A Study of Traffic-Related Particulate Air Pollution

By

Anne Marie Desmond

Submitted in part fulfilment of the NCEA requirements for the award of Master of Science in Environmental Protection

at

Institute of Technology, Sligo.

June 2001

Project Supervisor: Dr. John Bartlett

ABSTRACT

Particulate air pollution (PM_{10}) was monitored at a busy urban intersection using a continuous method. Meteorological conditions: wind speed, wind direction, air temperature and percent relative humidity were measured continuously, in conjunction with PM_{10}. Traffic volume counts were obtained for part of the monitoring period.

Motor vehicles were identified as being the main contributor to the overall PM_{10} level at the intersection. The meteorological parameters were assessed to establish their effect on the measured PM_{10} mass concentration values. A model of the relationships was developed and evaluated. The model was found to be reasonable in predicting PM_{10} values based on historical data.

An empirical technique was investigated based on the developed model to assess the suitability to prediction of daily PM_{10} average values. The empirical technique was concluded to be unacceptable when applied to PM_{10} due to the nature of the pollutant itself.

ACKNOWLEDGEMENTS

I would like to thank my supervisor, Dr. John Bartlett, for his assistance throughout the course of writing this thesis.

I wish to thank Professor S. H. Perry for making available the air monitoring facilities of The Department of Civil, Structural and Environmental Engineering, Trinity College.

I am grateful to my colleagues at The Department of Civil, Structural and Environmental Engineering for their help and encouragement over the last year. In particular Dr. Margaret O'Mahony, Mr. Paul Johnston and Mr. Chris O'Donovan.

I wish to acknowledge the contribution of data from The of The Department of Civil, Structural and Environmental Engineering, Trinity College, Met Eireann and Dublin Corporation.

I would like to thank my family and friends for their encouragement over the last year. Finally I would like to express my sincere gratitude to Tom for his patience and support.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS

As	Arsenic
Ca	Calcium
Cd	Cadmium
°C	degrees Celsius
cm^{-3}	cubic centimetre
CO	carbon monoxide
EPA	Environmental Protection Agency
Fe	Iron
g sec^{-1}	grams per second
H_2So_4	sulphuric acid
l min^{-1}	litres per minute
m^3 sec^{-1}	cubic meter per second
µg m^{-3}	micrograms per cubic meter
µm	micrometer
No_2	nitrogen dioxide
No_x	oxides of nitrogen
PGE	platinum group elements
PM_{10}	particulate matter with an aerodynamic diameter less than 10 µm
$PM_{2.5}$	particulate matter with an aerodynamic diameter less than 2.5 µm
ppm	parts per million
SCATS	Sydney co-ordinated adaptive traffic systems
Se	selenium
Si	silica
So_2	sulphur dioxide
TEOM	tapered element oscillating microbalance
UK DETR	United Kingdom Department of the Environment, Transport and the Regions
US EPA	United States Environmental Protection Agency
VOCs	volatile organic compounds
Zn	zinc

Chapter 1. Introduction

Ambient particulate matter is not a well-defined entity. Over the years different terms have been used to describe particulate matter. Some of these terms refer to methods of monitoring particulate air pollution others are based on the health effects. Differing terms also reflect the change in the nature of the pollutant. Originally it was described as soot or black smoke, later the concept of total suspended particulates was introduced. Since 1990 particulate matter is defined in terms of size fractions. The size fractions of most concern in air quality assessments are PM_{10} and $PM_{2.5}$. PM_{10} is that fraction of particulate matter with a diameter of 10 μm or less, $PM_{2.5}$ is defined as particulate matter with a diameter of 2.5 μm or less. (United States Environmental Protection Agency (US EPA), 1996). In recent years there has been increasing interest in the assessment of PM_{10} in air quality surveys, studies have shown a positive association between daily PM_{10} and non-accident mortality.

The nature of particulate air pollution has changed over the last number of decades. Such changes are mainly as a result of stricter legislative control over air quality. Improvements in air pollution control technologies and regulation of industries through licence agreements has greatly reduced particulate emissions from many industrial sources. Soot and black smoke were major problems in urban areas due to the burning of coal as a domestic fuel. The introduction of the bituminous coal regulations in 1990 and the change over to cleaner domestic fuels has greatly reduced this problem. The lead content of particulate matter has fallen to very low levels in recent years (Environmental Protection Agency (EPA), 1996). The reductions of lead emissions resulted from the introduction, in 1985, of the lead content of petrol regulations.

Increased economic growth and urban development of the type that Ireland has experienced in the last number of years can have a pronounced effect on the environment. Urban population growth combined with changes in land use is a major probable reason for air quality problems. In built up urban areas the major sources of particulate air pollution are transport and domestic and commercial space heating.

At present in Ireland there is more urban and industrial development and people commuting to work than ever before. With the development of large industrial estates on the edges of towns and cities and the spread of urban development distance travelled and commuting times have increased dramatically. The complete separation of industry and habitation, originally envisaged as an environmental improvement and a solution in societies with heavily polluting industries is now outdated as it has only lead to increased commuting traffic and congestion (Fenger, 1999). There has been a major increase in car ownership, statistics from The Department of the Environment show a 25% increase in car ownership from 1998 to 1999. Air pollution is becoming an increasing problem in urban centres due to increased volumes of motor vehicles and traffic congestion.

Control of particulate air pollution was exercised under Council Directive 80/779/EEC, which set standards for both sulphur dioxide and suspended particulates. This standard has been improved since the introduction of the Council Directive 96/62/EC on ambient air quality assessment and management, (The Air Quality Framework Directive). The first Daughter directive, adopted under the framework directive in April 1999, covers: sulphur dioxide, nitrogen oxides, particulate matter and lead. Threshold limits have been set with respect to daily and annual averages that must be attained by certain dates. The implementation of this directive calls for stricter control of particulate air pollution by putting in place stricter air quality standards, (EPA, 1997).

The level of monitoring and assessment of air quality required by this directive has increased considerably from previous regulations. Such assessments of air quality must be representative of entire areas. Based on such assessments strategies and mitigating plans must be put in place to reduce air pollution levels to those required by legislation. Such plans need to be assessed and updated accordingly to ensure that various targets set will be achieved. There is a major requirement for continuous monitoring under this directive thus considerable investment on the part of local authorities is required. Continuous monitoring equipment is far more expensive then previously used methods.

It is not possible to monitor air quality at all places at the one time, making it difficult to assess the temporal and spatial variations of pollutants at high resolution. The major constraints are the expense of continuous monitoring equipment and the difficulties in acquiring secure locations with power supplies (Croxford and Penn, 1998). In order for air quality measurements to be representative of pollution levels, details of local climatic conditions are necessary. Measuring of local climatic data is of extreme importance in air quality assessments as the topography of a region has an effect on climatic conditions that in turn effect the dispersion and dilution of air pollutants in the atmosphere (Croxford and Penn, 1996).

The real time data from continuous monitoring provides vast amounts of information on pollutant levels on a daily basis. In order to be proactive in providing information for change and to assess if plans implemented to reduce air pollution will be achieved, continuous monitoring combined with modelling and predictive techniques are required.

There are two main objectives to this thesis

1. To investigate the factors that affects the measured level of PM_{10} at a busy city centre traffic intersection.
2. To investigate the suitability of an empirical model (persistence factors) in predicting the 24-hour average PM_{10} concentration value.

The following steps have been identified to achieve the above objectives.

- Measure and assess the level of ambient PM_{10} in a built up urban area.
- Investigate the effects of local topography on the regional meteorological conditions.
- Evaluate the temporal characteristics of the pollutant level.
- Assess the effect of motor vehicles on the measured level of PM_{10}.
- Analyse the effects of the local meteorological conditions on the measured pollutant levels.
- Assess how representative the measured levels are of the true pollution level.
- Model the relationship between the measured pollutant level, meteorological conditions and pollutant source.

- Use the compiled information of the effects of meteorology and source on the level of PM_{10} to develop persistence factors as a method of predicting PM_{10}.
- In developing this method identify its suitability to the prediction of the average daily PM_{10} concentration.

This thesis is set out as follows

- Chapter 2 presents the literature review, which provides background information on particulate air pollution. The policies and standards governing particulate air pollution are reviewed along with measurement and assessment methods.
- Chapter 3 describes the monitoring programme and the monitoring equipment employed.
- Chapter 4 illustrates the measurements obtained from the monitoring programme.
- Chapter 5 investigates the relationships between PM_{10} levels and meteorological variables and pollutant source, i.e. motor vehicles.
- Chapter 6 describes the development of the persistence factors and investigates the suitability of this method for predicting PM_{10}.
- Chapter 7 concludes this study.

Chapter 2. Literature Review

This chapter covers four main points. Namely, airborne particulate pollution, current policies and standards, a review of current methods employed in monitoring and assessing particulate air pollution and finally air pollution modelling and forecasting techniques.

2.1 Particulate Matter

Airborne particulate matter consists of a mixture of solid and liquid particles suspended in the air. Particulate matter is defined as any substance, except pure water, that exists as a liquid or solid in the atmosphere under normal conditions and is microscopic or sub-microscopic size, but larger molecular dimensions (about 2Å), (Seinfeld, 1986). Particulate matter is unique in its complexity; it is not a single pollutant but rather a mixture of many subclasses of pollutants with each subclass containing many different chemical species (US-EPA), 1996). A full description of atmospheric particles requires specification of not only their concentration but also their size, chemical composition, phase (liquid or solid) and morphology.

Atmospheric particles originate from a variety of sources and possess a range of morphological, chemical, physical and thermodynamic properties. An important distinction in relation to the origin of particulate matter, they may be of primary or secondary origin (Harrison *et al.*, 1997, Kleeman and Cass, 1998).

- Primary Sources: particulates that are emitted directly into the atmosphere.
- Secondary Sources: particulates that are formed in the atmosphere as a result of chemical or physical transformations.

Examples of particulate matter which affect ambient air quality include combustion-generated particles such as diesel soot, fly ash, photo chemically produced particles such as those found in urban haze, salt particles formed from sea spray, and soil-like particles from suspended dust.

Atmospheric particles contain inorganic ions and elements, elemental carbon, organic compounds, and crustal compounds. The organic fraction is especially complex, containing hundreds of organic compounds. (US EPA 1996, Wrobel *et al.*, 2000)

2.1.1. Particulate Characteristics

Particle sizes span more than four orders of magnitude, from a few nanometers to one hundred nanometers (US-EPA, 1996). Combustion generated particles, such as those from power generation, automobiles and in tobacco smoke, can be as small as 0.003 μm and as large as 1μm. Particles produced in the atmosphere by photochemical processes range in diameter from 0.003 to 2 μm. Fly ash produced in the atmosphere by coal combustion ranges from 0.1 to 50 μm or more. Wind blown dust, pollens, plant fragments and cement dusts are generally above 2μm in diameter. Particles as small as a few nanometers and as large as 100 μm have been measured in the atmosphere (Covert *et al.*, 1992).

2.1.2. Particle Size

The main determinant of the behaviour of an atmospheric particle is its size and chemical composition (Environmental Protection Agency (EPA), 1996). Particle size is usually expressed in terms of its aerodynamic diameter, which refers to the unit density spherical particles with the same aerodynamic properties such as the falling speed. In practice the aerodynamic diameter is very similar to the geometric diameter as might be measured with a light or electron microscope (Seinfeld, 1986).

The atmospheric deposition rates of particles, and therefore, their residence time in the atmosphere, are strong functions of particle size. Size influences deposition patterns of particles within the lung (Fenger, 1999, Harrison *et al.*, 1997). Light scattering is strongly dependent on particle size having a strong influence on atmospheric visibility, through their affect on radiative balance on climate (US EPA, 1996).

Once aerosols are in the atmosphere, their size, number and chemical composition are changed, "aging" by several mechanisms until ultimately they are removed from the atmosphere by natural processes (Seinfeld, 1986). Such aging effects include gas to particle conversion processes (Kleeman and Cass, 1998), adsorption or desorption of semi-volatile material on or from particle surface (Weingarter *et al.*, 1997). These processes change the physical and chemical properties of the particles, which have an influence on the lifetime of the particles in the atmosphere (Weingarter *et al.*, 1997).

The particle size of which this study is concerned with is that of 10 μm aerodynamic diameter, normally referred to as PM_{10}. PM_{10} is that fraction of particulate matter having an aerodynamic diameter of 10μm or less. A definition of PM_{10}: particulate matter which passes through a size-selective inlet with a 50% cut-off at 10 μm aerodynamic diameter (US EPA, 1996). It is a well-documented fact that particles in this size fraction have the greatest impact on human health (US EPA, 1996, Fadel and Massoud, 2000, Harrison *et al.*, 1997, Fenger 1999). Atmospheric PM_{10} are also referred to as atmospheric aerosols. An aerosol describes any liquid or solid particles that are so fine they remain suspended indefinitely. The haze that exists in many industrial areas can be described as an aerosol. Other terms encountered are grit, which are small, hard, incombustible pieces of material produced by coal burning ranging from 500 μm down to 76 μm. Also dust ranging from 75 to 1 μm and fume below 1 μm aerodynamic diameter (US EPA, 1996).

2.1.3. Particle Size Distributions

Complex size distributions of airborne particles in ambient systems result from a multiplicity of sources generating particles and as mentioned above different composition and size characteristics (Morawska *et al.*, 1999).

Usually particles are grouped into three modes; ultrafine, fine and coarse (Fenger, 1999). The ultrafine particles are chemically formed or condensed from hot vapour e.g., from diesel exhaust and coagulate into fine particles (Whitby and Severdrup, 1980 as reported by US EPA, 1996).

Fine particles can often be roughly divided into two modes, nuclei mode and accumulation mode.

- The nuclei mode: This mode extends from about 0.005 to 0.1μm in diameter, and accounts for the preponderance of particles by number. But because of their small size, these particles rarely account for more than a few percent of the total mass of airborne particles. Particles in the nuclei mode are formed from condensation of hot vapours during combustion processes and from nucleation of atmospheric species to form fresh particles.
- The accumulation mode; This mode extends from 0.1 to about 1μm in diameter, usually accounts for most of the aerosol surface area and a substantial part of aerosol mass. The source of particles in the accumulation mode is the coagulation of particles in the nuclei mode and from condensation of vapours onto existing particles, causing them to grow into this size range. The accumulation mode is so named because particle removal mechanisms are least efficient in this regime, causing particles to accumulate there (Seinfeld, 1986).

The coarse mode extends from 1 to 100μm in diameter, such particles are formed by mechanical processes and usually consists of man-made and natural dust particles (Morawska *et al.*, 1999)

Whether or not nuclei and accumulation modes are present, the existence of a bimodal distribution with fine and coarse modes has been clearly demonstrated in atmospheric measurements (Seinfeld, 1986, Morawska *et al.*, 1999).

Studies in which chemical composition has been demonstrated as a function of particle size also demonstrate the division between fine and coarse modes and show a difference in chemical composition of the two modes. On the basis of such studies, it is possible to divide the major chemical species observed in atmospheric aerosols. The major components of the fine fraction of the atmospheric aerosol are sulphate, ammonium, nitrate ions, lead, carbon containing material including soot and condensed organic matter. Several studies have shown that toxic species, such as polynuclear aromatic compounds, As, Se, Cd and Zn are more concentrated in the fine particle fraction.

The coarse fraction consists mainly of crustal material, such as Fe, Ca and Si. The major sources are wind erosion products, primary emissions, sea spray and volcanic eruptions. (Seinfeld, 1986).

2.1.4. Classification of Particles

The aerosol community uses a number of different approaches in the classification of particles by size (US EPA, 1996). Such approaches are based on observed size distributions and formation mechanisms and include the coarse mode, fine mode, accumulation mode and nuclei mode discussed above. Dosimetry is another approach used, which is based on entrance into various compartments of the respiratory system. Here size fraction definitions are based on human health significance (US EPA, 1996). This approach classifies particles into:

- Inhalable Particles: Particles that enter the respiratory tract, including the head airways region.
- Thoracic Particles: Particles that reach the lung airways and the gas-exchange region of the lung.
- Respirable Particles: Particles that reach the gas exchange region of the lung.

2.2 Sources of Airborne Particulates

Particles are ubiquitous in the atmosphere. The lowest concentrations are found in background marine environments, where particle concentrations range from 100 cm^{-3} to 400 cm^{-3}. In the background continental environments, particle concentrations vary from 100 cm^{-3} to 5,000 cm^{-3}, while in urban areas of the United States concentrations may be as high as 4,000,000 cm^{-3}, (Willeke and Whitby, 1975, Whitby and Severdrup, 1980 as reported by US EPA, 1996). Particles account for a mass of a few $\mu g\ m^{-3}$ near surface over dry continental areas to several hundred $\mu g\ m^{-3}$ in polluted urban areas.

Airborne particulate matter can be anthropogenic or natural in origin, and as previously noted, can occur from primary and secondary sources. Particles of natural origin include wind-blown soil and sea spray, tend to be concentrated in the coarse particle fraction (>2.5μm) (Harrison *et al.*, 1997). Significant natural sources include soil and rock debris, volcanic action, sea spray, wild fires and reactions between natural gaseous emissions.

Anthropogenic sources are classified as stationary and mobile ones. In recent years a general downward trend in aerosol emissions from stationary sources has been observed in most of the developed countries (Wróbel *et al.*, 2000). Primary sources of PM_{10} from stationary sources include combustion sources such as domestic fires, industrial processes such as construction works, pharmaceutical industries, power stations, oil storage depots (EPA, 1996). These stationary sources are significant contributors to PM_{10} emissions and they account for about one half of PM_{10} emissions in the UK (EPA, 1996). However, in cities, these stationary source contributions have become quite small, in London, they account for about 5% of all PM_{10} emissions (QUARG 1996 as reported by Harrison *et al.*, 1997).

Secondary particulates from stationary sources include the formation of particles within the atmosphere from condensation of vapours or as a result of chemical reaction processes. Major sources of secondary particulates include the oxidation of sulphur dioxide to sulphuric acid and the oxidation of nitrogen dioxide to nitric acid (EPA, 1996). These secondary particles are mainly concentrated in the fine particle fraction (<2.5 μm), (Harrison *et al.*, 1997).

Many industrial sources and burning of carboniferous fuels produce PM_{10} (EPA, 1996). Combustion is a complex process that produces distinct particles in a number of ways including vaporisation followed by condensation to yield particles in a particular size range (Bridgeman H., 1990 as reported by EPA 1996).

Mobile source emissions are represented by transportation. In the urban environment this source contributes the most to levels of PM_{10}. The UK emissions inventory shows road transport to be a major source of PM_{10} (25%), other significant contributions result from public power (15%), commercial, institutional and residential combustion plants (16%). The large contribution which transport makes to PM_{10} is emphasised by the emission inventory for London. This shows road transport to be responsible for 80% of PM_{10} emissions, with medium-sized lorries accounting for nearly quarter of the total (QUARG 1996 as reported by Harrison *et al.*, 1997).

Studies in the U.S. have examined PM_{10} and $PM_{2.5}$, (Chow *et al.*, 1996a). Over the course of this study it was found that $PM_{2.5}$ contributed between 30 and 70 % of the PM_{10}. The author also concluded that meteorology was an important factor controlling concentrations of both $PM_{2.5}$ and PM_{10}. Characterising of the sources of PM_{10} in Santa Barbara County, California in 1989 using a chemical mass balance receptor model, found motor vehicles to be the largest source of PM_{10}, accounting for 30 to 42 % of PM_{10} mass concentration, (Chow *et al.*, 1996b). A similar study in Birmingham, U.K., estimated that vehicle exhaust emissions contribute an average 32% of PM10 and 41% of $PM_{2.5}$ during a six month winter study period (Harrison *et al.*, 1997). Also in winter months $PM_{2.5}$ contributing about 80% of PM_{10}, in summer months $PM_{2.5}$ contributing 50% of PM_{10}.

Prior to 1996 there were no comprehensive measurement of the concentrations of PM_{10} particulate species in urban air in Dublin. In 1996 The EPA undertook a baseline study of PM_{10} and VOCs in Dublin. In this study at city centre locations experiencing high volumes of traffic some 31% of measurements for 1996 exceeded the maximum 24-hour average limit value of 50μg m^{-3} (EPA, 1996).

2.2.1. Particulate Emissions from Transport

A typical engine exhaust gas can comprise a particulate emission of up to 4,000μg m^{-3} (Butler, 1979). Motor vehicles emit about 500 different compounds (Fenger, 1999). There are two categories of transport emissions sources:

(i) Vehicle related particles from tyre, clutch and brake wear.

(ii) Particles generated from vehicle exhaust.

Particulate material from vehicular sources range in size from large flakes to submicron particles and varies in consistency from hard and brittle to oil mists (Seinfeld, 1986). Mass and composition fall into two groups; coarse particles (> 2.5μm) and smaller fine particles, $PM_{2.5}$. Particles emitted by vehicles include unburnt hydrocarbons, oxygenated hydrocarbons, and inorganic species such as So_2, No_2, H_2So_4 (EPA, 1996). Some of the particles are generated in the engine combustion chamber, and nucleated and agglomerated in the vehicle exhaust pipe. Particles formed in the engine can also settle on interior surfaces, to flake off at a later stage and be emitted with exhaust gas (Seinfeld, 1986).

Diesel vehicle emissions emit approximately 30 to 70 times more PM_{10} than petrol vehicles equipped with catalytic converters and burning unleaded petrol (Wiengartner *et al.*, 1997, Walsh M.P. 1987 as reported by EPA, 1996). Previously lead particles were a problem with petrol-fuelled vehicles, since the change over to unleaded petrol levels of lead have been reduced to very low levels (EPA, 1996). Petrol fuelled automobiles are now fitted with catalytic converters in which platinum-group elements (PGEs) are the active components eliminating several noxious components from exhaust fumes. Studies suggest that PGEs have become a major source of urban pollution (Palacios *et al.*, 2000).

An exponent rise in environmental concentrations has been observed as the number of cars fitted with catalytic converters has grown (Hodge, 1986 as reported by Palacios *et al.*,2000). The mobility of PGEs on/from the catalyst surface is the most important factor for the increase in their environmental background level and can be mainly attributed to thermal sintering, evaporation and mechanical or thermal erosion, (Stenbom, 1994 as reported by, Palacios *et al.*, 2000).

About 40% of particles from tyre wear are less than 10μm (20% less than 1μm) and are primarily carbon. Particles from brake linings are less than 1μm in diameter and are composed mainly of asbestos and carbon. Carbonaceous particles in the atmosphere consist of two major components: graphite or black carbon (elemental or free carbon) and organic material, black carbon originating only from combustion processes. (Novakov, 1982 as reported by US EPA, 1996)

2.3 Factors Affecting The Ambient Concentration of Particulate Matter

The resulting ambient air quality in the urban environment from various sources depends not only on the quantity of emissions, but also on the external factors subsequent to the moment of emission. Factors such as mixing induced by vehicle motion, wind speed and direction, turbulence, chemistry and rate of pollutant removal, and the geometry of buildings and roads, all affect ambient concentration of a pollutant (Berkowicz *et al.*, 1996, Theurer, 1999). Thus affecting the pollutant level to which the general population is exposed.

The Role of Meteorology and Topography

Infrastructure and town planning determine emission patterns, through the siting of industrial areas, traffic routes and urban development. Meteorology and topography determine the extent of dispersion and transformation of pollutants emitted from these sources. Both of these factors have a critical effect on pollutant concentration and topographical features of an area have an effect on the microscale meteorology. This has been well documented by many authors (Croxford *et al.*, 1996, Harrison *et al.*, 1997, Scaperdas et al., 1999, Mayer, 1999).

Many of the major air pollution disasters of the past have been associated with calm weather conditions, where prevailing conditions did not allow for dispersal of pollutants. Such disasters include:

- The Meuse Valley Incident, Belgium, December 1930

  The topography of this region is characterised by a valley shaped like a trench, with steep sides about 400ft deep and about 1km wide. The population density is high, with much heavy industry. At the time of the incident very prolonged, stable weather conditions (anticyclonic inversion and fog) caused cold air to drain down into the valley, therefore emissions from heavy industries and domestic fires could not disperse. This resulted in at least sixty deaths due to respiratory illnesses and up to 6000 reported respiratory related illnesses.

- London Smog Disaster, December, 1952

  As with the previous example prolonged stable weather conditions and high levels of smoke and So_2 from domestic coal burning resulted in prolonged smog.

  This caused between 3,000 and 4,000 deaths. This incident led to an effort to control domestic air pollution in the UK by means of implementation of the Clean Air Act 1956. In 1991, London experienced another major air pollution incident during prolonged stable weather conditions. On this occasion the pollution was from vehicular sources, resulting in elevated levels of particulate and NOx. 160 deaths were reported.

- Dublin Smog Incident, January 1982.

  At this time alarmingly high levels of smog were witnessed in Dublin. A twofold increase in deaths resulted as a result of smog related illnesses. This incident was one of the factors that led to the implementation of The Air Pollution Act, 1987.

  (IT, Sligo, 1999)

2.3.1. The Effect of Meteorology

Meteorology affects the dispersion, dilution, transformation and transport of air pollutants. Meteorological scales of motion can be categorised into three main scales of systems.

- Macroscale / Synoptic; Scales of 100 to 1000 km and greater
- Mesoscale; Scales of 10 km up to 100km
- Microscale; Scales of up to 1 km and 10 km.

Each of these scales of motion plays a role in air pollutant dispersion and transformation over different periods of time. Microscale takes place over scales in the order of minutes to hours, where the initial break up and dilution of pollutant occurs. Mesoscale may influence transport and dispersal over time scales of hours to days. In this scale chemical transformations may occur. Synoptic scales of motion occurring over times of days to weeks, long-range transport of pollutant occurs. (Zannetti, 1990, Seinfeld, 1986).

The region of the atmosphere governing dispersion and transport of pollutants is the planetary boundary layer this is roughly the lowest 500m of the atmosphere. It represents the extent of influence of the earth's surface on wind structure in the atmosphere (Zannetti, 1990).

The following are some of the meteorological conditions that play a role in pollutant dispersion and transformation.

- **Wind**

  Wind Speed determines the degree to which pollutants will be dispersed and diluted once emitted into the atmosphere. In general high wind speeds results in low pollutant concentrations and vice-versa (Harrison *et al.*, 1997). Wind can also play another role other than mixing and dispersal with respect to PM_{10} concentrations. Under certain wind conditions higher wind speeds can result in re-circulation and re-suspension of particles (Lam et al., 1999)

  Wind Direction determines the initial transportation of pollutants from their source. The population downwind of the source are those that are most affected by the pollutant. If the wind is blowing from a point source directly to a receptor, a change in wind direction as little as 5 degree can lead to reductions in concentration, at the receptor in the order of 10% under unstable conditions, 50% under neutral conditions and 90% under stable conditions (Boubel, 1994).

  Ireland and the rest of Western Europe are influenced by predominantly southwesterly winds, which brings moist air from the sea and favours long-range transport of pollutant. In the northern part of Europe, the small amount of sunlight favours persistent inversions with poor dispersion conditions. In central and Eastern Europe, high pressures with air stagnation and accumulation of local pollution are frequent. (Fenger, 1999)

- **Precipitation and Relative Humidity**

  Precipitation has a significant effect on particles. Precipitation reduces particulate mass concentration by washout through agglomeration with raindrops (Harrison *et al.*, 1997). Different opinions exist on the effect of relative humidity on particle size. Some studies have indicated that relative humidity (Monn *et al.*, 1997) affects PM_{10} concentration in the ambient air, whereas in others (Lam *et al.*, 1999), very little correlation was found. At relative humidity below 85% freshly emitted combustion aerosols do not absorb water (Weingarter *et al.*, 1995).

- **Temperature**

  Ambient air temperature in urban areas affects level air pollutants through its demand on fuel consumption. At times of lower ambient temperature demands on space heating increase (Fenger, 1999). Air temperature is inter-related with atmospheric turbulence and stability as discussed below.

- **Turbulence**

  Turbulence is concerned with the highly irregular motions of the wind. Turbulent flows are irregular and random, so that velocity components at any location vary randomly with time (Seinfeld, 1986). Turbulence results in wind gusts and many eddies superimposed on the overall mean wind flow and are quite difficult to measure precisely (Seinfeld, 1986). These eddies are mainly generated from topographical features, buildings and changes in temperature. These gusts may aid in dispersal and dilution, or in some cases, bring about re-circulation of particles (Lam *et al.*, 1999).

  Two types of turbulence occur:

  1. Mechanical Turbulence: This is induced by vertical shear from horizontal flow. Eddies are small and relatively constant in size, have a high frequency and occur in the lowest km of the atmosphere.
  2. Thermal Turbulence: Here eddies are large and variable in size and they have a low frequency. (IT, Sligo, 1999)

- **Atmospheric Stability**

  In air pollution terms atmospheric stability is a measure of the atmospheres ability to disperse pollutants. A stable atmosphere will dampen the movement of an air parcel, whereas an unstable atmosphere will enhance the movement of a displaced air parcel (IT, Sligo, 1999). More stable conditions result in a build up of pollutant levels, as discussed previously with various air pollution incidents.

  There are three broad categories of atmospheric stability based on the atmospheric temperature profile:

  1. Neutral: This occurs when the environmental lapse rate (ELR) is equal to the dry adiabatic lapse rate (DALR). That is that the rate of cooling is equal to 1°C/100m, and a parcel of air at any height has no tendency to rise or fall.
  2. Stable: This occurs when the ELR is less than the DALR, under such conditions air parcels are inhibited from upward and downward motion. Dispersion of pollutants is restricted.
  3. Unstable: This occurs when the ELR is greater than the DALR, under such conditions air parcels at any height are displaced either upward or downward, and will continue their movement in the direction in which they were displaced. Unstable conditions are the most favourable for dispersion and dilution of pollutants.

     (Seinfeld, 1986, Zannetti, 1990).

Mixing height quantifies the vertical mixing in the atmosphere. Mixing heights are estimated from measurements of the vertical temperature profile, a radiosonde is sent into the atmosphere and temperature profiles at various altitudes are radioed back. The altitude at which the dry adiabatic line intersects the radiosonde measurements is taken as the maximum mixing height. The mixing height is a function of atmospheric stability. Under unstable conditions the mixing height is higher than it is under stable conditions (IT Sligo, 1999). This results in seasonal variations in mixing height, generally over summer months the mixing height is higher than it is in winter. Often pollution measurements are higher during the winter months, this is not necessarily a result of increased emission of pollutants, but that the mixing height is lower. Therefore pollutants are trapped in a smaller volume with less opportunity for dispersal and dilution (Morel *et al,* 1999)

- **Temperature Inversions**

Temperature inversions occur when the vertical temperature profile increases with height, resulting in cooler temperatures nearer to the ground level. An inversion results in poor vertical and lateral mixing of pollutants (Zannetti, 1990), with pollutants trapped under the inversion layer. Inversions generally occur at night-time under dry clear weather conditions. Inversions can be prominent in the early morning prior to sunrise, when the ground is at its coolest and there is minimal wind creating turbulence to mix the colder air with the warmer air above. Once the sun rises the ground warms and the inversion gradually disappears (Crosgrove, 1997).

2.3.2. The Effect of Topography

The region of the atmosphere governing the transport and dilution of air pollution is the planetary boundary layer, mentioned above. This layer is not the same as the mixing height. The depth of the boundary layer depends on the roughness of the surface over which the wind is flowing. This surface will have a frictional effect on the wind speed nearest the surface. The depth of the boundary layer is deeper for large surface roughness (urban areas) and shallower for small roughness (grassland, water) (Zannetti, 1990). The gradient wind occurs at the height above the surface where the effects of the surface are no longer felt. Thus the degree of ground turbulence in urban areas is much greater than in rural areas (UK Department of the Environment, Transport and the Regions (UK DETR), 2000a).

Another difference between urban and rural areas is that of air temperatures. Buildings in urban areas are composed of such materials as steel and concrete that heat quickly and retain heat. Subsequent re-radiation of this heat at night-time maintains warmer air temperatures, with urban areas remain warmer than surrounding rural areas (IT Sligo 1999, Seinfeld, 1986). Under conditions of light winds, this warm air rises and cooler air is drawn in. Subsidence replaces this air resulting in a closed unconditioned circulation, known as the urban heat island. This island traps and re-circulates pollutants within the urban area. (IT Sligo, 1999, Seinfeld, 1986, Zannetti, 1990).

The presence of buildings and tall structures in built up urban areas affects the wind flow around urban street-building configurations. This has an important influence on the micro-scale dispersion of vehicular pollution. Affecting the overall dilution and creating localised spatial variation of pollutant concentration (Scaperdas *et al.*, 1999). This type of urban configuration is known as a street canyon, a relatively narrow street between buildings that line up continuously along both sides (Sini *et al.*, 1996). In such street canyons a vortex re-circulation pattern is set up, creating a small-scale spatial variation of concentration, characterized by significant differences in concentration between the leeward and wind ward sides of the canyon (Berkowicz *et al.*, 1996, Croxford *et al.*, 1996).

Air quality measurements made at different urban sites may be strongly influenced by the way the street-building geometry interacts with the wind flow and the effects on pollutant emission and dispersion at a localized scale. (Scaperdas *et al.*, 1999, Sini *et al.*, 1996).

2.4 Impacts of Particulate Matter

The exact health and environmental impacts of various pollutants are still uncertain. Air quality is affected by a large number of different pollutants, both particulates (liquid and solid) and gases. Reports suggest that health hazards due to individual pollutants be increased by the combined pollution in the environment (Fenger, 1999).

2.4.1. Health Impacts

Suspended particulate matter is viewed as the most important air pollutant in terms of health (El-Fadel and Massoud, 2000). As already noted, particles vary widely in chemical composition and toxicity. The depth of penetration into the lungs is dependent on particle size. The physical properties of particles affect the transport and deposition of particles in the human respiratory system and chemical composition determine their impact on health (El-Fadel and Massoud, 2000).

Several high quality epidemiological studies from the US and Europe have indicated a strong link between increased PM_{10} concentration and increased mortality and morbidity as a result of short-term exposures (Harrison, 1997). The most important size fractions causing health effects are that of the thoracic particles (aerodynamic $<10\mu m$) and fine particles ($<2.5\mu m$) (Monn *et al.*, 1997, El-Fadel and Massoud, 2000). Figure 2.1 presents a schematic of particulate deposition in the human respiratory system. The ultrafine and fine particles, which are predominantly of anthropogenic origin are deposited in the lower parts of the human respiratory system therefore, having the greatest impact (Fenger, 1999, El-Fadel and Massoud, 2000).

