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VEHICLE EMISSIONS AND ROADSIDE AIR QUALITY

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ACRONYMS AND ABBREVIATIONS

A:F	Air Fuel ratio
ARN	Automated Rural Network
AUN	Automated Urban Network
CARB	California Air Resources Board
CAWRSS	Clark And Washoe Remote Sensing Study
CA	California
co	Colorado
CO	Carbon monoxide
CO_2	Carbon dioxide
COHb	Carboxyhaemoglobin
DC	Direct current
DoE	Department of Environment
DoT	Department of Transport
DVLA	Drivers Vehicle Licensing Authority
EC	European Community
EPAQS	Expert Panel on Air Quality Standards
EU	European Union
EUN	Enhanced Urban Network
FEAT	Fuel Efficiency Automobile Test
FTP	Federal Test Procedure
GM	General Motors
GMRL	General Motors Research Laboratories
GMOB	General Motors On-Board measurement
g/gal	grams of pollutants emitted per gallon of fuel used
g/mile	grams of pollutants emitted per mile travelled
gross polluting vehicle	any vehicle that falls within the top 20% of emitters when individual model years of the vehicle fleet are rank ordered
high emitting vehicle	any vehicle that falls within the top 10% of emitters when the vehicle fleet is rank ordered
HNO_2	Nitrous acid
HNO_3	Nitric acid
HC	Hydrocarbons
Hz	Hertz
IAPSC	Investigation of Air Pollution Standing Conference
IL	Illinois
I/O	input/output
ISATA	International Symposium on Advanced Transportation Applications
IR	Infrared
kerbside	monitoring site 3 metres from the road/pavement interface
kmph	Kilometres per hour
LCD	Liquid Crystal Display
MOT	Ministry of Transport

MPG	Miles per gallon
MT	Mega Tonnes
NDIR	Non-dispersive infrared
NO	Nitric oxide
NO_2	Nitrogen dioxide
NO_3	Nitrogen trioxide
NO_x	Nitrogen oxides
N_2O	Dinitrogen oxide
N_2O_3	Dinitrogen trioxide
N_2O_4	Dinitrogen tetroxide
N_2O_5	Dinitrogen pentoxide
NSCA	National Society for Clean Air and environmental protection
O_3	Ozone
ppm	parts per million
PM_{10}	Particulate Matter less than 10 microns in diameter
ppb	parts per billion
Q	$CO:CO_2$ ratio as measured by FEAT
Q^1	$HC:CO_2$ ratio as measured by FEAT
QUARG	Quality of Urban Air Review Group
r	Coefficient of correlation
r^2	Coefficient of determination
RAF	Royal Air Force
RAM	Random Access Memory
roadside	monitoring site on the kerb
ROM	Read Only Memory
RSDs	Remote Sensing Devices
SCAQMD	South Coast Air Quality Management District
SO_2	Sulphur dioxide
SUN	Statutory Urban Network
TWC	Three Way Catalyst
TRL	Transport Research Laboratory
UPRC	Urban Pollution Research Centre
USA	United States of America
USEPA	United States Environmental Protection Agency
UK	United Kingdom
vehicle fleet	those vehicles which pass the remote sensor during sampling
VOCs	Volatile Organic Compounds
WSL	Warren Spring Laboratory
WHO	World Health Organisation

Site abbreviations

A	Bounds Green Road, Haringey
B	Dixons Bank, Middlesborough
BEX	Bexley, London

C	Abbey Street, Southwark
CLL2	London Bloomsbury
CRD	Cromwell Road, central London
D	Uppingham Road, Leicester
LEIC	Welford Place, Leicester
MID	Longlands college, Middlesborough
WL	West London, Earls Court

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ABSTRACT

Individual carbon monoxide and hydrocarbon emissions were monitored from passing vehicles using the Fuel Efficiency Automobile Test at four survey sites (Bounds Green Road, Haringey (site A); Dixons Bank, Middlesborough (site B); Abbey Street, Southwark (site C); Uppingham Road, Leicester (site D)). The remotely measured emissions data is described in terms of fleet emissions, model year emissions and model year contribution to fleet emissions. It was found that there were a large majority of low emitting vehicles contributing little to fleet emissions and a small minority of high emitting vehicles contributing significant proportions to fleet emissions. Model year analysis suggested a low association between vehicle age and mean emissions prior to 1983 but a much improved relationship after 1983. Analysis of model year contributions to fleet emissions shows new gross polluters to be the largest contributors and older vehicles playing only a minor role.

The concentrations of carbon monoxide and nitrogen oxides in air were monitored, in conjunction with the FEAT measurement, at various distances from the road (roadside (on the kerb), kerbside (3 metres from the road), 7.5 metres and 15 metres from the road). A decrease of carbon monoxide and nitric oxide concentrations with distance from the road was noted for all sites with the exception of site D where meteorological parameters exerted a greater influence upon air quality than did distance from the road. The expected increase of NO_2 concentration with distance from the road, as NO is oxidised to NO_2, did not occur. Moreover, NO_2 concentrations decreased with distance from the road. However, the production of NO_2 by oxidation of NO can be inferred in two ways. Firstly, a much more gradual decline in concentrations with distance from the road was noted for NO_2 compared to CO and NO, possibly due to NO_2 production counteracting the reduction in concentration caused by dispersion. Secondly, an analysis of the change of ratios between nitrogen dioxide and nitric oxide with distance from the road reveals a relative increase of NO_2 with distance.

The air quality data were compared with the remotely measured vehicle emissions data, wind speed and wind direction. A statistical examination of the data was undertaken on a half-hourly and five minute basis (no wind data was available on a five minute basis). The half-hourly analyses for both CO and NO_x produced positive correlations between vehicle emissions data and air quality, and predominantly negative correlations between wind speed and air quality. Both positive and negative correlations were observed between wind direction and CO/NO_x air quality . Regression analyses were undertaken where the results were statistically significant at a 0.1 level. This reduced the sample size for CO to data collected on eight individual sampling days and to only two days for NO_x. All the analysed CO sampling days recorded r^2 values of greater than 0.5, such that for each sampling day at least half the variation in CO air quality is explained by the variation in on-road vehicle emissions, wind speed and wind direction. The two analysed NO_x sampling days recorded r^2 values of approximately 0.8. The five minute analyses produced were less statistically significant giving only a low degree of correlation between CO and NO_x air quality and on-road vehicle emissions. Regression analyses were undertaken for only two days for CO and only one day for NO_x.

CHAPTER 1. INTRODUCTION

1.1 Background

The nature of atmospheric pollution in the UK (United Kingdom) has changed dramatically over the past 40 years. The London smogs of the early 1950s led to legal and technological control of coal combustion in industrial and domestic premises, and resulted in substantial reductions in the levels of smoke and sulphur dioxide. Although these pollutants are now largely controlled, and the London smogs eliminated, other atmospheric pollutants have, and are, giving cause for concern. In the 1970s, lead from vehicle exhaust emissions was implicated in adverse human health effects, especially involving children. This problem has been significantly reduced by the considerable reduction in the lead content of petrol and the introduction of lead-free petrol.

However, road traffic is the main source of carbon monoxide, nitrogen oxides, hydrocarbons, particulate matter and still remains the main source of lead. The steady annual increase in the number of on-road vehicles (up by 31% over the period 1980-1993)*(DoT 1995)* has resulted in a parallel increase in the above pollutants, with the exception of lead. Carbon monoxide, hydrocarbons and nitrogen oxides are of particular interest to the work described in this thesis. The consequences of increased air pollution concentrations were demonstrated dramatically by the severe pollution episode in London in December 1991, when poor air quality, caused to a large degree by high nitrogen dioxide levels, was associated with numerous respiratory illnesses.

Traffic related air pollution problems tend to be concentrated in urban areas, and the extent of these problems led the UK Department of the Environment (DoE) to establish a Quality of Urban Air Review Group (QUARG), which first met in January 1992. Further, to this the *Environment Act 1995* included a requirement for the development of a strategy to improve areas of poor air quality. A consultation draft strategy has been recently issued *(August 1996)* by the DoE for this purpose with the intent of reducing any significant risks to health and achieving sustainable development in relation to air quality.

In 1987, the University of Denver developed the Fuel Efficiency Automobile Test (FEAT) to remotely measure on-road vehicle exhaust emissions. Most studies using the FEAT system have been undertaken in the United States of America with little work undertaken in the UK or Europe. Moreover, no work has been undertaken on either side of the Atlantic to relate remotely measured on-road vehicle emissions to roadside air quality. This research project was established to provide information in these areas.

This research project could not have been completed without the loan of the FEAT system from the Transport Research Laboratory (TRL) and their generosity with regard to this is gratefully acknowledged. The Urban Pollution Research Centre (UPRC) has had a number of research links with TRL over the past 10 years through collaborative projects involving CASE studentships and these are continuing. This project represents an extension of this cooperation although on a more informal basis.

1.2 Aims and objectives

The primary aims and objectives of the research are:

- to remotely sense carbon monoxide and hydrocarbon on-road emissions from individual vehicles. This study utilises the Fuel Efficiency Automobile Test (FEAT), whereby the concentrations of carbon monoxide, carbon dioxide and hydrocarbons in individual vehicle exhaust emissions are measured by the absorption produced in an infrared beam directed across the carriageway. The objective of this section of the research is to provide detailed information on the on-road emission characteristics of individual fleets in the UK.
- to monitor carbon monoxide and nitrogen oxides levels at roadside, thereby providing information as to their concentrations and how these concentrations change with distance from the road. This involves the installation and calibration of air quality monitors and the subsequent monitoring of air quality over a period when remote sensing data are being collected.
- to investigate the relationships between remotely sensed on-road vehicle emissions and roadside air quality. The aim of this section of the research is to examine the trends in air quality data and to attempt to explain these in terms

of on-road emissions and meteorological characteristics using regression analysis. From this analysis an attempt will be made to develop a model that explains air quality in terms on-road vehicle emissions, wind speed and wind direction.

1.3 Outline to thesis

The thesis consists of seven chapters. **Chapter 1** contains background information relating to the research project and lists the main aims and objectives of the research. A summary of the thesis structure together with publications and conferences attended is also included.

Chapter 2 discusses the scientific principles, applications and typical results for the Fuel Efficiency Automobile Test (FEAT) in three major sections. The first section gives an overview of the FEAT system, the principles behind its operation and the major components of the system. The second section discusses the system's ability to accurately measure exhaust emissions from passing vehicles and compares measurements made by different sensors for the same exhaust emissions. Comparisons are also made of remote sensing measurements to idle emission measurements and constant volume sampling techniques. The third section presents an analysis and comparison of fleet profiles as measured by FEAT from different regions around the world.

Chapter 3 has two main sections. The first section gives an overview of the air quality measurement of carbon monoxide (CO), using a non-dispersive infrared gas analyser, and nitrogen oxides (NO_x), using a chemiluminescent gas analyser, together with data acquisition. The second section is a review of urban air quality beginning with a general introduction to vehicle pollutants which is followed by more detailed information regarding the sources, typical concentrations, air quality legislation and health effects for CO and NO_x.

Chapter 4 primarily discusses the results from the FEAT system. The chapter begins with a consideration and description of the monitoring sites with regard to FEAT measurement and is followed by a discussion of the pilot survey and its implications for the subsequent monitoring programme. The third and final section presents the carbon monoxide and hydrocarbon results gained using the remote sensor. The discussion of the results focusses on the fleet emissions, the effect that the model year of a vehicle has on emissions and the contribution that model year has to fleet emissions.

The first section in **Chapter 5** provides a description of the monitoring sites with regard to air quality measurement. This is followed by a presentation of the carbon monoxide and nitrogen oxides air quality data, providing concentrations at roadside (on the kerb), kerbside (3 metres from the road), 7.5 metres from the road and 15 metres from the road. Comparison of the data from the monitoring sites is then made with air quality standards and data from the nearest Automated Urban Network (AUN) site.

Chapter 6 investigates the relationship between remotely sensed on-road vehicle emissions and roadside air quality. In order to accomplish this aim the roadside air quality data (presented in **Chapter 5**) have been compared with the remotely measured vehicle emissions data (presented in **Chapter 4**). Meteorological data, in particular wind speed and wind direction, from the nearest weather station to the sampling site has been included in the development of any established relationship.

Chapter 7 is the final chapter, summarising the main findings of the research project and recommending possible avenues for further research.

1.4 Publications

During the course of the research a number of refereed and non-refereed publications were produced. These publications are listed below.

1.4.1 Refereed publications

Muncaster G M*, Hamilton R S, and Revitt D M (1996). Remote sensing of carbon monoxide vehicle emissions. *The Science of the Tot. Env. Vol. 189/190. 149-154.*

Muncaster G M[*], Hamilton R S, Revitt D M, Stedman D H and Vanke J (1994). Individual emissions from on-road vehicles. *International Symposium on Advanced Transportation Applications (ISATA): The motor vehicle and the environment - demands of the nineties and beyond.* 29. 267-274

[*] Principal author

1.4.2 Non-refereed publications

Muncaster G M (1995). Cleaner exhausts: A remote chance ? *Journal of Environmental Health 51-52.*

Muncaster G M (1995). Remote sensing of vehicle emissions, Middlesborough. Report for Middlesborough County Council.

1.5 Conferences attended

- **Vehicle Emissions Meeting**
  Swansea, 16 April 1996.
  The Aerosol Society

- **Investigation of Air Pollution Standing Conference (IAPSC)**
  London, 5 December, 1995.
  IAPSC

- **The Fifth International Symposium on Highway and Urban Pollution**
  Copenhagen, 22-24 May, 1995.
  Middlesex University, University of Birmingham & University of Aalborg

- **The National Society for Clean Air and Environmental Protection (NSCA): Targeting Traffic Pollution - Options for Local Air Quality Management**
  Birmingham, 8 December, 1994.
  NSCA

- **27th International Symposium on Advanced Transportation Applications: The motor vehicle and the environment - demands of the nineties and beyond.**
  Aachen, 31 October - 4 November, 1994.
  ISATA

- **The Instrumented City**
  Nottingham, 21 September, 1994.
  University of Nottingham

- **Vehicle Emissions and Air Quality**
  London, 28 March, 1994
  Middlesex University/ NSCA

- **Something in the Air - Industrial Atmospheric Monitoring**
  London, 16 February, 1993.
  The Royal Society of Chemistry

- **Investigation of Air Pollution Standing Conference (IAPSC)**
  London, 7 December, 1992.
  IAPSC

- **The National Society for Clean Air and Environmental Protection (NSCA) Annual Conference**
  Bournemouth, 19-22 October, 1992.
  NSCA

CHAPTER 2. REMOTE SENSING OF VEHICLE POLLUTION

2.1 Introduction

This chapter has three major sections. The first section gives an overview of the Fuel Efficiency Automobile Test (FEAT), the principles behind its operation and the major components of the system. The second section discusses the ability of the system to accurately measure exhaust emissions from passing vehicles and compares measurements made by different sensors for the same exhaust emissions. Moreover, comparison of FEAT to more conventional methods, such as idle emission measurement and constant volume sampling, is also made. Finally in this section there is a discussion of the inherent instability of vehicle emissions. The third section presents an analysis and comparison of fleet profiles as measured by FEAT from different regions around the world. The pattern of emissions distributions and contributions is investigated, both through a hierarchial cluster analysis and a quintile analysis.

2.2 Remote vehicle emission measurement

In 1987 the University of Denver, with the support of the Colorado Office of Energy Conservation, developed an infrared (IR) remote monitoring system for automobile carbon monoxide (CO) exhaust emissions, as an approach to promote improved vehicle fuel economy. Thus, as the result of this original purpose, the University of Denver's remote sensor was given the acronym FEAT (Fuel Efficiency Automobile Test). The measuring principle of the instrument was based upon a conventional non-dispersive infrared (NDIR) exhaust gas analyser.

Since 1987 further emission channels have been developed for hydrocarbons, nitrogen oxides and smoke. The nitrogen oxides (NO_x) and smoke channels, however, differ in that they use an ultra-violet source in place of the infrared source. However, the NO_x and smoke channels were not used in this project as they remain in a developmental stage.

The FEAT system utilises standard spectroscopic principles and combustion equations to measure emissions from passing vehicles in under one second and consists of four major components *(Figure 2.1)*:

- infrared source
- detector
- computer
- video camera.

The basic instrument, as used, measures the carbon monoxide to carbon dioxide (CO/CO_2) ratio and the hydrocarbon to carbon dioxide (HC/CO_2) ratio in vehicular exhaust emissions. These ratios provide a qualitative assessment of vehicle emissions, a high ratio being indicative of a highly polluting vehicle and conversely a low ratio representing a clean vehicle.

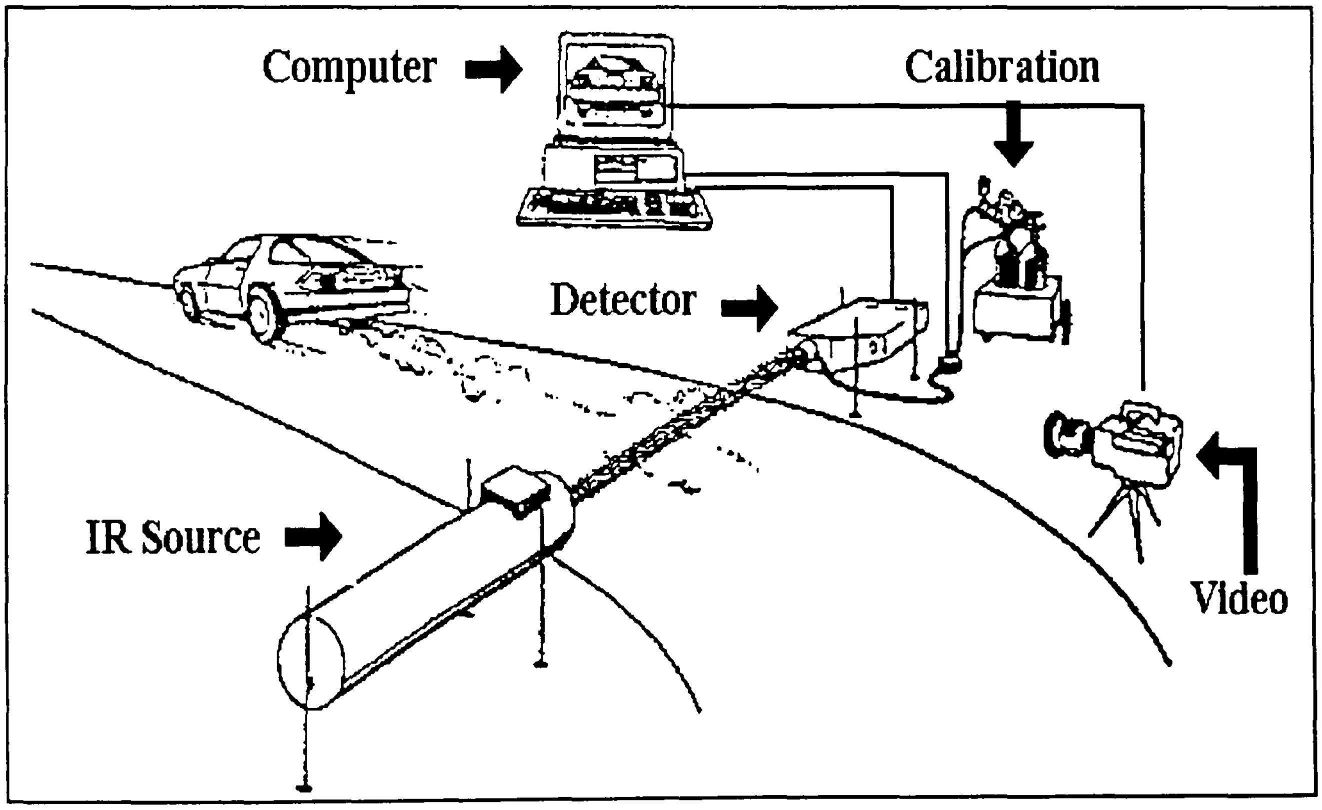


Figure 2.1 A schematic diagram of the University of Denver on-road emissions monitor *(Lyons & Stedman 1991)*.

Measurement is accomplished by directing a horizontal beam of infrared radiation across a single lane of traffic approximately 30 cm above the road surface; the height of most exhausts. This beam is focused into the detector unit and divided between four individual detectors; CO, CO_2, HC and a reference. An optical filter that transmits IR light of a wavelength uniquely absorbed by the molecule of interest is placed in front of each detector, determining its specificity. Signal reduction as a consequence of the absorption of infrared radiation produces the ratios mentioned above.

The CO/CO_2 and HC/CO_2 ratios, termed Q and Q^1 respectively, are the only valid measurement that can be made. Ratio measurements are only possible because the effective plume pathlength and amount of plume observed depend upon turbulence and wind speed *(Stedman & Bishop et al. 1994).* A Q value of zero is only possible in a vehicle with a catalyst working at maximum efficiency thereby converting all CO to CO_2. A high Q value corresponds to an engine running with a fuel rich air:fuel (A:F) ratio. A fuel lean air:fuel ratio produces limited amounts of CO, but it does impair driveability. A high Q^1 value can identify either a fuel rich or fuel lean air:fuel ratio combined with a missing or malfunctioning emission control system. A fuel rich air:fuel ratio has insufficient air to burn all the fuel and hence large amounts of HCs may be found in the exhaust. A fuel lean air:fuel ratio, if it is lean enough, can produce misfire and therefore a large quantity of hydrocarbons (unburned fuel) can be found in the exhaust manifold. If the vehicle has no catalyst or it has failed, then large quantities of hydrocarbons can be found in the exhaust emissions without large quantities of CO.

Using combustion equations and their derivatives (Appendix A), values can be derived for:

- percentage CO or HC
- the emissions of CO or HC in grams per gallon (g/gal) of fuel *(Bishop & Stedman 1990)*
- the emissions of CO or HC in grams per mile *(Cadle & Stephens 1994).*

Remote-sensing results are generally reported as percentage CO/HC in the literature and will be reported as such in chapters 4 and 6. The more traditional g/mile measurement has not been used because an estimate of the fuel economy for a vehicle at the time of a remote sensing measurement is very difficult. Thus, g/mile emission rates for individual vehicles are highly uncertain. Conversion factors for 1% CO and 0.1% HC at 20 mpg are 17.4 g/mile and 2.73 g/mile, respectively *(Cadle & Stephens 1994).*

2.2.1 Instrument overview and major components

The IR radiation source *(Figure 2.1)* is a silicon nitride gas drier igniter that when energized gives a stable temperature of 1400°C. The radiation then emitted is collimated by a gold plated mirror (f4, 6" diameter) into a parallel beam *(Stedman et al. 1991A)*.

The detector unit *(Figure 2.1)* is positioned directly across the roadway from the source so that the infrared beam is accurately focused upon the detector. The detector *(Figure 2.2)* contains three Peltier-cooled lead selenide detectors, equipped with interference filters at wavelengths of 4.6 (2174 cm^{-1}), 4.3 (2326 cm^{-1}) and 3.4 (2941 cm^{-1}) microns, for CO, CO_2 and HC detection respectively and a 3.9 (2564 cm^{-1}) micron wavelength filter that acts as a background channel. Each detector provides a current pulse at 2400 Hz, equivalent to the intensity of the infrared radiation detected at its specific wavelength. Electronic circuitry averages twenty four of these pulses and subtracts the background signal (the ambient gas concentrations). The averaged direct current (DC) level produced is then fed to three signal ports that are connected to the computer via an analogue to digital converter.

The internal optics of the detector include a germanium beam splitter, a 12 faceted polygon sapphire-windowed rotating mirror *(not shown in Figure 2.2)* and several focusing elements. The reflected light from each facet of the rotating mirror sweeps across a series of four focusing mirrors that in turn direct the light to the four detectors. Thus, each detector receives an equal amount of signal in sequential order.

There are two calibration techniques performed upon each remote sensor. The fundamental sensitivity of the instrument is calibrated in the laboratory. This is achieved by exposure of the sensor, at a path length of 6.7 metres, to known concentrations of CO, CO_2, and propane in an 8 cm IR flow cell. The calibration curves generated are used to derive equations relating the lowered voltage signals observed to the known concentrations. The CO and CO_2 curves are non-linear but because of the lower calibration concentrations of hydrocarbons employed the hydrocarbon calibration curves are closer to linearity.

