.[8]

In addition to the estimated coefficient, the standard errors, and the relative risk factor, as well as the time lag for each pollutant which represented best the delayed effect between exposure to air pollution and health impact, i.e. maximised the overall model fit according to the AIC. Since potential health effects caused by air pollution are generally assumed to be very small or not identifiable at all, hypothesis tests results for both the 5 per cent and the 10 per cent significance level are reported.[9] Also, for those regression coefficients that are significant at least at the 5 per cent level, the 95 per cent confidence interval (CI) has been computed and is presented in brackets.

---

[8] By using mean daily temperature, rather than maximum daily temperature already reduces the risk that the regression results are driven by these extreme values.

[9] This is in line with similar studies. See, for instance, Spix and Wichmann (1996).

For the case where the estimated regression coefficient was significant on the 10 per cent level the 90 per cent confidence interval is reported respectively. For the case where the estimated coefficient was not significant on either significance level, only the coefficient and respective standard error are reported, but no relative risk factor.

The following section is subdivided in three sub-sections. First, the results of the core model are given. Second, a summary of the pollution ranges of the respective air pollutants for each city studied is produced. This information is then used to finally compute the relative risk factors, taking the unchanged regression coefficients into account.

10.2.1 Birmingham

10.2.1.1 Parameter Estimates of the Core Model

Table 94: Parameter Estimates of the Core Model for Birmingham.

Variable	Coefficient	Standard Error
C	2.871513	0.132923
Trend	0.000209	0.000140
Squared Trend	-1.94E-07	4.30E-08
2-year sine-cosine wave	0.020514	0.007006
1-year sine-cosine wave	0.091697	0.018519
6-month sine-cosine wave	0.007278	0.12176
4-month sine-cosine wave	0.022259	0.007632
3-month sine-cosine wave	0.013530	0.006125
2.4-month sine-cosine wave	0.012520	0.006562
2-month sine-cosine wave	0.017242	0.006764
Dummy 1992	Reference category	
Dummy 1993	0.168630	0.042058
Dummy 1994	0.113348	0.071738
Dummy 1995	0.256591	0.097751
Dummy 1996	0.294064	0.118773
Dummy 1997	0.387602	0.142095
Dummy Winter	-0.039457	0.027813
Dummy Spring	Reference category	
Dummy Summer	0.007319	0.039832
Dummy Autumn	0.142468	0.025097
Dummy Sunday	-0.016888	0.020018
Dummy Monday	Reference category	
Dummy Tuesday	0.009022	0.019819
Dummy Wednesday	-0.013158	0.020163
Dummy Thursday	0.015388	0.020135
Dummy Friday	-0.001080	0.020266
Dummy Saturday	0.000654	0.020325
Dummy Epidemics (Lag 7)	0.306041	0.029143
Dummy Epidemics2	0.192547	0.051362
Dummy Epidemics3	0.641786	0.065459
Dummy Epidemics4	0.990169	0.098580
Dummy Epidemics5	0.184016	0.085257
Mean Temperature (Lag 1)	-0.021047	0.003461
Squared mean Temperature (Lag 1)	0.000670	0.000117
Mean rel. Humidity (Lag 1)	0.001205	0.000669
Dummy Hot & Humid	0.044931	0.025491

10.2.1.2 Pollutant Range applied for Birmingham

Table 95: Range of Pollutants for which Relative Risk is calculated.

Pollutant	5th percentile	95th percentile	Range (5th - 95th percentile)
PM_{10}	10.0	51.0	41.0
NO_2	17.82	81.18	63.36
SO_2	5.52	60.72	55.2
CO	0.3	1.15	0.85
O_3	6.21	66.24	60.03

10.2.1.3 Estimation Results of the Final Model

Table 96: Estimation Results of the Final Poisson Model for Birmingham.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.000903*	1	0.000458	1.04 (1.00; 1.08)
NO_2	0.000797*	1	0.000336	1.05 (1.00; 1.10)
SO_2	0.000542*	1	0.000283	1.03 (0.99; 1.62)
CO	0.016780**	1	0.009239	1.01 (1.00; 1.03)‡
O_3	0.000888*	1	0.000314	1.05 (1.02; 1.09)

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

For Birmingham, the strongest relationship on mortality was found for both NO_2 and ozone. The relative risk for the range between the 5^{th} and 95^{th} percentile, both NO_2 and ozone predicted a 5 per cent increase in mortality, followed by PM_{10} with 4 per cent and SO_2 with 3 per cent. For carbon monoxide, the result was less significant and showed an increase of 1 per cent for the chosen range. All pollutants fitted best to the model for 1 day lags. The results of the estimations are like those found in a similar study by Wordley et al.[10]

10.2.2 Bristol

10.2.2.1 Parameter Estimates of the Core Model

Table 97: Parameter Estimates of the Core Model for Bristol.

Variable	Coefficient	Standard Error
C	1.913073	0.104099
Trend	0.000226	0.000213
Squared Trend	-9.01E-08	6.32E-08
2-year sine-cosine wave	0.022312	0.010967
1-year sine-cosine wave	0.096297	0.027273
6-month sine-cosine wave	0.048092	0.016454
4-month sine-cosine wave	0.037404	0.010117
3-month sine-cosine wave	0.022485	0.008792
2.4-month sine-cosine wave	-0.004953	0.009239
2-month sine-cosine wave	0.039529	0.009659
Dummy 1992	Reference category	
Dummy 1993	0.117416	0.072769
Dummy 1994	-0.055878	0.129209
Dummy 1995	0.014281	0.179860
Dummy 1996	-0.096675	0.228904
Dummy 1997	-0.032918	0.278483
Dummy Winter	-0.064844	0.042355
Dummy Spring	Reference category	
Dummy Summer	-0.010713	0.057407
Dummy Autumn	0.141544	0.041490
Dummy Sunday	-0.058353	0.31026
Dummy Monday	Reference category	
Dummy Tuesday	-0.039671	0.030909
Dummy Wednesday	-0.061467	0.031083
Dummy Thursday	-0.063142	0.031049
Dummy Friday	-0.009487	0.030594
Dummy Saturday	-0.029235	0.030779
Dummy Epidemic (Lag 1)	0.369945	0.053925
Dummy Epidemics2	0.525437	0.230424
Dummy Epidemics3	1.247258	0.176560
Mean Temperature (Lag 2)	-0.024560	0.006714
Squared mean Temperature (Lag 2)	0.0007778	0.000300
Mean rel. Humidity (Lag 2)	0.000330	0.000950

[10] See: Wordley, Walters and Ayres (1997).

10.2.2.2 Pollutant Range applied for Bristol

Table 98: Range of Pollutants for which Relative Risk is calculated.

Pollutant	5th percentile	95th percentile	Range (5th - 95th percentile)
PM_{10}	11.0	50.0	39.0
NO_2	19.8	85.14	65.34
SO_2	5.52	35.88	30.36
CO	0.2	1.5	1.3
O_3	8.28	64.17	55.89

10.2.2.3 Estimation Results of the Final Model

Table 99: Estimation Results of the Final Poisson Model for Bristol.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001024**	2	0.000581	1.04 (1.01; 1.08)‡
NO_2	0.000879*	0-3	0.000227	1.06 (1.04; 1.08)
SO_2	0.001826*	0-3	0.000927	1.06 (1.04; 1.08)
CO	0.009621*	1	0.004730	1.01 (1.00; 1.03)
O_3	0.000180	1	0.000459	N/A

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

The estimation results for Bristol show the strongest positive association between both, NO_2, and SO_2 and mortality, with an increase of 6 per cent. For both pollutants, the strongest relationships were found when using the moving average concentrations of the previous three days. Carbon monoxide also showed a positive significant increase of 1 per cent. Particulates are also positively but less significant associated with mortality (4% increase). For ozone a positive but not significant association was estimated.

10.2.3 Cardiff

10.2.3.1 Parameter Estimates of the Core Model

Table 100: Parameter Estimates of the Core Model for Cardiff.

Variable	Coefficient	Standard Error
C	1.510702	0.113303
Trend	5.06E-05	0.000152
2-year sine-cosine wave	0.028777	0.013584
1-year sine-cosine wave	0.114413	0.034962
6-month sine-cosine wave	0.008293	0.020965
4-month sine-cosine wave	0.038550	0.012722
3-month sine-cosine wave	0.007914	0.010849
2.4-month sine-cosine wave	0.019445	0.011476
2-month sine-cosine wave	0.016467	0.011945
Dummy 1992	Reference category	
Dummy 1993	0.178858	0.068486
Dummy 1994	0.029786	0.116185
Dummy 1995	0.123437	0.174858
Dummy 1996	0.035958	0.226432
Dummy 1997	-0.029529	0.282728
Dummy Winter	0.028216	0.057841
Dummy Spring	Reference category	
Dummy Summer	0.104812	0.067700
Dummy Autumn	0.146202	0.048015
Dummy Sunday	-0.027647	0.040068
Dummy Monday	Reference category	
Dummy Tuesday	-0.025724	0.038716
Dummy Wednesday	-0.019637	0.038693
Dummy Thursday	-0.010497	0.038634
Dummy Friday	0.044729	0.038715
Dummy Saturday	-0.013761	0.039396
Dummy Epidemics (Lag 6)	0.245231	0.067476
Mean Temperature (Lag 2)	-0.031682	0.008026
Squared mean Temperature (Lag 2)	0.001045	0.000344
Mean rel. Humidity (Lag 2)	0.000846	0.001130

10.2.3.2 Pollutant Range applied for Cardiff

Table 101: Range of Pollutants for which Relative Risk is calculated.

Pollutant	5^{th} percentile	95^{th} percentile	Range (5^{th} - 95^{th} percentile)
PM_{10}	13.0	49.0	36.0
NO_2	21.78	67.32	45.54
SO_2	5.52	33.12	27.6
CO	0.3	1.3	1.0
O_3	6.21	66.24	60.03

10.2.3.3 Estimation Results of the Final Model

Table 102: Estimation Results of the Final Poisson Model for Cardiff.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001747**	1	0.001035	1.06 (1.00; 1.13)‡
NO_2	0.001612*	1	0.000817	1.08 (1.00; 1.16)
SO_2	-0.000661	2	0.001102	N/A
CO	0.018871**	2	0.010743	1.02 (1.00; 1.04)‡
O_3	0.000836	2	0.000722	N/A

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

The estimation results for Cardiff are somewhat mixed. A strong positive relationship could be found for nitrogen dioxide (8% increase). Both, particulates and carbon monoxide showed a positive but less significant effect with an increase of 6 % and 2 % respectively. For SO_2, a negative and insignificant relationship was found, while the effect for ozone is positive but also insignificant.

10.2.4 Edinburgh

10.2.4.1 Parameter Estimates of the Core Model

Table 103: Parameter Estimates of the Core Model for Edinburgh.

Variable	Coefficient	Standard Error
C	2.049517	0.096832
Trend	88.25E-05	0.000193
Squared Trend	-8.36E-08	5.60E-08
2-year sine-cosine wave	0.035990	0.009706
1-year sine-cosine wave	0.120781	0.0245569
6-month sine-cosine wave	0.043630	0.013962
4-month sine-cosine wave	0.030514	0.008949
3-month sine-cosine wave	0.012772	0.007776
2.4-month sine-cosine wave	0.015008	0.008195
2-month sine-cosine wave	0.032889	0.008574
Dummy 1992	Reference category	
Dummy 1993	0.083153	0.065671
Dummy 1994	-0.070685	0.117809
Dummy 1995	0.043154	0.165104
Dummy 1996	0.005940	0.210604
Dummy 1997	0.090042	0.255942
Dummy Winter	-0.050878	0.037360
Dummy Spring	Reference category	
Dummy Summer	0.008163	0.051169
Dummy Autumn	0.120640	0.036370
Dummy Sunday	0.008768	0.27666
Dummy Monday	Reference category	
Dummy Tuesday	-0.002675	0.027731
Dummy Wednesday	0.024884	0.027538
Dummy Thursday	0.016623	0.027609
Dummy Friday	0.029541	0.027493
Dummy Saturday	-0.014671	0.027817
Dummy Holidays	-0.036459	0.021247
Dummy Epidemics (Lag 1)	0.284306	0.048707
Mean Temperature (Lag 1)	-0.007109	0.003006
Mean rel. Humidity (Lag 2)	0.000672	0.001008

10.2.4.2 Pollutant Range applied for Edinburgh

Table 104: Range of Pollutants for which Relative Risk is calculated.

Pollutant	5^{th} percentile	95^{th} percentile	Range (5^{th} - 95^{th} percentile)
PM_{10}	10.0	37.0	27.0
NO_2	29.7	77.22	47.52
SO_2	2.76	44.16	41.4
CO	0.3	1.0	0.7
O_3	8.28	53.82	45.54

10.2.4.3 Estimation Results of the Final Model

Table 105: Estimation Results of the Final Poisson Model for Edinburgh.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001972*	3	0.000940	1.05 (1.00; 1.10)
NO_2	0.001088**	2	0.000605	1.05 (1.01; 1.10)‡
SO_2	0.001213*	3	0.000612	1.05 (1.00; 1.11)
CO	0.054420*	1	0.027197	1.04 (1.00; 1.08)
O_3	-0.001592*	0	0.000629	0.93 (0.88; 0.98)

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

The estimation results for Edinburgh showed a significant association with mortality for all pollutants examined. Increments (range 5^{th} to 95^{th} percentile) with PM_{10} and SO_2 were significantly associated with a 5 per cent increase in mortality. Carbon monoxide also showed a positive and significant relationship of 4 per cent, while the association of nitrogen dioxide was less significant, but positive (5%). On the other hand, in respective increments in ozone on the same day were significantly but negative associated with increased in mortality. These results are in line with those of a similar study.[11]

10.2.5 Glasgow

10.2.5.1 Parameter Estimates of the Core Model

Table 106: Parameter Estimates of the Core Model for Glasgow.

Variable	Coefficient	Standard Error
C	2.673204	0.087876
Trend	0.000481	0.000168
Squared Trend	-1.57E-07	5.7E-08
2-year sine-cosine wave	0.031296	0.007816
1-year sine-cosine wave	0.072346	0.019388
6-month sine-cosine wave	0.041617	0.011530
4-month sine-cosine wave	0.013995	0.007342
3-month sine-cosine wave	0.007353	0.006429
2.4-month sine-cosine wave	0.023575	0.007015
2-month sine-cosine wave	0.017131	0.007330
Dummy 1992	Reference category	
Dummy 1993	0.006279	0.056334
Dummy 1994	-0.249819	0.097077
Dummy 1995	-0.244043	0.138306
Dummy 1996	-0.335124	0.171229
Dummy 1997	-0.378203	0.244342
Dummy Winter	0.040686	0.048631
Dummy Spring	Reference category	
Dummy Summer	0.073657	0.041874
Dummy Autumn	0.123279	0.035957
Dummy Sunday	-0.035642	0.023577
Dummy Monday	Reference category	
Dummy Tuesday	-0.045564	0.022579
Dummy Wednesday	-0.008773	0.021992
Dummy Thursday	-0.083254	0.022966
Dummy Friday	-0.034856	0.021943
Dummy Saturday	-0.047651	0.022596
Dummy Epidemics (Lag 8)	0.262929	0.118361
Mean Temperature (Lag 2)	-0.010823	0.002187
Mean rel. Humidity Lag1)	-0.000460	0.000845
Dummy Hot & Humid	0.052487	0.046407

[11] See: Prescott et al. (1998).

10.2.5.2 Pollutant Range applied for Glasgow

Table 107: Range of Pollutants for which Relative Risk is calculated.

Pollutant	5^{th} percentile	95^{th} percentile	Range (5^{th} - 95^{th} percentile)
PM_{10}	10.0	38.7	28.7
NO_2	23.76	81.18	57.42
SO_2	2.76	24.84	22.08
CO	0.3	1.8	1.5
O_3	4.14	55.89	51.75

10.2.5.3 Estimation Results of the Final Model

Table 108: Estimation Results of the Final Poisson Model for Glasgow.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}†	0.001117**	2	0.000638	1.03 (1.00; 1.06)‡
NO_2	0.000363*	2	0.000139	1.02 (1.01; 1.04)
SO_2	N/A			N/A
CO	0.019680**	2	0.011719	1.03 (1.00; 1.06)‡
O_3	N/A			N/A

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
†: Data based on BS, converted to PM_{10} using formula PM_{10} = BS x 0.9.
‡: 90 per cent confidence interval.

Due to a significantly large number of missing values for SO_2 and O_3, no estimation for these pollutants was possible for Glasgow. Likewise, the data available for PM_{10} was equally incomplete and qualitatively not good enough to be included in the estimation. However, as a 'substitute' time-series, data for Black Smoke (BS) could be obtained from the AIRBASE run by the European

Environment Agency and European Commission Directorate General XI-33. AIRBASE contains meta information on air quality monitoring networks and stations and on air quality data for a selection of stations of components.[12] The estimation result was then transformed into PM_{10} measures. Both BS and CO estimates showed an increase of 3 per cent at the 10 per cent significance level. The estimation result for NO_2 indicated a more significant increase of 2 per cent in mortality.

10.2.6 Kingston upon Hull

10.2.6.1 Parameter Estimates of the Core Model

Table 109: Parameter Estimates of the Core Model for Kingston upon Hull.

Variable	Coefficient	Standard Error
C	1.020648	0.131778
Trend	0.001526	0.000229
Squared Trend	-3.6E-07	7.89E-08
2-year sine-cosine wave	0.061119	0.013488
1-year sine-cosine wave	0.159562	0.021513
6-month sine-cosine wave	0.054996	0.011120
4-month sine-cosine wave	0.014559	0.010887
3-month sine-cosine wave	0.009230	0.010570
2.4-month sine-cosine wave	0.033711	0.010460
2-month sine-cosine wave	0.019350	0.010477
Dummy 1992	Reference category	
Dummy 1993	-0.340521	0.074862
Dummy 1994	-0.759877	0.125519
Dummy 1995	-0.945030	0.166883
Dummy 1996	-1.241108	0.201970
Dummy 1997	-1.233640	0.240591
Dummy Sunday	-0.057318	0.038128
Dummy Monday	Reference category	
Dummy Tuesday	-0.042982	0.038025
Dummy Wednesday	-0.083657	0.038360
Dummy Thursday	-0.049400	0.038044
Dummy Friday	-0.006995	0.037742
Dummy Saturday	-0.037394	0.038139
Dummy Epidemics (Lag 3)	0.174840	0.066483
Mean Temperature (Lag 3)	-0.005307	0.003654
Mean rel. Humidity (Lag 2)	0.002543	0.001319

[12] See: AIRBASE website at: http:// etcaq.rivm.nl.

10.2.6.2 Pollutant Range applied for Kingston upon Hull

Table 110: Range of Pollutants for which Relative Risk is calculated.

Pollutant	5th percentile	95th percentile	Range (5th - 95th percentile)
PM_{10}	12.0	48.0	36.0
NO_2	21.76	71.28	49.52
SO_2	5.52	46.92	41.4
CO	0.3	1.2	0.9
O_3	8.28	62.1	53.82

10.2.6.3 Estimation Results of the Final Model

Table 111: Estimation Results of the Final Poisson Model for Kingston upon Hull.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001320*	1	0.000671	1.05 1.00; 1.11)
NO_2	0.000636**	2	0.000367	1.03 (0.99; 1.07)‡
SO_2	0.000420	2	0.000986	N/A
CO	0.009708*	1	0.004925	1.01 (1.00; 1.02)
O_3	0.001529**	0	0.000880	1.09 (1.05; 1.14)‡

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

For Kingston upon Hull, the association between particulates and mortality indicated a 5 per cent increase for the centile range shown in the table above, while CO showed a 1 per cent increase at the same 5 per cent significance level. For nitrogen dioxide and ozone a positive, but less significant relationship was found (3 % and 9 % respectively). No significant estimation results could be obtained for sulphur dioxide.

10.2.7 Leeds

10.2.7.1 Parameter Estimates of the Core Model

Table 112: Parameter Estimates of the Core Model for Leeds.

Variable	Coefficient	Standard Error
C	2.338682	0.077922
Trend	0.000420	0.000164
Squared Trend	-8.8E-08	4.96E-08
2-year sine-cosine wave	0.027531	0.008464
1-year sine-cosine wave	0.153770	0.021513
6-month sine-cosine wave	0.046707	0.012310
4-month sine-cosine wave	0.030854	0.007959
3-month sine-cosine wave	0.023738	0.006882
2.4-month sine-cosine wave	0.024812	0.007242
2-month sine-cosine wave	0.010901	0.007614
Dummy 1992	Reference category	
Dummy 1993	0.010583	0.056039
Dummy 1994	-0.186918	0.098001
Dummy 1995	-0.246472	0.136773
Dummy 1996	-0.310202	0.173747
Dummy 1997	-0.391613	0.211293
Dummy Winter	-0.092457	0.051995
Dummy Spring	Reference category	
Dummy Summer	-0.076640	0.045234
Dummy Autumn	0.054911	0.039081
Dummy Sunday	-0.071964	0.024521
Dummy Monday	Reference category	
Dummy Tuesday	-0.026717	0.024224
Dummy Wednesday	-0.049938	0.024366
Dummy Thursday	-0.009752	0.024144
Dummy Friday	-0.011934	0.024152
Dummy Saturday	-0.033058	0.024289
Dummy Epidemics (Lag 5)	0.289901	0.042740
Mean Temperature (Lag 3)	-0.002664	0.002428
Squared mean Temperature (Lag 5)	-0.000199	0.000117
Dew Point Temperature (Lag 5)	-0.006012	0.002310

10.2.7.2 Pollutant Range applied for Leeds

Table 113: Range of Pollutants for which Relative Risk is calculated.

Pollutant	5^{th} percentile	95^{th} percentile	Range (5^{th} - 95^{th} percentile)
PM_{10}	12.0	58.0	46.0
NO_2	29.7	83.16	53.46
SO_2	2.76	66.24	63.48
CO	0.3	1.3	1.0
O_3	6.21	55.89	49.68

10.2.7.3 Estimation Results of the Final Model

Table 114: Estimation Results of the final Poisson Model for Leeds.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	-0.000650	1	0.000553	N/A
NO_2	0.000683	3	0.000500	N/A
SO_2	0.000598*	3	0.000294	1.04 (1.00; 1.08)
CO	0.038416*	2	0.018244	1.04 (1.00; 1.08)
O_3	-0.000820**	3	0.000423	0.96 (0.93; 0.99)‡

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

The estimation results for Leeds show a mixed association of air pollution and mortality. No statistically significant relationship could be found either for PM_{10} or for NO_2. Two other pollutants, SO_2 and CO, indicated an increase of 4 per cent each time the their concentrations increase by the centile range stated above. Ozone showed a negative relationship that is significant on the 10 per cent level.

10.2. 8 Leicester

10.2.8.1 Parameter Estimates of the Core Model

Table 115: Parameter Estimates of the Core Model for Leicester.

Variable	Coefficient	Standard Error
C	1.721657	0.119498
Trend	-0.00184	0.000192
2-year sine-cosine wave	0.013764	0.013161
1-year sine-cosine wave	0.130031	0.031283
6-month sine-cosine wave	0.056769	0.019555
4-month sine-cosine wave	0.024227	0.012059
3-month sine-cosine wave	0.041734	0.010274
2.4-month sine-cosine wave	0.006444	0.010925
2-month sine-cosine wave	0.040331	0.011471
Dummy 1992	Reference category	
Dummy 1993	0.172772	0.043180
Dummy 1994	0.071419	0.056450
Dummy 1995	0.119432	0.096735
Dummy 1996	0.151972	0.147405
Dummy 1997	0.195723	0.219053
Dummy Winter	-0.051536	0.049370
Dummy Spring	Reference category	
Dummy Summer	-0.090213	0.067615
Dummy Autumn	0.145713	0.045301
Dummy Sunday	-0.058530	0.036618
Dummy Monday	Reference category	
Dummy Tuesday	-0.017000	0.038677
Dummy Wednesday	-0.070432	0.036865
Dummy Thursday	-0.035568	0.036470
Dummy Friday	-0.02974	0.036009
Dummy Saturday	-0.000109	0.036076
Dummy Epidemics (Lag 1)	0.270573	0.063450
Mean Temperature (Lag 1)	-0.019151	0.006087
Squared mean Temperature (Lag 1)	0.001168	0.000317
Mean rel. Humidity (Lag 2)	-0.001266	0.001248

10.2.8.2 Pollutant Range applied for Leicester

Table 116: Range of Pollutants for which Relative Risk is calculated.

Pollutant	5^{th} percentile	95^{th} percentile	Range (5^{th} - 95^{th} percentile)
PM_{10}	9.4	41.0	33.16
NO_2	21.78	73.26	51.48
SO_2	5.76	41.64	35.88
CO	0.2	1.2	1.0
O_3	6.21	68.31	62.1

10.2.8.3 Estimation Results of the Final Model

Table 117: Estimation Results of the final Poisson Model for Leicester.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.000297*	2	0.000147	1.01 (1.00; 1.02)
NO_2	0.000690	1	0.000844	N/A
SO_2	0.001057	1	0.000775	N/A
CO	0.023693	1	0.030465	N/A
O_3	0.000188**	1	0.000104	1.01 (1.00; 1.02)‡

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

The estimations for Leicester show relatively weak associations between the air pollutants examined and mortality. No statistically significant association could be found for NO_2, SO_2, and CO. Particulates showed a small but significant increase in mortality (1%), as did ozone, but being only significant on the 10 per cent level. The relatively weak results may be due to the very small number of cardio-respiratory deaths occurring per day during the study period.

10.2.9 Liverpool

10.2.9.1 Parameter Estimates of the Core Model

Table 118: Parameter Estimates of the Core Model for Liverpool.

Variable	Coefficient	Standard Error
C	2.156072	0.091712
Trend	0.000458	9.62E-05
2-year sine-cosine wave	0.030878	0.009268
1-year sine-cosine wave	0.069338	0.015929
6-month sine-cosine wave	0.036940	0.007983
4-month sine-cosine wave	0.018921	0.007678
3-month sine-cosine wave	0.030443	0.007476
2.4-month sine-cosine wave	0.023061	0.007403
2-month sine-cosine wave	0.006693	0.007352
Dummy 1992	Reference category	
Dummy 1993	-0.024946	0.043749
Dummy 1994	-0.298100	0.074792
Dummy 1995	-0.401417	0.110367
Dummy 1996	-0.677089	0.141907
Dummy 1997	-0.872941	0.177181
Dummy Sunday	-0.036241	0.027192
Dummy Monday	Reference category	
Dummy Tuesday	0.015741	0.026879
Dummy Wednesday	-0.024819	0.027099
Dummy Thursday	0.000134	0.026958
Dummy Friday	0.002253	0.026942
Dummy Saturday	0.012402	0.026851
Dummy Epidemics (Lag 1)	0.282391	0.047250
Dummy Epidemics 2	0.697314	0.177694
Mean Temperature (Lag 1)	-0.016873	0.002719
Mean rel. Humidity (Lag 1)	0.001345	0.001009
Dummy Hot & Humid	0.128104	0.048476

10.2.9.2 Pollutant Range applied for Liverpool

Table 119: Range of Pollutants for which Relative Risk is calculated.

Pollutant	**5^{th} percentile**	**95^{th} percentile**	**Range (5^{th} - 95^{th} percentile)**
PM_{10}	11.0	52.0	41.0
NO_2	17.82	86.33	68.51
SO_2	2.76	82.8	80.04
CO	0.2	1.0	0.8
O_3	6.21	66.24	60.03

10.2.9.3 Estimation Results of the Final Model

Table 120: Estimation Results of the Final Poisson Model for Liverpool.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001254*	0-2	0.000629	1.05 (1.00; 1.10)
NO_2	0.001173**	0-3	0.000643	1.08 (1.00; 1.15)‡
SO_2	0.001181*	0-3	0.000452	1.10 (1.05; 1.16)
CO	0.060836*	2	0.029564	1.05 (1.00; 1.10)
O_3	-0.000781	1	0.000611	N/A

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

The estimation results for Liverpool show a fairly strong association between air pollution and mortality. The mean average of PM_{10} for the previous 2 days was associated with a 5% increase in mortality. The same increase was found for carbon monoxide, while the strongest impact was found for SO_2 (10 %) when examining the mean concentration of the previous three days. A similarly large impact of 8 per cent was found for nitrogen dioxide, although only significant at the 10 per cent level. No statistically significant association could be found for ozone.

10.2.10 London

10.2.10.1 Parameter Estimates of the Core Model

Table 121: Parameter Estimates of the Core Model for London.

Variable	Coefficient	Standard Error
C	4.593970	0.031919
Trend	0.000231	7.02E-05
Squared Trend	-8.83E-08	2.07E-08
2-year sine-cosine wave	0.018228	0.003478
1-year sine-cosine wave	0.137651	0.008833
6-month sine-cosine wave	0.023965	0.005271
4-month sine-cosine wave	0.029347	0.003285
3-month sine-cosine wave	0.010783	0.002806
2.4-month sine-cosine wave	0.013229	0.002920
2-month sine-cosine wave	0.011198	0.003076
Dummy 1992	Reference category	
Dummy 1993	0.080113	0.023920
Dummy 1994	-0.034185	0.042654
Dummy 1995	-0.001907	0.059744
Dummy 1996	-0.069483	0.076230
Dummy 1997	-0.098969	0.093288
Dummy Winter	-0.026458	0.013360
Dummy Spring	Reference category	
Dummy Summer	0.041605	0.018457
Dummy Autumn	0.137546	0.014306
Dummy Sunday	-0.047589	0.009979
Dummy Monday	Reference category	
Dummy Tuesday	-0.024307	0.009961
Dummy Wednesday	-0.029716	0.009965
Dummy Thursday	-0.013702	0.009899
Dummy Friday	-0.019411	0.009883
Dummy Saturday	-0.037871	0.009959
Dummy Holiday	-0.010495	0.006128
Dummy Epidemics1(Lag 7)	0.332014	0.018779
Dummy Epidemics2	0.278648	0.050219
Dummy Epidemics3	0.327646	0.085424
Dummy Epidemics4	0.477738	0.081583
Dummy Epidemics5	0.233636	0.104857
Mean Temperature (Lag 1)	-0.024185	0.002369
Squared mean Temperature (Lag 1)	0.000881	9.95E-05
Mean rel. Humidity (Lag 1)	0.000767	0.000326
Dummy Hot & Humid	0.060016	0.021922

10.2.10.2 Pollutant Range applied for London

Table 122: Range of Pollutants for which Relative Risk is calculated.

Pollutant	5^{th} percentile	95^{th} percentile	Range (5^{th} - 95^{th} percentile)
PM_{10}	13.8	55.0	41.8
NO_2	38.6	106.9	68.3
SO_2	5.91	60.11	54.2
CO	0.48	2.08	1.6
O_3	5.4	53.59	48.19

10.2.10.3 Estimation Results of the Final Model

Table 123: Estimation Results of the Final Poisson Model for London.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.000572*	1	0.000174	1.02 (1.01; 1.04)
NO_2	0.000487*	1	0.000113	1.03 (1.02; 1.04)
SO_2	0.000543*	3	0.000117	1.03 (1.02; 1.04)
CO	0.018590*	1	0.005734	1.03 (1.01; 1.05)
O_3	0.000692*	1	0.000241	1.03 (1.01; 1.06)

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.

The multiple Poisson regressions conducted using the London time-series data found the strongest and most consistent results. All air pollutants examined showed a positive and statistically significant association with mortality. As for all cities, the relative risk was calculated to indicate the percentage increase of the mortality rate each time the concentration of the respective pollutant increases by the range between the 5th and 95th percentile. These calculations suggest an increase of 2 per cent for PM_{10}, and an increase of 3 per cent for all the other air pollutants, NO_2, SO_2, CO, and ozone. These results are similar to those of previous studies, thus confirming their findings.

10.2.11 Newcastle upon Tyne

10.2.11.1 Parameter Estimates of the Core Model

Table 124: Parameter Estimates of the Core Model for Newcastle upon Tyne.

Variable	Coefficient	Standard Error
C	1.793192	0.104284
Trend	0.000400	0.000161
2-year sine-cosine wave	0.003224	0.11905
1-year sine-cosine wave	0.053197	0.029363
6-month sine-cosine wave	0.038952	0.017457
4-month sine-cosine wave	0.010237	0.011171
3-month sine-cosine wave	0.026733	0.009700
2.4-month sine-cosine wave	0.027436	0.010272
2-month sine-cosine wave	0.013184	0.010786
Dummy 1992	Reference category	
Dummy 1993	-0.124974	0.068129
Dummy 1994	-0.298171	0.121975
Dummy 1995	-0.460568	0.180344
Dummy 1996	-0.625421	0.237598
Dummy 1997	-0.794151	0.296348
Dummy Winter	0.072412	0.046625
Dummy Spring	Reference category	
Dummy Summer	-0.061043	0.062731
Dummy Autumn	-0.008245	0.044060
Dummy Sunday	-0.011067	0.034145
Dummy Monday	Reference category	
Dummy Tuesday	0.021476	0.034219
Dummy Wednesday	-0.048907	0.034455
Dummy Thursday	-0.104319	0.034986
Dummy Friday	-0.047125	0.034464
Dummy Saturday	-0.011067	0.034145
Dummy Epidemics (Lag 2)	0.165746	0.061783
Mean Temperature (Lag 3)	-0.009461	0.003581
Mean rel. Humidity (Lag 3)	-0.000986	0.001061

10.2.11.2 Pollutant Range applied for Newcastle upon Tyne

Table 125: Range of Pollutants for which Relative Risk is calculated.