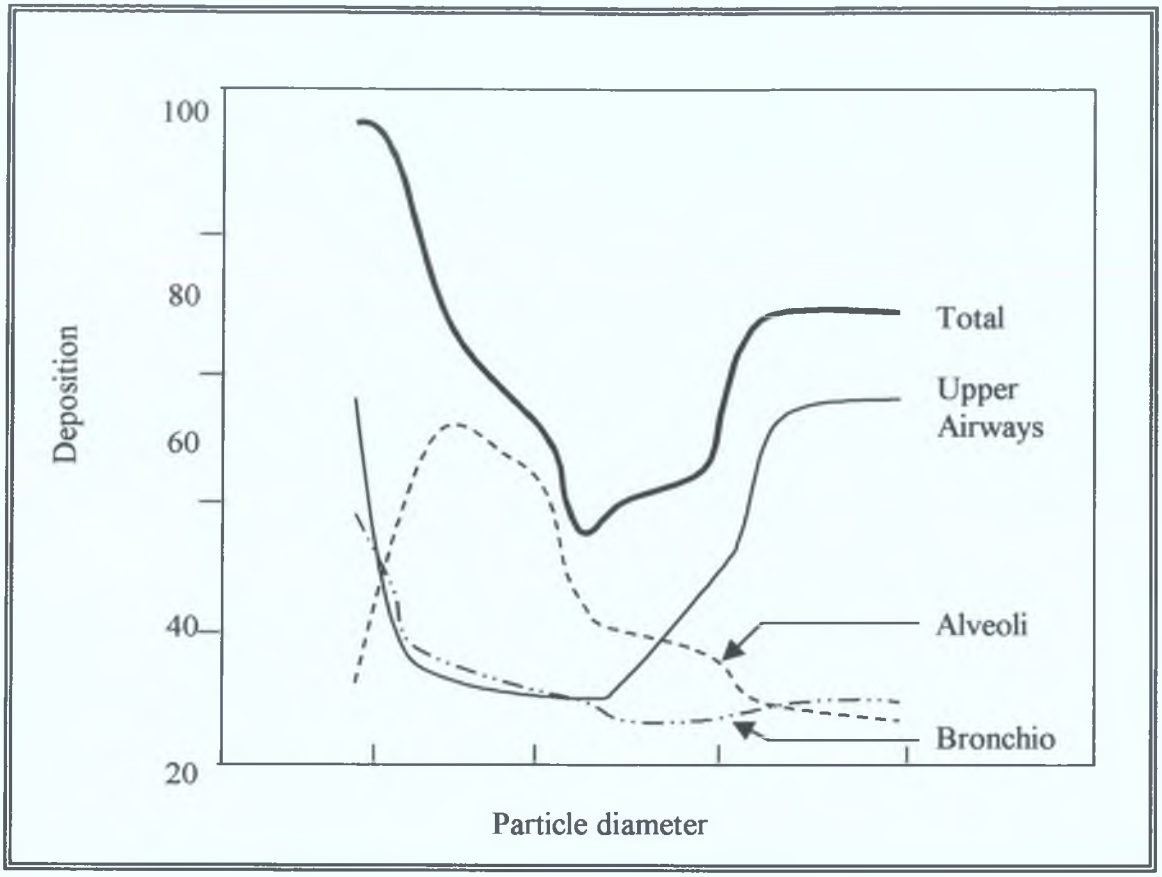

Figure 2.1 Size distributions of Particulate Matter and the Corresponding Deposition in the Human Respiratory System (Fenger, 1999).

Natural sources are quantitatively important as contributions to the total load of particulates in the atmosphere, their environmental and health implications are generally considered to be relatively small. Far more important are those derived from manmade sources, in particular in the urban environment, namely transportation (Fenger, 1999).

Many studies have shown there is a relationship between PM_{10} concentrations and various health indicators, such as increased hospital admissions, frequency of respiratory illness, reduced lung function and death. Analysis of epidemiological data from USA has suggested a rise of 10μg m^{-3} in PM_{10} levels is accompanied by an increase in the relative risk of mortality of about 1% in the exposed population. The PEACE (Pollution Effects on Asthmatic Children in Europe) study that was carried out in 10 European countries suggests that increased PM_{10} levels are followed by an increase in hospital admissions for respiratory conditions. Increased PM_{10} levels also lead to symptom exacerbation among children with chronic respiratory symptoms. (US EPA, 1996)

Bronchitis is often associated with long-term exposure to particulate laden air. Bronchitis is a lung condition characterised by a cough and difficulty breathing. Continual exposure to particulate laden air leads to chronic inflammation of the bronchial tubes and excess mucous production that initiates a cough (US EPA, 1996).

Particulate matter deposited on lung may remain in lung tissue acting as an irritant and may affect structure of lung over a period of time e.g. asbestos, metal fumes (beryllium). Particulate matter that is absorbed from the lungs into the bloodstream can cause damage elsewhere in the body. The significance of these particles is that they may contain absorbed material that are biologically and biochemically active e.g. heavy metals, lead, mercury and cadmium (US EPA, 1996).

2.4.2. Impact on Local Climate

The atmosphere is the source of the earth's weather and climatic systems. When air pollutants are emitted intro the atmosphere they have the potential to cause climatic change. Air pollution effects seen at both local and global effects. Local climatic changes are caused by a number of air pollutants; the most significant of which is particulate matter (Seinfeld, 1986). Airborne particles reduce the intensity of sunlight at ground level and this can cause local climatic change. This may occur due to a number of reasons as presented in figure 2.2.

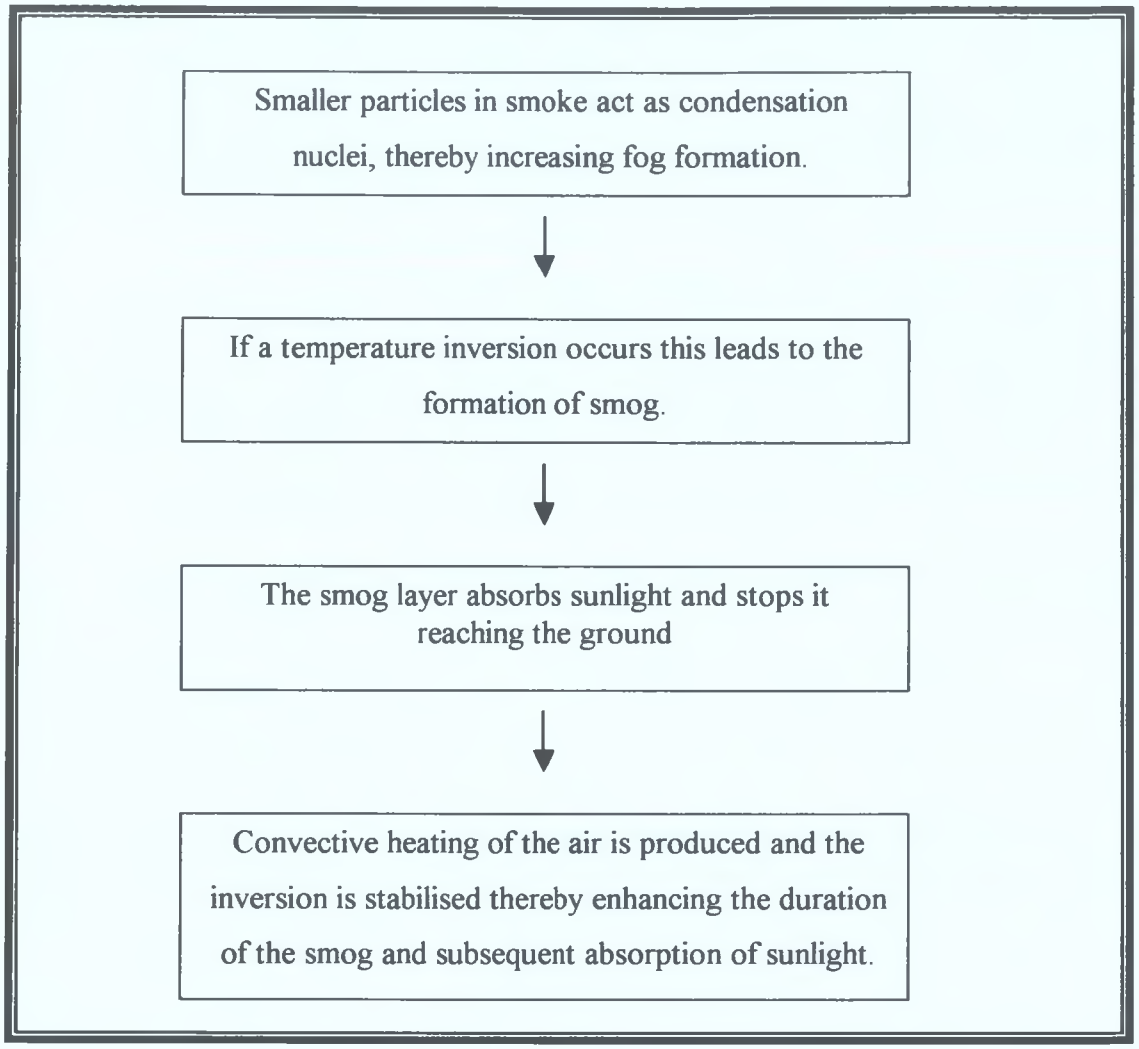

Figure 2.2 Impact of Particulate Pollution on Climate (IT, Sligo, 1999)

2.4.3. Impact on Materials

The deposition of airborne particles on the surface of buildings and culturally important articles can cause damage and soiling, thus reducing the life usefulness and aesthetic appeal of such structures (Baedecker *et al.*, as reported by US EPA, 1996). The presence of particles on surfaces may exacerbate the physical and chemical degradation of materials that normally occur when these materials are exposed to environmental factors such as wind, sun, temperature fluctuations and moisture (US EPA, 1996).

2.5 Policies and Standards

Human health is the main concern in the regulation of urban air quality (Fenger, 1999).

2.5.1. Air Quality Standards

Based on clinical, toxicological and epidemiological evidence, guidelines values of ambient particulate concentrations were established by determining concentrations with the lowest observed adverse effect and adjusted by an arbitrary margin of safety factor to allow for uncertainties in extrapolation (El-Fadel and Massoud, 2000). Generally, the most frequently used reference guidelines for particulate pollution are those set by The World Health Organisation (WHO), the European Union (EU), and the United States Environmental Protection Agency (USEPA). While WHO guidelines are based on health considerations only. Standards determined by the EU and US EPA reflects the technological feasibility of attainment as well. Many countries adopt these guidelines or else they establish their own ambient air quality standards to achieve the same overall objective (El-Fadel and Massoud, 2000).

2.5.2. Irish Policies and Standards

There are a number of statutory instruments in place in the Irish system for the control of emissions from domestic, vehicular and industrial sources. Prior to the Air Pollution Act 1987 there was no specific legal system for control of air pollution. Some control was exercised by the following.

- **Alkali etc. Works Regulation Act 1906**
- **The Public Health (Ireland) Act, 1878**
- **Local Government (Planning and Development) Act, 1963**
- **Local Government (Sanitary Services) Act, 1962**

  Two sets of regulations have been made under this section 10 of this act:

  1. **The Control of Atmospheric Pollution Regulations, 1970**
  2 **The Control of Atmospheric Pollution (Licensing) Regulations, 1985**

- **Regulations under the European Union (EU) Communities Act, 1972**

These regulations provided for reducing the sulphur content of gas oil, the lead content of leaded petrol and the benzene content of unleaded petrol

- **European Union Directives**

In addition to the above several other European Union Directives relating to air pollution were in existence prior to the Air Pollution Act, 1987 that set air quality objectives to control particulate air pollution.

- Council Directive 80/779/EEC on air quality limit values and guide values for sulphur dioxide and suspended particulates
- Council Directive 82/884/EEC on limit value for lead in air.

2.5.3. Air Pollution Act, 1987

The Air Pollution Act, 1987 provided a comprehensive framework for the control of atmospheric and pollutants. It enabled full effect to be given to various pieces of EU legislation on air pollution including Directive 84/360/EEC on Combating of Air pollution from Industrial Plants and the various Directives in relation to ambient air quality previously mentioned.

Parts of this act deal specifically with ambient air quality;

- Part IV deals with Special Control Areas, Air Quality Standards and Emission Limit Values.

  i. Special Control Areas: These are used as means of restricting the use of certain fuels to control air emissions affecting local air quality in specified areas. The initial idea of introducing Special Control Area Orders was as a means of controlling emissions from bituminous coal burning in domestic dwellings.

  ii. Air Quality Standards (AQS): Specifies the maximum allowed concentration of substances allowable in ambient air. Council Directive 80/779/EEC sets limit and guide values for Sulphur Dioxide and Suspended Particulates.

  iii. Emission Limit Values: State the maximum pollutant level that can be emitted into the environment, normally set as part of licence conditions.

- Part V deals with Air Quality Management Plans

  Under Section 46 of the act local authorities may, and shall if the Minister so direct make an Air Quality Management Plan (AQMP) in relation to all or part of their functional area. An AQMP must contain objectives to prevent and limit air pollution or to preserve air quality within the area to which the plan relates.

  (IT, Sligo, 1999, Dublin Corporation, 1999).

2.5.4. Air Quality Framework Directive

Council Directive 96/62/EC of 27 of September 1996 on Ambient Air Quality Assessment and Management provides the framework for future legislation on air quality within the Europe Community.

The objectives of this directive are to:

- Define and establish objectives for ambient air quality in the European Community that will avoid, prevent and reduce harmful effects on human health and the environment.
- Assess ambient air quality in Member States on the basis of common methods and criteria
- Obtain adequate information on ambient air quality and ensure that it is made available to the public.
- Maintain ambient air quality where it is good and improve it in other cases.

The principle of the European Commissions assessment and management approach can be described in figure 2.3. Each country will be divided into zones. The monitoring, assessment, management and reporting of air quality will be undertaken in relation to these zones and to limit values, assessment thresholds and margins of tolerance to be set for the listed pollutants. Alert thresholds will also apply for some of the pollutants for the purpose of taking immediate action to inform the public of particular pollution incidences as they occur.

For each individual pollutant, the limit values will have to be met everywhere by a specified attainment date. A margin of tolerance, which decreases in equal annual increments to zero at the attainment date, will apply in the worst affected areas where concentrations are considerably in excess of the limit value. In such cases specific action plans will be needed to bring the concentrations below the limit value by the attainment date. Details of these action plans and their effects are part of the reporting obligations related to the framework directive.

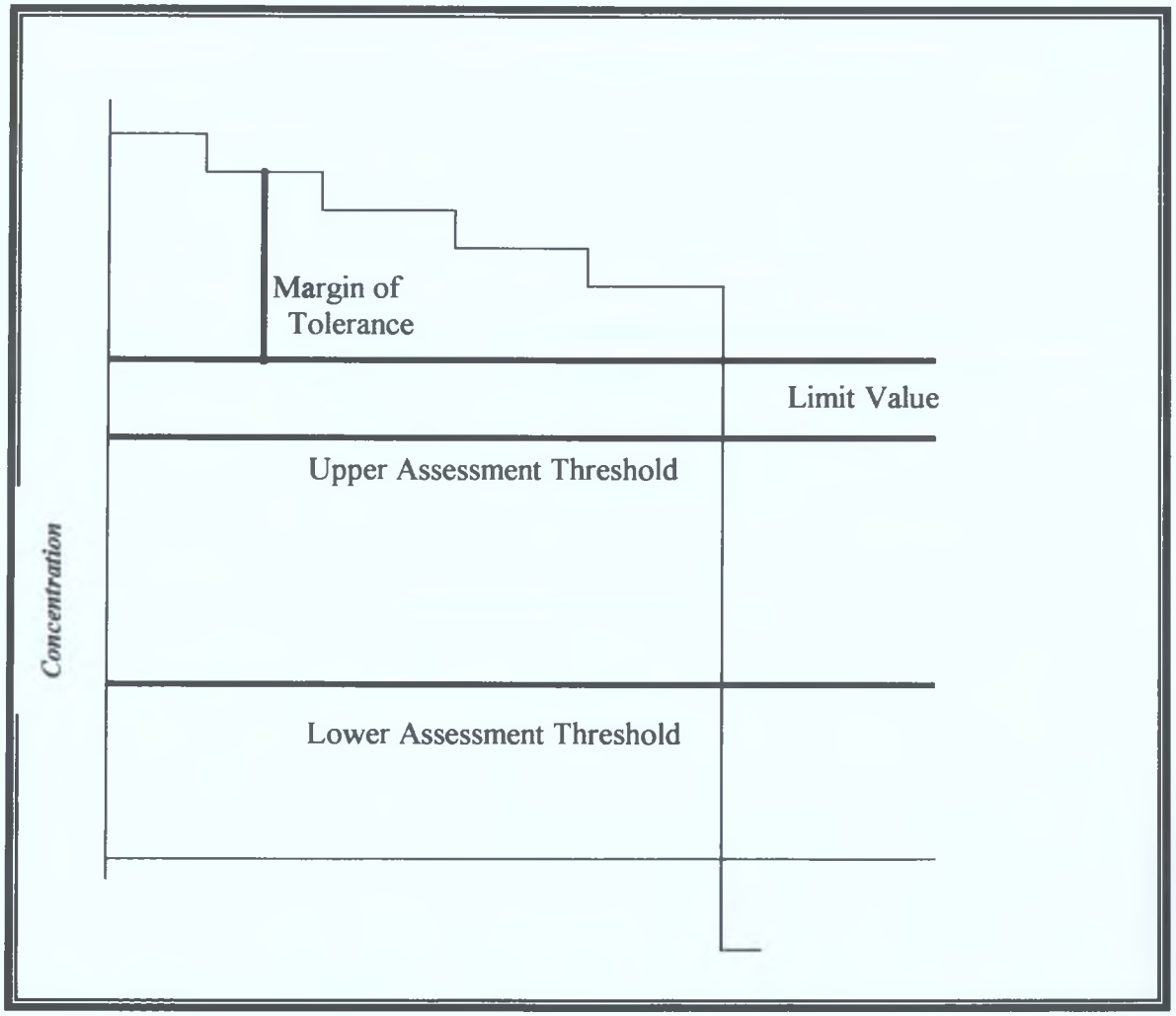

Figure 2.3 Limit Value, Assessment Thresholds and Margin of Tolerance (EPA, 1997)

Monitoring and assessment of pollutant levels are major requirements of this directive. The extent of which in any zone will be determined mainly by population size and air quality status. Monitoring is mandatory wherever concentrations are above the lower assessment threshold and the greatest monitoring effort applies if concentrations are above the upper assessment threshold. Where concentrations are between the two thresholds, less intensive measurement combined with other assessment methods (e.g. air quality modelling) will suffice.

Daughter Directive on $S0_2$, NOx, Particulate Matter and Lead

The first daughter directive, adopted in April 1999 covers sulphur dioxide, nitrogen oxides, particulate matter and lead. Following the concepts in the Framework directive, this directive sets out limit values, margins of tolerance, assessment thresholds and other parameters against which monitoring, assessment and management of these pollutants are to be undertaken. Limit values, margins of tolerance and assessment thresholds are summarised on Table 2.1.

The Framework Directive requires that this daughter directive should:

i. Set limit values for the protection of human health and ecosystems, including the attainment dates by which they should be met.
ii. Specify, as appropriate, the temporary margins of tolerance to apply during the period between the coming into force of the Directive and the limit value attainment date.
iii. Define upper and lower assessment thresholds for the determination of the assessment requirements applicable in each zone.
iv. Set alert thresholds, where appropriate, and list details to be supplied to the public if an alert threshold is exceeded.
v. Specify the minimum number of monitoring stations
vi. Specify the criteria and techniques for measurement
vii. Set out criteria for use of other techniques for assessing ambient air quality, such as dispersion modelling.

Table 2.1. Limit Values, Attainment Dates, Margins of Tolerance for PM_{10} (EPA, 1997)

Pollutant	**PM_{10} Stage 1**	**PM_{10}-Stage 1**	**PM_{10}-Stage2[A]**	**PM_{10}-Stage2[A]**
Limit Value Objective	Protection of Human Health	Protection of Human Health	Protection of Human Health	Protection of Human Health
Averaging Period	24-Hours	Calender Year	24-Hours	Calender Year
Limit Value ug/m^3	50	40	50	20
Basis of Application of Limit Value	No more than 35 exceedences in a calender year	Annual Mean	No more than 7 exceedences in a calender year	Annual Mean
Limit Value Attainment Date	01-Jan-2005	01-Jan-2005	01-Jan-2010	01-Jan-2010
Assessment Thresholds[b] Upper:	30ug/m^3	14ug/m^3	To be derived	To be derived
Lower:	20ug/m^3	10ug/m^3	To be derived	To be derived
Margins of Tolerance[c]	25ug/m^3	8ug/m^3	To be derived	10ug/m^3

A: indicative limit value to be reviewed in the light of further information on health and environmental effects

B: to be applied on the same basis as the corresponding limit value except in the case of PM_{10} where they are based on the indicative limits for 2010.

C: on entry into force of the Directive and reducing to zero by the attainment date

The primary particulate form, for which limit values are proposed is PM_{10}. Measurement of $PM_{2.5}$ will also be required. It is now generally accepted that measurements of these particle fractions are better indicators of the health effects of particulate air pollution, than those provided by the black smoke or total suspended particulate methods.

The objective of keeping the general public adequately informed about the state of air quality is a particularly important aspect of the directive. Information on ambient air quality is to be made available on a daily basis via broadcast media, computer network services or other suitable means. (Source EPA, 1997, 1998)

2.6 Monitoring and Assessment of PM_{10}

Assessment of health risks associated with airborne aerosols implies that measurements be made. Measurements defining the aerosol characteristics, concentrations and exposures that contribute to or simply correlate with, adverse health effects (US EPA, 1996). The assumption that fixed-location measurements are representative of inhalation exposure implies that the effects of local spatial and temporal gradients are understood and appropriately applied to the sampling siting criteria (Spengler *et al.*, 1994 as reported by US EPA 1996). Development of relationships between aerosol characteristics and health or ecological responses requires that the aerosol sampling and analysis processes are truly representative and adequately defined.

2.6.1. Measurement of Particulate Matter

Air borne particle monitoring instruments share a number of common features. A pump draws air of a known flow rate through a validated sampling head, which selects a particular size fraction of the ambient airborne particles. The collected particles are then analysed in a number of ways. The mass of particles can be determined gravimetrically and can also be determined directly by continuous methods. Continuous assessment may take the form of, the change in a property of the filter due to the presence of the particles or passing of the filter through an optical sensing device (UK DETR, 2000b).

Because of the diverse nature (chemical and physical) of particles in the atmosphere, there are many possible measurement procedures. As with other pollutants, such as So_2 may be measured by several methods, but in principle all should yield the same measurement of concentration. Many different techniques are also available to quantify different properties of the airborne particles, and a number of different methods are available for expressing the mass concentration of airborne particulate matter. This can provide problems in that conversion from one metric to another is rarely possible, as the relationships are site-and-time specific (UK DETR, 2000b). On the other hand this variety can offer opportunities for selection of the metric that will best describe the property of the particles being investigated.

2.6.2. Black Smoke Method

Historically particulate matter was measured using the British standard smoke stain method, BS 1747:Pt1969. (Black Smoke). This method has the advantage of simplicity and low cost and has been used extensively in national networks (EPA, 1996). This has resulted in a large body of data being available for the measurements of smoke and particulate mass loading.

The method is sensitive only to light absorbing components of particulate matter, which tend to be dominated by carbon in its elemental form, present in soots from combustion processes. These particles tend to be predominantly of submicrometer size and will not contribute completely to the total particle mass. The black smoke method generally underestimates the PM_{10} mass concentration typically by a factor of 2 to 4 than that obtained gravimetrically (EPA, 1996). As this method only accounts for coloured particulate matter, this can lead to differences in winter/summer measurements, due to the predominance of lighter coloured particles being collected in summer (EPA, 1996). Levels of particulates measured using this method have decreased in Dublin City following the introduction of regulations to ban the use of bituminous coal in the city since 1990. This method when introduced in 1960s was suited to the coal smoke pollution of the time (QUARG, 1993 as reported by UK DETR, 2000b). Many of the components that contribute to PM_{10} nowadays do not absorb visible light in particular secondary particles such as ammonium sulphate and ammonium nitrate. They cannot be measured by this method i.e. the transport generated particulates. As the nature of particulate material has changed since then this method does not provide a reliable quantification of the mass concentration of particles in the atmosphere, but reflects the contribution of combustion sources, particularly diesel vehicles. The Black smoke method has been shown to approximate $PM_{3.2}$ and is essentially a measurement of diesel emissions (Reponen *et al.*, 1996).

2.6.3. Gravimetric Methods

Gravimetric methods of particle measurements are based on weighing the mass of particles collected by drawing a known volume of air through a filter paper for a specified time period.

- **Measurement of Total Suspended Particulate Matter (TSP)**

Measurement of TSP is performed using a High Volume Sampler (HiVol) that samples air at a rate of 1-2 cubic metres a minute, thereby collects milligram quantities of particles over a 24-hour period. TSP includes all particles with diameters <100 μm, this measure is an unsatisfactory indicator of health effects as only particles <10 μm are expected to penetrate into the tracheo-bronchial system and TSP samplers can sample particles sizes in excess of 10 μm (Simpson, 1990 as reported by EPA, 1996). Furthermore investigation of the inlet characteristics of the TSP sampler shows them to be highly sensitive both to orientation with respect to wind direction and to wind speed. Particles in excess of 20 μm diameter being sampled efficiently under some conditions, whilst under others, efficiency was poor even for particles below 10 μm diameter (Wedding *et al.*, 1977 as reported by UK DETR, 2000b). This HiVOL sampler was designated as the reference method in 1971 for measurement of particulate matter, which collects particulate matter up to a nominal size of 25 or 45 μm (US EPA, 1996).

- **Size Selective Samplers**

The poor characteristics of the TSP sampler led the US EPA to promote the development of size selective samplers. This led to the adoption of a PM_{10} sampler (US EPA, 1996). This sampler has a symmetrical inlet, the properties of which are independent of wind direction and wind speed, which excludes particles by having a 50% cut point at 10μm, thus sampling all particles with an aerodynamic diameter of 10μm or less. From this PM_{10} is defined as particulate matter that passes through size selective inlet with a 50% efficiency cut-off at 10 μm aerodynamic diameter.

The original PM_{10} inlets were designed for use with high volume samplers, they are now available for samplers operating at lower volume rates. These samplers can also be fitted with inlets that incorporate a cyclone giving a 50% cut–off at 2.5 μm or 1μm. This allows for collection of size fractions $PM_{2.5}$ and PM_1 (US EPA, 1996).

- **Partisol PM_{10} Sampler**

The Partisol PM_{10} sampler (manufactured by Rupprecht and Patashnick Co. Inc.) is one such size selective PM_{10} sampler; it is now the US EPA reference method for PM_{10} mass concentration. This instrument is used by the Irish EPA and also in the UK as part of the national monitoring network. The Partisol sampler is a low flow gravimetric PM_{10} sampler. Particles are collected over a 24-hour period onto a glass fibre filter to yield a 24-hour average PM_{10} mass concentration value. Filters can be retained if further chemical analysis is required. Sampling periods, of week, month or year, are also used, in assessing long-terms trends and compliance purposes. This instrument is also used to measure $PM_{2.5}$, by using a $PM_{2.5}$ size selective inlet (Rupprecht and Patashnick Co. Inc, 1998)

2.6.4. Continuous Measuring Methods

Continuous measuring equipment has the advantage over gravimetric methods in that they provide real time information of pollutant levels as they occur. The Beta (β)-Gauge Monitor is one such example of a continuous method (EPA, 1996). Direct monitoring demonstrates the levels a person is exposed to can vary drastically during the day, and that ambient, outdoor air quality, regulated by various limit values do not describe the overall load, even in industrialised countries (Fenger, 1999). The most commonly used continuous instrument for PM_{10} measurements is the Tapered Element Oscillating Membrane (TEOM), manufacture by Rupprecht and Patashnick Co. Inc (UK DETR, 2000b).

- **Tapered Element Oscillating Membrane, TEOM**

The TEOM was designated by the US-EPA in October 1990 as an equivalent method for measurement of 24-hour mean PM_{10} for compliance purposes (US EPA, 1996). The TEOM is a frequently used and cost-effective method for measurement of PM_{10} when daily or hourly concentrations are needed for regulatory purposes, atmospheric chemistry studies, source apportionment modelling, or epidemiological studies of human health effects (Allen G. *et al.*, 1997). The TEOM instruments were adopted for use in the UK network, because of their ability to measurement data in real time, as compared to measurements from traditional gravimetric methods, which are available only after the period required for collection and weighing (UK DETR, 2000b). This instrument has the ability to produce a 30-minute, 1-hour, 8-hour and 24-hour average. As with the Partisol sampler, this instrument samples by means of a size-selective inlet that can be adapted to sample $PM_{2.5}$ and PM_1.

In theory such continuous measurements should be directly equivalent to those of a gravimetric procedure using the same sample inlet characteristics, but in practice divergence occurs in the presence of semi-volatile components of particulate matter (UK DETR 2000b, Allen *et al.*, 1997). This is because the TEOM, as used in national networks, pre-heats the sample air stream to 50°C prior to particle collection in order to drive off associated particle bound water (PBW) and other components that might lead to inconsistent and variable mass measurements. This causes loss of some semi-volatile particulate components (e.g. ammonium nitrate and some organic compounds). Removal of PWB is critical to the operation of the TEOM, which is based on the measure of incremental mass, because particulate matter is accumulated on the oscillating filter up to two to three weeks before a new filter is used. If collected particulate matter is not heated or desiccated sufficiently to remove all of the water, any change in the ambient water vapour concentration will produce a change in PBW, which will in turn change the collected particulate matter on the TEOM filter (Allen *et al.*, 1997).

A number of studies have investigated this problem. It is well documented by a number of authors, that the TEOM records slightly lower concentration values than those determined by reference sampler methods (Allen *et al.*, 1997, King e*t al.*, 2000, UK DETR 2000b, EPA, 1996). Factors have been determined in the United States, to achieve US EPA certification (EQPM-1090-079), a re-scaling factor of 1.03 and a +3μg m^{-3} offset are applied to make average allowances for lost material (King e*t al.*, 2000). The difference in measurements between the two methods appears to depend upon the composition of the particulate matter, the sampling site and season, and no universally applicable correction is available (UK DETR 2000b). In measuring PM_{10} the divergence is typically of the order of 20%, but rarely exceeds 40%. At the Marylebone Lane monitoring site in The UK a study investigated these difference between the TEOM and gravimetric methods (UK DETR 1998a and 2000b). The TEOM PM_{10} results were multiplied by the 1.3 re-scaling factor, this revealed that using the 1.3 factor overestimates the number of exceedences compared to the gravimetric sampler by 45% (UK DETR 2000b).

An evaluation of the TEOM method by Allen *et al.*, 1997, revealed good agreement between TEOM and manual gravimetric methods during summer months and not such good agreement during winter months. Differences were also noted between different monitoring sites, sites dominated by vehicular contributions and industrial emissions, particles at such sites should contain a large fraction of semi-volatile compounds. A comparison of the TEOM and gravimetric methods by Salter *et al.*, 1999 also showed good agreement between the two methods. In this study the measurements were made in an area where levels of volatile organic compounds were low, this may explain to some extent the difference between the TEOM and manual gravimetric methods.

For the purposes of this study the TEOM is the more appropriate method. Real time monitoring is required to assess the effect of changing traffic volumes and varying meteorological conditions on the PM_{10} concentration on an hourly basis.

2.7 Methods for Predicting Ambient PM_{10} Levels

Environmental evaluation from air pollution can take the form of two approaches. The more comprehensive would consist of large scale atmospheric monitoring carried out at a wide range of sites and maintained for a long duration at each. This involves significant expense, both initial and recurrent. Also it is difficult to make future projections or evaluate abatement strategies from historic pollutant concentrations alone as future conditions change. (Seinfeld, 1986, Zannetti, 1990)

The second option is to develop either a physical or mathematical model of the problem and conduct a more limited programme of monitoring. Development of models to predict air pollution is necessary for many reasons

1. Establish emission control legislation
2. Evaluating emission control techniques and strategies
3. Select locations of future sources of pollution
4. Planning the control of air pollution episodes
5. Assessing responsibility for existing air pollution levels.

A physical model can be a small scale laboratory representation of phenomena e.g. a wind tunnel, water tank. A mathematical model consists of a set of analytical algorithms that describe the physical and chemical aspects of the problem. (Benarie, 1980, Zannetti, 1990).

There are two types of mathematical models:

- Deterministic Models: These models are based on a fundamental mathematical description of atmospheric processes in which effects are generated by a cause e.g. a dispersion model. The output from such models i.e. pollutant concentration is computed from mathematical inputs which include, source composition, emission rates and meteorological conditions (Zannetti, 1990)
- Statistical Models: These models are based on semi-empirical statistical relations among available data and measurements. An example of a statistical model is a forecast, which is that concentrations of pollutants over the next few hour/days can be given as a statistical function of 1. Currently available measurements and 2. Past correlation between these measurements. (Zannetti, 1990, Benarie, 1980).

Potentially these are the most accurate method as they are based on real time data. However they are not always used due to the lack of suitable data with which to create the models. Large quantities of simultaneously monitored pollution, meteorological and traffic parameters are generally not available.

Mixed deterministic-statistical models are also available. Some dispersion models are based on statistical diffusion theories and the performance of a statistical model is always improved when some deterministic information is included in its structure (Zannetti, 1990).

2.7.1. Dispersion Models

Dispersion modelling is very commonly used in evaluating air quality from a variety of sources. The Gaussian plume model is one of the most commonly used dispersion models (UK DETR, 2000a, Zannetti, 1990). This is based on a simple formula that describes the three-dimensional concentration field generated by a point source under stationary meteorological and emission conditions. Pollutant emissions from road traffic can be predicted using dispersion models by treating the emission as a line source. A large car park or airport area can be modelled by treating the emission as an area source. Pollutant emission rates are calculated from emission factors. These emission factors are based on fuel type and class of vehicle and are expressed as weight of pollutant emitted per distance travelled. These emission factors are input into the model, along with meteorological data, and information on the local topography to calculate the ambient pollutant level. Such emission factors are updated as changes to fuel composition and vehicle performance are introduced e.g. low sulphur diesel fuel, the introduction of catalytic converters (US EPA, 1995).

Dispersion models require accurate input data in order to accurately predict pollutant values. The concentration of pollutant predicted by most models is directly proportional to the pollutant emission rate, if all other parameters are constant (Zannetti, 1990). There is a hierarchy of models in increasing order of complexity, choice of model is dependent on the level of assessment required and the availability of input data.

Some models may only require regional wind speeds while others require very accurate details of a great number of meteorological parameters and topographical features. Inputs of topographical features may be as simple as urban or rural inputs or they may require exact building dimensions. (UK DETR, 2000a).

The least complex of dispersion models are screening models. Design for Roads and Bridges (DMRB) and CAR are two such screening models. Such models are used to give an indication of air quality near roads. Intermediate models include such models as Assessing the Environment of Locations in Urban Streets (AEOLIUS) and Operational Street Pollution Model (OSPM). Both of these models can predict pollutants within a street canyon and at the kerbside. AEOLIUS is used to calculate NOx, NO2, CO, SO2, PM_{10} and 1,3-butadiene. Advanced models include California Line Source Model (CALINE), this model includes a photochemistry model and can handle up to 20 road links and 20 receptors. (UK DETR, 2000a). The US EPA PART5 particulate emission factor model is used for the analysis of the particulate air pollution impact of gasoline-fuelled and diesel-fuelled motor vehicles (US EPA, 1995).