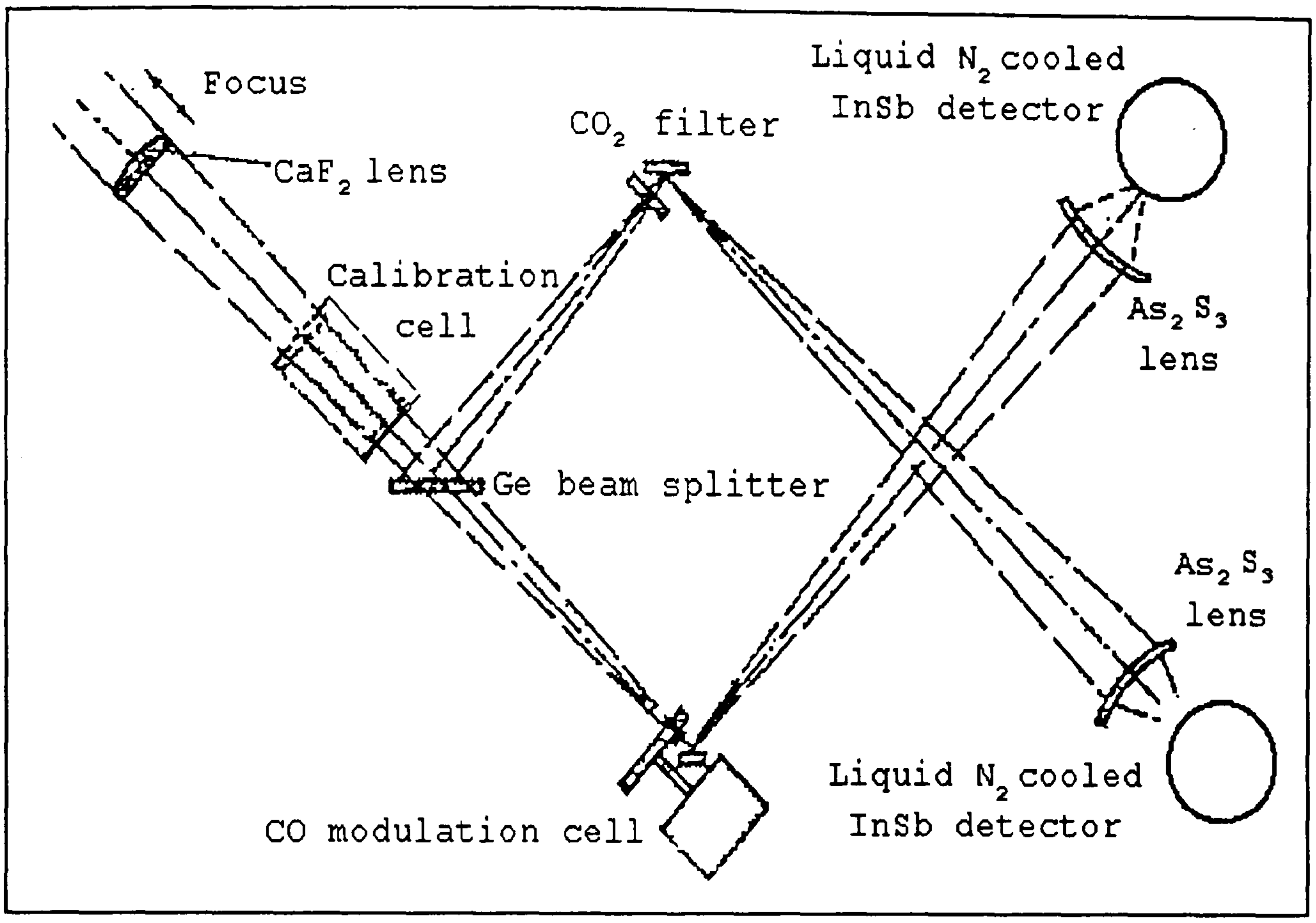


Figure 2.2 Optics and detector configuration for original carbon monoxide channel *(Bishop et al 1989).*

Prior to field measurements, the instrument undergoes a quality assurance calibration performed with the system set up at the path length to be used at the location. A 10-cm tube, in front of the focusing lens, serves as a chamber into which puffs of certified amounts of CO, CO_2 and propane gas are introduced. Due to the curvature of the response functions the field calibrations (near sea level) usually show higher ratios to CO_2 than those derived from laboratory equations at 5300 ft in Denver, USA. The data for each day are adjusted by that day's correction factor.

Although propane is the gas used to calibrate for hydrocarbons, other hydrocarbons have different sensitivities at 3.4 microns and therefore the results are given as propane equivalents. To obtain a value for hexane the propane value must be doubled because the sensor is approximately twice as sensitive to hexane as to propane. Therefore, a reading of 1200 ppm 'propane equivalents' would be read as 2400 ppm hexane.

The video camera *(Figure 2.1)* records 60 frames per second *(Bishop et al. 1992)*. It is focused on the rear of the vehicles as they pass the sensor and the image of each vehicle is recorded; written to this image are the date, time, percentage CO, HC, and

CO_2 in the emissions. The video information is recorded on tape and if required, through the Drivers Vehicle Licensing Authority (DVLA) records, the vehicle can be identified according to model, make and age. At present license plate data has to be read manually from video records. In the future this laborious process will be eliminated with the planned introduction of an automated license plate reader.

The software that operates the system was developed to ensure that erroneous measurements were not recorded. When errors are detected they lead to rejection of the measurement. A rejection sets an invalid flag in the database. There are two major rejection criteria. One rejection criterion is that there has been insufficient signal change to measure any exhaust components accurately. The second is that there has been too much scatter in the correlations of the 100-200 independent measurements of HC, CO and CO_2 conducted during any one data collection period, from which the CO/CO_2 and HC/CO_2 ratios are derived. Insufficient signal change could occur for passing pedestrians, passing motor cyclists, vehicles with an elevated exhaust or any other occasion on which the beam is blocked without the appearance of exhaust. The second criterion is based upon the expected signal to noise ratio of the system. The CO channel has a least squares slope error of less than 20% for all readings above 1%, and for those below 1%, the rejection threshold is set at 0.2% CO (absolute). Rejection criteria for the HC channel are currently set at less than 20% for HC readings greater than 0.375 %HC and 0.075 %HC for readings less than 0.375 %HC *(Bishop et al. 1989; Guenther et al. 1991).*

The FEAT system operates most effectively under dry conditions. Rain, snow and very wet road/pavement surfaces cause scattering of the IR beam. These interferences can cause the frequency of invalid readings to increase, to the point at which all data are rejected as being contaminated by too much noise.

2.3 Comparison and validation of FEAT measurements

The FEAT system has undergone rigorous testing, comparison and validation. Initial testing concentrated on the system's ability to accurately measure emissions from passing vehicles and whether those results were reproducible. Comparisons with other vehicle emission measurement techniques were then investigated. Studies both in the USA and Europe have compared FEAT to idle emission tests and to dynamometer testing, with mixed success. This mixed success is in part due to the inherent instability in some vehicles' emissions.

2.3.1 Validation of the remote sensor

In December 1989, the California Air Resources Board (CARB), the South Coast Air Quality Management District (SCAQMD) and General Motors Research Laboratories (GMRL) jointly sponsored a study to investigate the reasons for persistent high carbon monoxide concentrations near Lynwood in the Los Angeles basin. As part of this study validation of the FEAT system was undertaken *(Ashbaugh et al.1992)*.

Validation of the remote sensor involved the repeated monitoring of a vehicle under controlled conditions in a variety of operating modes. The study was undertaken in a large empty car park where it was possible to obtain a wide variety of controlled operating conditions. Verification of the remote sensing CO and HC channels was conducted through the use of a General Motors (GM) 1989 Pontiac SSE instrumented vehicle. The vehicle was equipped with two Horiba MEXA non-dispersive infrared analysers (one of which measured HC and CO, while the other measured CO and CO_2) and a laptop computer. The laptop computer enabled air:fuel ratio, vehicle speed and engine rpm to be varied via the engine computer. All measurements involving the Pontiac were made with the car cruising at 30 mph.

The CO and the HC measurements for FEAT unit 3002 (FEAT units are numbered sequentially from 3000) and the General Motors Research Laboratories (GMRL) remote sensor are plotted against the GM On-Board (GMOB) measurements in *Figure 2.3.* A wide range (0 - 10%) of carbon monoxide emissions was attained by altering the A:F ratio on the instrumented vehicle. However, this was not possible for HC emissions. HC emissions could not be increased to levels equivalent to those found from some high emitting vehicles on-road, even when misfire was induced.

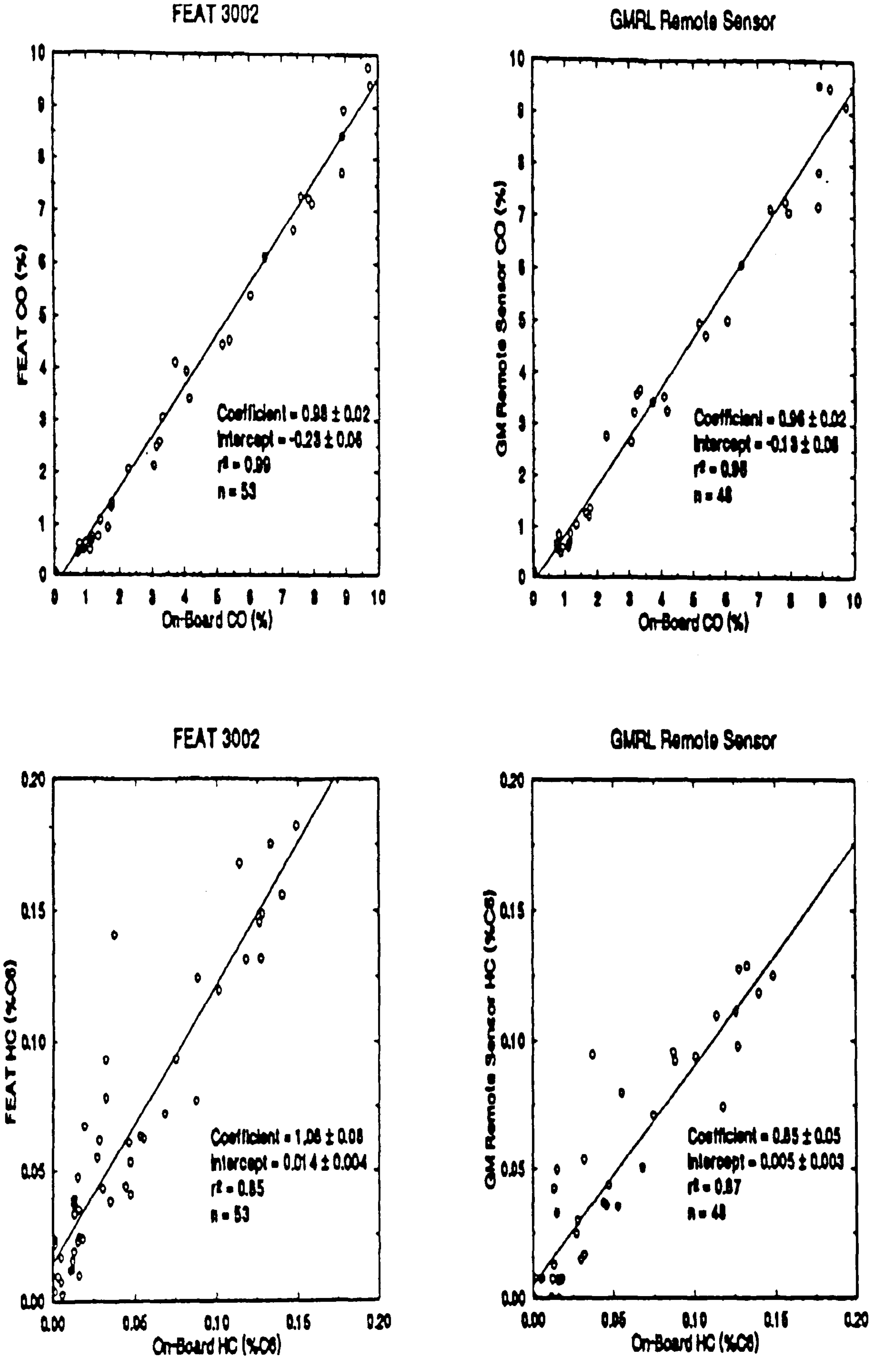


Figure 2.3 Comparison of FEAT unit 3002 & GMRL remote sensor to on-board measurements of carbon monoxide and hydrocarbons *(Ashbaugh et al. 1992)*

The FEAT units and the GM remote sensor *(Stephens and Cadle, 1990)* compared very well to the onboard measurements, with r^2 values of 0.99 and 0.98 recorded for carbon monoxide and 0.85 and 0.87 for hydrocarbons. The HC measurements, given as hexane equivalents in *Figure 2.3*, exhibited more scatter than the CO measurements, in part due to the generally low HC emissions. However, despite the scatter, the remote sensors measured HCs to within ±15% of the calibrated on-board measurements and the remotely measured CO values were within ±5% of the on-board measurement for CO.

2.3.2 Comparison of measurements made using different remote sensors

A comparison of remote sensors was conducted in an empty car park at the Santa Anita race track in Arcadia, California *(Ashbaugh et al. 1992)*. Four remote sensors were compared, FEAT units 3002, 3004 and 3005 and the fourth manufactured by General Motors Research Laboratories (GMRL). A FEAT unit was placed at each end of the test course, with another FEAT unit and the GMRL sensor side by side near the middle of the test course. These two sensors were separated by 11 feet. FEAT unit 3004 was located at the west end of the test course with FEAT unit 3005 at the east end and FEAT unit 3002 in the middle.

FEAT units 3004 and 3005 and the GMRL remote sensor are plotted against FEAT unit 3002 in *Figure 2.4*. The three FEAT units and the GMRL system compared very well to one another for CO measurement, with consistent r^2 values of 0.99. However, the HC measurement comparisons exhibited more scatter with r^2 values of 0.87, 0.85, and 0.76. FEAT unit 3005 did not measure HC as accurately as the other two FEAT systems and this is responsible for the lower r^2 value. Before the survey FEAT unit 3005 lost the mirror that focuses the IR beam on the HC detector leading to the discrepancy between systems.

A further evaluation of the reproducibility of the FEAT system was undertaken in the UK by the Transport Research Laboratory (TRL) with comparison of measurements made using FEAT units 3007 and 3002 *(Hickman & McCrae 1995)*. The two units compared quite well, giving r values of 0.96 (r^2 = 0.92) for CO and 0.88 (r^2 = 0.77)

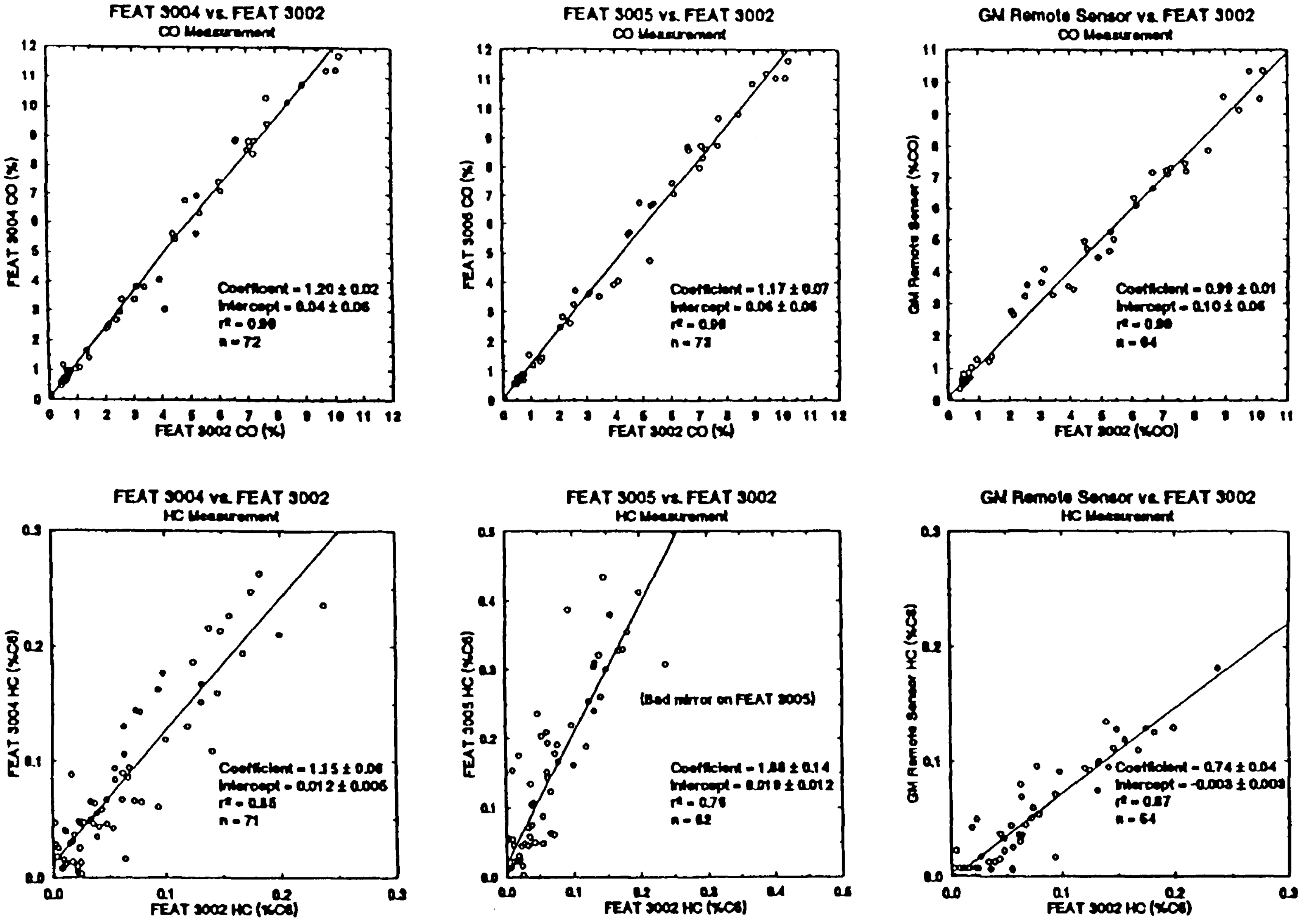
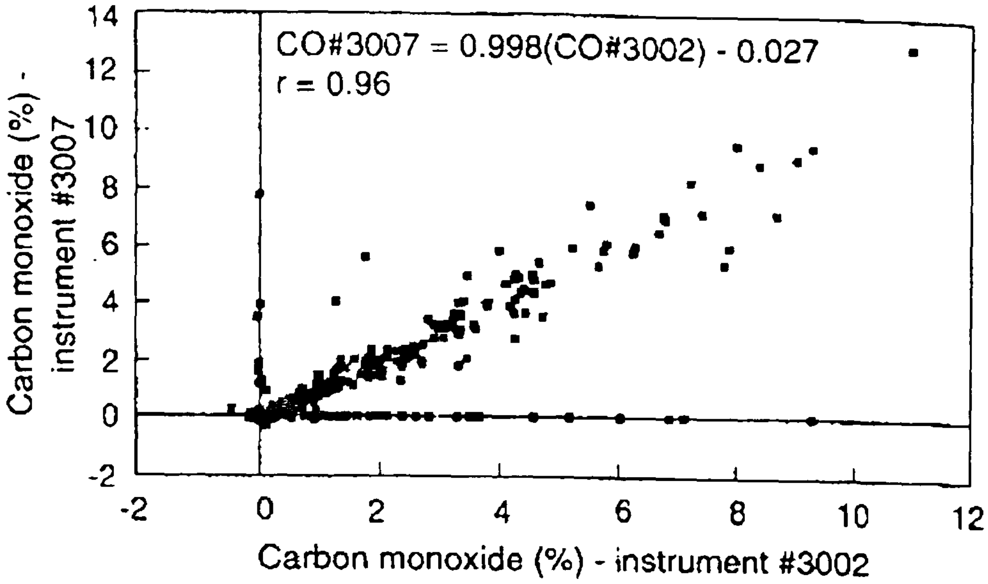


Figure 2.4 Comparison of FEAT units 3004/3005 & GMRL remote sensor to FEAT unit 3002 for CO & HC measurement *(Ashbaugh et al. 1992)*. NB. The sensors were not aligned to measure the same exhaust.



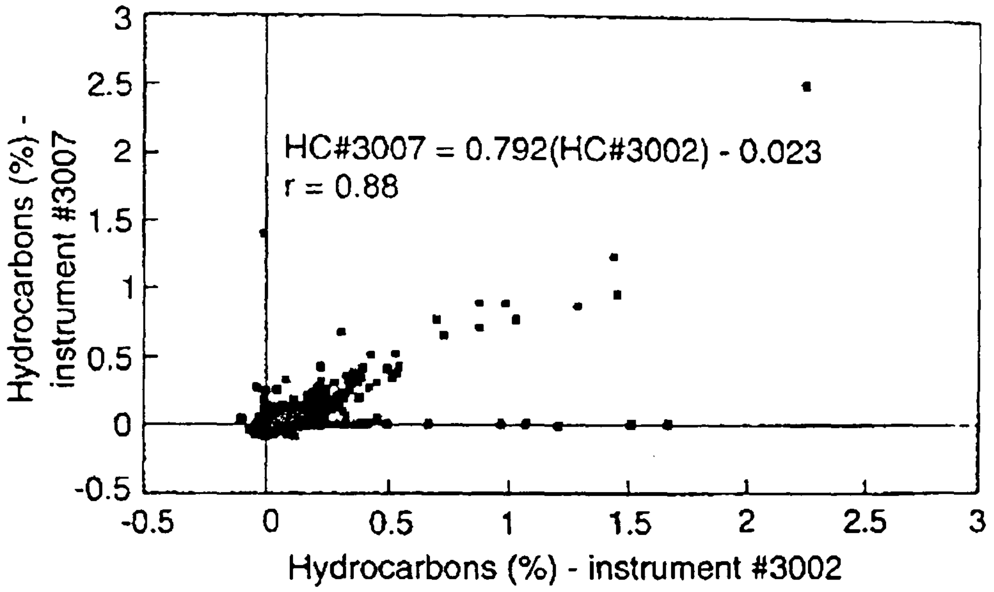


Figure 2.5 Comparison of FEAT unit 3002 to FEAT unit 3007 for CO & HC measurements *(Hickman & McCrae 1995).*

for HC *(Figure 2.5)*. There are several outlying points on both the horizontal and vertical axes. These occur when one instrument rejected the data for a particular vehicle and because of this have been excluded from the regression analysis.

It is possible that the comparison for HC was not as good as for CO because FEAT unit 3002 was tuned to 3.3 microns for the detection of HC whereas FEAT unit 3007 was tuned to 3.4 microns. The resulting different responses of the exhaust gas mixture, relative to that of the propane used for calibration, would lead to a departure from 1:1 equivalence of the results.

2.3.3 Comparison of FEAT to idle emissions measurement

A further component of the TRL study *(Hickman & McCrae 1995)* was to compare FEAT system on-road measurements to idle emission test measurements. Surveys were conducted at Liberton Gardens, Edinburgh, Victoria St and London Wall, London, whereby vehicle emissions were measured using both the FEAT system and conventional idle exhaust emissions check equipment. Vehicles that provided valid emissions data from the FEAT system were randomly selected and requested to stop for an idle emissions check; approximately 420 vehicles had an idle tailpipe emissions check. The results of the London Wall roadside survey are presented in *Figure 2.6.*

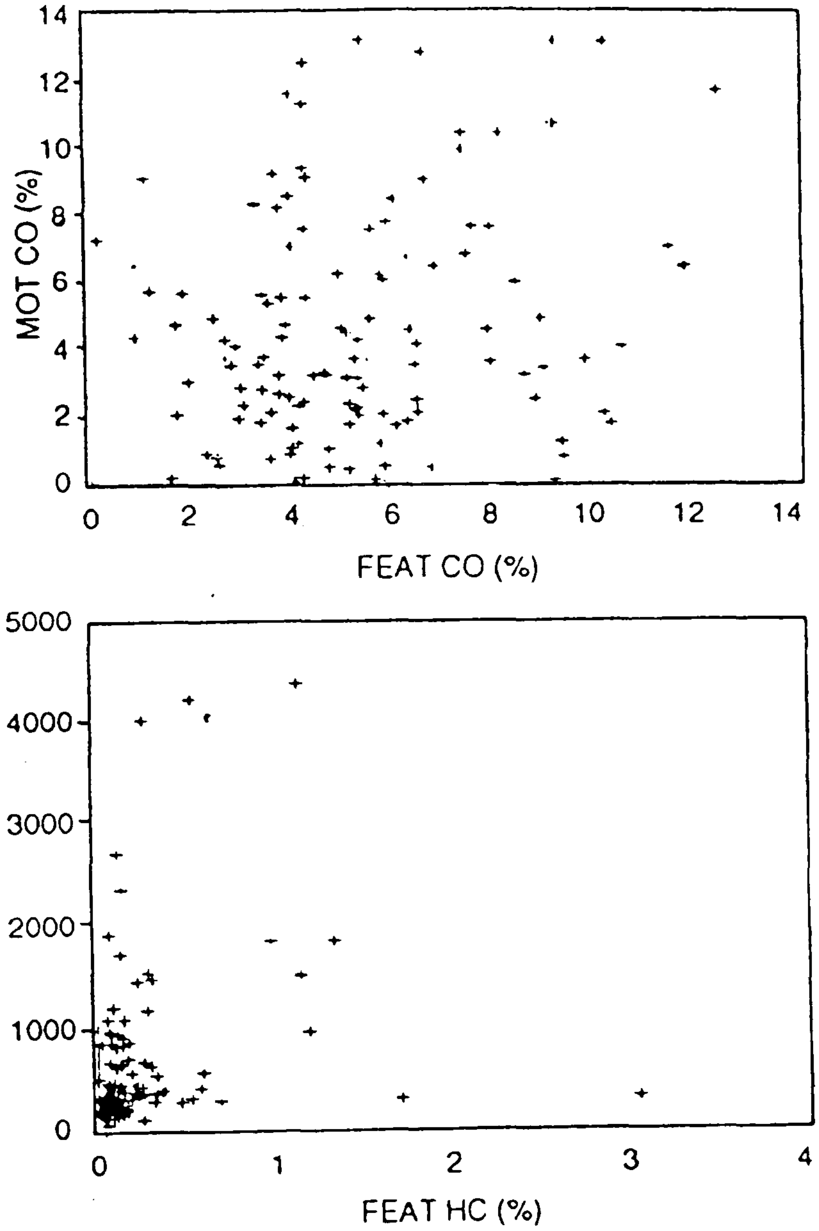


Figure 2.6 Comparison of emission rates by remote sensing (FEAT unit 3007) and at idle, London Wall, May 1993 *(Hickman & McCrae 1995).*

The graphs show CO and HC emission data, as determined by FEAT, plotted aga the same parameters determined using a standard idle test. There is no apparent vi correspondence between idle emission rates and those found by remote sensing either CO or HC. This is confirmed by regression analysis with an r^2 value of found for London Wall being typical for all sites.