Pollutant	5th percentile	95th percentile	Range (5th - 95th percentile)
PM_{10}	11.0	49.6	38.6
NO_2	21.78	75.24	53.46
SO_2	5.52	46.92	41.4
CO	0.3	1.2	0.9
O_3	6.21	62.1	55.89

10.2.11.3 Estimation Results of the Final Model

Table 126: Estimation Results of the Final Poisson Model for Newcastle upon Tyne.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001140*	3	0.000562	1.04 (1.00; 1.09)
NO_2	0.001629*	1	0.000556	1.09 (1.04; 1.15)
SO_2	0.000750**	3	0.000425	1.03 (1.00; 1.06)‡
CO	0.006603	2	0.004137	N/A
O_3	0.000924**	2	0.000533	1.05 (1.00; 1.11)‡

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

For Newcastle upon Tyne, the largest impact could be found for nitrogen dioxide, which was associated with a 9 per cent increase in mortality. Particulate matter was also statistically significantly associated, indicating 4 per cent increase in the mortality rate each time the concentration increases by the value of the 5th to 95th percentile range. The two pollutants SO_2 and ozone showed also a positive relationship although only significant at the 10 per cent level. The estimation showed a 3 per cent increase for SO_2 and a 5 per cent increase for ozone. No statistically significant association could be found for carbon monoxide.

10.2.12 Sheffield

10.2.12.1 Parameter Estimates of the Core Model

Table 127: Parameter Estimates of the Core Model for Sheffield.

Variable	Coefficient	Standard Error
C	2.283484	0.053969
Trend	0.000224	0.000123
2-year sine-cosine wave	0.14772	0.008849
1-year sine-cosine wave	0.070591	0.021891
6-month sine-cosine wave	0.070357	0.013042
4-month sine-cosine wave	0.011773	0.008358
3-month sine-cosine wave	0.016146	0.007297
2.4-month sine-cosine wave	0.009182	0.007705
2-month sine-cosine wave	0.029845	0.008090
Dummy 1992	Reference category	
Dummy 1993	0.004538	0.051506
Dummy 1994	-0.176898	0.092580
Dummy 1995	-0.188363	0.136908
Dummy 1996	-0.306367	0.181632
Dummy 1997	-0.421412	0.225453
Dummy Winter	-0.070578	0.035183
Dummy Spring	Reference category	
Dummy Summer	-0.1333843	0.047517
Dummy Autumn	0.048908	0.033361
Dummy Sunday	-0.048756	0.026192
Dummy Monday	Reference category	
Dummy Tuesday	0.000864	0.025841
Dummy Wednesday	-0.019165	0.025987
Dummy Thursday	-0.003196	0.025901
Dummy Friday	-0.004059	0.025908
Dummy Saturday	-0.002959	0.025882
Dummy Epidemics (Lag 6)	0.376045	0.044780
Mean Temperature (Lag 3)	-0.008012	0.002498
Mean rel. Humidity (Lag 3)	0.000651	0.000282

10.2.12.2 Pollutant Range applied for Sheffield

Table 128: Range of Pollutants for which Relative Risk is calculated.

Pollutant	5th percentile	95th percentile	Range (5th - 95th percentile)
PM_{10}	11.0	54.0	43.0
NO_2	26.73	87.12	60.39
SO_2	5.52	57.96	52.44
CO	0.2	1.4	1.2
O_3	8.28	60.03	51.75

10.2.12.3 Estimation Results of the Final Model

Table 129: Estimation Results of the Final Poisson Model for Sheffield.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001086**	3	0.000562	1.05 (1.00; 1.10)‡
NO_2	0.000693*	1	0.000296	1.04 (1.01; 1.08)
SO_2	0.001128*	3	0.000558	1.06 (1.00; 1.12)
CO	0.020887**	4	0.012474	1.03 (1.00; 1.05)‡
O_3	0.001093	2	0.000927	N/A

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

For Sheffield, nitrogen dioxide and sulphur dioxide showed positive and significant effects on the daily number of deaths. The impacts range from 1 per cent to 8 per cent for NO_2 and from 0 per cent to 12 per cent for SO_2. These relationships were significant on the 5 per cent significance level. Two other pollutants, PM_{10} and CO, showed a positive influence, which was significant on the 10 per cent significance level. The relative risks indicate a 5 per cent increase in mortality for PM_{10} and a 3 per cent increase for CO. For ozone, no statistically significant association between the daily mean concentrations and daily mortality could be found.

10.2.13 Southampton

10.2.13.1 Parameter Estimates of the Core Model

Table 130: Parameter Estimates of the Core Model for Southampton.

Variable	Coefficient	Standard Error
C	1.375297	0.120656
Trend	-8.03E-05	0.000209
2-year sine-cosine wave	-0.010438	0.014343
1-year sine-cosine wave	0.105047	0.038373
6-month sine-cosine wave	0.034594	0.021879
4-month sine-cosine wave	0.008261	0.013869
3-month sine-cosine wave	-0.001443	0.011781
2.4-month sine-cosine wave	0.025618	0.012391
2-month sine-cosine wave	0.019815	0.013044
Dummy 1992	Reference category	
Dummy 1993	0.120734	0.086143
Dummy 1994	0.104784	0.157553
Dummy 1995	0.130947	0.232449
Dummy 1996	0.102832	0.307813
Dummy 1997	0.019403	0.384190
Dummy Winter	-0.012805	0.057301
Dummy Spring	Reference category	
Dummy Summer	-0.034285	0.077836
Dummy Autumn	0.160742	0.058534
Dummy Sunday	0.009318	0.042569
Dummy Monday	Reference category	
Dummy Tuesday	0.034239	0.042390
Dummy Wednesday	-0.002268	0.042677
Dummy Thursday	0.008114	0.042500
Dummy Friday	0.050967	0.042145
Dummy Saturday	0.051013	0.042059
Dummy Epidemics (Lag 1)	0.521929	0.072237
Mean Temperature (Lag 2)	-0.014385	0.005714
Squared mean Temperature (Lag 1)	0.000531	0.000254
Mean rel. Humidity (Lag 2)	-0.001093	0.001212

10.2.13.2 Pollutant Range applied for Southampton

Table 131: Range of Pollutants for which Relative Risk is calculated.

Pollutant	5^{th} percentile	95^{th} percentile	Range (5^{th} - 95^{th} percentile)
PM_{10}	11.55	44.0	32.45
NO_2	23.76	77.22	53.46
SO_2	2.76	24.84	22.08
CO	0.3	1.6	1.3
O_3	6.21	62.1	55.89

10.2.13.3 Estimation Results of the Final Model

Table 132: Estimation Results of the Final Poisson Model for Southampton.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.002689*	0-3	0.001341	1.08 (1.00; 1.16)
NO_2	0.002005*	0-3	0.000958	1.11 (1.00; 1.23)
SO_2	0.001178*	1	0.000597	1.03 (1.00; 1.05)
CO	0.017128*	2	0.006498	1.02 (1.01; 1.04)
O_3	0.002351*	0-4	0.001085	1.14 (1.01; 1.28)

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

In Southampton, the estimation results for all air pollutants showed a positive significant association between the respective daily mean concentrations and the daily number of deaths. Using the range between the 5^{th} and 95^{th} percentile for the respective pollutant as a base, the relative risk functions indicated a 14 per cent increase in mortality related with ozone, 11 per cent for NO_2, 8 per cent for PM_{10}, 3 per cent for SO_2 and 2 per cent for carbon monoxide. All estimation results were significant on the 5 pert cent level. For PM_{10}, NO_2 and ozone, the moving average concentrations for the previous 3 days (4 days for ozone) fitted best.

10.2.14 Swansea

10.2.14.1 Parameter Estimates of the Core Model

Table 133: Parameter Estimates of the Core Model for Swansea.

Variable	Coefficient	Standard Error
C	1.208152	0.123071
Trend	0.000422	0.000174
2-year sine-cosine wave	0.036394	0.013535
1-year sine-cosine wave	0.070209	0.038264
6-month sine-cosine wave	0.036619	0.022182
4-month sine-cosine wave	-0.002681	0.014242
3-month sine-cosine wave	0.040970	0.011445
2.4-month sine-cosine wave	-0.007313	0.012205
2-month sine-cosine wave	0.042003	0.012606
Dummy 1992	Reference category	
Dummy 1993	-0.009687	0.074981
Dummy 1994	-0.281164	0.133920
Dummy 1995	-0.327974	0.194269
Dummy 1996	-0.387767	0.258338
Dummy 1997	-0.576945	0.320610
Dummy Winter	-0.006247	0.057414
Dummy Spring	Reference category	
Dummy Summer	-0.181708	0.079470
Dummy Autumn	0.039185	0.054105
Dummy Sunday	0.005689	0.041630
Dummy Monday	Reference category	
Dummy Tuesday	0.009721	0.041162
Dummy Wednesday	0.018592	0.042408
Dummy Thursday	-0.001359	0.041401
Dummy Friday	0.044470	0.039411
Dummy Saturday	0.034659	0.042851
Dummy Holiday	-0.123458	0.070854
Dummy Epidemics1(	0.441749	0.067977
Mean Temperature (Lag 2)	-0.028875	0.009199
Squared mean Temperature (Lag 2)	0.001445	0.000440
Mean rel. Humidity (Lag 3)	0.000861	0.001324

10.2.14.2 Pollutant Range applied for Swansea

Table 134: Range of Pollutants for which Relative Risk is calculated.

Pollutant	5^{th} percentile	95^{th} percentile	Range (5^{th} - 95^{th} percentile)
PM_{10}	11.0	46.0	35.0
NO_2	15.84	69.3	53.46
SO_2	2.76	44.18	41.42
CO	0.2	1.18	0.98
O_3	10.35	74.52	64.17

10.2.14.3 Estimation Results of the Final Model

Table 135: Estimation Results of the Final Poisson Model for Swansea.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.000971*	1	0.000478	1.03 (1.00; 1.08)
NO_2	0.001078*	3	0.000472	1.06 (1.01; 1.11)
SO_2	0.001688**	4	0.000888	1.07 (1.01; 1.14)‡
CO	0.021821*	2	0.010968	1.02 (1.00; 1.04)
O_3	0.001064*	1	0.000400	1.07 (1.02; 1.13)

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

The estimation results for Swansea indicate an increase in mortality by 7 per cent due to ozone, 6 per cent for NO_2, 2 per cent for carbon monoxide, and 3 per cent for PM_{10}. All these results were statistically significant at the 5 per cent level.

sulphur dioxide also showed a positive impact on mortality (7 %), which is, however, only significant at the 10 per cent level.

10.3 Discussion and Summary of the Estimation Results

Having presented and briefly described the estimation results of the multiple Poisson regressions, this section will first provide a summary of these estimation results, and second discuss them. The following table summarises all estimation results for all air pollutants in all 14 British cities, which have been analysed in this study. The calculations that will be made in Chapter 11 of this study will be based on these results. Hence, the following table serves not only as an overview of the manifold estimation results but also as a compact base of reference.

Table 136: Summary of Estimation Results.

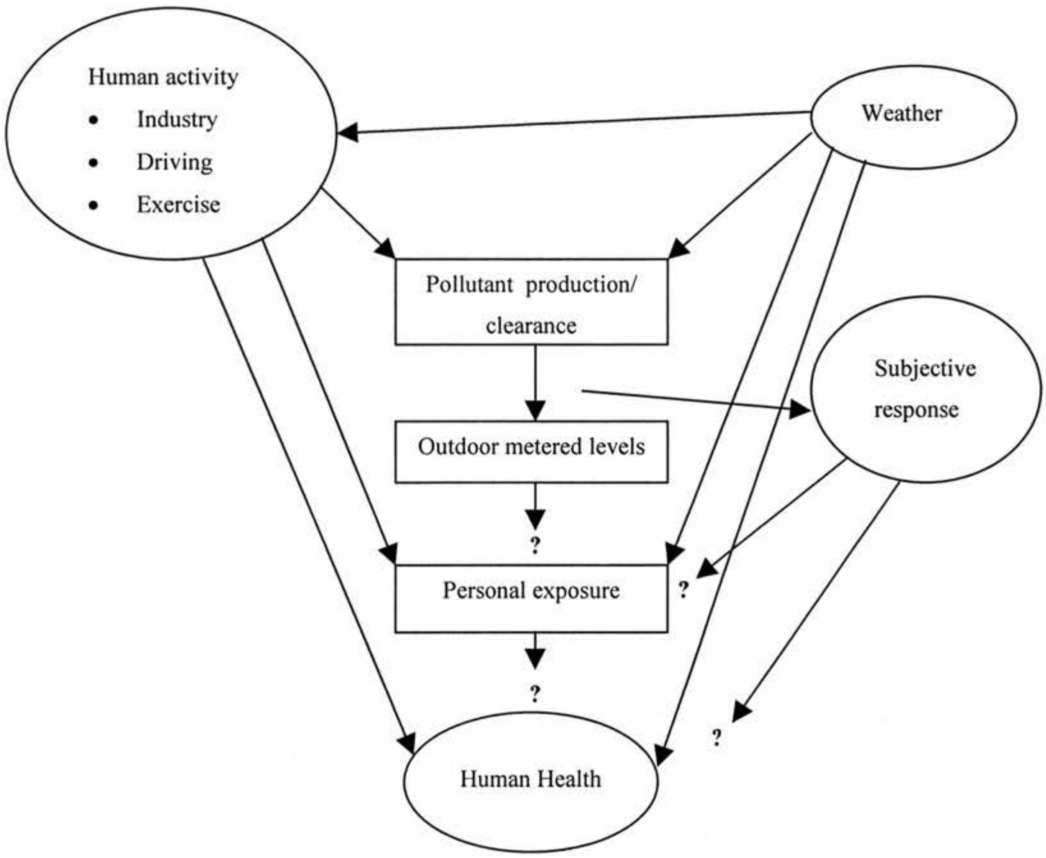
	PM_{10}		NO_2		SO_2		CO		O_3	
	Coefficient (S.E.)	RR	Coefficient (S.E.)	RR	Coefficient (S.E.)	RR	Coefficient (S.E.)	RR	Coefficient (S.E.)	RR
Birmingham	0.000903 (0.000458)	1.04	0.000797 (0.000458)	1.05	0.000542 (0.000283)	1.03	0.016780 (0.009239)	1.01	0.000888 (0.000314)	1.05
Bristol	0.001021 (0.000581)	1.04	0.000879 (0.000227)	1.06	0.0001826 (0.000927)	1.06	0.009621 (0.004730)	1.01	0.000180 (0.000459)	N/A
Cardiff	0.001747 (0.001035)	1.06	0.001612 (0.000817)	1.08	-0.000661 (0.001102)	N/A	0.018871 (0.010743	1.02	0.000836 (0.000772)	N/A
Edinburgh	0.001972 (0.000940)	1.05	0.001088 (0.000605)	1.05	0.001213 (0.000612)	1.05	0.054420 (0.027197)	1.04	-0.001592 (0.000629)	0.93
Glasgow	0.001117 (0.000638)	1.03	0.000363 (0.000139)	1.02	N/A	N/A	0.019680 (0.011719)	1.03	N/A	N/A
Kingston upon Hull	0.001320 (0.000671)	1.05	0.000636 (0.000367)	1.03	0.000420 (0.000986)	N/A	0.009708 (0.004925)	1.01	0.001529 (0.000880)	1.09
Leeds	-0.000650 (0.000553)	N/A	0.000683 (0.000500)	N/A	0.000598 (0.000294)	1.04	0.038416 (0.018244)	1.04	-0.000820 (0.000423)	0.96
Leicester	0.000297 (0.000147)	1.01	0.000690 (0.000844)	N/A	0.001057 (0.000775)	N/A	0.023693 (0.030465)	N/A	0.000188 (0.000104)	1.01
Liverpool	0.001254 (0.000629)	1.05	0.001173 (0.000643)	1.08	0.001181 (0.000452)	1.10	0.0660836 (0.029564)	1.05	-0.000781 (0.000611)	N/A
London	0.000572 (0.000174)	1.02	0.000487 (0.000113)	1.03	0.000543 (0.000117)	1.03	0.018590 (0.005734)	1.03	0.000692 (0.000241)	1.03
Newcastle upon Tyne	0.001140 (0.000532)	1.04	0.001629 (0.000556)	1.09	0.000750 (0.000425)	1.03	0.006603 (0.004137)	N/A	0.000924 (0.000533)	1.05
Sheffield	0.001086 (0.000562)	1.05	0.000693 (0.000296)	1.04	0.001128 (0.000558)	1.06	0.020887 (0.012474)	1.03	0.001093 (0.000927	N/A
Southampton	0.002689 (0.001341)	1.08	0.002005 (0.000958)	1.11	0.001178 (0.000597)	1.03	0.017128 (0.006498)	1.02	0.002351 (0.001085)	1.14
Swansea	0.000971 (0.000478)	1.03	0.001078 (0.000472)	1.06	0.001688 (0.000888)	1.07	0.021821 (0.010968)	1.02	0.001064 (0.000400)	1.07

In recent years many studies have reported the relations between daily variations in the exposure to various air pollutants and short-term variations in mortality rates. The results of this study agree with those findings that also investigated relative low levels of ambient outdoor concentrations of these air pollutants. Concentrating on cases of mortality due to respiratory and cardiovascular disease (ICD-9 390-529), may explain the somewhat stronger effects found in this study compared to others, since air pollution is considered to have the largest impact on these symptoms.

When using routinely collected time-series data, the validity of the data set is an important factor that has to be considered. This study has, in common with the majority of other studies, used a single point measurement of pollutants as an average exposure for a large area of residence for most of the cities examined. As a consequence, the estimated regression results are likely to be underestimates, since personal exposure, the most important exposure estimate, is usually not available on a large scale. Hence, the implicit assumption made in this study is that there is a close correlation between personal exposure and the fixed-point sampling.

However, the question remains as to whether the associations found in the estimation procedure are in fact causal, or the result of residual confounding. As discussed in detail in Chapter 9, the analysis of possible adverse effects of air pollution on human health faces the problem of potential impacts of confounding factors. The following figure illustrates short-term inter-relationships between outdoor metered air pollution and human health, indicating that some of these inter-relationships are still not entirely understood.

Figure 52: Short-term Relations between Outdoor Metered Air Pollution and Human Health.



Source: Poloniecki et al. (1997), p. 539.

The above figure shows that factors that affect creation and clearance of air pollution, including meterological factors and driving motor vehicles, can have a direct effect on both human behaviour and acute health outcomes (e.g. death). Factors such as wind[1] and rain may not only affect air pollution levels but also "the amount of physical exercise taken and the duration of time spent outdoors, where levels of air pollution are generally different from indoors."[2] Further factors such as aeroallergens (e.g. pollens) may also be associated with impaired health.[3]

---

[1] However, in the process of establishing the estimation models, wind speed data was originally also considered, but finally dropped since no improvement of the overall model fit could be achieved, with the respective estimated coefficients being insignificant.

[2] Poloniecki et al. (1997), p. 539.

[3] See, for instance, Anderson et al. (1998).

The problem of confounding was recognised and carefully included in the procedure of the model building employed in this study. The estimation models used in this study controlled for potential causes of confounding such as long term trends, seasonality, and meteorological factors. Furthermore, systematic components such as calendar specific effects including day of the week and holidays, as well as unsystematic components, that is influenza epidemics have been considered, since they may also bias the outcome of time-series regressions. Assuming that the mortality data is Poisson distributed, calls for a different approach to de-seasonalising the data.

Basically, since no pre-filtering is possible, some other form of filtering is necessary. For this study it had been decided to use a parametric approach, where sinusoidal terms were included to model cyclical seasonal patterns. The length of these cycles range from two months to two years, leaving other short term fluctuations unchanged. In this, the filter is actually included into the regression model 'competing' with the other variables included in the model. Additionally, indicators for season were also included.

Further potential for confounding to occur is in the treatment of the meteorological factors, including temperature and humidity. This study, in common with the majority of other studies, used 24-hour mean temperature, rather than 24-hour maximum temperature. Also, the pollution data included, was restricted to the range between the 5^{th} and 95^{th} percentile, in order to achieve more robust estimates. Furthermore, mean relative humidity was included. For both variables, same day, as well as previous days were examined when constructing the core model.

This study has a large number of similarities with other studies in respect of data collection of pollutants, confounders, and health outcomes. Further, since the analyses, and particularly the procedure to control for confounding effects, have been heavily based on the procedure suggested by the APHEA project, this allows comparison of the estimation results here with a number of other studies. Generally, the regression results found in this study are consistent and, basically, of the same order compared with the findings of those of previously published reports, especially from other European urban areas including the few British locations studied. This is particularly true when comparing the results of those studies that analysed cause-specific mortality, i.e. respiratory and/or cardiovascular disease. Also, the consistency in the regression results across all 14 cities analysed, covering different air quality and meteorological conditions, gives confidence in the estimation procedure chosen, excluding the case that the effects were found purely by chance.

In summary, the results of this study support previous findings of a significant association between a range of air pollutants and health effects, i.e. cardio-respiratory deaths. Considering current ambient concentration of the air pollutants studied, these effects are relatively small accounting for relatively small proportions of total annual deaths in Great Britain. Nevertheless, although these effects seem to be relatively small, they result in significant economic costs; this will be illustrated in the following chapter.

11. Quantification of Air Pollution related Health Effects

11.1 Introduction

The previous chapters first discussed and established a sound methodology to estimate the association between air pollution concentrations and acute health effects, i. e. premature death due to respiratory and cardiovascular disease, and second, presented the estimation results of the applied multiple Poisson regressions. These so-called exposure-response functions now serve as the basis for further calculations in the attempt to quantify external health costs associated with air pollution, and in particular relate them to road transport. This calculation is a two step procedure and is the main focus of this chapter.

First, based on the exposure-response coefficients estimated in the previous chapter, the absolute number of deaths due to air pollution per city examined is calculated. These figures are presented for each air pollutant examined together with its percentage share of total annual mortality in the respective city. These figures serve as a basis for further calculations, where they are adjusted in a way that the motor traffic's contribution of the total air pollution concentrations is taken into account. The resulting figures present the numbers of premature deaths attributable to traffic-induced air pollution.

Second, based on these calculations, the willingness to pay estimates for the value of a statistical life, as they have been derived in Chapter 6, are applied, resulting in a final monetary value. As described in detail in Chapters 4 and 5, the negative effects arising from road traffic generated air pollution are not incorporated in the market prices paid by the road users. These effects, such as premature deaths as analysed in this study, represent external costs that are imposed on society, or more specifically external costs imposed on the economy of Great Britain.

The procedure to estimate these external costs implies the monetary valuation of the number of deaths calculated in this chapter, i.e. putting a money value on the absolute number of premature deaths brought forward in association with air pollution. This monetary value is the value of a statistical life based on the willingness to pay to reduce the risk of dying prematurely, as discussed in detail in Chapter 6. However, having adopted a fairly conservative approach throughout the entire study, a range of values including a low, central, and high estimate, as also presented in Chapter 6, will be used rather than one single estimate. The resulting best monetary estimates associated with the estimated total annual mortality cases due to air pollution, and road transport in particular, for each of the 14 British cities included in this study are given.

However, no attempt has been made to simply add up the effects of the individual pollutants for an overall effect of air pollution. These results could be misleading, since there may be additive or synergistic effects of air pollution.[1] This is due to the fact that the population under consideration in fact inhales a mix of pollutants, rather than one single pollutant. However, the individual effects per pollutant for each city examined are summarised in order to give an overall figure. This is undertaken for both the effects expressed in the total annual number of premature deaths related to the exposure of air pollution (differentiated according to total air pollution and motor traffic induced air pollution), as well as the monetary valuation of these effects. The findings of this chapter are finally summarised in the concluding section.

---

[1] The Committee on the Medical Effects of Air Pollution, however, states that these effects are likely to be small. See Department of Health (1998), p. 59.

11.2. Quantification of Overall Health Effects

Using the exposure-response coefficients presented in the previous chapter, the absolute number of premature deaths expected as a result of a change in the concentration of the respective air pollutant in a given city can be calculated. The following model taken from Ostro (1994), and Ostro and Chestnut (1998) has been adopted in this analysis.[2] This procedure has also been applied in previous similar studies.[3] Based on the daily times-series findings, the total annual number of expected cases of premature deaths is calculated as follows:

$$dH_{ij} = b_{ij} \times POP_j \times CMR \times dC_j \times 1/100 \qquad (42)$$

where:

- d = change in.
- H = health outcome.
- i = health effect i (i.e. premature death).
- j = particular pollutant j.
- b_j = slope of exposure-response function for pollutant j.
- POP_j = population exposed to pollutant j.
- CMR = Crude mortality rate.
- C = ambient concentration of pollutant j.
- 1/100 = converts from percentages to absolute numbers.

The equation above represents the change in annual premature mortality associated with a certain air pollutant. For each city included in this study, separate calculations are made. Instead of applying a single average crude mortality rate for all locations, the product of the population at risk and the crude

---

[2] See: Ostro (1994), pp. 5-6 and Ostro and Chestnut (1998), pp. 100-101.
[3] See, for instance, Maddison et al. (1996) and Pearce and Crowards (1996).

mortality rate is substituted by the total number of annual non-accidental deaths for each city.[4] This may be a more accurate approach, especially when analysing individual cities rather than the urban areas of Great Britain as a whole. The numbers of annual non-accidental deaths for each of the cities examined are summarised in the following table:

Table 137: Total Number of Non-Accidental Deaths in British Cities in 1997.

City	# of non-accidental deaths in 1997
Birmingham	10,756
Bristol	4,154
Cardiff	3,084
Edinburgh	5,238
Glasgow	9,552
Kingston upon Hull	2,870
Leeds	7,534
Leicester	3,056
Liverpool	5,806
London	65,962
Newcastle upon Tyne	3,384
Sheffield	6,275
Southampton	2,189
Swansea	2,437

The following table recapitulates the summary of the annual mean concentrations of the air pollutants measured in the respective British cities, as illustrated in Chapter 8.

4 The Department of Health (1998), for instance, applied a crude mortality rate for Great Britain of 1,106.4 per 100,000 population.

Table 138: Summary of Annual Means of Air Pollution.

City	PM_{10}	NO_2	SO_2	CO	O_3
Birmingham	24.20	47.9	21.9	0.62	34.09
Bristol	24.17	46.96	15.36	0.65	34.23
Cardiff	25.49	42.65	14.03	0.63	32.45
Edinburgh	20.05	49.62	17.64	0.57	29.52
Glasgow	14.64	51.06	7.19	0.83	18.99
Kingston upon Hull	24.24	43.28	21.09	0.58	32.48
Leeds	25.84	52.82	21.66	0.71	28.01
Leicester	21.48	43.38	17.98	0.54	33.13
Liverpool	25.41	48.54	27.96	0.46	33.60
London	27.79	66.72	23.74	1.08	25.79
Newcastle upon Tyne	24.48	46.38	17.57	0.63	32.52
Sheffield	24.79	55.91	23.51	0.58	30.80
Southampton	22.55	45.2	10.71	0.73	31.66
Swansea	24.10	41.4	16.43	0.49	38.79

Applying the exposure-response coefficients, the following tables present the total annual number of expected cases of premature mortality using the equation presented above. For each air pollutant, the absolute number of annual deaths that can be associated to air pollution using the central (point) estimated coefficient given in the previous chapter.

An alternative way to present the estimation results is to calculate the numbers of reduction in deaths that can be brought forward per μg/m^3 reduction of a specific pollutant. The results of these calculations are presented in Appendix 6 of this study. For cases where no significant or a negative regression coefficients were estimated, no figures could be calculated.

11.2.1 Particulate Matter

Table 139: Best Estimates for Premature Mortality from PM_{10} per Year.

City	Annual # of premature deaths	Percentage of total annual # of deaths
Birmingham	235	2.18
Bristol	103	2.48
Cardiff	137	4.44
Edinburgh	207	3.95
Glasgow	156	1.63
Kingston upon Hull	92	3.21
Leeds	N/A	N/A
Leicester	19	0.62
Liverpool	185	3.19
London	1,049	1.59
Newcastle upon Tyne	94	2.78
Sheffield	169	2.69
Southampton	133	6.08
Swansea	57	2.34
Total 14 Cities	2,636	1.99

The estimated figures show that the total annual number of deaths brought forward possibly associated with the exposure to PM_{10}, is roughly 2 per cent of the annual number of non-accidental deaths. The largest percentage (6.08%) can be found for Southampton, whereas in Leicester just over half a per cent (0.62%) of the deaths can be related to particular matter.

11.2.2 Nitrogen Dioxide

Table 140: Best Estimates for Premature Mortality from NO_2 per Year.

City	Annual # of premature deaths	Percentage of total annual # of deaths
Birmingham	411	3.82
Bristol	171	4.12
Cardiff	212	6.87
Edinburgh	283	5.40
Glasgow	177	1.85
Kingston upon Hull	79	2.75
Leeds	N/A	N/A
Leicester	N/A	N/A
Liverpool	331	5.70
London	2,143	3.25
Newcastle upon Tyne	256	7.57
Sheffield	243	3.87
Southampton	198	9.05
Swansea	109	4.47
Total 14 Cities	4,613	3.49

The figures for nitrogen dioxide indicate an annual increase in mortality due to the exposure of this pollutant by around 3.5 per cent. The highest share is given for Southampton (9.05%), while the figures indicate that less than 2 per cent of total annual death may be associated with NO_2 in Glasgow. Compared with the results for PM_{10} presented in the previous sub-section, the number of premature deaths brought forward related with NO_2 is significantly larger; the figures for London and Newcastle upon Tyne are more than double.

11.2.3 Sulphur Dioxide

Table 141: Best Estimates for Premature Mortality from SO_2 per Year.

City	Annual # of premature deaths	Percentage of total annual # of deaths
Birmingham	128	1.19
Bristol	117	2.82
Cardiff	N/A	N/A
Edinburgh	112	2.14
Glasgow	N/A	N/A
Kingston upon Hull	N/A	N/A
Leeds	98	1.30
Leicester	N/A	N/A
Liverpool	192	3.31
London	850	1.29
Newcastle upon Tyne	45	1.33
Sheffield	166	2.65
Southampton	28	1.28
Swansea	68	2.79
Total 14 Cities	1,804	1.36

For sulphur dioxide, somewhat similar results to PM_{10} are observed. The annual average of all cities examined is with 1.36 per cent around half a per cent smaller than for PM_{10}. Liverpool indicates the largest share with 3.31 per cent; in Birmingham, sulphur dioxide contributes just over 1 per cent to total annual non-accidental deaths. As noted above, the similarity to PM_{10} may indicate that similar effects have been captured. It is assumed that PM_{10} measurements are likely to incorporate compounds such as sulphates and ammonium salts.[5] Hence,

[5] See: QUARG (1993).

this confirms that simply adding up the figures for the pollutants would lead to overestimates for the actual effects.

11.2.4 Carbon Monoxide

Table 142: Best Estimates for Premature Mortality from CO per Year.

City	Annual # of premature deaths	Percentage of total annual # of deaths
Birmingham	112	1.04
Bristol	26	0.63
Cardiff	37	1.20
Edinburgh	162	3.09
Glasgow	156	1.63
Kingston upon Hull	16	0.56
Leeds	205	2.72
Leicester	N/A	N/A
Liverpool	176	3.03
London	1,324	2.01
Newcastle upon Tyne	N/A	N/A
Sheffield	76	1.21
Southampton	27	1.23
Swansea	26	1.07
Total 14 Cities	2,343	1.77

The estimates for carbon monoxide show that less than two per cent (1.77) of total annual non-accidental deaths may be attributable to the exposure of CO. The contribution of individual cities ranges from 0.56 per cent in Hull to just over 3 per cent in Edinburgh and Liverpool.

11.2.5 Ozone

Table 143: Best Estimates for Premature Mortality from O_3 per Year.

City	**Annual # of premature deaths**	**Percentage of total annual # of deaths**
Birmingham	326	3.03
Bristol	N/A	N/A
Cardiff	N/A	N/A
Edinburgh	N/A	N/A
Glasgow	N/A	N/A
Kingston upon Hull	143	4.98
Leeds	N/A	N/A
Leicester	19	0.62
Liverpool	N/A	N/A
London	1,177	1.78
Newcastle upon Tyne	102	3.01
Sheffield	N/A	N/A
Southampton	163	7.45
Swansea	101	4.14
Total 14 Cities	2,031	1.54

The rather inconsistent estimation results for ozone, make it more difficult to state an 'overall' estimate. Half of the examined cities do not show a significant adverse impact on human health, indicated by premature death. For those that do, Southampton shows the largest impact, accounting for and 7.45 per cent of the total annual number of deaths. In Leicester, a much smaller share of total deaths could be associated with ambient levels of ambient ozone concentrations (0.62%).