2.7.2. Statistical Models

The major distinction between deterministic and statistical models is that deterministic models initiate their calculations at the pollution source and aim to establish a cause to effect relationship (Zannetti, 1990). The most complicated statistical model is based on a group of observations and is essentially empirical (Benarie, 1980). Statistical models are characterised by their direct use of, air quality measurements to infer semiempirical relationships. Such models include statistical forecasts and empirical models. These models are developed using methods that include some of the following:

- Frequency Distributions
- Time Series Analysis
- Mixed Deterministic and Statistical Techniques
- Receptor Modelling Techniques

Frequency distributions are commonly used to assess the probability of occurrence of high concentration values. The measurement of the probability distribution density allows for the prediction of extreme events and predictions of violations of an air quality standard (Georgopoulos and Seinfeld, 1982). Selecting an appropriate frequency that best fits air quality measurements is a powerful tool to analyse pollution data and design mitigation strategies (Morel *et al.*, 1999). Several frequency distributions have been proposed and used to fit air quality measurements including, log-normal, gamma and weibull frequency distributions. (Zanetti, 1990, Seinfeld, 1986).

Time series analysis may include spectral analysis, regression analysis, trend analysis and principal component analysis (Zannetti, 1990). These methods can be used in batch or real time mode. Batch simulations perform statistical analysis of past measurements in order to establish empirical relationships. Real-time applications require the availability of on-line data and provide forecasts that can be used for real-time intervention strategies, in an effort to mitigate incoming pollution episodes. Real-time applications have been investigated by Pérez *et al.*, 2000, to predict several hours in advance of the occurrence of high $PM_{2.5}$ pollution episodes in Santiago, Chile. Spectral analysis techniques allow the identification of cycles in meteorological and air quality time-series measurements (Zannetti, 1990).

Time series methods can be applied in two modes, fitting mode and forecasting mode The fitting mode tends to overestimate model performance (Zannetti, 1990). In this mode the same set of measurements is used to estimate the model parameters (e.g. regression coefficients) and model performance evaluation. Whereas in the forecasting mode one set of measurements are used to estimate model parameters and the model is then applied to a different set of measurements to evaluate performance (Benarie, 1980).

Regression analysis is a type of multiple time-series analysis in which independent variables (e.g. meteorological measurements) are statistically related to air quality concentrations, which are considered the dependent variable. Trend and seasonal variations can be assessed using multiple regression models. Principal component models have also been used to calculate pollutant distributions and predictions. (Zanetti, 1990, Benarie, 1980).

Receptor Modelling can be considered the opposite to dispersion modelling. A receptor model starts with the observed ambient pollutant level at the receptor and seeks to apportion the observed concentration among several source types based on known chemical compositions of source and receptor materials (Zannetti, 1990). Such models have been developed in the UK for predicting future concentrations of PM_{10}, benzene and 1,3-butadiene for comparison with air quality standard objectives (Stedman and Dore 1998a, 1998b). The receptor modelling technique for predicting PM_{10} levels enables the measured daily PM_{10} concentration at a monitoring site to be divided into three components, primary combustion particles, secondary particles and other particles. Either black smoke measurements or oxides of nitrogen measurements are used as an indicator for primary combustion particles and rural sulphate measurements are used as an indicator for secondary particles. The difference between these measurements determines the concentration of other particles. (Airborne Particle Expert Group (APEG), 1998, Stedman, 1997 as reported by Stedman and Dore 1998b). This receptor model enables relatively sophisticated estimates of both annual mean and high percentile PM_{10} concentrations for future years (Stedman *et al.*, 1998b).

Persistence Factors

Dispersion models output pollutant concentrations as average hourly values because meteorological conditions are likely to remain constant over this period (UK-DETR, 2000a). 15-minute mean and 8-hour mean are less usual averaging periods. In order to model different time periods according to air quality standards quite often data (e.g. traffic data) does not exist for 8-hour and 24-hour periods. Many dispersion models only predict traffic data for peak and inter-peak hours, measurements of traffic data indicate that traffic activity and composition varies over a 24-hour period (Reynolds, 2000). Thus estimating concentrations for every hour involves considerable computer processing and input data to generate varying time-averaged pollutant concentrations. The use of persistence factors is one such method of predicting varying time-averaged concentrations of pollutants (Cooper, 1989). Persistence factors are used to convert one-hour values to other time period averages. The persistence factor method has the advantage over modelling sequential hours of data in that it includes the effects of both traffic and meteorological conditions (Cooper, 1989).

Cooper *et al.*, (1992), recommend using a persistence factor of 0.55 to convert 1-hour carbon monoxide to 8-hour concentration in Orlando, Florida. This value is based on comprehensive statistical analysis of 10 years of meteorological data, coupled with detailed hourly modelling results.

The UK-DETR (1998b) state that the maximum 8-hour mean for carbon monoxide within urban areas is unlikely to exceed the air quality objective of 10 ppm if the maximum 1-hour concentration is less than 12 ppm. This is equivalent to a conservative persistence factor of approximately 0.83. The UK-DETR (1998b) also state that the 15-minute mean for So_2 within urban areas is approximately 1.36 times the 1-hour mean.

This study will investigate if persistence factors are applicable to the prediction of the 24-hour average PM_{10} mass concentration.

Chapter 3. The Monitoring Programme

This chapter presents the details of monitoring that were undertaken for this study and descriptions of the equipment employed.

Continuous air pollution monitoring requires reliable equipment that can be left unattended for varying lengths of time. One of the major constraints to continuous air monitoring besides after cost considerations is locating a safe and secure site with adequate power supplies. Along with the security requirements, the monitoring site needs to be located in such a way that the measured level of pollutant is representative of the area as a whole (Croxford and Penn, 1998). The site at which this monitoring study was undertaken fulfills the facilitation of security and power supply. Analysis of the measured data in the forthcoming chapters will assess how well this site represents the level of pollution occurring at the particular location.

The 1998 EU common position (57/98) specifies criteria for monitoring sites, in that traffic-oriented samplers should be at least 25 meters from major junctions and should be no less than 4 meters from the centre of the nearest traffic lane (Dublin Corporation, 1999). At this monitoring site the location of sampling equipment meets these requirements, in that they approximately 4 meters from the nearest lane.

3.1 The Monitoring Site

Air quality monitoring was carried out at the intersection of Pearse Street and Westland Row in Dublin City Centre. This is a four-way intersection and a main artery through the city centre. This intersection is characterised by high volumes of traffic and high levels of congestion during peak hours. To the south and west of this intersection is Trinity college and to the north and east commercial buildings and residential housing. This is a busy pedestrian route owing mainly to the presence of Pearse Street Dart Station and the university. This intersection can be considered to be representative of many others in the city that experience a high flow of traffic.

Figure 3.1 presents a schematic of the intersection. Pearse Street has four traffic lanes, one of which is a bus lane. Westland Row and Lombard Street have three traffic lanes, one of the lanes in Westland Row is also a bus lane. Westland Row can be characterised as a street canyon, as it is a relatively narrow street with tall buildings either side. Pearse Street East and Lombard Street suffer less from the street canyon effect as the street is wider and the buildings are not as tall. Pearse Street West is the most open approach of the intersection with three storey buildings to one side and the open space of the northeast Trinity College car park to the other.

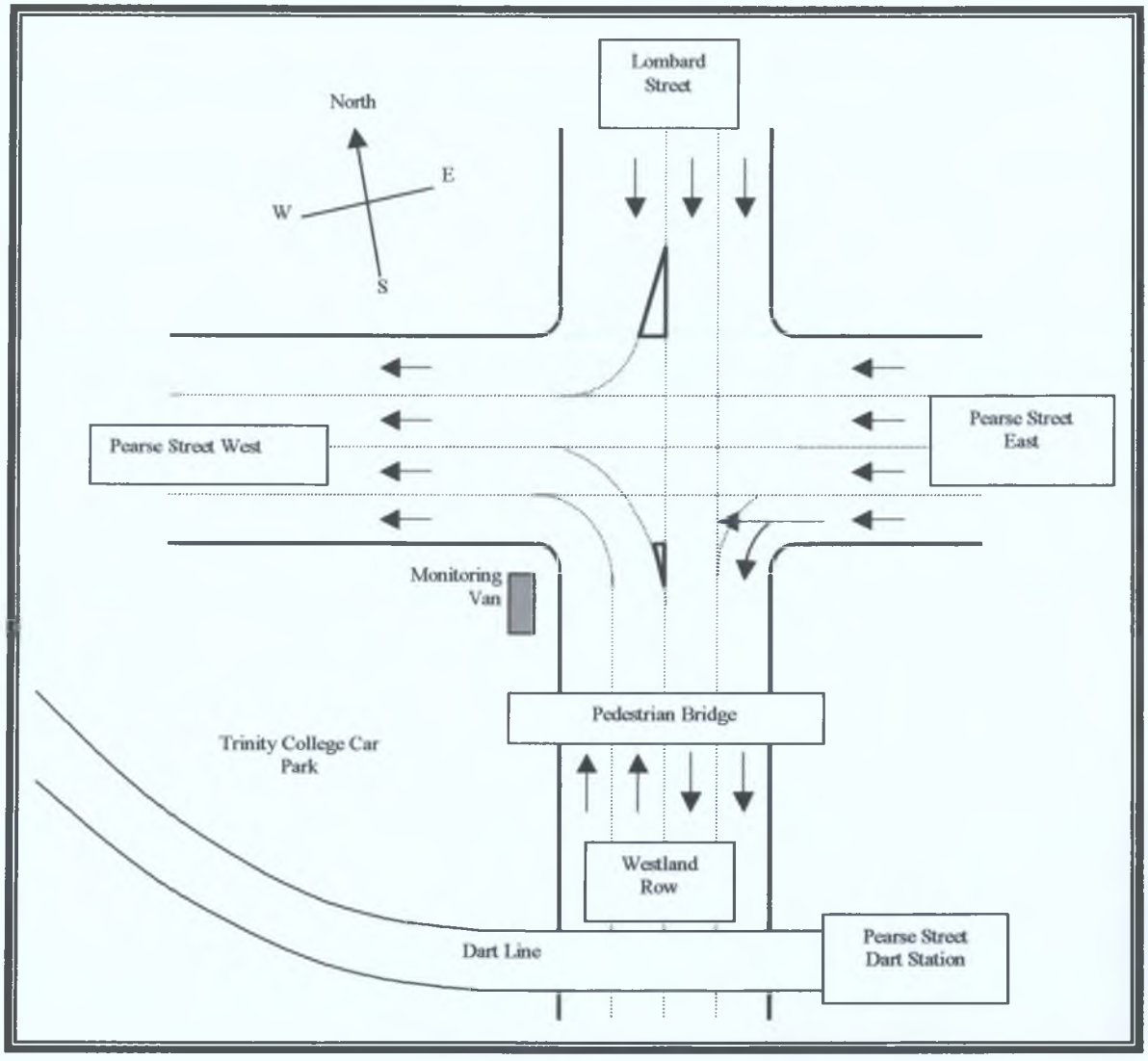

Figure 3.1 Schematic of The Monitoring Site

3.2 The Monitoring Programme

Monitoring of particulate air pollution (PM_{10}) and meteorological parameters was carried out using the Trinity College Mobile Air Monitoring unit. This was located in the northeast car park of Trinity College. This unit houses a number of items of air monitoring equipment and the meteorological equipment. The PM_{10} sampling inlet and meteorological equipment are located on the roof on this unit at a distance of 4 meters from the edge of the roadside and heights of 3 meters and 4.5 meters respectively. Traffic volumes were recorded by means of the SCATS urban traffic control system that is operated by Dublin Corporation.

The duration of the monitoring programme is presented in table 3.1. Data from time periods before 2000 was obtained from an earlier air monitoring study carried out by The Civil Engineering Department at Trinity College. Traffic volume data was source from Dublin Corporation by means of the urban traffic control system. Meteorological Data recorded at Dublin Airport was obtained from Met Eireann. This data was obtained to make comparisons between the regional and local meteorological conditions.

Table 3.1. The Monitoring Programme

Parameter	**PM_{10}**	**Meteorological Parameters**	**Traffic Volumes**
Time Period	Dec-97 to Mar-99	Dec-97 to Mar-99	* Jan-98
Time Period	Mar-00 to May 00 Sep-00 to Dec-00	Mar-00 to May-00 Sep-00 to Dec-00	
Time Period	Feb-01 to Apr-01	Feb-01 to Apr-01	Feb-01 to Apr-01

* Traffic volumes from January 1998 were measured for one week only.

3.3 Particulate Air Pollution (PM_{10}) Monitoring

Ambient levels of PM_{10} were measured using a Rupprecht and Pataschnick TEOM (**T**apered **E**lement **O**scillating **M**embrane) Series 1400a PM_{10} monitor. The TEOM PM_{10} monitor is a real-time device for measuring the concentration of particles smaller than 10μm diameter in outdoor and indoor ambient air.

The TEOM is composed of two major components

1. The TEOM Sensor Unit: This contains the mass measurement hardware and allows for continuous monitoring of accumulated mass on the system's exchangeable filter cartridge
2. The TEOM Control Unit: This houses the microprocessor system, flow control hardware, a gauge to determine filter lifetime, transformers and power supplies.

   A schematic of the TEOM system is presented in figure 3.1.

The ambient air is drawn through the PM_{10} size selective inlet at a flow rate of 16.7 l/min, at the exit of the PM_{10} inlet this flow is isokinetically split into a 3 l/min sample stream that is sent to the instrument's mass transducer (sensor unit) and a 13.7 l/min exhaust stream.

Inside the mass transducer the sample air stream passes through Teflon coated borosilicate glass filter that is attached to the tip of a hollow, tapered, oscillating, glass rod. The accumulation on the filter is measured every 2 seconds. The microprocessor computes from this weight the mass rate in g/sec. And finally the mass concentration in $\mu g\ m^{-3}$ by dividing the mass rate by the flow rate (corrected to standard temperature and pressure and expressed in $m^3\ sec^{-1}$).

Internal Temperatures: Temperatures are controlled to minimise the effects of changing ambient conditions. The sample stream is heated to 50°C so that the sample filter always collects under conditions of low constant humidity.

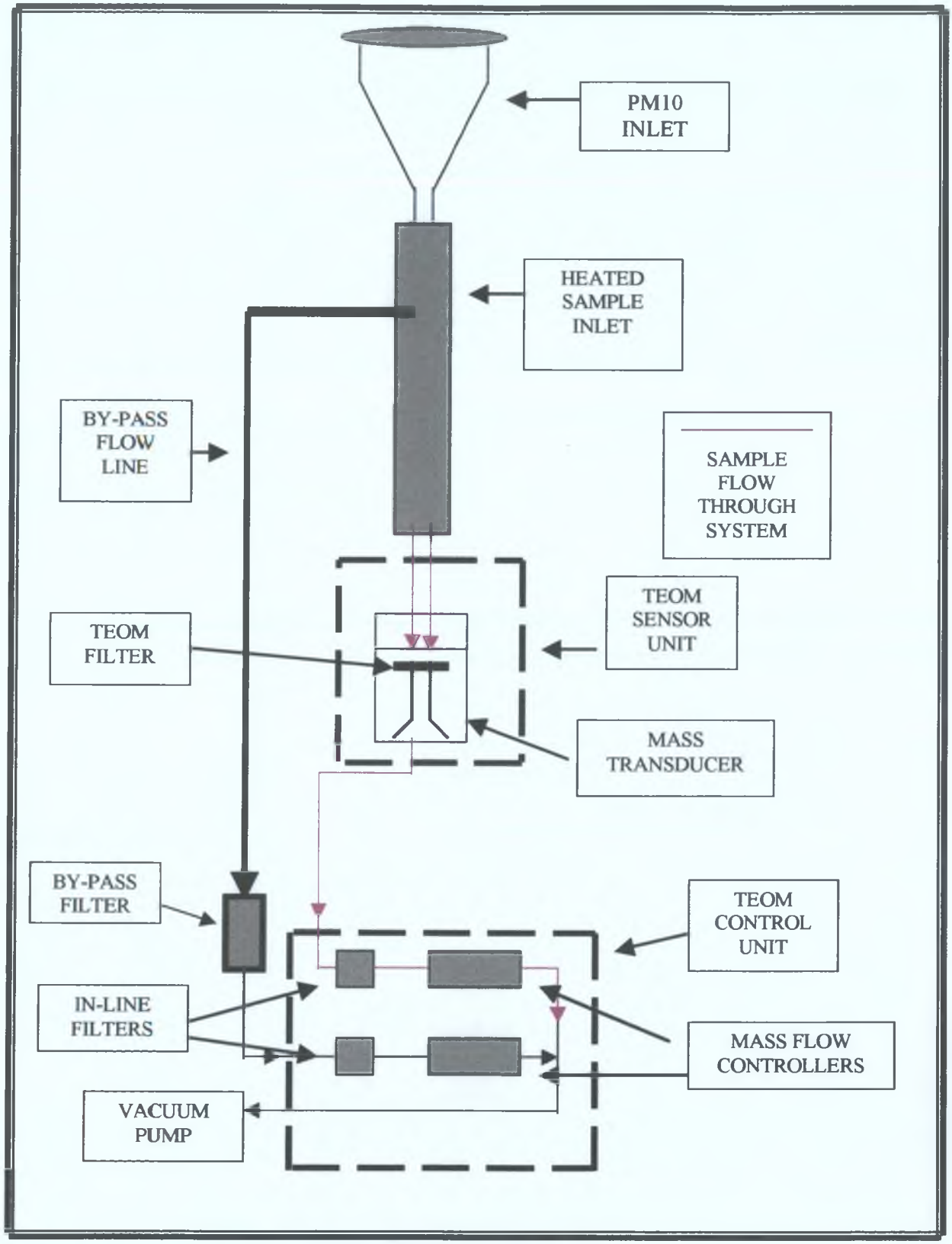

Figure 3.2 Schematic of flow system through the TEOM

(Source TEOM Operating Manual, 1993)

Flow System: Particle size separation takes place as the sample proceeds through the PM_{10} inlet. A flow splitter separates the total flow of 16.7 l/min into a main flow of 3 l/min that enters the sensor unit and an auxiliary flow of 13.7 l/min. The main flow passes through the exchangeable filter and then proceeds through an air tube and an in-line filter to the mass flow controller. The auxiliary flow is filtered in a by-pass fine particulate filter, then in an in-line filter and finally it enters a second mass flow controller. These mass flow controllers' control the flow through the system, maintaining constant flows and correct for local temperature and barometric pressure. A single pump provides the vacuum to draw the sample air stream through the system.

Mass Transducer Operation: The weighing principle used in the TEOM mass transducer consists of a tapered element, which is a hollow tube. This element is clamped at one end and free to vibrate at the other end. An exchangeable filter is placed over the tip of the element at the free end. The sample stream is drawn through the filter and then down the tapered element. The tapered element vibrates at its own natural frequency and is in essence a hollow cantilever beam with an associated spring rate and mass. When additional mass is added the frequency of vibration decreases. The electronic control circuit detects this change in the frequency of vibration. A precision counter measures the frequency with a two second sampling period, and the system the computes from this the mass concentration of PM_{10}.

The exchangeable filter is changed when the instrument indicates that its mass loading has been met. The time between filter changes depends on the ambient levels of PM_{10}, generally in urban areas filter exchange is required every 10 to 14 days. Data is stored within the instruments data logger, and is easily down loaded using a computer or modem

(Source, Rupprecht and Patashnick Co. Inc, TEOM Series 1400a Operating Manual, 1993)

3.4 Meteorological Parameters

The meteorological parameters measured are wind speed, wind direction, air temperature and relative humidity. Wind speed and wind direction are measured by a wind sensor system. This incorporates an anemometer for wind speed measurements reported in m/sec and a wind vane for direction measurements. Wind speed is recorded in meters per second (m/sec). Wind direction is recorded in degrees from true north. A combined temperature and humidity sensor measure air temperature and relative humidity. Air temperature is recorded in degrees Celsius (°C) and relative humidity is recorded as percent relative humidity (%RH).

These sensors are attached to a telescopic mast that is attached to the roof of the monitoring unit. These sensors are connected to a datalogger that records the meteorological parameters. Measurements can be logged over any time frequency, for the purposes of this study measurements were recorded every 15 minutes.

3.5 Traffic Measurements

Traffic volumes were obtained from Dublin Corporation by way of the SCATS (Sydney Co-ordinated Adaptive Traffic Sytems) urban traffic control system. The Roads Authority of New South Wales, Australia, developed this system, it is the most advanced and widely used urban traffic control system available today. It is a computer-based system incorporating hardware, software and traffic control philosophy. It operates in real-time, adjusting signal timings throughout the system in response to variations in traffic demand in order to optimise pedestrian, public transport and general traffic flows over the network. The system is software orientated and consists of a network of microprocessor based traffic controllers and detectors located at road intersections, with a detector in each traffic lane.

Each SCATS traffic controller is equipped with a built in modem that allows it to communicate with the SCATS central computer via a dedicated telephone line. SCATS measures traffic volumes and flows at intersections and the information gathered is continuously sent to the central computer, where the systems software can calculate the optimum method of running the intersection. The total volume of motor vehicles passing through the intersection is logged every 15 minutes.

The installation of this system in Dublin began in 1989 covering 32 sets of traffic signals in the city centre. Following a review in 1998, it has been decided to extend SCATS to every main route into the city. (SACTS Dublin Area Traffic Control System (Brochure)).

Chapter 4. Monitoring Results

This chapter presents the results of the monitoring programme. A summary of the PM_{10} data is presented here and it is compared with air quality standards. Details of the meteorological parameters and traffic volumes are also presented. The data presented here is analysed and interpreted in chapter 5 in order to assess the major trends in the data and to establish the relationships between the measured PM_{10} levels, the meteorological parameters and the pollutant source i.e. traffic volumes.

Monitoring of PM_{10} and meteorological parameters was carried out for varying time periods from December 1997 to April 2001, the times at which levels were measured and percentages data capture for these time periods is tabulated in table 4.1. In general data capture rates were quite good, data was unavailable during periods of instrument downtime, instrument calibration and periods when the monitoring unit was at other monitoring sites.

It should be noted that the measurements recorded in this study might slightly underestimate the levels of particulate pollution occurring at the intersection owing to the operating conditions of the TEOM instrument. The differences that occur between the TEOM and gravimetric methods have been previously discussed in section 2.6 of chapter 2.

Table 4.1 Percentage Data Capture of Monitoring Programme

Time Period	PM_{10} TEOM	Meteorological Parameters	Traffic Volumes
Dec-97	99%	93%	0%
Jan-98	90%	100%	13%
Feb-98	99%	100%	0%
Mar-98	100%	100%	0%
Apr-98	89%	100%	0%
May-98	81%	100%	0%
Jun-98	93%	100%	0%
Jul-98	100%	100%	0%
Aug-98	100%	100%	0%
Sep-98	100%	100%	0%
Oct-98	97%	100%	0%
Nov-98	100%	95%	0%
Dec-98	71%	100%	0%
Jan-99	81%	100%	0%
Feb-99	82%	100%	0%
Mar-99	97%	78%	0%
Mar-00	49%	100%	0%
Apr-00	80%	33%	0%
Sep-00	37%	70%	0%
Oct-00	100%	82%	0%
Nov-00	100%	20%	0%
Dec-00	50%	33%	0%
Feb-01	60%	60%	100%
Mar-01	100%	68%	100%
Apr-01	100%	68%	100%

4.1 Particulate Data.

A summary of the monthly and hourly statistics of the ambient PM_{10} levels, are tabulated in table 4.2 and 4.3. This data is in compliance with the Ambient Air Quality Directive for particulate air pollution, which has been discussed previously in section 2.5 of chapter 2. With regard to the 24-hour average the PM_{10} levels measured in 1998 are in compliance as there are less than 35 exceedences of the 24-hour average mass concentration value of 50 $\mu g\ m^{-3}$. The annual average of 40 $\mu g\ m^{-3}$ is also in compliance.

Table 4.2 Monthly Summary Statistics of PM_{10} Values ($\mu g\ m^{-3}$)

Month	Mean	Max Daily Average	Min Daily Average	Median Daily Average	98th Percentile	95th Percentile	No. of Exceedences
Dec-97	21.36	45.8	6.8	18.93	42.05	39.11	None
Jan-98	21.8	44.03	8.4	19.9	42.5	40.83	None
Feb-98	16.75	42.52	9.45	14.36	36.76	29.85	None
Mar-98	19.85	36.56	8.75	20.55	32.91	29.98	None
Apr-98	18.74	34.03	9.09	16.57	33.15	32.32	None
May-98	21.62	39.72	7.35	20.4	37.17	34.5	None
Jun-98	16.3	35.2	7.94	15.06	35.05	34.4	None
Jul-98	11.48	22.93	5.1	10.66	22.85	19.32	None
Aug-98	14.4	26.94	7.2	12.83	26.5	24.55	None
Sep-98	20.5	44.3	5.72	18.14	41.4	38.88	None
Oct-98	13.16	25.53	6	11.38	24.86	23.4	None
Nov-98	14.54	41.8	7.14	12.64	37.55	31.5	None
Dec-98	25.19	65.97	9.25	22.82	56.7	44.75.	1
Jan-99	26.65	51.63	9.4	26.12	51.54	50.96	2
Feb-99	30.1	52.6	14.55	29.96	47.63	41.75	1
Mar-99	27.56	45.54	12.09	25	42.95	41.2	None
Mar-00	28.7	64.2	12.5	27.66	58.77	50.64	1
Apr-00	23.34	40.63	11.5	21.84	38.45	35.97	None
Sep-00	34.6	44.73	20.33	36.2	44.12	43.23	None
Oct-00	21.78	34.25	9.65	21.65	33.23	21.8	None
Nov-00	24.23	48.67	10.99	22.95	48	46.17	None
Dec-00	30.14	82.57	16.8	24.26	71.52	54.96	1
Feb-01	52.9	70.91	11.72	48.7	70.74	70.4	6
Mar-01	39.96	86.69	18.36	36.67	82.11	71.41	3
Apr-01	28.1	52.26	6.68	26.66	49.58	44.8	1

The levels recorded for 1999, 2000 and 2001 do not cover the entire twelve months for any year, but there seems to be a general upward trend in the levels recorded. For the three months of this year there are already nine exceedences of the 24-hour average mass concentration value. With the next round of standards that will be set under the directive and with conditions continuing in this trend there may be problems in future years in meeting the standards, in particular the annual average of 20 $\mu g\ m^{-3}$ that is to be attained by January 2010. There appears to be no significant variation between summer and winter monthly average mass concentration although the higher levels do tend to occur during the winter months. Such temporal trends will be discussed in more detail in the next chapter.

The 24-hour average mass concentration value tends to be higher during the winter periods, the highest value of 86.69 μg m^{-3} occurred in March-2001 and the lowest of 5.1 μg m^{-3} occurred in July-1998.

Table 4.3. Hourly Summary Statistics of PM_{10} Values (μg m^{-3})

Month	Maximum 1 Hour	Minimum 1 Hour	Median 1 Hour	98th Percentile Hourly Values	95th Percentile Hourly Values
Dec-97	128.60	2.30	62.52	62.52	50.20
Jan-98	147.20	15.20	17.10	65.20	55.00
Feb-98	102.00	2.00	14.20	52.10	38.66
Mar-98	71.20	4.10	16.85	50.64	40.60
Apr-98	73.30	1.10	16.00	47.90	42.00
May-98	74.20	4.40	18.70	56.40	47.30
Jun-98	127.80	2.10	13.20	48.52	40.93
Jul-98	42.50	2.50	16.60	31.20	24.80
Aug-98	43.80	2.20	12.30	35.30	31.30
Sep-98	69.00	3.40	17.30	52.00	46.00
Oct-98	115.10	2.30	10.80	35.42	29.30
Nov-98	79.10	1.60	11.25	57.30	40.70
Dec-98	195.40	3.10	20.40	93.60	56.00
Jan-99	141.30	3.10	20.40	87.66	63.10
Feb-99	140.00	1.40	26.40	75.88	66.90
Mar-99	162.80	1.30	25.70	65.30	53.30
Mar-00	234.00	2.00	22.80	90.60	59.40
Apr-00	131.50	3.00	21.60	55.60	46.96
Sep-00	128.00	4.50	31.20	71.40	67.71
Oct-00	87.60	3.00	20.40	52.74	44.22
Nov-00	145.90	2.00	20.10	82.41	53.10
Dec-00	177.20	3.20	23.40	121.34	68.64
Feb-01	220	7.4	40	187.66	137.98
Mar-01	210.00	8.80	31.70	113.42	89.40
Apr-01	112.10	4.00	24.80	77.00	57.10

The statistics presented in table 4.3 highlight the peak hourly concentrations that occur on a daily basis, the effects of which are cancelled out by lower night-time levels and the 24-hour averaging periods. These peak hour levels of particulate pollution generally occur in the morning or evening normally coinciding with rush hour traffic.

Figure 4.1 shows typical daily measurements of PM_{10} mass concentrations. It is characterised by a morning, lunchtime to mid afternoon and evening peak. This is typical of a week-day. A weekend day or public holiday differing as there is no significant morning peak as shown in figure 4.2.

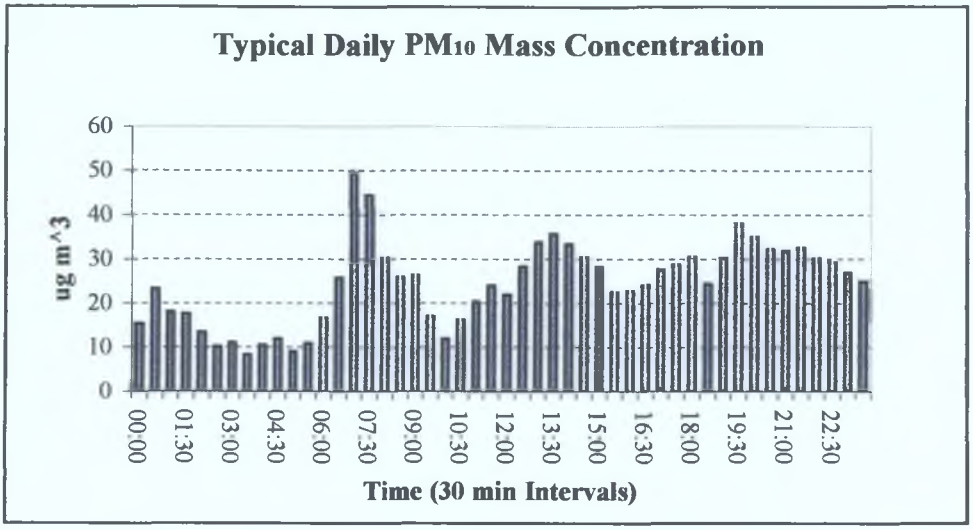

Figure 4.1. Typical Week Day of PM_{10} Mass Concentration Values

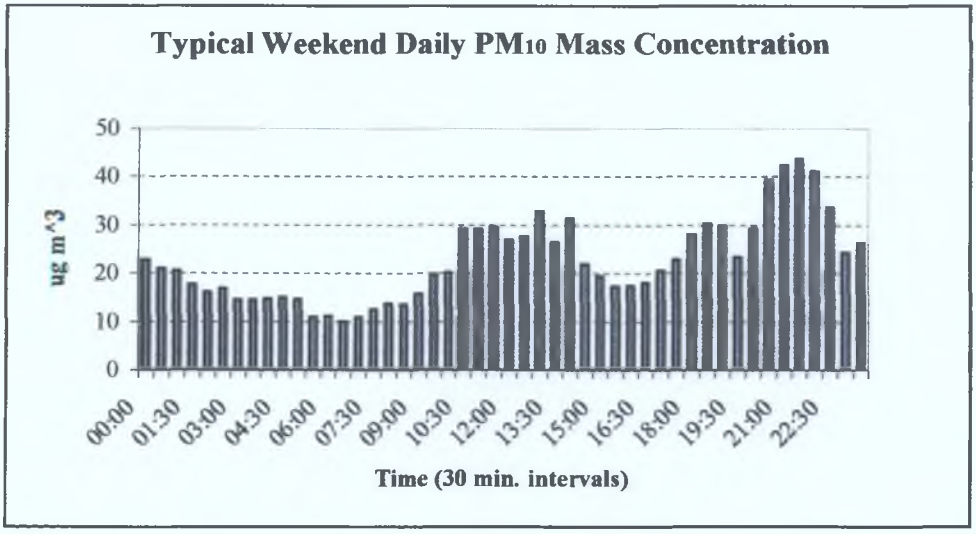

Figure 4.2 Typical Weekend Day of PM_{10} Mass Concentration Values

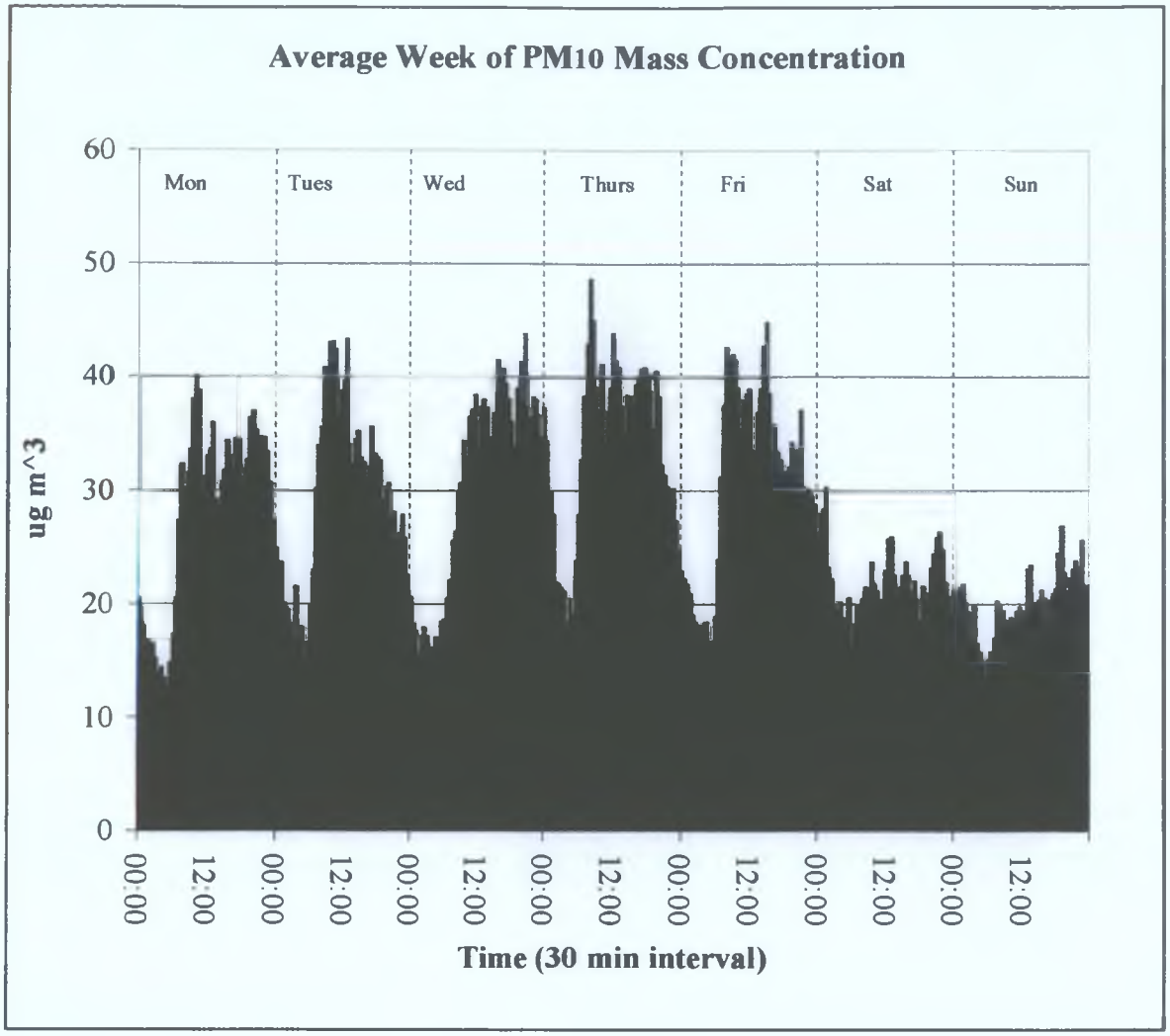

Figure 4.3 Typical Week of PM_{10} Mass Concentration Values

Figure 4.3 is a graph of an average week of PM_{10} mass concentration measurements. This graph gives an indication of the differences that occur between week and weekend days.