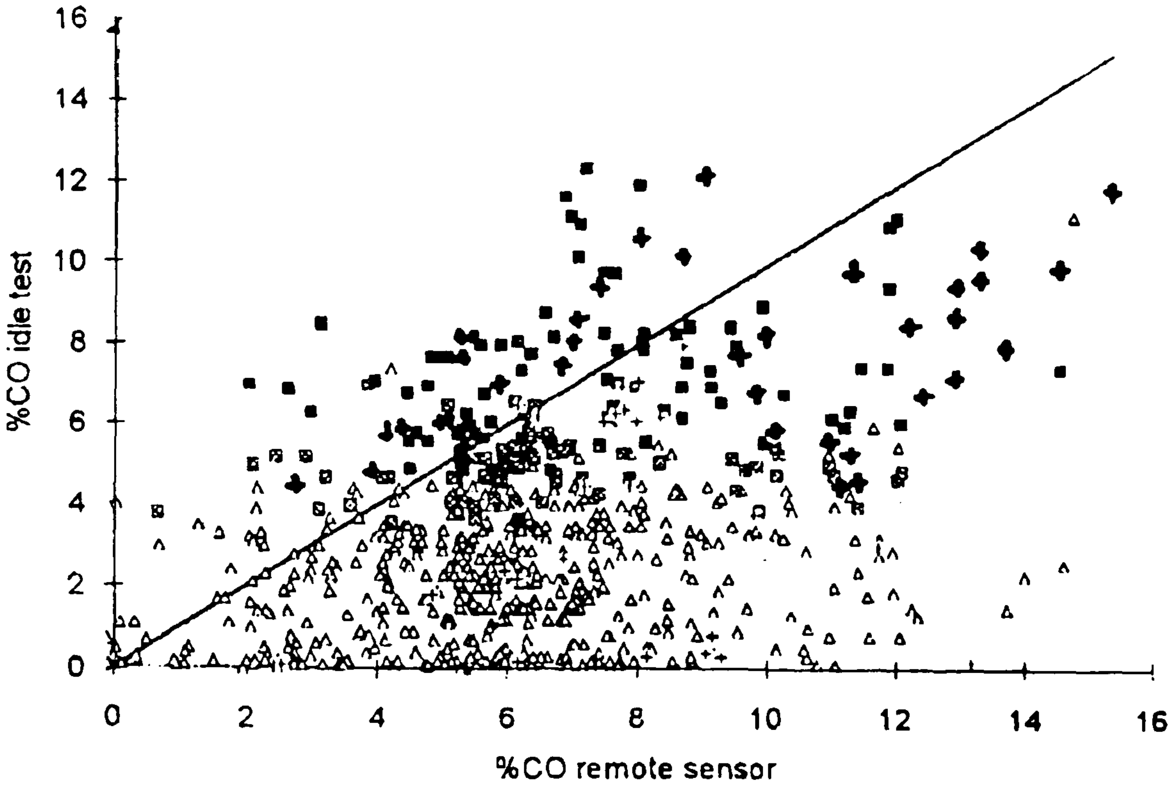


Figure 2.7 Plot of idle test %CO readings vs remote sensor %CO readings for 7 vehicles; the solid line is the line of best fit *(Sjodin 1994)*

A similar study conducted in Sweden *(Sjodin 1994)* found a stronger relationshi CO. The results of idle test %CO measurements plotted against FEAT measurements for 771 vehicles are shown in *(Figure 2.7)*. The correlation coeffi calculated for this data set was 0.36 and although this represents a stronger relatio than found for the UK data set, it is still not significant. This is not particu surprising as it is recognized that emissions vary according to a vehicles mo operation and speed, and the remote sensor is sampling emissions from higher emi modes (engine under load, accelerations and decelerations) than during the no-loa test. Moreover, the discrepancies between these two types of tests can be us question the application of an idle emissions test for enforcing the non-exceeden on-road emissions *(Vanke & Bidgood 1992)*.

Work carried out by *Walsh et al. (1996)* in the Clark and Washoe Remote Sens Study (CAWRSS) presented another picture. The study compared the Nevada s approved inspection and maintenance test (a no-load, two-speed idle test) with remote sensing devices (RSD). The remote sensors were supplied by the United St Environmental Protection Agency (USEPA) and Denver University; the authors did include the FEAT serial numbers. *Figure 2.8* shows the results of a compari between the averaged RSD readings and the averaged idle test values for CO. coefficient of correlation between the calculated average idle emissions and the obser RSD CO measurement was 0.50, much higher than found in the UK and Swe studies. One possible reason for this could be that because the idle test in Nevada a two speed component it is more accurately reflecting on-road emissions, and there remote sensing measurements.

The data obtained in the Nevada study was further analysed by comparing F readings for high emitters and low emitters to idle values. The relationship between two distributions was quite different above and below the arbitrary 4.0% cutoff to identify high emitters. For the 115 points with an RSD reading of less than 4 CO, r was equal to 0.57. For RSD readings greater than or equal to 4.0% CO, r equal to 0.29 for the 131 points. The lower correlation coefficient above the 4 cutpoint may be due to differences in operating conditions including heavy accelera or deceleration or cold start operation. Higher emitting vehicles are also likely to maintenance and have much more variable engine response.

The 4.0% cutoff value used to describe high emitters in the US would be inapprop for the UK. The UK emission standard of 3.5% or 6% CO - depending on the a the vehicle - could mean that a vehicle that passes the emissions check could be cl as a high emitter in the US. The scope for tightening emission limits in the U therefore very large.

A parallel analysis of HC measurements was also conducted. The results were variable and less predictable than CO measurements with a coefficient of correlati 0.38 being found for the full fleet.

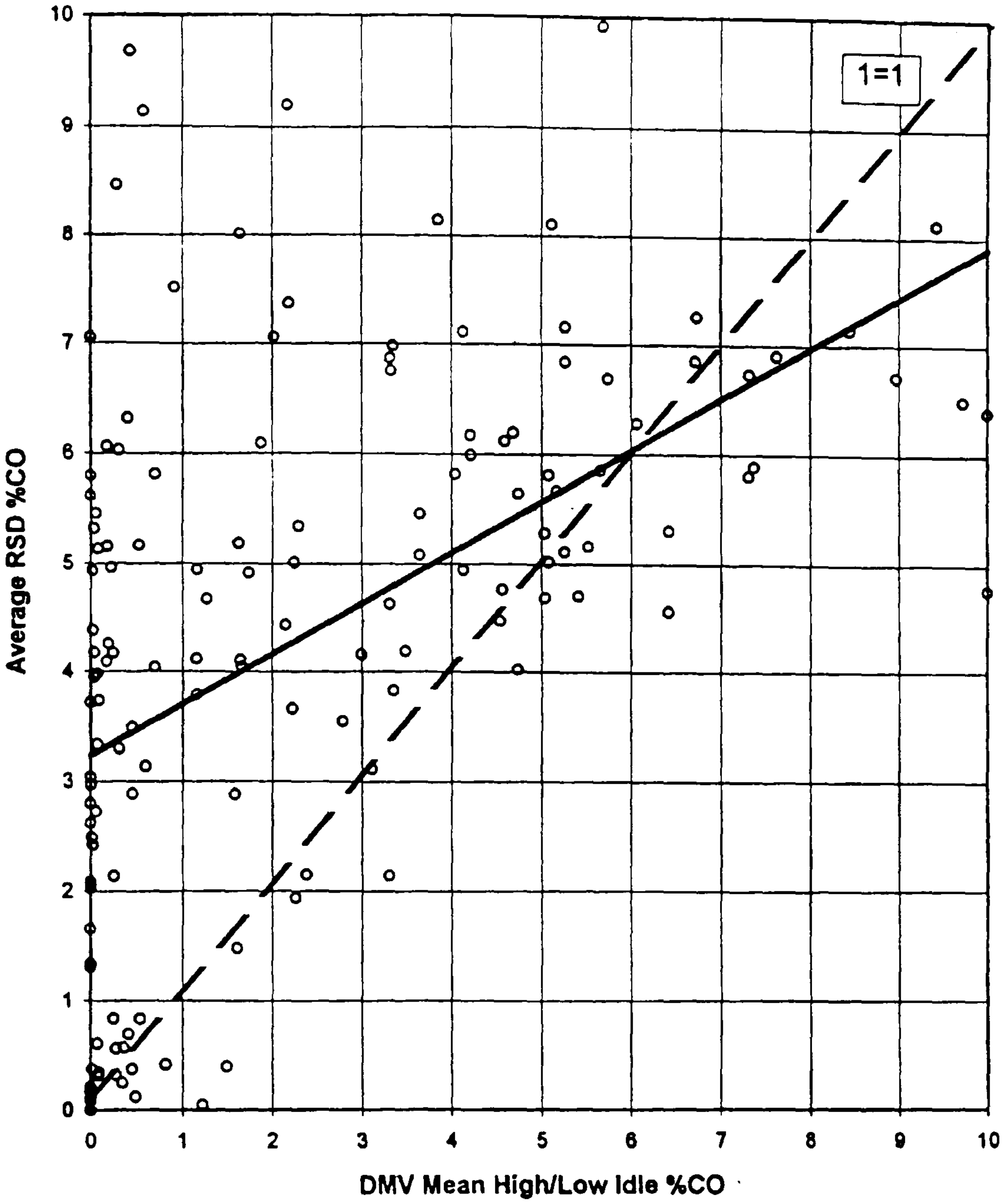


Figure 2.8 Comparison of %CO remote sensing measurement to mean (high an low speed) idle measurement; the solid line is the line of best fit *(Walsh et a 1996).*

2.3.4 Continuous comparison of FEAT and conventional measurement

A series of measurements was undertaken by TRL to compare FEAT to a conventi system operated on a chassis dynamometer. Data from both systems were reco continuously. *Figures 2.9 & 2.10* show concentrations recorded throughout tests the EC test cycle from a cold start *(Figure 2.9)* and a hot start *(Figure 2.10)* by the different systems. There is quite a high degree of agreement between the two se results visually. Both measurement procedures show emission rates varying in the : way, following the operational changes in the test cycle, and numerically the rat emission determined by the two methods are similar.

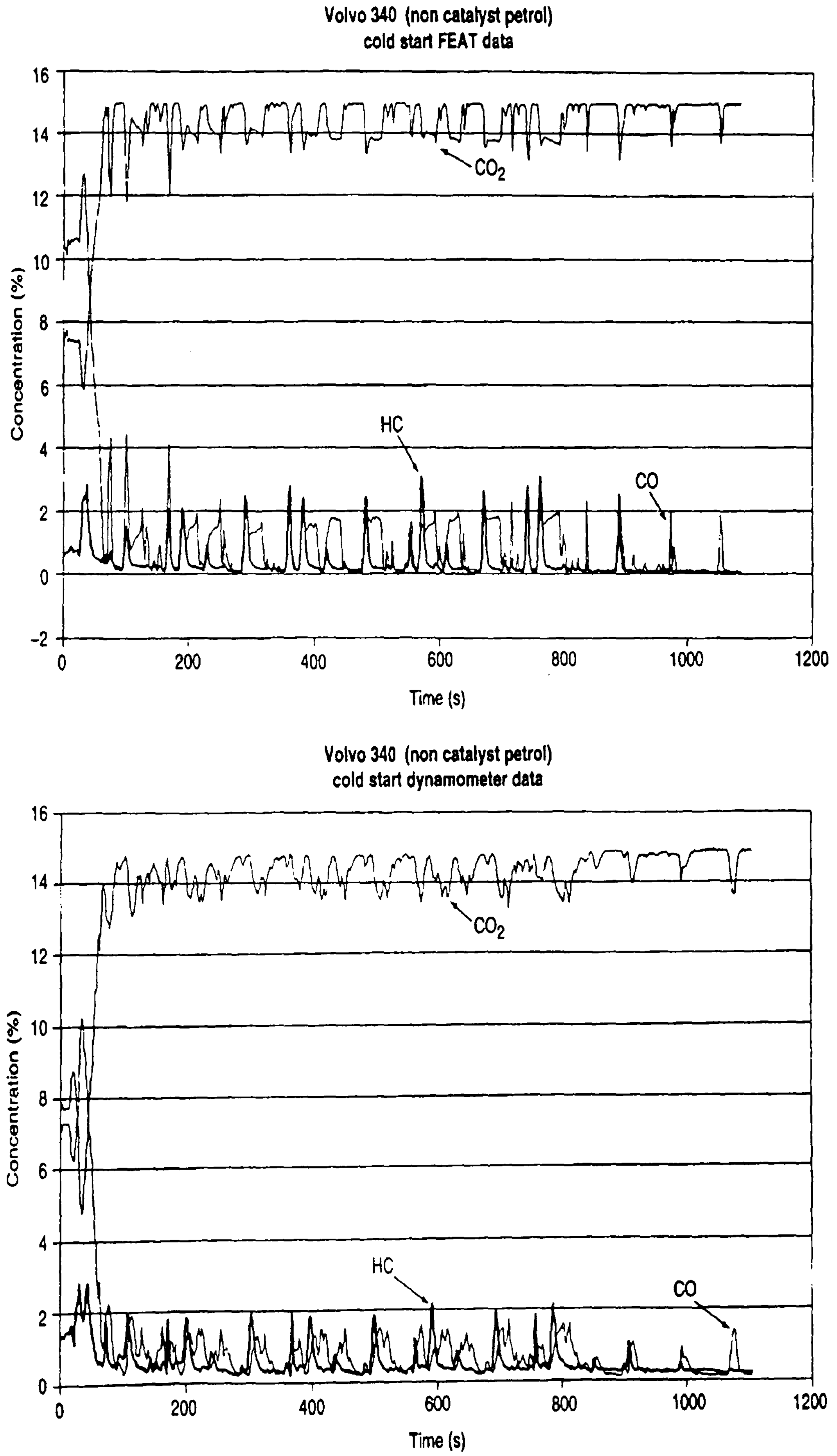


Figure 2.9 Emission rates measured by remote sensing and conventional laborato sampling *(Hickman & McCrae 1995).*

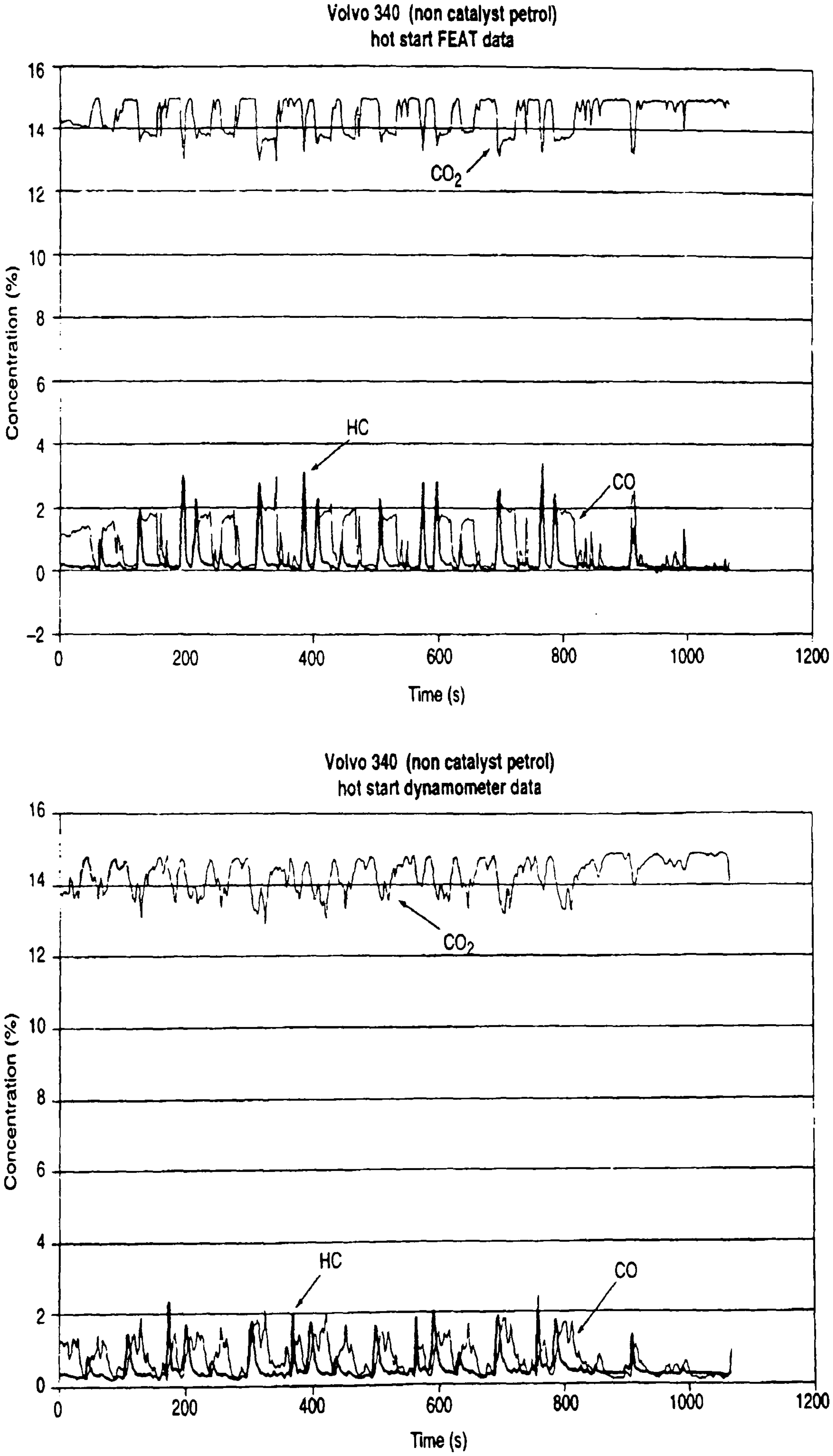


Figure 2.10 Emission rates measured by remote sensing and convention laboratory sampling *(Hickman & McCrae 1995)*.

A more quantitative analysis of other results from this series of tests is illustrated *Figures 2.11 & 2.12*. For the analysis, data from one car were chosen, as it was tes over the European Community (EC) cycle and the US Federal Test Procedure (FT Each of the figures shows carbon dioxide, carbon monoxide and hydrocarbon emissi over one test cycle: *Figure 2.11* uses the EC cycle and *Figure 2.12* the US FTP cycl

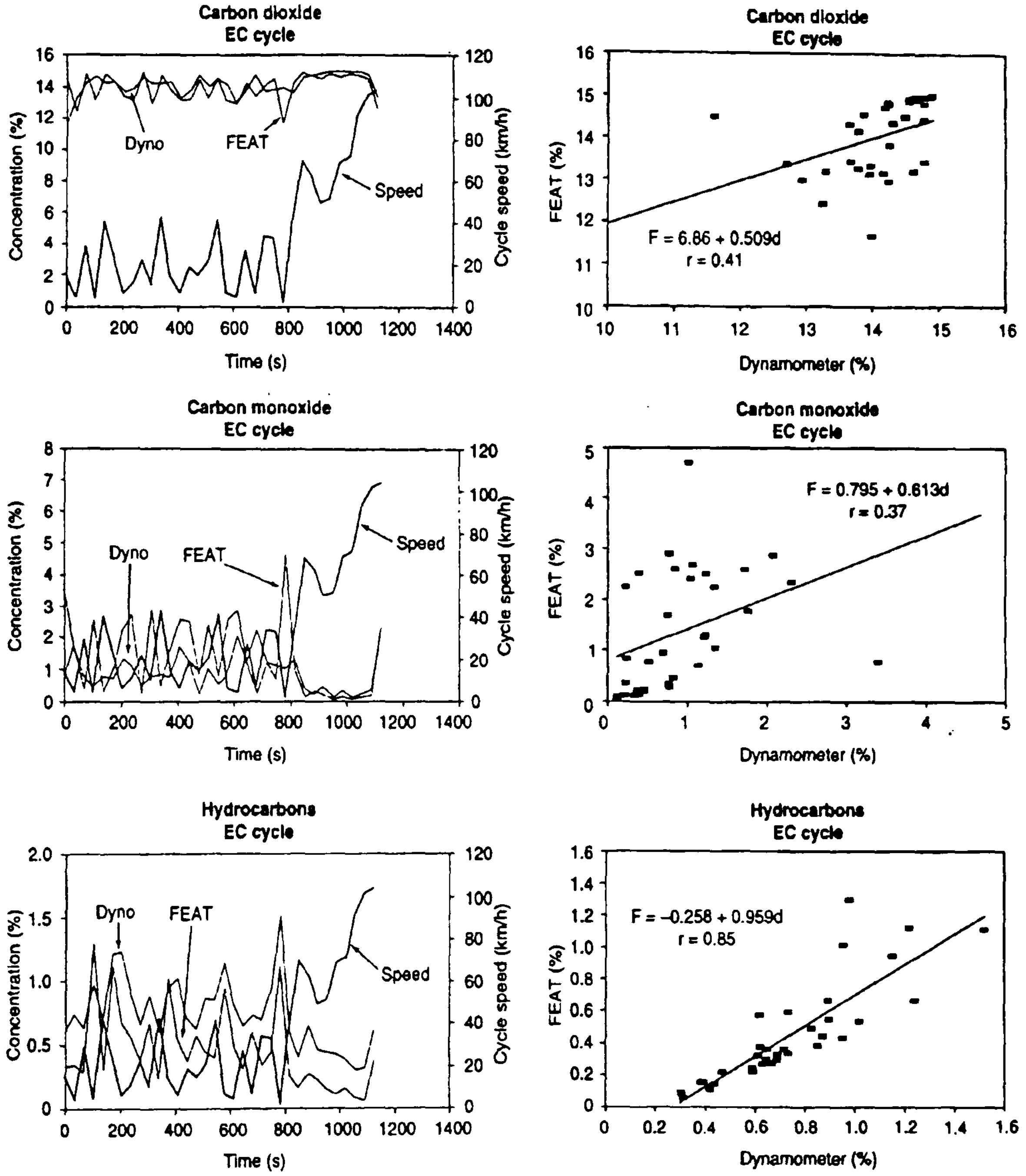


Figure 2.11 Comparison between emissions measured by remote sensing (FEA unit 3002) and chassis dynamometer for the EC test cycle *(Hickman & McCr 1995)*.