11.3 Quantification of Road Traffic-Induced Health Effects

As mentioned in Chapter 3, when describing the main traffic-related air pollutants, the source of pollution may widely vary depending upon the location. Whereas power stations may be the dominant source of a specific air pollutant in a rural area, concentrations of the same pollutant in urban inner city areas may be almost entirely generated by road traffic exhaust emissions. In other words, since this study is concerned with the impact of traffic-related air pollution on human health in various cities, it is important to account for the fact that road traffic is by far the largest contributor to urban concentrations for all pollutants included in this study.

Although data for the specific contribution of road transport is provided for London, as presented in Chapter 3, no such information is currently available for the other British cities included in this study. However, a study by Eyre et al. (1996), provides estimates for urban-rural dose ratios, which can be used to make these adjustments. These ratios, which will be applied to the national figures broken down to main sources, do account for the higher doses arising from urban sources than rural sources. The ratios presented by Eyre et al. (1996) are reproduced in the following table.

Table 144: Estimated Urban-Rural Dose Ratios.

Air Pollutant	Urban-rural dose ratio
PM_{10}	6.4
NO_2	4.6
SO_2	19
CO	1.8

Source: Eyre et. al. (1996), pp. 13 & 28-30.

Before applying these conversion factors, the contribution of the main sources to national emissions of the respective air pollutants is summarised in the following table. These figures are based on the detailed discussion of traffic-related air pollutants in Chapter 2 of this study.

Table 145: Percentage Contribution of Road Traffic to Total National Emissions.

Pollutant	**Road Transport**	**Domestic Sources**	**Power Stations**	**Others Sources**
PM_{10}	26	15	12	47
NO_x	48	N/A	20	32
SO_2	2	4	62	32
CO	75	5	1	19‡

Note: ‡: This includes industrial combustion (12%) and other sources which predominantly comprises other transport.

Combining the information from both tables can then be used to produce the following partial responsibility of the various sources, specifically in urban areas. The procedure follows the approach by Pearce and Crowards (1996), and Maddison et al. (1996) and will be applied to the pollutants stated above. Applying a weight of 1.0 indicates that all emissions from the respective source are generated in rural areas, whereas higher weights reflect a mixed contribution of rural and urban areas to total national emissions.

Before proceeding, a special note has to be made for the secondary pollutant ozone. As described in Chapter 3 of this study, the oxidant ozone is formed through a chemical process involving the pollutant groups of oxides of nitrogen and sunlight in the presence of volatile organic compounds. Further, both ozone as well as its precursors can be transported over long distances, which makes it extremely difficult to calculate an urban-rural ratio. Also, it appears rather difficult to attribute ozone concentrations to specific sources of emissions.

As previously described, concentration levels of ozone are generally lower in urban areas due to scavenging by nitric oxide from vehicle exhausts. In other words, road traffic is not only a main source of the precursors' emissions to create ozone, but it also contributes to the 'reduction' of ozone concentrations. Thus, exposure to ozone levels in highly trafficked inner city areas are generally lower than in rural areas, which may also be reflected in the rather inconsistent estimation results shown in the previous chapter. As a consequence, no further calculations for the secondary pollutant ozone were performed.

The following parts illustrate the procedure used in order to derive the percentage share of pollution emissions, road traffic is responsible for in urban areas. The calculations were made for the pollutants particular matter, nitrogen dioxide, sulphur dioxide and carbon monoxide.

- Particulate Matter

Road vehicles	= 26% of emissions * 6.4	= 1.664
Domestic sources	= 15% of emissions * 1.0	= 0.150
Power stations	= 12% of emissions * 1.0	= 0.120
Other emissions	= 47% of emissions * 1.0	= 0.470
Sum of weighted percentages:		= 2.404

From this, the revised weighted contribution of road transport in urban areas can be calculated as follows: 1.664 / 2.404 = 69.2 %. In other words, road vehicles generate nearly seventy percent of PM_{10} concentrations in urban areas.

- Nitrogen Dioxide

Road vehicles	= 48% of emissions * 4.6	= 2.208
Power stations	= 20% of emissions * 1.0	= 0.200
Other emissions	= 32% of emissions * 2.3	= 0.736
Sum of weighted percentages:		= 3.144

From this, the revised weighted contribution of road transport in urban areas can be calculated as follows: 2.208 / 3.144 = 70.1 %. In other words, road vehicles generate just over seventy percent of NO_2 concentrations in urban areas.

- Sulphur Dioxide

Road vehicles	= 2% of emissions * 19.0	= 0.380
Domestic sources	= 4% of emissions * 8.0	= 0.320
Power stations	= 62% of emissions * 1.0	= 0.620
Other emissions	= 32% of emissions * 1.0	= 0.320
Sum of weighted percentages:		= 1.640

From this, the revised weighted contribution of road transport in urban areas can be calculated as follows: 0.380 / 1.640 = 23.2%. In other words, road vehicles generate just over twenty-three percent of SO_2 concentrations in urban areas.

- Carbon Monoxide

Road vehicles	= 75% of emissions * 1.8	= 1.35
Domestic sources	= 5% of emissions * 1.0	= 0.05
Industrial combustion	= 12% of emissions * 1.0	= 0.12
Power stations	= 1% of emissions * 1.0	= 0.01
Other emissions	= 7% of emissions * 1.0	= 0.07
Sum of weighted percentages:		= 1.60

From this, the revised weighted contribution of road transport in urban areas can be calculated as follows: 1.35 / 1.60 = 84.4%. In other words, road vehicles generate nearly 85 per cent CO concentrations in urban areas.

The following table provides a summary of the road traffic's share of pollution emissions in urban areas. These figures are used to desscribe the responsibility of road traffic in general to urban ambient concentrations of pollution. However, when using the presented percentages, one has to acknowledge that they are best estimates rather than a precise replication of the 'true' urban-rural ratios.

Nevertheless, comparing the presented percentages with the actual figures for the share of the road transport sector of total concentrations of pollution in London support the assumption that these estimates are correct.[6]

Table 146: Percentage of Pollution Associated with Road Traffic in Urban Areas.

Pollutant	**Proportion of road traffic**
PM_{10}	69.2
NO_2	70.1
SO_2	23.2
CO	84.4

Applying these percentages to the figures of total numbers of additional premature deaths due to air pollution as computed and presented in the previous section, results in the number of such deaths attributable to road traffic in the examined 14 British cities.

[6] It had been suggests that the contribution of road traffic in London may be even larger than applied in this study.

11.3.1 Particulate Matter

Table 147: Best Estimates for Premature Mortality from Road Traffic Related PM_{10}.

City	Annual # of premature deaths	Percentage of total annual # of deaths	# of deaths per 100,000 inhabitants
Birmingham	163	1.51	16
Bristol	71	1.72	18
Cardiff	95	3.07	30
Edinburgh	143	2.73	32
Glasgow	108	1.13	18
Kingston upon Hull	64	2.22	24
Leeds	N/A	N/A	N/A
Leicester	13	0.43	4
Liverpool	128	2.20	27
London	726	1.10	10
Newcastle upon Tyne	65	1.92	23
Sheffield	117	1.86	22
Southampton	92	4.20	43
Swansea	39	1.62	17
Total 14 Cities	1,824	1.38	

Across the cities examined in this study, road traffic related PM_{10} is responsible for total annual non-accidental deaths ranging from 0.43 per cent in Leicester to 4.20 per cent in Southampton, while the average for all 14 cities is 1.38 per cent. The number of deaths brought forward per 100,000 inhabitants is greatest in Southampton with 43, and Edinburgh and Cardiff, with 32 and 30 respectively. In Leicester, on the other hand, 4 deaths per 100,000 may be attributable to the exposure of particular matter generated by road traffic.

11.3.2 Nitrogen Dioxide

Table 148: Best Estimates for Premature Mortality from Road Traffic Related NO_2.

City	Annual # of premature deaths	Percentage of total annual # of deaths	# of deaths per 100,000 inhabitants
Birmingham	288	2.68	28
Bristol	120	2.89	30
Cardiff	149	4.82	47
Edinburgh	198	3.79	44
Glasgow	124	1.30	20
Kingston upon Hull	55	1.93	21
Leeds	N/A	N/A	N/A
Leicester	N/A	N/A	N/A
Liverpool	232	4.00	50
London	1,502	2.28	21
Newcastle upon Tyne	179	5.30	64
Sheffield	170	2.71	32
Southampton	139	6.34	65
Swansea	76	3.14	33
Total 14 Cities	3,234	2.44	

The analysis for nitrogen dioxide generated by road traffic shows the largest adverse effect both in relative and absolute terms. In Southampton and Newcastle upon Tyne, the number of deaths brought forward is greatest with 6.34 per cent and 5.30 percent respectively of total annual non-accidental deaths, which represents 65 and 64 deaths per 100,000 inhabitants respectively. In contrast, the impact in Glasgow, Kingston upon Hull, and London is around one third of the size with around 20 deaths brought forward per 100,000 inhabitants.

The total number of road traffic-related deaths brought forward for all 14 cities examined in this study is 3,234, which represents 2.44 per cent of the total annual number of deaths in these cities.

11.3.3 Sulphur Dioxide

Table 149: Best Estimates for Premature Mortality from Road Traffic Related SO_2.

City	Annual # of premature deaths	Percentage of total annual # of deaths	# of deaths per 100,000 inhabitants
Birmingham	30	0.28	3
Bristol	27	0.65	7
Cardiff	N/A	N/A	N/A
Edinburgh	26	0.50	6
Glasgow	N/A	N/A	N/A
Kingston upon Hull	N/A	N/A	N/A
Leeds	23	0.30	3
Leicester	N/A	N/A	N/A
Liverpool	45	0.77	10
London	197	0.30	3
Newcastle upon Tyne	10	0.31	4
Sheffield	39	0.61	7
Southampton	6	0.30	3
Swansea	16	0.65	7
Total 14 Cities	419	0.32	

Concentrations of sulphur dioxide generated by motor vehicles have the smallest adverse impact found in the analysis. This may be mainly due to the fact that road traffic accounts for just over 20 per cent of all SO_2 emissions, whereas motor vehicles contribute more than two thirds of total emissions for each of the other air pollutants examined in this study. Nevertheless, road traffic SO_2 emissions account for 0.77 per cent of the total annual non-accidental deaths in Liverpool, which is equivalent to 10 deaths per 100,000 inhabitants. The impact in each of the other cities analysed ranges from 0.28 per cent in Birmingham (3 deaths per 100,000) to 0.65 per cent in Bristol and Swansea (7 deaths per 100,000). The overall total number of premature deaths associated to road traffic for all 14 cities is 419, which is equivalent to 0.32 per cent of the total annual number of deaths (excluding accidental deaths).

11.3.4 Carbon Monoxide

Table 150: Best Estimates for Premature Mortality from Road Traffic Related CO.

City	Annual # of premature deaths	Percentage of total annual # of deaths	# of deaths per 100,000 inhabitants
Birmingham	95	0.88	9
Bristol	22	0.53	5
Cardiff	31	1.01	10
Edinburgh	137	2.61	30
Glasgow	132	1.38	21
Kingston upon Hull	14	0.47	5
Leeds	173	2.30	24
Leicester	N/A	N/A	N/A
Liverpool	149	2.56	32
London	1,117	1.69	16
Newcastle upon Tyne	N/A	N/A	N/A
Sheffield	64	1.02	12
Southampton	23	1.04	11
Swansea	22	0.90	10
Total 14 Cities	1,977	1.49	

The calculations for carbon monoxide indicate that road traffic may be responsible for 32 deaths per 100,000 in Liverpool and 30 deaths in Edinburgh. The lowest figures based on the regression results presented in the previous chapter are those for Bristol and Kingston upon Hull with 5 deaths brought forward per 100,000 inhabitants that can be associated with road traffic generated CO emissions. The estimated total number of premature deaths brought forward due to the exposure to traffic-related carbon monoxide is 1.977 per year.

11.3.5 Summary of Estimated Effects

The previous sub-section presented the number of deaths brought forward, based on the regressions results illustrated in the previous chapter. Both overall figures as well as those that may be related to pollution emission from road traffic have been calculated. As noted above, these figures are not added up across the different pollutants in order to avoid double counting due to additive or synergistic effects. However, total figures for the different cities studied were produced.

The following table presents the overall numbers of deaths brought forward.

Table 151: Summary of Total Deaths brought forward.

Pollutant	Total	Road Traffic Related
PM_{10}	2,636	1,824
NO_2	4,613	3,234
SO_2	1,804	419
CO	2,343	1,977

The presented values are based on the results of the analysis of 14 British cities, covering a population of 12.889 million. Assuming that about 80 per cent of the population in Great Britain live in urban areas, the figures may be extrapolated in order to obtain overall values for urban areas of Great Britain.[7] Obviously, these

[7] In 1997, the population of Great Britain was 57.335m. In fact, almost 90 per cent of people in Great Britain live in urban areas, while just over half of the total population is resident in the urban areas with populations of 100 thousand or more. See: ONS (1999b), pp. 29-30. Hence, assuming a percentage of 80 per cent excludes that part of the population that lives in very small urban areas with rather 'rural' characteristics.

values are only a crude estimate, since this assumes similar conditions and exposure-response functions across all urban areas.

However, since the results of up to 14 different data sets for a particular air pollutant have been analysed, the problem of transferability may be less severe, and the estimate produced for the general urban Great Britain population may in fact be fairly reliable. The following table presents such an estimate for the number of deaths brought forward in urban areas of Great Britain.

Table 152: Extrapolated Annual Number of Deaths in Urban Great Britain.

Pollutant	Urban Great Britain	
	Total	Road Traffic Related
PM_{10}	9,358	6,475
NO_2	16,376	11,481
SO_2	6,404	1,487
CO	8,318	7,018

11.4. Calculation of External Costs

The previous two sections presented the figures of the absolute numbers of deaths brought forward that can be associated to the various air pollutants included in this study. Both the total annual numbers of deaths as well as those attributable to road transport emissions have been calculated. These results are now used for further calculations to quantify these effects in monetary terms.

The resulting values are the external costs that arise from premature deaths caused by the exposure to air pollution. In other words, as described in detail in Chapter 5, these costs are imposed on the society as a whole regardless of who is generating them. In the case examined in this study, the owner and user of motor vehicles are not made responsible for the adverse effects caused by the generated emissions, such as the related premature deaths, i.e. the polluter-pays-principle is not fulfilled.

The following sections present the external costs that can be associated with the exposure to air pollution. The valuation process is based on the discussion in Chapter 6, where a range of values for a statistical life was derived. These values have been adjusted to the specific case of air pollution, taking such factors as the quality of life, age, and impaired health status into account.

Following the conservative approach applied in this study, a range of estimates was presented rather than one single value. The values including a low, central and upper estimate are as follows:

Table 153: Summary of Air Pollution Related VOSL.

Lower bound	Central value	Upper bound
£3,000	£126,000	£1.645m

The figures for the deaths brought forward are multiplied with these monetary values for a statistical life and presented in the following sub-sections. Differentiated by air pollutants, for each city, low, central, and high estimates are provided. A further distinction is made between the overall number of premature deaths and those that can be related to road traffic.

11.4.1 Particular Matter

Table 154: External Costs Associated with Premature Deaths due to PM_{10} (million £).

City	Low estimate		Central estimate		High estimate	
	Total	*Road*	*Total*	*Road*	*Total*	*Road*
Birmingham	0.705	0.489	29.610	20.538	386.575	268.135
Bristol	0.309	0.213	12.978	8.946	169.435	116.795
Cardiff	0.411	0.285	17.262	11.97	225.365	156.275
Edinburgh	0.621	0.429	26.082	18.018	340.515	235.235
Glasgow	0.468	0.324	19.656	13.608	256.62	177.66
Kingston upon Hull	0.276	0.192	11.592	8.064	151.34	105.28
Leeds	N/A	N/A	N/A	N/A	N/A	N/A
Leicester	0.057	0.039	2.394	1.638	31.255	21.385
Liverpool	0.555	0.384	23.31	16.128	304.325	210.56
London	3.147	2.178	132.174	91.476	1,725.605	1,194.27
Newcastle upon Tyne	0.282	0.195	11.844	8.19	154.63	106.925
Sheffield	0.507	0.351	21.294	14.742	278.005	192.465
Southampton	0.399	0.276	16.758	11.592	218.785	151.34
Swansea	0.171	0.117	7.182	4.914	93.765	64.155
Total 14 Cities	7.908	5.472	332.136	229.824	4,336.22	3,000.48

11.4.2 Nitrogen Dioxide

Table 155: External Costs Associated with Premature Deaths due to NO_2. (million £).

City	Low estimate		Central estimate		High estimate	
	Total	*Road*	*Total*	*Road*	*Total*	*Road*
Birmingham	1.233	0.864	51.786	36.288	676.095	473.76
Bristol	0.513	0.36	21.546	15.12	281.295	197.4
Cardiff	0.636	0.447	26.712	18.774	348.74	245.105
Edinburgh	0.849	0.594	35.658	24.948	465.535	325.71
Glasgow	0.531	0.372	22.302	15.624	291.165	203.98
Kingston upon Hull	0.237	0.165	9.954	6.93	129.955	90.475
Leeds	N/A	N/A	N/A	N/A	N/A	N/A
Leicester	N/A	N/A	N/A	N/A	N/A	N/A
Liverpool	0.993	0.696	41.706	29.232	544.495	381.64
London	6.429	4.506	270.018	189.252	3,525.235	2,470.79
Newcastle upon Tyne	0.768	0.537	32.256	22.554	421.12	294.455
Sheffield	0.729	0.51	30.618	21.42	399.735	279.65
Southampton	0.594	0.417	24.948	17.514	325.71	228.655
Swansea	0.327	0.228	13.734	9.576	179.305	125.02
Total 14 Cities	13.839	9.702	581.238	407.484	7,588.385	5,319.93

11.4.3 Sulphur Dioxide

Table 156: External Costs Associated with Premature Deaths due to SO_2 (million £).

City	Low estimate		Central estimate		High estimate	
	Total	*Road*	*Total*	*Road*	*Total*	*Road*
Birmingham	0.384	0.09	16.128	3.78	210.56	49.35
Bristol	0.351	0.081	14.742	3.402	192.465	44.415
Cardiff	N/A	N/A	N/A	N/A	N/A	N/A
Edinburgh	0.336	0.078	14.112	3.276	184.24	42.77
Glasgow	N/A	N/A	N/A	N/A	N/A	N/A
Kingston upon Hull	N/A	N/A	N/A	N/A	N/A	N/A
Leeds	0.294	0.069	12.348	2.898	161.21	37.835
Leicester	N/A	N/A	N/A	N/A	N/A	N/A
Liverpool	0.576	0.135	24.192	5.67	315.84	74.025
London	2.55	0.591	107.1	24.822	1,398.25	324.065
Newcastle upon Tyne	0.135	0.03	5.67	1.26	74.025	16.45
Sheffield	0.498	0.117	20.916	4.914	273.07	64.155
Southampton	0.084	0.018	3.528	0.756	46.06	9.87
Swansea	0.204	0.048	8.568	2.016	111.86	26.32
Total 14 Cities	5.412	1.257	227.304	52.794	2,967.58	689.255

11.4.4 Carbon Monoxide

Table 157: External Costs Associated with Premature Deaths due to CO (million £).

City	Low estimate		Central estimate		High estimate	
	Total	*Road*	*Total*	*Road*	*Total*	*Road*
Birmingham	0.336	0.285	14.112	11.97	184.24	156.275
Bristol	0.078	0.066	3.276	2.772	42.77	36.19
Cardiff	0.111	0.093	4.662	3.906	60.865	50.995
Edinburgh	0.486	0.411	20.412	17.262	266.49	225.365
Glasgow	0.468	0.396	19.656	16.632	256.62	217.14
Kingston upon Hull	0.048	0.042	2.016	1.764	26.32	23.03
Leeds	0.615	0.519	25.83	21.798	337.225	284.585
Leicester	N/A	N/A	N/A	N/A	N/A	N/A
Liverpool	0.528	0.447	22.176	18.774	289.52	245.105
London	3.972	3.351	166.824	140.742	2,177.98	1,837.465
Newcastle upon Tyne	N/A	N/A	N/A	N/A	N/A	N/A
Sheffield	0.228	0.192	9.576	8.064	125.02	105.28
Southampton	0.081	0.069	3.402	2.898	44.415	37.835
Swansea	0.078	0.066	3.276	2.772	42.77	36.19
Total 14 Cities	7.029	5.931	295.218	249.102	3,854.235	3,252.165

11.4.5 Summary of External Costs

The values calculated in the previous sub-sections indicate that the external costs that can be made attributable to the emission of air pollutants are rather significant. The following table gives the values for each air pollutant, summing up the estimates across the 14 cities for each air pollutant.

Table 158: Summary of External Costs for 14 British Cities (in million £).

Pollutant	**Total External Costs**			**Road Traffic Related External Costs**		
	Low	Central	Upper	Low	Central	Upper
PM_{10}	7.908	332.136	4,336.22	5.472	229.824	3,000.48
NO_2	13.839	581.238	7,588.385	9.702	407.484	5,319.93
SO_2	5.412	227.304	2,967.58	1.257	52.794	689.255
CO	7.029	295.218	3,854.235	5.931	249.102	3,252.165

The figures for the external costs arising from air pollution induced premature death can not be neglected. Even the estimates presented using the lower bound for the value of a statistical life indicate that a significant extra burden is put on society as a whole. Clearly, pollution emissions generated by road traffic are a major contributor to these costs.

Likewise to the procedure described above, the extrapolation of the city results estimated in this study, leads to a fairly large figure of external costs imposed on the Great Britain economy by air pollution in general and from the road transport sector in particular. These extrapolated figures, assuming that 80 per cent of the population of Great Britain live in urban areas, are presented in the following table.

Table 159: Extrapolated External Costs in Urban Great Britain (in million £).

Pollutant	Total External Costs			Road Traffic Related External Costs		
	Low	Central	Upper	Low	Central	Upper
PM_{10}	28.074	1,179.108	15,393.91	19.425	815.85	1,0651.38
NO_2	49.128	2,063.376	26,938.52	34.443	1,446.606	18,886.25
SO_2	19.212	806.904	10,534.58	4.461	187.362	2,446.115
CO	24.954	1,048.068	13,683.11	21.054	884.268	11,544.61

According to these figures, road traffic alone may cause, for instance, more than 800 million pounds (central estimate) of external costs per year due to the premature death associated with the exposure to PM_{10}, with nitrogen dioxide causing even greater damage (£1.4 billion).

11.5 Summary

This chapter first quantified the total annual deaths due to the exposure of various air pollutants brought forward, and based on these figures, subsequently quantified the external costs associated with these deaths.

In the first part, the exposure-response coefficients as estimated and presented in Chapter 10, were used to derive the total number of deaths that may be associated with the exposure to air pollution. The procedure applied took the total number of non-accidental annual deaths and the annual mean pollution concentrations recorded in the respective British city into account.

Since the main analysis of this study was focused on the road transport sector, a further adjustment has been made. In doing so, the percentage contribution of each pollutant to overall urban concentrations of these pollutants was calculated applying urban-rural dose ratios as suggested by Eyre et al. (1996). The figures were then applied to the estimated total number of deaths previously calculated, resulting in the annual number of premature deaths brought forward due to road traffic related pollution for each pollutant and in each of the cities examined.

Although the estimated figures are rather small, they cannot be neglected. This is especially true when comparing the estimated figures with the number of road accident casualties. In 1997, a total number of 3,599 people were killed on British roads.[8]

Although the numbers have been added up across the various cities for each air pollutant, adding up the figures across the various pollutants for each city would be misleading; it is rather difficult to 'isolate' individual effects attributable to one pollutant solely. In other words, due to co-variation of pollutants, it appears extremely difficult to judge which individual pollutant has caused the recorded effect (i.e. premature death), especially, since it may be possible that some additive or synergistic effects may indeed be responsible for this effect. The figures presented in this chapter represent the total annual number of deaths brought forward due to the exposure to a certain air pollutant per city taking into account current levels of air pollution concentrations.

In the second part, the figures generated in the first part were combined with the estimated values of preventing a statistical fatality, as presented in Chapter 6. Here, following the conservative approach applied throughout the study, a range

[8] See: Department of the Environment, Transport and the Regions (1999), p. 102.

of values including a low, central, and high estimate were computed. Values were produced for the total annual number of deaths due to air pollution, as well as for the specific cases which can be made attributable to the road transport sector as the main source of pollution emission. Again, no attempt to summarise the values for the respective pollutants across each city has been made, in order to avoid double counting due uncertainties regarding the 'responsibility' of the pollutants as well as possible additive or synergistic effects. Furthermore, it can be assumed that the presented figures for the external costs may indeed underestimate the 'true' values.

Besides the uncertainties regarding the estimation of the exposure-response functions, the uncertainties associated with the valuation procedure also support this argument. As discussed previously, some studies suggest a value of a statistical life that is a multiple of that applied in this study.

Further, the approach suggested by Schwab-Christe (1995) proposes to increase the VOSL by up to 100 per cent when including such factors as like the risk of grief and bereavement a person would experience in the event of a relative's death.[9] Nevertheless, the estimates derived still represent a significant amount of external costs in general, and in particular associated with the road transport sector. This holds true, especially, since other important pollutants, such as benzene, lead, 1.3-butadiene and ozone have not been included in the final analysis.

Furthermore, the presented figures 'only' illustrate the estimated external costs imposed on the UK economy associated with premature mortalities due to air pollution. No attempt has been made to also quantify the effects air pollution can

---

[9] See: Schwab-Christe (1995).

have on morbidity. It can be assumed that this would increase the presented estimates significantly.[10]

[10] See, for instance, Maddison et al. (1996).

12. Conclusion

The aim of this study was twofold: first, the estimation of the relationship between daily levels of air pollution concentrations and the daily number of deaths resulting in so-called exposure-response functions; and second, the quantification and valuation of the estimated relationships. The analysis was undertaken in a stepwise manner.

Exposure-response functions were estimated using daily data on mortality and air pollution concentrations, for the six years from 1992 through 1997. The results of multiple Poisson regression analysis indicate an overall strong and positive association between daily concentrations of the different air pollutants examined and day to day changes in mortality. These results could be shown to hold for the majority of the 14 British cities analysed, and are well in line with those of similar studies, specifically those conducted in Western Europe.

The majority of previous studies have been conducted in environments with different air pollution levels to those found in the United Kingdom and thus raised the question of whether these results can be extrapolated to the UK. Hence, the present study is a major contribution to this field, not only confirming the results of previous studies, but also adding a significant amount of new empirical evidence to the existing literature.

Following the methodological framework set by the APHEA project, where the latest findings in epidemiological research are combined with advanced econometric methodology, guarantees reliable results that are easily comparable with those reported in other studies.

Since this study investigated acute effects, the modelling method applied is such that all long term tendencies were eliminated regardless of whether they are created by secular trends, smoking prevalence trends, or air pollution, etc. leaving only short term effects. Thus no attempt has been made to either analyse chronic (long-term) health effects of air pollution, or to examine potential morbidity effects. Chronic health effects of long term exposure to air pollution may lead to a shortened life expectancy. In other words, long term exposure to air pollution may gradually increase the vulnerability of people leading to severe cardio-respiratory problems or even premature death ('creation of new cases').

Further, it can be assumed that the estimated acute health effects, i.e. the few people that are so badly affected they died prematurely, are only a small proportion of the total health effects of air pollution. For a much larger group of people the exposure to air pollution 'only' exacerbates their already existing conditions. Hence, from a public point of view, these effects including additional emergency room visits and hospital admission, as well as days off work and extra therapy, may present an even larger problem.

Based on the exposure-response functions derived in the first part of this study, the absolute number of premature deaths attributable to current air pollution levels was calculated. Combining these figures with the share of air pollution concentration attributable to motor vehicles, the annual number of premature deaths brought about by road traffic induced air pollution was calculated. The figures computed for the different air pollutants can in no means be ignored, neither in absolute terms nor in terms of lives saved per $\mu g/m^3$ reduction of the various air pollutants.

These figures were then 'translated' into monetary terms in order to reflect the external costs inflicted on society by the users of motor vehicles.[1]

The procedure of monetarizing health effects, such as premature mortality, is often criticised for the ethical implications of placing a monetary term on human lives. However, this claim is based on a misunderstanding that not the value of a specific life is being assessed, but rather the benefit of a risk reduction (e.g. improving air quality) that is monetarized (e.g. the number of fatalities reduced by reducing the pollution concentrations by a certain amount). Hence, the terms value of a statistical life (VOSL) or value of preventing a statistical fatality (VPF) reflect more adequately the fact that a decrease in risk is valued before the negative results have already taken place, and not that a specific human life is valued after the impact of air pollution provoked this specific person's death.

For this study, starting from a baseline VPF for road accident fatalities provided by the Department of the Environment, Transport and the Regions, various scenarios of adjustments incorporating the possible loss of life expectancy and quality of life due to impaired health caused by air pollution, lead to a range of estimates for the willingness to pay for reductions in air pollution mortality risks. Combining these values with the estimated absolute number of premature deaths resulted in the total external costs imposed on society by air pollution both in general and from the road transport sector.

These total estimates attributable to the road sector can form the basis for further calculations. Breaking down these figures by different fuel types, for example, gives the marginal external costs per litre of fuel associated with premature

[1] As an 'intermediate' step, the respective costs of air pollution in general, regardless of the original source of emission were also calculated.

mortality. Other such breakdowns may include the marginal external costs by vehicle type per kilometre travelled, or by vehicle type per litre fuel. The resulting figures play important parts in possible tax considerations by the responsible authorities.[2]

An alternative way of expressing the estimated results is by illustrating the benefits of a lower mortality risk due to a reduction in air pollution concentrations by a certain amount. This knowledge is essential for the authorities when, as in the UK, a National Air Quality Strategy is formed where objectives for further reductions in the levels of air pollution are established and assessed.

The respective estimates produced by this study are presented in the following table.

Table 160: Summary of Estimated Benefits per Unit Pollutant Concentration per Year.

Benefits per $\mu g/m^3$ reduction of pollutant (acute deaths) - million £					
City	**PM_{10}**	**NO_2**	**SO_2**	**CO^a**	**Ozone**
Birmingham	0.03 -16.45	0.027 - 14.805	0.018 - 9.87	0.054 - 29.61	0.03 - 16.45
Bristol	0.012 -6.58	0.012 - 6.58	0.024 - 13.16	0.012 - 6.58	N/A
Cardiff	0.015 - 8.225	0.015 - 8.225	N/A	0.018 - 9.87	N/A
Edinburgh	0.03 - 16.45	0.018 - 9.87	0.018 -9.87	0.084 - 46.06	N/A
Glasgow	0.033 - 18.095	0.009 - 4.935	N/A	0.057 - 31.225	N/A
Kingston	0.012 - 6.58	0.006 - 3.29	N/A	0.009 - 4.935	0.012 - 6.58
Leeds	N/A	N/A	0.015 - 8.225	0.087 - 47.705	N/A
Leicester	0.003 - 1.645	N/A	N/A	N/A	0.003 - 1.645
Liverpool	0.021 - 11.515	0.021 - 11.515	0.021 - 11.515	0.114 -62.51	N/A
London	0.114 - 62.510	0.096 - 52.64	0.108 - 59.22	0.369 - 202.335	0.138 - 75.67
Newcastle	0.012 - 6.58	0.018 - 9.87	0.009 - 4.935	N/A	0.009 - 4.935
Sheffield	0.021 - 11.515	0.012 - 6.58	0.021 - 11.515	0.039 - 21.385	N/A
Southampton	0.018 - 9.87	0.012 - 6.58	0.009 - 4.935	0.012 - 6.58	0.015 - 8.225
Swansea	0.006 - 3.29	0.009 - 4.935	0.012 - 6.58	0.015 - 8.225	0.009 - 4.935

a: Calculated per 0.1 ppm reduction.

[2] See, for instance, Maddison et al. (1996) and Mayeres (1993).

Having adopted a conservative approach throughout this study, these values are most likely underestimates. Nevertheless, providing these figures for 14 of the major cities in Britain is an important contribution to the current efforts by the Government to 're-define' transport policy. The publication of the White Paper *A New Deal for Transport: Better for Everyone*, reflects the Government's commitment to giving transport the highest possible priority in shaping a new future for sustainable transport in the UK.[3] The centrepiece of the White Paper is the introduction of local transport plans, where local authorities set out strategies for transport for improving air quality, road safety and public transport and reducing road traffic.[4]

In order to achieve the set targets, they are allowed a wide range of powers at their disposal including road user charging and levies on workplace parking to tackle congestion and pollution with the revenues helping to fund further transport improvements.[5] Additionally, these plans provide the context for bus quality partnerships and include strategies to increase walking and cycling, and the promotion of green transport plans. Since it is for the respective local authority to decide precisely what elements it puts into local transport plans, a sound air pollution policy appraisal is necessary. Such a policy appraisal may include a full cost-benefit analysis with costs and quantified benefits expressed in monetary terms. To do so, the decision-makers need to be fully informed about the detailed consequences of a proposed policy, i.e. have full and clear information on the importance of the health benefits of reducing air pollution, especially since air quality considerations need to be integrated into local authorities' decision making process. Following the Department of the

---

[3] See: Department of the Environment, Transport and the Regions (1998d).