4.2 Meteorological Parameters.

The monthly averages of the meteorological parameters are presented in table 4.4. The meteorological data measured at the city centre location was compared with regional data recorded at Dublin Airport.

Table 4.4 Meteorological Parameters

Month	Wind Speed (m/sec)	Predominant Wind Direction	Temperature (degrees C)	% Relative Humidity
Dec-97	1.3	South-West	8.9	73.9
Jan-98	1.6	South-West	7.45	71.5
Feb-98	2.5	South-West	9.6	71.6
Mar-98	1.96	South/South-West	9.56	71.46
Apr-98	1.8	South-West	7.99	71.88
May-98	1.66	South-West/East	13.06	71.11
Jun-98	1.72	South-West	14.4	71.11
Jul-98	2.04	South-West	15.65	70.66
Aug-98	1.95	South-West	15.6	69.9
Sep-98	1.55	South-West/East	14.7	76
Oct-98	2.22	South-West	11.65	71.15
Nov-98	1.9	South-West	8.75	72.65
Dec-98	1.91	South-West	8.08	71.6
Jan-99	2.3	South-West	6.98	71.12
Feb-99	1.6	South-West	6.95	69.85
Mar-99	0.95	North-East	9.31	66.85
Mar-00	2	West/South-West	9.6	71.5
Apr-00	1	South West/North East	10.5	67.53
Sep-00	1.5	West/North-West	15.34	58.18
Oct-00	1.23	North-West	10.76	56.72
Nov-00	Insufficient Data	Insufficient Data	Insufficient Data	Insufficient Data
Dec-00	1	North-West	6.26	62.3
Feb-01	1.43	North-West	5.65	53.1
Mar-01	2.2	North-East	6.67	51.32
Apr-01	1.25	South West/North East	8.73	47.77

4.2.1 Wind Speed

The wind speeds recorded at the city centre location were generally low, the average over the monitoring period is in the order of 2 m/sec. The highest wind speeds recorded were 8 m/sec. The mean wind speed values do not necessarily indicates the most frequently occurring wind speed (Croxford *et al.*, 1996). The discrete frequency distribution of the wind speed measurements is presented in fig 4.4. It is seen from this distribution how the wind speeds of 1.5m/s to 2 m/s are most common.

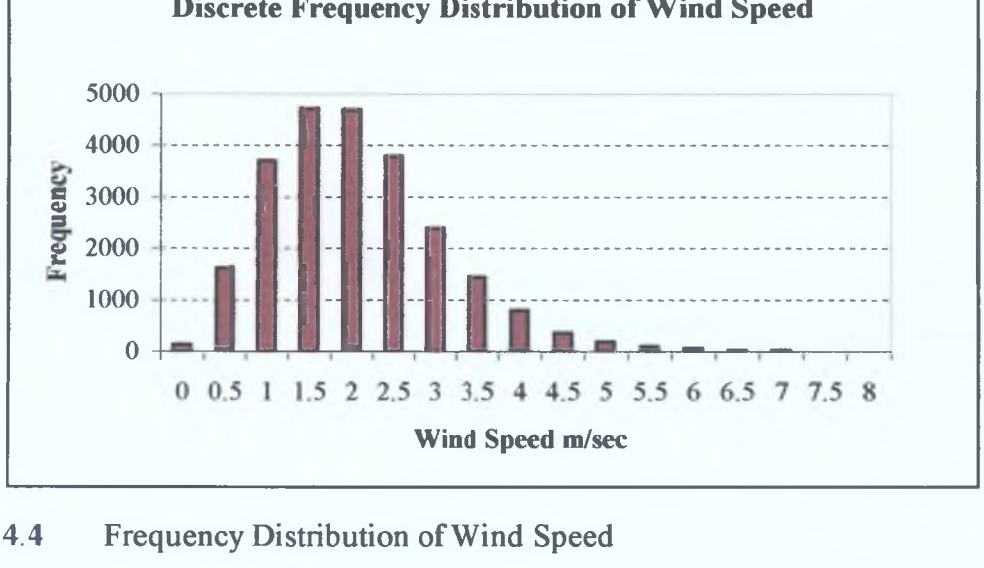

Figure 4.4 Frequency Distribution of Wind Speed

It was necessary to investigate which distribution curve best fitted the frequency distribution of wind speed. Table 4.5 lists two possible curves used to approximate the distribution. The poisson distribution is discrete and therefore no predictions for the poisson distribution are calculated in table 4.5.

Table 4.5 Comparison of Theoretical Distributions for Wind Speed

Wind Speed	Frequency	Normal	Poisson
0	0.00470	0.029328	0.073474
0.5	0.05380	0.066835	
1	0.12350	0.11677	0.154295
1.5	0.15703	0.156409	
2	0.15640	0.160621	0.162010
2.5	0.12630	0.126459	
3	0.07947	0.076331	0.113407
3.5	0.04820	0.035323	
4	0.02637	0.012532	0.059539
4.5	0.01220	0.003409	
5	0.00623	0.000711	0.025006
5.5	0.00317	0.000114	
6	0.00190	1.39E-05	0.008752
6.5	0.00093	1.31E-06	
7	0.00063	9.43E-08	0.002626
7.5	0.00013	5.21E-09	
8	0.00010	2.21E-10	0.000689
8.5	0.00000	7.16E-12	
9	0.00000	1.78E-13	0.000161

Figure 4.5 below is a graph of the data presented in table 4.5 and gives an indication of the closeness of each theoretical frequency distribution. The normal distribution falls down due to the none-symmetrical shape of the measured frequency distribution. But instead of using either the poisson or normal distribution it was decided to simply use the measured discrete values to decide the highest frequency of occurrence.

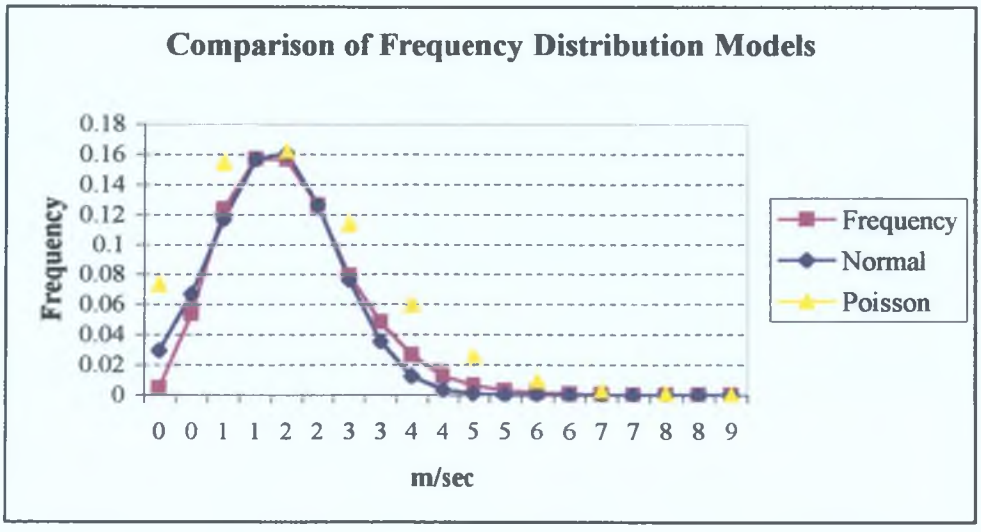

Figure 4.5 Theoretical Frequency Distributions For Wind Speed

Regional wind speeds recorded at Dublin Airport were considerably higher, averaging about 8 m/sec. The difference between the two data sets is most likely a result of the difference in the monitoring locations. Buildings and street geometry in the city centre have a shielding and channelling effect on wind flows resulting in lower measurements in the city centre (Croxford *et al.*, 1996). The regional meteorological data from Dublin Airport is recorded at a more open site and higher altitude i.e. due to wind gradient.

4.2.2 Wind Direction

The predominant wind directions recorded at the city centre location were from the south and south-west. This compares well with the regional wind direction data recorded at Dublin Airport. Although there is a lot more deviation about the mean wind direction at the city centre location. These deviations from the regional wind direction are again due to the city centre location. Tall structures and street geometry have the effect of distorting the wind flows. The effect of buildings and urban structures often result in a local micrometeorological climate being created in such locations, which can be quite different to the regional climate (Berkowiz *et al.*, 1996, Croxford *et al.*, 1996).

4.2.3 Air Temperature

Average monthly temperature ranged from 5°C to 15.5°C. The local temperature values compare quite well with values recorded at Dublin airport. Air temperature values tend to be consistently higher at the city centre location, as displayed in figure 4.6. This is possibly due to the urban heat island effect that has previously been discussed in section 2.3 of chapter 2.

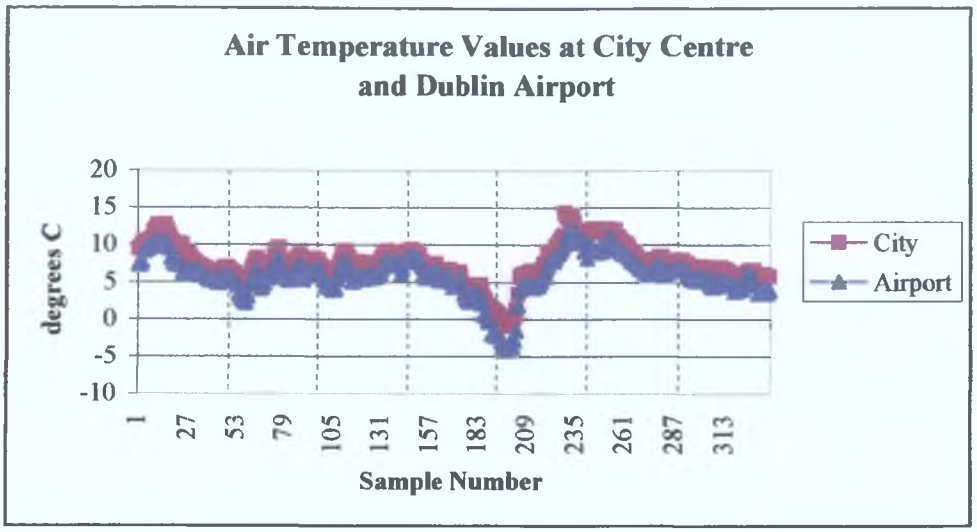

Figure 4.6 Plot of Local and Regional Air Temperature Values

Bivariate correlation analysis was used to establish how closely related the local and regional meteorological air temperatures are. Figure 4.7 presents a scatter plot of the two data sets, which shows good correlation, with an R^2 value of 0.9463.

The r value is the Pearsons correlation coefficient, this describes how strongly the x and y values in a sample of pairs are related to one another. The R^2 value is the coefficient of determination, which is the square of the correlation coefficient. This value gives the proportion of variation that is attributable to the linear relationship between the two variables. The r value is given by the equation;

$$r = \frac{\sum(x_i - \bar{x})(y_i - \bar{y})}{\sqrt{(x_i - \bar{x})^2}\sqrt{\sum(y_i - \bar{y})^2}} \qquad \text{.....Equation 4.1}$$

Where

(x_1,y_1), (x_2,y_2),...., (x_n,y_n) denote a sample (x,y) pairs and $\bar{x}$, $\bar{y}$ are the averages of the x samples and y samples

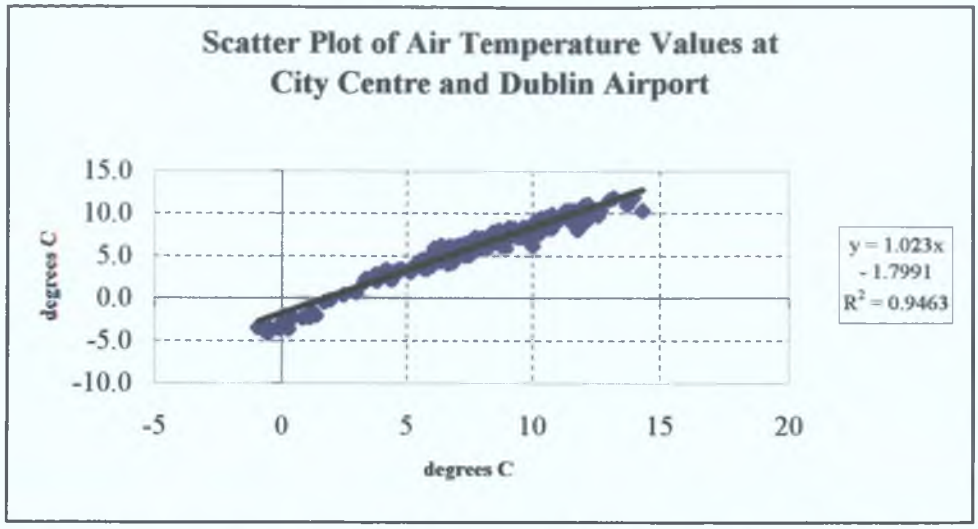

Figure 4.7 Scatter Plot of Local and Regional Air Temperature Values

4.2.4 Percent Relative Humidity

Average monthly percent relative humidity values recorded ranged from 47 to 74 % relative humidity. The values recorded at the city centre compare quite well with regional data recorded at Dublin Airport. Figure 4.8 displays the regional values to be consistently higher than those at the city centre location.

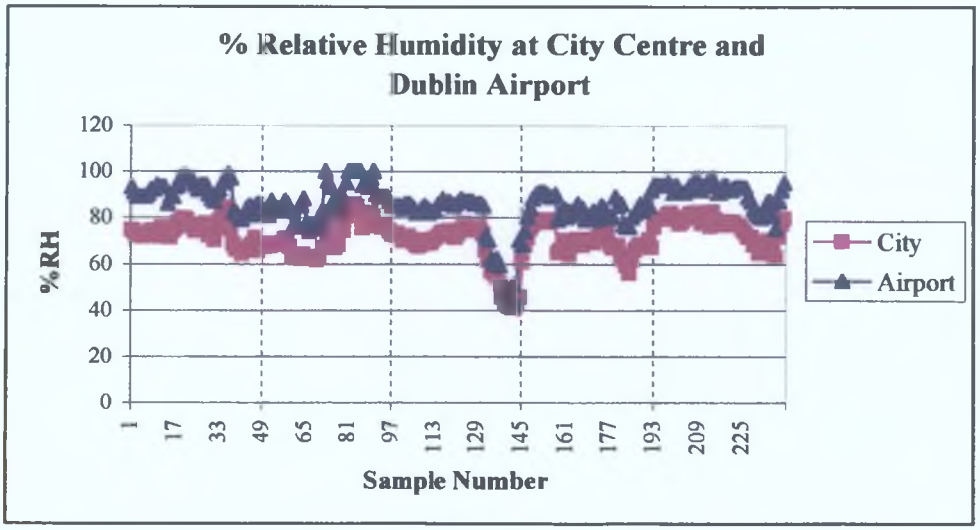

Figure 4.8 Plot of Local and Regional % Relative Humidity Values

Bivariate correlation analysis was used here again to establish how closely related the local and regional meteorological percent relative humidity values are. Figure 4.9 shows good correlation between the two data sets, producing an R^2 value of 0.84.

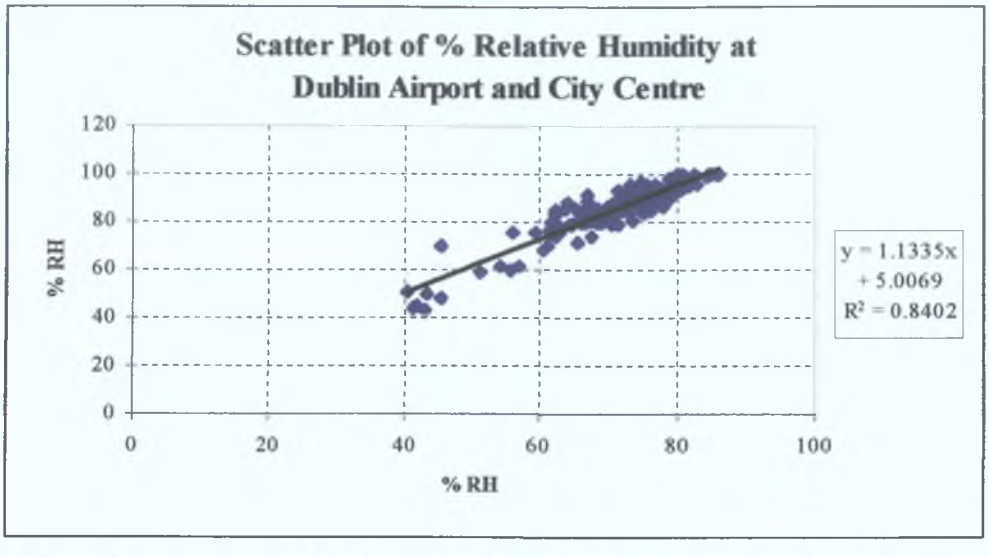

Figure 4.9 Scatter Plot of Local and Regional % Relative Humidity Values

The good level of correlation between the local and regional values for air temperature and percent relative humidity values reveal that the urban location does not alter the mean regional meteorological conditions examined above to any great extent. It also shows good reliability in the measurements recorded as part of this monitoring study.

4.3 Traffic Flow Characteristics

Daily traffic volumes recorded at this intersection by means of the SCATS urban traffic control system range from 49,000 to 80, 000 vehicles. The average daily traffic volumes are presented in table 4.6. The maximum and minimum volumes tend to occur at the same time each day, with the exception on weekends when there is no distinct morning peak. This is clearly seen in figure 4.10, which presents a typical average week of traffic volumes. From observation of the weekly traffic counts, it is clear that week days follow very similar daily cycles i.e. peak in the morning and evening, flat in the afternoon and dropping off after nightfall.

The measurement period included three public holidays and one period of school holidays. The public holidays exhibit the same traffic patterns as a Saturday and Sunday and the period of school holidays did not exhibit any decrease in traffic volumes at this intersection.

Table 4.6 Average Daily Volumes of Traffic (25-Jan-01 to 9-May-01)

Day	Average Volume	Max Volume	Time of Occurrence	Min Volume	Time of Occurrence
Mon	75,263	2,977	08:30-09:00	129	05:00-05:30
Tues	75,830	2,844	08:00-08:30	108	05:00-05:30
Wed	77,043	2,974	08:30-09:00	123	05:00-05:30
Thurs	79,295	3,062	08:30-09:00	155	05:00-05:30
Fri	80,027	2,949	08:30-09:00	203	05:00-05:30
Sat	59,115	1,997	13:00-13:30	312	05:00-05:30
Sun	49,765	1,798	14:30-15:00	269	06:00-06:30

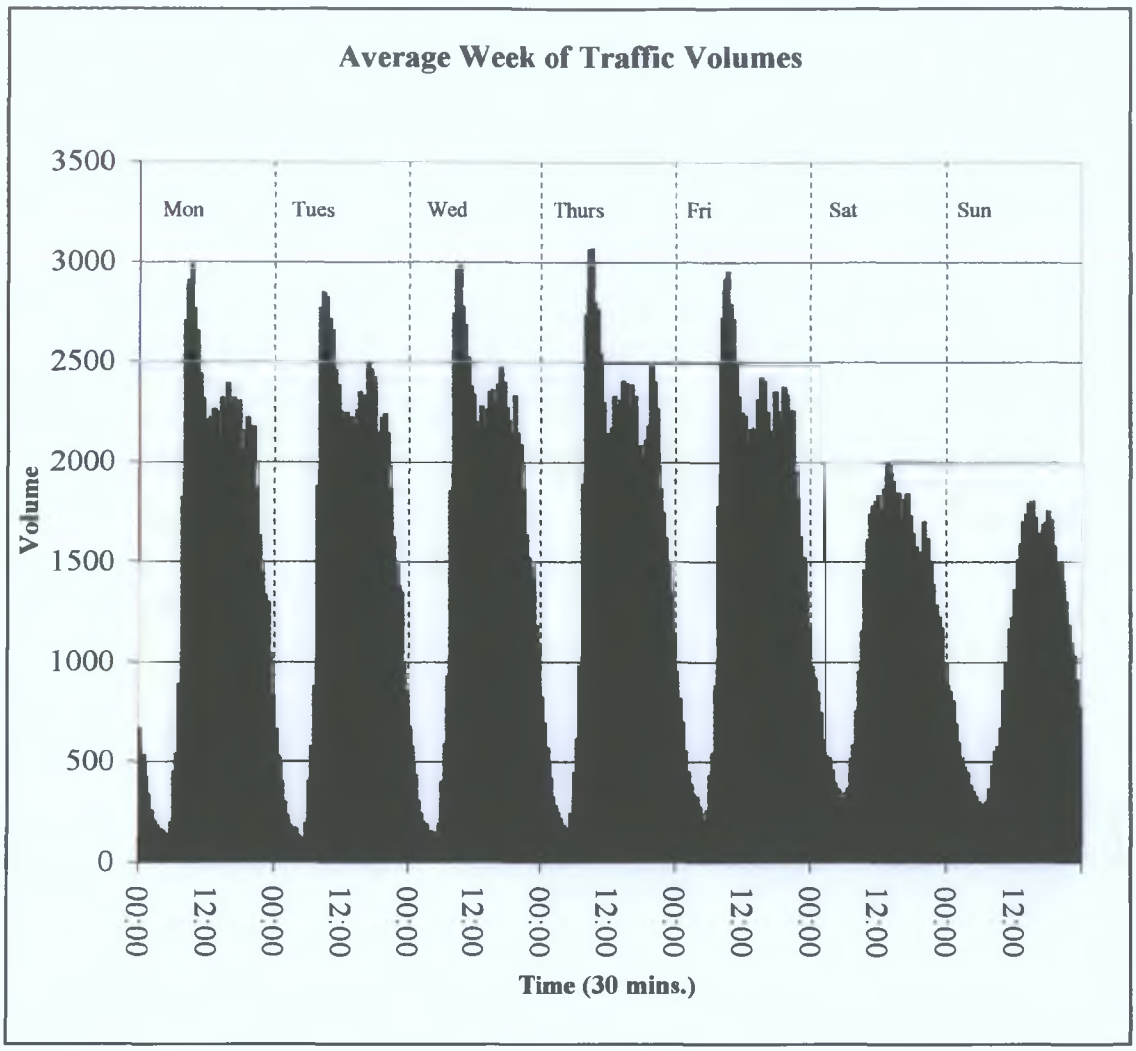

Fig. 4.10 Average Weekly Traffic Volumes.

The volumes recorded in late January of 2001 were compared with volumes recorded over the same time period in 1998. Traffic volumes from 1998 were also recorded using by the SCATS system. There appears to be a significant increase in volumes through the junction since that time. As noted earlier in section 4.1 the levels of PM_{10} recorded since 1998 have increased, this would fit in well with the increase in traffic volumes through the intersection. Based on the available data from 1998, there is an increase in the volumes of motor vehicles by approximately 35%.

There is still the same daily trend in traffic volumes, figure 4.11 shows an average week day of traffic volumes for January 1998 and 2001. There have been no changes to the layout of this junction since 1998, i.e. no quality bus corridors or other traffic control measures have been put in place.

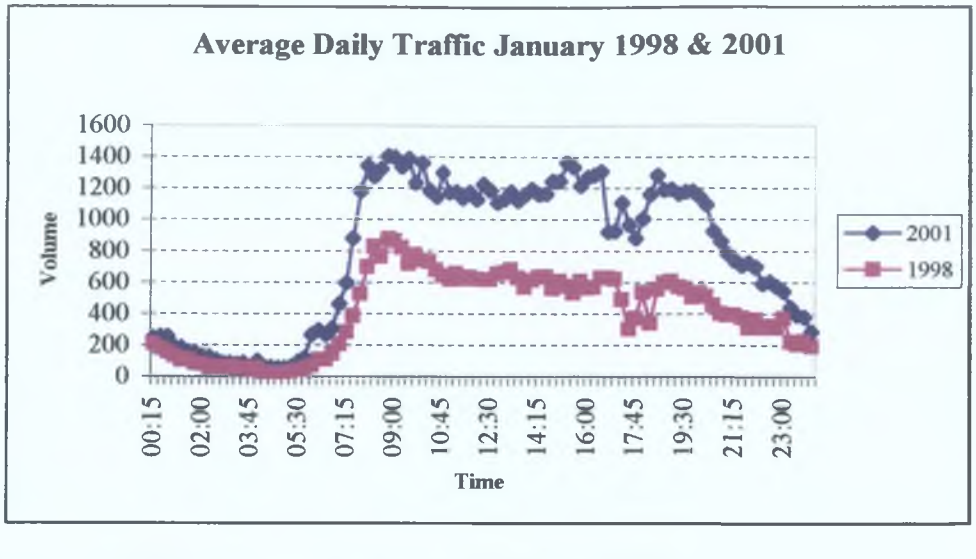

Fig. 4.11 Traffic Volumes for January 1998 and 2001.

The SCATS system does not give a breakdown of vehicle type. Figure 4.12 presents are plot of volumes of cars and other vehicles passing through the junction between 7:00 am and 6:00 pm. This graph is produced from manual counts taken at the junction for a typical weekday (Marnane, 2000). These counts were taken on a weekday and give an indication of the breakdown of traffic composition. Other vehicles include buses and goods vehicles that are mainly diesel fuelled, such vehicles emit greater amounts of PM_{10} than petrol fuelled vehicles. This graph indicates that the number of other vehicles tend to increase as the number of cars decrease.

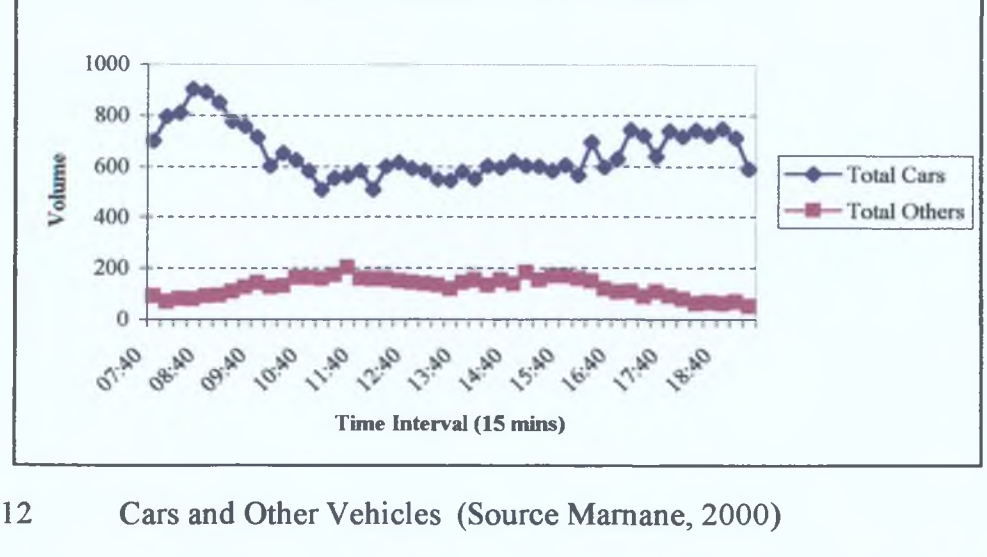

Fig. 4.12 Cars and Other Vehicles (Source Marnane, 2000)

4.3.1 Vehicle Ownership Statistics

The latest vehicle ownership statistics are from 1999. These statistics show a 24% increase since 1998 in vehicle ownership across all vehicle categories. The greatest increase in 1999 was noted in motorcycle ownership, an increase of 59% from the previous year. The total vehicle ownership in Ireland for 1999 was 1,608,156 vehicles with private cars contributing 1,269,245 to this total. Statistics for the greater Dublin area are presented in table 4.6.

Table 4.7 Fleet Composition in The Greater Dublin Area, 1999 (Source DOE, 1999)

	Total Vehicles 436,171 100%	
Petrol 93.29% 339,575	Private Cars 364,295 83.50% Diesel 6.64% 24,189	LPG 0.07% 255
Petrol 2.55%	Goods Vehicles 42,978 9.85% Diesel 97.39%	LPG 0.06%
Buses 2,461	Public Service Vehicles 8,600 1.97% Taxi & Hackney 6,139	
	Motorcycles 10,135 2.32%	

The fleet composition can be broadly split into 71.17% petrol fuelled, 28.77% diesel fuelled and 0.06% fuelled by liquid petroleum gas (LPG). Table 4.6 accounts for 97.64% of vehicles in the Greater Dublin area, the remaining 2.35% is made up of other classes. Other classes may include government owned vehicles, fire brigade vehicles and ambulances. (Department of The Environment, (DOE), 1999).

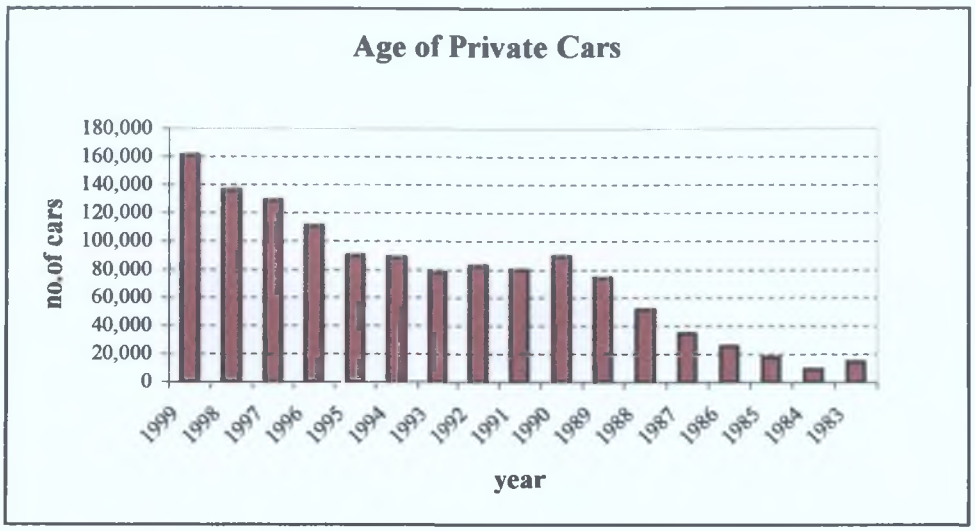

Figure 4.13 Age of Private Cars in Ireland (Source DOE, 1999).

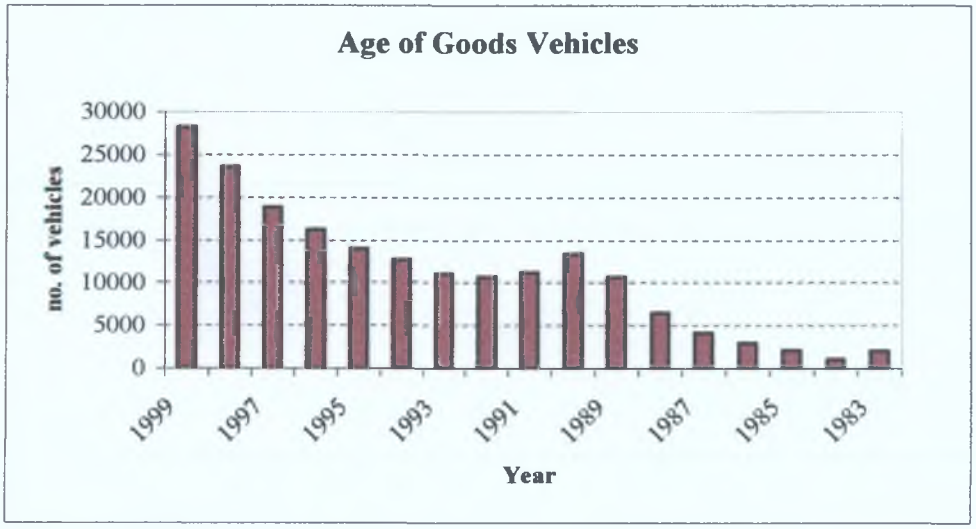

Figure 4.14 Age of Goods Vehicles in Ireland (Source DOE, 1999)

Figures 4.13 and 4.14 show the age of vehicles in the State. Over the last few years numbers of older aged cars has been noted to decrease significantly. This is partly due to increased prosperity, the introduction of the government scrapage scheme, and the many incentives offered by cars dealers encouraging the public to purchase new cars. This should be better for the environment as newer cars incorporate cleaner technologies into vehicles engine design (Fenger, 1999).

4.3.2 PM_{10} and Traffic Volumes

Average daily PM_{10} mass concentration values are compared with average daily traffic volumes. Figure 4.15 presents a plot of this comparison. The peak in PM_{10} levels tends to follow the peak in traffic volume, the traffic generated pollution being picked up by the particulate monitoring equipment a short time later. This distinct pattern points to traffic being the major contributor to ambient PM_{10} levels at this location.

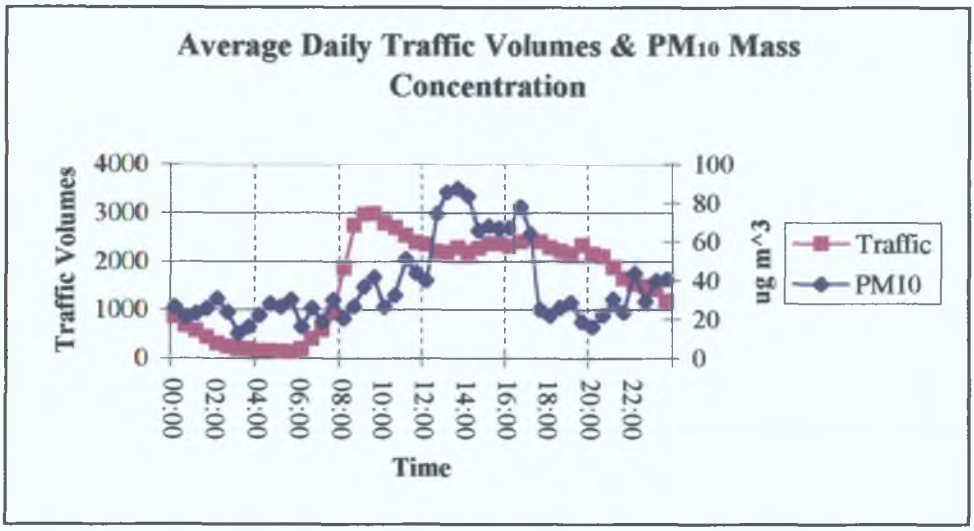

Fig. 4.15 Average Daily PM_{10} Values and Average Daily Traffic Volume.