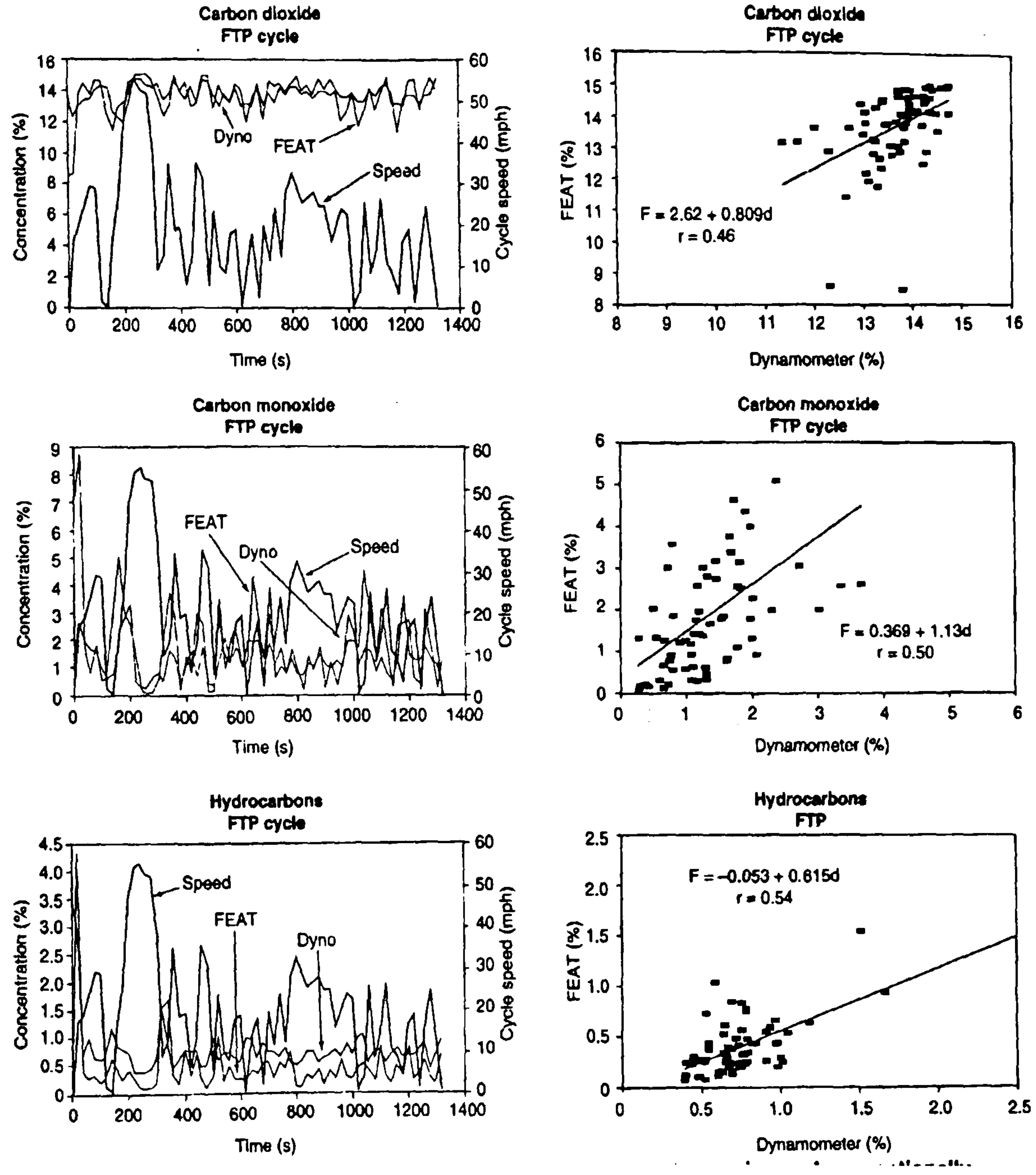


Figure 2.12 Comparison between emissions measured by remote sensing (FEAT unit 3007) and chassis dynamometer for the FTP test cycle *(Hickman & McCrae 1995)*.

Data recorded by both methods of measurement are shown together with the spee profile during the driving cycle. Again, the general features discussed earlier may observed, with emission rates as shown by both methods responding similarly changes in vehicle speed, and concentration levels being of similar magnitude. The da have also been examined statistically using a simple regression procedure. The resul of the analysis are also shown in *Figures 2.11 & 2.12*, with equations of the lines best fit and their correlation coefficients shown on the scatter plots. Correlati coefficients vary between 0.37 and 0.85 with an of average approximately 0.50 at statistical significance level of 0.05.

2.3.5 Variability in emissions from individual vehicles

The appearance of the same vehicle showing widely divergent emissions on duplic remote sensing systems has been seen as a weakness of FEAT. This could h resulted from an inability of the sensor to accurately measure instantaneous exh emissions. Typical data *(Stedman & Bishop et al. 1994)* obtained from 4122 vehi with matched license plates is shown in *Figure 2.13.* The data were collected Rosemead Boulevard, Los Angeles with two FEAT units 100 ft apart. There several outlying points on both the horizontal and vertical axes. These occur when instrument rejected the data for a particular vehicle and because of this were exclu from the regression analysis, which gave an r^2 of 0.54.

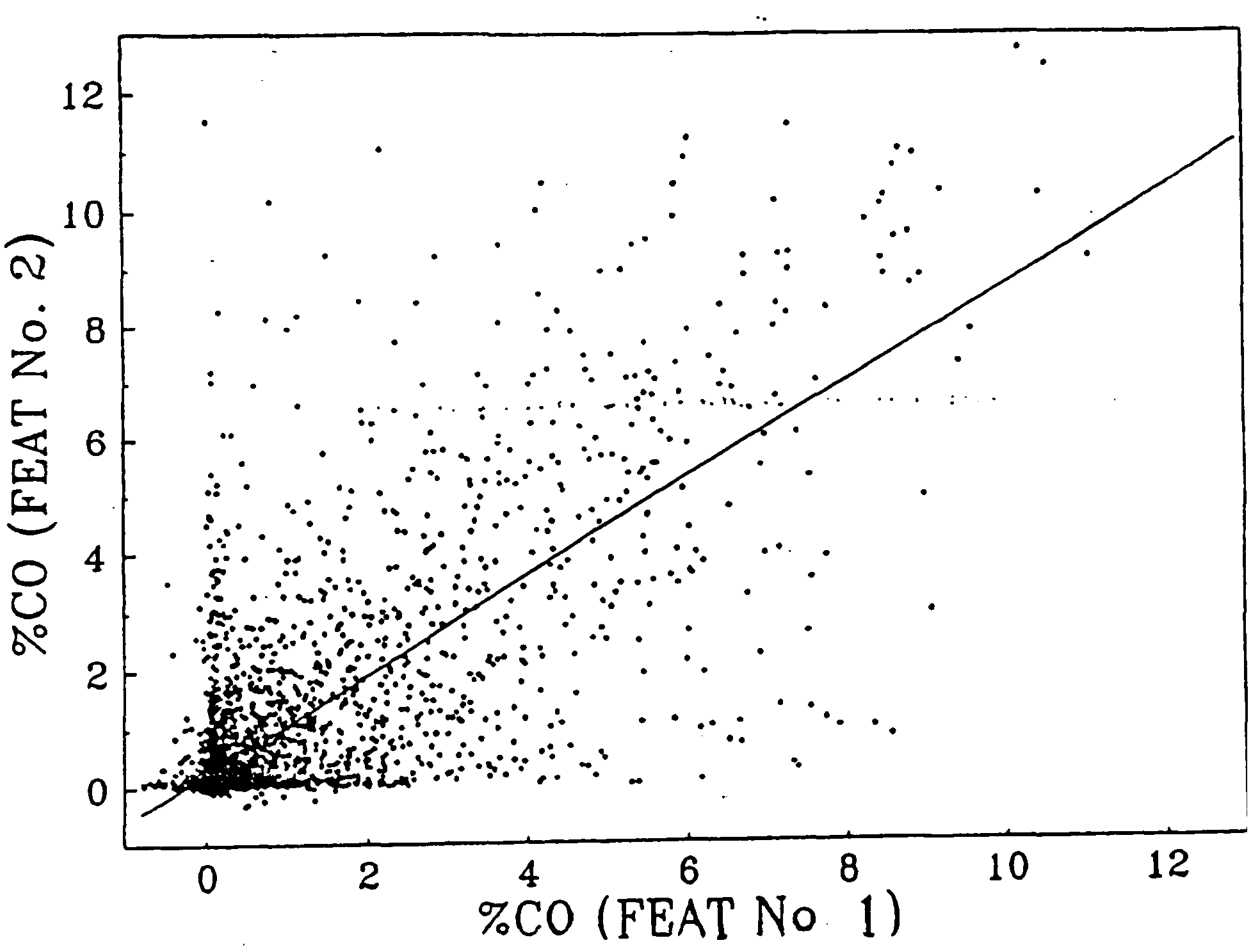


Figure 2.13 Data collected from two FEAT units approximately 100 ft apart. T equation of the regression line is:
FEAT(2) = 0.23 + 0.85 *FEAT(1) *(Stedman & Bishop et al. 1994).*

However, blind on-road comparison *(Figure 2.3)* to vehicles with known on-rc emissions has proved that FEAT can accurately measure instantaneous exha emissions. A possible reason for the lack of correlation could be that for some vehicl emissions variability is intrinsic to the vehicle. If this is the case, emissions variabil should be noted whatever the measurement technique.

In 1992, a major consortium of automobile manufacturers and oil companies undertc a study of the effects of proposed fuel modifications on emissions from late model ca Vehicles were recruited and segregated into two categories, normal emitters and h emitters. There were seven high emitting vehicles (defined as vehicles emitting > g/mile CO and >1 g/mile HC as measured by an IM240 test; an IM240 test is a t bag mass emission test conducted on a dynamometer over a transient driving cyc which were subjected to 14 separate Federal Test Procedure (FTP) tests.

The FTP test is conducted upon a chassis dynamometer, under a series of acceleratio decelerations, stops and starts. Inertia and friction are set for each vehicle. 7 emissions from each phase within the driving cycle are collected at a constant volu into three sample bags and the concentrations of each species determined. The driv cycle is modeled on a typical summertime commuter journey to work in Los Ange in the early 1970's. Each test takes at least 12 hrs to complete and costs over $10 Precision of the results for a given vehicle are reported as ±20% *(Berg, 1978)*.

Figure 2.14 shows the results for CO and HC emissions from five of these tests vehicles used the same base fuel). The absolute run to run variability of these rep measurements is quite high with the extreme emission levels varying by an orde magnitude. If we assume each measurement was correct and accurately represents emissions at the time of the test, then the vehicle must be responsible for the variab in emissions not the measurement technique.

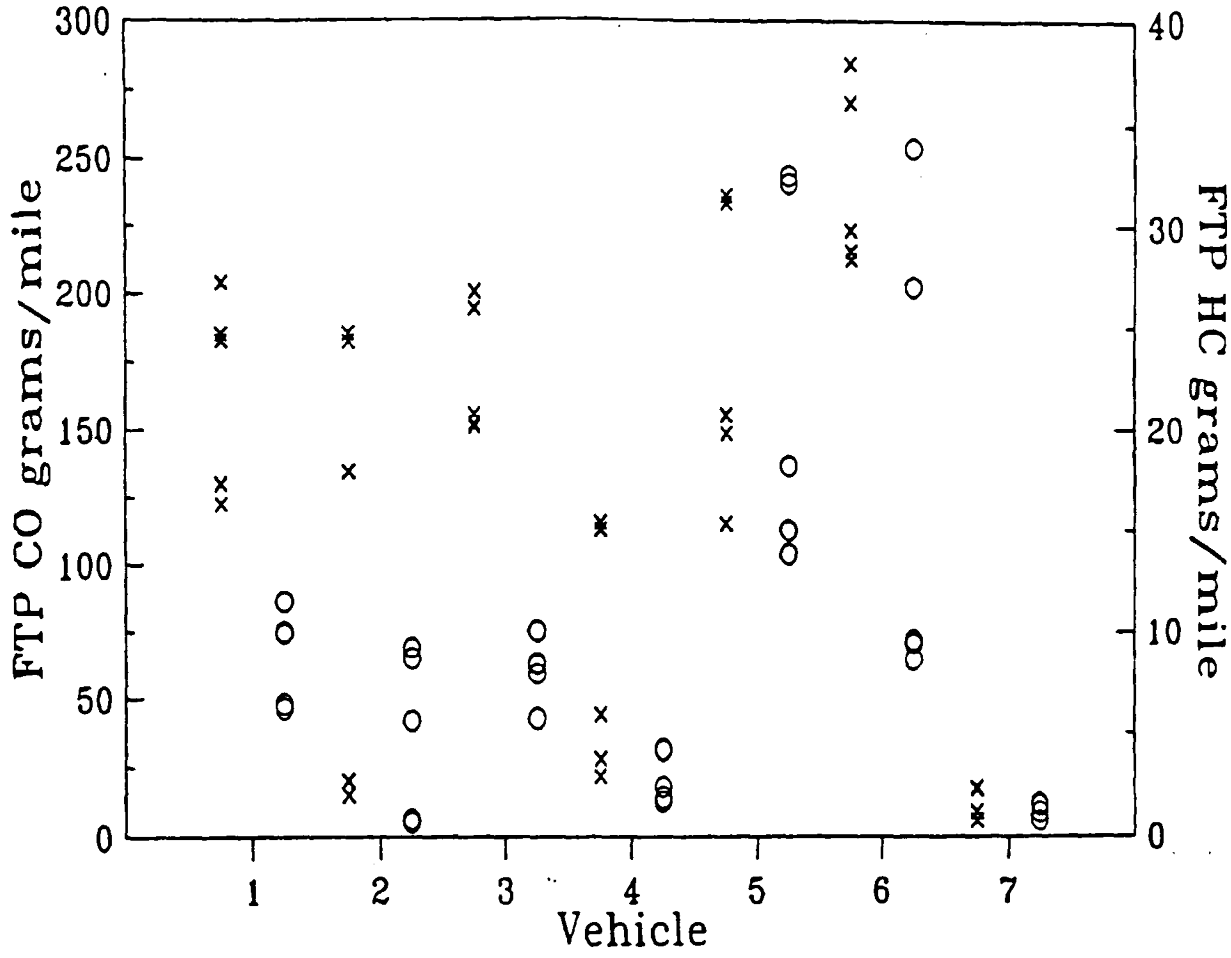


Figure 2.14 F.T.P. data for CO and HC emissions from 1989 and newer mod year high emitters. CO plotted as x and HC data signified by o *(Stedman & Bisho et al 1994)*.

2.4 Global on-road vehicle exhaust emissions

The University of Denver's FEAT system has been used to measure emissions in m locations around the world, logging data for more than 1,000,000 vehicles. *Zhan al. 1995* reviewed data from 22 of those locations with approximately 300,000 vel measurements recorded. The sampling sites were chosen on the basis of possessi good urban traffic flow and being representative of different regions around the w with correspondingly different fleet profiles. A fleet is defined as that populatio motor vehicles that pass by the remote sensor at a particular location during a de period. The sampling location, sites, measuring dates and number of records are sh in **Tables 2.1A and 2.1B.**

Table 2.1A Remote sensing sampling locations *(Zhang et al. 1995)*

Location	Site description	Date	Records
Bangkok,Thailand	westbound single lane on the south frontage of Si Ayutthaya Rd, level and high-traffic density with many motorcycles	Aug 1993	5260
Chicago,IL	uphill, traffic light-controlled, on-ramp to eastbound 1-290 from Central Ave., straight and busy with some hard accelerating vehicles	Jun 1992	8733
Denver,CO	tightly curved, uphill 4% grade off-ramp from southbound I-25 to southbound Speer Blvd., speed about 30 mph	Oct 1991	35945
Edinburgh, UK	generally level, straight and high-traffic density sites	Nov 1992	4524
Gothenburg, Sweden	uphill, curved freeway interchange ramp at Gullbergsmotet, the last exit northbound before tunnel to Hissingen Island	Sep 1991	10285
Hamburg, Germany	level, curved on-ramp to southbound freeway	May 1994	11128
Hong Kong	ramp from Queensway southbound onto Cottontree Dr. at the top of the steepest uphill section, high fraction of diesel powered vehicles	Aug 1993	5891
Kathmandu, Nepal	uphill section of Ram Shah Path Rd. and a level section westbound at Martyr's Gate, many motorcycles	Aug 1993	11227
Leicester, UK	similar site as in Edinburgh, but older fleet with many motorcycles	Nov 1992	4992
Lisbon, Portugal	slightly uphill, exit from Tagus Bridge auto route to north Lisbon	May 1994	10426
London, UK	similar site as in Edinburgh, but younger fleet, many motorcycles and diesel powered taxis	Nov 1992	11666

Table 2.1B Remote sensing sampling locations *(Zhang et al. 1995)*

Location	Site description	Date	Record
Los Angeles, CA	Rosemead Blvd., southbound to the Pomona Freeway (I-60) in El Monte, three-lane surface street narrowed by cones into a single	Jun 1991	42546
Lyons, France	uphill, interchange ramp from northwest bound freeway to westbound circular	May 1994	14276
Melbourne, Australia	combination of on- or off-ramps of a highway or a two-way road divided by a solid traffic island	May 1992	15908
Mexico City, Mexico	five sites, mostly highway exit or entry ramps, traffic flow cruising at about 25 mph	Feb 1991	31838
Milan, Italy	uphill, curved on ramp to northbound freeway to Turin	May 1994	13943
Rotterdam, The Netherlands	Kleinpolderplatz, uphill, interchange ramp for eastbound vehicles turning north towards Amsterdam	May 1994	12882
Seoul, South Korea	construction site just after the toll booths of Nam San Park Tunnel 1 southbound, about 2% uphill grade, slow moving congested traffic	Aug 1993	3104
Taipei, Taiwan	northbound on-ramp to the major north-south freeway, motorcycles are not allowed on the freeway	Aug 1993	12062
Thessaloniki, Greece	straight and level single lane of downtown one-way street, many motorcycles	Sep 1992	10536
Toronto, Canada	Bay view Extension, northbound, 2% upgrade ramp onto Don Valley Parkway	Apr 1990	11290
Zurich, Switzerland	uphill section of a single-lane road leaving Zurich to southeast Switzerland	Mar 1994	11298

2.4.1 Exhaust emission distributions

A summary of CO and HC emissions data is shown in **Tables 2.2 and 2.3** *(Zhang al. 1995)*. The mean values, for CO emissions, vary between 0.55% and 4.30%, w an average value of 1.5% for all fleets. The median value for each fleet varies betw 0.1% and 3.8%. The median of each fleet is lower than the mean for each locati indicating that average CO emissions of fleets throughout the world are dominated high emitting vehicles. The larger the difference between the mean and median for e fleet the greater the skewed nature of the distribution. Mexico City and Kathman stand out as having the highest CO emission levels and a near normal distributi Rotterdam, Denver and Gothenburg are cities that have emission levels below average value and a marked skewness in their distribution indicating that most vehic are low emitters.

A small fraction of the vehicles, in all fleets, referred to as gross polluters/h emitters, is responsible for the majority of total measured on-road exhaust emissions. In Hamburg, 7% of the vehicles were responsible for 50% of the emissions whereas in Kathmandu 25% of the vehicles were responsible for 50% of CO emissions. On average those fleets that are lower emitting have a lower percent of gross polluters with a lower gross polluter cutpoint (a gross polluter cutpoint is lowest concentration of any vehicle contributing half the total emitted pollutants) t those of the higher emitting fleets.

The mean values, for HC emissions, vary between 0.03% and 0.95%, with an aver value of 0.14%. The median value for each fleet varies between 0.02% and 0.57%. with CO, the median of each fleet is lower than the mean for each location, indica that average HC emissions of fleets throughout the world are dominated by h emitting vehicles. Bangkok and Kathmandu are outliers for HC emissions becaus the high percentage of two-cycle engines. These included three-wheeled motorcy many of which were poorly maintained and badly tuned. *Chan et al. (1995)* found CO and HC emissions from one such motorcycle are comparable to emissions fro catalyst equipped cars. Thessaloniki also contains a significant fraction of two-st vehicles, which emit high HC levels in their exhausts.

Table 2.2. Remote sensing CO emission data summary *(Zhang et al. 1995)*.

Location	Mean (%CO)	Median (%CO)	% Gross Polluter	Cutpoint (%
Bangkok[a]	3.04	2.54	21.65	5.24
Chicago	1.04	0.25	7.50	4.20
Denver	0.74	0.11	6.68	3.42
Edinburgh	1.48	0.69	13.40	3.44
Gothenburg	0.71	0.14	8.38	2.58
Hamburg	0.57	0.12	6.80	2.43
Hong Kong	0.96	0.18	9.17	3.41
Kathmandu[a]	3.85	3.69	24.88	5.85
Leicester[a]	2.32	1.61	18.15	4.33
Lisbon	1.48	0.38	12.00	4.21
London[a]	0.96	0.17	8.38	3.58
Los Angeles	0.79	0.15	6.97	3.46
Lyons	0.97	0.22	8.79	3.52
Melbourne	1.42	0.57	12.43	3.52
Mexico City	4.30	3.81	24.29	6.58
Milan	1.25	0.39	11.99	3.42
Rotterdam	0.55	0.13	6.99	2.31
Seoul	0.82	0.26	9.83	2.62
Taipei	1.49	0.88	16.78	3.09
Thessaloniki[a]	1.40	0.55	13.32	3.46
Toronto	0.75	0.15	7.93	2.85
Zurich	0.83	0.17	6.93	3.66

Locations marked with an [a] indicate significant fractions of two-stroke vehicles.

Table 2.3. Remote sensing HC emission data summary *(Zhang et al. 1995)*.

Location	Mean (%HC)	Median (%HC)	% Gross Polluter	Cutpoint (%
Bangkok[a]	0.95	0.57	17.93	1.98
Chicago	0.09	0.06	15.89	0.13
Denver	0.06	0.03	10.92	0.10
Edinburgh	0.13	0.08	11.87	0.23
Gothenburg	0.06	0.05	19.71	0.09
Hamburg	0.04	0.02	9.88	0.09
Hong Kong	0.05	0.04	15.87	0.08
Kathmandu[a]	0.76	0.36	16.15	1.06
Leicester[a]	0.21	0.13	13.78	0.33
Lisbon	0.06	0.03	10.42	0.12
London[a]	0.14	0.07	10.00	0.29
Los Angeles	0.07	0.04	10.27	0.15
Lyons	0.07	0.04	11.07	0.14
Melbourne	0.11	0.06	11.05	0.19
Mexico City	0.21	0.11	10.20	0.41
Milan	0.06	0.04	10.74	0.12
Rotterdam	0.04	0.02	9.91	0.08
Seoul	0.04	0.02	4.80	0.15
Taipei	0.06	0.05	20.09	0.09
Thessaloniki[a]	0.16	0.08	9.86	0.33
Zurich	0.03	0.02	9.34	0.06

Locations marked with an [a] indicate significant fractions of two-stroke vehicles.

It has been claimed by *Zhang et al. 1995* that London and Leicester also contain significant fraction of two-stroke vehicles. These would most likely be motorcycles. T FEAT equipment has difficulty measuring such vehicles because of their morpholog Therefore, the explanation that these fleets, because of their high fraction of two-stro vehicles, emit high HC levels is not wholly satisfactory.

2.4.2 Correlation of CO and HC emissions

When mean fleet emissions for CO were plotted against mean fleet emissions for H for these locations (fleets with significant fractions of two-stroke vehicles were omitte a regression equation was derived *(Zhang et al. 1995):*

$$\%HC = 0.046 * \%CO + 0.019 \qquad (r^2 = 0.81)$$

The strength of the relationship implies that mean HC exhaust emissions could approximately predicted by establishing the mean CO emissions for a given fleet. fleet with high CO emissions can also be expected to have a large number of high H emitters. This is because the factors that effect CO emissions in a fleet will also aff HC emissions in that fleet; differing levels of maintenance, average fleet age a imposition of compulsory emission standards. However, as has been noted *Muncaster et al. 1994,* an individual vehicle that is a CO high emitter is not necessar a high emitter of HC.

In contrast to fleet mean data, individual HC emissions and CO emissions sh considerable scatter *(Figure 2.15)*. There are some HC gross polluters with very l CO emissions. These vehicles most likely suffer from misfire (which can be caused ignition system failures) or lean misfire (where the air:fuel mixture has too little f to ignite). For rich air:fuel mixtures (where the air:fuel mixture has too little air to b all the fuel) the percentage CO increases as combustion becomes less complete, more unburned fuel passes through the cylinders. If there is a partially functio catalytic converter, it will convert the fuel that was not burned in the cylinder into C resulting in very high CO emissions and low HC levels. A fully functional emissi system with air addition will also oxidise the CO to CO_2, resulting in a low emitt vehicle. A very rich mixture combined with an ineffective/absent catalytic conve will result in both high CO and HC emissions. This is possible with a poorl maintained vehicle where one cylinder may be misfiring due to ignition problems whil the other cylinders are running rich.