[4] A summary of different approaches of sustainable road transport policies, in various European cities, is given in the Proceedings for the 24th European Transport Forum (1996).

[5] These tools are in combination with those provided in The Road Regulation Act 1984, The Environment Act 1995, The Road Traffic Reduction Act 1997.

Environment, Transport and the Regions "a review and assessment of air quality is the first, and key step in local air quality management regime. It will provide the benchmark for action by local authorities."[6]

Consequently, this study serves as a significant background in assessing the full costs of air pollution due to road traffic. To complete the picture, future research should concentrate on the benefits of reducing the morbidity risks related to air pollution attributable to road transport, again analysing the respective health impacts (hospital admissions, emergency room visits) on a local level (i.e. per city).[7]

Overall, the present study offers probably the most comprehensive analysis of traffic-related mortality effects attributable to air pollution in Britain, covering 14 of the largest cities across the country. So far, studies trying to quantify and value the short-term mortality effects of air pollution have almost solely relied on exposure-response functions derived from non-UK data. This raises doubts on the reliability (in quantity and quality terms) of the resulting estimates. Further, the quantification process in these studies applied average national (or average urban) figures of air pollution concentrations, making no account for air quality differences occurring across the country, specifically in urban areas. Both these serious issues have been addressed in this study, filling the gap of British epidemiological evidence regarding air pollution and mortality as well as the respective quantified effects in terms of numbers of premature deaths and the related external costs.

---

[6] See: DETR website at: http://www.enviornment.detr.gov.uk.

[7] Furthermore, respective effects in rural areas research, especially the adverse effects of ground level ozone being the main rural pollutant, needs to be address in future research.

The significance of this study's results is enhanced by a recent report published by the green lobby group Friends of the Earth who claim that UK air quality in 1999 shows the biggest deterioration since modern records began. According to this report, the average number of days on which air pollution levels were above the Government's air quality standard as measured at a number of key monitoring sites around Britain, rose between 20 and 53 per cent compared with 1998 figures.[8]

Motor vehicles, particularly cars, have revolutionised the way how people live, bringing great flexibility and widening horizons. Road transport largely determines people's quality of life, having great positive impacts on growth and prosperity. However, this all comes with a cost. The adverse effects road transport has, especially on the environment, people's health and in turn on the economy, are often not realised. In the framework of a sustainable transport system, it is for the authorities, based on research evidence, to raise awareness in the public for these negative impacts, encouraging people to take alternative means of travel into consideration in their day to day life.

---

8 See: Friends of the Earth (2000).

References

Abbey, D., B. Hwang, R. J. Burchette, T. Vancuren and P. K. Mills. 1995. Estimated Long-Term Ambient Concentrations of PM_{10} and Development of Respiratory Symptoms in a Nonsmoking Population. *Archives of Environmental Health* 50: 139-153.

Aberle, G. and M. Engel. 1992. Theoretishe Grundlagen zur Erfassung und Bewertung des volkswirtschaftlichen Nutzens. *Internationales Verkehrswesen.* 24 (3): 169-175.

Acton, J. P. 1973. *Evaluating Public Programs to Save Lives: The Case of Heart Attacks*. Research Report R-73-02. Santa Corporation: Santa Monica.

Akaike, H. 1970. Statistical Predictor Identification. *Annals of the Institute of Statistics and Mathematics* 22: 203-217.

Alfaro, J-L., Chapuis, M. and F. Fabre. 1994. *Socio-economic Cost of Road Accidents.* Luxembourg: Cost 313. Commission of the European Communities, Directorate-General XIII.

Allgemeiner Deutscher Automobil Club (ADAC). 1992. *Motorwelt.* Issue 6/92, 39.

Allred, E. N. et al. 1989. Short-term Effects of Carbon Monoxide Exposure on the Exercise Performance of Subjects with Coronary Artery Disease. *New England Journal of Medicine*. 321: 1426-1432.

Allred, E. N. et al. 1991. Effects of Carbon Monoxide on Myocardial Ischaemia. *Environmental Health Perspectives* 91: 89-132.

Al-Osh, M. A. and A. A. Alzaid. 1987. First-Order Integer Valued Autoregressive (INAR(1)) Process. *Journal of Time Series Analysis* 8 (30): 261-275.

Alzaid, A. A. and M. A. Al-Osh. 1988. First-Order Integer Valued Autoregressive (INAR(1)) Process: Distributional and Regression Properties. *Statistica Neerlandica* 42 (1): 53-61.

Anderson, H. R., A. Ponce de Leon, J. M. Bland, J. S. Bower and D. P. Strachan. 1996. Air Pollution and Daily in London: 1987-1992. *British Medical Journal* 312: 665-669.

Anderson, H. R., A. Ponce de Leon, J. M. Bland, J. S. Bower, J. Emberlin and D. P. Strachan. 1998. Air Pollution, Pollen and Daily Admissions for Asthma in London 1987-1992. *Thorax* 53: 842-848.

Anto, J. and Jordi S. 1995. Nitrogen Dioxide and Allergic Asthma: Starting to Clarify an Obscure Association. *Lancet* 345: 402-404.

Arrow, K. J. 1970. The Organization of Economic Activity: Issues Pertinent to the Choice of Market Versus Non-Market Allocation. In *Public Expenditures and Policy Analysis*, ed. Haveman, R. H. and J. Margolis. Chicago: Markham.

Aunan, K. 1996. Exposure-Response Functions for Health Effects of Air Pollution based on Epidemiological Findings. *Risk Analysis* 16 (5): 693-709.

Ayres, J. G. 1997. The Health Effects of Air Pollution in the United Kingdom. In *Air Pollution in the United Kingdom*, ed. Davidson, G. and C. N. Hewitt, 70-84. Cambridge: The Royal Society of Chemistry.

Bachárovarová, L., K. Fandakova, J. Bratinka, M. Budinska, J. Bachar and M. Gudeba. 1996. The Association between Air Pollution and the Daily number of Deaths: Findings from the Slovak Republic Contribution to the APHEA Project. *Journal of Epidemiology & Community Health* 50 (Suppl. 1): S19-S21.

Ball, D., J .P. Brimblecombe and F.M. Nicholas. 1991. *Review of Air Quality Criteria for the Assessment of Near-Field Impacts of Road Transport.* TRRL Report CR 240. Crowthorne: Transport and Road Research Laboratory.

Ball, D. J., R. S. Hamilton and R. M. Harriosn. 1991. The Influence of Highway-Related Pollutants on Enviornmnetal Quality. In *Highway Pollution,* ed. Hamilton,. R. S. and R. M. Harrison, 1-48. Amsterdam, London: Elsevier.

Bator, Francis M. 1958. The Anatomy of Market Failure. *Quarterly Journal of Economics* 72 (August): 351-379.

Baumol, W. J. and W. E. Oates. 1990. The Theory of Environmental Policy. 3rd Edition. Cambridge: Cambridge University Press.

Baxter, Laurance A., Stephen J. Finch, Frederick W. Lipfert and Qiqing Yul. 1997. Comparing Estimates of the Effects of Air Pollution on Human Mortality Obtained Using Different Regression Methodologies. *Risk Analysis* 17 (3): 273-278.

Beattie, Jane, Judith Covey, Paul Dolan, Lorraine Hopkins, Michael Jones-Lee, Graham Loomes, Nick Pidgeon, Angela Robinson and Anne Spencer. 1998. On the Contingent Valuation of Safety and the Safety of Contingent Valuation: Part 1 - *Caveat Investigator*. *Journal of Risk and Uncertainty* 17: 5-25.

Blomquist, G. C. 1981. The Value of Human Life: An Empirical Perspective. *Economic Inquiry* 19: 157-164.

Blomquist, G. C., T. R. Miller and D. T. Levy. 1996. Values of Risk Reduction Implied by motorist Use of Protection Equipment: New Evidence from Different Populations. *Journal of Transport Economics and Policy* 30: 55-66.

Bonnafous, A. 1994. Summary and Conclusions to Internalising the Social Costs of Transport. In *Internalising the Social Costs of Transport*, ed. The European Conference of Ministers of Transport (ECMT), 181-191. Paris: OECD

Bown, William. 1994. Dying from too much Dust. *New Scientist* 12: 13.

Brännäs, Kurt. 1995. Prediction and Control for a Time-Series Count Data Model. *International Journal of Forecasting* 11: 263-270.

Bremner, S. A, H. R. Anderson, R. W. Atkinson, A. J. McMichael, D. P. Strachan, J. M. Bland and J. S. Bower. 1999. Short Term Associations between Outdoor Air Pollution and Mortality in London 1992-94. *Occupational Environmental Medicine* 56: 237-244.

Breyer, F. and P. Zweifel. 1996. *Gesundheitsökonomie. Zweite überarbeitete und erweiterete Auflage*. Berlin: Springer-Verlag.

British Road Foundation. 1998. *Basic Road Statistics 1998*. London: British Road Foundation.

Bromley, D. W. 1989. *Economic Interests and Institutions.* Oxford: Basil: Blackwell.

Broome, J. 1978a. Trying to Value a Life. *Journal of Public Economics* 9: 91-100.

Broome, J. 1978b. The Economic Value of Life: A Reply. *Economica* 54 (215): 240.

Broome, J. 1979. Trying to Value a Life: A Reply. *Journal of Public Economics* 12: 259-262.

Broome, J. 1982. Uncertainty in Welfare Economics and the Value of Life. In *The Value of life and Safety: Proceedings of a Conference held by the Geneva Association*, ed. Jones-Lee, M. W., 201-216. Amsterdam: North Holland Publishing Company.

Broome, J. 1985. The Economic Value of Life. *Economica* 52: 281-294.

Broughton, G. F. J., J. S. Bower, P. G. Willis and H. Clark. 1997. *Air Pollution in the UK 1995.* National Environmental Technology Centre. Oxfordshire: AEAT.

Brunekreef, Bert, Douglas Dockery, et. al. 1995. Epidemiologic Studies on Short-Term Effects of Low Levels of Major Ambient Air Pollution Components. *Environmental Health Perspectives* 103: 3-14.

Buchanan J. M. and W. E. Stubblebine. 1962. Externality. *Economica* 29 (Nov.): 371-384.

Buchanan, J. M. and G. Tullock. 1962. *The Calculus of Consent.* Ann Arbor, Michigan: University of Michigan Press.

Buchanan, J. M. and Roger L. Faith. 1979. Trying Again to Value a Life. *Journal of Public Economics* 12: 245-248.

Burnett, R. T., R. E. Dales, M. E. Raizenne, D. Krewski, P. W. Summers, G. R. Roberts, M. Raad-Young, T. Dann and J. Brook. 1994. Effects of Low Ambient Levels of Ozone and Sulfates on the Frequency of Respiratory Admissions in Ontario Hospitals. *Environmental Research* 65: 172-194.

Button, K. J. 1993. *Transport Economics. 2nd Edition*. Cambridge: Edward Elgar Publishing Limited.

Button, K. J. 1994. Overview of Internalising the Social Costs of Transport. In *Internalising the Social Costs of Transport*, ed. The European Conference of Ministers of Transport, 7-31. Paris: OECD.

Calthrop, E. and D. Maddison. 1996. The Dose-Response Function Approach to Modelling the Health Effects of Air Pollution. *Energy Policy*. 24 (7): 599-607.

Cameron, A. Colin and Pravin K. Trivedi. 1986. Econometric Models Based on Count Data: Comparisons and Applications of some Estimators and Tests. *Journal of Applied Econometrics* 1: 29-53.

Cameron, A. Colin and Pravin K. Trivedi. 1990. Regression-Based Tests for Overdispersion in the Poisson Model. *Journal of Econometrics* 46: 347-364.

Cameron, A. Colin and Pravin K. Trivedi. 1998. *Regression Analysis of Count Data*. Cambridge: Cambridge University Press.

Carthy, Trevor, Susan Chilton, Judith Covey, Lorraine Hopkins, Michael Jones-Lee, Graham Loomes, Nick Pidgeon and Anne Spencer. 1998. On the Contingent Valuation of safety and the Safety of Contingent Valuation: Part 2 - The CV/SG "Chained" Approach. *Journal of Risk and Uncertainty* 17 (3): 187-213.

Chambers, J. M., and W. S. Cleveland, B. Kleiner and P. A. Tukey. 1983. *Graphical Methods for Data Analysis*. Boston: Duxbury Press.

Chappie, Mike and Lester Lave. 1982. The Health Effects of Air Pollution: A Re-analysis. *Journal of Urban Economics* 12: 346-376.

Charlton, John. 1996. Which Areas are Healthiest? *Population Trends* 83: 17-24.

Cheung, Steven N. S. 1973. The Fable of the Bees: An Economic Investigation. *The Journal of Law and Economics* 16 (April): 11-33.

Chilton, S, L., M. W. Jones-Lee, G. Loomes, A. Robinson, R. Cookson, J. Covey, A. Spencer, L. Hopkins, N. Pidgeon and J. Beattie. 1998. *Valuing Health and Safety controls: A Literature Review*. Contract Research Report 171/1998. London: Health and safety Executive.

Christopherson, O. 1997. Mortality during the 1996/97 Winter. In *Population Trends 90*, ed. Office for National Statistics, 11-19. London: HMSO.

Clapham, J. H. 1922. Of Empty Economic Boxes. *The Economic Journal* 32 (Sept.): 305-314.

Cleveland, W. S. 1979. Robust Locally Weighted Regression and Smoothing Scatterplots. *Journal of the American Statistical Association* 74: 829-838.

Cleveland, W. S. 1984. Graphs in Scientific Publications. *The American Statistician* 74: 261-269.

Cleveland, W. S. 1985. *The Elements of Graphing Data*. Murray Hill, New Jersey: Bell Telephone Laboratories, Inc.

Cleveland, W. S. 1993. *Visualizing Data*. New York: Hobart Press.

Coase, R. H. 1960. The Problem of Social Cost. *The Journal of Law and Economics* III: 1-44.

Cohen, A. J., and C. A. Pope III. 1995. Lung Cancer and Air Pollution. *Environmental Health Perspectives* 103: 219-225.

Complainville, C. and J. O. Martins. 1994. *NO_x / SO_x Emissions and Carbon Abatement*. OECD Working Paper No. 69, Paris.

Conley, Bryan. 1978. The Value of Human Life in the Demand for Safety: Extension and Reply. *The American Economic Review* 68 (4): 717-720.

Cook, Philip. 1978. The Value of Human Life in the Demand for Safety: Comment. *The American Economic Review* 68 (4): 710-711.

Cooper, J. and J. Loomis. 1992. Sensitivity of Willingness to Pay Estimates to Bid Design in Dichotomous Choice Contingent Valuation Models. *Land Economics* 68 (2): 211-224.

Cooper, J. and J. Loomis. 1993. Sensitivity of Willingness to Pay Estimates to Bid Design in Dichotomous Choice Contingent Valuation Models: A Reply. *Land Economics* 69 (2): 203-208.

Cornes R. and T. Sandler. 1996. *The Theory of Externalities, Public Goods and Club Goods*. 2nd Edition. Cambridge: Cambridge University Press.

Coursey, Don L., John L. Hovis and William D. Schulze. 1987. The Disparity Between Willingness to Accept and Willingness to Pay Measures of Value. *The Quarterly Journal of Economics* 102: 679-690.

Covey, J., M. W. Jones-Lee, G. Loomes and A. Robinson. 1995. The Exploratory Empirical Study. In *Exploratory Study of Consumers' Willingness to Pay for Food Risk Reduction*, ed. Ives, D. B. Soby, G. Goats, D. J. Ball, J. Covey, M. W. Jones-Lee, G. Loomes and A. Robinson. Report to MAFF. University of East Anglia.

Cropper, M. L., S. K. Aydede and P. R. Portney. 1992. Rates of Time Preference for Saving Lives. *American Economic Review* 82: 469-472.

Cropper, Maureen L., Sema K. Aydede and Paul R. Portney. 1994. Preferences for Life Saving Programs: How the Public Discounts Time and Age. *Journal of Risk and Uncertainty* 8: 243-265.

Cummings, R. G., D. S. Brookshire and W. D. Schulze. 1986. *Valuing Public Goods: A State of the Arts Assessment of the Contingent Valuation Method*. Totowa, NJ: Rowman and Allanheld.

Cummings, R. G. and L. Osborne Taylor. 1998. Does Realism Matter in Contingent Valuation Surveys? *Land Economics* 74 (2): 203-215.

Dab, W., S. Medina, P. Quenel, Y. Le Moullee, A. Le Tetre, B. Thelot, C. Monteil, P. Lameloise, P. Pirard, I. Momas, R. Ferry and B. Festy. 1996. Short Term Respiratory Health Effects of Ambient Air Pollution: Results of the APHEA Project in Paris. *Journal of Epidemiology & Community Health* 50 (Suppl. 1): S42-S46.

Dahlman, Carl J. 1979. The Problem of Externality. *The Journal of Law and Economics* 22 (April): 141-162.

Dalvi, M. Q. 1988. *The Value of Life and Safety: A Search for a Consensus Estimate*. London: HMSO.

Dardis, Rachel. 1980. The Value of a Life: New Evidence from the Marketplace. *The American Economic Review* 70 (5): 1077-1082.

Davis, R. K. 1964. The Value of Big Game Hunting in a Private Forrest. In *Transactions of the Twenty-ninth North American Wildlife Conference*. Washington, D.C.: Wildlife Management Institute.

Dean, C., and J. F. Lawless. 1989. Tests for detecting Overdispersion in Poisson Regression Models. *Journal of the American Statistical Association* 84 (406): 467-472.

Delgado, Miguel A. and Peter M. Robinson. 1992. Nonparametric and Semiparametric Methods for Economic Research. *Journal of Economic Surveys* 6 (3): 201-249.

Delucchi, M. A. 1997. The Annualized Social Cost of Motor-Vehicle Use in the U.S., Based on 1990-1991 Data: Summary of Theory, Data, Methods, and Results. In *The Full Costs and Benefits of Transportation: Contributions to Theory, Method and Measurement,* ed. Greene, D. L., Jones, R. W. and M. A. Delucchi, 27-69. Berlin, New York: Springer-Verlag.

Demsetz, H. 1967. Towards a Theory of Property Rights. *American Economic Review* 57: 347-359.

Department of Commerce, National Oceanic and Atmospheric Administration (NOAA). 1993. Report of the NOAA Panel on Contingent Valuation. *Federal Register* 58 (10): 4602-4614

Department of Health. 1998. Committee on the Medical Effects of Air Pollution. *Quantification of the Effects of Air Pollution on Health in the United Kingdom*. London: The Stationary Office.

Department of Health. 1999. *Economic Appraisal of the Health Effects of Air Pollution*. Ad-Hoc Group on the Economic Appraisal of Health Effects of Air Pollution. London: The Stationary Office.

Department of the Environment, Transport the Regions. 1998a. *National Travel Survey*. London: The Government Statistical Service. The Stationary Office.

Department of the Environment, Transport the Regions. 1998b *Transport Statistics Great Britain: 1998 Edition.* 24th Edition. London: The Government Statistical Service. The Stationary Office.

Department of the Environment, Transport the Regions. 1998c *Digest of Environmental Statistics 1997.* No. 19 Vol. 1. London: The Government Statistical Service. The Stationary Office.

Department of the Environment, Transport and the Regions. 1998d. *A New Deal for Transport: Better for Everyone*. London: The Stationary Office.

Department of the Environment, Transport the Regions. 1999. *Transport Statistics Great Britain: 1999 Edition.* 25th Edition. London: The Government Statistical Service. The Stationary Office.

Department of Trade and Industry. 1999. *Digest of United Kingdom Energy Statistics 1999*. London: The Government Statistical Service. The Stationary Office.

Desaigues, B. and A. Rabl. 1995. Reference Values for Human Life: An Econometric Analysis of a Contingent Valuation in France. In *Contingent Valuation, Transport Safety and the Value of Life*, ed. Schwab-Christe, N. G. and N. C. Soguel, 85-112. Boston: Kluwer.

Deutsche Straβenliga. 1992. Straβe, Verkehr und Wirtschaft. in *SVW-Info-Diens*t. 2 July .

Diamond, Peter. 1996. Testing the Internal Consistency of Contingent Valuation Surveys. *Journal of Environmental Economics and Management* 30: 337-347.

Diggle, P.J. 1990. *Time-Series: A Biostatistical Introduction*. Oxford: Oxford Science Publications.

Diggle, P. J, K-Y Liang and S. L. Zeger. 1994. *Analysis of Longitudinal Data.* Oxford: Clarendon Press.

Dockery, D. W., J. Schwartz and J. D. Spengler. 1992. Air Pollution and Daily Mortality: Association with Particulates and Acid Aerosols. *Environmental Research* 59: 362-373.

Dockery, D. W., C. Arden Pope III, Xipeng XU, John D. Spengler, James H. Ware, Martha E. Fay, Benjamin G. Ferris and Frank E. Speizer. 1993. An Association between Air Pollution and Mortality in Six U.S. Cities. *The New England Journal of Medicine* 329 (24): 1753-1760.

Dockery, D. W. and C. A. Pope III. 1994. Acute Respiratory Effects of Particulate Air Pollution. *Annual Review of Public Health* 15: 107-132.

Dockery, Douglas and Arden Pope III. 1996. Epidemiology of Acute Health Effects: Summary of Time-Series Studies. In *Particles in Our Air: Concentrations and Health Effects*, ed. Wilson, J. and J. Spengler, 123-149. Cambridge, MA: Harvard School of Public Health.

Donnelly, John C. 1994. Is the Indoor Air you Breathe Safe? *Safetyline:* 12-14.

Drever, Frances, Maragret Whitehead and Murray Roden. 1996. Current Patterns and Trends in Male Mortality by Social Class (based on Occupation). *Population Trends* 83: 15-35.

Dreyfus, M. and W. K. Viscusi. 1995. Rates of Time Preference and Consumer Valuation of Automobile Safety and fuel efficiency. *Journal of Law and Economics* 38: 79-105.

Dubourg, W. R., M. W. Jones-Lee and Graham Loomes. 1994. Imprecise Preferences and the WTP-WTA Disparity. *Journal of Risk and Uncertainty* 9: 115-133.

Dubourg, W. R. 1996. Estimating the Mortality Costs of Lead Emissions in England and Wales. *Energy Policy* 24 (7): 621-625.

Ebert, Udo. 1998. *A General Approach to the Evaluation of Nonmarket Goods*. Universität Oldenburg Working Paper, V-184-98.

ECOPLAN. 1992. *Externe Kosten im Agglomerationsverkehr*. Nationales Forschungsprogramm Stadt und Verkehr. Fallbeispiel Region Bern, Band 15B. Zürich.

ECOPLAN. 1993. *Externe Nutzen des Verkehrs*. . Nationales Forschungsprogramm Stadt und Verkehr. Fallbeispiel Region Bern, Band 39. Zürich.

Edwards, J., S. Walters and R. K. Griffiths. 1994. Hospital Admissions for Asthma in Preschool Children: Relationship to Major Roads in Birmingham, United Kingdom. *Archives of Environmental Health* 49: 223-228.

Ellis, Howard S. and William Fellner. 1943. External Economies and Diseconomies. *The American Economic Review* XXXIII (3): 493-511.

Elvik, Rune. 1995. An Analysis of Official Economic Valuations of Traffic Accident Fatalities in 20 Motorised Countries. *Accident Analysis and Prevention* 27 (2): 237-247.

European Commission. 1995a. *Externalities of Energy: ExternE Project. Volume 1: External Costs of Energy.* DGXII, Science, Research and Development; JOULE Programme. Brussels: Office for Official Publications of the European Communities.

European Commission. 1995b. *Externalities of Energy: ExternE Project. Volume 2: External Costs of Energy.* DGXII, Science, Research and Development; JOULE Programme. Brussels: Office for Official Publications of the European Communities.

European Conference of Ministers of Transport (ECMT). 1992. *Benefits of Different Transport Modes: Report of the Ninety-Third Round Table on Transport Economics.* Lyon: European Conference of Ministers of Transport (ECMT)

European Transport Forum. 1996. *Proceedings of Seminar C: Planning for Sustainability.* Brunel University: PTRC Education and Research Services Ltd.

Evans, J. S., T. Tosteson and P. L. Kinney. 1994. Cross-Sectional Mortality Studies and Air Pollution Risk Assessment. *Environmental International* 10: 55-83.

Ewetz, L. and P. Camner. 1983. *Motor Vehicles and Cleaner Air.* Stockholm: Swedish Ministry of Agriculture.

Eyre, N.J., E. Ozdemiroglu, D. W. Pearce and P. Steele. 1996. *Damage Costs of Transport Emissions - Geographical and Fuel Dependence.* Working Paper WM 96-02. University College London and University of East Anglia: CSERGE.

Fairly, D. 1990. The Relationshipt of Daily Mortality to Suspended Particulates in Santa Clara County, 1980-1986. *Environmental Health Perspectives* 89: 159-168.

Farr, W. 1876. *Contribution to 39th Annual Report of the Registrar General of Births, Marriages, and Deaths in England and Wales*. London.

Firth, D. 1991. Generalized Linear Models. In *Statistical Theory and ModellingL in Honor of Sir David Cox, FRS*, ed. Hinkley D. V., N. Reid and E. J. Snell, 55-82. London: Chapman and Hall.

Fischer, Gregory W., M. Granger Morgan, Baruch Fischhoff, Indira Nair and Lester B. Lave. 1991. What Risks are People Concerned about? *Risk Analysis* 11 (2): 303-314.

Fisher, Ann, Lauraine G. Chestnut and Daniel M. Violette. 1989. The Value of Reducing Risks of Death: A Note on New Evidence. *Journal of Policy Analysis and Management* 8 (1): 88-100.

Ford, J. L., P. K. Pattanaik and X. Wei. 1995. On Measuring the Value of Life. *Economics Letters* 49: 223-230.

Forman, Richard T. T. and Lauren E. Alexander. 1998. Roads and their Major Ecological Effects. *Annual Review of Ecological Systems* 29: 207-231.

Frankel, M. 1979. *Hazard Opportunity and the Valuation of Life*. Mimeo, University of Illinois at Urbana-Champagne.

Fraser, Clive D. 1984. Optimal Compensation for Potential Fatality. *Journal of Public Economics* 23: 307-332.

Freeman, A. M. 1993. *Measurement of Environmental and Resource Values: Theory and Methods*. Washington, DC: Resources for the Future.

Friends of the Earth. 2000. *1999 Grim Year for Air Pollu*tion. Press Release. Friends of the Earth.

Gastaldi, M., J-P. Pradayrol, E. Quinet and M. Rega. 1996. Valuation of Environmental Externalities: From Theory to Decision-Making. *Transportation Planning and Technology* 19: 207-219.

Gee. David. 1997. Approaches to Scientific Uncertainty. In *Health at the Crossroads: Transport Policy and Urban Health*, ed. Fletcher, Tony and Anthony J. McMichael, 27-51. Chichester, New York: John Wiley & Sons.

Gerking, S., M. De Haan and W. Schulze. 1988. The Marginal Value of Job Safety: A Contingent Valuation Study. *Journal of Risk and Uncertainty* 2: 185-199.

Gilbert, R. F. 1995. The Alleged Persistent Misapplication of Economics Versus The Economic Value Of Life: A Comment. *Journal of Forensic Economics* 8 (3): 279-286.

Gilbert, R. F. 1998. Special Problems with Value-Of-Life Estimates: A Reappraisal. *Journal of Forensic Economics* 11 (1): 47-50.

Gourieroux, C., Montfort A. and A. Trognon. 1984a. Pseudo Maximum Likelihood Methods: Theory. *Econometrica* 52 (3): 681-700.

Gourieroux, C., Montfort A. and A. Trognon. 1984b. Pseudo Maximum Likelihood Methods: Applications to Poisson Models. *Econometrica* 52 (3): 701-720.

Graedel, T. E., D. T. Hawkins and L. D. Claxon. 1986. *Atmospheric Chemical Compounds*. Orlando: Academic Press.

Green. Malcolm. 1994. Introduction to How Vehicle Pollution Affects Our Health. In *How Vehicle Pollution Affects Our Health*, ed. Read Cathy, 2-4. London: The Ashden Trust.

Greene, W. 1997. *Econometric Analysis*. 3rd Edition. New York: Prentice Hall Inc.

Gregory, Robin, Sarah Lichtenstein and Pail Slovic. 1993. Valuing Environmental Resources: A Constructive Approach. *Journal of Risk and Uncertainty* 7: 177-197.

Gurmu, Shiferaw. 1991. Tests for detecting Overdispersion in the Positive Poisson Regression Model. *Journal of Business & Economic Statistics* 9 (2): 215-222.

Halkos, G. E. 1998. Evaluating the Direct Costs of Controlling NOx Emissions in Europe. *Energy Source* 20 (3): 223-239.

H. M. Treasury. 1991. *Economic Appraisal in Central Government: A Technical Guide for Government Department*. London: HMSO.

Hamilton, R. S. and R. M. Harrison. 1991. Highway Pollutant Monitoring. In *Highway Pollution*. Hamilton, ed. R. S. and R. M. Harrison, 48-100. Amsterdam, London: Elsevier.

Hanemann, M. 1994. Valuing the Environment through Contingent Valuation. *Journal of Economic Perspectives* 8 (4): 19-43.

Hanley, N. D. 1989. Valuing Non-Market Goods Using Contingent Valuation. *Journal of Economic Surveys* 3 (3): 235-252.

Hanley, N. and C. L. Spash. 1993. *Cost-Benefit Analysis and the Enviornment*. Aldershat: Elgar.

Hanley, N., J. Shogren and B. White. 1997. *Environmental Economics in Theory and Practice*. London: Macmillan.

Harrison, R. M. 1997. Urban Air Pollution in the United Kingdom. In *Air Pollution in the United Kingdom*, ed. Davidson, G. and C. N. Hewitt, 22-39. Cambridge: The Royal Society of Chemistry.

Harvey, A. C. and C. Fernandes. 1989. Time Series Models for Count or Qualitative Observations. *Journal of Business & Economic Statistics* 7 (4): 407-417.

Harvey, A. C. 1990. *The Econometric Analysis of Time Series. 2nd edition.* New York, London: Philip Allan.

Hastie, T. J. and R. J. Tibshirani. 1990. *Generalized Additive Models*. London, New York: Chapman and Hall Ltd.

Hatzakis, Angelos, Klea Katsouyanni, Anna Kalandidi, Nicholas Day and Dimitros Trichopoulos. 1986. Short-Term Effects of Air Pollution on Mortality in Athens. *International Journal of Epidemiology* 15 (1): 73-81.

Hauer, E. 1994. Can One Estimate the Value of Life or Is it Better to be Dead than Stuck in Traffic? *Transport Research-A* 28A (2): 109-118.

Hausman, J., B. H. Hall and Z. Griliches. 1984. Econometric Models for Count Data with an Application to the Patents-R&D Relationship. *Econometrica* 52 (4): 909-938.

Heller, W. P. and D. A. Starrett. 1976. On the Nature of Externalities. In *Theory and Measurement of Economic Externalities*. ed. Lin, S. A. Y. New York: Academic Press.

Hexter, A. C. and J. R. Goldsmith. 1971. Carbon Monoxide: Association of Community Air Pollution with Mortality. *Science* 172: 265-267.

Hodgson, T. A. 1983. The State of the Art of Cost-of-Illness Estimates. *Advances in Health Economics and Health Services Research* 4: 129-164.

Hoevenagel, R. An Assessment of the Contingent Valuation method. In *Valuing the Environment: Methodological and Measurement Issues*, ed. Pethig, R, 195-228. Dordrecht: Kluwer Academic Publishers.

Hoevenagel, R. A Comparison of Economic Valuation Methods. In *Valuing the Environment: Methodological and Measurement Issues*, ed. Pethig, R, 251-270. Dordrecht: Kluwer Academic Publishers.

Hohmeyer, O., R. L. Ottinger and K. Rennings. 1995. *Social Costs and Sustainability. Proccedings of an International Conference, held at Ladenburg, Germnay, May 27-30, 1995*. Berlin: Springer-Verlag.

Holland, W. W., A.E. Bennett, I. R. Cameron, C. du Florey, S. R. Leeder, R. Schilling, A. Swan and R. Waller. 1979. Health Effects of Particulate Pollution: Reappraising the Evidence. *American Journal of Epidemiology* 110 (5): 527-659.

Holman C. 1989. *Particulate Pollution from Diesel Vehicles*. London: Friends of the Earth.

Holman, C. 1991. *Air Quality & Health*. London: Friends of the Earth.

Holman, Claire. 1994. How much does Road Traffic contribute to Air Pollution? In *How Vehicle Pollution Affects Our Health*, ed. Read Cathy, 5-9. London: The Ashden Trust.

Horowitz, John K. and Richard T. Carson. 1990. Discounting Statistical Lives. *Journal of Risk and Uncertainty* 3: 403-413.