4.4 Conclusions

This chapter presented the data recorded in the monitoring study. In section 4.1 the monthly statistics of the PM_{10} mass concentration values were complied and compared with the air quality standard for particulate air pollution. The PM_{10} data recorded at the junction was in compliance with the directive. The levels recorded however are seen to have increased. The meteorological parameters were summarised and compared with regional meteorological data recorded at Dublin Airport. Air temperature and percent relative humidity measurements were found to be well correlated with the regional measurements. There was a significant variation between local and regional wind speed and wind direction measurements. This variation was noted to result from the differences in locations. The frequency distribution of the wind speed measurements was investigated. The most frequently occurring wind speeds were found to occur between 1.5 and 2.0 m/sec and are not significantly different from the mean wind speed. The traffic flow characteristics of the junction were summarised in section 4.3. When compared with limited data from 1998 there is a significant increase in the volumes of traffic passing through this junction on a daily basis.

The data that has been presented here will be analysed in chapter 5 to establish the exact underlying trends in the data.

Chapter 5. Interpretation of Monitored Data

This chapter investigates the relationships between the various parameters presented in chapter 4. As previously discussed in section 2.3 of chapter 2, the measured ambient level of PM_{10} is not only as a result of the amount of pollutant emitted into the atmosphere, but also the effect of the prevailing meteorological conditions and the topography of the area. Dispersion of pollutant is greatly influenced by wind direction, wind speed, turbulence and atmospheric stability. Topographical siting and urban structures like street canyons for example have a great effect on these meteorological parameters. These effects on meteorological parameters result in spatial variation in pollutant level (Croxford *et al.*, 1996). In chapter 4 the differences that occur between the city centre and Dublin Airport meteorological conditions have been noted, particularly noticeable is the wind speed and wind direction measurements. Air quality in a city is often assessed based on data from only a few urban monitoring sites; therefore it is important to assess the small-scale effects that buildings and street geometry have on the wind flow so that the data measured is representative of the actual pollution level. (Croxford *et al.*, 1996, Scaperdas *et al.*, 1999, Berkowicz *et al.*, 1996) The objective of this chapter is to investigate the relationship between the measured PM_{10} data; meteorological parameters and pollutant source i.e. motor vehicles. The results of the data analysis will then be used to calculate the persistence factors in chapter 6.

5.1 Data Analysis

Time series analysis was applied to the data to give an initial visual indication as to what trends were present and then to begin analysing the relationship between the measured PM_{10} levels and the meteorological variables and volumes of motor vehicles. Such methods as regression analysis and trend analysis were used extensively to model these relationships.

5.1.1. Regression Analysis

Regression analysis attempts to use the information from the explanatory variable to explain some of the variability in the response variable (Hogg and Ledolter, 1987). In this study the response variable is the PM_{10} mass concentration and the explanatory variables are traffic volumes and the meteorological parameters. The simple linear regression model and the polynomial regression model were investigated.

Simple Linear Regression Model

This model is written as; $Y_i = \beta_0 + \beta_1 x_i + \varepsilon_i$ $i = 1,2,\ldots n$ .....Equation 5.1

Where:

1. Y_i = the response variable that corresponds to the setting in x_i of the explanatory variable, $i = 1,2,\ldots n$.
2. x_i is the ith observation on the explanatory variable
3. β_0 and β_1 are the coefficients (parameters) in the linear relationship. β_0 is the intercept and β_1 is the slope. A change of one unit in the explanatory variable X translates into a change of β_1 units in the response variable.
1. The random variables $\varepsilon_1, \varepsilon_2,\ldots, \varepsilon_n$ are errors that create the scatter around the linear relationship $\beta_0 + \beta_1 x_i$, $i = 1,2,\ldots, n$, respectively. It is assumed that these errors are mutually independent and normally distributed with mean zero and variance σ^2
2. $E(\sigma^2) = 0$ and var $(\varepsilon_1) = \sigma^2$, $i=1,2,\ldots, n$

   The information from one error does not imply something about the other.

The response $Y_i = \beta_0 + \beta_1 x_i + \varepsilon_i$ is the sum of two components. The first, $\beta_0 + \beta_1 x_i$, is not random because β_0 and β_1 are parameters and x_i is a known constant. The second component ε_i is a random variable and hence Y_i is also a random variable. $Y_1, Y_2,\ldots, Y_n$ are mutually independent random variables that have the respective distributions, $N(\beta_0 + \beta_1 x_i, \sigma^2)$, $i = 1,2,\ldots,n$, $E(Y) = \beta_0 + \beta_1 x$ is called the regression function.

(Hogg and Ledolter, 1987)

Polynomial Regression Model

The polynomial regression model is used in situations where the response is curvilinear. A second order polynomial regression model in the one variable is given by

$$Y = \beta_0 + \beta_1 x^2 + \beta_2 x^2 + \varepsilon \quad \text{.....Equation 5.2}$$

This is sometimes called the quadratic model. β_1 is the linear effect parameter and β_2 is the quadratic effect parameter.

To estimate the closeness between observations Y_i and a regression line $\beta_0 + \beta_1 x_i$ is to consider the square of each deviation. The sum of the squares is a measure of the closeness between the regression line and the observations. There are three sums of squares, which are

1. Total Sum of Squares SSTO
2. Sum of Squares due to Regression SSR
3. Sum of Squares due to Error SSE

The coefficient of determination R^2 provides a summary statistic that measures how well the regression equation fits the data. This is given by

$$R^2 = SSR / SSTO = 1 - SSE / SSTO$$

This coefficient expresses the variability that is explained by the regression model as a fraction of the total sum of squares. The R^2 value lies between 0 and 1. The higher this value is the more the explanatory variable explains the variability in the response variable, an R^2 value of 0.6 or greater is generally accepted. (Hogg and Ledolter, 1987, Montgomery and Peck, 1992).

5.1.2. Mathematical Application

The response variable and explanatory can also be termed the predictand and prescriptor respectively (Benarie, 1980). The condition of a prescriptor whose variation greatly influences the predictand will cause a decrease in the variance of the latter. The evaluation of importance of different prescriptors is impeded by the correlation between the regression coefficients and it is further subject to an uncertainty, which cannot be expressed statistically. The addition of one or more prescriptors may change the previous conclusions completely since the regression coefficients are altered.

From this point the number of prescriptors should not be too large. Excessive scatter indicates almost automatically the prescriptors that are useless and should be eliminated. In assessment of data there should be a development sample and a test sample. The development sample should be used to investigate and establish the relationship between the predictand and precsriptor. The test sample should then be used to validate the relationship between the two. (Benarie, 1980, Zannetti, 1990).

In this study the monitored data was split into a development sample and a test sample, the data from the development sample was used for the data analysis. The data from the development sample was filtered in such a way as to identify only one dependent and independent variable. This is to say that such a large volume of data allows the relationship of PM_{10} levels for say 1000-1500 vehicles, at a wind speed of 3m/sec from a north-easterly direction to be identified. This allowed for more detailed single regression analysis to be performed on the data, yielding a more accurate model. This involved extensive data manipulation as any missing data had to be identified and not recorded as zero levels.

Analysis of data was based on the half hour PM_{10} mass concentration value. An average traffic week was calculated based on the available traffic data (Jan to May 2001) with public holidays and weekends accounted for. This average week was then assigned to all weeks of PM_{10} data for which traffic was unavailable. This average week is possibly an over estimation of the traffic volumes that would have been occurring for the earlier PM_{10} measurements, but it is valid for the following reasons. In section 4.3 of chapter 4 traffic volumes from January 1998 are compared with volumes from the same period in January 2001, the same pattern in traffic volumes was observed, low night-time levels, early morning peaks. Volumes from this year show that patterns are repeated on a day to day basis. Thus, when PM_{10} data was filtered, PM_{10} levels occurring at night-time and early morning fell into the same patterns, also these times have similar weather patterns. Generally more stable conditions with lower air temperatures and lower wind speeds occur at night-time than during the day (Seinfeld, 1986).

The data is firstly assessed for temporal trends, i.e. diurnal cycles, weekly and seasonal patterns. Then the effect of traffic volumes and of each individual meteorological variable is investigated to assess its effect on the measured pollutant level.

5.2 Temporal Variability of Ambient PM_{10} Levels

Temporal variations in air pollution measurements are related to diurnal cycles, weekly and seasonal patterns. The emission of traffic related air pollution is strongly linked to movement of people (Fenger, 1999).

5.2.1. Diurnal Variation

Measured levels of PM_{10} display a distinct diurnal variation, with a distinct morning peak followed by a series of smaller peaks throughout the day. This pattern in PM_{10} levels tends to follow the pattern of traffic volumes very well, as displayed in figure 5.1. Diurnal variability in PM_{10} mass concentration is influenced by two separate processes, which give rise to primary and secondary particulates (Seinfeld, 1986). Primary particulates emitted directly, and secondary particulates being formed as a result of chemical reactions in the atmosphere, for example the action of sunlight on gaseous emissions, resulting in photochemical smog or through agglomeration processes. Meteorological conditions in Dublin do not give rise to the formation of photochemical smog (Marnane, 2000), this results in PM_{10} levels following the traffic pattern very closely.

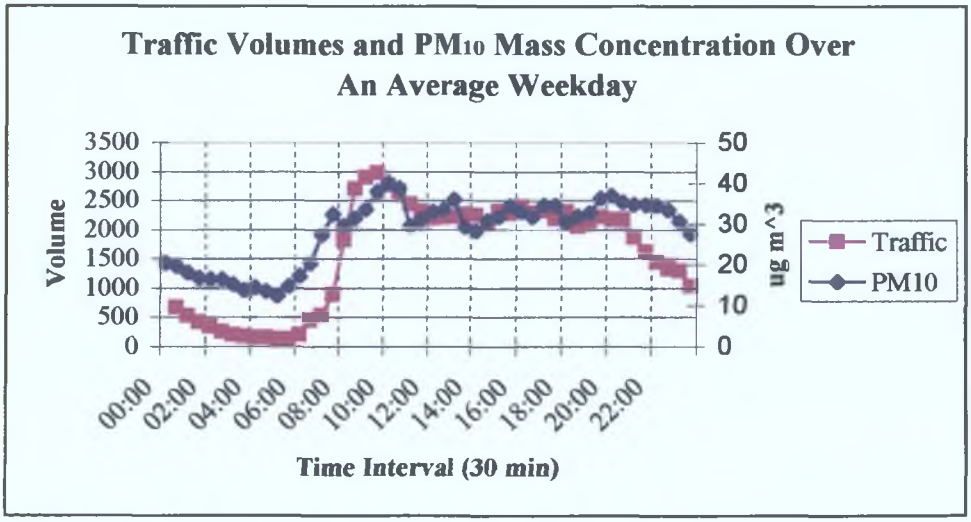

Figure 5.1 Average Weekday of Traffic Volumes and PM_{10} Values

Traffic volumes in the peak morning rush hour (8:00 to 9:00 am) are higher than that of the evening and this corresponds to a high morning peak in PM_{10} mass concentration value. Night-time levels of PM_{10} mass concentration do not drop off as quickly as morning levels. Even though traffic volumes in the morning are higher, they are seen to occur over a smaller time frame. In the evening traffic volumes are more spread out, thus the pollutant is continually emitted. It is also noted that the temperature normally drops and the winds calm towards nightfall, which are known to be situations for increased pollution levels as weather conditions are not conducive for dispersion and dilution. Occasions when night-time levels remain high right through the night to early morning are most likely as a result of temperature inversions.

There is a distinct variation between week-days and weekend days with respect to PM_{10} levels. A weekend day displays a higher peak of PM_{10} in the afternoon and evening time rather than the morning as presented in figure 5.2. Also night time and early morning (23:00 to 03:00) PM_{10} levels are higher on weekends than weekdays, indicating social activities and movement of people.

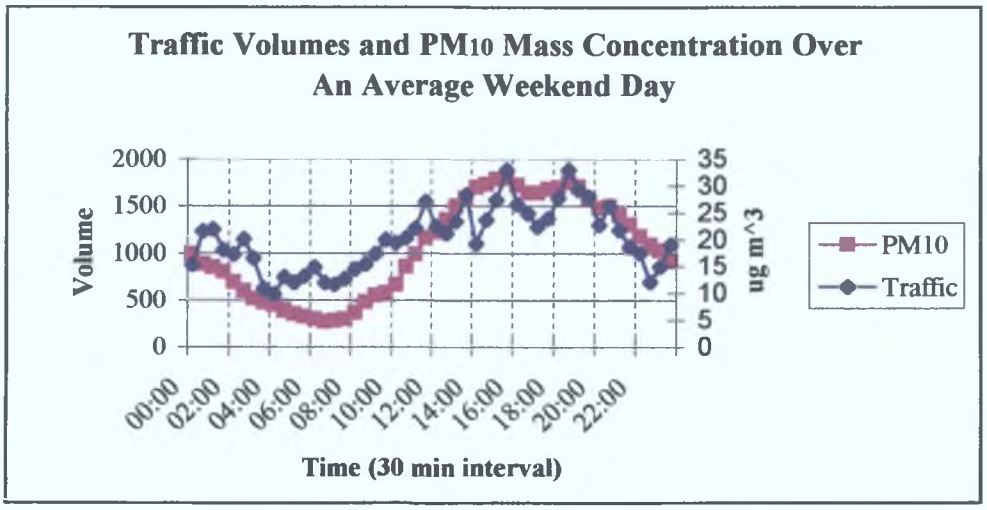

Figure 5.2 Average Weekend Day of Traffic Volumes and PM_{10} Values

5.2.2. Weekly Variation

The difference between a week-day and weekend day has been discussed above and this is clearly seen in figure 5.3, which presents a average week of PM_{10} values. This weekly pattern was shown to be very repeatable throughout the monitoring program.

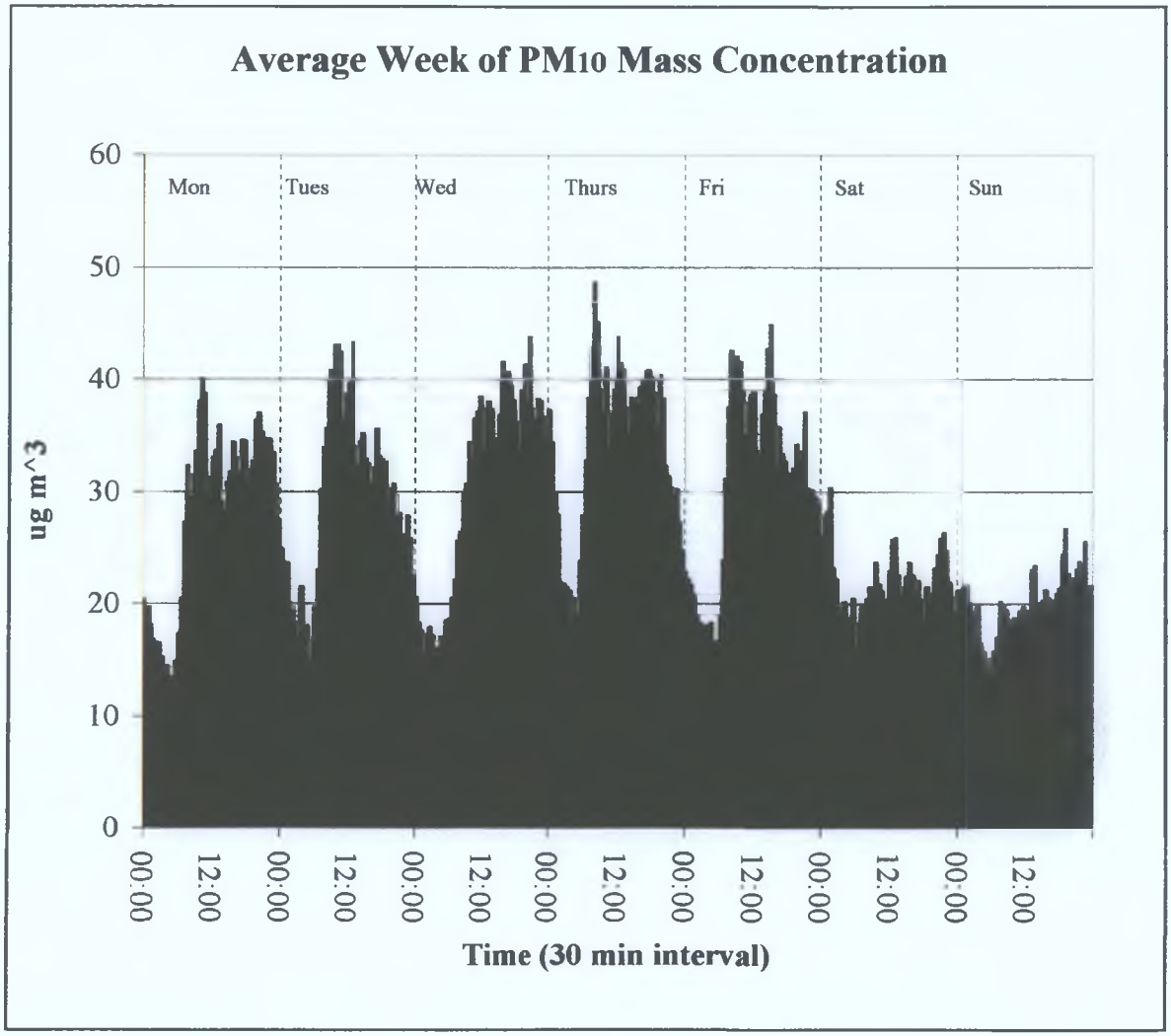

Figure 5.3 Weekly PM_{10} Mass Concentration Values

5.2.3. Seasonal Variation

Many studies have found a strong seasonal variation in PM_{10} data. The major cause for this difference may not be attributable to changes in emission patterns, but to weather conditions (Morel et al., 1999). For example: in the case of inversions, particulates are trapped in smaller volumes of air because the inversion temperature is at a lower level in winter.

Figure 5.4 is based on an average day of two summer months and two winter months of PM_{10} data from the same year. Towards night-time the wintertime levels appear to be higher than the summertime levels, this can mainly be attributed to differences in meteorological conditions. During summer months the mixing height is higher than in winter months resulting in better dispersal of the pollutant. Also during winter months there are more periods of stagnation and lower inversion layer resulting in higher levels of pollution being recorded. Furthermore a greater number of temperature inversions occur during the winter months than summer months thus there is less dispersion of pollutant, and higher levels than normal are recorded (Seinfeld, 1986, Morel *et al.*, 1999).

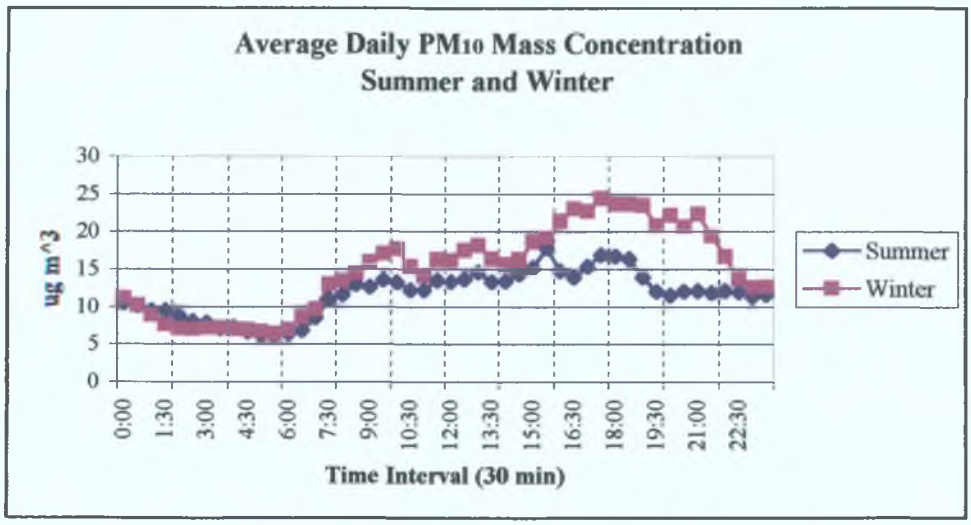

Figure 5.4. Seasonal Variation of PM_{10} Mass Concentration Values

From the monthly ambient temperatures presented in chapter 4 there is no major extremes in temperatures between winter and summer months. There are a higher percentage of easterly winds occurring during the winter months and these are normally associated with lower air temperatures. It is evident from the data that under such conditions higher levels of PM_{10} occur. At this particular site when the wind is from the north-east it is blowing the pollution from the junction directly toward the monitoring equipment, thus recording higher levels during the winter months. This observation is investigated further in this chapter in the analysis of the effect of wind direction on PM_{10} concentration values.

It is difficult to reveal the true seasonal dependency on the PM_{10} levels, as the levels recorded at this junction since 1998 have increased significantly, even when winter months are compared the is no major correlation in the data.

5.3 The Effect of Pollutant Source on PM_{10} Mass Concentration Values

The temporal trends in the data show a distinct relationship between pollutant source i.e. volumes of motor vehicles and the measured level of PM_{10}. PM_{10} data for which traffic data was available was analysed to model relationship between the two data sets. PM_{10} data was filtered with respect to traffic volumes into categories of 100 vehicles. Then the average PM_{10} mass concentration was isolated for each category and a scatter plot of the data is presented in figure 5.5.

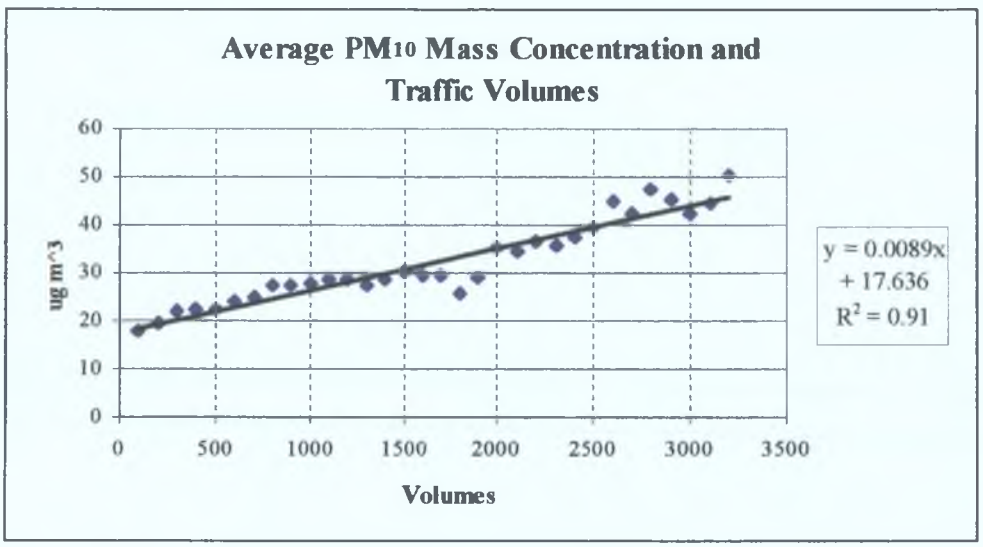

Figure 5.5. Plot of PM_{10} Mass Concentration Values and Traffic Volumes

High correlation was found between the two data sets a linear trend gives an R^2 value of 0.91. A polynomial trend fits this series quite well also, producing an R^2 value of 0.93.

Traffic speed at this intersection is quite low, below 20 miles per hour, except at times of very low traffic volumes namely the midnight to early morning, therefore there is very little turbulence due to speed of vehicles. Vehicles at higher speeds, add turbulence thus leading to greater dispersion and dilution of pollutant. Turbulence due to traffic may become important at very low wind speeds (Berkowicz *et al.*, 1996).

5.4 The Effect of Meteorology on PM_{10} Mass Concentration Values

PM_{10} mass concentration values are compared with meteorological data to establish the relationship between the measured PM_{10} mass concentration and the corresponding meteorological conditions. Already noted in chapter 4 there is variation between meteorological conditions recorded in the city centre and Dublin Airport, in particular, differences between the wind speed and wind direction measurements. These differences can be explained due to the nature of the location of the monitoring site.

The urban heat island effect gives rise to higher air temperatures in the city center (Sini *et al.*, 1996). The presences of buildings distort the mean wind direction and mean wind speeds. These effects are well documented in many air pollution studies (Hao *et al.*, 2000, Demirci *et al.*, 2000, Berkowicz *et al.*, 1996).

5.4.1. Filtering of Data

Data was filtered so as to identify one dependent and one independent variable based on the half hour PM_{10} mass concentration value, previously mentioned in section 5.1.2. Data was filtered as follows

1. Filtered with respect to traffic volumes into six categories increasing in increments of 500 vehicles
2. To assess the effects of air temperature, percent relative humidity and wind speeds each category from 1 above was filtered with respect to the independent variable that was being investigated.
3. To assess the effect of wind direction, each of the categories in 1 above was sorted with respect to wind speed increments of 0.5 m/sec intervals. Then in each of these sub-categories the average PM_{10} mass concentration from each wind direction was isolated. This allows for the effect of each wind direction angle to be investigated over different wind speeds, and it also allows for wind speed to be investigated in each individual direction (Berokwicz *et al.*, 1996).

5.4.2. The Effect of % Relative Humidity on PM_{10} Mass Concentration

Regression analysis was used to identify if a relationship exists between percent relative humidity and PM_{10} mass concentration values. The data from each of the traffic volume categories mentioned in section 5.4.1 were filtered with respect to percent relative humidity into sub-categories of 10 % relative humidity. The average PM_{10} mass concentration for each 10% increment in relative humidity was then isolated. This resulted in six scatter plots with respect to percent relative humidity based on increasing volumes of traffic. Very low correlation was found between the two data sets. This may be due in part to the lack of variation in humidity data, very few measurements occurring below 45% humidity and above 85%. When regression analysis was performed on data occurring within this data range of 45% and 85% relative humidity, quite good correlation was revealed. Under conditions of increased relative humidity the measured PM_{10} level was seen to decrease. The average of the six plots for this range of data is presented in figure 5.6. The polynomial trend best describes the relationship between the two variables, producing an R^2 value of 0.94.

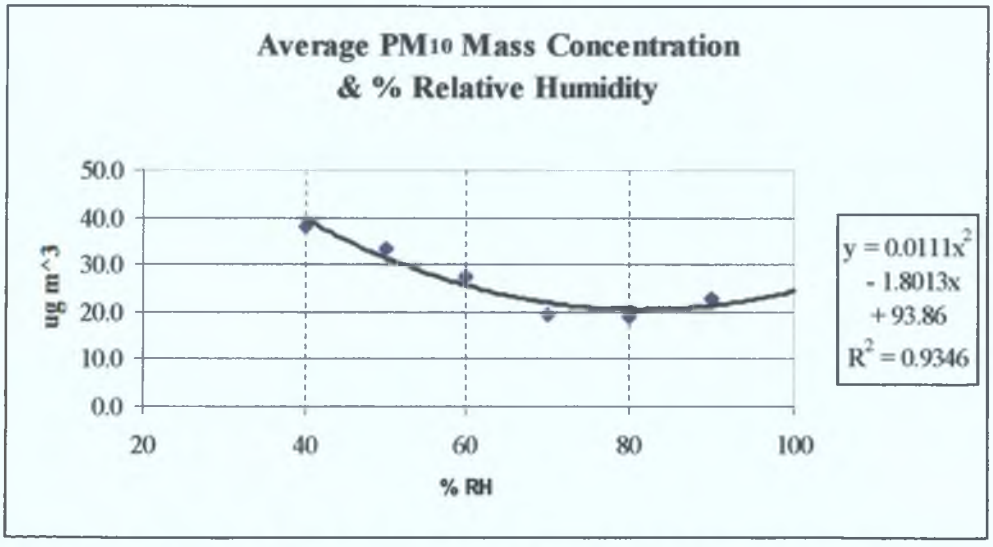

Figure 5.6 Average PM_{10} Mass Concentration and % Relative Humidity

5.4.3. The Effect of Air Temperature on PM_{10} Mass Concentration

Due to its geographical location Ireland does not suffer from extremes in air temperature between winter and summer. The monthly average temperatures have been presented in chapter 4 they range from 6°C to 16°C. Regression analysis was used to establish if there was a distinct relationship between PM_{10} levels and air temperature.

The PM_{10} data from each of the traffic volume categories mentioned in section 5.4.1 were filtered with respect to air temperature into sub-categories of 2°C. The average PM_{10} mass concentration for each 2°C increment was then isolated. This resulted in six scatter plots with respect to air temperature based on an increasing source of pollution (i.e. traffic volumes). The average of these six plots is presented in figure 5.7 a polynomial trend best fits the data with an R^2 value of 0.9414. This high R^2 value indicates that the trend line models the relationship between the PM_{10} levels and air temperature. Table 5.1 presents the coefficients of determination obtained for the each regression trend line that best fitted the relationship for the six plots. The polynomial trend line best describes the relationship between PM_{10} and air temperature, as it produces higher R^2 values than the linear, exponential and other trend lines. At very low temperatures the PM_{10} mass concentration tends to be at it's highest. When air temperatures rise above 20°C PM_{10} mass concentration tends to increase, this may be due to dry conditions at higher temperatures and particulates being re-suspended.

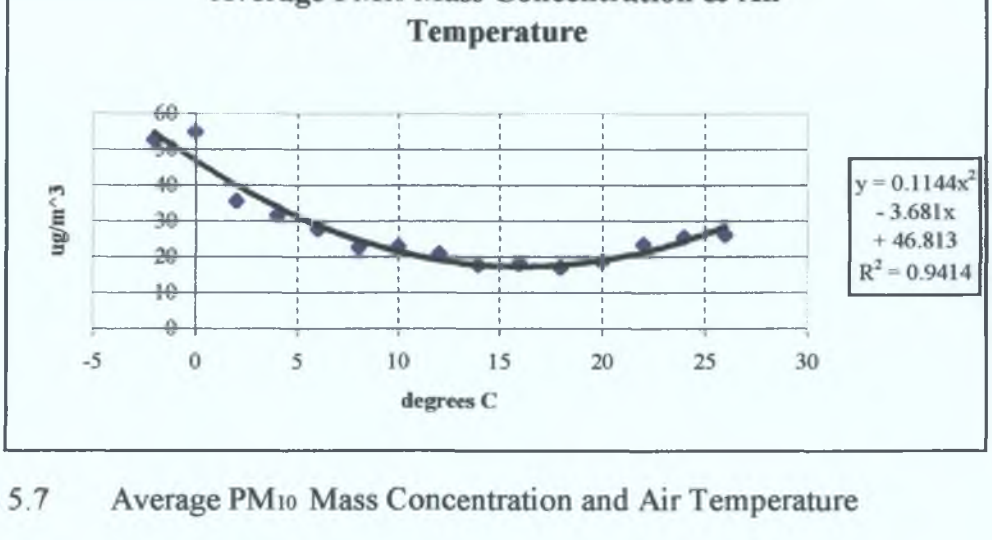

Figure 5.7 Average PM_{10} Mass Concentration and Air Temperature

Table 5.1 R^2 Values For Air Temperatures and PM_{10}

Traffic Volume	Polynomial Trend R^2 Value	Exponential Trend R^2 Value
500	0.9322	0.7642
1000	0.8753	N/A
1500	0.9318	0.5258
2000	0.9777	0.4646
2500	0.7614	0.7825
3000	0.7338	0.7816

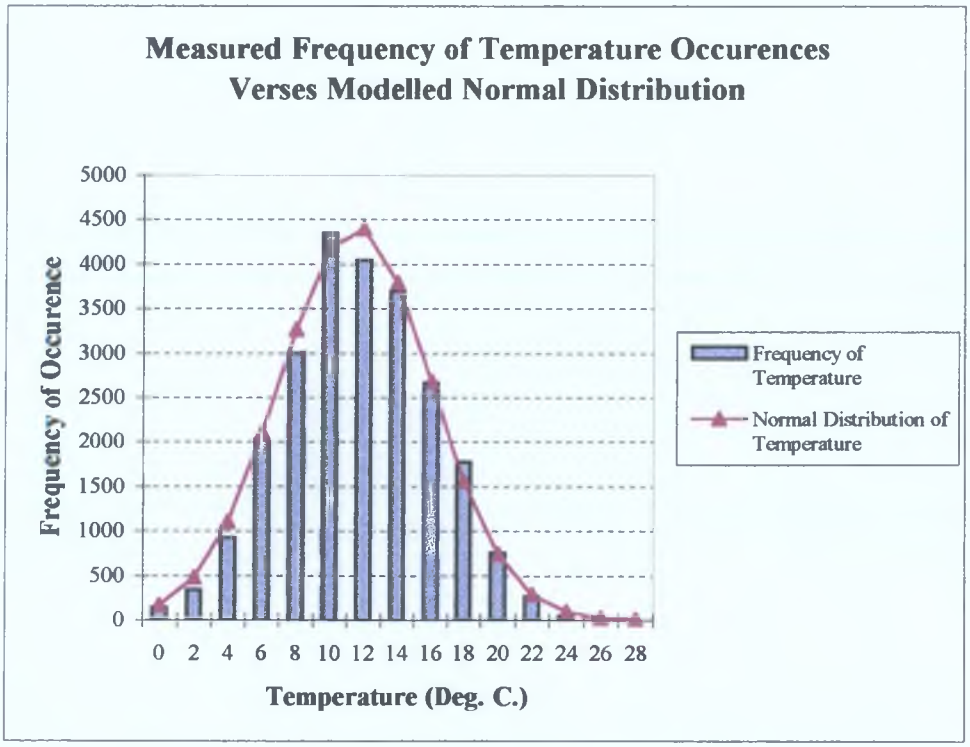

Figure 5.8 Frequency Distribution of Air Temperature Values

From figure 5.8 above it is seen that the frequency of air temperatures has a mean of 11.5°C and standard deviation of 4.5. This illustrates how moderate the climate is in Ireland as the majority of temperature readings lie between 6 and 16°C. It also indicates the extremities have fewer data points by which to model the trend.

5.4.4. The Effect of Wind Speed on PM_{10} Mass Concentration

As previously mentioned wind speed has the effect of increasing the dispersion and dilution of pollutants. To model the trend between PM_{10} and wind speed regression analysis is applied. As with humidity and temperature, the half hour PM_{10} values from each traffic volume category were filtered with respect to wind speed with increments of 0.5 m/sec. The average PM_{10} mass concentration value for each wind speed increment is then isolated.

When the data sets are plotted the polynomial regression model best fits the data. This plot is presented in figure 5.9. The average PM_{10} mass concentration over all traffic volumes plotted against wind speed produces an R^2 value of 0.8764. Table 5.2 presents the R^2 values produced for each category of traffic volume. As the wind speed increases the PM_{10} level tends to decrease, but when the wind speed gets to a certain level (approximately 6 m/sec) the pollutant level begins to increase again. This effect is strongly seen in times of dry conditions and higher air temperatures when particles may be re-circulated.