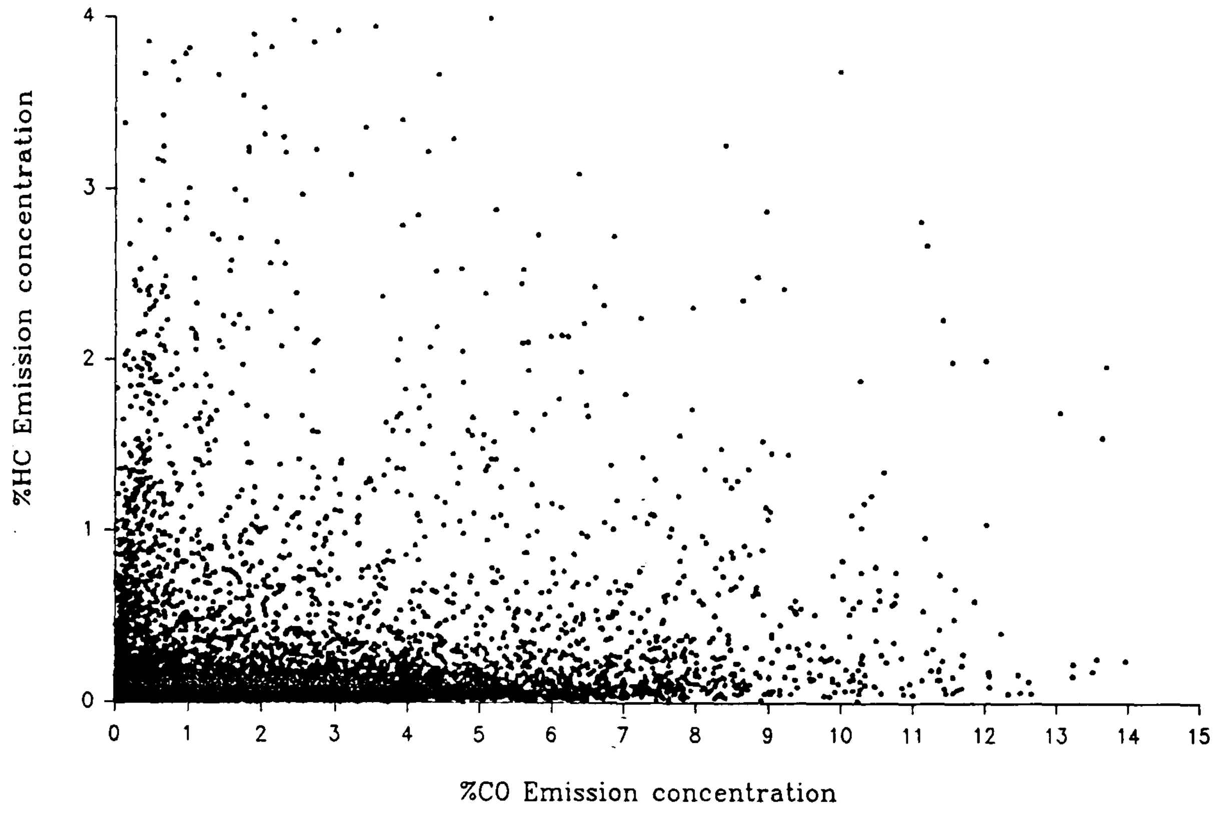


Figure 2.15 Relationship between individual vehicle HC emissions and CO emissions *(Muncaster et al. 1994).*

2.4.3 Cluster analysis for identifying the similarities in emission distributions of tl fleets

A hierarchial aggregative cluster analysis was undertaken by *Zhang et al. 1995* differentiate fleet distributions from around the globe. In the analysis each locati represented a sample, and the fraction of vehicles in each given emission ran represented the value of the variable that characterises the sample distribution. Tl dissimilarity between locations can be measured as the distance between samples; t larger the distance, the more dissimilar the two locations are.

The results of a hierarchical clustering are displayed in a dendrogram, which helps indicate the relationships between individual samples. The dendrogram is interpret by examining the way in which the samples group together to form the clusters and order in which the clusters combine to form large groups.

Figures 2.16 and 2.17 are dendrograms of the CO and HC emissions distributions. T cluster patterns for both CO and HC distributions have a major division into three lar clusters. The samples in cluster X generally have lower emissions than those in clus

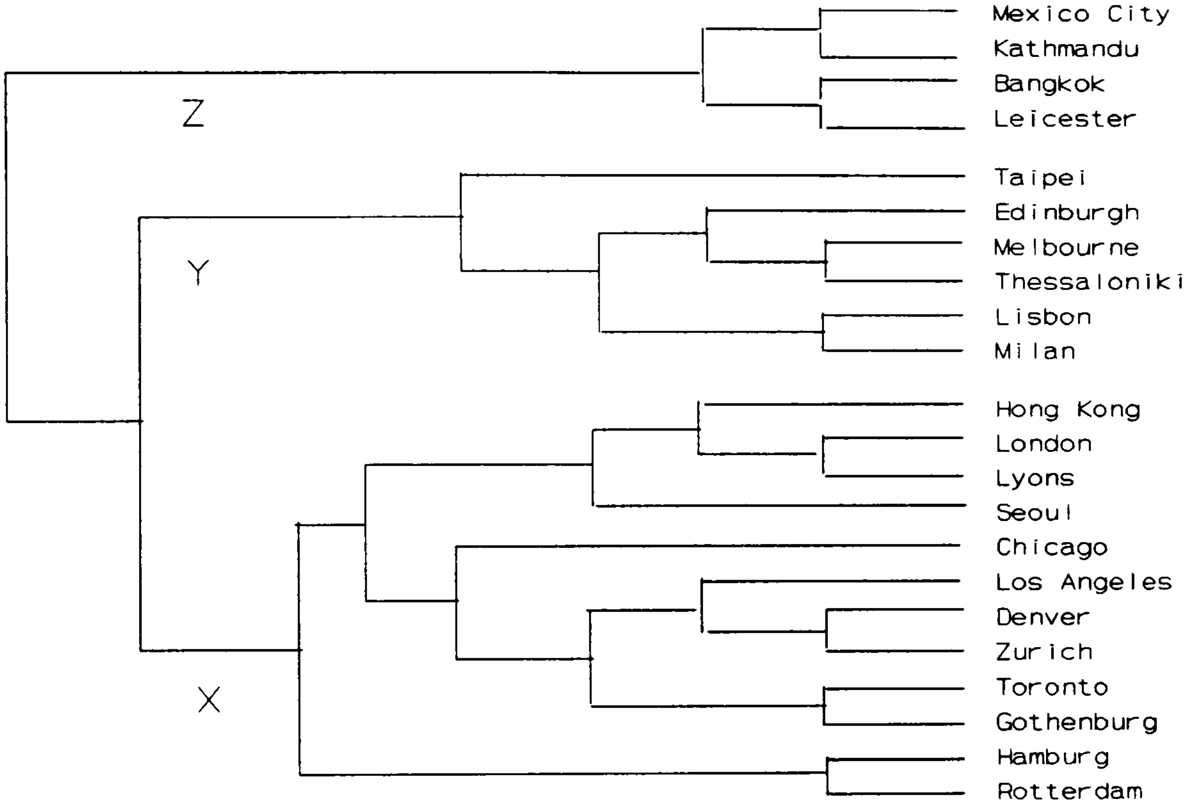


Figure 2.16 Dendrogram for the cluster analysis of CO emission distributions from 22 locations *(Zhang et al. 1995)*.

Z for both CO and HC's (cluster Z contains the samples with the highest aver emission rates). Most of the locations in cluster X have a new fleet and strict inspecti programmes. Clusters Y and Z have older fleets and, especially in cluster Z, the fl is composed of older vehicles that are less well maintained and are not subjected emission control legislation or new emission standards. However, age alone cannot fu explain all the differences. For example, pre-1970 USA vehicles have similar emiss technology to the vehicles in Mexico City, but the older vehicles from USA are average a factor of two lower in CO emissions *(Zhang et al. 1995)*. This implies t a difference in how mechanics tune and maintain the vehicles may be involved *(Bea et al. 1991, Beaton et al. 1992)*.

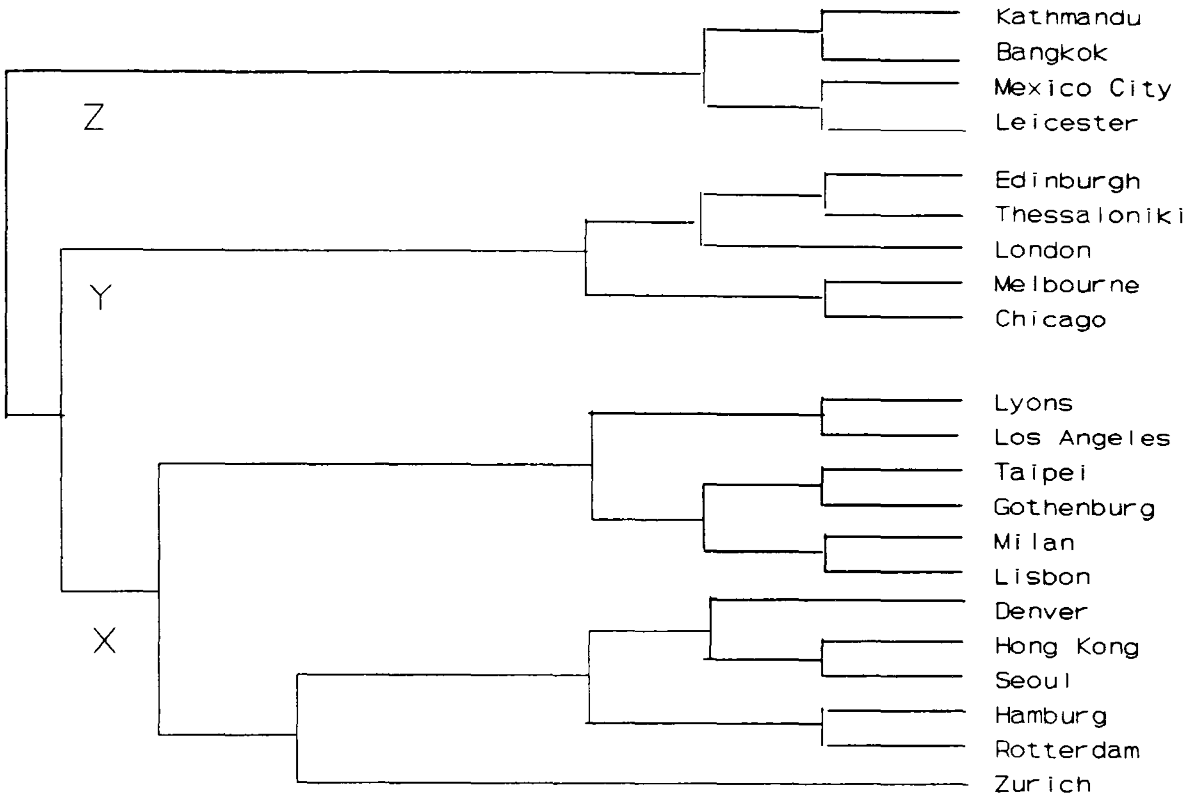


Figure 2.17 Dendrogram for the cluster analysis of HC emission distributions from 21 locations. No HC data are available for the Toronto fleet *(Zhang et al. 1995)*.

To assist in the understanding of the clusters three typical individual samples we chosen to illustrate the different cluster pattern for CO and HC, respectively *(Figur 2.18 & 2.19)*. The Denver fleet (cluster X) presents a highly skewed distribution i CO *(Figure 2.18)* such that about 80% of the fleet emitted less than 1% CO a contributed only 18% of total fleet emissions. In contrast, the Kathmandu fleet (clus Z) only has 20% of the fleet within the less than 1% CO emission category. T skewness of the distribution is decreasing in the order cluster X, cluster Y and clus Z. The lower the CO emissions of the fleet are, the more skewed the fleet distributio with fleet CO emissions being represented by a gamma distribution *(Zhang et al. 199*

Therefore, the moment of skewness may be used as an indicator of the avera emissions of a fleet. The distribution of the total CO emission contribution also chang with the cluster. In cluster X, the major contribution to total emissions is from th vehicles measured in the less than 1% CO category since the majority of vehicles res in this category. As the vehicle distribution becomes less skewed, gradually the emission contribution tends to be more normally distributed.

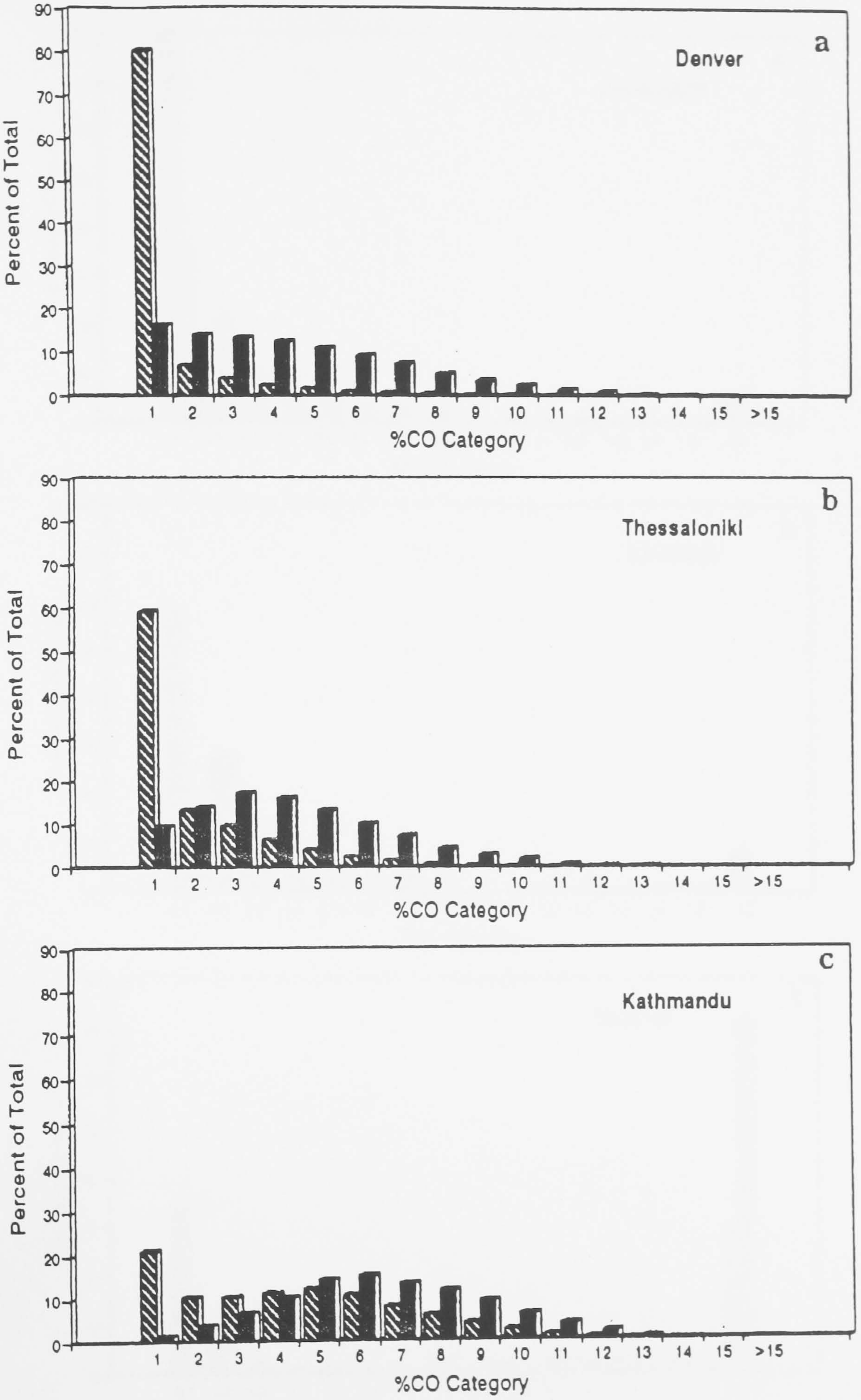
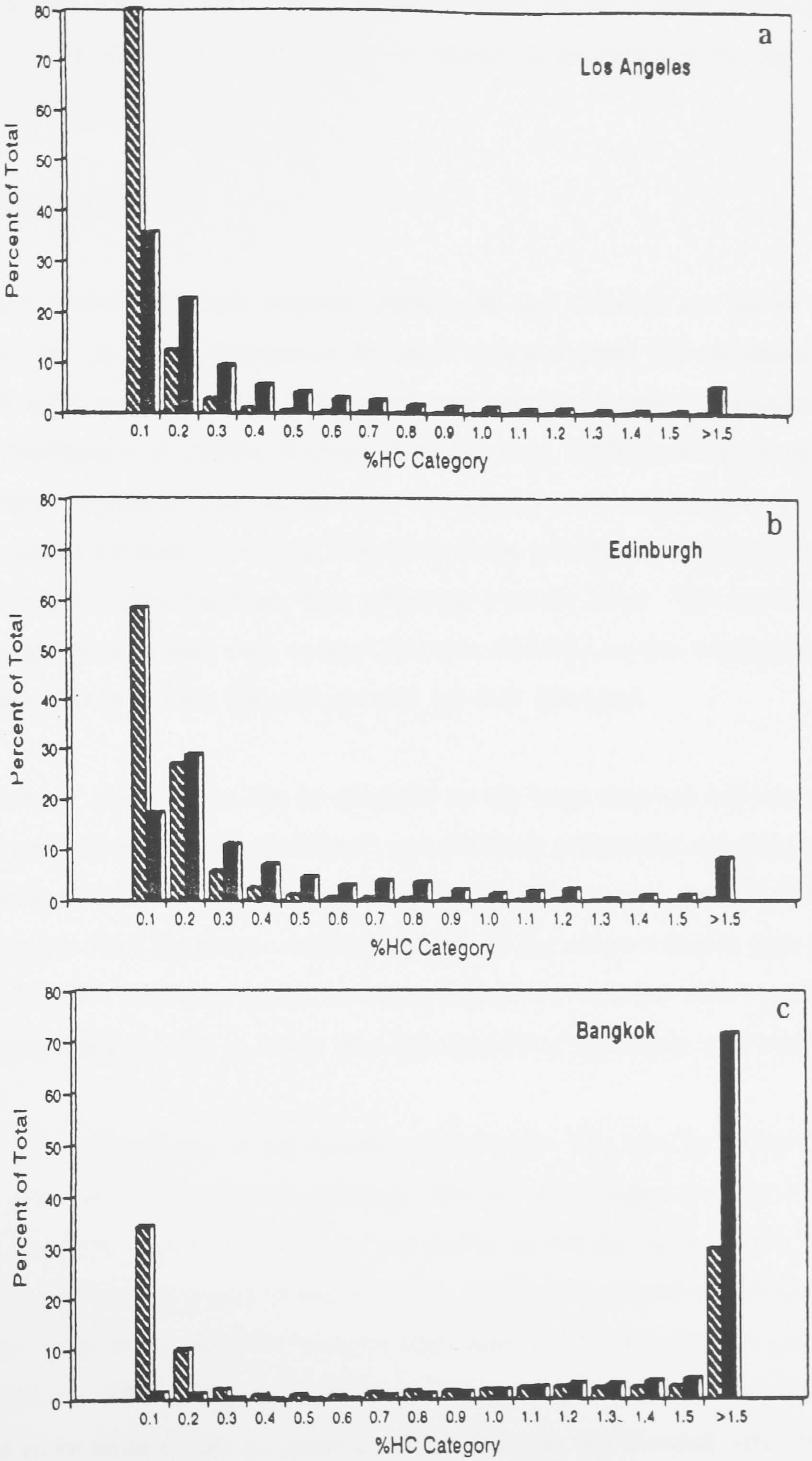


Figure 2.18 Histogram for Denver (a), Thessaloniki (b) and Kathmandu (c) CO data. The striped bars representing the % of the fleet and the solid bars the % of fleet emissions *(Zhang et al. 1995)*.



igure 2.19 Histogram for L.A. (a), Edinburgh (b) and Bangkok (c) HC data. he striped bars representing the % of the fleet and the solid bars the % of fleet nissions *(Zhang et al. 1995).*

For HC, the skewness of the vehicle distribution changes in a similar manner as for CO. The exception for HC is that the contribution of the highest emitting vehicles (mostly two-stroke) is extremely large in cluster Z as indicated by the data from Bangkok *(Figure 2.19)*.

2.4.4. Quintile analysis

Data from Gothenburg, Los Angeles, Melbourne and Leicester are shown in *Figure 2.20* with age of vehicle production on the X axis and mean CO emissions on the Y axis. All vehicles over 15 years old are gathered into the 15-year-old group as previous studies *(Stedman et al. 1991b, Stedman & Bishop et al. 1994)* have shown little further deterioration beyond 15 years of age. The increasingly poor maintenance, which causes the increase in emissions with age, also leads to the increased potential for the vehicle to remove itself from the fleet, thus achieving a steady state. The emissions within each age group have been rank ordered and then divided into five commensurate parts (quintiles). The emissions for each quintile are then averaged.

The effects of maintenance can be observed by the large absolute difference between the well maintained vehicles (quintile 1) and the badly maintained vehicles (quintile 5). The highest emitting quintile of the newest vehicles (age group 1,quintile 5) has higher CO emissions than the lowest emitting quintile of the oldest vehicles (age group 15, quintile 1). This indicates in-use emission differences between well maintained and badly maintained vehicles is larger than age dependent deterioration of emissions.

All four fleets deteriorate as the vehicle age increases. The average CO emission rate and the deterioration rate for the Leicester fleet are much higher than the rates for the Gothenburg fleet. However, the fleets are similar in that the same models of vehicles are driven and similar grades of fuel are used. Sweden introduced closed-loop catalytic converters over three years for vehicles representing the 4-6 years age group (model years 1986 to 1988) in *Figure 2.20.* The effect of this is clearly noticeable with the decrease in emissions from all quintiles of the 4-6 year old Swedish vehicles. The big differences observed between the Leicester and Gothenburg fleets older than 5 years presumably arise from better maintenance in Sweden.

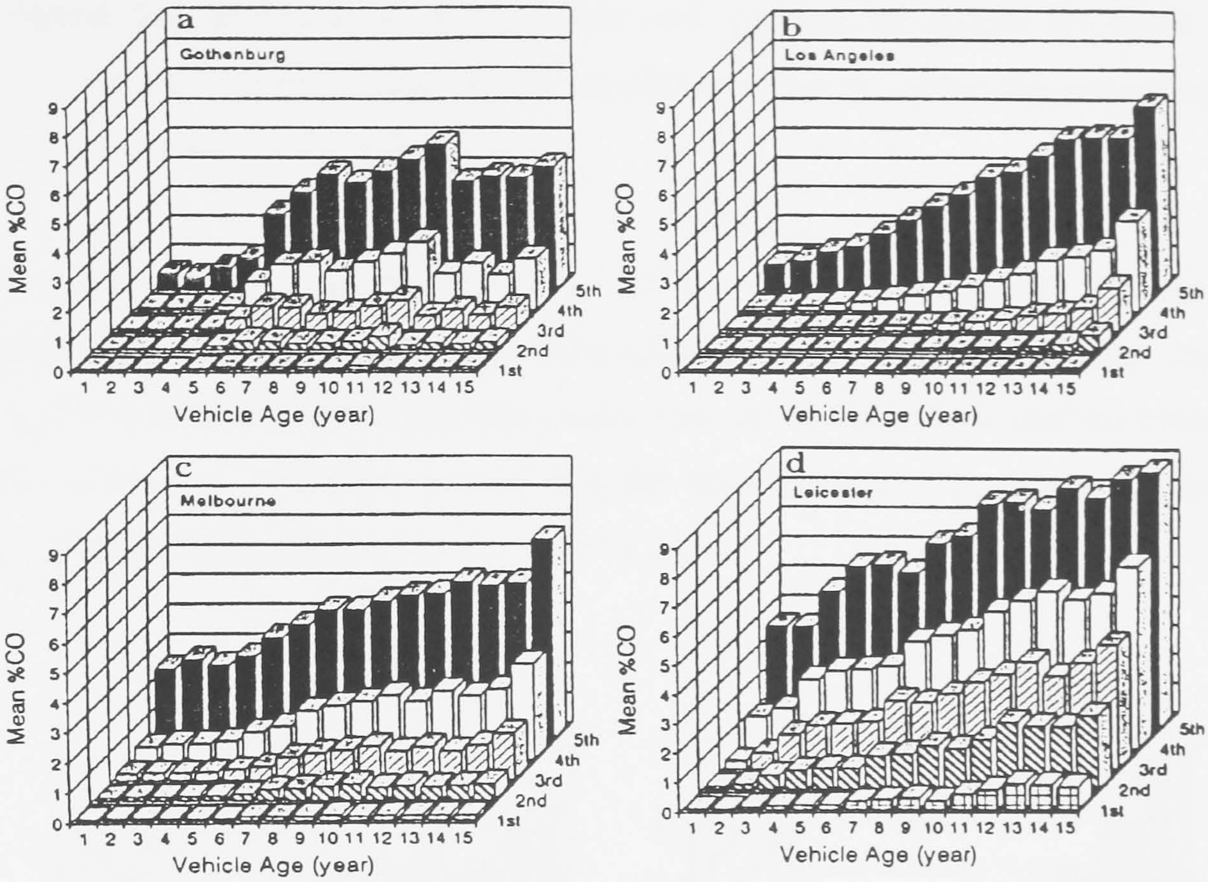


Figure 2.20 Gothenburg (a), Los Angeles (b), Melbourne (c) and Leicester (d) CO data presented as emission factors by age divided into quintiles *(Zhang et al. 1995)*.