Hughes, William T. and C. F. Sirmans. 1992. Traffic Externalities and Single-Family House Prices. *Journal of Regional Science* 32 (4): 487-500.

Ippolito, Pauline M. and Richard A. Ippolito. 1984. Measuring the Value of Life Saving from Consumer Reactions to New Information. *Journal of Public Economics* 25: 53-81.

Isham, V. 1991. Modelling Stochastic Phenomena. In *Statistical Theory and ModellingL in Honor of Sir David Cox, FRS*, ed. Hinkley D. V., N. Reid and E. J. Snell, 177-203. London: Chapman and Hall.

Ito, K., P. Kinney and G. D. Thurston. 1995. Variations in PM_{10} Concentrations within Two Metropolitan Areas and their Implication for Health Effects Analysis. *Journal of Inhalation Toxicology* 7 (5): 735-745.

Jantunen M. 1997. *Socioeconomic and Cultural Factors in Air Pollution Epidemiology*. Report Number 8. Directorate-Generale XII. Brussels: European Commission.

Johannesson, Magnus and Per-Olov Johansson. 1997b. Quality of Life and the WTP for an Increased Life Expectancy at an Advanced Age. *Journal of Public Economics* 65: 219-228.

Johannesson, Maguns and Per-Olov Johannson. 1997a. Saving Lives in the Present versus Saving Lives in the Future – Is there a Framing Effect? *Journal of Risk and Uncertainty* 15: 167-176.

Johannesson, Magnus, Per-Olov Johannson and Richard M. O'Conor. 1996. The Value of Private Safety versus the Value of Public Safety. *Journal of Risk and Uncertainty* 13: 263-275.

Johannesson, Magnus, Per-Olov Johansson and Karl-Gustaf Löfgren. 1997. On the Value of Changes in Life Expectancy: Blips versus Parametric Changes. *Journal of Risk and Uncertainty* 15: 221-239.

Johansson, P. O. 1993. *Cost-Benefit Analysis of Environmental Change*. Cambridge: Cambridge University Press.

Johansson, P. O. 1994. Altruism and the Value of Statistical Life: Empirical Implications. *Journal of Health Economics* 13: 111-118.

Johnson, David B. 1973. Meade, Bees, and Externalities. *The Journal of Law and Economics* XVI: 35-52.

Jones, Lisa. 1995. Dirty Indoor Air can Cause Illness and Irritation during the Winter. *Total Health* 17: 29-32.

Jones-Lee, M. W. 1976. *The Value of Life: an Economic Analysis*. London: Martin Robertson.

Jones-Lee, M. W. 1978. The Value of Human Life in the Demand for Safety: Comment. *The American Economic Review* 68 (4): 712-716.

Jones-Lee, M. W. 1979. Trying to Value a Life: Why Broome Does not Sweep Clean. *Journal of Public Economics* 12 (October): 249-256.

Jones-Lee, M. W. 1982. *The Value of Life and Safety: Proceedings of a Conference held by the Geneva Association*. Amsterdam: North Holland Publishing Company.

Jones-Lee, M. W. 1987. The Economic Value of Life: A Comment. *Economica* 54: 397-400.

Jones-Lee, M. W. 1989. *The Economics of Safety and Physical Risk*. Oxford: Basil Blackwell.

Jones-Lee, M. W. 1990. The Value of Transport Safety. *Oxford Review of Economic Policy* 6 (2): 39-59.

Jones-Lee, M. W. 1991. Altruism and the Value of Other People's Safety. *Journal of Risk and Uncertainty* 4: 213-219.

Jones-Lee. M. W. 1992. Paternalistic Altruism and the Value of Statistical Life. *The Economic Journal* 102 (January): 80-90.

Jones-lee, M. W., M. Hammerton and P. R. Philips. 1985. The Value of Safety: Results of a National Sample Survey. *Economic Journal* 95: 49-72.

Jones-Lee, M. W. and G. Loomes. 1994. Towards a Willingness to Pay Based Value of Underground Safety. *Journal of Transport Economics and P*olicy 28: 83-98.

Jones-Lee, M.W. and G. Loomes. 1995. Scale and Context Effects in the Valuation of Transport Safety. *Journal of Risk and Uncertainty* 11: 183-203.

Jones-Lee, M. W., G. Loomes and P. R. Philips. 1995. Valuing the Prevention of Non-Fatal Road Injuries: Contingent Valuation vs. Standard Gambles. *Oxford Economic Papers* 47: 676-695.

Jones-Lee, M. W. and G. Loomes. 1996. *Theoretical and Empirical Issues Concerning the Valuation of Safety*. ESRC Report 234987.

Judge, G. G., W. E. Griffiths, R. Carter Hill, H. Lütkepohl and T-C. Lee. *The Theory and Practice of Econometrics*. 2nd Edition. New York: John Wiley & Sons.

Kågeson, P. 1994. Effects of Internalisation on Transport Demand and Modal Split. In *Internalising the Social Costs of Transport*, ed. The European Conference of Ministers of Transport (ECMT), 77-95. Paris: OECD

Kahn, Shulamit. 1986. Economic Estimates of the Value of Life. *IEEE Technology and Society Magazine,* 24-31.

Kahneman, D. and A. Tversky. 1979. Prospect Theory: An Analysis of Decision under Risk. *Econometrica* 47: 263-291.

Kalkstein, L. S., G. Tan and J. Skindlov. 1987. An Evaluation of Objective Clustering Procedures for Use in Synoptic Climatological Classification. *Journal of Climate and Applied Meterology* 26: 717-730.

Kalkstein, L. S. 1993. Direct Impacts in Cities. *The Lancet* 342: 1397-1399.

Kanninen, Barbara. 1995. Bias in Discrete Response Contingent Valuation. *Journal of Environmental Economics and Management* 28 (1): 114-125.

Kapp, K. William. 1969. On the Nature of Significance of Social Costs. *Kyklos* 22: 334-347.

Katsouyanni, K. 1995. Health Effects of Air Pollution in Southern Europe: Are there Interacting Factors? *Environmental Health Perspective* 103: 23-28.

Katsouyanni, K., J. Schwartz, C. Spix, G. Touloumi, D. Zmirou, A. Zanobetti, , B. Wojtyniak, J.M. Vonk, A. Tobias, A. Ponka , S. Medina, L. Bacharova, H.R. Anderson. 1996. Short term effects of air pollution on health: A European approach using epidemiologic time series data: the APHEA protocol, *Journal of Epidemiology & Community Health* 50 (Suppl. 1): S12-S18.

Katsouyanni, K. 1997. Research Methods in Air Pollution Epidemiology. In *Health at the Crossroads - Transport Policy and Urban Health*, ed. Fletcher, T. and A. J. McMichael, 51-61. Chichester, New York: John Wiley & Sons Ltd..

Katsouyanni, K., G. Touloumi, C. Spix, J. Schwartz, F. Balducci, S. Medina, G. Rossi, B. Wojtyniak, J. Sunyer, L. Bacharova, J. P. Schouten, A. Ponka and H. R. Anderson. 1997a. Short term effects of Ambient Sulphur Dioxide and Particulate Matter on Mortality in 12 European Cities: Results from Time Series Data from the APHEA Project. *British Medical Journal* 314: 1658-1663.

Katsouyanni, K., D. Zmirou, C. Spix, J. Sunyer, J. P. Schouten, A. Ponka, H. R. Anderson, Y. Le Moullec, B. Wojtyniak, M. A. Vigotti, L. Bacharova and J. Schwartz. 1997b. Short term Effects of Air Pollution on Health: A European Approach Using Epidemiologic Time Series Data: The APHEA Project. *Public Health Reviews* 25: 7-18.

Khaw, Kay-Tee. 1995. Temperature and Cardiovascular Mortality. *Lancet* 345: 337-339.

Kidholm, K. 1995. Assesing the Value of Traffic Safety Using the Contingent Valuation Technique: The Danish Survey. In *Contingent Valuation, Transport Safety and the Value of Life*, ed. Schwab-Christe, N. G. and N. C. Soguel, 45-62. Boston: Kluwer.

King, Gary. 1989. Variance Specification in Event Count Models: From Restrictive Assumptions to a Generalized Estimator. *American Journal of Political Science* 33 (3): 762-784.

Kinney, P. L. and Halûk Özkaynak. 1991. Associations of Daily Mortality and Air Pollution in Los Angeles County. *Environmental Research* 54: 99-120.

Kinney, P. L., K. Ito and G. D. Thurston. 1995. A Sensitivity Analysis of Mortality/ PM_{10} Associations in Los Angeles. *Inhalation Toxicology* 7: 59-69.

Kleinman, Michael, T. and William J. Mautz. 1996. *How Does Exercise Affect the Dose of Inhaled Air Pollution?* Research Report Number 45. The Health Effects Institute.

Kleinman, M. T. et al. 1997. Effects of Short-Term Exposure to Carbon Monoxide in Subjects with Coronary Artery Disease. *Archives of Environmental Health.* 44: 361-369.

Knetsch, Jack and J. A. Sinden. 1984. Willingness to Pay and Compensation Demanded: Experimental Evidence of an Unexpected Disparity in Measures of Value. *The Quarterly Journal of Economics* 98: 507-521.

Knight, F. H. 1924. Some Fallacies in the Interpretation of Social Cost. *Quarterly Journal of Economics* XXXVIII: 582-606.

Koenig J. Q. 1995. Effect of Ozone on respiratory Responses in Subjects with Asthma. *Environmental Health Perspectives* 103: 103-106.

Komanoff, Charles. 1994. Pollution for Roadway Transportation. *PACE Environmental Law Review* 12 (1): 121-160.

Korc, Marcelo E. 1996. A Socioeconomic Assessment of Human Exposure to Ozone in the South Cost Air Basin of California. *Journal of Air and Waste Management Association* 46: 547-557.

Koutrakis, P. and C. Sioutas. Physico-Chemical Properties and Measurement of Ambient Particles. 1996. In *Particles in Our Air: Concentrations and Health Effects*, ed. Wilson, J. and J. Spengler, 15-40. Cambridge, MA: Harvard School of Public Health.

Kram, T.,E. Schol, A. Stoffer, W. Rothengatter, A. Günnemann, M. M. Sørensen, A. Stouge, S. Suter and F. Walter. 1996. *External Costs of Transport and Internalisation: Synthesis Report on Topic A: External Cost of Transport.* Petten: Netherlands Energy Research Foundation ECN.

Krupnick, Alan J. and Maurren L. Cropper. 1992. The Effect of Information on Health Risk Valuation. *Journal of Risk and Uncertainty* 5: 29-48.

Kullback, S. 1968. *Information Theory and Statistics.* New York: Dover Publications Inc.

Landefeld, J. Steven and Eugene P. Seskin. 1982. The Economic Value of Life: Linking Theory to Practice. *American Journal of Public Health* 72 (6): 555-566.

Lanoie, P. C. Petro and R. Latour. 1995. The Value of Statistical Life: A Comparison of Two Approaches. *Journal of Risk and Uncer*tainty 10: 235-257.

Lawless, J. F. 1987. Regression Methods for Poisson Process Data. *Journal of the American Statistical Association* 82 (399): 808-815.

Le Guen, J. 1999. *Reducing Risk, Protecting People*. London: Health & Safety Executive, Risk Assessment Policy Unit.

Lee, D. 1997. Uses and Meanings of Full Social Cost Estimates. In *The Full Costs and Benefits of Transportation: Contributions to Theory, Method and Measurement,* ed. Greene, D. L., Jones, R. W. and M. A. Delucchi, 113-149. Berlin, New York: Springer-Verlag.

Lee, Lung-Fei. 1986. Specification Test for Poisson Regression Models. *International Economic Review* 27 (3): 689-706.

Leigh, P. 1991. No Evidence of Compensating Wages for Occupational Fatalities. *Industrial Relations* 30 (3): 382-395.

Leigh, J. Paul. 1995. Compensating Wages, Value of a Statistical Life, and Inter-industry Differentials. *Journal of Environmental Economics and Management* 28: 83-97.

Liang, Kung-lee and Scott l. Zeger. 1986. Longitudinal Data Analysis Using Generalized Liner Models. *Biometrika* 73 (1): 13-22.

Linnerooth, Joanne. 1979. The Value of Human Life: A Review of the Models. *Economic Inquiry* 17: 52-74.

Linneroth, J. 1982. Murdering Statistical Lives . . . ? In *The Value of life and Safety: Proceedings of a Conference held by the Geneva Association,* ed. Jones-Lee, M. W, 229-261. Amsterdam: North Holland Publishing Company.

Linster, M. 1990. Background Facts and Figures to: Transport and the Environment. In *Transport and the Environment*, ed. The European Conference of Ministers of Transport (ECMT), 9-46. Paris: OECD.

Lipfert, F. W. 1980. Differential Mortality and the Environment: The Challenge of Multicollinerarity in Cross-Sectional Studies. *Energy Systems and Policy* 3 (4): 367-400.

Lipfert, F. W. 1984. Air Pollution and Mortality: Specification searches Using SMSA-Based Data. *Journal of Environmental Economics and Management* 11: 209-243.

Lipfert F. W. 1993. *Air Pollution and Community Health: A Critical Review and Data Sourcebook.* New York: Van Nostrand Reinhold.

Litai, D. 1980. *A Risk Comparison Methodology for the Assessment of Acceptable Risk.* PhD Thesis. Massachusetts Institute of Technology.

Litman, T. 1997. *Transportation Cost Analysis: Techniques, Estimates and Implications*. Victoria, BC: Victoria Transport Policy Institute.

Liu, Jin-Tan, James K. Hammit and Jin-Long Liu. 1997. Estimated Hedonic Wage Functions and Value of Life in a Developing Country. *Economic Letters* 57: 353-358.

Loomis, John, George Peterson, Patricia Champ, Thomas Brown and Beatrice Lucero. 1998. Paired Comparison Estimates of Willingness to Accept Versus Contingent Valuation Estimates of Willingness to Pay. *Journal of Economic Behavior and Organization* 35: 501-515.

Lynam, D. R. and G. D. Pfeifer. 1991. Human Health Effects on Highway-Related Pollutants. In *Highway Pollution*. Hamilton, ed. R. S. and R. M. Harrison, 256-281. Amsterdam, London: Elsevier.

MAAPE. 1995. *Health Effects of Exposure to Mixtures of Pollutants*. Fourth report prepared by the Department of Health's Advisory Group on Medical Aspects of Air Pollution Episodes. London: HMSO.

Mackenbach et al. 1993. Air Pollution, Lagged Effects of Temperature and Mortality: Netherlands, 1979-1987. *Journal of Epidemiology and Community Health* 47: 121-126.

Maclean, A. D. 1979. *The Value of Public Safety: Results of a Pilot-Scale Survey*. London: Home Office Scientific Advisory Branch.

Maddala, G. S. 1983. *Limited-Dependent and Qualitative Variables in Econometrics*. London: Cambridge University Press.

Maddison, D., D. Pearce, O. Johansson, E. Calthrop, T. Litman and E. Verhoef. 1996. *Blueprint 5: The True Costs of Road Transport.* London: Earthscan Publications Ltd.

Maibach, M., S. Mauch, R. Iten, S. Banfi, W. Ott, E. Ledergerber and K. P. Masuhr. 1996. *Die vergessenen Milliarden - externe Kosten im Energie- und Verkehrsbereich*. Bern, Stuttgart: Paul Haupt Verlag.

Maier, G., S. Gerking and P. Weiss. 1989. *The Economics of Traffic Accidents on Austrian Roads: Risk Lovers of Policy Deficit*? Mimeo, Wirtschaftsuniversität Wien.

Marshall, A. 1898. Principles of *Economics. Volume 1*. London: MacMillan & Co.

Mauch, S. F. and W. Rothengatter. 1995. *External Effects of Transport: Project for UIC Paris: Final Report*. Paris: International Union of Railways (UIC).

Mayeres, I. 1993. The Margibnal External Cost of Car Use - With an Application to Belgium. *Tijdschrift voor Economie en Managem*ent XXXVIII (3): 225-258.

McCullagh, P and J.A. Nelder. 1989. *Generalized Linear Models. 2nd Edition*. London, New York: Chapman and Hall Ltd.

McDaniels, Timothy, Mark S. Kamlet and Gregory W. Fischer. 1992. Risk Perception and the Value of Safety. *Risk Analysis* 12 (4): 495-503.

McFadden, D. and G. Leonard. 1992. Issues in the Contingent Valuation of Environmental Goods: Methodologies for Data Collection and Analysis. In *Contingent Valuation: A Critical Assessment*, ed. McFadden, D. and G. Leonard, 74-121. Cambridge: Cambridge Economics Inc.

McMichael, A. J. 1997. Transport and Health; Assessing the Risks. In *Health at the Crossroads - Transport Policy and Urban Health*, ed. Fletcher, T. and A. J. McMichael, 9-27. Chichester, New York: John Wiley & Sons Ltd..

Meade, J. E. 1952. External Economies and Diseconomies In a Competitive Situation. *The Economic Journal* 62 (March): 54-67.

Melinek, S. J., S. K. D. Wooley and R. Baldwin. 1973. *Analysis of a Questionnaire on Attitudes to Fire Risk.* Fire Research Note 962. Borehamwood: Joint Fire Research Organisation.

Miller, T. and J. Guria. 1991. *The Value of Statistical Life in New Zealand: Market Research on Road Safety*. Land Transport Division, Ministry of Transport, Wellington.

Miller, T. R. 1990. The Plausible Range for the Value of Life - Red Herrings Among the Mackerel. *Journal of Forensic Economics* 3 (3): 17-39.

Miller, T. R. 1997. Societal Costs of Transportation Crashes. In *The Full Costs and Benefits of Transportation: Contributions to Theory, Method and Measurement,* ed. Greene, D. L., Jones, R. W. and M. A. Delucchi, 281-315. Berlin, New York: Springer-Verlag.

Ministry of Health. 1954. *Mortality and Morbidity during the London Fog of December 1952*. London: HM Stationary Office.

Mishan, E. J. 1969. The Relationship between Joint Products, Collective Goods, and External Effects. *Journal of Political Economy* 69: 329-348.

Mishan, E. J. 1971a. *Cost-Benefit Analysis: An Informal Introduction.* London: George Allen and Unwin Ltd.

Mishan, E. J. 1971b. Evaluation of Life and Limb: A Theoretical Approach. *Journal of Political Economy* 79: 133-137.

Mishan, E. J. 1972. The Postwar Literature on Externalities. *Journal of Economic Literature 9*: 1-28.

Mishan, E. J. 1981. The Value of Trying to Value Life. *Journal of Public Economics* 15: 133-137.

Mishan, E. J. 1985. Consistency in the Valuation of Life: A Wild Goose Chase?. *Social Philosophy & Policy* 2 (2): 152-167.

Mitchell, C. G. B. and A. J. Hickman. 1990. Air Pollution and Noise from Road Vehicles. In *Transport and the Environment*, ed. The European Conference of Ministers of Transport (ECMT), 46-76. Paris: OECD.

Mitchell, R. C. and R. T. Carson. 1989. *Using Surveys to Value Public Goods*. Washington, DC: Resources for the Future.

Moore, M. J. and W. K. Viscusi. 1988. The Quantity-Adjusted Value of Life. *Economic Inquiry* 26: 369-388.

Moore, M. J. and K. K. Viscusi 1990. Models for Estimating Discount Rates for Long-Term Health Risks using Labor Market Data. *Journal of Risk and Uncertainty* 3: 381-401.

Office for National Statistics. 1998a. *Family Spending: A Report on the 1997-98 Family Expenditure Survey.* London: The Government Statistical Service. The Stationary Office.

Office for National Statistics. 1998b. *Regional Trends 33 - 1998 Edition.* London: The Government Statistical Service. The Stationary Office.

Office for National Statistics. 1999a. *Economic Trends: Annual Supplement 1999.* London: The Government Statistical Service. The Stationary Office.

Office for National Statistics. 1999b. *Social Trends 29; 1999 Edition.* London: The Government Statistical Service. The Stationary Office.

Organization for Economic Cooperation and Development. 1993. *OECD Environmental Data 1993: Compendium 1993.* Paris: OECD.

Organization for Economic Cooperation and Development. 1995a. *OECD Environmental Data: Compendium 1995.* Paris: OECD.

Organization for Economic Cooperation and Development. 1995b. *Motor Vehicle Pollution: Reduction Strategies beyond 2010.* Paris, France: OECD Publication Service.

Ospelt, Walter. 1994. *Road Vehicles - Efficiency and Emissions.* at website: http://www.bmu.gv.at/embu_new/conf1/ospelt.htm.

Ostro, Bart and Lauraine Chestnut. 1998. Assessing the Health Benefits of Reducing Particulate Matter Air Pollution in the United States. *Environmental Research, Section A* 76: 94-106.

Ostro, Bart. 1994. *Estimating the Health Effects of Aiir Pollution: A Method with an Application to Jakarta*. Policy Research Working Paper. New York: The World Bank.

Özkaynak, Halûk and George D. Thurston. 1987. Association between 1980 U. S. Mortality Rates and Alternative Measures of Airborne Particle Concentration. *Risk Analysis* 7 (4): 449-461.

Papandreou, A. A. 1994. *Externality and Institutions*. Oxford: Clarendon Press.

Parijs, Philippe Van. 1992. Ethical Aspects of the Economic Value of Human Life: A Symposium. *Recherches Economiques de Louvain* 58 (2): 121-124.

Pearce, David and Tom Crowards. 1996. Particulate Matter and Human Health in the United Kingdom. *Energy Policy* 24 (7): 609-619.

Persson, U. and M. Cedervall. 1991. *The Value of Risk Reduction: Results of a Swedish Sample Survey.* IHE Working Paper 6-1991. Lund: The Swedish Institute of Health Economics.

Persson, U., A. Lugner Norinder and M. Svensson. 1995. Valuing the Benefits of Reducing the Risk of Non-Fatal Road Injuries: The Swedish Experience. In *Contingent Valuation, Transport Safety and the Value of Life*, ed. Schwab-Christe, N. G. and N. C. Soguel, 63-84. Boston: Kluwer.

Peters, A and A. Doring. 1997. Increased Plasma Viscosity during an Air Pollution Episode: A Link to Mortality? *Lancet* 349: 1582-1588.

Petty, W. 1699. *Political Arithmetick, or a Discourse Concerning the Extent and Value of Lands, People, Buildings, Etc.* London: Robert Caluel.

Plagiannakos, T. and J. Parker. 1988. *An Assessment of Air Pollution Effects on Human Health in Ontario.* Report No. 706.01 (#260), Energy Economics Section, Economics and Forecast Division, Ontario Hydro, Toronto.

Pigou, A. C. 1922. Empty Economic Boxes: A Reply. *Economic Journal*, 34 Dec), 485-465.

Pigou, A. C. 1932. *The Economics of Welfare*. 4th Edition. London: Macmillan.

Pindyck R. S. and D. L Rubinfeld. 1989. *Microeconomics*. New York: Macmillan.

Pindyck, .R. S. and D. L. Rubinfeld. 1991. *Econometric Models & Economic Forecasts 3rd edition*. New York: McGraw Hill Inc.

Poloniecki, Jan D., Richard W. Aitkinson, Antonio Ponce de Leon and H. Ross Anderson. 1997. Daily Time Series for Cardiovascular Hospital Admissions and Previous Day's Air Pollution in London, UK. *Occupational and Environmental Medicine* 54: 535-540.

Ponce de Leon, A, H.R. Andreson, J.M. Bland, D.P. Strachan and J. Bower. 1996. Effects of Air pollution on Daily Hospital Admissions for Respiratory Disease in London between 1987-88 and 1991-92. *Journal of Epidemiology & Community Health* 50 (Suppl. 1): S63-S70.

Pope, C. A., J. Schwartz and M. R. Ransom. 1992. Daily Mortality and PM_{10} Pollution in Utah Valley. *Archives of Environmental Health* 47: 211-217.

Pope, C. A., D. W. Dockery and J. Schwartz. 1995. Review of Epidemiological Evidence of Health Effects of Particulate Air Pollution. Inhalation Toxicology 7 (1): 1-18.

Pope, C. A. and L. S. Kalkstein. 1996. Synoptic Weather Modelling and Estimates of the Exposure-Response Relationship between Daily Mortality and Particulate Air Pollution. *Environmental Health Perspectives* 104: 414-420.

Portney, P. R. 1981. Housing Prices, Health effects and Valuing Reductions in Risk of Death. *Journal of Environmental Economics and Management* 8: 72-78.

Portney, P. R. 1994. The Contingent Valuation debate: Why Economists Should Care. *The Journal of Economic Perspectives* 8 (4): 3-18.

Prentice, Ross L. and Mark W. Mason. 1986. On the Application of Linear Relative Risk Regression Models. *Biometrics* 42: 109-130.

Prescott, G. J., G. R. Cohen, R. A. Elton, F. G. R. Fowkes and R. M Agius. 1998. Urban Air Pollution and Cardiopulmonary Ill Health: A 14.5 Years Time Series Study. *Occupational and Environmental Medicine* 55: 697-704.

Propsero, J.M., R.J. Charlston, V. Mohnen, R. Janicke, A.C. Delany, J. Moyers, W. Zoller and K Kahn. 1983. The Atmospheric Aerosol System: an Overview. *Review of Geophysiscs and Space Physics* 21: 1607-1629.

Quality of Urban Air Review Group (QUARG). 1993. *Diesel Vehicle Emissions and Urban Air Quality. 2nd Report*. London: Department of Environment.

Quality of Urban Air Review Group (QUARG). 1996. *Airborne Particulate Matter in the United Kingdom. 3rd Report*. London: Department of Environment.

Quantitative Micro Software 1998. *EViews User's Guide*. Irvine, CA: Quantitative Micro Software.

Quinet, E. 1997. Full Social cost of Transportation in Europe. In *The Full Costs and Benefits of Transportation: Contributions to Theory, Method and Measurement,* ed. Greene, D. L., Jones, R. W. and M. A. Delucchi, 69-113. Berlin, New York: Springer-Verlag.

Randall, A. 1986. The Possibility of Satisfactory Benefit Estimation with Contingent Markets. In *Valuing Public Goods: A State of the Arts Assessment of the Contingent Valuation Method, ed.* Cummings, R. G., D. S. Brookshire and W. D. Schulze. Totowa, NJ: Rowman and Allanheld.

Rice, D. and B. Cooper. 1967. The Economic Value of Human Life. *American Journal of Public Health* LVII: 1954-1986.

Rietveld, P. 1989. Infrastructure and Regional Development: A Survey of Multiregional Economic Models. *The Annals of Regional Science* 23: 255-274.

Robertson, D. H. 1924. Those Empty Boxes. *The Economic Journal* 36 (March): 16-31.

Robinson, J. 1941. The Rising Supply Price. *Economica* 8: 1-8.

Römer, A. U., W. W. Pommerehne and L. P. Feld. 1998. Revealing Prefernces for Reductions of Public Risks: An Application of the CV Approach. *Journal of Environmental Planning and Management* 41 (4): 477-503.

Rosen, H.S. 1995. *Public Finance.* 4th Edition. Irwin: Chicago.

Rothengatter, W. 1993. Externalities of Transport. In *European Transport Economics*, ed. Polak, J. and A. Heertje, 81-129. Oxford: Blackwell Publishers.

Rothengatter, W. 1994. Do External Benefits Compensate for External Costs of Transport? *Transport Research A* 28A (4): 321-328.

Rowe et al., 1995. *The New York Electricity Externality.* Study Volume 1. Oceana Publications.

Royal Commission on Environmental Pollution. 1994. *Eighteenth Report: Transport and the Environment.* London: HMSO.

Royal Commission on Environmental Pollution. 1996. *Twentieth Report: Transport and the Environment - Developments since 1994.* London: HMSO.

Royal Society (1992). *Risk: Analysis, Perception and Management.* London: The Royal Society.

Saldiva, P. H. N., C. A. Pope, J. Schwartz, D. W. Dockery, A. J. Lichtenfels, J. M. Salge, I. Barone and G. M. Bohm. 1995. Air Pollution and Mortality in Elderly People: A Time Series Study in Sao Paulo, Brazil. *Archives of Environmental Health* 50: 159-163.

Samet, J., S. Zeger, J. Kelsall, J. Xu and L. Kalkstein. 1998. Does Weather Confound or Modify the Association of Particulate Air Pollution with Mortality? *Environmental Research, Section A* 77: 9-19.

Samuels, W. J. 1972. Welfare Economics, Power and Property. In *Perspectives of Property*, ed. Wunderlich, G. and W. L. Gibson. Pennsylvania State University.

Savage, Ian. 1993. An Empirical Investigation into the Effect of Psychological Perceptions on the Willingness to Pay to Reduce Risk. *Journal of Risk and Uncertainty* 6: 75-90.

Schelling, T. C. 1968. The Life You Safe May be Your Own. *In Problems in Public Expenditure Analysis*. ed. Chase, S. B. 127-176. Washington, D.C.: The Brookings Institution.

Schkade, David A. and John W. Payne. 1994. How People Respond to contingent Valuation Questions: A Verbal Protocol Analysis of Willingness to Pay for an Environmental Regulation. *Journal of Environmental Economics and Management* 26: 66-109.

Schouten, J. P; Vonk, J. M. and A. de Graaf. 1996. Short Term Effects of Air Pollution on Emergency Hospital Admissions for Respiratory Disease: Results of the APHEA Project in Two Major Cities in The Netherlands, 1977-1989. *Journal of Epidemiology & Community Health* 50 (Suppl. 1): S22-S29.

Schuman H. and S. Presser. 1981. *Questions and Answers in Attitude Surveys*. New York: Academic Press.

Schwab-Christe, N. G. 1995. The Valuation of Human Costs by the Contingent Valuation Method: the Swiss Experience. *In Contingent Valuation, Transport and the Value of Life*, ed. Schwab-Christe, N. G. and N. C. Soguel, 19-44. Boston: Kluwer.

Schwab-Christe, N. G. and N. C. Soguel. 1996. The Pain of Road-Accident Victims and the Bereavement of their Relatives: A Contingent-Valuation Experiment. *Journal of Risk and Uncertainty* 13: 277-291.

Schwartz, J. 1991. Particulate Air Pollution and Daily Mortality in Detroit. *Environmental Research* 56: 204-213.

Schwartz, J. 1991/92. Particulate Air Pollution and Daily Mortality: A Synthesis. *Public Health Review* 19: 39-60.

Schwartz, J. 1993. Air Pollution and Daily Mortality in Birmingham, Alabama. *American Journal of Epidemiology* 137 (10): 1136-1147.

Schwartz, J. 1994a. Particulate Air Pollution and Daily Mortality in Cincinnati, Ohio. *Environmental Health Perspectives* 102: 186-189.

Schwartz, J. 1994b. Nonparametric Smoothing in the Analysis of Air Pollution and Respiratory Illness. *The Canadian Journal of Statistics*. 22 (4): 471-487.

Schwartz, J. 1994c. Air Pollution and Daily Mortality: A Review and Meta Analysis. *Environmental Research* 64: 36-52.

Schwartz, J. and A. Marcus. 1990. Mortality and Air Pollution in London: A Time Series Analysis. *American Journal of Epidemiology* 131 (1): 185-194.

Schwartz, J., C. Spix, H. E. Wichmann and E. Malin. 1991. Air Pollution and Acute Respiratory Illness in Five German Communities. *Environmental Research* 56: 1-14.

Schwartz, J. and D. W. Dockery. 1992a. Increased Mortality in Philadelphia Associated with Daily Air Pollution Concentrations. *American Review of Respiratory Disease* 145: 600-604.

Schwartz, J. and D. W. Dockery. 1992b. Particulate Air Pollution and Daily Mortality in Steubenville, Ohio. *American Journal of Epidemiology* 136 (1): 12-19.

Schwartz, J., D. Slater, T. Larson, W. E. Pierson and J. Q. Koenig. 1993. Particulate Air Pollution and Hospital Emergency Visits for Asthma in Seattle. *American Review of Respiratory Disease* 147: 826-831.

Schwartz J. C. Spix, G. Touloumi, L. Bacharova, T. Barumamdzadeh, A. Le Tetre, T. Piekarksi, A. Ponce de Leon, A. Ponka, G. Rossi, M. Saez and J.P. Schouten. 1996. Methodological Issues in Studies of Air Pollution and Daily Counts of Deaths or Hospital Admissions. *Journal of Epidemiology and Community Health* 50 (Suppl. 1): S3-S11.

Scitovsky, Tibor. 1954. Two Concepts of External Economies. *Journal of Political Economy* 62: 143-151.

Seaton, Anthony, William MacNee, Kenneth Donaldson and David Godden. 1995. Particulate Air Pollution and Acute Health Effects. *The Lancet* 345: 176-178.

Sexton, Ken, Henry Gong Jr., John C. Bailar, III, Jean G. Ford, Diane R. Gold, William E. Lambert and Mark J. Utell. 1993. Air Pollution Health Risks: Do Class and Race Matter? *Toxicology and Industrial Health* 9 (5): 843-878.

Shepard, D. and R. Zeckhauser. 1982. Life-Cycle Consumption and Willingness to Pay for Increased Survival. In *The Value of Life and Safety: Proceedings of a Conference held by the Geneva Association*, ed. Jones-Lee, M. W. Amsterdam: North Holland Publishing Company.