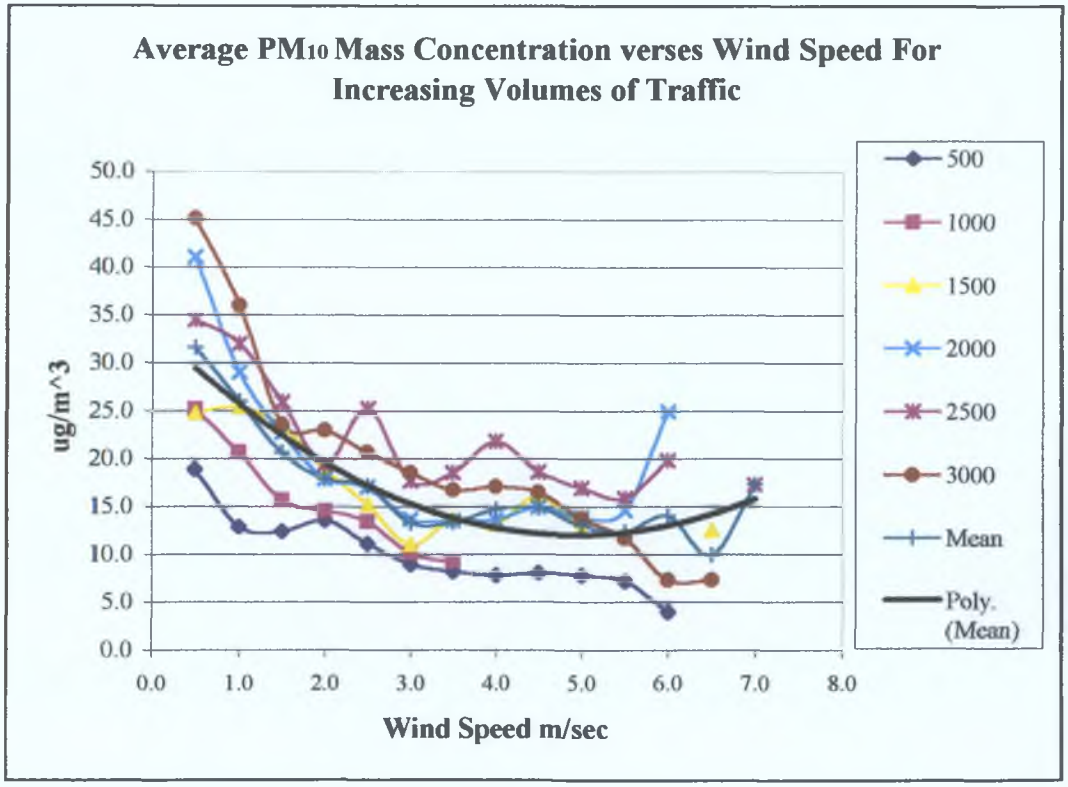

Figure 5.9 Plot of PM_{10} Mass Concentration and Wind Speed for Each Traffic Volume Category

Table 5.2 R^2 Values for PM_{10} and Wind Speed in each Traffic Volume Category

Traffic Volume	500	1000	1500	2000	2500	3000	Mean Over All Volumes
R^2 Value	0.8764	0.977	0.8344	0.9364	0.8163	0.8901	0.8764

5.4.5. The Effect of Wind Direction on PM_{10} Mass Concentration

The measured level of PM_{10} is highly dependent on wind direction, this is one of the most site-specific parameters (Berkowciz *et al.*, 1996). Wind direction has the effect of indicating the area of highest impact as a result of an air pollution episode. That is if the prevailing wind direction is from a southerly direction then the impact will occur north of the point of emission.

The position of the monitoring equipment at this particular site has shown an almost discrete number of wind directions, this is not surprising considering the wind is being channeled around buildings and the effect of the street canyons. Chapter 4 has already examined the differences between the local wind direction and the measured regional directions. The mean wind direction for both local and regional measurements are in good agreement, but there appears to be a lot more variation about this mean direction in the local data. This variation is seen as a result of the urban setting.

The wind direction data from this monitoring program indicate that the wind direction is predominantly from the south and south-west. Thus the majority of the time the pollutant was directed away from the monitor rather than towards the monitor, thus a true representation of the actual pollutant is difficult to establish.

To observe the effect of wind direction on PM_{10} mass concentration numerous wind roses were plotted. A wind rose illustrates pictorially the PM_{10} values measured verses wind direction. A 45-degree spacing has been used in the wind rose. The data was filtered as described in section 5.4.1. This allowed the effect of wind direction to be observed for constant wind speed and pollution level. This resulted in a series of approximately 72 wind roses. Figure 5.10 presents a wind rose, here the average PM_{10} mass concentration value for each wind direction over all traffic volumes is plotted. A wind speed of 1.5 to 2.0 m/sec was chosen to be the most representative wind speed as it is the most frequently occurring. The shape of this rose indicates that the highest levels of PM_{10} are recorded when the wind direction is from the north-east, and the second highest levels occur when the wind is from the north and east. When the wind is from these directions a true representation of the amount of pollution generated is by the traffic is given, as the pollution is going directly to the receptor i.e. the PM_{10} monitor.

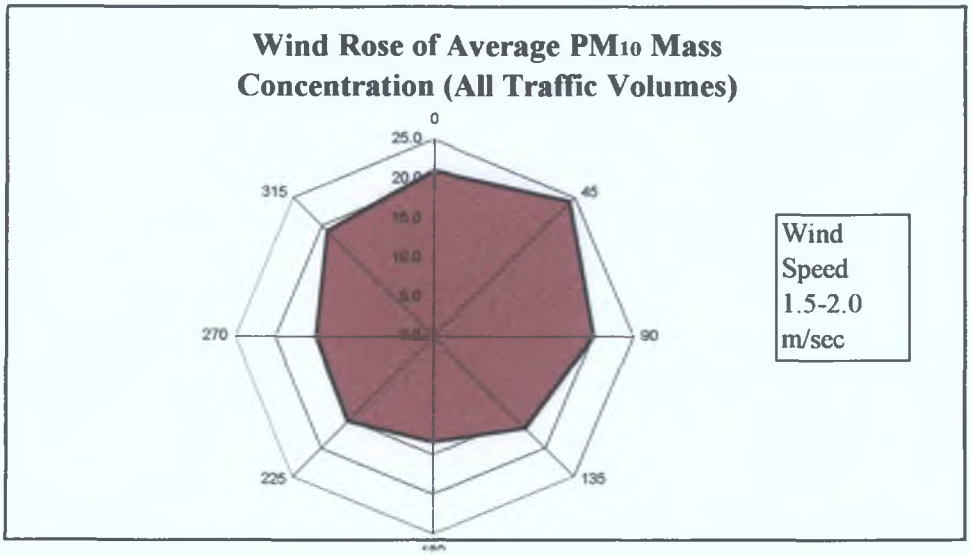

Figure 5.10 Wind Rose of Average PM_{10} Mass Concentration Values

At very low wind speeds or on occasions of no wind speeds at all, a wind rose is seen to take the shape presented in figure 5.11. Such incidences rarely occur. This plot shows that on such occasions that an almost equal level of pollution will be picked up from any direction when the effect of wind speed and in turn direction are removed from PM_{10} measurements. The highest PM_{10} mass concentration still tends to occur when the wind direction is from the north-east and east.

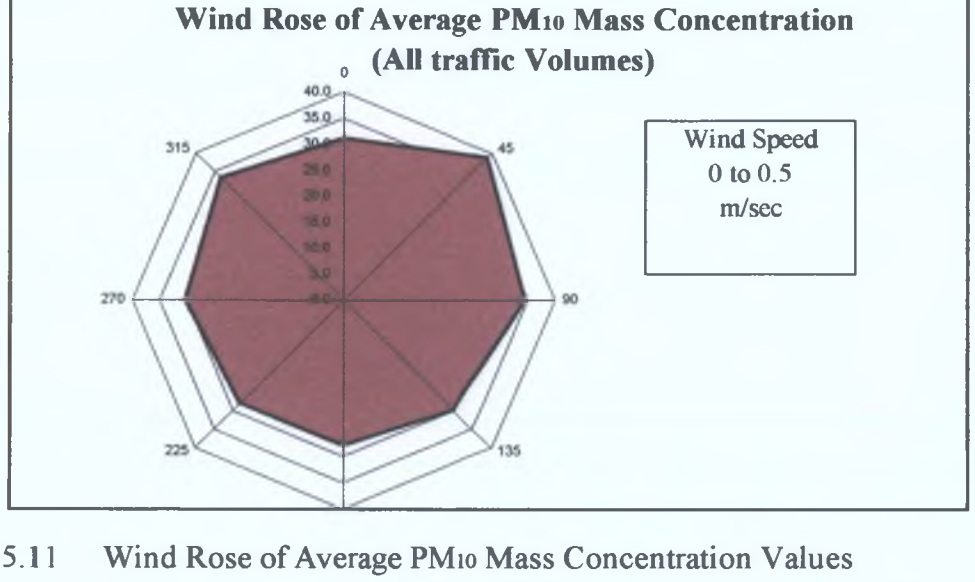

Figure 5.11 Wind Rose of Average PM_{10} Mass Concentration Values At Low Wind Speed

5.5 Discussion of Data Analysis

The variables that have been analysed in the preceding sections of this chapter are discussed here. The variables having the greatest influence on the measured level of PM_{10} at this intersection are identified.

5.5.1. Temporal Trends In The Data

The temporal trends presented in section 5.2 show the measured levels of PM_{10} to be very closely related to changes in traffic volumes. The movement of people expressed as the variation in traffic volumes and flow patterns has a significant effect on the level of measured PM_{10}. This is particularly evident in the differences that occur between week-day and weekend day PM_{10} levels. In section 5.2.3 the seasonal variation in PM_{10} levels was investigated. There appears to be a slight seasonal variation particularly seen in the evening and night-time levels of PM_{10}, this variation is a result of varying weather conditions between the summer and winter months rather than changes in emission patterns. The higher frequency of temperature inversions and lower mixing heights during winter months will inhibit the dispersion of the pollutant. When investigating the effect of air temperature on PM_{10}, the measured value seemed to increase at higher air temperatures, such higher temperatures occurring during the summer months. Eventhough the mixing height is higher during the summer months such dry warm conditions seem to result in increased levels of PM_{10}, this is most likely due to re-circulation and re-suspension of dry particles (Lam *et al.*, 1999). Also there is generally less rainfall during the summer months, precipitation plays a significant role in the removal of particles from the atmosphere (Seinfeld, 1986).

5.5.2. Pollutant Source (Motor Vehicles)

Very good correlation was found between the measured PM_{10} mass concentration values and the volumes of motor vehicles. An R^2 value of 0.91 resulted from the linear regression model and 0.93 from the polynomial regression model. When the data was further analysed to assess the effects of the meteorological variables, the effect of motor vehicles on PM_{10} levels was also evident. The data analysis involved filtering the data based on traffic volumes into categories ranging from 500 vehicles to 3000 vehicles. Each traffic volume category in increasing order corresponded to higher levels of PM_{10}. From evaluating the temporal trends and the effects of traffic volumes on PM_{10} concentrations it is evident that transport is the major source of source of particulate air pollution at this intersection.

To ascertain the exact contribution made by motor vehicles to the overall PM_{10} level would require the use of a receptor modelling technique. Such a technique attempts to approximate the contribution of each source to the overall pollutant level (Zannetti, 1990, Stedman and Dore, 1998b). A more simplified approach is to subtract the background level from the measured pollutant level. Such an approach could be applied to this intersection, as there appears to be no other major contributor to the PM_{10} levels. The background level could be taken as the level that is measured when no motor vehicles are passing through the junction. The monitoring data has shown that such a situation does not occur. The PM_{10} levels recorded at 5:00 am each day possibly approximate background levels as the lowest traffic volumes are recorded at this time. At these times motor vehicles are most likely traveling at higher speeds than during the day, turbulence induced by higher vehicle speeds will have a positive effect on pollutant dispersion and dilution (Berkowicz *et al.*, 1996). This will result in less pollution from the same volume of traffic with engines idling at the intersection. The nearest urban background measurements to this intersection are those recorded at The Phoenix Park by Dublin Corporation (Dublin Corporation, 1999). Average monthly background levels measured at this site for 1999 range from 15 to 20 $\mu g\ m^{-3}$ (Dublin Corporation, 1999).

It is difficult to assess if these background levels are representative of the background levels occurring at the intersection in this study. The distance between the two locations is approximately 3 miles. The nature of the two locations is very different, this may result in more natural sources, such as soil-like particles contributing to the PM_{10} levels at the background location. Such sources are not present at the Pearse Street and Westland Row intersection. Furthermore, a gravimetric method (R&P Partisol Air Sampler) is employed for the measurement of PM_{10} at The Phoenix Park (Dublin Corporation 1999), it has been previously highlighted in section 2.6 of chapter 2 that such gravimetric methods tend to record higher levels of PM_{10} than the TEOM instruments. Possibly the only means by which to assess the true background level is to restrict the movement of motor vehicles over a specific time period, a situation that is unlikely to occur.

5.5.3. Meteorological Effects on PM_{10} Mass Concentration

1. Percent Relative Humidity and Air Temperature

The effects of percent relative humidity and air temperature were presented in section 5.4.2 and 5.4.3 respectively. Very low correlation was found between percent relative humidity and PM_{10} mass concentrations over the entire range of data. The data sets did fit the polynomial regression model quite well when the least frequently occurring percent relative humidity measurements were excluded. No conclusions were drawn on the exact nature of the relationship.

Air temperature seems to be well correlated with the PM_{10} mass concentration measurements, with increasing temperature the levels of PM_{10} seem to decrease to quite low levels. There may be a number of reasons to explain the apparent strength of this relationship.

- The higher ambient temperatures occur during the summer months when the mixing layer is higher than it is during the winter months, therefore a larger volume is available for dispersion and dilution of pollutants and lower levels of PM_{10} are recorded.

- During the winter months there are more periods of stagnation and greater occurrences of temperature inversions, such conditions inhibit effective dispersal and dilution of PM_{10}, therefore higher levels of the pollutant are measured. Both of these facts are well documented in air pollution literature (Seinfled, 1986, Croxford et al., 1996, Sini et al., 1996).
- The prevailing wind is from the south-west, bringing warm and moist air. However, colder winds from the north are more in the direction of accurate source to receptor measurement pathway. Hence higher PM_{10} values tend to be recorded in colder periods, especially in extremes of cold.

On review of these points the air temperature does not seem to have a critical effect on PM_{10} levels at this particular location. If there was a greater volume of data available, it may be possible to isolate only those temperatures that occur when the wind direction is directing the pollution at the receptor (i.e. the PM_{10} monitor) and then a better representation of the effect of temperature could be investigated. Form the data available in this study when isolating temperatures from a north-east direction the data range is too small to investigate the relationship thoroughly.

It is seen that in figure 5.6 the polynomial curve is

A. relatively linear across the average air temperature range of 6 to 16 degrees and
B. based on far fewer measurements in the extremities of hot and cold weather.

Considering the points above and the frequency distribution of temperature in figure 5.8, it is therefore reasonable to assume that PM_{10} measurements do not have any major dependence on air temperature in this model.

2. Wind Speed and Wind Direction

The relationship between PM_{10} mass concentration and wind speed produced good correlation. The trend of decreasing PM_{10} levels with increasing wind speeds and then a tendency of the pollutant levels to increase again when wind speeds get to a higher level has been noted in air pollution literature (Lam *et al.*, 1999)

The wind roses presented above in section 5.4.4 show that the most realistic source to receptor relationship occurs when the wind direction is from the north, north-east and east. This raises the question as to how representative the measured levels of PM_{10} are of the amount of pollution from the intersection when the wind blows from other wind directions. The highest PM_{10} levels are recorded when the wind is from the north, east and in particular from the north-east. From the north-east the wind is coming right from the center of the traffic junction. The highest levels are also noted to occur during the winter months, when the majority of the north-easterly winds occur. When wind is from other directions particularly from the south and south-west the pollution is being blown away from the PM_{10} monitor. As a result the actual levels due to the traffic from the junction are not recorded. Perhaps if the receptor (i.e. the PM_{10} monitor) were at the opposite side of the junction higher levels during the summer months would be measured. Siting the monitoring equipment at the opposite side of the intersection is not possible due to security problems and the availability of power supplies.

At the beginning of this chapter it was highlighted that air quality assessment in an urban area is often based on only a few urban monitoring sites, therefore it is important that the measurements made at these sites are representative of the actual air quality. The effects that buildings and street-geometry have on wind flow are very important in this assessment. For this reason the emphasis is placed on wind speed and wind direction in this study. Air temperature is also affected by buildings and street-geometry and it too has an effect on pollutant levels but to investigate air temperature thoroughly detailed information of atmospheric stability is required (Sini *et al.*, 1996). This is beyond the scope of this study.

The intersection at which this study was undertaken has some of the characteristics of a street canyon. The approach from Westland Row and Pearse Street East can be considered to have canyon characteristics and Lombard Street to a lesser extent. The remaining corner of the junction backs onto the north-east Trinity College car park, which is an open flat area. This type of intersection layout and the associated meteorological conditions may result in high spatial variation in PM_{10} levels. To provide an accurate and detailed description of the effect of this street-geometry on the wind flow pattern would require some of the following methods to be applied. Modelling the flow and dispersion around arbitrary, complex street-building geometry, wind tunnel simulations and computational fluid dynamics (Scaperdas *et al.*, 1999, Berkowicz *et al.*, 1996, Hao *et al.*, 2000, Demirci *et al.*, 2000). Such methods are beyond the scope of this study.

From the findings of studies by these authors a more simplified approach was adopted to assess the representativeness of the measured PM_{10} mass concentration values, based on the wind flow pattern at this intersection. This approach is based on the assumption that the true pollution level is recorded when the wind direction is from the north-east and the wind speed is approximately 1.5 m/sec. A wind speed of 1.5 m/sec was identified in chapter 4 as being the most frequently occurring wind speed. This approach incorporates the relationships that have been established in the previous sections for PM_{10} with the meteorological variables and traffic volumes. This will then to allow all the measured data to be adjusted accurately to be representative of the true pollution level.

5.6 Adjustment of Data

The objective is to find a true wind rose shape. This shape is different for every wind speed and every traffic volume, thus every wind rose should then be used to adjust the PM_{10} measurement differently for all wind speeds and all traffic volumes. This calculation task would be phenomenal and would require a software program to calculate. The intention now is to produce a shape that closely matches the effect of measuring PM_{10} levels at varying traffic volumes irrespective of wind speed.

This is performed as follows:

1. Find the shape of the wind rose for every traffic volume category in steps of 500 vehicles across all wind speeds. Why, because there is now a constant source of emission to characterise each wind rose shape.

2. This results in 12 separate wind roses for wind speed by 6 traffic volume categories, which is a total of 72 wind roses. From these, it is possible to find out the best average shape wind rose that reflects the effect of measuring the wind from different directions apart from down wind.

3. The best way to average such a large number of wind roses are to normalise every rose so that a comparison can be made without the effect of wind speed being included. That is if there are small PM_{10} levels measured around the 360 degrees of the wind rose due to high winds then this cannot be compared to the high PM_{10} levels of the low wind speed wind rose.

4. It is known that the PM_{10} level is a true reading when the wind is coming from the north-east i.e. 45 degree angle, therefore this is the direction to which to normalize the wind roses. The normalisation process is completed as follows: All measurements of PM_{10} at 45 degree angle are divided by themselves to give 1 and each angle around the roses are divided by this measurement (PM_{10} at 45 degree), to retain the same proportion. At this stage there are 72 wind roses that are at the same scale and they can now be compared.

5. For each given traffic volume, there are 12 roses with varying shape. The question here is what shape is the best or most accurate. Firstly the shape that has the greatest number of data points but also each wind speed has information to contribute to the "true" shape.

6. This section is where the frequency distribution of occurrences of wind speed is incorporated. This distribution is to provide the information as to what shape is most common. The discrete distribution in figure 5.12 is now used to weight how much each wind rose attributes to the final rose for each constant source of traffic volume. Figure 5.12 shows that the highest frequency is approximately 1 to 2 m/sec. The wind rose shapes at these wind speeds are going to be the most prevalent and are given the greatest weighting. The weighting equation multiplies all normalised wind roses by its equivalent normalised frequency distribution value (figure 5.13). These values are presented in table 5.3. The 12 wind roses are then averaged to produce a final wind rose for each traffic volume category.

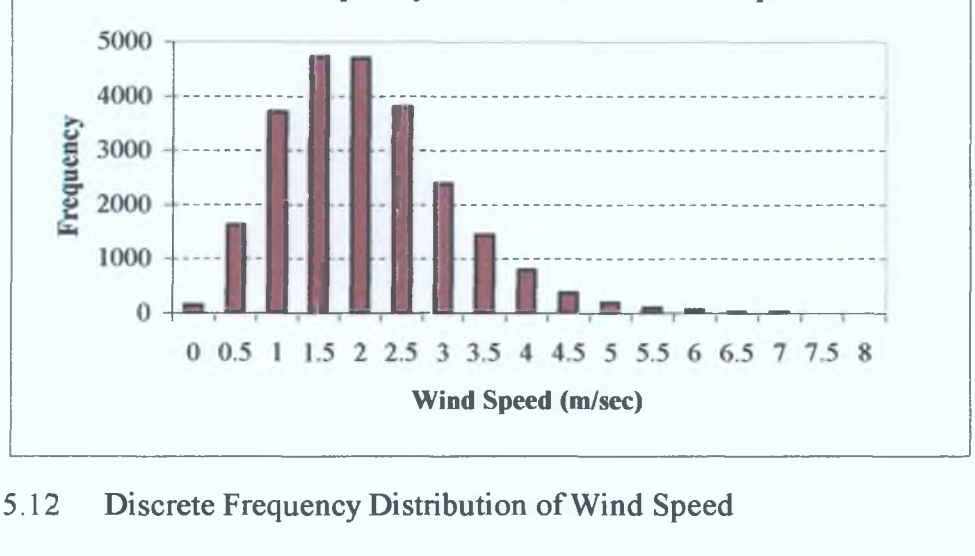

Figure 5.12 Discrete Frequency Distribution of Wind Speed

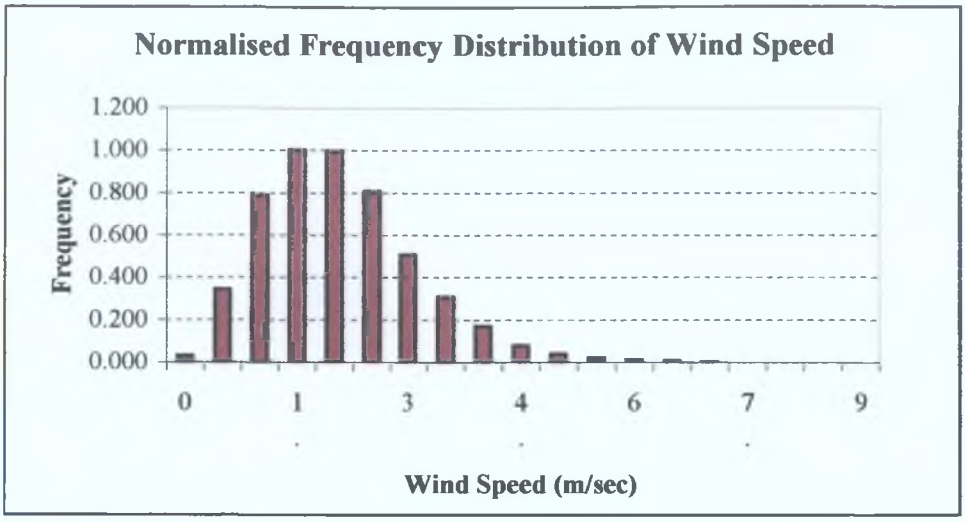

Figure 5.13 Normalised Frequency Distribution

Table 5.3 Weighting Factors for Wind Speed

Wind Speed m/sec	Weighting Factor	Wind Speed m/sec	Weighting Factor
0-0.5	0.02993	3.0-3.5	0.50605
0.5-1.0	0.342602	3.5-4.0	0.306941
1.0-1.5	0.786457	4.0-4.5	0.167905
1.5-2.0	1	4.5-5.0	0.077691
2.0-2.5	0.995967	5.0-5.5	0.039694
2.5-3.0	0.804288	5.5-6.0	0.020166

7. There are now 6 normalised wind roses for each traffic volume category that were analysed separately. The conclusion drawn from looking at all six roses is that there was not much difference between the shapes of each one. They also show that comparing low and high volumes of traffic the position of the receptor was not swamped by PM_{10} in situations of high emission or too far form the source in situations of low emission of pollutant. Figure 5.14 presents a plot of each normalised wind rose.

8. From observation it was decided to average all of the 6 normalised wind roses to one wind rose, as the individual shapes were quite similar. This led to the final wind rose shape in figure 5.15.

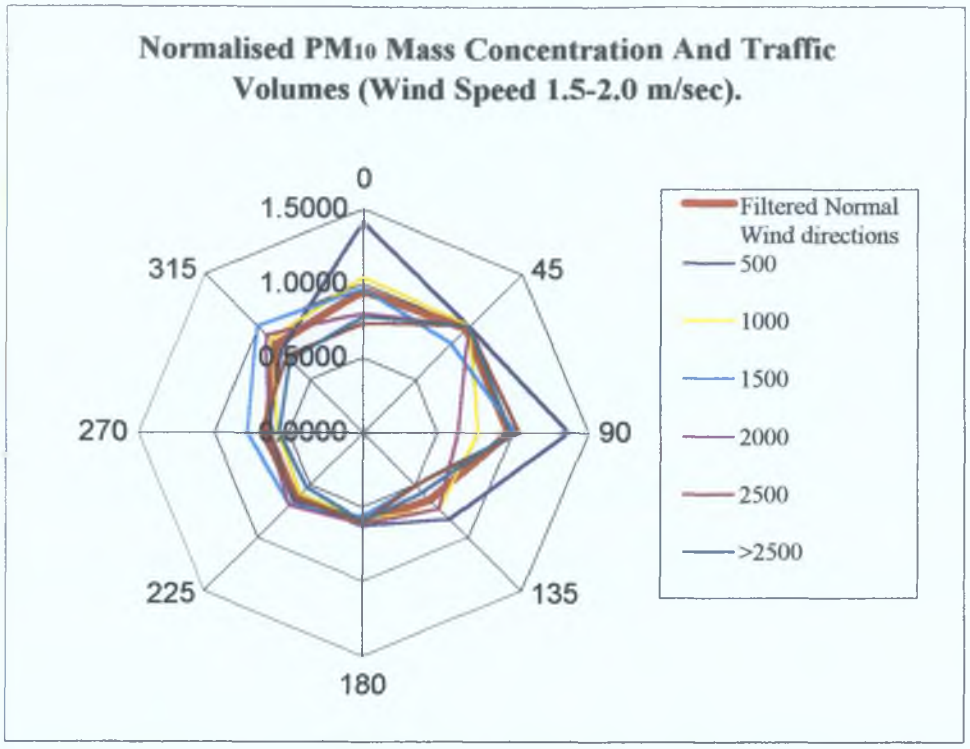

Figure 5.14 Wind Rose of Normalised PM_{10} Mass Concentration Values for all Traffic Volume Categories

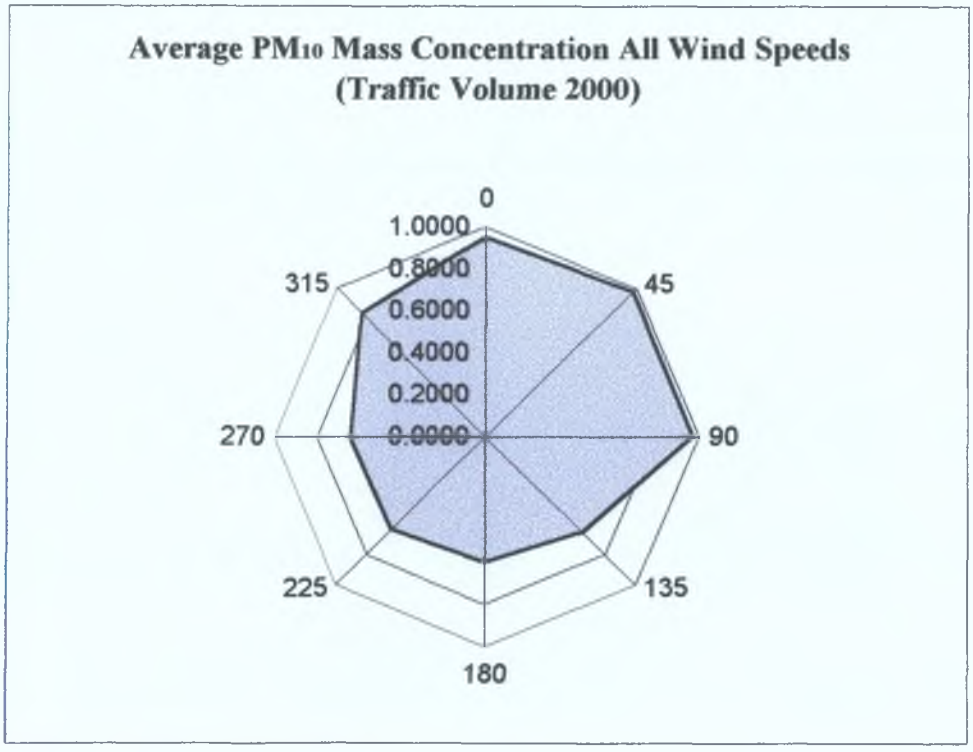

Figure 5.15 Average Normalised Wind Rose

5.6.1. Effect on PM_{10} After Adjusting Data

Having adjusted the PM_{10} data, based on the normalised wind rose it is now possible to look again at the relationship between PM_{10} and wind speed. Figure 5.16 is the plot of average PM_{10} mass concentration and wind speed for each traffic volume category i.e. pollutant source. The two sets of PM_{10} verses wind speed graphs are now available for comparison, namely the unadjusted (figure 5.9) and wind direction adjusted (figure 5.16) graphs. The comparison of the R^2 values of the fitting polynomials show a higher correlation to adjusted data supporting the principle of accounting for wind direction. Table 5.4 presents the comparison of the R^2 values for the adjusted and unadjusted data. These adjusted polynomials now become the means of prediction from here on. This means the true ambient level of PM_{10} as a result of 2500 vehicles for example at wind speed conditions of 1.5 m/sec can now be calculated.

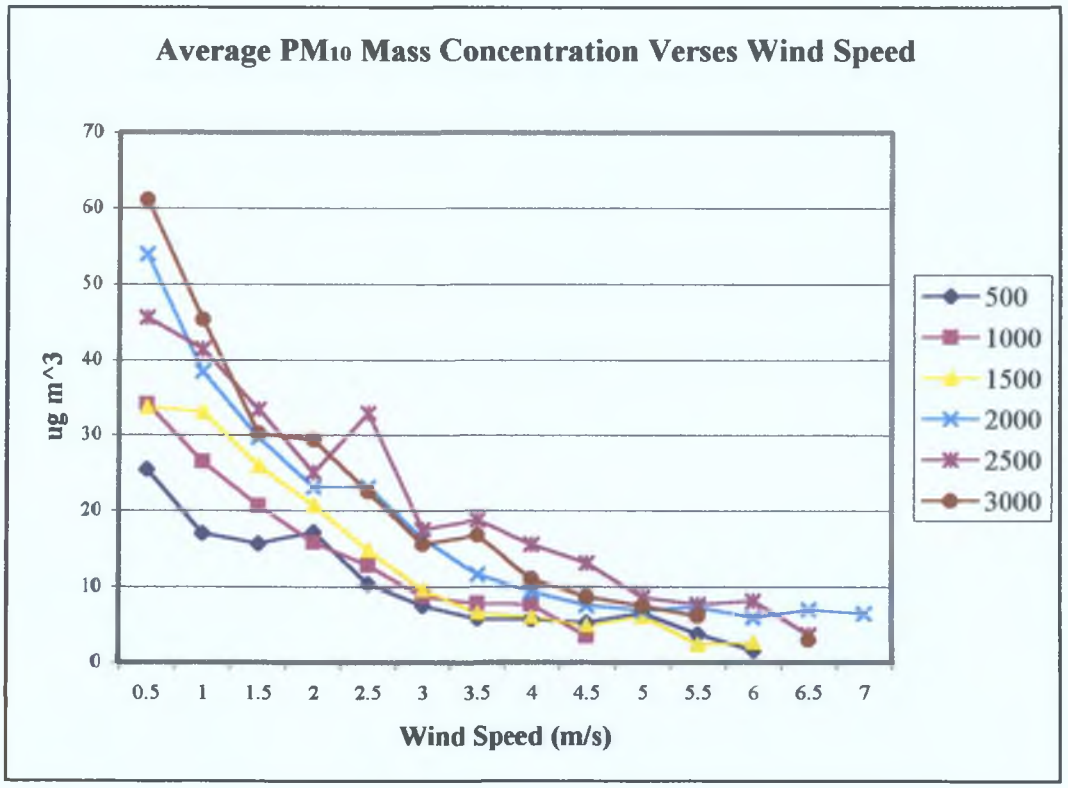

Figure 5.16 Average PM_{10} Mass Concentration verses Wind Speed for each Traffic Volume Category (Adjusted Data)

Table 5.4 Comparison of R^2 Values for Unadjusted and Adjusted Data with respect to Wind Speed.

Traffic Volume	500	1000	1500	2000	2500	3000	Mean Over All Volumes
R^2 Value Unadjusted	0.8764	0.977	0.8344	0.9364	0.8163	0.8901	0.8764
R^2 Value Adjusted	0.9261	0.9883	0.9759	0.9737	0.952	0.9627	0.9637

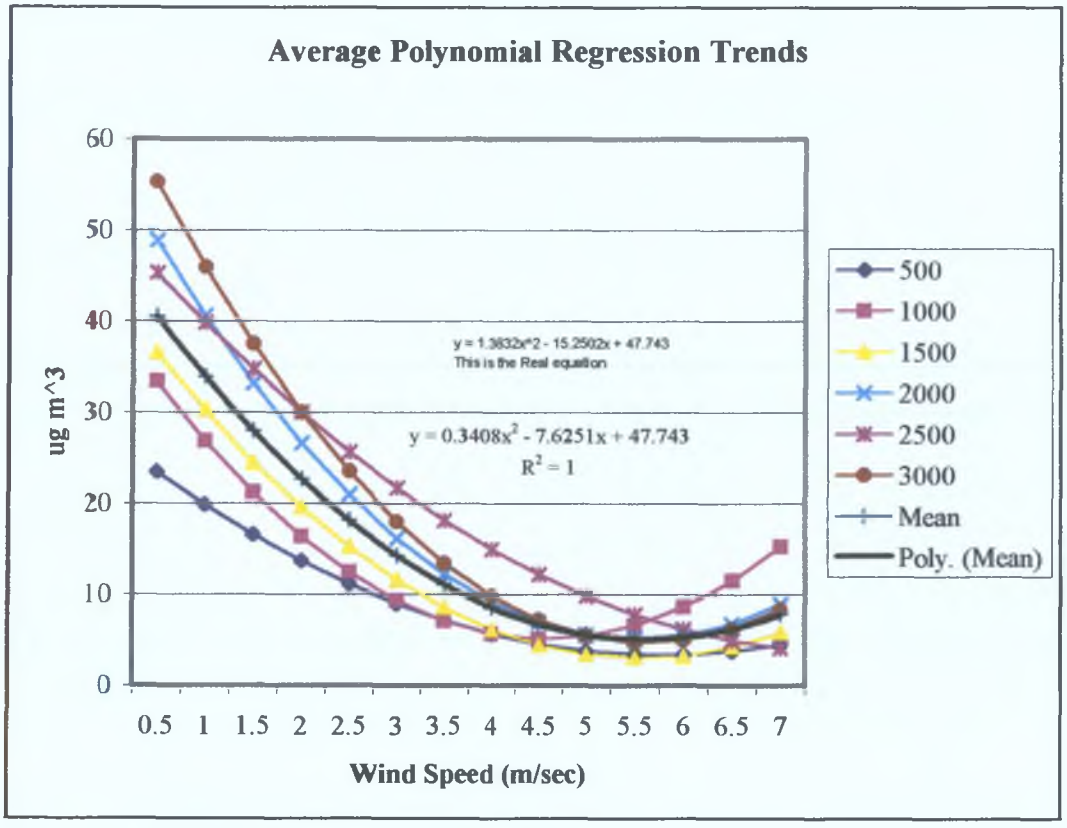

Figure 5.17 Final Polynomial Trend Lines

Figure 5.17 above represents the 7 polynomial curves that were found to best fit the measured curves in figure 5.16. The curve titled "Poly. (Mean)" represents the mean of these 7 traffic volume polynomial curves. The equation from this plot $y=1.3632x^2-15.2502x + 47.743$, can be used to adjust measured PM_{10} values for wind speed.