The Australian fleet introduced catalytic converters to vehicles representing the 6 to 7 years age group (model year 1985 and 1986) in *Figure 2.20.* Zhang et al. 1995 claim that, in contrast to Gothenburg, the effects of catalysts in Melbourne are detectable only among the lower emitting quintiles whereas in Gothenburg there is a dramatic drop through all quintiles. However, close examination of *Figure 2.20* does not show the effects of catalysts to be detectable only among the lower emitting quintiles. The effect of the catalysts appear to visible across all quintiles with a decrease in emissions from quintile 5 being especially apparent.

In the United States catalysts were introduced in the early 1970s and then updated in the early 1980s but there are no signs of any sharp breaks in the Los Angeles plot to coincide with these changes. This observation suggests that advanced emission control technology is most effective only when combined with proper vehicle maintenance. To support the assumption that vehicles are better maintained in Sweden, vehicles over 10 years old that are not catalyst equipped show lower average CO emissions than vehicles of a corresponding age, but originally catalyst equipped, in the USA. This supports the observation that deterioration of emissions with age can be mainly attributed to a progressive lack of maintenance and an increase in emission system tampering *(Motor Vehicle Tampering Survey 1989, 1990)*.

Figure 2.21 shows the same organisation of plots as in *Figure 2.20* but for HC emissions. Zhang et al. 1995 provide little discussion of the HC emissions data, merely stating that most observations and deductions from the CO emissions analysis hold true for HC emissions. However, a more detailed description of HC emissions may be obtained from careful inspection of *Figure 2.21*.

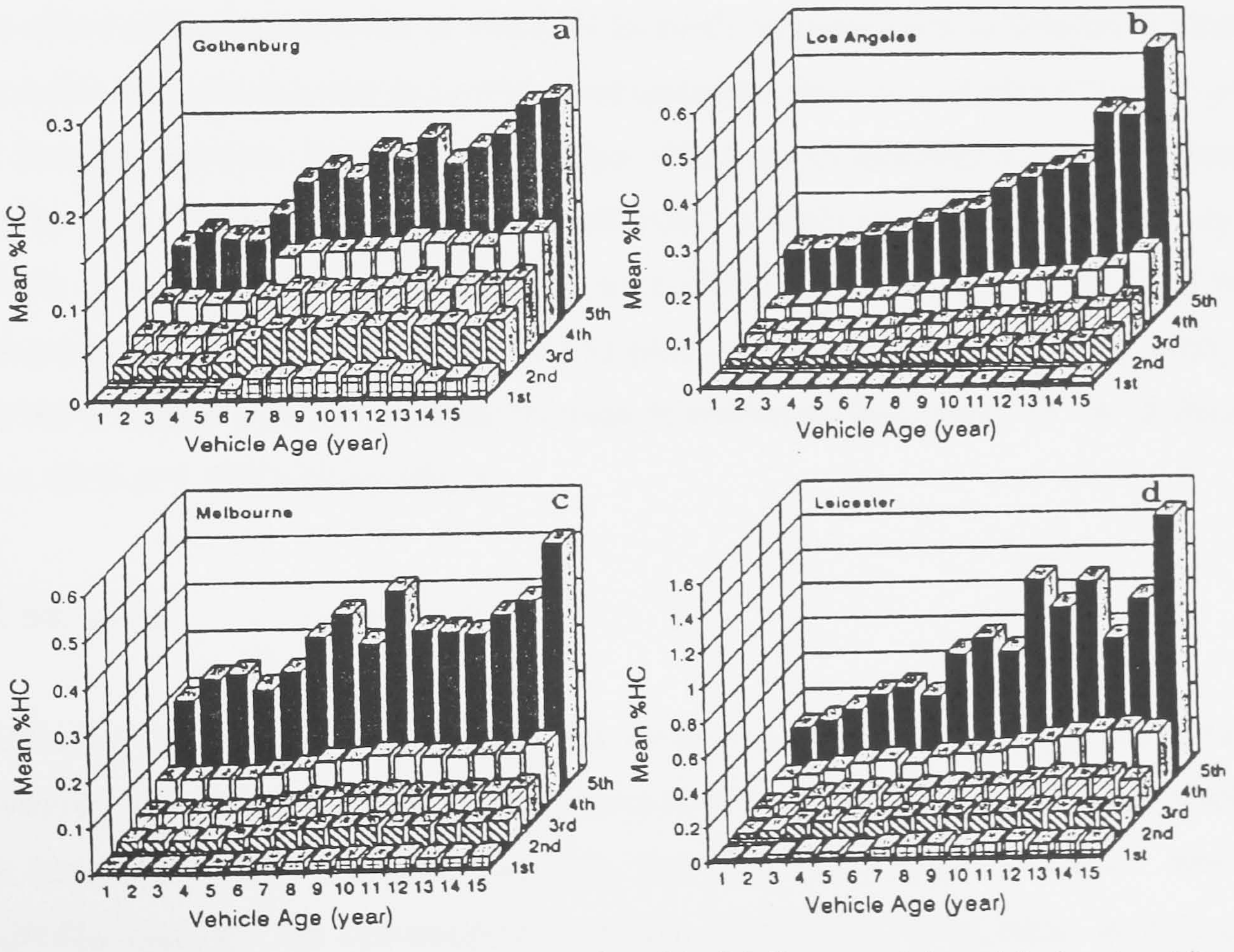


Figure 2.21 Gothenburg (a), Los Angeles (b), Melbourne (c) and Leicester (d) HC data presented as emission factors by age divided into quintiles *(Zhang et al. 1995)*.

The effects of maintenance upon emissions are significant, with a large noticeable difference between the well maintained vehicles in quintile 1 and the badly maintained vehicles in quintile 5. The highest emitting quintile of the newest vehicles (age group 1, quintile 5) has higher HC emissions than the lowest emitting quintile of the oldest vehicles (age group 15, quintile 1). This indicates that in-use emission differences between well maintained and badly maintained vehicles are larger than age-dependent deterioration of hydrocarbon emissions.

All four fleets deteriorate as the vehicle age increases. The average HC emission rate and the deterioration rate for the Leicester fleet are much higher than the rates for the Gothenburg, Los Angeles and Melbourne fleet (note changes in Y axis). The Gothenburg fleet has the lowest HC emission rate with emissions significantly less than for the other fleets. The Gothenburg fleet also differs in that the contribution to total emissions from the high emitter component (quintile 5) is smaller for HC compared to CO whereas in every other fleet the situation is reversed.

The effect of the introduction of closed-loop catalytic converters in Sweden is clearly noticeable with the decrease in hydrocarbon emissions from all quintiles of the 4-6 year old Swedish vehicles. However, the catalyst effect across quintiles is less pronounced than for CO. The effects of catalyst introduction in Melbourne are detectable only in quintile 5. In the United States catalysts were introduced in the early 1970s and then updated in the early 1980s. Unlike for CO emissions the effects of updating catalysts in 1980 is visible with a noticeable decrease in emissions from years 13 to 12 (model years 1979 and 1980) in quintile 5.

2.5 Summary

The Fuel Efficiency Automobile Test was developed in 1987 to measure CO exhaust emissions. Since 1987 emission channels have been developed for HC, NO_x and smoke. The basic instrument, as used, measures the carbon monoxide to carbon dioxide (CO/CO_2) ratio and the hydrocarbon to carbon dioxide (HC/CO_2) ratio in vehicular exhaust emissions. These ratios provide a qualitative assessment of vehicle emissions, a high ratio being indicative of a highly polluting vehicle and conversely a low ratio representing a clean vehicle. Using combustion equations and their derivatives, values can be derived for: emissions in grams per gallon, emissions in grams per mile and percentage emission in the raw exhaust.

The FEAT system has undergone rigorous testing, comparison and validation. Validation of the remote sensor was undertaken by comparing remote sensing measurements to onboard measurements. The FEAT units compared very well to the onboard measurements, measuring HC's to within ±15% of the calibrated on-board measurements and CO values to within ±5% of the on-board measurement. Comparison of measurements made using different remote sensors to the same emissions has taken place both in the UK and the USA, providing r^2 values of between 0.99 and 0.92 for CO and values of between 0.87 and 0.77 for HC.

Comparison of FEAT emission measurement to idle emissions measurement has also taken place but with mixed success. This is not surprising as emissions vary according to a vehicles mode of operation and speed, and as the remote sensor is sampling emissions from higher emitting modes than the no-load idle test a close correlation would be unlikely. A series of measurements were undertaken by TRL to compare FEAT to a conventional system operated on a chassis dynamometer. It was found that there was quite a high degree of agreement between the two measurement procedures.

The University of Denver's FEAT system has been used to measure emissions in many locations around the world, logging data for more than 1,000,000 vehicles. *Zhang et al. 1995* reviewed data from 22 of those locations. A small fraction of the vehicles, in all fleets, referred to as gross polluters/high emitters, are responsible for the majority of total measured on-road exhaust emissions of CO and HC.

There is a strong relationship between mean fleet emissions for CO and mean fleet emissions for HC. This is because the factors that effect CO emissions in a fleet will also effect HC emissions in that fleet; differing levels of maintenance, average fleet age and imposition of compulsory emission standards. However, an individual vehicle that is a CO high emitter is not necessarily a high emitter of HC.

A hierarchial aggregative cluster analysis showed cluster patterns for CO and HC distributions to have a major division into three large clusters (X, Y, Z). Most of the locations in cluster X have a new fleet and strict inspection programmes. Clusters Y and Z have older fleets and, especially in cluster Z, the fleet is composed of older vehicles that are less well maintained and are not subjected to emission control legislation or new emission standards.

Quintile analysis was conducted on emissions data from Gothenburg, Los Angeles, Melbourne and Leicester. The effects of maintenance can be observed by the large absolute difference between the well maintained vehicles (quintile 1) and the badly maintained vehicles (quintile 5). The highest emitting quintile of the newest vehicles has higher CO and HC emissions than the lowest emitting quintile of the oldest vehicles indicating that in-use emission differences between well maintained and badly maintained vehicles is larger than age dependent deterioration of emissions. All four fleet emissions deteriorate as the vehicle age increases with the effect of catalysts most apparent for fleets with good vehicle maintenance.

CHAPTER 3. AMBIENT AIR QUALITY

3.1 Introduction

This chapter is composed of two distinct sections. The first of these gives an overview of the air quality measurement of carbon monoxide (CO), using a non-dispersive infrared gas analyser, and nitrogen oxides (NO_x), using a chemiluminescent gas analyser, together with data acquisition. The second section is a review of urban air quality beginning with a general introduction to vehicle pollutants which is followed by more detailed information regarding the sources, typical concentrations, air quality legislation and health effects for CO and NO_x.

3.2 Ambient air quality measurement

Ambient air quality measurement can be classified into essentially two types, passive and active measurement. Passive measurement is less sensitive and has longer averaging times, typically 24 hours to one month. Passive measurement also has the disadvantage of not being able to be connected to a telemetry system, making data acquisition difficult. The more sophisticated active measurement techniques such as continuous monitoring, used in the Department of Environment (DoE) Automated Urban Network, do not have these disadvantages and can give almost instantaneous measurements and provide real time peak values. Continuous electronic monitoring was employed in this study.

3.2.1 Carbon monoxide

Carbon monoxide (CO) was monitored using a Horiba Instruments (model APMA-350E) cross-flow modulation, non-dispersive infrared (NDIR) gas absorption analyser which allows continuous real time measurements. In the traditional NDIR method *(Stern 1976)*, infrared radiation is split into two parallel beams, one passing through the sample cell and one through a reference cell *(Figure 3.1)*. The reference cell contains a non-infrared absorbing gas while the sample cell is continuously flushed with the sample atmosphere. The detector consists of a two-compartment gas cell (both compartments filled with CO) separated by a diaphragm, any movement of which causes a change of electrical capacitance in an external circuit, and ultimately an amplified electrical signal which is suitable for input to a recorder.

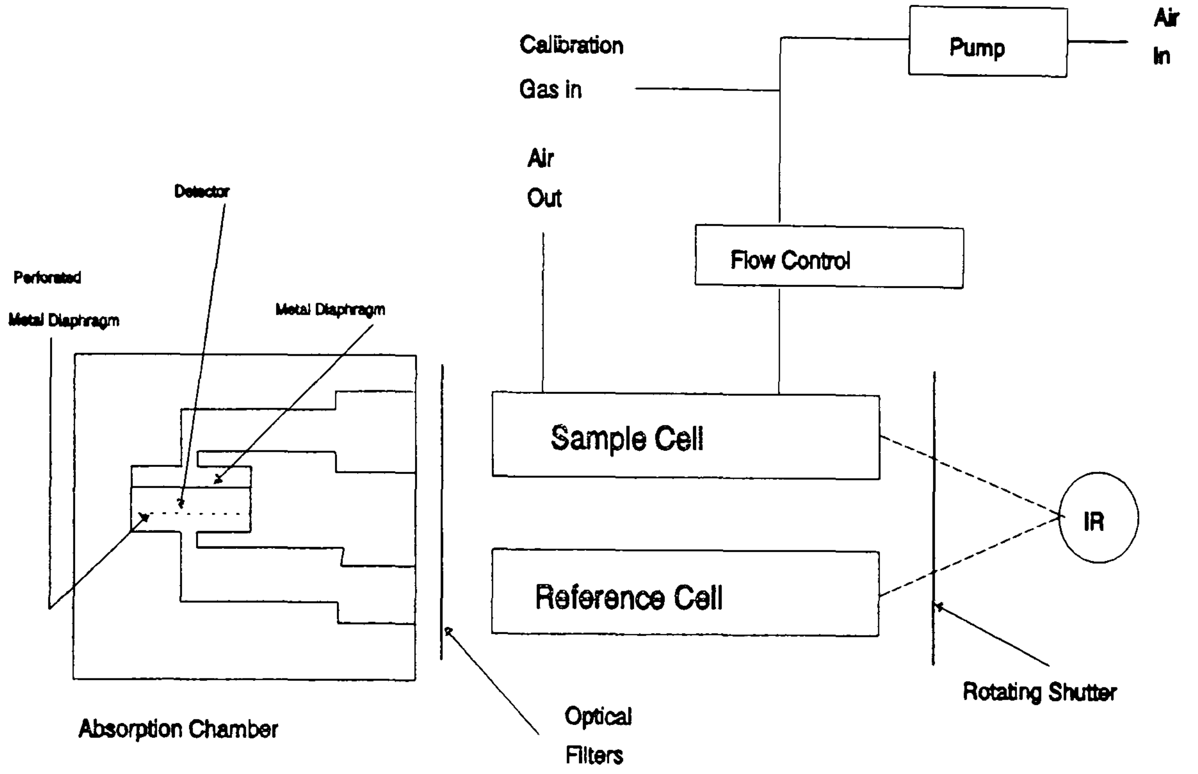


Figure 3.1 Schematic diagram of a traditional NDIR analyser *(McCrae 1988).*

During analyser operation a rotating shutter alternately exposes the reference cell and the sample cell to the infrared source. At the frequency of the shutter a constant amount of infrared energy passes through the reference cell to one compartment of the detector cell while a varying amount of infrared energy, indirectly proportional to the CO concentration in the sample cell, reaches the other detector cell compartment. These unequal amounts of residual infrared energy reaching the two compartments of the detector are reflected by unequal transformation of infrared energy to heat and hence to mechanical energy in the form of detector gas expansion. This unequal expansion is the force which causes the movement in the detector cell diaphragm and the resulting variation in the electrical signal.

One problem that this traditional technique can suffer from is the presence of water vapour in the sample. However, in the Horiba model APMA-350E instrument this is minimised by passing the sample through a silica gel drying agent. Interference is given as within ± 0.4 ppm.

Minimum detectable limits for this technique are nominally 0.5 ppm. However, using the cross flow modulation technique the detection limit is lowered to 0.06 ppm with noise within ± 0.015 ppm. In this technique there is only one infrared beam in the analyser unit *(Figure 3.2)* which passes through the cell to the detector. During measurement a rotary valve alternately introduces the sample gas and reference gas (ambient air with all CO removed by an internal cutter) into the cell. The presence of CO in the sample gas generates a difference in the intensity of light reaching the detector, between when the cell is filled with sample gas and when it is filled with reference gas. This difference causes a metallic membrane in the detector to move back and forth according to the alternating gas flow *(cross-flow modulation)*.

The analyser requires neither an optical chopper nor optical adjustment. Furthermore, this technique virtually eliminates zero drift and greatly enhances analyser sensitivity *(Horiba Instruments 1992)*. Zero and span calibrations are performed by introducing the appropriate blank and standard gases. Zero drift is specified as within ± 0.2 ppm/day and span drift as within ± 2% full scale measurement/day which equates to ± 0.16 ppm as calibration was undertaken with a 8 ppm standard. Carbon monoxide air quality data in Chapter 5 is therefore quoted to one decimal place only. The monitor has four selectable ranges from 0.06 ppm to 100 ppm with a response time (T_{90} ; time required to reach 90% of the maximum value) of one minute *(Horiba Instruments 1992)*.

3.2.2 Nitrogen oxides

Nitrogen oxides (NO_x) were monitored using a Horiba Instruments (model APNA-350E) chemiluminescent gas analyser. The principle behind the technique is that when NO reacts with ozone to form NO_2, some NO_2 is excited to a higher electronic state (equation 3.1), and as these excited molecules return to the ground state, detectable chemiluminescence is emitted (equation 3.2). As the activated NO_2 reverts to a lower energy state it emits radiation in the range of 600 to 3000 nm with a maximum intensity at 1200 nm *(Harrison and Perry 1986)*. The emitted radiation consequently produces a current via a detector that is directly proportional to the emission intensity and thus to the NO concentration in the sample.

Equation 3.1 $NO + O_3 = NO_2^{*} + O_2$

Equation 3.2 $NO_2^{*} = NO_2 + hv$

The instrument operates by dividing the filtered sample gas into two lines *(Figure 3.3)*. One line passes the gas on unaltered whereas in the second, the gas flows through a converter that reduces NO_2 to NO. The gas flow to the reaction chamber is switched between the two separate flows by a solenoid valve with the rate of flow controlled by a capillary. Ozone is supplied to the reaction chamber at a constant rate by an internal ozonator that uses dehumidified ambient air as feed gas.

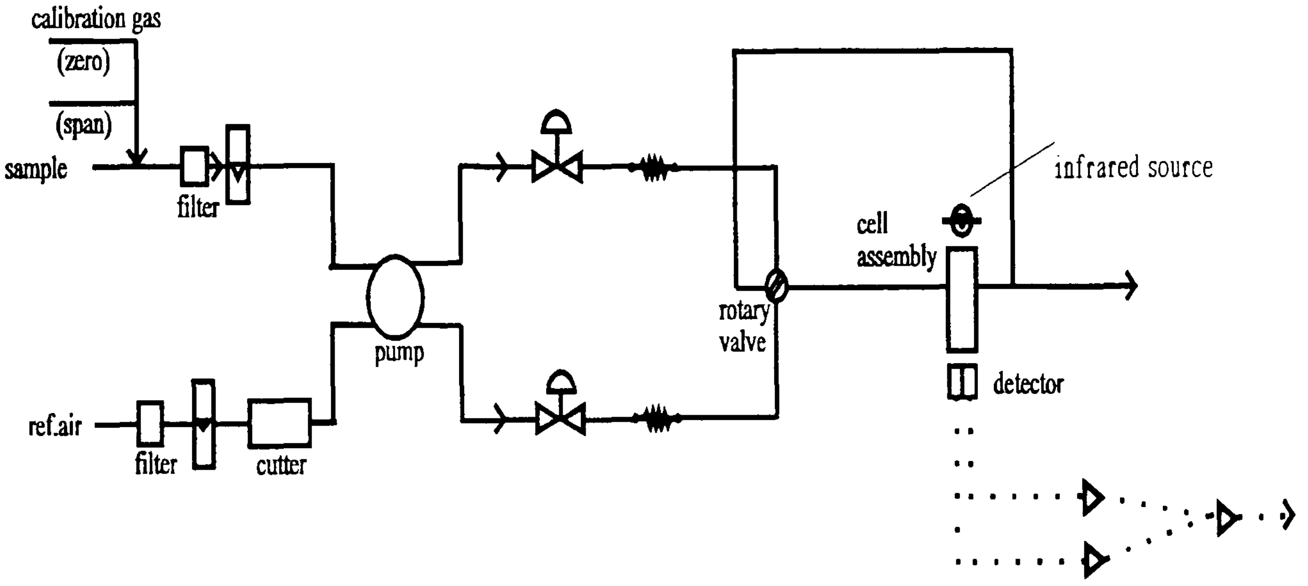


Figure 3.2 Schematic diagram of a cross-flow modulated non-dispersive infra-red analyser, Horiba Instruments model APMA-350E. *(Horiba 1992)*.

Inside the reaction chamber, NO reacts with ozone to form excited NO_2 and the emitted chemiluminescence is measured through an optical filter by a sensitive solid detector. The output of the detector is linearly proportional to the NO concentration when flow line 1, which underwent no conversion, is operational, and the NO_x concentration when flow line 2 is operational. The NO_2 concentration is determined by an electronic subtraction circuit that subtracts the NO signal from the NO_x signal.

The continuous monitor has four selectable full scale ranges from 0.1 to 10 ppm. It has a lower detectable limit of 0.002 ppm, with a response time (T_{90}) of within three minutes for the 0.1 ppm range and two minutes for the other ranges *(Horiba Instruments 1992)*. Zero drift is specified as within ± 0.002 ppm/day and span drift as within ± 2% full scale measurement/day which equates to ± 0.016 ppm as calibration was undertaken with a 0.8 ppm standard. Noise is given as within ± 0.0005 ppm and interference as within ± 0.008 ppm.

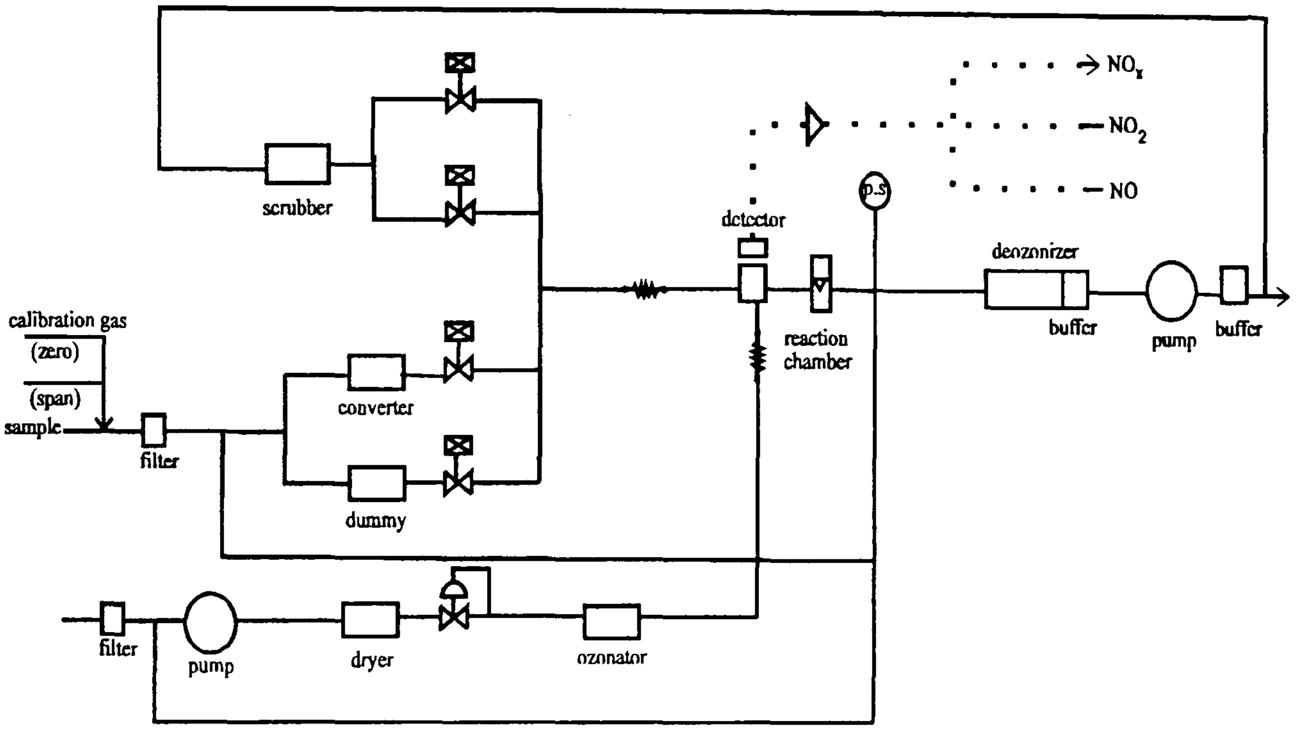


Figure 3.3 Schematic diagram of a chemiluminescent NO_x gas analyser, Horiba Instruments model APNA-350E *(Horiba 1992)*.