Shepard, D. and R. Zeckhauser. 1984. Survival versus Consumption. *Management Science* 30 (4): 423-439.

Shumway, R.H., R. Tai, L. Tai and Y. Pawitan. 1983. Statistical Analysis of Daily London Mortality and Associated Weather and Pollution Effects. *Technical Report 53*, Division of Statistics, University of California, Davis.

Shumway, R.H., A. S Azari and Y. Pawitan. 1988. Modelling Mortality Fluctuations in Los Angeles as Functions of Pollution and Weather effects. *Enviornmental Research* 45: 224-242.

Small, Kenneth A. and Camilla Kazimi. 1995. On the True Costs of Air Pollution from Motor Vehicles. *Journal of Transport Economics and Policy* 29: 7-32.

Smith, V. Kerry and Ju-Chin Huang. 1995. Can Markets Value Air Quality? A Meta-Analysis of Hedonic Property Value Models. *Journal of Political Economy* 103 (1): 2 09-227.

Soby, B. A. and D. J. Ball. 1991. Consumer Safety and the Valuation of Life and Safety. Environmental Risk Assessment Unit, School of Environmental Science. Norwich: University of East Anglia.

Soguel, N. C. 1995. Contingent Valuation, Transport Safety and the Value of Life. In *Contingent Valuation, Transport and the Value of Life*, ed. Schwab-Christe, N. G. and N. C. Soguel, 1-19. Boston: Kluwer.

Sommer, H and R. Neuschwander. 1996. Monetarisiereung der Verkehrsbedingten Externen Gesundheitskosten. GVF-Report No. 272. Altdorf, Berne: ECOPLAN.

Spengler, J. and R. Wilson. 1996. Emissions, Dispersion, and Concentration of Particles. In *Particles in Our Air: Concentrations and Health Effects*, ed. Wilson, J. and J. Spengler, 41-63. Cambridge, MA: Harvard School of Public Health.

Spix, C., J. Heinrich, D. Dockery, J. Schwartz, G. Volksch, K. Schwinkowski and C. Collen. 1993. Air Pollution and Daily Mortality in Erfurt (East Germany) from 198901989. *Environmental Health Perspectives* 101: 518-526,

Spix, C., J. Heinrich, D. Dockery, J. Schwartz, G. Volksch, K. Scwinkowski, C. Collen and H. E. Wichmann. 1994. Summary of the Analysis and Reanalysis Corresponding to the Study on Air Pollution and Daily Mortality in Erfurt, E. Germany 1980-1989. *Critical Evaluation Workshop on Particulate Matter: Mortality Epidemiology Studies*: November, Raleigh, NC, USA.

Spix, C and H. E. Wichmann. 1996. Daily Air Mortality and Air Pollutants: Findings from Köln, Germany. *Journal of Epidemiology & Community Health* 50 (Suppl. 1): S52-S58.

Staller, J. M., B. P. Sullivan and E. A. Friedman. 1994. Value of Life Estimates - Too Imprecise for Courtroom Use: A Note. *Journal of Forensic Economics* 7 (2): 215-219.

Stedman, J. R., E. Linehan, S. Espenhahn, T. Bush and T. Davies. 1998. *Predicting PM_{10} Concentrations in the UK*. Abingdon: AEA Technology.

Stern, F. et al. 1988. Heart Disease Mortality among Bridge and Tunnel Officers exposed to Carbon Monoxide. *American Journal of Epidemiology* 128: 1276-1288.

Stewart, J. and L. Gill. 1998. *Econometrics*. 2nd Edition. London: Prentive Hall Europe.

Sunyer, J., J. Castellsague, M. Saez, A. Tobias and J.M. Anto. 1996. Air Pollution and Mortality in Barcelona. *Journal of Epidemiology & Community Health* 50 (Suppl. 1): S76-S80.

Szklo, G. 1987. Design and Conduct of Epidemiologic Studies. *Preventive Medicine* 16: 142-149.

TEST. 1991. *Wrong Side of the Tracks? Impacts of Road and Rail Transport on the Environment: A Basis for Discussion. Test Report No. 100.* London: TEST

Thaler, R. 1982. Precommitment and the Value of Life. In *The Value of life and Safety: Proceedings of a Conference held by the Geneva Association*, ed. Jones-Lee, M. W. 171-183. Amsterdam: North Holland Publishing Company.

Tinch, R. 1995. *The Valuation of Environmental Externalities.* Full Report prepared for the Department of Transport. London: HMSO.

Tolley, G. S. and R. G. Fabian. 1998. Issues in Improvement of the Valuation of Non-Market Goods. *Resource and Energy Economics* 20: 75-83.

Toulomi, G., S. J. Pocock, K. Karsouyanni and D. Trichopolous. 1994. Short-Term Effects of Air Pollution on Daily Mortality in Athens: A Time-Series Analysis. *International Journal of Epidemiology* 23: 957-967.

Touloumi, G., Samoli, E. and K. Katsouyanni. 1996. Daily mortality and "Winter Type" Air Pollution in Athens, Greece - A Time Series Analysis within the APHEA Project. *Journal of Epidemiology & Community Health* 50 (Suppl. 1): S47-S51.

Touloumi, G., K. Katsouyanni, D. Zmirou, J. Schwartz, C. Spix, A. Ponce de Leon, A. Tobias, P. Quennel, D. Rabczenko, L. Bacharova, L. Bisanti, J. M. Vonk and A. Ponka. 1997. Short-term Effects of Ambient Oxidant Exposure on Mortality: a Combined Analysis within the APHEA Project. *American Journal of Epidemiology* 146 (2): 177-185.

Trivedi, Pravin K. 1997. Econometric Models of Event Counts. *Journal of Applied Econometrics* 12: 199-201.

Tukey, J. W. 1962. The Future of Data Analysis. *Annals of Mathematics and Statistics* 33: 1-67.

Turvey, R. 1963. On Divergences between Social Cost and Private Cost. *Economica* (August): 309-313.

U.S. Department of Transportation. 1997. *National Transportation Statistics 1996*. Washington, DC: Bureau of Transportation Statistics.

Ulph, Alistair. 1982. The Role of Ex Ante and Ex Post Decisions in the Valuation of Life. *Journal of Public Economics* 18: 265-276.

Usher, Dan. 1985. The Value of Life for Decision Making in the Public Sector. *Social Philosophy & Policy* 2 (2): 168-191.

Van Houtven, George and Maureen L. Cropper. 1996. When is a Life Too Costly to Save? The Evidence from U.S. Environmental Regulations. *Journal of Environmental Economics and Management* 30: 348-368.

Vatn, Arild and Daniel W. Bromley. 1997. Externalities – A Market Model Failure. *Environmental and Resource Economies* 9: 135-151.

Verhoef, E. 1994. External Effects and Social Costs of Road Transport. *Transport Research A* 28A (4): 273-287.

Verhoef, E. 1996. The External Costs of Road Transport in The Netherlands. In *The True Costs of Road Transport: Blueprint 5*, ed. Maddison, D., D. Pearce, O. Johansson, E. Calthrop, T. Litman and E. Verhoef, 203. London: Earthscan Publications Ltd.

Verhoef, A. P., G. Hoek, J. Schwartz and J. H. van Wijen. 1996. Air Pollution and Daily Mortality in Amsterdam, The Netherlands. *Epidemiology* 7: 225-230.

Vigotti, M.A., G. Rossi, L. Bisanti, A. Zanobetti and J. Schwartz. Short term Effects of Urban Air Pollution on Respiratory Health in Milan, Italy, 1980-1989. *Journal of Epidemiology & Community Health* 50 (Suppl. 1): S71-S75.

Viner, Jacob. 1931. Cost Curves and Supply Curves. *Zeitschrift für Nationalökonomie* 3: 23-46.

Viscusi, K. W. 1986. The Valuation of Risks to Life and Health: Guidelines for Policy Analysis. *In Benefits Assessment: The State of the Art*, ed. Bentkover, J. D., V. T. Covello and J. Mumpower. Dordrecht: D. Reidel Publishing company.

Viscusi, K.W. 1990. The Econometric Basis for Estimates of the Value of Life. *Journal of Forensic Economics* 3 (3): 61-70.

Viscusi, K. W. 1992. *Fatal Tradeoffs*. Oxford: Oxford University Press.

Viscusi, K. W. 1993. The Value of Risks to Life and Health. *Journal of Economic Literature* 31: 1912-1946.

Viscusi, K. W. 1994. Risk-Risk Analysis. *Journal of Risk and Uncertainty* 8: 5-17.

Viscusi, K. W. 1998. *Rational Risk Policy*. Oxford: Clarendon Press.

Viscusi, K. W., W. A Magat and A. Forrest. 1988. Altruistic and Private Valuations of Risk reduction. *Journal of Policy Analysis and Management* 7 (2): 227-245.

Viscusi, K. W. and Michael J. Moore. 1989. Rates of Time Preference and Valuations of the Duration of Life. *Journal of Public Economics* 38: 297-317.

Viscussi, W. K, W. A. Magat and J. Huber. 1991. Pricing Environmental Health Risks: Survey Assessments of Risk-Risk and Risk-Dollar Trade-Offs for Chronic Bronchitis. *Journal of Environmental Economics and Mana*gement 21: 35-51.

Viscusi, K. W., J. K. Hakes and A. Carlin. 1997. Measures of Mortality Risk. *Journal of Risk and Uncertainty* 14: 213-233.

Walters, Sarah. 1994. What are the Respiratory Health Effects of Vehicle Pollution? In *How Vehicle Pollution Affects Our Health*, ed. Read Cathy, 9-12. London: The Ashden Trust.

Watkins, L. H. 1991. *Air Pollution from Road Vehicles*. Transport and Road Research Laboratory, Department of Transport. London: HMSO.

Watsham, Terry J. and Keith Parramore 1997. *Quantitative Methods in Finance*. London: International Thomson Business Press.

Wedel, M., W. S. Desarbo, J. R. Bult and V. Ramaswamy. 1993. A Latent Class Poisson Regression Model for Heterogeneous Count Data. *Journal of Applied Econometrics* 8: 397-411.

Weinstein, Milton C., Donald S. Shepard and Joseph S. Pliskin. 1980. The Economic Value of Changing Mortality Probabilities: A Decision-Theoretic Approach. *The Quarterly Journal of Economics* 94: 373-396.

Whitcomb, David K. 1972. *Externalities and Welfare*. New York, London: Columbia University Press.

Whitelegg, J., A. Gatrell and P. Naumann. 1993. *Traffic and Health: A Report for Greenpeace Environmental Trust*. University of Lancaster: Environmental Epidemiology Research Unit.

Willeke, R. 1991. Soziale Kosten und Nutzen des Kraftfahrzeugsverkehrs. *Schriftenreihe der DVWG B* 138: 49-60.

Willeke, R. 1992. *Benefits of Different Transport Modes*. Round Table 92. European Conference of Ministers of Transport. Paris: OECD.

Willeke, R. 1996. *Mobilität, Verkehrsmarktordnung, Externe Kosten und Nutzen des Verkehrs*. Frankfurt: Verband der Automobilindustrie e. V.

Williams, M. L. 1997. Current and Future Legislation United Kingdom and Europe. In *Air Pollution in the United Kingdom*, ed. Davidson, G. and C. N. Hewitt, 85-95. Cambridge: The Royal Society of Chemistry.

Williamson, J. G. 1984. British Mortality and the Value of Life, 1781-1931. Population Studies 38: 157-172.

Winkelmann, Rainer. 1995. Recent Developments in Count Data Modelling: Theory and Application. *Journal of Economics Surveys* 9 (1): 1-24.

Winkelmann, Rainer. 1997. *Econometric Analysis of Count Data*. Berlin, Heidelberg: Springer-Verlag.

Winkelmann, R. and K.F. Zimmermann. 1992. *Recent Developments in Coun tData Modelling: Theory and Applications*. Münchner Wirtschaftswissenschaftliche Beiträge 92-05.

Winkelmann, R. and K. F. Zimmermann. 1995. Recent Developments in Count Data Modelling: Theory and Application. *Journal of Economic Surveys* 9: 1-24.

Wittenberg, Beatrice A. and Jonathan B. Wittenberg. 1997. *Effects of Carbon Monoxide on Heart Muscle Cells*. Research Report Number 62. The Health Effects Institute.

Worcester, Dean A. Jr. 1969. Pecuniary and Technological Externality, Factor Rents, and Social Costs. *The American Economic Review* 59: 873-885.

Wordley, J, Walters, S,. and J.G. Ayres. 1997. Short Term Variations in Hospital Admissions and Mortality and Particulate Air Pollution. *Occupational and Environmental Medicine* 54: 108-116.

World Health Organization. 1999. *Health Costs Due to Road Traffic-related Air Pollution. An Impact Assessment Project of Austria, France, and Switzerland. Synthesis Report prepared for the WHO Ministerial Conference on Environment and Health.* London: WHO.

Wyzga, Ronald E. 1978. The Effect of Air Pollution upon Mortality: A Consideration of Distributed Lag Models. *Journal of the American Statistical Association* 73 (363): 463-472.

Young, A. A. 1917. Pigou's Wealth and Welfare. *Quarterly Journal of Economics* 27: 672-686.

Zeger, L. Scott, Kung-Yee Liang. 1986. Longitudinal Data Analysis for discrete and Continuous Outcomes. *Biometrics* 42: 121-130.

Zerbe, R. O. 1976. The Problem of Social Cost: Fifteen Years Later. In *Theory and Measurement of Economic Externalities*. ed. Lin, S. A. Y. New York: Academic Press.

Zmirou, D, T. Barumandzadeh, F. Balducci, P. Ritter, G. Laham and J-P. Ghilardi. 1996. Short term effects of air pollution on mortality in the city of Lyon, France, 1985-1990. *Journal of Epidemiology & Community Health* 50 (Suppl. 1): S30-S35.

Zmirou, D., J. Schwartz, M. Saez, A. Zanobetti, B. Wojtyniak, G. Touloumi, C. Spix, A. de Leon, Y. Le Moullec, L. Bacharova, J. Schouten, A. Ponka and K. Katsouyanni. 1998. Time-Series Analysis of Air Pollution and Cause-Specific Mortality. *Epidemiology* 9 (5): 495-503.

Zweifel, P. and F. Breyer. 1997. *Health Economics*. New York: Oxford University Press Inc.

Referenced Internet Websites

The Department of the Environment, Transport and the Regions.
http://www.detr.gov.uk.

AEA Technology plc. The National Environmmental Technolgy Centre (NETCEN).
http://www.aeat.co.uk

NSCA.
http://www3.mistral.co.uk

Umweltbundesamt Germany.
http://www.umweltbundesamt.de

The European Environment Agency.
http://www.eea.dk

The Manchester Metropolitan University: Atmospheric Research & Information Centre.
http://www.doc.mmu.ac.uk

City of Westminster.
http://www.wcceh.gov.uk

The London Research Centre.
http://www.london-research.gov.uk

Rheinisch-westfälische Technische Hochschule Aachen.
http://www.imib.rwth-aachen.de

Umweltbundesamt Austria.
http://www.bmu.gv.at

The University of Teeside.
http://www.scm.tees.ac.uk

The Warwick District Council.
http://www.warwickdceh.demon.co.uk

British Columbia Ministry of Environment, Lands & Parks.
http://www.env.gov.be.ca

The University of Edinburgh (The Faculty of Medicine).
http://www.med.ed.ac.uk

South East Institute of Public Health
http://www.seiph.ac.uk

The Health Effects Institute (HEI).
http://www.healtheffects.org

U.S. Environmental Protection Agency.
http://www.epa.gov

Environment Agency U.K.
http://www.environmnet-agency.gov.uk

California Air Resources Board. California Environmental Protection Agency.
http://arbis.arb.ca.gov

American Lung Association.
http://www.lungusa.org

World Resources Institute.
http://www.igc.apc.org

AIRBASE run by the European Environment Agency and European Commission Directorate General XI-33.
http:// etcaq.rivm.nl.

Appendix 1

As described in Chapter 9, the individual air pollutants examined in this study are assumed to be highly correlated. The following tables illustrate the correlation matrices for the five pollutants per city. The figures presented below prove that these pollutants do correlate.

- **Birmingham**

Table 161: Correlation Matrix for Air Pollutants in Birmingham.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.73	0.65	0.55	-0.25
NO_2		1	0.66	0.65	-0.48
SO_2			1	0.54	-0.41
CO				1	-0.42
Ozone					1

- **Bristol**

Table 162: Correlation Matrix for Air Pollutants in Bristol.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.72	0.45	0.50	-0.36
NO_2		1	0.41	0.61	-0.58
SO_2			1	0.26	-0.29
CO				1	-0.53
Ozone					1

➢ Cardiff

Table 163: Correlation Matrix for Air Pollutants in Cardiff.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.48	0.54	0.37	-0.02
NO_2		1	0.59	0.56	-0.49
SO_2			1	0.47	-0.29
CO				1	-0.44
Ozone					1

➢ Edinburgh

Table 164: Correlation Matrix for Air Pollutants in Edinburgh.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.53	0.44	0.41	-0.18
NO_2		1	0.50	0.57	-0.57
SO_2			1	0.54	-0.44
CO				1	-0.53
Ozone					1

➢ Glasgow

Table 165: Correlation Matrix for Air Pollutants in Glasgow.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.58	0.50	0.47	-0.21
NO_2		1	0.44	0.77	-0.57
SO_2			1	0.38	-0.22
CO				1	-0.56
Ozone					1

➢ Kingston upon Hull

Table 166: Correlation Matrix for Air Pollutants in Kingston upon Hull.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.47	0.40	0.39	-0.13
NO_2		1	0.46	0.59	-0.55
SO_2			1	0.25	-0.27
CO				1	-0.45
Ozone					1

➢ Leeds

Table 167: Correlation Matrix for Air Pollutants in Leeds.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.54	0.49	0.52	-0.25
NO_2		1	0.50	0.62	-0.53
SO_2			1	0.42	-0.37
CO				1	-0.47
Ozone					1

➢ Leicester

Table 168: Correlation Matrix for Air Pollutants in Leicester.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.57	0.43	0.45	-0.12
NO_2		1	0.58	0.65	-0.56
SO_2			1	0.42	-0.35
CO				1	-0.43
Ozone					1

➢ Liverpool

Table 169: Correlation Matrix for Air Pollutants in Liverpool.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.74	0.67	0.46	-0.40
NO_2		1	0.68	0.53	-0.63
SO_2			1	0.44	-0.54
CO				1	-0.50
Ozone					1

➢ London

Table 170: Correlation Matrix for Air Pollutants in London.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.72	0.64	0.55	-0.11
NO_2		1	0.70	0.71	-0.23
SO_2			1	0.58	-0.31
CO				1	-0.50
Ozone					1

➢ Newcastle upon Tyne

Table 171: Correlation Matrix for Air Pollutants in Newcastle upon Tyne.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.64	0.43	0.49	-0.22
NO_2		1	0.55	0.70	-0.55
SO_2			1	0.40	-0.37
CO				1	-0.51
Ozone					1

➢ Sheffield

Table 172: Correlation Matrix for Air Pollutants in Sheffield.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.51	0.58	0.55	-0.36
NO_2		1	0.31	0.66	-0.36
SO_2			1	0.40	-0.47
CO				1	-0.59
Ozone					1

➢ Southampton

Table 173: Correlation Matrix for Air Pollutants in Southampton.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.70	0.48	0.52	-0.24
NO_2		1	0.56	0.64	-0.46
SO_2			1	0.45	-0.32
CO				1	-0.52
Ozone					1

➢ Swansea

Table 174: Correlation Matrix for Air Pollutants in Swansea.

	PM_{10}	NO_2	SO_2	CO	Ozone
PM_{10}	1	0.43	0.61	0.29	0.04
NO_2		1	0.54	0.71	-0.53
SO_2			1	0.52	-0.34
CO				1	-0.60
Ozone					1

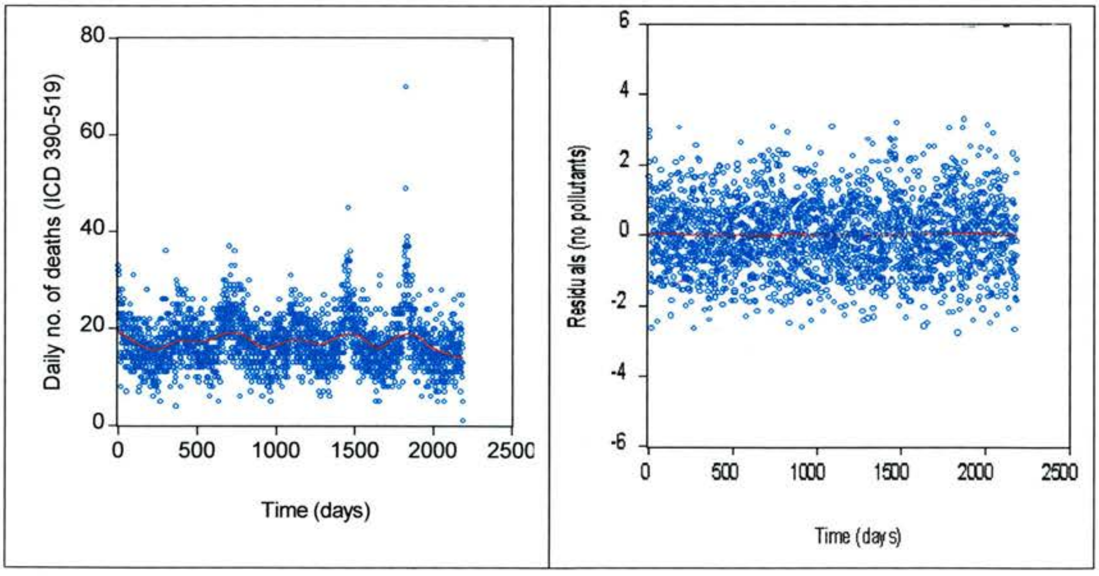
Appendix 2

As described in detail in Chapter 9, a specific procedure had been applied in order to control for the possibility of confounding that may result from the shared cycles of the dependent variable (daily death counts) and the independent variables, such as temperature, humidity, season, and epidemic periods. In doing so, a core model was composed and fitted to the data.

After testing the residuals of the core model for goodness of fit, the air pollution data was finally also included. The following figures illustrate the original unfiltered data, as well as the residuals of the core model where all variables have been included except the air pollution data for all 14 British cities analysed in this study. The coloured lines indicate fitted non-parametric LOESS functions as described in detail in Chapter 8 of this study.

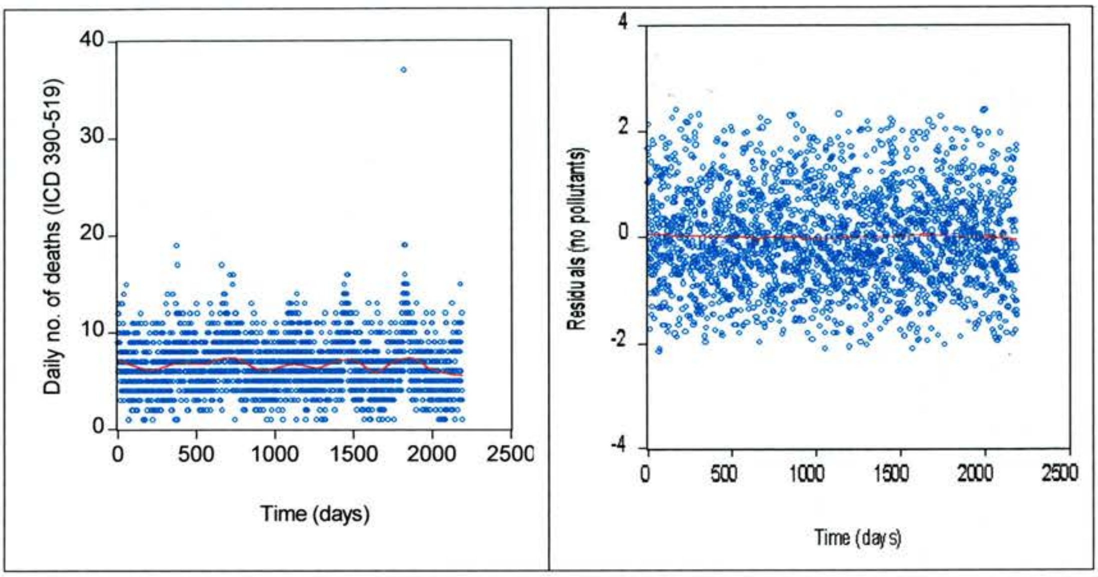
- **Birmingham**

Table 175: Unfiltered Daily Death Counts & Residuals of Core Model for Birmingham.



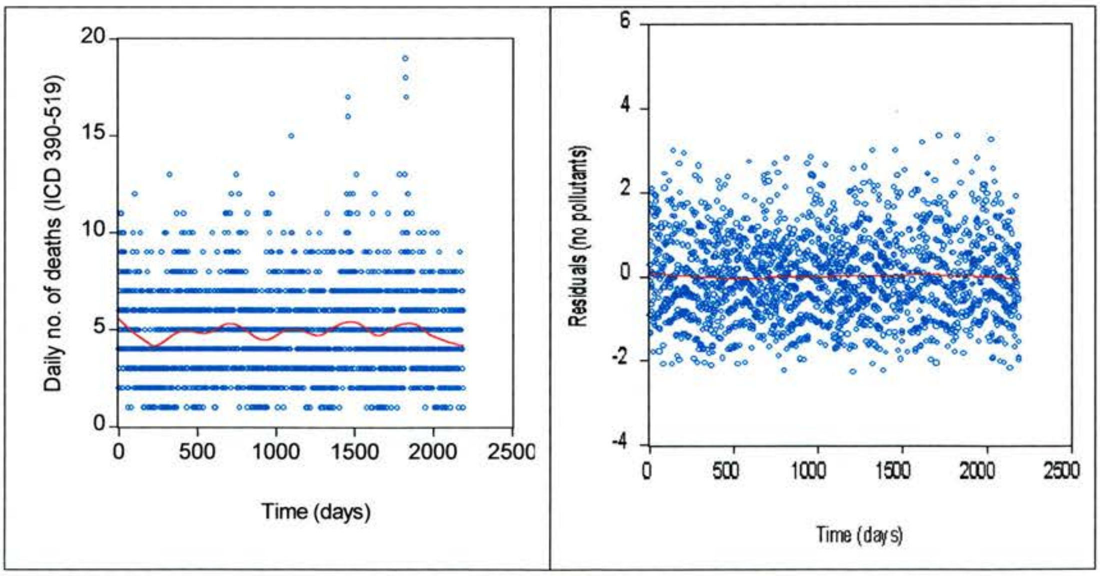
➢ Bristol

Table 176: Unfiltered Daily Death Counts & Residuals of Core Model for Bristol.



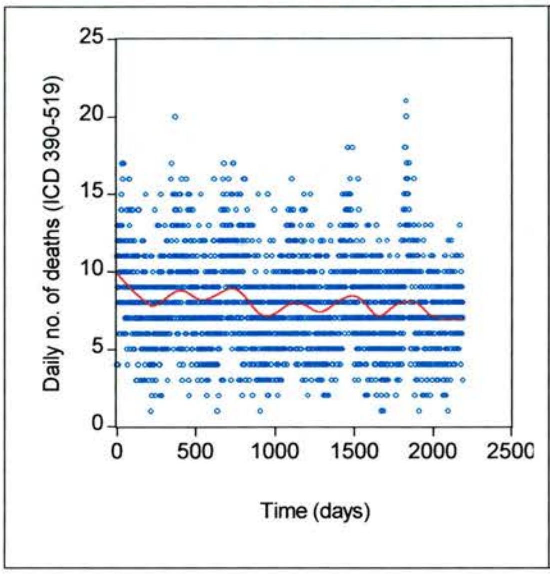
➢ Cardiff

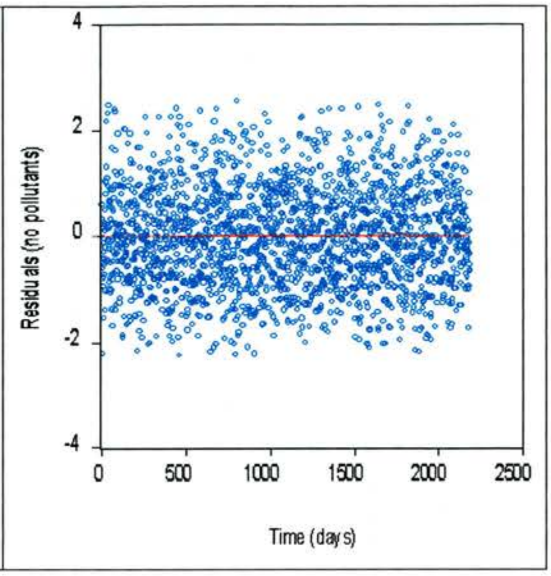
Table 177: Unfiltered Daily Death Counts & Residuals of Core Model for Cardiff.



➢ Edinburgh

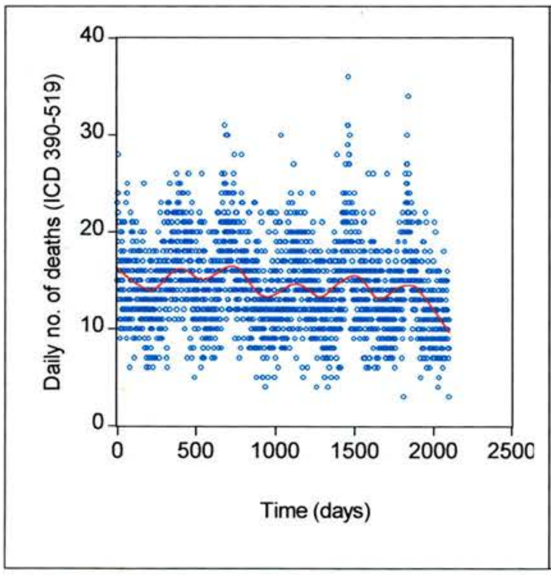
Table 178: Unfiltered Daily Death Counts & Residuals of Core Model for Edinburgh.

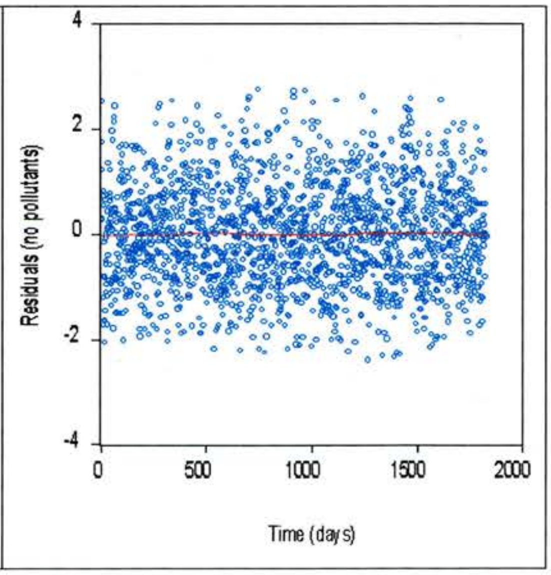




➢ Glasgow

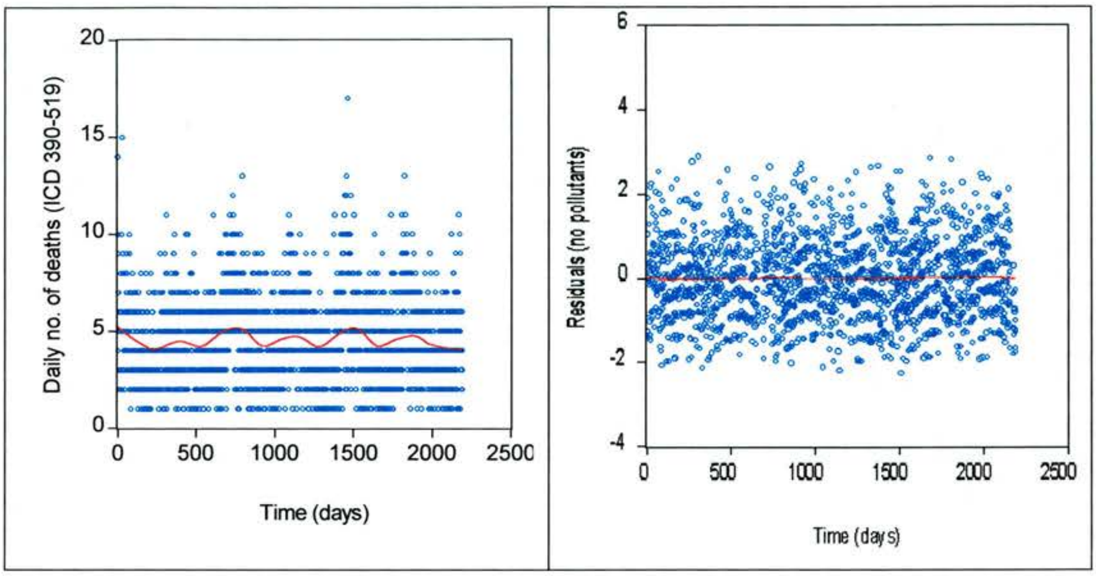
Table 179: Unfiltered Daily Death Counts & Residuals of Core Model for Glasgow.