5.7 Measured Versus Predicted PM_{10} Values

The series of graphs presented below are a means of illustrating the accuracy of the modelling technique. The data used to compile these graphs is from the test sample of data that has not been used in developing the model. These comparisons between the unadjusted measured data and predicted data are typical examples.

Figures 5.18 and 5.20 are comparisons between unadjusted measured data and predicted data as per the developed model. Figures 5.19 and 5.21 are the corresponding scatter plots for these sets of data. The coefficients of determination i.e. the R^2 value from these plots are 0.6235 and 0.637 respectively. The prediction method can be considered reasonable as the R^2 values are greater than 0.6 that is to say that the prediction method accounts for greater than 60% of the variation in the PM_{10} value.

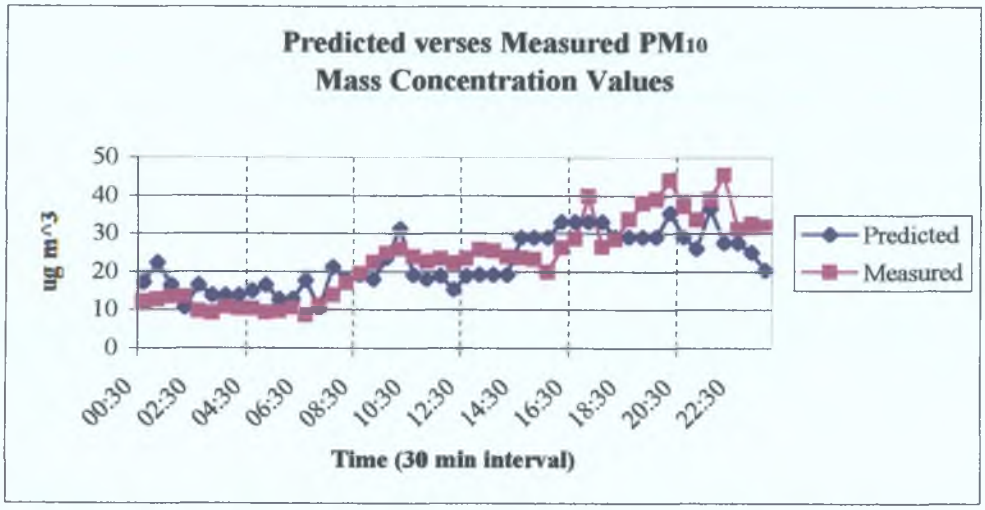

Figure 5.18 Predicted and Measured PM_{10} Mass Concentration Values

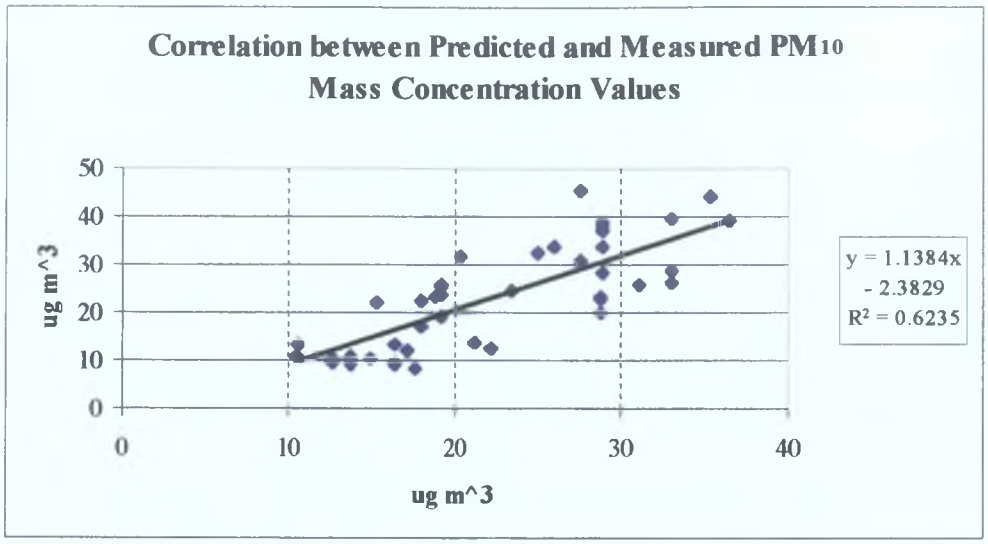

Figure 5.19 Correlation Between Predicted and Measured Values

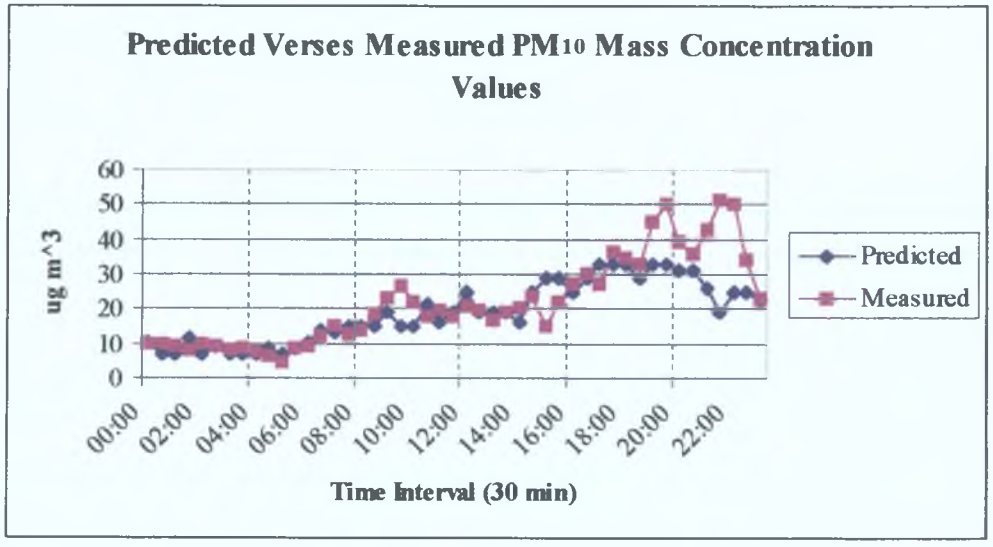

Figure 5.20 Predicted and Measured PM10 Mass Concentration Values

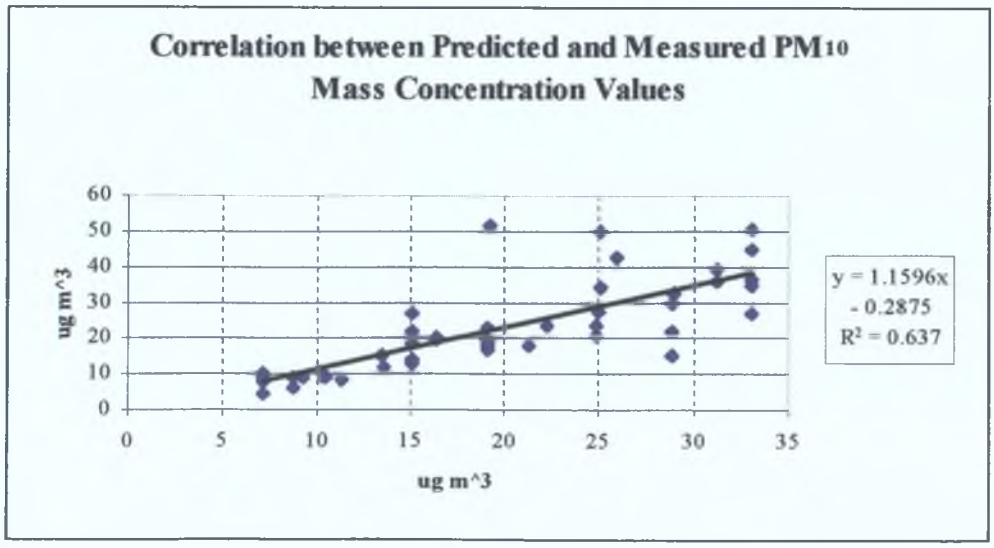

Figure 5.21 Correlation Between Predicted and Measured Values

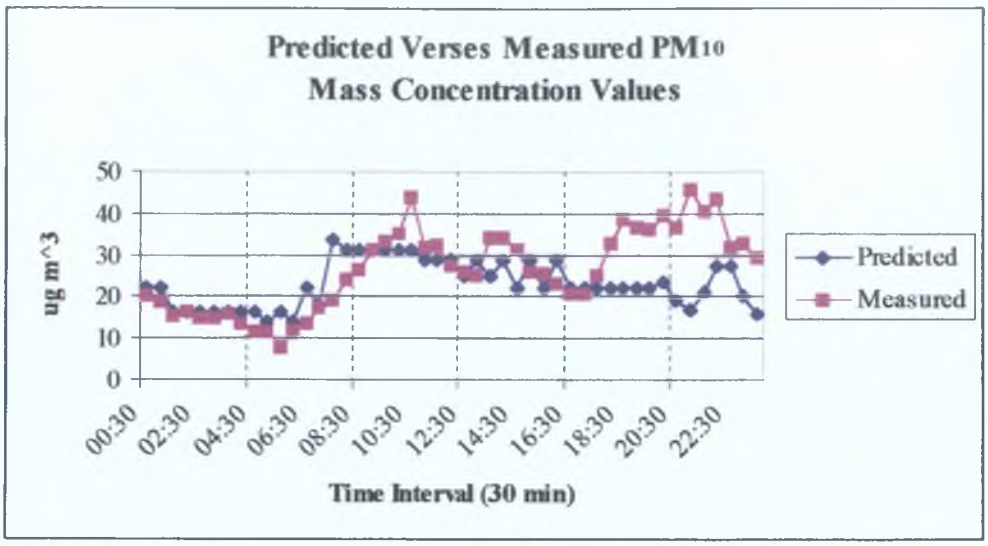

Figure 5.22 Predicted and Measured PM_{10} Mass Concentration Values

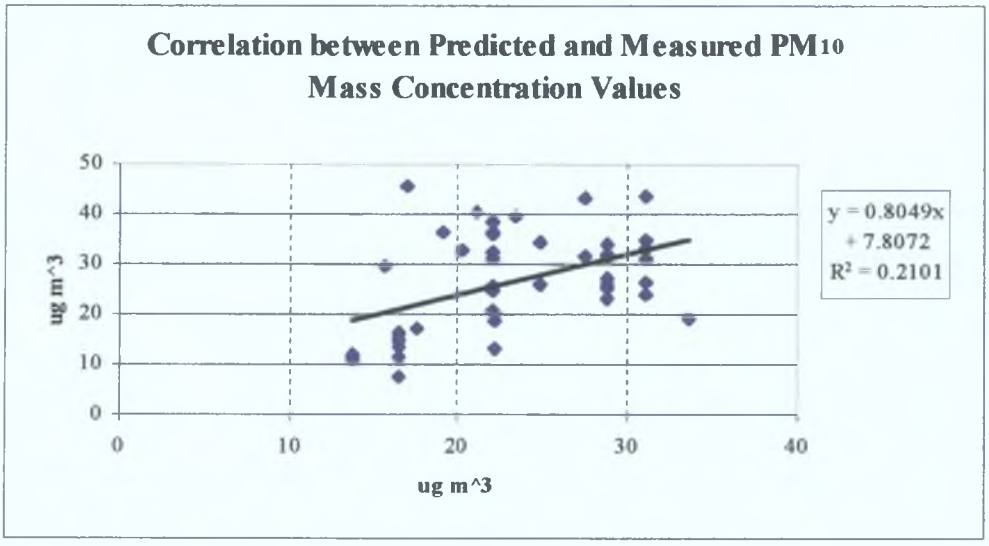

Figure 5.23 Correlation Between Predicted and Measured Values

The comparisons in figures 5.18 and 5.20 above between the actual and predicted PM_{10} values show a reasonable correlation supporting the prediction method. However, a closer examination of interesting comparisons between the measured and predicted graphs shows that on some occasions the prediction method falls down. Figure 5.22 above is one such example.

On this particular day the prediction method seemed to be reasonable up until 7:00 pm, after this time the measured value remains much higher than the predicted value. This difference is a result of very calm conditions and or the occurrence of a temperature inversion. The measured data is from the month of October when temperature inversions are known to occur at high frequencies (Crosgrove, 1997). The prediction method falls down at this point, as it does not account for air temperature and atmospheric stability. Subsequently the scatter plot presented in figure 5.23 produces a very low R^2 value of 0.21, this indicates that the prediction method only accounts for 21% of the variation in the measured PM_{10} values.

The comparison between the predicted and measured PM_{10} levels presented below in figure 5.24, indicate a delay between the prediction levels and actual levels. There is approximately a one-hour delay in the reduction in PM_{10} level in the period of 9am to 10 am. It was only when spike-like graphs displayed in figure 5.24 below was observed was the delayed response was noted. Figure 5.24 reveals that the pollution level due to traffic is not always picked up immediately by the PM_{10} monitor. If very calm and stable meteorological conditions persist, there may be some time delay in the pollutant reaching the monitor i.e. the receptor. Figure 5.25 presents a scatter plot of the measured and predicted values, as with the figure 5.23 a very low R^2 value of 0.2164 was produced.

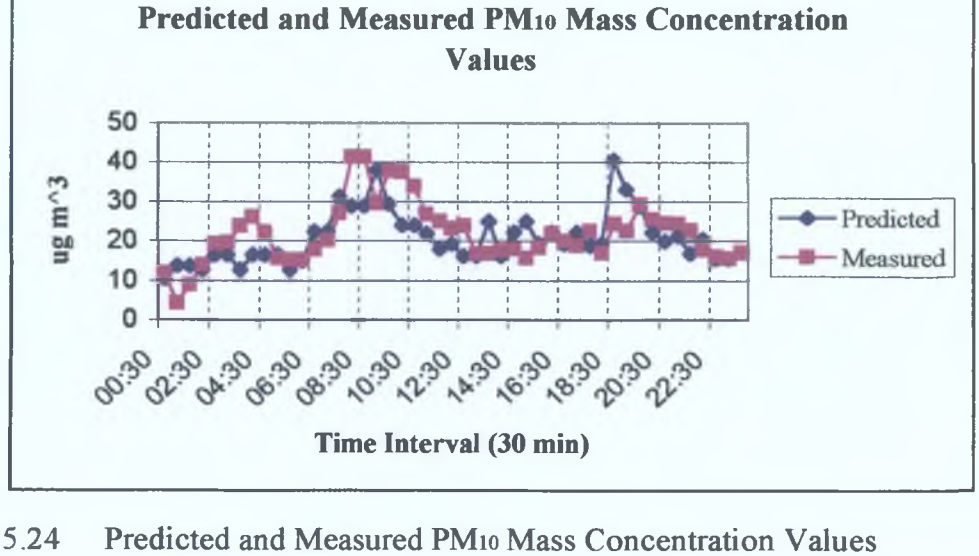

Figure 5.24 Predicted and Measured PM_{10} Mass Concentration Values

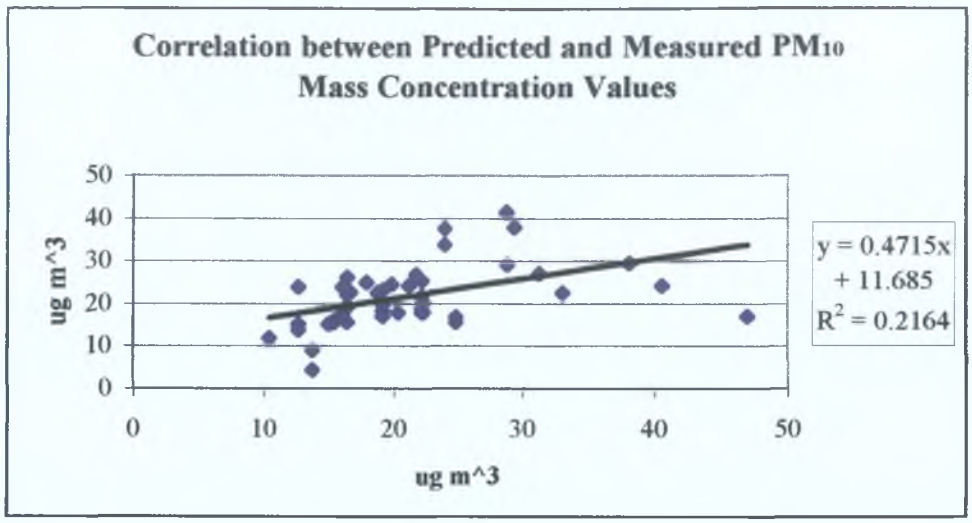

Figure 5.25 Correlation Between Predicted and Measured Values

5.8 Discussion of Model

The model presented above has been developed based on the regression analysis carried out in the preceding sections of this chapter. It has been shown here to work quite well under normal meteorological conditions i.e. in the absence of calm and stable weather conditions. In general with models of this type once they are extrapolated beyond the input data used to construct them, there is no guarantee of reliability (Chock *et al.*, 1975 also as reported by and Harrison *et al.*, 1997). The data set used here to check the reliability of the model is from the test sample, which increases the confidence in the model reliability, as it seemed to work reasonably well when extrapolated beyond the input data set.

The reliability of such a model also decreases when extrapolated to a different location, this is due in part to the effect of topography on meteorological conditions and different emission patterns (Benarie, 1980). In such situations the relationships between the pollutant level and meteorological conditions may vary. The reliability of this model to perform at a different site is dependent on employing it at a site that has similar characteristics to the intersection of this study or to apply adjustments to the model to suit a particular location.

5.9 Conclusions

The main objective of this chapter was to establish the relationship between the measured PM_{10} data, the meteorological parameters and traffic volumes. Through extensive data analysis these relationships were investigated.

In section 5.2 temporal trends in the data were established. These trends show that the PM_{10} level is greatly influenced by the volumes of motor vehicles passing through the junction on a daily basis, with week-days and weekend-days having distinct patterns. These temporal trends identify transport as the major source of pollution at this monitoring site. This is seen by the PM_{10} level closely following traffic volumes and PM_{10} concentrations dropping to very low levels at night-time. In section 5.3 it was mathematically established using regression analysis that increased volumes of traffic have a pronounced effect on the measured level of PM_{10}.

To investigate the relationship between PM_{10} levels and the meteorological variables regression analysis was again applied to the data. This revealed reasonable correlation between PM_{10} concentrations and air temperature and percent relative humidity. It was decided that in the region of interest the effect of air temperature was minimal and the correlation with relative humidity was only good for a specific range of data. It was therefore decided that wind direction and wind speed would be the meteorological parameters that most emphasis was to be placed on. These two parameters showed the greatest variation with regional meteorological data, this shows the effect that the topography of the area has on wind flow characteristics.

Investigation of wind speed established that increasing wind speed has the effect of decreasing the pollutant level to a certain point and when wind speeds increases to a certain point the pollutant level begins to rise again. This is possibly due to re-circulation and re-suspension of particles due to increased wind velocity.

Analysis of wind direction data and PM_{10} mass concentration values allows the true source to receptor relationship to be identified. This identified that when the wind was from a north-east direction a true representation of the pollutant level was established. Unfortunately the predominant wind direction is from the south-west and the location of the monitoring equipment at this site did not allow a true representation of the pollution levels to be made based on monitoring PM_{10} alone. By identifying the wind direction that gave a true representation of the pollution level due to the intersection it was possible to adjust all measured values in such a way that they became more representative of the pollution level due to the intersection

From this data analysis, a modelling technique based on wind direction, wind speed and traffic volume was developed. This model was applied to the test sample of data and comparisons were made. The model worked well in most situations. The model fails to account for air temperature and extremely calm atmospherically stable conditions when the pollution is not dispersed. Under such calm stable conditions the model was not very reliable.

The relationships established and the model developed in this chapter based on PM_{10} concentrations, traffic volumes and meteorological conditions will be used in the next chapter to calculate persistence factors for predicting the 24-hour average PM_{10} mass concentration.

Chapter 6. Calculation of Persistence Factors for PM_{10}

This chapter describes how to calculate persistence factors for PM_{10}. The persistence factors can then be used as a prediction method, to predict the 24-hour average mass concentration value of PM_{10} based on the wind direction adjusted worse case one-hour pollution episode. A persistence factor is the product of two components, a meteorological persistence factor (MPF) and a vehicle persistence factor (VPF). The MPF is designed to allow for the effect of meteorological conditions varying over the period of measurement i.e. 24 hours. The VPF allows for the change in pollutant source, i.e. traffic volumes, over the same time period. The relationships between these components and PM_{10} mass concentration have been established in chapter 5.

This method has been used in the US to relate a modelled 1-hour carbon monoxide (CO) concentration to a corresponding 8-hour average concentration (Cooper, 1989, Cooper *et al.*, 1992). Eight hours being the time averaging period on which the CO air quality standard is based. However in this study the method is applied to relate a 1-hour PM_{10} mass concentration value to a 24-hour average value, this being the time averaging period for the PM_{10} standard. The objective is to identify the 1-hour pollution episode based on the peak traffic hour. This 1-hour PM_{10} concentration is then adjusted to predict the worse case scenario from that measurement. This adjustment is necessary in situations where the wind direction is not from the source to receptor and wind speeds are higher than average. The ratio of the 24-hour average mass concentration level to this peak hour is calculated to produce the MPF. Similarly the ratio of the peak traffic hour volume to the average hourly volumes results in the VPF. Therefore, the 24-hour average mass concentration can be predicted as follows.

$$PM_{10\,24\text{-}h} = PM_{10\,1\text{-}h} * MPF * VPF$$
$$= PM_{10\,1\text{-}h} * TPF \quad \text{.....Equation 6.1}$$

Where TPF = Total Persistence Factor

The Meteorological Persistence Factor (MPF) accounts for the fact that the wind speed and direction, the ambient temperature and atmospheric stability do not persist. The Vehicle Persistence Factor (VPF) accounts for the fact that the hourly average of the 24-hour vehicle counts are always less than the peak hour vehicle count.

To calculate the MPF actual PM_{10} measurements must be used. However in using the persistence factors, prediction models are used to calculate what is expected to be the 1-hour worst case PM_{10} mass concentration. The literature available on use of persistence factors as applied to carbon monoxide, demonstrates that a 1-hour modelled CO concentration is employed. This 1-hour concentration is modelled using emission factor models and dispersion models, MOBILE3 and CALINE3 (Cooper, 1989). In this study the emission factor models and dispersion models are effectively combined in the developed prediction method outlined in chapter 5. The method used to calculate persistence factors in this study is based on the method used by Cooper, 1989 and Cooper *et al.*, 1992.

6.1 Calculation of Meteorological Persistence Factor

The meteorological persistence factor (MFP) for each day must be calculated. The method is outlined as follows:

1. Take one days PM_{10} data along with traffic volumes, wind speed, wind direction.
2. Observe the peak traffic volume hour in that same day, this identifies the peak hour for calculation purposes.
3. Adjust the measured PM_{10} level in this peak hour for worse-case conditions, i.e. low wind speed (0.5 m/sec), direction north-east, slightly stable conditions. The correction factors for wind direction are presented in table 6.1. Note the change in direction applied in order to adjust the wind direction to produce the worse case hour.
4. Change all other PM_{10} levels around by the same angle, relative to the worse case direction (north-east in this study). The values for all other measurements taken are appropriately adjusted by the same angle to allow for the change in direction. This angle of adjustment needs to be calculated for each day.

5. Exceptions to above, if 6 or more readings in the 24 hours are missing the calculation should ignore this set of data. If there is 6 or more hours of calm in that same 24 hours then ignore also.

6. These adjusted measurements are the basis for calculating the MPF discussed. The average of the 24 hours now becomes the numerator and the worse case hour is then used as the denominator in calculating the MPF. Note that the predicted worse case PM_{10} level is included in this averaging process. Each MPF is calculated by the following equation:

$$MPF = \frac{\left(\sum_{i=j}^{k} PM10_i\right) / N}{PM10_p} \quad \text{.....Equation 6.2}$$

Where: $PM10_i$ = concentration of PM10 for the *i*th hour

$PM10_p$ = concentration of PM10 for the peak hour

MPF = meteorological persistence factor for the 24 hour period.

N = number of valid PM10 measurements that 24 hour period.

7. All calculated persistence factors are then compiled in a spreadsheet to give a daily factor. From this the highest, second highest, mean and standard deviations are displayed. The MPF is achieved from this analysis.

Table 6.1 Correction Factors for Wind Direction Adjustment

Angle (degrees)	Correction factor to NE	Correction factor from NE
0	1.050455834	0.952
45	1.02620082	0.9745
90	1.026994082	0.9737
135	1.552598707	0.6441
180	1.670616179	0.5986
225	1.594042919	0.6273
270	1.562961981	0.6398
315	1.203806395	0.8307

Table 6.1 presents the correction factors by which the measured data is adjusted for wind direction to correspond to the rotation of the peak hour's measurement. The two columns in table 6.1 above are used to correct all PM_{10} levels. The need for both factors is to allow for the fact that when adjusting the level to the normal direction (i.e. 45 degrees) the PM_{10} level is increased. This process is then followed by a reduction as each new wind direction moves away from the normal direction. A sample calculation using the above correction factors is presented in table 6.2.

Table 6.2 Sample Calculation of AdjustingPM_{10} Values for Wind Direction

	RAW DATA				ADJUSTED DATA			
Hour	**Speed**	**Direction**	**Traffic**	**PM_{10}**	**Speed**	**Direction**	**Traffic**	**PM_{10}**
01:00:00	1.07	8.7	600	26.4	1.07	83.3	600	27
02:00:00	0.647	357.5	379	27.6	0.647	72.1	379	28.2
03:00:00	1.3435	0.8	225	22	1.3435	75.4	225	22.5
04:00:00	1.0205	2.8	168	15	1.0205	77.4	168	15.3
05:00:00	0.7965	352.5	143	13.2	0.7965	67.1	143	13.5
06:00:00	0.846	65.6	162	13.6	0.5	140.2	162	9
07:00:00	1.3185	358.8	491	20.1	1.3185	73.4	491	20.5
08:00:00	1.1945	350.2	1352	28.7	1.1945	64.8	1352	29.4
09:00:00	1.6175	330.4	2869	29.5	1.6175	45	2802	52.5
10:00:00	0.896	118.6	2802	35.1	0.896	193.2	2869	32.6
11:00:00	0.7465	101.3	2544	33.8	0.7465	175.9	2544	20.7
12:00:00	1.02	268.5	2253	32.8	1.02	343.1	2253	48.7
13:00:00	1.344	195.53	2211	36.3	1.344	270.13	2211	38.7
14:00:00	1.6175	193.28	2251	35.9	1.6175	267.88	2251	38.4
15:00:00	1.319	351.2	2215	34.6	1.319	65.8	2215	35.4
16:00:00	1.02	40.2	2331	30.8	1.02	114.8	2331	20.3
17:00:00	1.8415	158.5	2301	29.5	1.8415	233.1	2301	30.9
18:00:00	1.443	194.52	2243	36.1	1.443	269.12	2243	38.5
19:00:00	1.269	183.344	2101	45.9	1.269	257.944	2101	49
20:00:00	0.846	183.344	2195	67.6	0.846	257.944	2195	72.3
21:00:00	0.3735	176.4	2018	52.9	0.3735	251	2018	56.5
22:00:00	0.7465	176.4	1534	55.5	0.7465	251	1534	59.3
23:00:00	0.3235	31.2	1310	64.9	0.3235	105.8	1310	64.8
00:00:00	0.7465	46.7	934	41.7	0.7465	121.3	934	27.6

The actual angle of adjustment required to bring the worse case 1-hour PM_{10} measurement to 45 degrees, shown at 9 am in table 6.2 above, was –285 degrees. All other data was then adjusted by that angle before being adjusted by the wind rose factors. The above example resulted in the MPF of 0.676 being calculated. This process was repeated for each day in the measurement database.

Table 6.3 PM_{10} Values at Various Wind Speeds based on Polynomial Equation

Speed	Correction Factor	Speed	Correction Factor
0.5	40.4587	**4**	8.5534
1	33.856	**4.5**	6.7219
1.5	27.9349	**5**	5.572
2	22.6954	**5.5**	5.1037
2.5	18.1375	**6**	5.317
3	14.2612	**6.5**	6.2119
3.5	11.0665	**7**	7.7884

Table 6.3 above lists sample values of PM_{10} for given wind speed using the polynomial equation $y = 1.3632x^2 - 15.2502x + 47.743$ developed in chapter 5. This polynomial is used to adjust the peak 1-hour PM_{10} mass concentration from its measured level (angle adjusted) at the measured wind speed back to the predicted worse case of 0.5 m/sec. Table 6.3 is based on an average traffic volume of 2000 vehicles.

A sample of some of the MPF calculated is presented in table 6.4. The final MPF is the mean of the highest MPFs across all the months measured. The final MPF calculated from all data measured 0.849.

Table 6.4 Sample of Calculated Meteorological Persistence Factors

Week Number	Date	Worst weekly Worst factor	Mean	Standard Deviation
37	11-Sep-00	0.875	0.723	0.27
40	07-Oct-00	0.749	0.614	0.19
41	13-Oct-00	0.604	0.402	0.23
42	19-Oct-00	0.756	0.354	0.101
43	25-Oct-00	0.995	0.523	0.007
44	02-Nov-00	0.891	0.753	0.102
50	17-Dec-00	0.261	0.2	0.01
51	18-Dec-00	0.329	0.301	0.012
6	10-Feb-01	0.884	0.61	0.213
9	04-Mar-01	0.768	0.33	0.17
10	11-Mar-01	0.63	0.512	0.117
11	14-Mar-01	0.931	0.62	0.261
12	19-Mar-01	0.851	0.812	0.09
16	21-Apr-01	0.793	0.459	0.123
17	28-Apr-01	0.791	0.395	0.01
18	04-May-01	0.747	0.599	0.1
19	07-May-01	0.981	0.71	0.194

6.2 Calculation of Vehicle Persistence Factor

The vehicle persistence factor (VPF) is a measure of how the highest traffic count persists over the period of time measured, it is the ratio of the highest hours traffic volume to the average traffic volume. The VPF is calculated as follows:

1. Compile the hourly traffic counts on a daily basis.
2. Choose the hour of greatest traffic volumes. This then becomes the peak hour.
3. The average of the 24 hours now becomes the numerator and the worse case peak hour is then used as the denominator in calculating the VPF.
4. VPFs are calculated using equation 6.3 below for every day. The mean daily VPFs from these are presented in table 6.4. The mean of these seven values from table 6.4 is now the VPF, which is 0.565.

The VPF is calculated by the following equation:

$$VPF = \frac{\left(\sum_{i=j}^{k} VC_i\right) / N}{VC_p} \qquad \text{.....Equation 6.3}$$

Where: VC_i = vehicle count for the *i*th hour

VC_p = vehicle count for the peak hour

VPF = vehicle persistence factor for the 24 hour period.

N = number of valid vehicle counts in the 24 hour period.

Table 6.5 Average Vehicle Persistence Factor for Each Day

Day	Average Hourly Volume	Peak Hour Volume	VPF
Mon	3165	5741	0.551339
Tues	3187	5870	0.54293
Wed	3194	5874	0.543752
Thurs	3344	6099	0.548287
Fri	3359	5877	0.57155
Sat	2564	4131	0.620682
Sun	2182	3775	0.578183

The VPF calculated for week days are seen to be lower than those of the weekend days. The peak volume of traffic during the weekend is lower when compared to a week day. The time at which the peak hour traffic occurs at the weekend varies more than on a week day. Throughout the entire measuring period, the peak traffic hour occurs between 8:00 and 9:00 am on a week day as opposed to a weekend day when the peak hour can occur anytime between 2:00 pm and 7:00pm. The VPF calculated here is the mean of the entire vehicle persistence factor that is, 0.565.

6.3 Calculation of Total Persistence Factor

The Total persistence factor (TPF) is the product of the MPF and the VPF. From equation 6.1 the TPF is given by:

$$TPF = 0.849 * 0.565 = 0.48$$

This TPF of 0.48 can now be used in predicting the 24-hour average PM_{10} mass concentration from the measured 1-hour (wind direction adjusted) PM_{10} concentration. To establish the suitability of this prediction method to PM_{10}, the accuracy of the TPF developed here must be evaluated.

6.4 Evaluation of Persistence Factor Method

The total persistence factor of 0.48 was applied to adjusted peak 1-hour PM_{10} values to establish how well this method can predict the average 24-hour mass concentration. These results were then compared to the actual measurements. Samples of two separate comparisons are given in figures 6.1 and 6.3. The corresponding scatter plots for these data sets are presented in figures 6.2 and 6.4 respectively. These comparisons are made using data from the test sample that has not been used in developing the persistence factors and the prediction model in the previous chapter.

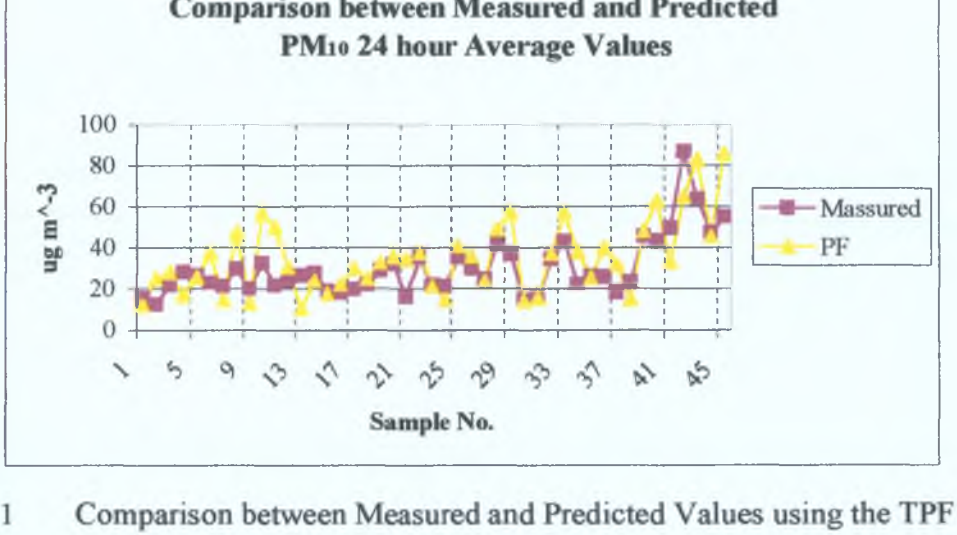

Figure 6.1 Comparison between Measured and Predicted Values using the TPF

From initial observation of figures 6.1 and 6.3, there seems to be a significant variation between the measured 24-hour average values and the values predicted using the persistence factor method. The persistence factor of 0.48 on some occasions over estimates and under estimates the 24-hour average by as much as a factor of 2. On other occasions it seems to work quite well, in particular when the measured PM_{10} levels drop off quickly approaching night-time. The correlation between the predicted and measured values is seen to be quite poor. The R^2 values for the two sets of data presented here in figures 6.2 and 6.4 are 0.58 and 0.49 respectively. These values indicate that there is significant variation between the two data sets.