3.2.3 Data acquisition

Data were recorded using a Campbell Scientific 21X Micrologger *(Figure 3.4)*. On the panel, the 16 digit keyboard is used to enter programs and commands. The 9-pin serial I/O connector provides connection to data storage peripherals or telecommunication interfaces for serial data transfer or remote programming. Data and programme information can be viewed on the 8 digit LCD display.

The micrologger can accept up to 8 differential measurements or 16 single ended measurements, using one differential channel for two single ended channels. The micrologger is a real time system that has a maximum programme execution rate of 80 programme tables per second which equates to a typical throughput rate of one measurement, with linear scaling and transfer to tape, every second *(Campbell Scientific 1984)*. Data throughput is the rate at which a signal can be measured, processed and stored in final memory. The rate is reduced by additional processing or when data is transferred to tape or through the 21X serial port.

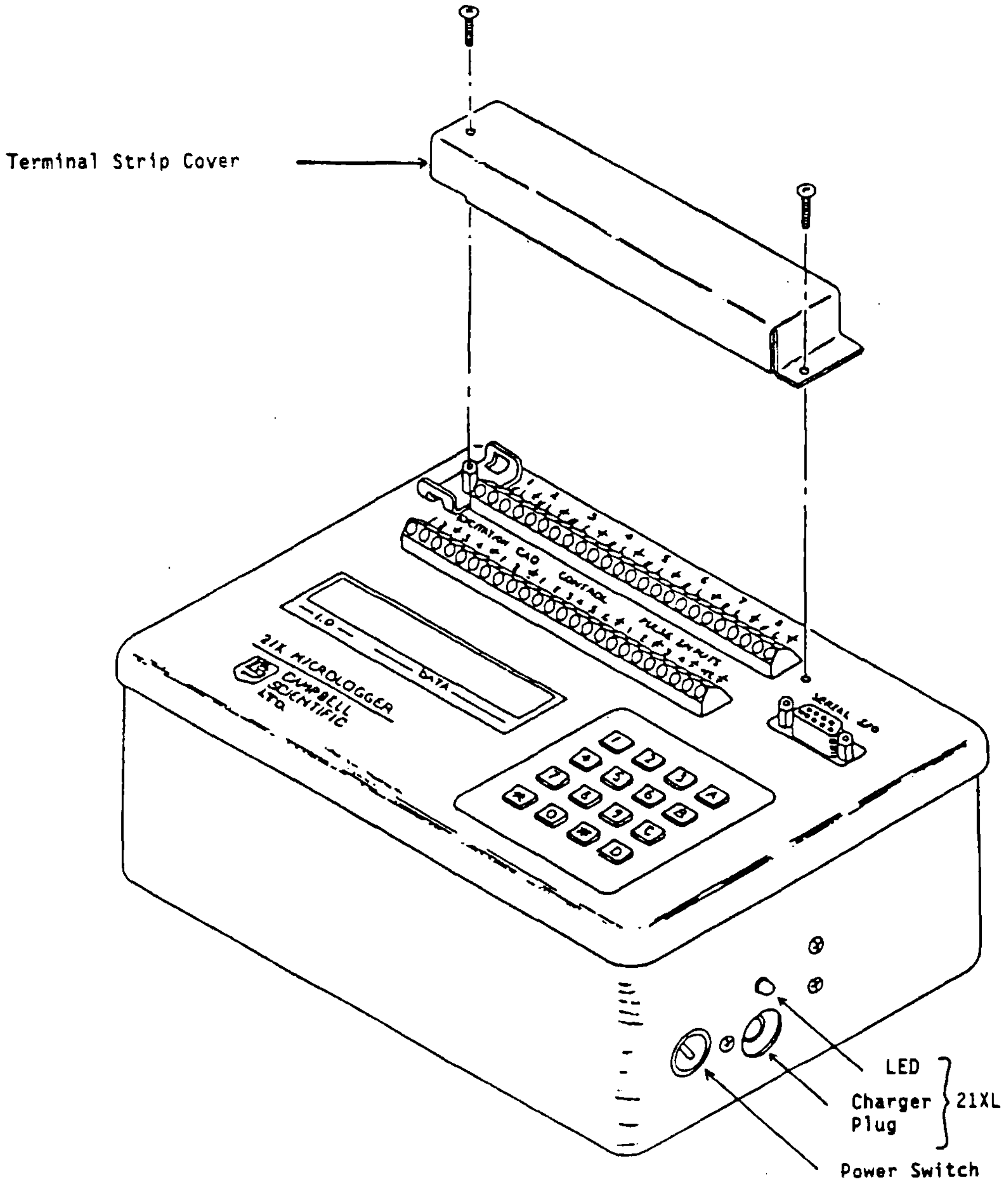


Figure 3.4 View of the operating functions and connection ports on the 21X Micrologger *(Campbell Scientific 1984)*.

The data logging program *(Tables 3.1 A, B, C, D)* sets the instructions to record the output voltages from the ambient air monitors every second. The voltages are then converted to the appropriate units. Further to this the logging program calculates 5 minute, 10 minute, 15 minute and hourly averages. The information was then downloaded to a personal computer using the 9 pin D-type port of the micrologger.

The memory of a standard 21X micrologger is 16K ROM and 40K RAM which allows the storage of 19,328 low resolution data points. This proved to be insufficient and to overcome this problem the memory was expanded to 24K ROM using storage module SM 716.

Table 3.1A Logger program with instructions

```
*1A   0A    1600A
01A   10A   *0
1A    10A
4A    P71A
5A    4A
1A    1A
1A    P77A
1A    11A
0A    P92A
P86A  0A
10A   60A
P71A  10A
4A    P71A
1A    4A
P77A  1A
11A   P77A
P92A  11A
0A    *0
5A    P71A
10A   *4
71A   01A
4A    02A
1A    *0
P77A  *5A
11A   95A
P92A  31A
```

Table 3.1B Key definition summary *(Campbell Scientific 1984).*

Key	Action
0-9	Enter numeric data
*	Enter mode
A	Advance through a program table or data storage, or enter the displayed number into memory
B	Back-up through a program table of data storage
C	Change sign of a floating point number or change 4 digit input location number to increment on loop
D	Enter decimal point of a floating point number
#	Clear right most digit
#D	Delete entire instruction
#A	Advance to next instruction or next final data storage array
#B	Back-up to previous instruction or previous final data storage array

Table 3.1C Key mode summary *(Campbell Scientific 1984).*

Key	Mode
*0	LOG data and indicate active tables
*1	Program table 1
*2	Program table 2
*3	Subroutine program table 3
*4	Enable/disable tape and/or printer output
*5	Display real time, set clock
*6	Display/alter input storage data, toggle flags
*7	Display final storage data
*8	Final storage dump to cassette tape
*9	Final storage dump to printer
*A	Memory allocation
*B	Signature test

Table 3.1D Key instruction summary *(Campbell Scientific 1984).*

Key	Instruction
P71	Stores the average value over the given output interval for each input location value specified.
P77	Stores the current time in final data storage.
P86	Unconditionally executes the specified command. Sets the output flag.
P92	If Time instruction.

3.3 Review of urban air quality

3.3.1 Introduction

Urban air pollutants arise from a wide variety of sources, and can be classified as either primary pollutants or secondary pollutants. Primary pollutants are emitted directly to the atmosphere whereas secondary pollutants are formed as a result of reactions between primary pollutants and, possibly, other secondary pollutants. Emissions of different pollutants are estimated from a knowledge of the processes that form them - the main process in urban areas being combustion - and are called emission inventories. The UK emission inventories for all pollutants are under continuous review as new methods, factors and statistics become available to improve the accuracy of the inventories. The data in *Table 3.2*, calculated from emission inventories produced by Warren Spring Laboratory, shows the relative importance of different pollutants in the United Kingdom in 1993.

In most parts of the United Kingdom the dominant pollutants arise from the motor vehicle although in certain areas, for instance Belfast, where coal is continuing to be used in both a domestic and commercial environment, this is not necessarily the case. However, at a national level, road transport is the single most important source of the pollutants. Moreover, in urban zones the contribution can be expected to be even higher. This is because there is typically more traffic and less industry. The emissions from traffic, being at a lower height, also have a greater impact on local air quality than do emissions from industrial stacks.

Table 3.2 Sources of the principal gaseous air pollutants - 1993. Emissions from power stations have been allocated, on an approximate basis, to the various categories according to their use of electricity generated *(DoE 1995)*.

Source	% of Total Emissions				
	Sulphur Dioxide	Black Smoke	Nitrogen Oxides	Carbon Monoxide	Volatile Organic Compounds
Road Transport	4	52	51	91	45
Other Transport	4	1	8	1	2
Commercial/ public service	9	1	4	-	-
Industry	41	5	16	1	1
Agriculture	1	-	1	-	-
Miscellaneous	12	1	5	-	-
Exports	1	-	1	-	1
Other	1	9	1	1	49
Domestic	27	30	12	5	2
Total (kT)	3188	444	2347	5641	2418

The major pollutants associated with road transport and their contributions to the total emissions in the UK and London are shown in *Figure 3.5 (Revitt 1995)*. Road transport was clearly a more important source of sulphur dioxide, black smoke, nitrogen oxides and volatile organic compounds (VOCs) in Greater London than in the UK as a whole.

In determining the contribution from traffic, the emissions from approximately 25 million vehicles (the current UK vehicle population) need to be considered. The difficulty in determining vehicular contributions is further compounded because vehicle emissions are dependent upon several factors: the fuel type used in the vehicle *(Table*

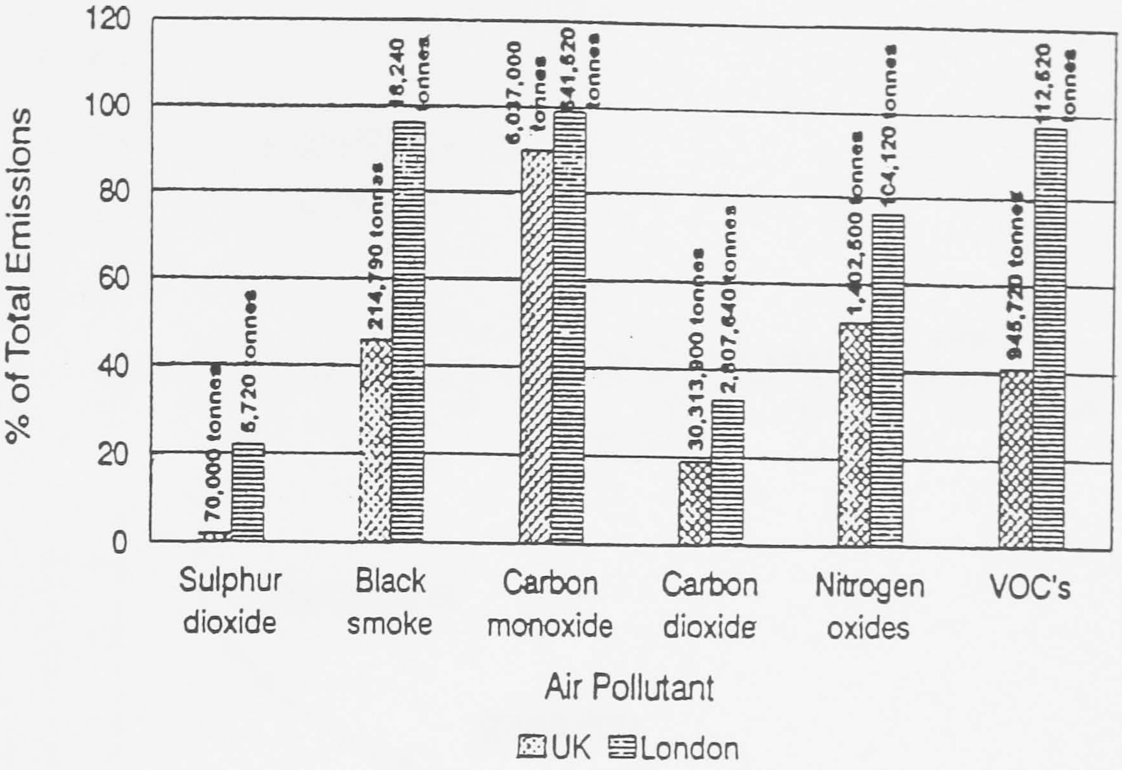


Figure 3.5 Road transport contribution to air pollutant emissions in the UK and London *(Revitt 1995)*.

3.3); whether the vehicle has engine/pollution control; how the vehicle is maintained; the driver's behaviour; traffic conditions; vehicle speeds and the number and mileage of different types of vehicles.

Table 3.3 A comparison of diesel and petrol exhaust emissions *(Acres 1991)*.

Fuel	Carbon monoxide (%)	Hydrocarbons (ppm)	Nitric oxide (ppm)	Sulphur dioxide (ppm)	Particulates (g/m^3)
Diesel Engine	0.1	300	4000	200	0.5
Petrol Engine	10.0	1000	4000	60	0.01

The effect of vehicle speeds upon emissions of various pollutants can be dramatic. *Figure 3.6* shows the effect of speed on emissions of carbon monoxide, nitrogen oxides and volatile organic compounds from petrol fuelled cars with and without catalytic

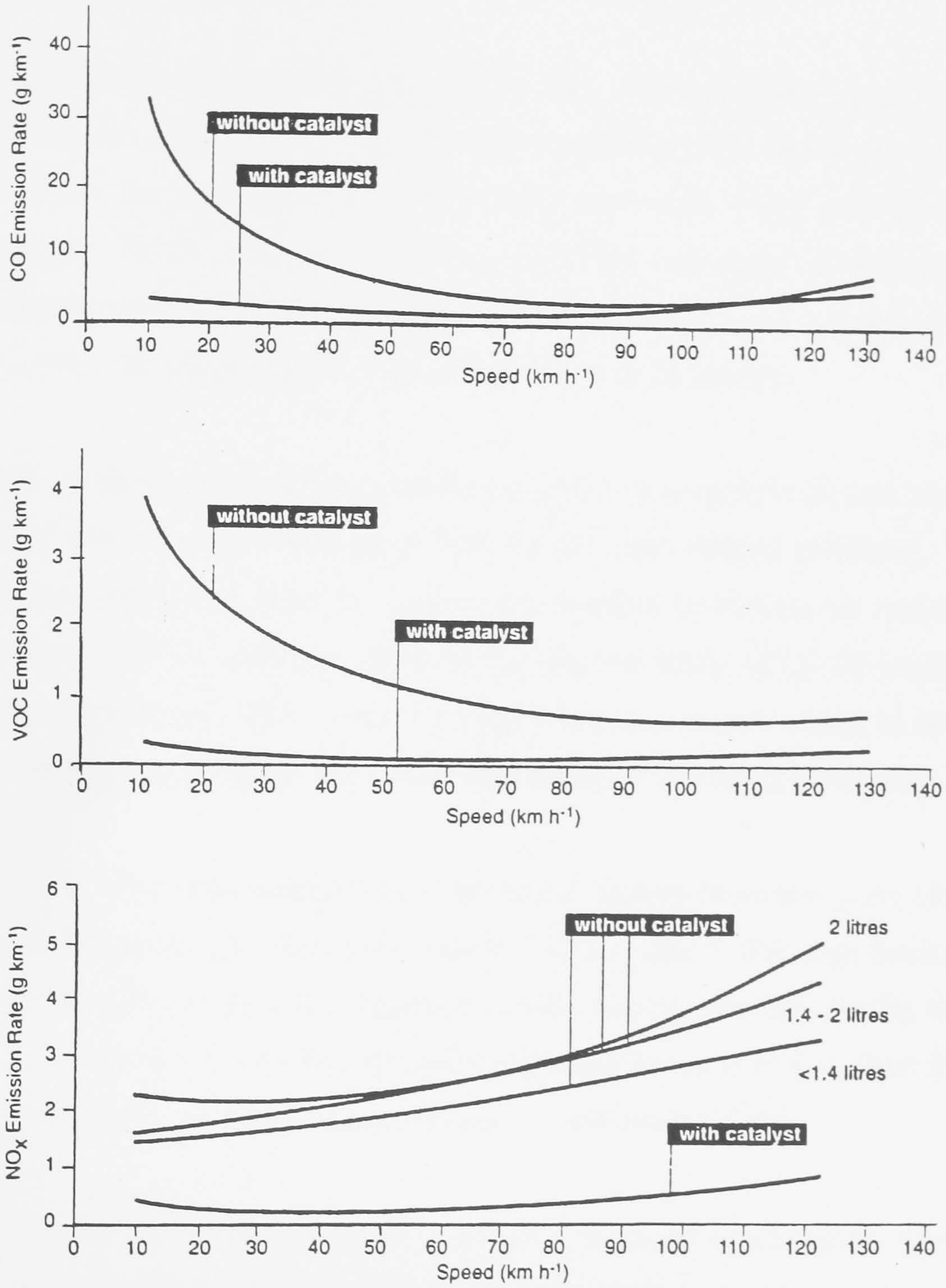


Figure 3.6 Speed dependent emission rates for CO, VOC and NO_x for petrol engined vehicles of less than 2.5 tonnes *(Eggleston et al. 1991).*

converters. For cars fitted with catalytic converters the reduction in emissions is significant, with up to a 90% reduction in emissions over those cars without a catalyst. However, CO emissions are not significantly decreased by the fitting of catalysts when vehicles are driven at high speeds and may even suffer a small increase in emissions. Those vehicles without catalysts show a decrease in carbon monoxide and VOC emissions with increasing speed although this is less clear with catalyst fitted cars. The emissions of CO and VOC are highest at the slow speeds associated with urban driving.

For nitrogen oxides the opposite generally occurs, with emissions lowest at the slow speeds associated with urban driving. However, large amounts of this pollutant are also emitted in the stop-start conditions often associated with urban driving. However, emissions per vehicle peak on motorways with the associated higher speeds. The relationship between speed and nitrogen oxides is less exaggerated in catalyst equipped cars for which the emissions are consistently lower at all speeds.

The speed emission curves do not include the effect on emissions of cold starts which for catalyst cars can contribute up to 50% of the total emitted pollutants. Emission reduction can only occur when the catalyst has reached its operational temperature of approximately 300°C, therefore, prior to the catalyst being lit-up the emissions are essentially uncontrolled. Thus, when the engine is cold, emissions may be as much as an order of magnitude higher than when the catalyst is operating efficiently.

UK emissions from road transport have increased rapidly in recent years (*Table 3.4*) despite the introduction of emissions controls. This is due to the large increase in the number of vehicles on the road, especially within the last decade. During the period 1980 to 1993 the UK's vehicle population increased by 31% to just over 25 million vehicles *(DoT 1995)* of which approximately 21 million were cars.

For most of the significant pollutants, emissions increased more rapidly than vehicle numbers between 1980 and 1993. This may be explained by a combination of factors, but particularly the increasing average distance travelled by vehicles, with vehicle usage going up by 70% from 1980-1993 to 422 billion kilometres *(DoT 1995)*. The large increase in nitrogen oxides can be further explained by the trade off between emission controls for different pollutants. The continued reduction of emission limits for other pollutants has lead to an increase in emissions of nitrogen oxides. The additional steep growth in black smoke emissions can be explained by the growth in the diesel powered component of the fleet as more people choose to buy diesel powered vehicles.

Table 3.4 The increase in UK estimated emissions from road transport (1980-1993) *(DoE 1995)*

Pollutant	% Increase
Black smoke	93
Nitrogen oxides	35
Sulphur dioxide	40
Carbon monoxide	30
Volatile organic compounds	9

Particles as measured in the air are defined by the method of measurement. Until recently they were measured in the UK by the 'Black Smoke' method and hence the reference in the text to black smoke emissions. The black smoke method is when air is drawn through a filter paper and the blackness of the stain measured. However, although this technique gives a good indication of the concentration of particles produced by coal burning it is not good at quantifying the particles produced by motor vehicles. Therefore, increasingly particles are being measured by a method that determines the mass of that fraction most likely to be deposited in the lung and cause ill health. These particles are called PM_{10} (Particulate Matter less than 10 microns in diameter). However, even this measure may be too big and it may be that $PM_{2.5}$ will be a more appropriate measure for the future *(EPAQS; Particles 1995)*.

3.3.2 Carbon monoxide

Carbon monoxide (CO) is produced by the incomplete combustion of fossil fuels. Incomplete combustion occurs when an 'engine' (combustion system) is not working at its stoichiometric air:fuel ratio. When an engine is operating at a stoichiometric air:fuel ratio, the CO emissions will normally be at a minimum. Incomplete combustion may occur when an engine system operates outside this ideal ratio, and can be said to be running rich or lean. In these conditions CO emissions can vary quite markedly. Incomplete combustion can occur in a mis-tuned motor vehicle or when starting a boiler from cold. It is apparent therefore, that CO emissions are largely controlled by the combustion conditions and therefore, reflect the state of wear or any modifications performed upon an engine.

3.3.2.1 Sources of carbon monoxide

The most significant natural source of carbon monoxide is probably the oxidation of atmospheric methane, with an estimate by *Tie et al. 1992* that 500 MTonnes are produced annually. Less significant natural inputs include forest fires, terpene oxidation and oceanic sources. Estimates for anthropogenic sources vary from 350 to 600 MTonnes per annum with the motor vehicle being the predominant source, specifically petrol engined vehicles *(Gilham et al. 1992)*. Although on a global scale natural inputs can outweigh anthropogenic inputs, their influence upon urban air quality is marginal *(NAS/NRC 1977; Crutzen & Gidel 1988; Cicerone 1988; Tie et al. 1992)*.

The increasing volume of traffic upon our highways has seen a subsequent rise in the importance of motor vehicles as a CO source, coupled with the decrease in domestic coal burning *(Figure 3.7)*. In 1970 coal accounted for 25% of the UK national CO emissions but by 1992 this had decreased to 4%. The trend is reversed for motor vehicles with 87% of emissions coming from road transport in 1992 compared to only 61% from the same source in 1970.

However, emissions from petrol engined vehicles can be expected to decline in the future. The newer engine designs that incorporate fuel injection and engine management systems should reduce emissions because the engine is more often working at stoichiometry. Older engine systems, that use carburettor based fuelling systems and do not have engine management systems, are less efficient in this regard. In addition, they may be poorly adjusted giving rise to quite large emissions, and are therefore being phased out. However, the biggest decline in emissions will come from the application of the EU Directive on exhaust emissions that came into force on the 1st of January 1993. This practically mandated the fitting of three way catalysts (TWC) to all new petrol engined vehicles, effectively lowering emissions by at least 5 to 10 times.

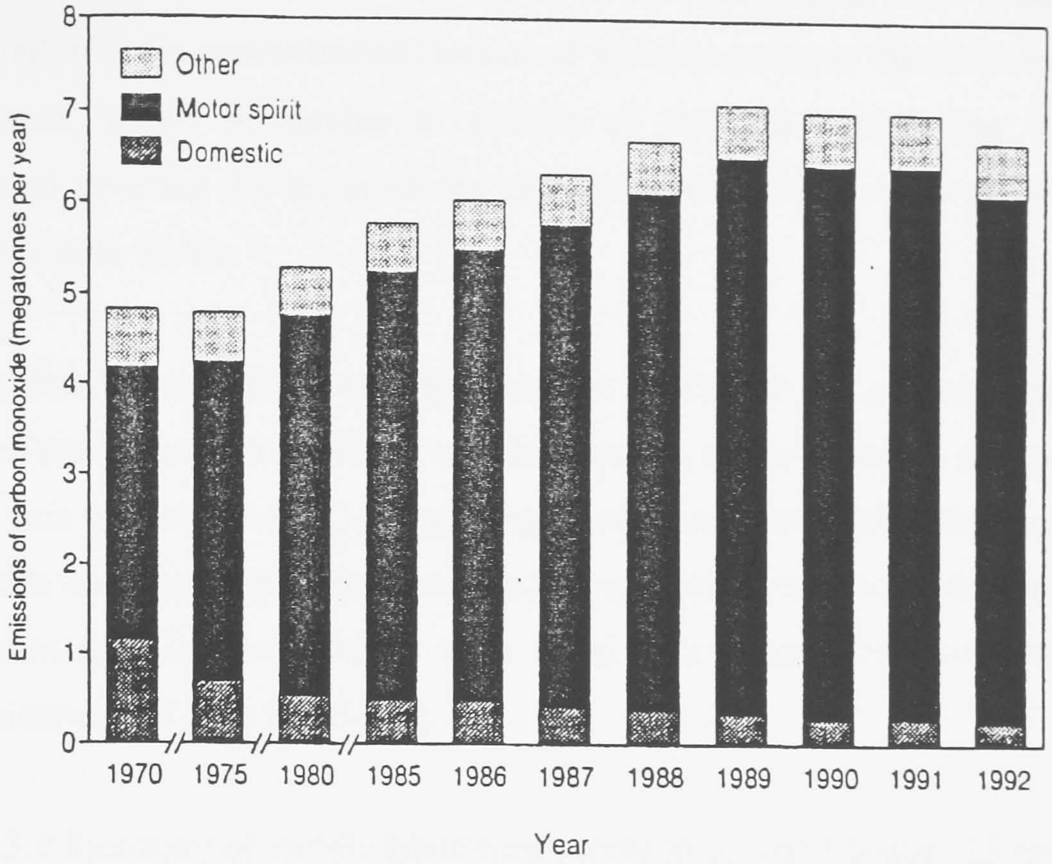


Figure 3.7 UK emissions of carbon monoxide (1970-1992) *(EPAQS; CO 1994).*

3.3.2.2 Roadside ambient air concentrations

Little work has been carried out with regard to roadside measurement of CO concentrations. There is insufficient data on the levels of CO near roads and how these levels change with distance away from the road. Work conducted by *London Scientific Services 1990* provided results for one busy roadside site in London (3 metres from the kerb) which showed annual mean concentrations to be 2-3 times higher than local background concentrations. Similar work conducted by Friends of the Earth, given as evidence to the *House of Commons Transport Committee (1994)*, claimed CO concentrations at kerbside were up to double the levels recorded 10 feet from the pavement edge.