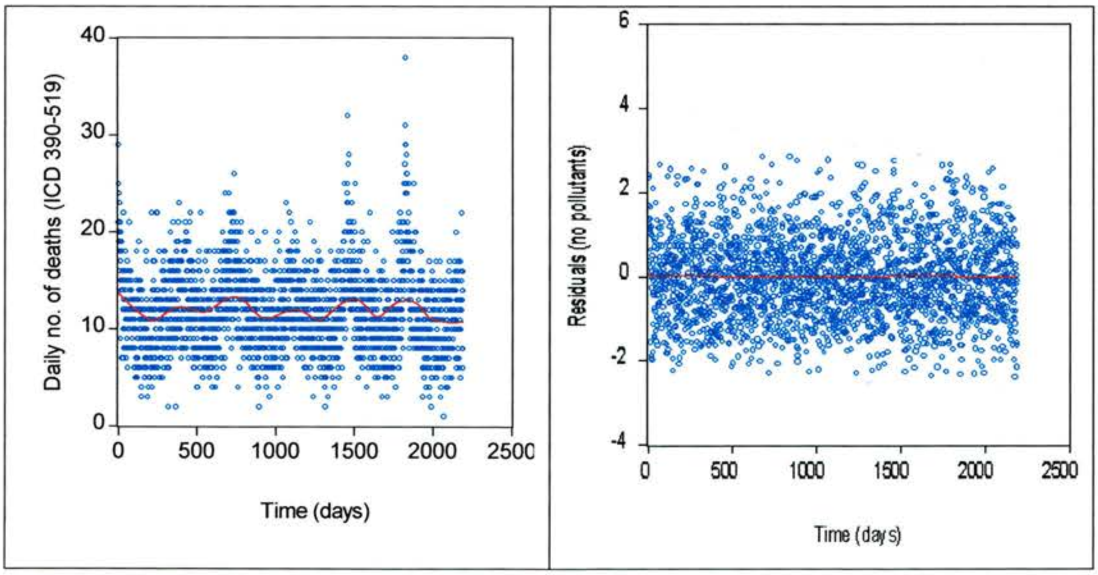
➢ Kingston upon Hull

Table 180: Unfiltered Daily Death Counts & Residuals of Core Model for Kingston upon Hull.



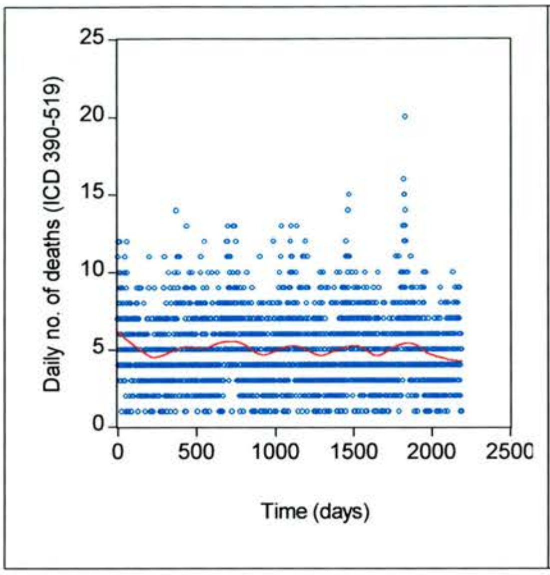
➢ Leeds

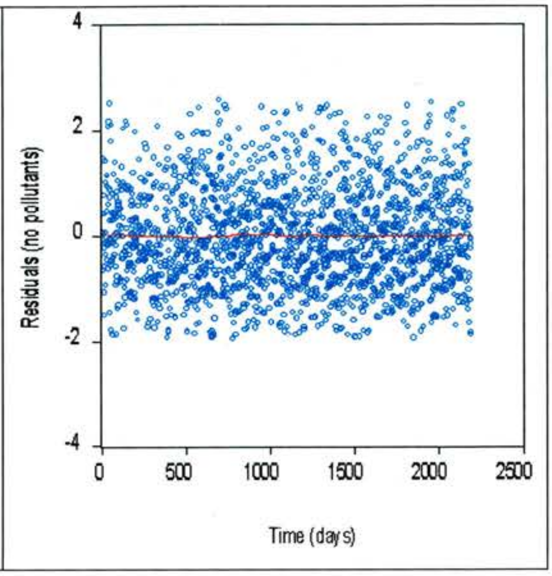
Table 181: Unfiltered Daily Death Counts & Residuals of Core Model for Leeds.



➢ Leicester

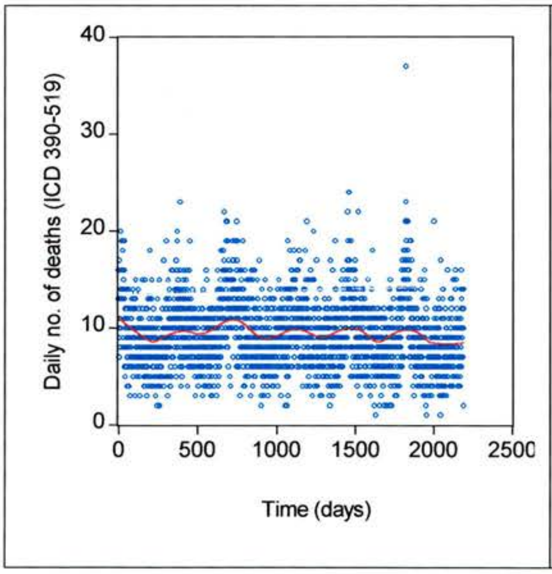
Table 182: Unfiltered Daily Death Counts & Residuals of Core Model for Leicester.

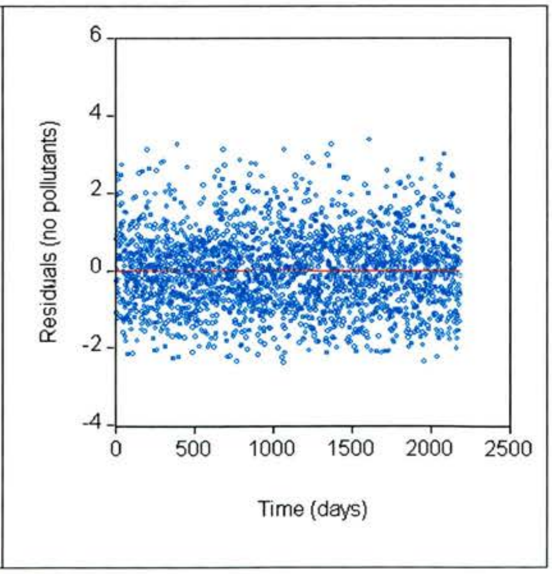




➢ Liverpool

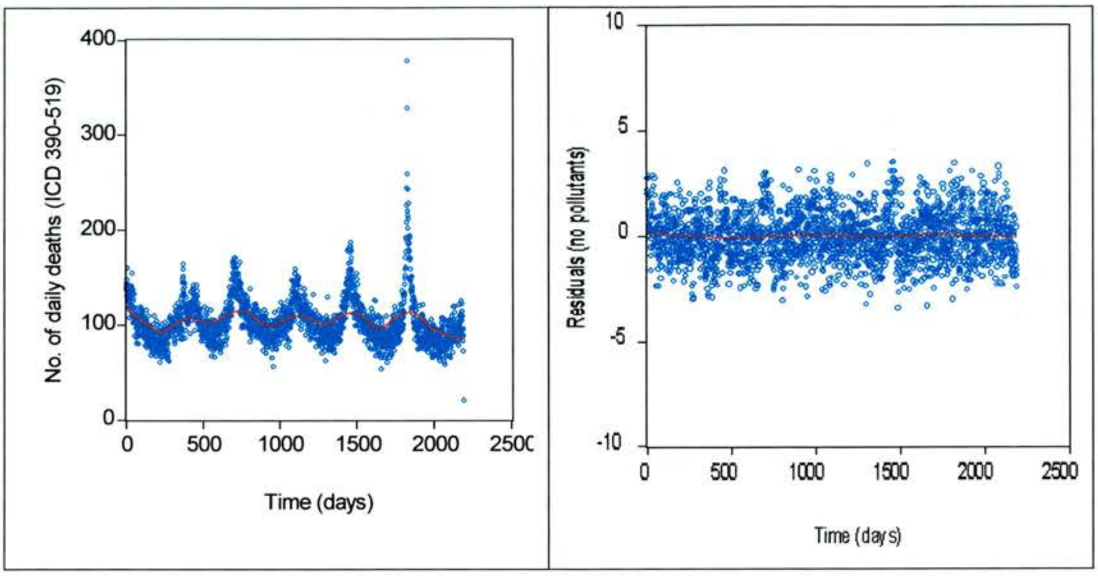
Table 183: Unfiltered Daily Death Counts & Residuals of Core Model for Liverpool.





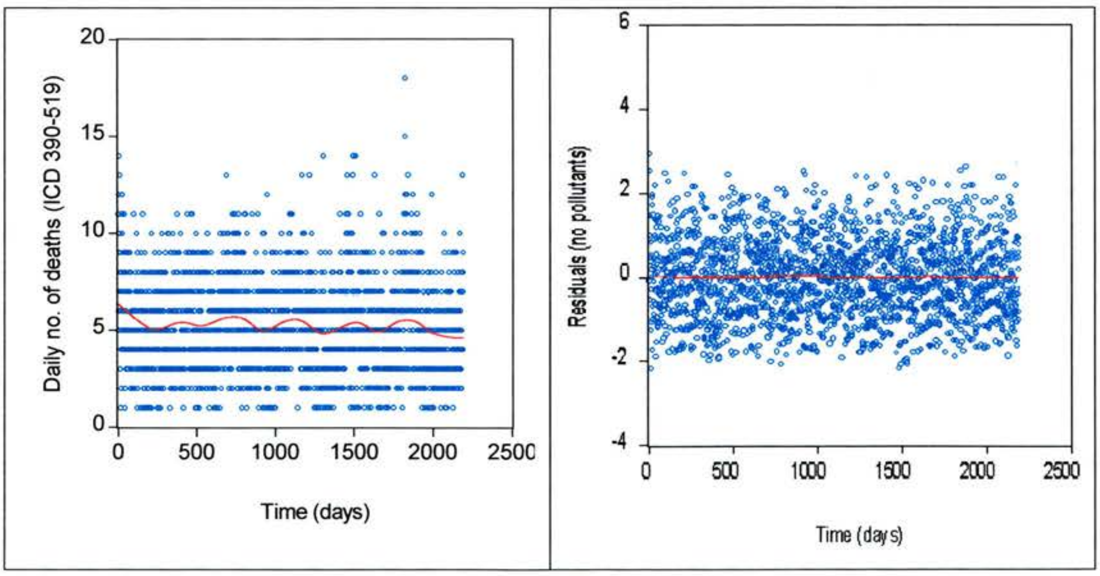
➢ London

Table 184: Unfiltered Daily Death Counts & Residuals of Core Model for London.



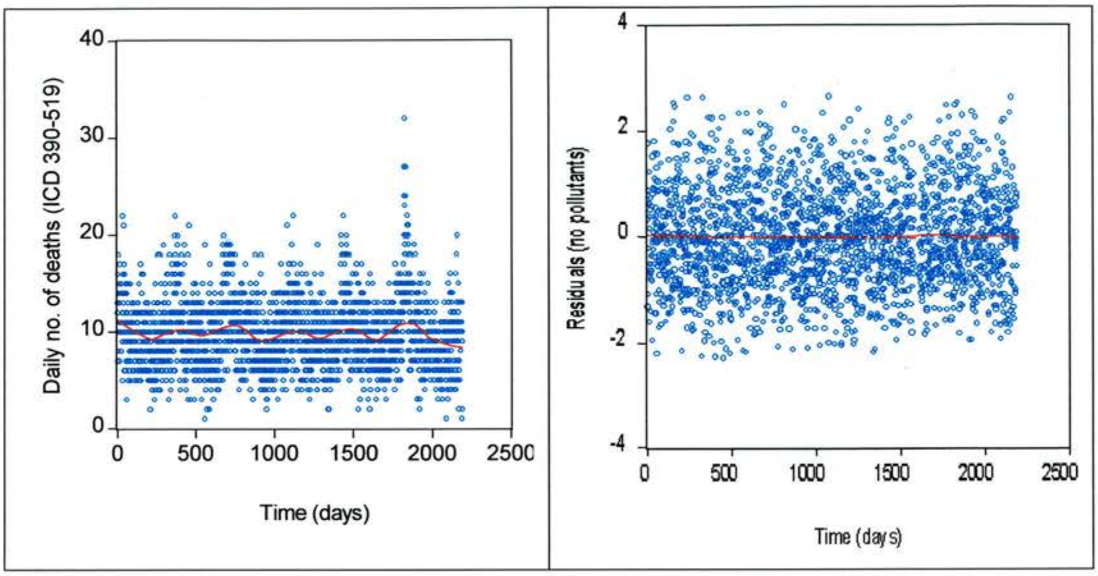
➢ Newcastle upon Tyne

Table 185: Unfiltered Daily Death Counts & Residuals of Core Model for Newcastle upon Tyne.



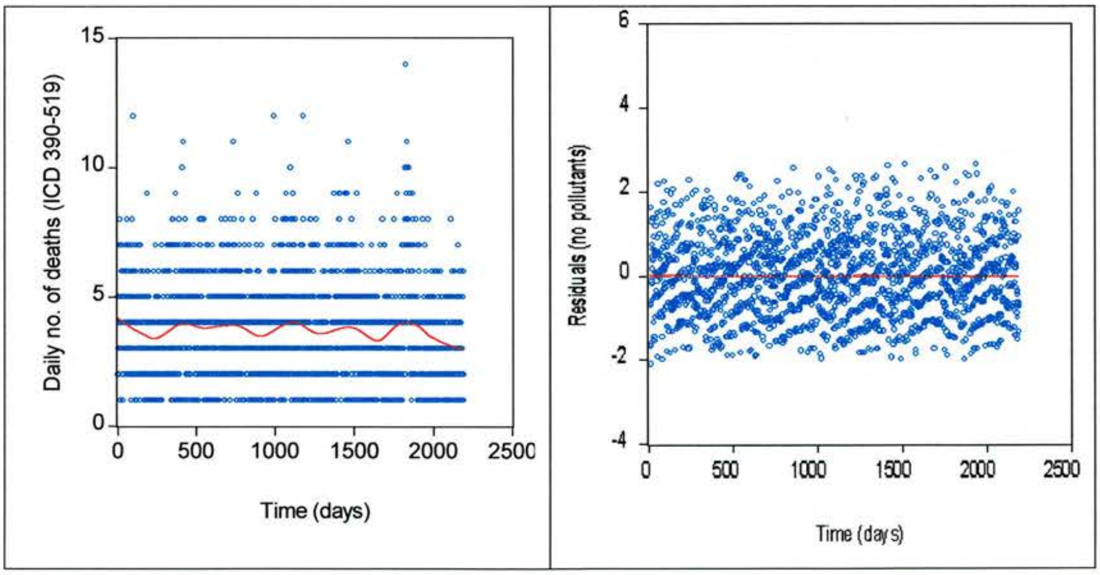
> **Sheffield**

Table 186: Unfiltered Daily Death Counts & Residuals of Core Model for Sheffield.



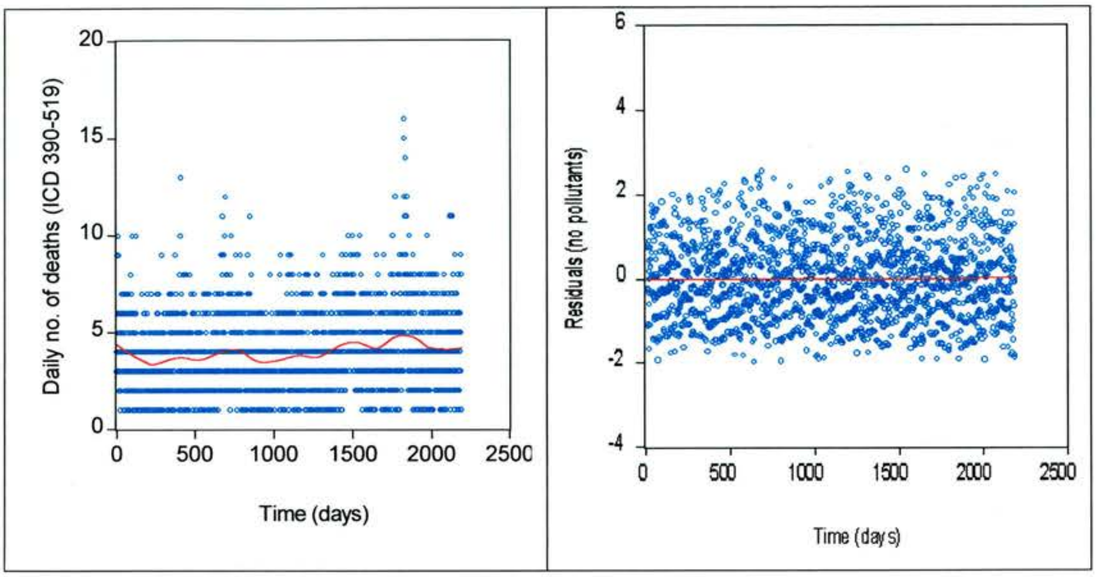
> **Southampton**

Table 187: Unfiltered Daily Death Counts & Residuals of Core Model for Southampton.



> **Swansea**

Table 188: Unfiltered Daily Death Counts & Residuals of Core Model for Swansea.



Appendix 3

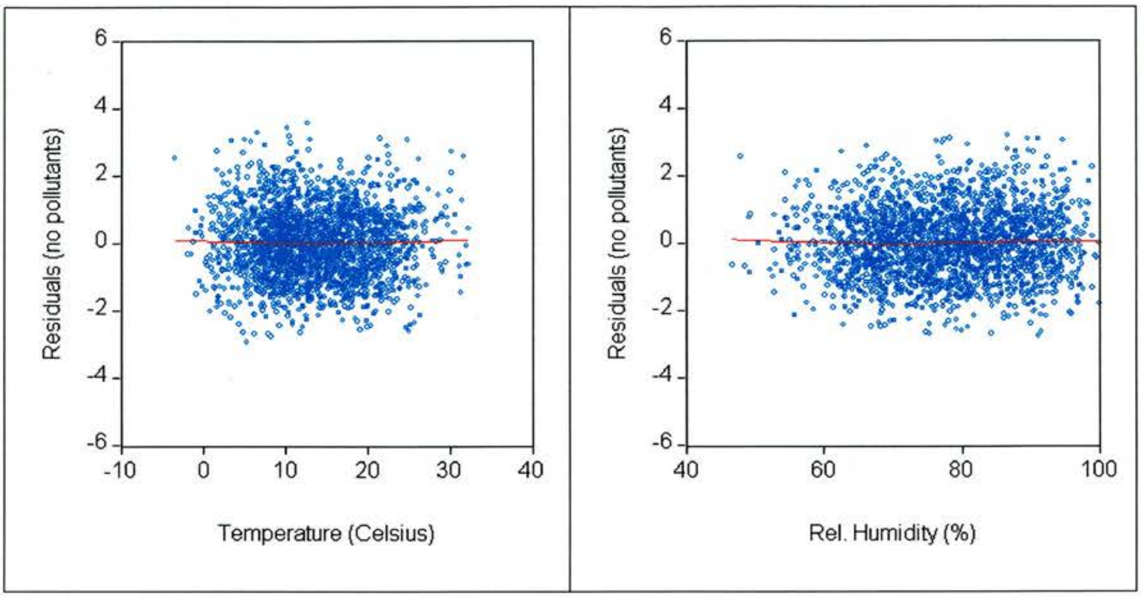
The following figures show the mortality deviance residuals versus both the temperature and relative humidity recorded during the study period in the respective city. The general issue is to examine whether there is some range of temperature or relative humidity where the core model (i.e. after controlling for all of the parametric covariates except air pollution) as described in chapter 9 does not fit well. If such ranges of 'poor fit' (i.e. high temperatures) would exist, the respective residuals (in that range) would be noticeably different from zero.[1] The non-parametric smoother used is LOESS, which has been described in detail in Chapter 9. The LOESS smoothing is represented by the plotted line and does not indicate any region of either temperature or relative humidity where a systematic over- or underprediction of daily mortality has occurred in the core model.

---

[1] See: Schwartz (1994b), p. 477-482.