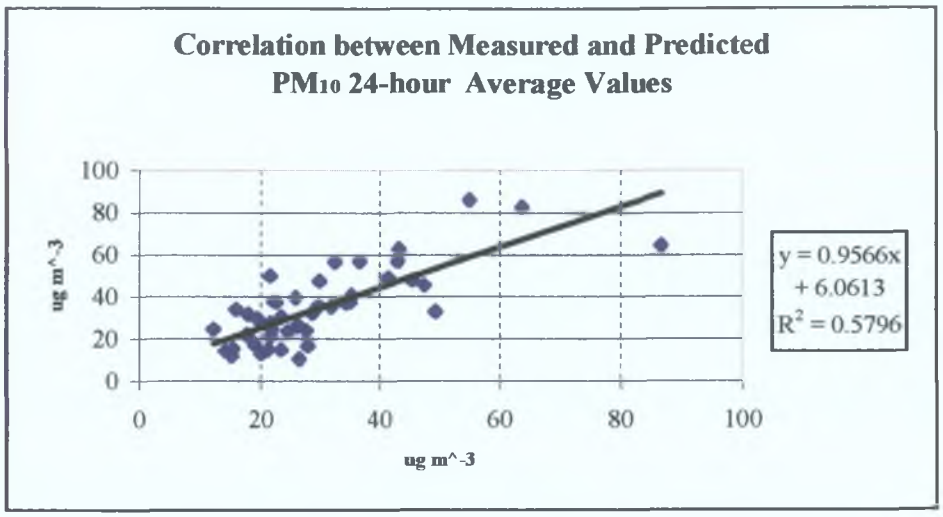

Figure 6.2 Scatter Plot of Measured and Predicted Values using the TPF

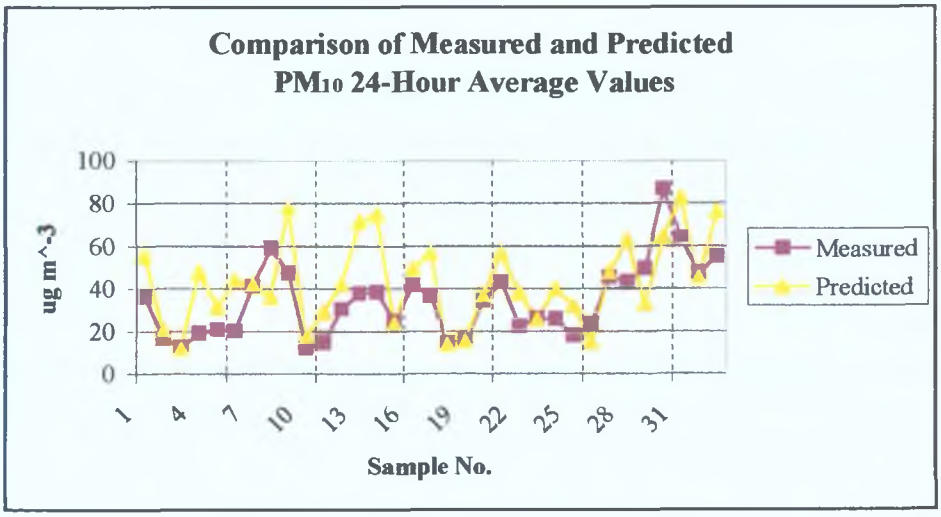

Figure 6.3 Comparison between Measured and Predicted Values using the TPF

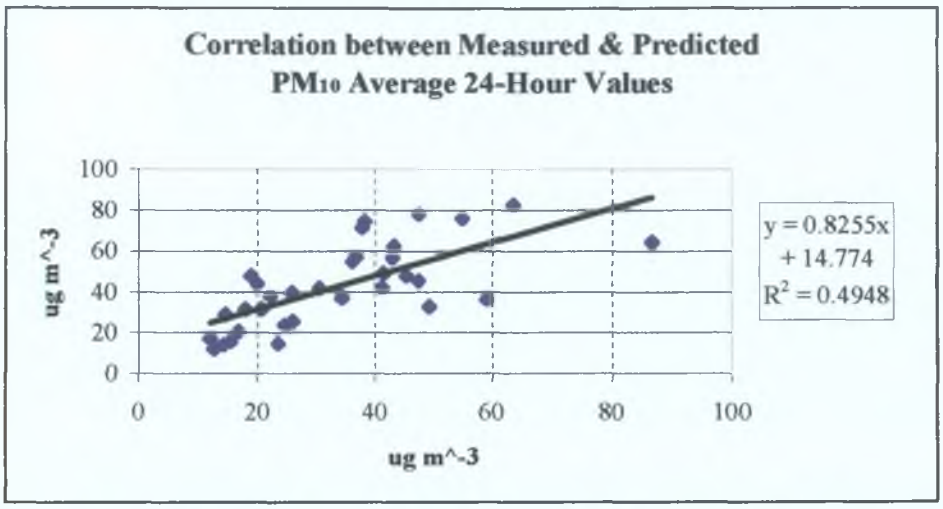

Figure 6.4 Scatter Plot of Measured and Predicted Values using the TPF

A point to note is that no similar study of the developing the persistence factor method for predicting PM_{10} concentrations has been discover in all searches of literature. The question proposed is how accurate the prediction technique is. If one is to base the decision criteria on R^2 values from the data comparison then the prediction method would not stand, given an R^2 value decision criteria of 0.6.

6.5 Discussion of Persistence Factors Method for Predicting PM_{10} Levels

This TPF of 0.48 was applied to the data to evaluate how well the method works to predict the average 24-hour mass concentration value. When the predicted value was compared to measured values the method did not seem to work very well. Literature available from the carbon monoxide study reveals that the method works quite well (Cooper, 1989). There are a number of factors that account for this method to work well for carbon monoxide and not for PM_{10}. These are discussed in the following sections.

6.5.1. Differing Characteristics of Carbon Monoxide and PM_{10}

Carbon monoxide and PM_{10} are two common pollutants generated by motor vehicles. The greatest amount of pollutant is emitted from motor vehicles when the engine is idling and decelerating. These driving modes occur during times of heavy traffic congestion and when vehicles are approaching an intersection (Butler, 1979, US EPA, 1992). These conditions are seen to occur quite a lot at the intersection of concern in this study. The intersection suffers from heavy congestion during the morning rush hour and to a lesser extent in the evenings. Up to 1,500 motor vehicles have been recorded by the SCATS system in a 15-minute time period between 8:00 and 9:00 am. Due to the nature of traffic intersections and traffic signal sequences, there is a considerable amount of decelerating and accelerating through out the day. Only late at night and early morning is the cruise driving mode possible. In this mode, the least amount of pollutant is emitted (US EPA 1992).

CO emissions result from incomplete combustion of fuel in the absence of oxygen, such emissions are highest when the engine is idling due to heavy congestion (Seinfeld, 1986, Butler, 1979). Under such conditions very high levels of CO pollution will occur, the engine exhaust gas may comprise of up to 69,000 ppm of CO (Butler, 1979). Once the congested traffic passes the levels of Co emitted decrease to low levels. The emitted Co is oxidised to carbon dioxide and in general, the pollutant episode passes very quickly (US EPA, 1992). Atmospheric processes are quite effective in removing CO, levels of CO are seen to drop off very quickly even with only slight increases in wind speed (Seinfeld, 1986).

The situation with PM_{10} is different as it may be composed of many different compounds and additional PM_{10} may be formed from reactions among gaseous emissions (Seinfeld, 1986). A typical engine exhaust gas may comprise PM_{10} of up to 4,000 $\mu g\ m^{-3}$ (Butler 1979). The measurements made during the course of this study indicate that the PM_{10} levels tend to remain suspended in air for longer periods, often noted at the onset of nightfall. Lighter night time wind speeds, extremely stable conditions and temperature inversions are known to suspend PM_{10} for longer (Seinfeld, 1986, Zannetti, 1990).

The measurement data in this study, at times indicate high levels of PM_{10} persisting for sometime after the high volumes of traffic have passed. These high levels of PM_{10} are not removed as quickly from the atmosphere as CO when compared with the Florida study. Under such conditions, higher MPFs were seen to be calculated, lower MPFs resulted on occasions when the PM_{10} concentration dropped to low levels at night time. This has the effect of moderating the mean MPF.

The method of calculating total persistence factors for CO tends to ignore data when very calm and stable weather conditions occur (Cooper, 1989). Such calm and stable conditions have been seen here to be very relevant to PM_{10} levels. It should be noted that calm stable weather conditions will inhibit the effective dispersal and dilution of almost every air pollutant, but such conditions tend to have a greater effect on particle removal from the atmosphere (Butler, 1979). This is most likely due to the diverse nature of PM_{10} composition.

In the CO monoxide study, details of stability class along with wind speed conditions were included in calculating the TPF. In this study wind speed conditions were included but stability class measurements were unavailable, therefore such effects could not be accounted for in the calculation. The result of omitting stability class measurements does not allow for the carry over of pollutant from the day time into the night time to be accounted for, this carry over seems to be quite relevant to the PM_{10} measurements. Figure 6.5 below taken from chapter 5 is a comparison of the measured and the modelled PM_{10} values based on the relationships derived in chapter 5. This graph clearly displays the levels of PM_{10} remaining high into the night. The developed model allows for higher PM_{10} levels in lower wind speed conditions but does now allow for the carry over from one hour to the next of high levels when a stable condition is present.

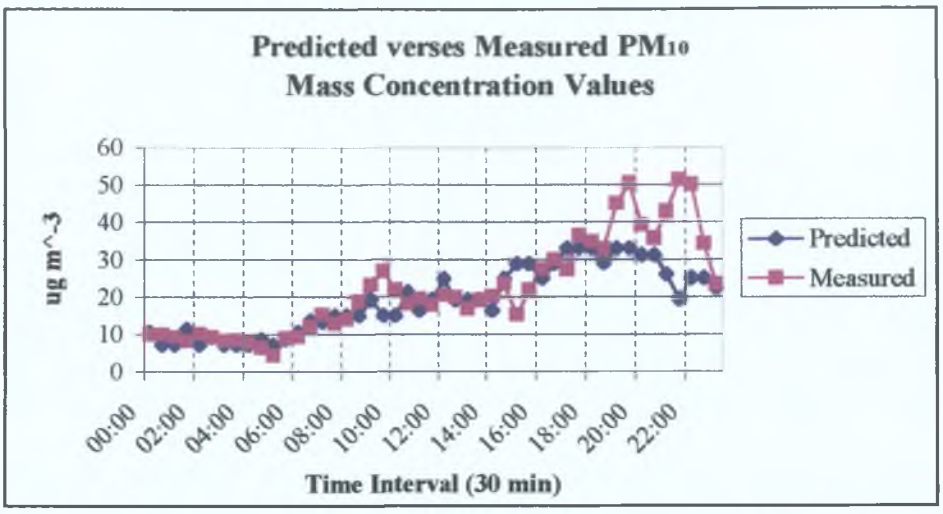

Figure 6.5 Predicted verses Measured PM_{10} Mass Concentration Values

6.5.2. The Effect of the Time Averaging Period

The averaging period used in the CO study was over 8 hours, as noted earlier this is the time averaging period set down in the CO air quality standard. In this study, the time averaging period stated in the PM_{10} air quality standard of 24 hours was adopted. This implies the 1-hour measurement has to account for 3 times the temporal variation than that of the CO study. The traffic count is greatly reduced during nighttime and in doing so reduces the VPF. This study produces a VPF of 0.565 over the 24 hours. When this is compared to an 8-hour day time averaging period in the CO study persistence factors in the order of 0.84 were calculated.

6.6 Conclusions

The persistence factor method seems to work on the assumption that all other levels of pollutant measured through out the day will be considerably lower than the peak pollution hour. This is relevant to CO as the level decreases to a reasonably low level outside of the peak pollution hour. In this study, after investigation of the nature of PM_{10} pollution and the effects of meteorology, such an assumption cannot be applied. Outside of the peak hour the level of PM_{10} can still be quite high if meteorological conditions are not conducive to effective dispersal and dilution of the pollutant.

Chapter 7. Conclusions

This thesis set out to achieve two main objectives, namely the investigation of the factors that affect the measured ambient level of PM_{10} at a busy city centre intersection and the application of the persistence factor method to PM_{10} prediction. These objectives were achieved through extensive data analysis and interpretation.

The data analysis revealed high temporal variation in the measured daily PM_{10} value. This variation was observed to be greatly influenced by the volumes of motor vehicles passing through the intersection. A slight seasonal variation was observed in the PM_{10} values, this was due to variations in meteorological conditions rather than variation in pollutant source. The urban site was seen to have an effect on the regional meteorological conditions. Significant variations between the regional and local wind speed and direction data were observed. This type of variation between regional and local meteorology was not observed among the air temperature and percent relative humidity measurements.

Motor vehicles were seen to be the predominant cause of pollutant variation with wind speed and wind direction largely accounting for the remaining variation. In attempting to isolate the effects of air temperature and percent relative humidity on the measured pollutant value no relationship was concluded. Based on the relationships established between traffic volumes, wind speed and wind direction it was possible to adjust the data so that it became more representative of source to receptor impact. The model developed allowed for the calculation of both MPF and VPF to achieve an overall TPF that could be used to predict the 24-hour average PM_{10} value. The TPF calculated was 0.48. It was established that this method is not ideally suited to the prediction of PM_{10}, due to the nature of the pollutant itself.

The main conclusions drawn from this study

1. The current levels of PM_{10} measured at this intersection have increased significantly from the measurements made since 1998. As of yet the air quality standard for particulate air pollution has not been breached.

2. The ambient PM_{10} level exhibited distinct temporal characteristics that were greatly influenced by the traffic flow characteristics through the intersection.

3. Seasonal variation was identified but not to any great extent, this was seen to be influenced by the meteorological variations between winter and summer months.

4. Motor vehicles were observed to be the main source of particulate air pollution at this intersection.

5. The street geometry and urban structures were observed to have an influence on the regional meteorological parameters resulting in different local micro-meteorological conditions. The effects of topography having the greatest influence on wind flow patterns.

6. The measured level of PM_{10} was greatly influenced by the wind flow patterns at the junction. PM_{10} was well correlated with wind speed. The relationship between PM_{10} and wind direction allowed the true source to receptor relationship to be identified.

7. Based on the evaluation of air temperature and percent relative humidity on PM_{10} levels the effects were deemed negligible.

8. The model developed based on wind speed, wind direction and traffic volume was shown to predict with reasonable accuracy the ambient level of PM_{10}.

9. The persistence factor method was deemed to be unsuitable for the prediction of the daily PM_{10} value due to the nature of particulate air pollution and the time averaging period required.

10. The study has also found the sheer immensity of calculations (approximately 1,128,950 data points) required in conducting this project should incorporate the skills of a software programmer.

11. The true effect of topography on meteorological conditions and in turn on pollutant level was difficult to assess as the monitoring equipment represented only one receptor point. To assess the spatial variations in pollutant level that result from such effects would require the co-locating of monitoring equipment at a number of points around this monitoring site.

Recommendations For Further Study:

1. Investigation of the spatial variability in pollutant level through the co-location of pollution monitors at individual sites.

2. Simultaneous measurement of PM_{10} by means of the TEOM and Gravimetric method, for comparison purposes.

3. Assess the contribution made by $PM_{2.5}$ to the overall PM_{10} mass concentration.

4. Inclusion of stability class measurements and rainfall data in particulate pollution studies.

5. Employ the SCATS system in conjunction with pollution prediction methods to abate high pollution episodes by means of traffic restrictions.

8. Appendix

Table 8.1 Normalised Wind Rose Values for each Traffic Volume Category

		Traffic Volumes					
Direction	**500**	**1000**	**1500**	**2000**	**2500**	**3000**	**Average**
0	1.41487	1.039693	0.961227	0.793711	0.728618	0.773687	0.952
45	1	1.008318	0.838491	1	1	1	0.9745
90	1.37173	0.771625	1.023629	0.642199	1.043067	0.990042	0.9737
135	0.823068	0.732143	0.505262	0.729697	0.50319	0.571128	0.6441
180	0.626702	0.583414	0.560915	0.621012	0.601487	0.597959	0.5986
225	0.64929	0.586381	0.689013	0.693042	0.621344	0.524944	0.6273
270	0.629862	0.562306	0.774694	0.628484	0.683037	0.56048	0.6398
315	0.775244	0.877238	1.007295	0.915963	0.719071	0.689378	0.8307

The values in Table 1 correspond to the wind roses in figure 5.14. The average wind rose in the final column forms the basis for correcting the measured PM_{10} values depending on the wind direction at the time of measurement. This allows all measured values to be representative of the true level of PM_{10} that would result from a particular volume of traffic and wind flow from any direction.

Table 8.2 Wind Speed verses Average PM_{10} Values for each Traffic Volume Category

Wind Speed	PM_{10} verses Wind Speed for each Traffic Volume Category					
	500	**1000**	**1500**	**2000**	**2500**	**3000**
m sec^{-1}	$\mu g\ m^{-3}$	$\mu g\ m^{-3}$	$\mu g\ m^{-3}$	$\mu g\ m^{-3}$	$\mu g\ m^{-3}$	$\mu g\ m^{-3}$
0.5	25.48241	34.11848	33.72864	53.9634	45.53334	61.16043
1	16.96487	26.47598	33.04158	38.47851	41.46733	45.31
1.5	15.67649	20.60993	25.86492	29.69331	33.39037	30.14325
2	17.1389	15.78012	20.70602	23.05825	25.0186	29.35021
2.5	10.33058	12.72761	14.73012	23.05423	32.85205	22.48722
3	7.401988	8.640795	9.543298	16.33371	17.48036	15.55441
3.5	5.722795	7.777442	6.550532	11.58146	18.74501	16.70758
4	5.679728	7.579732	6.006308	9.339437	15.57657	11.01486
4.5	5.230375	3.498437	4.867777	7.495256	13.12592	8.584896
5	6.530622		6.202435	6.848965	8.551365	7.470319
5.5	3.718223		2.411952	7.236772	7.602282	6.097636
6	1.59242		2.612658	5.888922	8.108433	
6.5				6.917274	3.600178	2.983787
7				6.442314		

The values for PM_{10} in Table 2 were then used to create the raw data curves in figure 5.16. These PM_{10} values have been adjusted for wind direction as per table 1 above.

Table 8.3 Best-Fit Polynomial Equations for PM_{10}

Traffic Volume	**Derived Polynomial Equations**	**Corresponding R^2 Values**
500	y = 0.1809x2 - 4.1745x + 27.456	$R^2 = 0.9261$
1000	y = 0.4273x2 - 7.8062x + 40.744	$R^2 = 0.9893$
1500	y = 0.3291x2 - 7.2993x + 43.474	$R^2 = 0.9759$
2000	y = 0.4326x2 - 9.566x + 57.973	$R^2 = 0.9737$
2500	y = 0.194x2 - 6.0808x + 51.191	$R^2 = 0.952$
3000	y = 0.4808x2 - 10.824x + 65.62	$R^2 = 0.9627$

The polynomial equations in Table 3 represent the PM_{10} level, y, associated with the change in wind speed x. These polynomials are used to predict the PM_{10} levels as part of the final model created.

References

Allen, G., Sioutas, C., Koutrakis, P., Reiss, R., Lurmann, F., Roberts, P.T. (1997) Evaluation of the TEOM method for measurement of ambient particulate mass in urban areas. Journal of Air and Waste Management Association Vol. 47, pp. 682-689

APEG (1998) Source Apportionment of Airborne Particulate Matter in the Umited Kingdom. Airborne Particles Expert Group (APEG).

Baedecker, P.A., Edney, E.O., Moran, P.J., Simpson, T.C., Williams, R.S. (1991) Effects of acidic deposition on materials. In: Irving, P.M., ed. Acidic deposition: state of science and technology, volume III: terrestrial, materials, health and visibility effects. Washington, DC: The U.S. National Acid Precipitation Assessment Program (State of Science and Technology Report no.19).

Benarie, M.M., (1980) Air Pollution Problems-1, Urban Air Pollution Modelling. The Macmillan Press LTD

Berkowicz, R., Palmgren, F., Hertel, O., Viagnati, E. (1996) Using measurements of air pollution in streets for evaluation of urban air quality – meteorological analysis and model calculations. The Science of the Total Environment Vol. 189/190, pp. 259-265

Boubel, R.W. (1994) Fundamentals of air pollution, 3rd Ed., Academic Press, London

Bridgeman, H., (1990) Air Pollution: problems for the 1990's. Belhaven Press, London

Butler, J.D. (1979) Air pollution chemistry. Academic Press, London

Chock, P.P., Terrell, T.R., Levitt, S.B. (1975) Time series analysis of riverside, California, air quality data. Atmospheric Environment Vol. 9, pp. 978-989

Chow, J.C., Watson, J.G., Zhigiang, L., Lowenthal., D.H., Frazier, A., Solomon, P.A., Thuillier., R.H. and Magliano, K., (1996a). Descriptive analysis of $PM_{2.5}$ and PM_{10} at regionally representative locations during STVAQS/AUSPEX. Atmospheric Environment Vol. 30, pp 2079-2112

Chow, J.C., Watson, J.G., Zhigiang, L., Lowenthal, D.H., and Countess, R.J., (1996b) Sources and chemistry of PM_{10} aerosol in Santa Barbara County, California Atmospheric Environment Vol. 30, pp 1489-1499

Cooper, C.D., (1989) Persistence factors for mobile source (roadway) carbon monoxide modelling. Journal of Air Pollution Control Association Vol. 39, pp. 714-720

Cooper, C.D., Malone, L.C., Lie, P.S., (1992) Identifying "Worst Case" persistence factors for carbon monoxide modelling near intersections in Orlando, Florida. Journal of Air Pollution Control Association Vol. 42, No.11, pp 1461-1465

Cosgrove, B. (1997) The World of Weather Swan Hill Press

Covert, D.S., Kapustin, V.N., Quinn, P.K and Bates, T.S., (1992) New particle formation in the marine boundary layer. Journal of Geophysics Res.{Atmos.} Vol. 97, pp. 20,581-20,589.

Croxford, B., Penn, A., Hillier, B., (1996) Spatial distribution of urban pollution: civilizing urban living. The Science of the Total Environment Vol. 189/190, pp. 3-9

Croxford, B. and Penn, A. (1998) Siting considerations for urban pollution monitors. Atmospheric Environment Vol. 32, No. 6, pp. 1049-1057

Demirci, E., and Cuhadaroglu, B. (2000) Statistical analysis of wind circulation and air pollution in urban Trabzon. Energy and Buildings, Vol. 31, No. 1, pp. 49-53

Department of The Environment (1999) Irish bulletin of vehicle and driver statistics 1999. Department of The Environment and Local Government, Custom House, Dublin

Dublin Corporation SCATS Dublin area traffic control system. Office of The Director of Traffic, Dublin Corporation, Civic Offices, Fishamble Street, Dublin

Dublin Corporation (1999) Air quality monitoring, annual report, 1999. The Air Quality and Noise Control Unit, Dublin Corporation, Civic Offices, Fishamble Street, Dublin

El-Fadel, M. and Massoud, M. (2000) Particulate matter in urban areas: health-based economic assessment. The Science of The Total Environment Vol. 257, No. 2-3, pp. 133-146

Environmental Protection Agency (EPA) (1996) A baseline study of the concentrations of volatile organic compounds and PM_{10}. Environmental Protection Agency, Headquarters, Ardcavan, Wexford

Environmental Protection Agency (EPA) (1997) Air quality monitoring annual report 1997 Environmental Agency, Headquarters, Ardcavan, Wexford

Environmental Protection Agency (EPA) (1998) Air quality monitoring, annual report, 1998. Environmental Agency, Headquarters, Ardcavan, Wexford

Fenger, J., (1999) Urban air quality. Atmospheric Environment, Vol. 33, No. 29, pp. 4877-4900

Georgopoulos, P.G. and Seinfeld, J.H. (1982) Statistical distributions of air pollutant concentrations. Environmental Science and Technology, Vol.16, No. 7, pp. 401A-416A

Hao, J., He., D., Wu, L. and He, K. (2000) A study of the emission and concentration distribution of vehicular pollutants in the urban area of Beijing. Atmospheric Environment Vol. 34, No. 3, pp. 453-465

Harrison, R.M., Deacon, A.R., Jones, M.R. and Appleby, R.S. (1997) Sources and processes affecting concentrations of PM_{10} and $PM_{2.5}$ particulate matter in Birmingham (U.K.). Atmospheric Environment Vol. 31 No. 24, pp. 4103-4117

Hogg, R.V., and Ledolter, J. (1987) Engineering Statistics. Macmillan Publishing Company, New York.

Hodge, V.F. and Stallard, M.O. (1986) Environmental Science and Technology. Vol, 20, pp. 1058-1060 EMBASE

IT, Sligo (1999) Air pollution control course notes from Postgraduate Diploma in Environmental Protection. Department of Environmental Science, Institute of Technology, Sligo

King, A.M., Pless-Mulloli, T., Merefield, J., Stone, I., (2000) New directions: TEOMs and the volatility of UK non-urban PM_{10} a regularity dilemma. Atmospheric Environment, Vol. 34, No. 19, pp. 3211-3212

Kleeman, M.J., Cass, G.R. (1998) Source contributions to the size and composition distribution of urban particulate air pollution. Atmospheric Environment, Vol. 32, No.16, pp. 2803-2816

Lam, G.C.K., Leung, D.Y.C., Niewiadomski, M., Pang, S.W., Lee, A.W.F., Louie, P.K.K. (1999) Street-level concentrations of nitrogen dioxide and suspended particulate matter in Hong Kong. Atmospheric Environment Vol.33, pp. 1-11

Marnane, I., (2000) Online Hydrocarbon Monitoring Ph.D. thesis, University of Dublin, Trinity College, Dublin

Mayer, H. (1999) Air pollution in cities. Atmospheric Environment Vol.33, pp. 4029-4037

Monn, C.H., Carabais, V., Junker, M., Waeber, R., Karrer, M., Wanner, H.U. (1997) Small-scale spatial variability of particulate matter <10μm (PM_{10}) and nitrogen dioxide. Atmospheric Environment Vol.31, No.15, pp. 2243-2247

Montgomery, D.C., and Peck, E.A., (1992) Introduction to Linear Regression Analysis. John Wiley & Sons Inc.

Morawska, L., Thomas, S., Jamriska, M. and Johnson, G., (1999) The modality of particle size distributions of environmental aerosols. Atmospheric Environment Vol. 33, No. 27, pp. 4401-4411

Morel, B., Yeh, S., Cifuentes, L., (1999) Statistical distributions for air pollution applied to the particulate problem in Santiago. Atmospheric Environment Vol. 33, pp. 2575-2585

Novakov, T. (1982) Soot in the atmosphere. In: Wolff, G.T., Klimisch, R.L., eds. Particulate carbon: atmospheric life cycle. New York, NY: Plenum Press: pp. 19-41

Palacios, M.A., Gómez, M.M., Moldovan, M., Morrison, G., Rauch, C., McLeod, C., Ma, R., Laserna, J., Lucena, P., Caroli, S., Alimonti, A., Petrucci, F., Bocca, B., Schramel, P., Lustig, S., Zischka, M., Wass, U., Stenbom, B., Luna, M., Saenz, J.C., Santamaría, J. and Torrens, J.M. (2000) Platinum-group elements: quantification in collected exhaust fumes and studies of catalyst surfaces. The Science of the Total Environment, Vol. 257, No. 1, pp 1-15

Pérez, P., Trier, A., Reyes,J. (2000) Prediction of $PM_{2.5}$ concentrations several hours in advance using neural networks in Santiago, Chile. Atmospheric Environment Vol. 34, No. 8, pp. 1189-1196

QUARG (1993) Urban air quality in the United Kingdom. Quality of Urban Air Review Group, United Kingdom.

QUARG (1996) Airborne particulate matter in the United Kingdom. Quality of Urban Air Review Group. Available from Institute of Public and Environmental Health, The University of Birmingham.

Reponen, A., Ruuskanen, J., Mirme, A. Pärjälä, E., Hoek, G., Roemer, W., Hosiokangas, J., Pekkanen., J., and Jantunen, M., (1996) Comparison of five methods for measuring particulate concentrations in cold winter climate. Atmospheric Environment Vol. 30. No. 22, pp. 3873-3879

Reynolds. A.W. (2000) Development and application of an integrated air quality modelling system for traffic related pollution in urban areas. Ph.D. thesis, University of Dublin, Trinity College, Dublin

Rupprecht and Patashnick Co., Inc. (1993) Operating manual TEOM series 1400a ambient particulate (PM_{10}) monitor. Rupprecht and Patashnick Co., Inc., 25 Corporate Circle, Albany, NY 12203, USA

Rupprecht and Patashnick Co., Inc. (1998) Operating manual Partisol-plus model 2025 sequential air sampler. Rupprecht and Patashnick Co., Inc., 25 Corporate Circle, Albany, NY 12203, USA

Salter, L.F. and Parsons, B. (1999) Field trials of the TEOM and Partisol for PM_{10} monitoring in the St Austell china clay area, Cornwall, UK. Atmospheric Environment Vol. 33, pp. 2111-2114

Scaperdas, A., Colvile, R.N., (1999) Assessing the representativeness of monitoring data from an urban intersection site in central London, UK. Atmospheric Environment, Vol. 33, pp. 661-674

Seinfeld, J.H. (1986) Atmospheric Chemistry and Physics of Air Pollution. John Wiley & Sons.

Shi, P.J. and Harrison, R.M. (1997) Regression modelling of hourly Nox and NO_2 concentrations in urban air in London. Atmospheric Environment Vol. 31, No. 24, pp. 4081-4094

Simpson, R.W. (1990) A model to control emissions which avoid violations of PM_{10} health standards for both short and long term exposures. Atmospheric Environment Vol. 24, pp. 917-924

Sini, J.F., Anquetin, S., Mestayer, P.G. (1996) Pollutant dispersion and thermal effects in urban street canyons. Atmospheric Environment Vol. 30, No.15, pp. 2659-2677

Spengler, J., Schwab, M., Ryan, P.B., Colome, S., Wilson, A.L, Bilick, I. And Becker, E. (1994) Personal exposure to nitrogen dioxide in the Los Angeles Basin. Journal of The Air and Waste Management Association Vol. 44, pp. 39-47

Stedman, J.R., (1997) The secondary particle contribution to elevated PM_{10} concentrations in the UK. AEA Technology, National Environmental Technology Centre. Report 20008001/007. AEAT – 2959.

Stedman, J.R., and Dore, C.J. (1998a) A Report for The UK Department of The Environment, Transport and The Regions, Benzene and 1,3-butadiene at the roadside. John Stedman AEA Technology Plc, National Environmental Technology Centre, E5 Culham, Abingdon, OX14 3DB.

Stedman, J.R., and Dore, C.J. (1998b) A Report for The UK Department of The Environment, Transport and The Regions, Predicting PM_{10} Concentrations in The UK. John Stedman AEA Technology Plc, National Environmental Technology Centre, E5 Culham, Abingdon, OX14 3DB.

Stenbom, B. (1994) Microstructural changes in deactivated noble metal catalysts. Ph.D. thesis, Chalmers University of Technology, Sweden.

Theurer, W. (1999) Typical building arrangements for urban air pollution modelling. Atmospheric Environment, Vol.33, pp. 4.57-4066

UK Department of the Environment, Transport and the Regions (UK DETR) (1998a) London Kerbside Air Quality Monitoring Site, Marylebone Road, Annual Report 1998. Commissioned by the London Local Authorities and the Department of the Environment, Transport and the Regions, London, UK.

UK DETR (1998b) Selection and use of dispersion models. Report No.LAQM.TG3(98), Department of the Environment, Transport and the Regions, London, UK.

UK DETR (2000a) Review and Assessment: Selection and use of dispersion models. Department of the Environment, Transport and the Regions, London, UK.

UK DETR (2000b) What is the appropriate measurement on which to base a standard? Expert Panel on Air Quality Standards (Air borne particles) Issued jointly by the Department of the Environment, Transport and the Regions, London, UK and The Scottish Executive.

United States Environmental Protection Agency (US-EPA) (1992) Guideline for modelling carbon monoxide from roadway intersections. US EPA, Office of Air Quality Planning and Standards, Research Triangle Park, NC 27711, USA

United States Environmental Protection Agency (US-EPA) (1995) PART5 User's Guide. US EPA, Assessment and Modelling Division, 2000 Traverwood Drive, Ann Arbor, MI 48105, USA

United States Environmental Protection Agency (US-EPA) (1996) Air quality criteria for particulate matter, Report No. EPA/600/P-95/001aF, Office of Research and Development, Washington DC 20460, USA

Walsh, M.P., (1987) Motor vehicle pollution in Europe- a problem still not solved. In: Proceedings of the Institute of Mechanical Engineers, Vehicle Emissions and their Impact on European Air Quality

Wedding. J.B., McFarland, A.R. and Cermak, J.E. (1977) Large particle collection characteristics of ambient aerosol samples. Environmetal Science and Technology, Vol. 11, pp. 387-390

Weingartner, E., Keller, C. Stahel, W.A., Burtscher, H. and Baltensperger, U. (1997) Aerosol emission in a road tunnel. Atmospheric Environment, Vol. 31, No. 3, pp. 451-462

Whitby, K.T. (1978) The physical characteristics of sulphur aerosols. Atmospheric Environment, Vol. 12, pp. 135-159

Whitby, K.T. and Severdrup, G,M) (1980) California aerosols: their physical and chemical characteristics. In: Hidy, G.M., Mueller, P.K., Grosjean, D., Appel, B.R., Wesolowski, J.J., eds. The character and origins of smog aerosols: a digest of results from the California Aerosol Characterization Experiment (ACHEX). New York, NY: John Wiley & Sons, Inc., pp 477-517. (Advances in Environmental Science and Technology, Vol. 9)

Willeke, K. and Whitby, K.T. (1975) Atmospheric aerosols: size distribution interpretation. Journal of Air Pollution Control Association, Vol. 25, pp. 529-534

Wróbel, A., Rokita, E., and Maenhaut, W. (2000) Transport of traffic-related aerosols in urban areas. The Science of The Total Environment Vol. 257, No. 2-3, pp. 199-211

Zannetti, P., (1990) Air Pollution Modelling. Theories, Computational Methods and Available Software. Computational Mechanics Publications, Southhampton, UK.