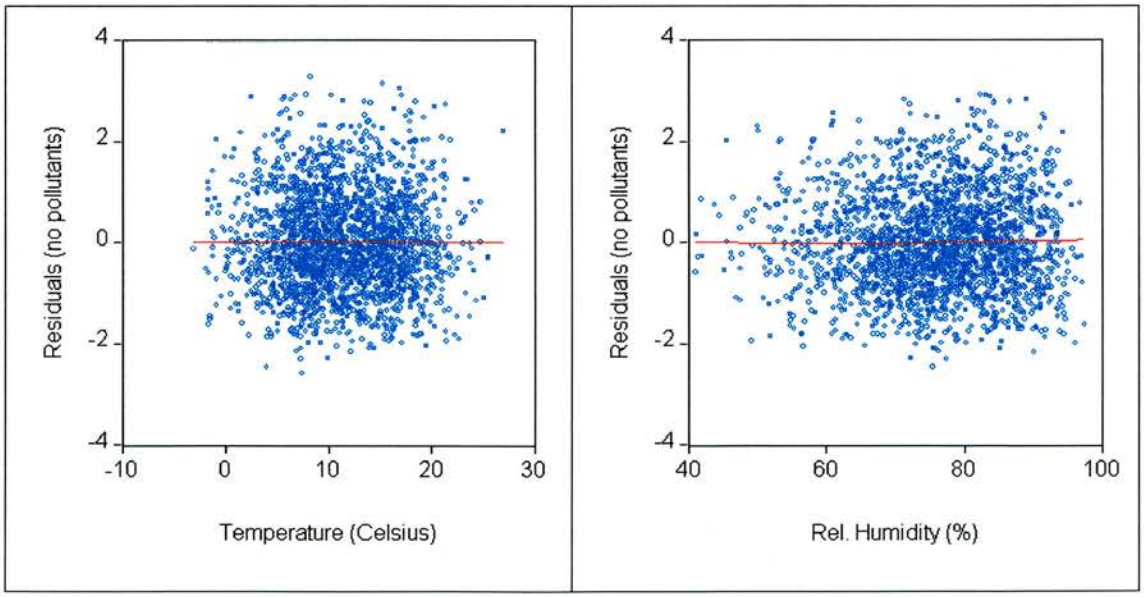
➢ Birmingham

Table 189: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



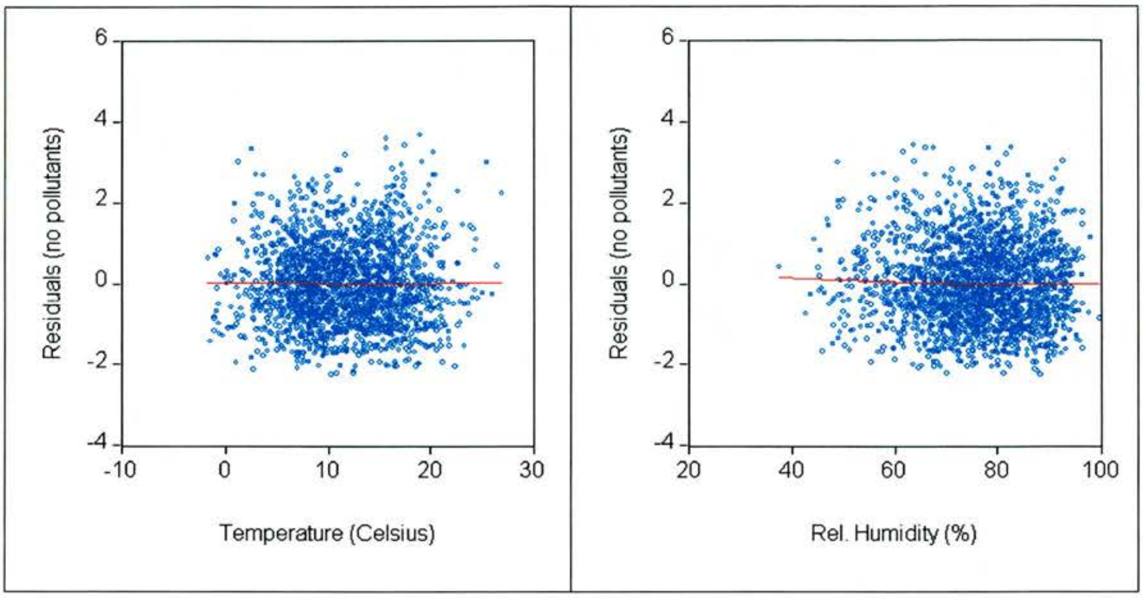
➢ Bristol

Table 190: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



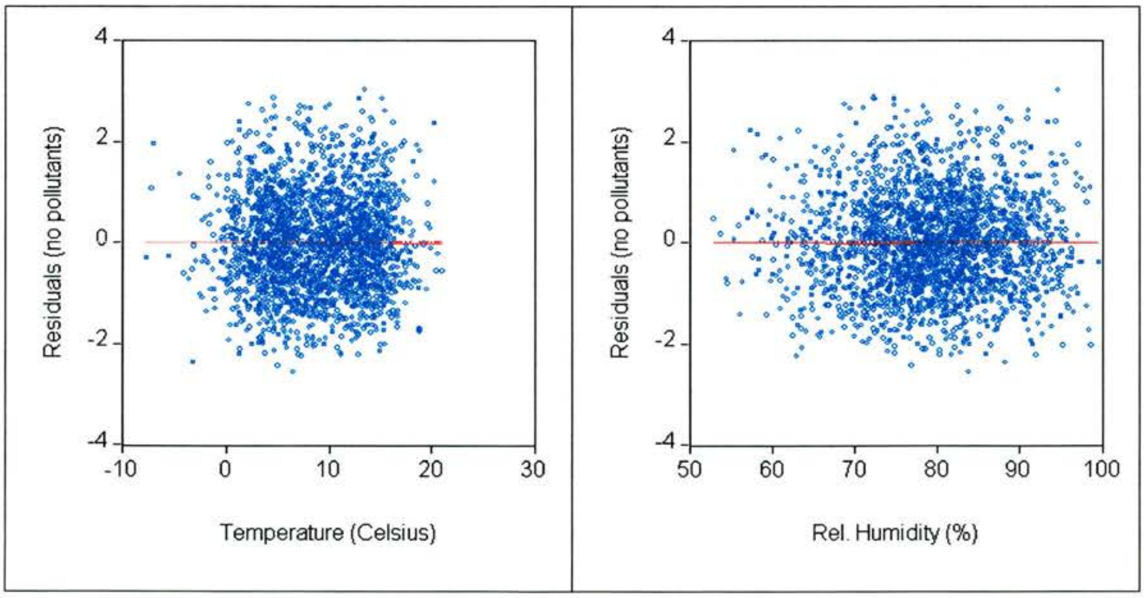
➢ Cardiff

Table 191: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



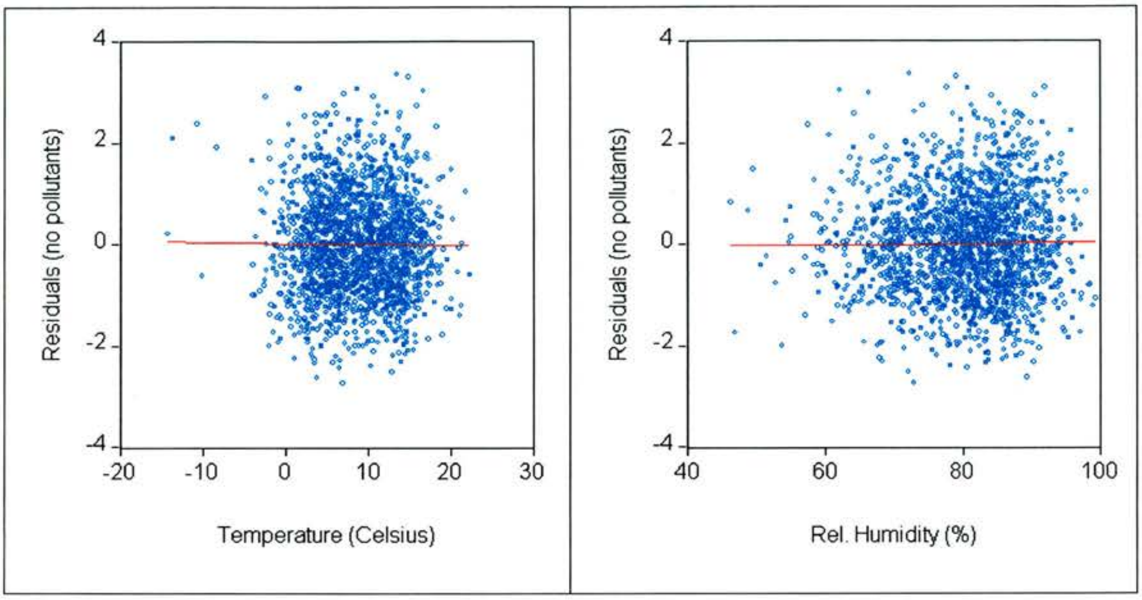
➢ Edinburgh

Table 192: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



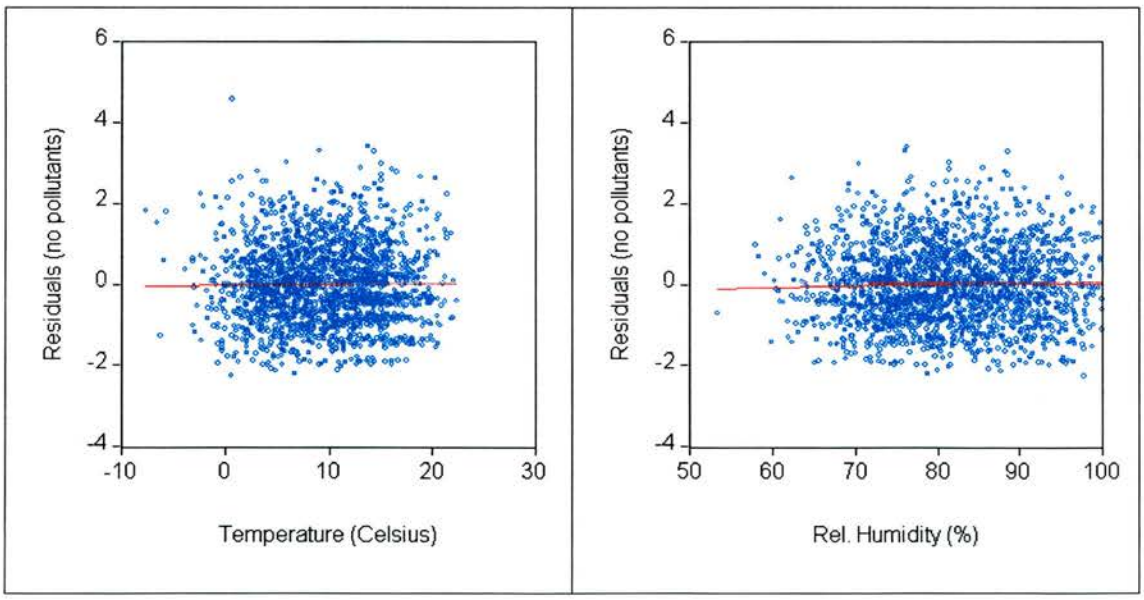
➢ **Glasgow**

Table 193: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



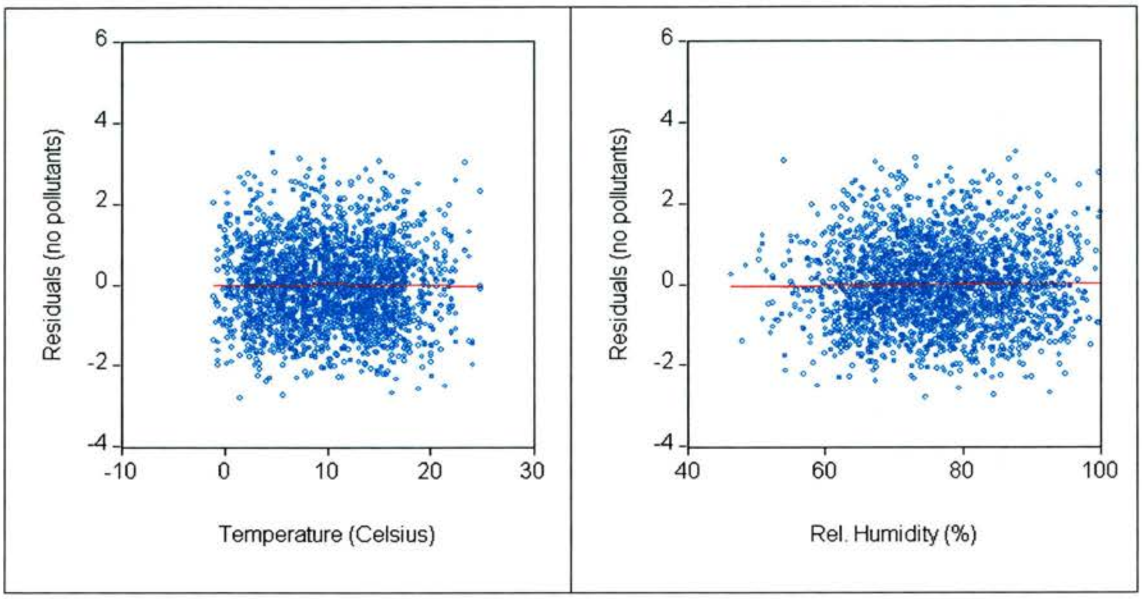
➢ **Kingston upon Hull**

Table 194: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



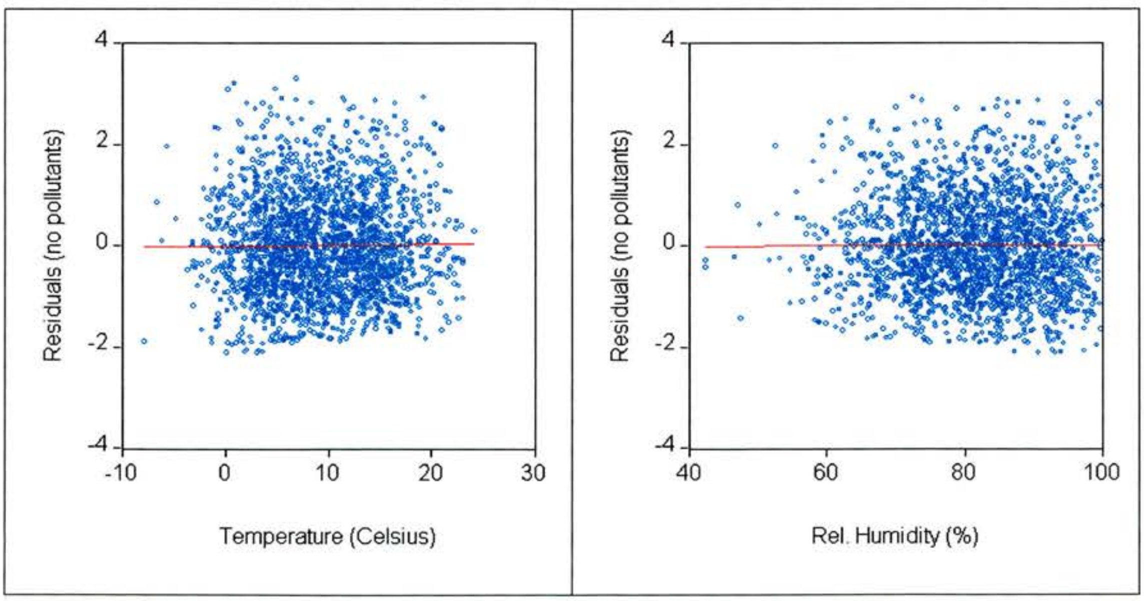
➢ Leeds

Table 195: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



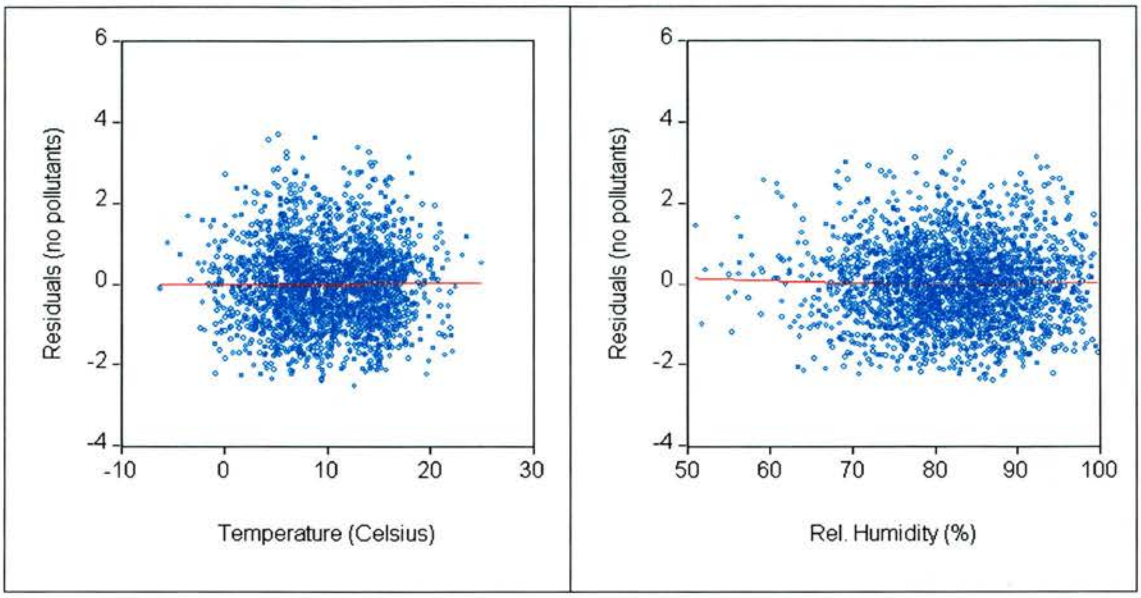
➢ Leicester

Table 196: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



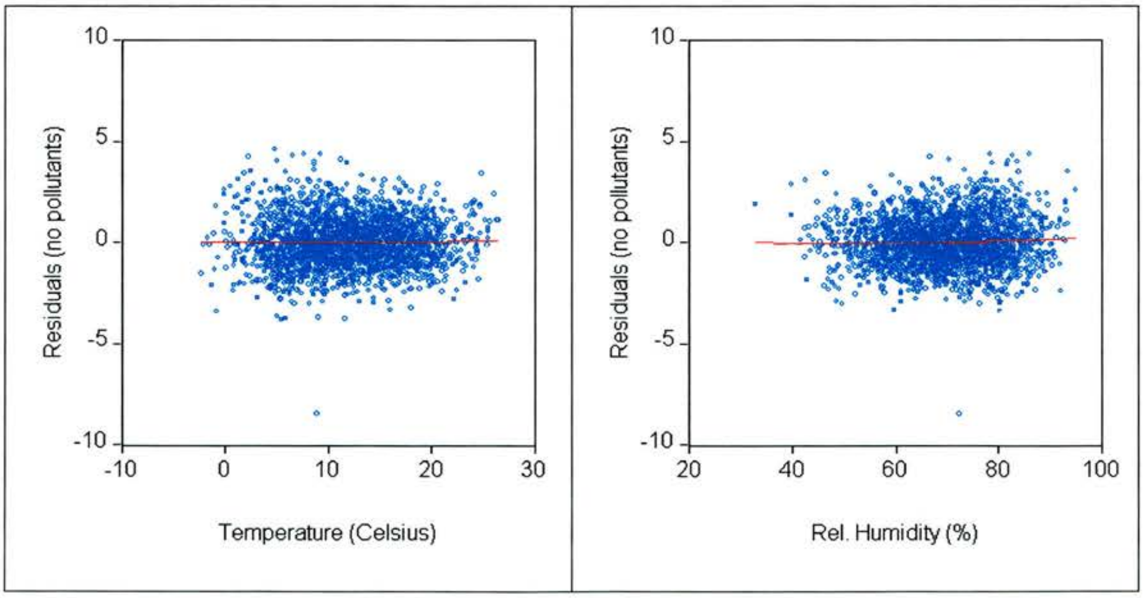
➢ Liverpool

Table 197: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



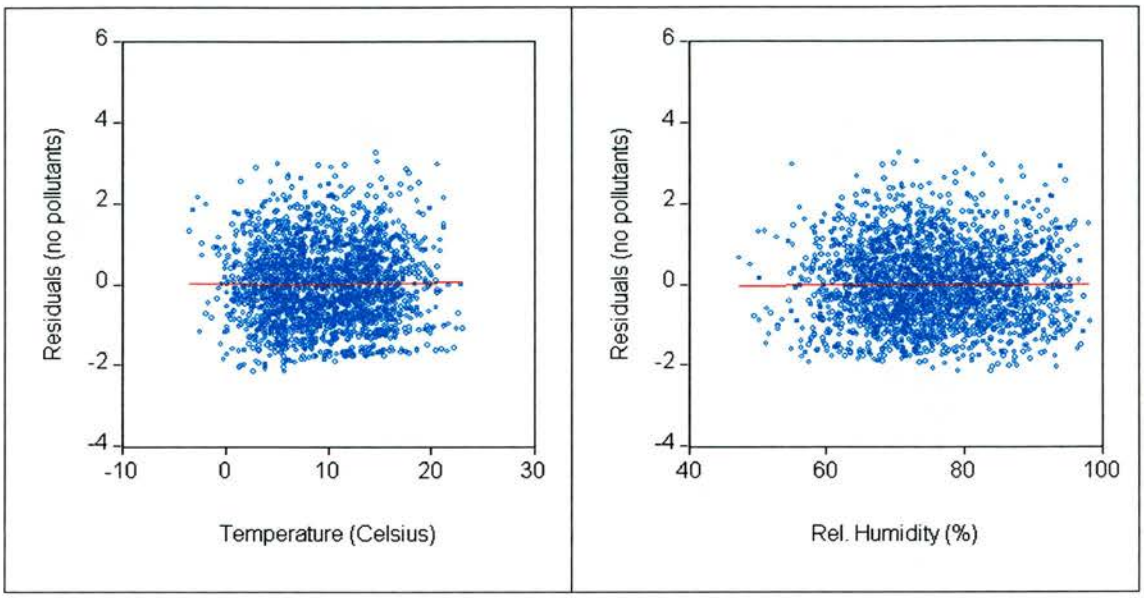
➢ London

Table 198: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



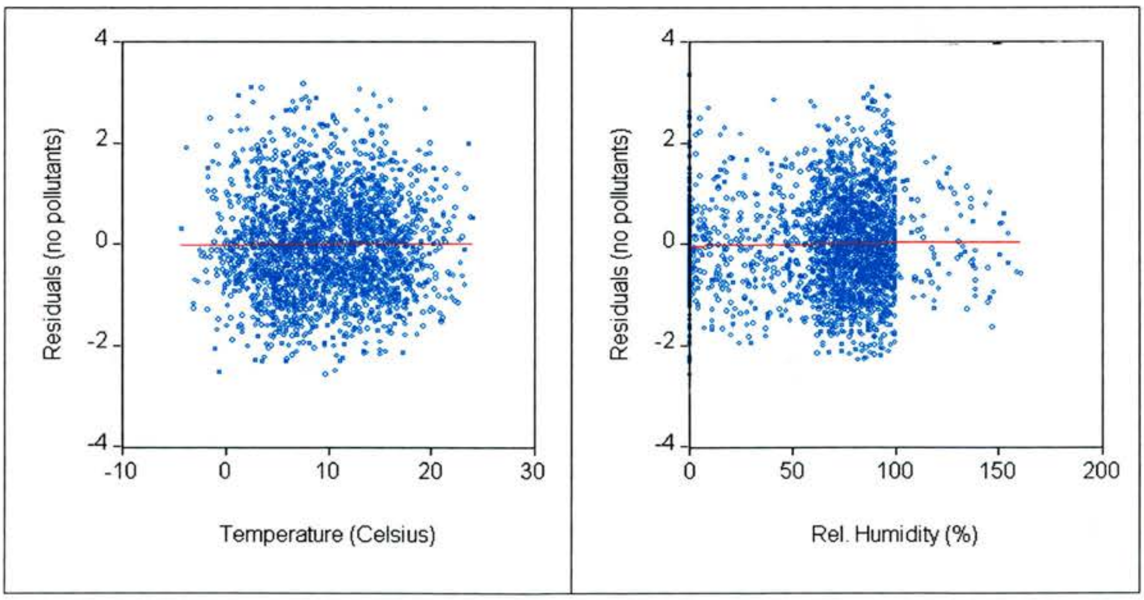
➢ Newcastle upon Tyne

Table 199: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



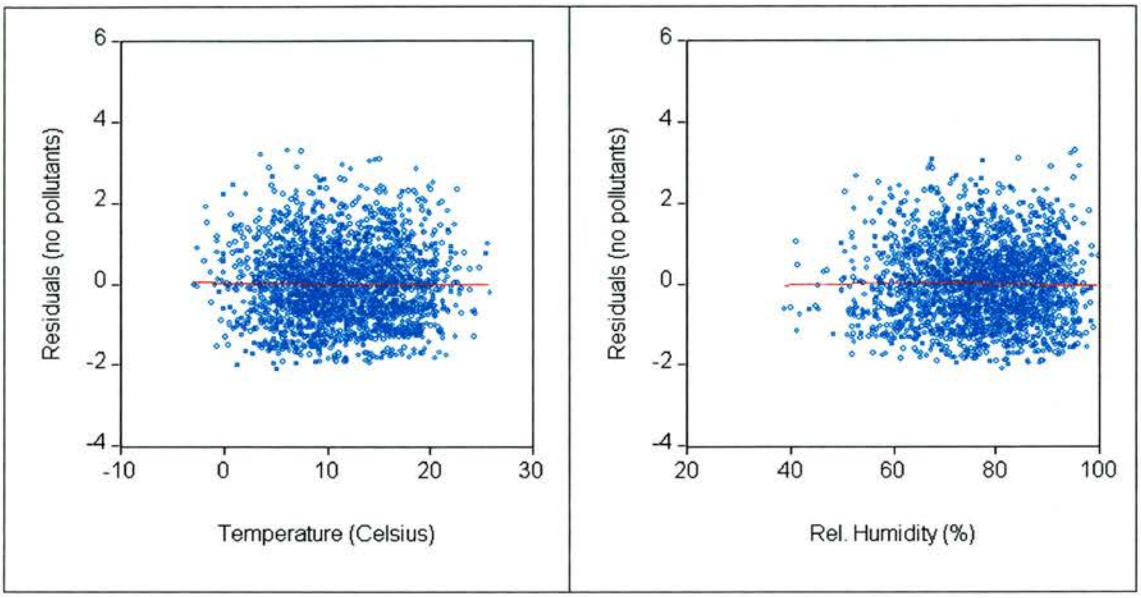
➢ Sheffield

Table 200: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



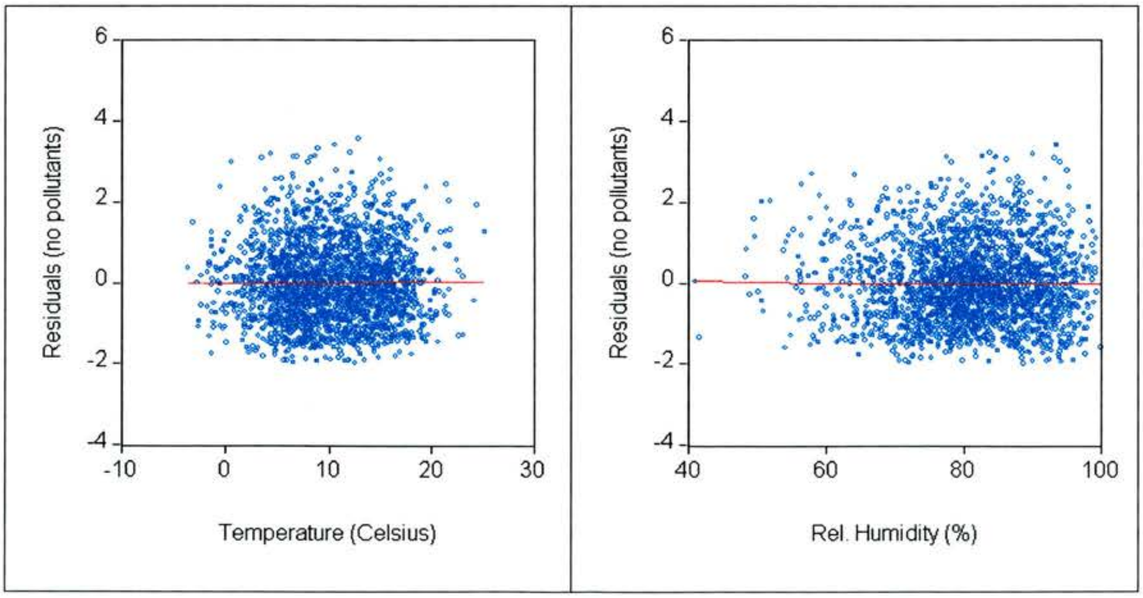
➢ Southampton

Table 201: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



➢ Swansea

Table 202: Mortality Deviance Residuals vs. Temperature & rel. Humidity (LOESS fit).



Appendix 4

As an alternative to the estimation model applied in this study, a somewhat less sophisticated model has also been tested. As described in Chapter 9, one method suggested to de-seasonalise and de-trend the respective time-series data is to pre-filter data before analysing it. Although it had been decided to apply the method where the respective 'filters', i.e. trigonometric functions and dummy variables, are included into the regression function in order to allow them to 'compete' with the unfiltered explanatory variables, the method of pre-filtering was also tested. This may be seen as a control for the final estimation results.

The following approach suggested by Kinney and Özkaynak (1991) has been also applied. In order to eliminate potential confounding effects resulting from shared seasonal cycles of the environmental variables, i.e. weather and pollution, and mortality, a specific filter has been designed. This filter consists of a centred weighted 19-day moving average of each variable. In other words, using specific weights, a moving average of each variable was computed and then subtracted from the variable value at lag 0 (centre) to produce a filtered time-series. For the dependent variable (daily death counts) however, this procedure is not possible, since filtering would lead to a 'non-count' variable, which in turn would not allow estimation of the Poisson regressions, as described in detail in Chapter 9. Therefore, for this variable the weighted filter was included as an independent variable leaving the dependent count variable unchanged. The weights employed are listed in the following table:

Table 203: Filter Coefficients Applied to Remove Seasonal Cycles.

Lag	Weight
-9	0.0123081
-8	0.0200389
-7	0.0295593
-6	0.0403846
-5	0.0518010
-4	0.0629365
-3	0.0728599
-2	0.0806902
-1	0.0857055
0	0.0874320
1	0.0857055
2	0.0806902
3	0.0728599
4	0.0629365
5	0.0518010
6	0.0403846
7	0.0295593
8	0.0200389
9	0.0123081

Source: Kinney and Özkaynak (1991), p. 104.

These filter coefficients were originally developed by Shumway et al. (1983).[1] Performing this filtering operation on each of the variables included in the analysis yields a substantial reduction in the variance contributed by annual and other long-period cycles as described in Chapter 9. The short-term day to day cycles, which are the focus of this analysis, are, however, left intact.

After pre-filtering the data, multiple Poisson regressions were fitted. As with the estimation procedure described and employed in Chapter 9, dummy variables for season, the day of the week, holiday, hot and humid days, and epidemic periods were also included in addition to the meteorological parameters i.e. temperature and humidity.[2] Additionally, the pollution data was transformed into $\mu g/m^3$ as the measurement unit using the same conversion factors as described in Chapter 8.

[1] See: Shumway et al. (1983).

[2] All these variables have been described in detail in Chapter 9.

The results of the multiple Poisson regressions are summarised in the following tables. It has been decided to present the respective estimation results for London and Birmingham only for illustrative purposes. The procedure to compute the relative risk factors has been previously described in Chapter 10.

➢ **London:**

Table 204: Estimation Results with Pre-filtering for London.

Pollutant	Coefficient	Standard Error	Relative Risk Factor (per $100\mu g/m^3$)
PM_{10}	0.000662*	0.000180	1.07
NO_2	0.000613*	0.000106	1.06
SO_2	0.000432*	0.000114	1.04
CO	0.012413*	0.004033	1.13†
Ozone	0.000389*	0.000164	1.04

Note: (*) significant at the 5 % level.
†: Relative Risk Factor per 10 ppm.

➢ **Birmingham**

Table 205: Estimation Results with Pre-filtering for Birmingham.

Pollutant	Coefficient	Standard Error	Relative Risk Factor (per $100\mu g/m^3$)
PM_{10}	0.000968*	0.000395	1.10
NO_2	0.000897*	0.000287	1.09
SO_2	0.000657*	0.000272	1.07
CO	0.027890*	0.013462	1.32†
Ozone	0.000765*	0.000355	1.08

Note: (*) significant at the 5 % level.
†: Relative Risk Factor per 10 ppm.

Comparing these results with those presented in Chapter 10 and Appendix 5, where multiple Poisson regression estimation have been used, close similarities can be observed. This may indicate that the health effects associated to air pollution are indeed not an artefact of the respective statistical model applied.

Appendix 5

In Chapter 10 of this study, the estimation results of the multiple Poisson regressions were presented and discussed. It had been decided to express the regression results in form of a relative risk factor associated with an increment between the 5^{th} and 95^{th} percentile of the respective pollution data recorded in the various cities.

However, as mentioned before, it was also decided to present an alternative to this procedure in order to make the estimation results obtained in this study more comparable to the results of previous studies as reviewed in Chapter 7. In doing so, the regression results are also expressed using the relative risk factors for an increment of 100 $\mu g/m^3$ for the pollutants Particular Matter, nitrogen dioxide, sulphur dioxide, and ozone, and 10ppm for carbon monoxide. Thus, the resulting relative risk factors express the percentage increase in mortality, assuming an increase in pollution concentration by 100 $\mu g/m^3$ and 10ppm respectively.

These factors are computed for each pollutant and city in the same way as described and illustrated in Chapter 10. Likewise to the procedure in Chapter 10, the 95 per cent confidence intervals are also calculated. For the case where the estimated regression coefficient is only significant on the 10 per cent level, the 90 per cent confidence interval is produced respectively.

➢ Birmingham

Table 206: Estimation Results of the Final Poisson Model for Birmingham.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.000903*	1	0.000458	1.10 (1.00; 1.20)
NO_2	0.000797*	1	0.000338	1.08 (1.01; 1.16)
SO_2	0.000542*	1	0.000283	1.06 (1.00;1.12)
CO	0.016780**	1	0.009239	1.18 (1.02; 1.37)‡
O_3	0.000888*	1	0.000314	1.09 (1.03; 1.16)

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

➢ Bristol

Table 207: Estimation Results of the Final Poisson Model for Bristol.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001024**	2	0.000581	1.11 (1.02; 1.23)‡
NO_2	0.000879*	0-3	0.000227	1.09 (1.06; 1.11)
SO_2	0.001826*	0-3	0.000927	1.20 (1.07; 1.43)
CO	0.009621*	1	0.004730	1.10 (1.00; 1.20)
O_3	-0.001886	1	0.000645	N/A

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

- **Cardiff**

Table 208: Estimation Results of the Final Poisson Model for Cardiff.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001747**	1	0.001035	1.19 (1.00; 1.41)‡
NO_2	0.001612*	1	0.000817	1.17 (1.00; 1.38)
SO_2	-0.000661	2	0.001102	N/A
CO	0.018871**	2	0.010743	1.20 (1.01; 1.44)‡
O_3	0.000836	2	0.000722	N/A

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

- **Edinburgh**

Table 209: Estimation Results of the Final Poisson Model for Edinburgh.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001972*	3	0.000940	1.21 (1.01; 1.46)
NO_2	0.001088**	2	0.000605	1.11 (1.01; 1.23)‡
SO_2	0.001213*	3	0.000612	1.12 (1.00; 1.27)
CO	0.054420*	1	0.027197	1.72 (1.01; 2.93)
O_3	-0.001592*	2	0.000629	0.85 (0.75; 0.96)

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

➢ Glasgow

Table 210: Estimation Results of the Final Poisson Model for Glasgow.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}†	0.001117**	2	0.000638	1.11 (1.01; 1.25)‡
NO_2	0.000363*	2	0.000139	1.04 (1.01; 1.07)
SO_2	N/A			N/A
CO	0.019680**	2	0.011719	1.21 (1.00; 1.48)‡
O_3	N/A			N/A

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
†: Data based on BS, converted to PM_{10} using formula PM_{10} = BS x 0.9.
‡: 90 per cent confidence interval.

➢ Kingston upon Hull

Table 211: Estimation Results of the Final Poisson Model for Kingston upon Hull.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001320*	1	0.000871	1.13 (0.96; 1.35)
NO_2	0.000636**	2	0.000367	1.06 (1.00; 1.13)‡
SO_2	0.000420	2	0.000986	N/A
CO	0.009708*	1	0.004925	1.10 (1.00; 1.21)
O_3	0.001529**	0	0.000880	1.16 (0.98; 1.34)‡

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

➢ Leeds

Table 212: Estimation Results of the Final Poisson Model for Leeds.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	-0.000650	1	0.000553	N/A
NO_2	0.000683	3	0.000500	N/A
SO_2	0.000598*	3	0.000294	1.06 (1.00; 1.12)
CO	0.038416*	2	0.018244	1.46 (1.03; 2.09)
O_3	-0.000820**	3	0.000423	0.92 (0.86; 0.99)‡

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

➢ Leicester

Table 213: Estimation Results of the Final Poisson Model for Leicester.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.000297*	2	0.000147	1.03 (1.00; 1.06)
NO_2	0.000690	1	0.000844	N/A
SO_2	0.001057	1	0.000775	N/A
CO	0.023693	1	0.030465	N/A
O_3	0.000188**	1	0.000104	1.02 (1.00; 1.03)‡

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

➢ Liverpool

Table 214: Estimation Results of the Final Poisson Model for Liverpool.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001254*	0-2	0.000627	1.13 (1.01; 1.29)
NO_2	0.001173**	0-3	0.000643	1.12 (1.00; 1.24)‡
SO_2	0.001181*	0-3	0.000452	1.12 (1.01; 1.23)
CO	0.060836*	2	0.029564	1.83 (1.02; 3.3)
O_3	-0.000781	1	0.000611	N/A

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

➢ London

Table 215: Estimation Results of the Final Poisson Model for London.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.000572*	1	0.000174	1.06 (1.02; 1.10)
NO_2	0.000487*	1	0.000113	1.05 (1.03; 1.07)
SO_2	0.000543*	3	0.000117	1.06 (1.03; 1.08)
CO	0.018590*	1	0.005734	1.20 (1.07; 1.35)
O_3	0.000692*	1	0.000241	1.07 (1.02; 1.12)

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.

➢ Newcastle upon Tyne

Table 216: Estimation Results of the Final Poisson Model for Newcastle upon Tyne.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001140*	3	0.000562	1.12 (1.00; 1.25)
NO_2	0.001629*	1	0.000556	1.18 (1.02; 1.38)
SO_2	0.000750**	3	0.000425	1.08 (1.00; 1.16)‡
CO	0.006603	2	0.004137	N/A
O_3	0.000924**	2	0.000533	1.09 (1.00; 1.20)‡

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

➢ Sheffield

Table 217: Estimation Results of the Final Poisson Model for Sheffield.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.001086**	3	0.000562	1.11 (1.00; 1.22)‡
NO_2	0.000693*	1	0.000296	1.07 (1.01; 1.16)
SO_2	0.001128*	3	0.000558	1.11 (1.00; 1.25)
CO	0.020887**	4	0.012474	1.23 (1.00; 1.51)‡
O_3	0.001093	2	0.000927	N/A

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

➢ Southampton

Table 218: Estimation Results of the Final Poisson Model for Southampton.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.002689*	0-3	0.000579	1.30 (1.14; 1.46)
NO_2	0.002005*	0-3	0.000958	1.22 (1.01; 1.47)
SO_2	0.001178*	1	0.000597	1.12 (1.00; 1.26)
CO	0.017128*	2	0.006498	1.18 (1.04; 1.35)
O_3	0.002351*	0-4	0.001085	1.26 (1.02; 1.56)

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

➢ Swansea

Table 219: Estimation Results of the Final Poisson Model for Swansea.

Pollutant	Coefficient	Lag	Standard Error	Relative Risk (CI)
PM_{10}	0.000971*	1	0.000478	1.10 (1.00;1.21)
NO_2	0.001078*	3	0.000472	1.11 (1.02; 1.22)
SO_2	0.001688**	4	0.000888	1.18 (1.02; 1.37)‡
CO	0.021821*	2	0.010968	1.24 (1.00; 1.54)
O_3	0.001064*	1	0.000400	1.11 (1.04; 1.19)

Note: (*) significant at the 5 % level; (**) significant at the 10 % level.
‡: 90 per cent confidence interval.

Likewise to the procedure in Chapter 10, the following table provides a summary of the multiple Poisson regressions. As described before, the results are expressed as relative risk functions, while the increment used in this Appendix is 100 $\mu g/m^3$ (10 ppm for carbon monoxide).

Table 220: Summary of Estimation Results.

	PM_{10}		NO_2		SO_2		CO		O_3	
	Coefficient (S.E.)	RR	Coefficient (S.E.)	RR	Coefficient (S.E.)	RR	Coefficient (S.E.)	RR	Coefficient (S.E.)	RR
Birmingham	0.000903 (0.000458)	1.10	0.000797 (0.000336)	1.10	0.000542 (0.000283)	1.06	0.016780 (0.009239)	1.18	0.000888 (0.000314)	1.09
Bristol	0.0010245 (0.000581)	1.11	0.000879 (0.000227)	1.09	0.001826 (0.000927)	1.20	0.009621 (0.004730)	1.10	-0.001886 (0.000645)	N/A
Cardiff	0.001747 (0.001035)	1.19	0.001612 (0.000817)	1.17	-0.000661 (0.001102)	N/A	0.018871 (0.010743	1.20	0.000836 (0.000.722)	N/A
Edinburgh	0.001972 (0.000940)	1.21	0.001088 (0.000605)	1.11	0.001213 (0.000612)	1.12	0.054420 (0.027197)	1.72	-0.001592 (0.000629)	0.85
Glasgow	0.001117 (0.000638)	1.11	0.000363 (0.000139)	1.04	N/A	N/A	0.019680 (0.011719)	1.21	N/A	N/A
Kingston upon Hull	0.001320 (0.000871)	1.13	0.000636 (0.000367)	1.06	0.000420 (0.000986)	N/A	0.009708 (0.004925)	1.10	0.001529 (0.000880)	1.16
Leeds	-0.000650 (0.000553)	N/A	0.000683 (0.000500)	N/A	0.000598 (0.000294)	1.06	0.038416 (0.018244)	1.46	-0.000820 (0.000423)	0.92
Leicester	0.000297 (0.000147)	1.03	0.000690 (0.000844)	N/A	0.001057 (0.000775)	N/A	0.023693 (0.030465)	N/A	0.000188 (0.000104)	1.02
Liverpool	0.001254 (0.000627)	1.13	0.001173 (0.000643)	1.12	0.001181 (0.000452)	1.12	0.0660836 (0.029564)	1.83	-0.000781 (0.000611)	N/A
London	0.000572 (0.000174)	1.06	0.000487 (0.000113)	1.05	0.000543 (0.000117)	1.06	0.018590 (0.005734)	1.20	0.000692 (0.000241)	1.07
Newcastle upon Tyne	0.001140 (0.000532)	1.12	0.001629 (0.000556)	1.18	0.000750 (0.000425)	1.08	0.006603 (0.004137)	N/A	0.000924 (0.000533)	1.09
Sheffield	0.001086 (0.000562)	1.11	0.000693 (0.000296)	1.07	0.001128 (0.000558)	1.11	0.020887 (0.012474)	1.23	0.001093 (0.000927	N/A
Southampton	0.002689 (0.000579)	1.30	0.002005 (0.000958)	1.22	0.001178 (0.000597)	1.12	0.017128 (0.006498)	1.18	0.002351 (0.001085)	1.26
Swansea	0.000971 (0.000478)	1.10	0.001078 (0.000472)	1.11	0.001688 (0.000888)	1.18	0.021821 (0.010968)	1.24	0.001064 (0.000400)	1.11

Appendix 6

In Chapter 11 of this study the estimation results have been expressed in form of annual total deaths brought forward in respect of current levels of air pollution concentrations. As noted before, an alternative procedure is to calculate the reduced numbers of deaths brought forward per 1 $\mu g/m^3$ per city per year. For illustration purpose, the results of such calculations are presented in the table below. For carbon monoxide, the calculations have been made per 0.1 ppm rather than 1ppm or 10 ppm since the annual mean concentrations are only in the range of around 0.65 ppm.

Table 221: Summary of Annual Reductions in Numbers of Deaths brought forward.

Reduction in deaths brought forward (acute)					
City	**PM_{10}**	**NO_2**	**SO_2**	**CO^a**	**Ozone**
Birmingham	10	9	6	18	10
Bristol	4	4	8	4	N/A
Cardiff	5	5	N/A	6	N/A
Edinburgh	10	6	6	28	N/A
Glasgow	11	3	N/A	19	N/A
Kingston upon Hull	4	2	N/A	3	4
Leeds	N/A	N/A	5	29	N/A
Leicester	1	N/A	N/A	N/A	1
Liverpool	7	7	7	38	N/A
London	38	32	36	123	46
Newcastle upon Tyne	4	6	3	N/A	3
Sheffield	7	4	7	13	N/A
Southampton	6	4	3	4	5
Swansea	2	3	4	5	3

[a]: Reductions are calculated for 0.1 ppm.