This is as expected as carbon monoxide is a primary pollutant for which motor vehicles are the dominant source. Carbon monoxide gradients are therefore, likely to be very marked near to roads. Mobile laboratory monitoring results demonstrate this clearly although few studies of this kind have been undertaken in the UK. One such study was undertaken in London Victoria by *Broughton (1988)* and involved a mobile laboratory equipped for measurement of NO_2, NO_x, SO_2, O_3 and CO being driven within a moving stream of traffic between 15 and 24 October 1985 *(Table 3.5)*. These data were compared with the concentrations measured simultaneously at the fixed monitoring site at Victoria Street. A number of circuits of the area surrounding Victoria were performed covering the major routes together with adjacent minor roads where there was very little traffic.

The results showed a relatively high gradient in NO and CO concentrations (30-40%) between the major thoroughfares and the adjacent minor roads. A shallower gradient (60%) was recorded for NO_2 suggesting a more uniform spatial distribution. Carbon monoxide concentrations were measured along minor roads towards major roads and significant rises in concentration were noted with a maximum concentration rise of approximately 10 ppm to 20 ppm.

Table 3.5 Summary of mobile laboratory survey in central London, 15 and 24 October 1985 *(modified after Broughton 1988)*.

Road Section	NO			NO_2			CO		
	Mean (ppb)	Rank	% relative to highest	Mean (ppb)	Rank	% relative to highest	Mean (ppm)	Rank	% relative to highest
Vauxhall Bridge Rd	315	4	63	84	3	90	6.5	3	66
Vauxhall Bridge Rd north	312	5	62	81	5	87	4.6	5	47
Hyde Park Corner Roundabout	501	1	100	93	1	100	9.8	1	100
Grosvenor Place	498	2	99	90	2	97	8.6	2	88
The minor roads	169	6	34	59	6	63	4.2	6	43
Victoria Street	409	3	82	82	4	88	6.1	4	62

Work undertaken by *Derwent et al. (1995)* presented CO air quality data, over the period July 1991 - June 1992, from a location 5 metres from the kerbside, adjacent to Exhibition Road, in central London. The data showed seasonal variation with mean quarterly weekday concentrations of 1.4, 2.1, 1.7 and 1.0 ppm for quarters 3, 4, 1 and 2 (Q3 = July 1991 - September 1991; Q4 = October 1991 - December 1991; Q1 = January 1992 - March 1992; Q2 = April 1992 - June 1992), respectively. This is typical of primary pollutants in urban areas with the highest concentrations observed during the winter months and the lowest concentrations in the summer months.

3.3.2.3 Air quality legislation

There are no limit or guide values specified by the European Union for CO in ambient air. The Expert Panel on Air Quality Standards *(EPAQS; carbon monoxide 1994)* has recommended an air quality standard for the UK of 10 ppm in 1994, measured as a running 8-hour average. The WHO (World Health Organisation) guidelines are considered in this thesis. The WHO guidelines were established with the aim of 'providing a basis for protecting human health' *(WHO 1987)*.

The WHO recommended a maximum carboxyhaemoglobin (COHb) level of 2.5-3.0% in blood derived from the Coburn equation. To fulfil this commitment the WHO have set the following time weighted exposure guidelines:

- 87 ppm over 15 minutes;
- 50 ppm over 30 minutes;
- 25 ppm over 1-hour;
- 10 ppm over 8-hours.

The basis for the guidelines was attained from the best available scientific knowledge on the health implications of CO exposure, that looked at cardiovascular, neurobehavioural, fibrinolysis and perinatal effects. The guidelines were then set to protect the more vulnerable parts of society, such as the elderly, pregnant women and people suffering from cardiac or respiratory difficulties.

Measured exceedences of the 15 minute, 30 minute or 1-hour guidelines are generally confined to pre 1980. However, the guide values have been approached in recent years, with an hourly peak of 24.8 ppm being recorded at Cromwell Road. These high values are strongly associated with winter stagnation/inversion episodes, when wind speeds are low and there is a stable atmospheric boundary layer. These conditions combine to make it difficult for pollutants to disperse. An example of such an episode occurred in December 1991 when CO hourly concentrations reached 18.0 ppm at Earls Court, London, 16.7 ppm kerbside at Cromwell Road, London and 15.2 ppm in Glasgow *(QUARG 1993)*.

The 8-hour guideline has been frequently exceeded in the past *(Table 3.6)* at both kerbside and urban background sites. Sites operated by Warren Spring Laboratory had one or more exceedences for the 8-hour average at 45% (9 out of 20) of urban sites and 71% (5 out of 7) of kerbside sites. These 8-hour exceedences are also strongly related to winter inversions. However, in more recent years, 1992 and 1993, there has only been one exceedence recorded, in Manchester.

3.3.2.4 Health effects

Carbon monoxide, by combining with haemoglobin to form carboxyhaemoglobin, interferes with the supply of oxygen to cells. Carbon monoxide has an affinity for haemoglobin approximately 250 times higher than that of oxygen and therefore, inhibits the formation of oxyhaemoglobin and so decreases the capacity of blood to transport oxygen from the lungs to the tissues. Those tissues most sensitive to hypoxia (the central nervous system and the cardiovascular system) are of most concern to high exposures to CO.

If the carboxyhaemoglobin concentrations in the blood exceed 5%, then statistically significant impairment of vigilance tasks can be measured. The central nervous system also has the possibility of reduction in performance at carboxyhaemoglobin concentrations of below 5% *(WHO, 1987)*. Therefore, to provide for this, WHO recommended COHb levels of between 2.5 and 3.0% in blood. The effects upon the cardiovascular system are experienced most by people who have impaired heart or lung

Table 3.6 CO concentrations, and exceedences of the WHO 8h guideline, at suburban and urban sites *(Broughton et al. 1991b, 1991c and 1992).*

Site & CO parameter	CO concentration (mg m^{-3})		
	1988	1989	1990
Stevenage (suburban)			
Annual average		0.6	0.8
Max 1h average		5.5	6.4
Max 8h average		3.3	2.1
No of 8h averages > 10 mg^{-3}		nil	nil
Central London (urban)			
Annual average	1.1	1.3	1.3
Max 1h average	15.1	20.9	11.7
Max 8h average	6.0	16.2	6.2
No of 8h averages > 10 mg^{-3}	one	one	nil
Cromwell Rd, London (kerbside)			
Annual average		3.6	2.9
Max 1h average		24.8	18.4
Max 8h average		18.1	11.2
No of 8h averages > 10 mg^{-3}		eight	three
West London (urban)			
Annual average			1.5
Max 1h average			13.2
Max 8h average			8.8
No of 8h averages > 10 mg^{-3}			one
Glasgow (urban)			
Annual average		1.6	1.1
Max 1h average		19.2	17.0
Max 8h average		11.6	7.8
No of 8h averages > 10 mg^{-3}		three	one

function. Utilisation of available oxygen by the heart muscle is near to maximum even at rest, and an increased demand for oxygen can only be met by increasing coronary blood flow. Therefore any person whose capacity to increase blood flow to the heart is reduced, can find even small decreases in available oxygen harmful. Two studies *(Anderson et al. 1973; Aronow & Isbell, 1973)* have shown that angina is aggravated by carboxyhaemoglobin concentrations of 3% and it has also been shown that exposure to carbon monoxide can aggravate peripheral arterial as well as coronary heart disease *(Aronow et al. 1974).*

Carbon monoxide may also influence foetal development. The incidence, for women who smoked during pregnancy, of perinatal deaths is higher and those babies born are of smaller birth weight. There could be several mechanisms at work as tobacco smoke contains many toxins but hypoxia caused by carbon monoxide is one possible mechanism. Animal experiments *(Longo 1976)* support this with a dose-related increase in the incidence of stillbirths and deaths among the newly born, down to carboxyhaemoglobin levels of about 10%.

3.3.3 Nitrogen oxides

The most significant nitrogen compounds related to road transport are nitric oxide (NO) and nitrogen dioxide (NO_2), known collectively as NO_x; other oxides of nitrogen, such as N_2O, N_2O_3 and N_2O_4, do not contribute significantly to urban air pollution. However, there are other compounds of significance, which may play a role in pollution episodes and which take several days to form. These include dinitrogen pentoxide (N_2O_5), nitrogen trioxide (NO_3), nitric acid (HNO_3) and nitrous acid (HNO_2). Organic nitrates also play a role in such episodes, most notably peroxyacetyl nitrate. Nitrogen compounds are also contributors to the wet and dry deposition of acidic compounds to vegetation and buildings.

Although NO has relatively innocuous health implications its relationship with NO_2 makes it important in this respect. NO_2 is primarily formed by the oxidation of NO in the atmosphere. The rate of conversion, and the proportion of NO converted to NO_2, is largely dependent on the supply of oxidants, such as ozone, and the original concentration of NO present. Although the primary oxidant is ozone (Equation 3.3), there are less well understood conditions, under which other molecules act as the oxidising agent. One other possible oxidising agent is molecular oxygen (Equation 3.4). This reaction was thought to be too slow to play a role in urban air quality, however, if NO concentrations exceed 500 pbb, it can play a significant role in generating NO_2.

$$NO + O_3 \rightarrow NO_2 + O_2 \qquad \text{Eq. 3.3}$$

$$NO + O_2 \rightarrow 2NO_2 \qquad \text{Eq. 3.4}$$

The reaction in equation 3.3 is quick and, provided the concentration of ozone is sufficient, approaches completion in approximately one minute. However, if the supply of oxidants becomes depleted, as can occur near large sources of NO, such as a busy highway, a considerable proportion can be left unoxidised. However, as the polluted air moves downwind and mixes with cleaner air and more oxidants (such as molecular oxygen (equation 3.4) or aldehydes) become available, the bulk of NO is converted to NO_2 *(Harrison 1994)*. The NO/NO_2 ratio is therefore significantly larger in urban areas than rural areas.

3.3.3.1 Sources of nitrogen oxides

The most significant natural sources of NO_x are volcanoes, lightning and bacterial action. These natural sources far outweigh anthropogenic inputs on a global scale but lag behind man-made inputs in the UK. The chief anthropogenic sources are from the combustion of fossil fuels both in power generation and in the internal combustion engine.

Nitrogen oxides are primarily formed at high temperatures during combustion processes from the oxidation of nitrogen in the air and less importantly any nitrogenous elements in the fuel. The proportion of NO_2 produced is difficult to accurately determine in motor vehicle exhaust emissions and is often assumed to be approximately 5% of total NO_x. Minor contributions to the atmosphere come from non-combustion industrial sources, such as the manufacture of nitric acid, the use of explosives and welding operations.

Estimated UK emissions of NO_x for the period 1970-93 are presented in *Table 3.7 (Gilham et al. 1992)*. The two principal sources are shown to be power generation and road transport. Power generation emissions have declined since 1970 while emissions from road transport have steadily grown. Emissions from road transport in 1970 were 0.61 MT and accounted for 27% of total emissions, whereas in 1990 road transport produced 1.38 MT which accounted for just over 50% of total emissions. As *Table 3.7* represents the position in the UK as a whole, it can be expected that road transport would have an even greater role in urban areas than predicted by the national average. This is clearly illustrated in *Figure 3.5*.

The European Union directive on exhaust emissions came into place on the 1 January 1993 and has effectively mandated the fitting of three way catalytic (TWC) converters to all new petrol engined vehicles. The TWC has the ability to reduce emissions, in comparison to a non-catalyst car, by up to ten times, even when deterioration and possible failure of catalysts are considered *(Gilham et al. 1992; Eggleston 1992)*. It has therefore been predicted, following the widespread fitting of catalysts, that NO_x emissions from the motor vehicle will have fallen by a factor of two in the period 1990-2010 *(Eggleston 1992)*.

3.3.3.2 Roadside ambient air concentrations

NO and NO_2 concentrations have been continuously monitored at two kerbside sites in London. The adjacent highways carry between 50,000 and 60,000 vehicles daily. The sites were located at Cromwell Road - operated by Warren Spring Laboratory - and a central London location - operated formerly by London Scientific Services. The Cromwell Road site, half a metre from the kerb, has recorded the highest concentrations of NO and NO_2 in the UK, peaking in 1983 at 2,200 ppb for NO and 1,817 ppb for NO_2 *(QUARG 1993)*. The central London site, situated 3m from the kerb, followed a similar pattern *(Table 3.8)*.

Table 3.7 Summary of estimated NO_x emissions between 1970 & 1993, MTonnes year^{-1} (*Gilham et al. 1992*).

	’70	’75	’80	’85	’89	’90	’91	’92	’93
By Source									
Power stations	0.84	0.84	0.88	0.78	0.77	0.78	0.67	0.66	0.57
Other industry	0.49	0.39	0.32	0.28	0.27	0.27	0.32	0.32	0.33
Road transport	0.61	0.69	0.80	0.97	1.38	1.38	1.32	1.25	1.14
Other	0.35	0.31	0.31	0.30	0.30	0.30	0.32	0.32	0.30
By Fuel									
Coal	0.98	0.83	0.91	0.77	0.79	0.79	0.73	0.71	0.59
Petrol and DERV	0.61	0.69	0.81	0.97	1.37	1.38	1.79	1.73	1.64
Other	0.70	0.71	0.59	0.59	0.56	0.56	0.11	0.11	0.11
TOTAL	2.29	2.23	2.31	2.33	2.72	2.73	2.63	2.54	2.35

The annual mean concentrations of NO and NO_2 at the Cromwell Road site show that concentrations of NO are over four times that of NO_2 *(QUARG 1993)*. This is because the primary emission from motor vehicles is NO and not NO_2. *Figure 3.8* further displays this point with the annual time series for 1991 showing NO concentrations to be greater than NO_2 on almost all occasions. If the data are then compared with those produced for the urban directive sites in Manchester and Glasgow *(Figures 3.9 and 3.10)* they show a similar scenario with the values for NO higher than for NO_2.

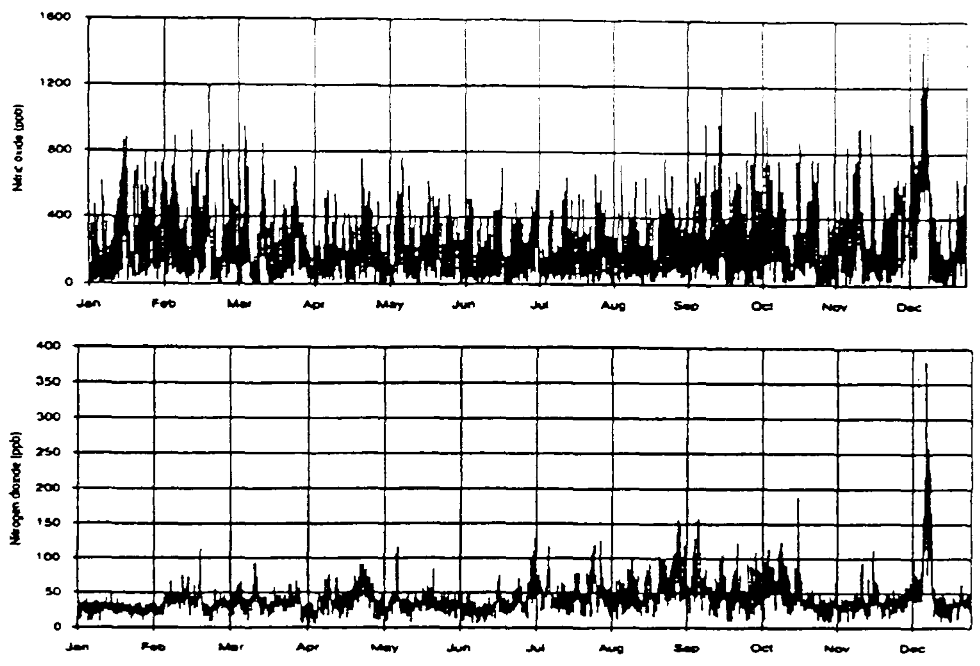


Figure 3.8 Time series of NO and NO_2 concentrations at Cromwell Road, London (1991) *(QUARG 1993).*

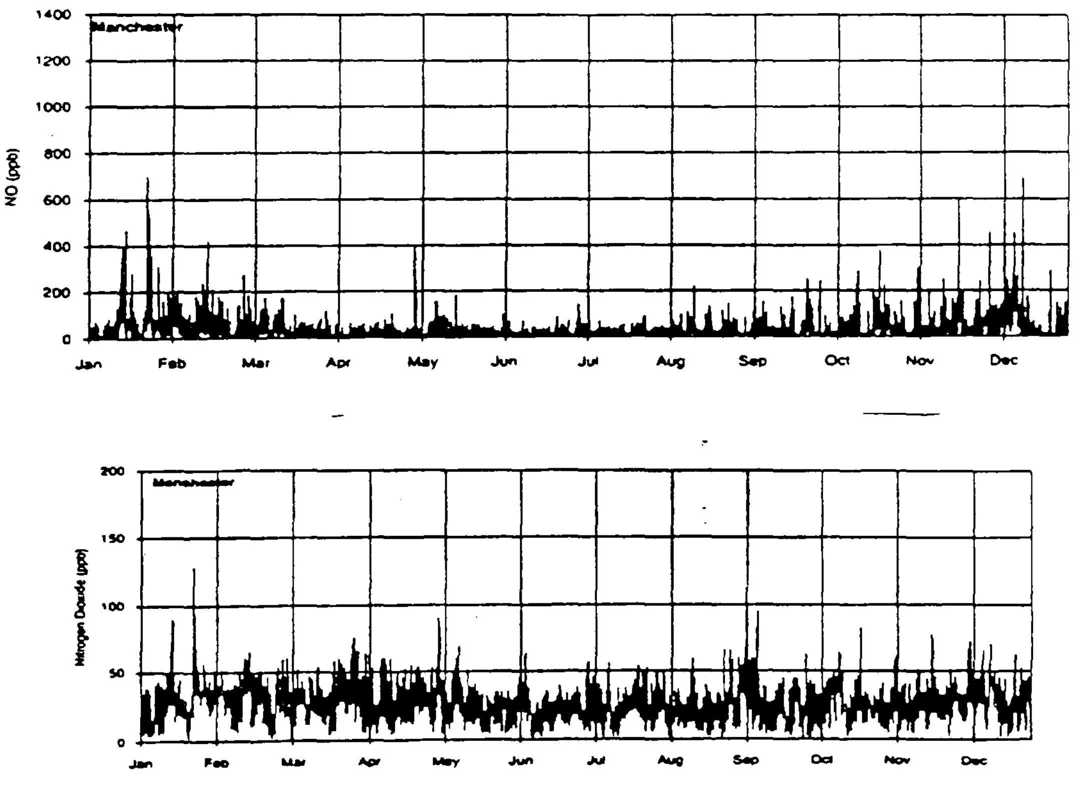


Figure 3.9 Time series of hourly NO and NO_2 concentrations at Manchester, 1991 *(QUARG 1993).*

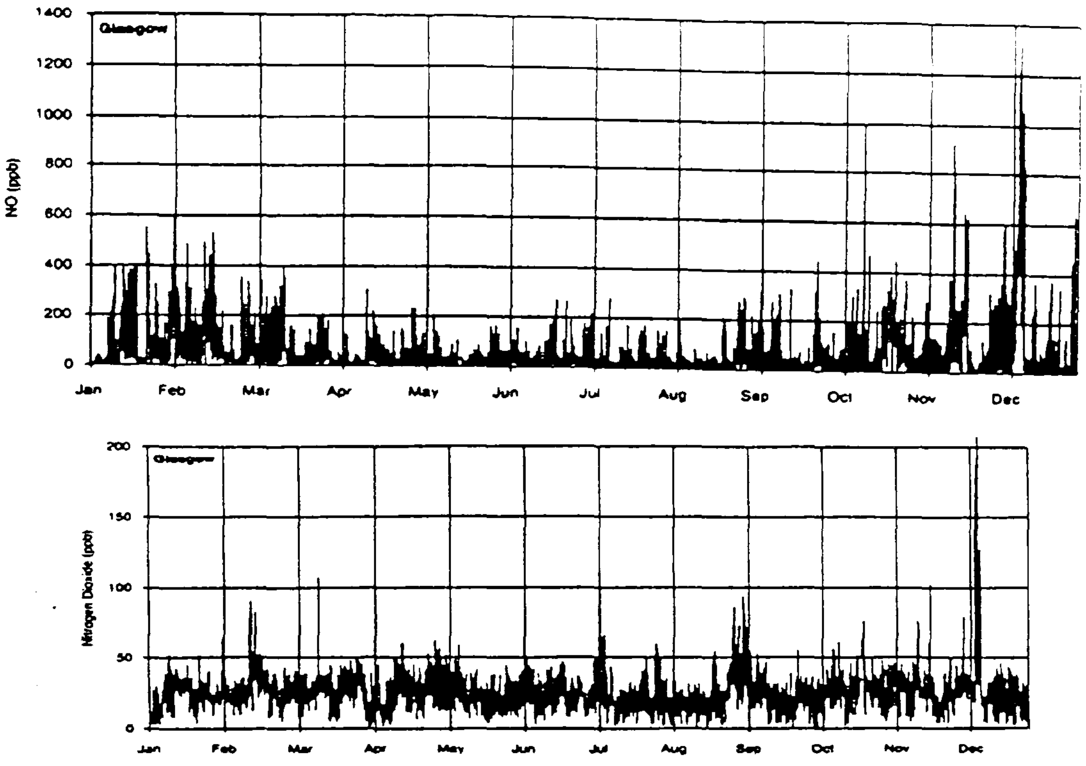


Figure 3.10 Time series of hourly NO and NO_2 concentrations at Glasgow, 1991 *(QUARG 1993)*.

Table 3.8 Peak hourly average NO and NO_2 concentrations (ppb) recorded at two roadside locations in London *(QUARG 1993)*.

Site	1986		1987		1988		1989		1990	
	NO	NO_2	NO	NO_2	NO	NO_2	NO	NO_2	NO	NO_2
Cromwell road	1126	195	-	-	-	-	1767	271	1336	245
Central London	-	468	-	264	-	345	-	212	-	-

A further study carried out using NO_2 diffusion tubes to derive concentrations, by the *Photochemical Oxidants Review Group (PORG 1990)*, found that for a road with traffic flows of 50,000 vehicles per day, the NO_2 concentrations varied as follows:

- 80-90 ppb in the centre of the road;
- 50-60 ppb at the kerb;
- 40-50 ppb at the back of the pavement, 3 metres from the kerb;
- 30-40 ppb in the local 'background environment'.

The review group also reported the presence of two maxima in the vertical concentration distribution of NO_2 in a central London street. A remote sensor was used to give the vertical distribution and found a maximum at ground level and a second above the height of the buildings. The first maximum is due to its proximity to the source of the emissions and the second is thought to have been caused by the additional oxidants present above the buildings.

Work undertaken by *Laxen and Noordally (1989)* near a heavily trafficked highway (traffic flows between 16,000 and 160,000) examined the decrease in NO_2 concentrations away from the road *(Figure 3.11)*. The study found that the NO_2 concentration became nearly indistinguishable from background levels at 20-30m from the kerb. However, the attempt to find a simple relationship between traffic flows and concentration was unsuccessful. The inability to find a simple relationship was attributed to the survey's incapacity to account for vehicle speed, engine mode, traffic flow or the limitation imposed by lack of ozone on NO_2 formation.

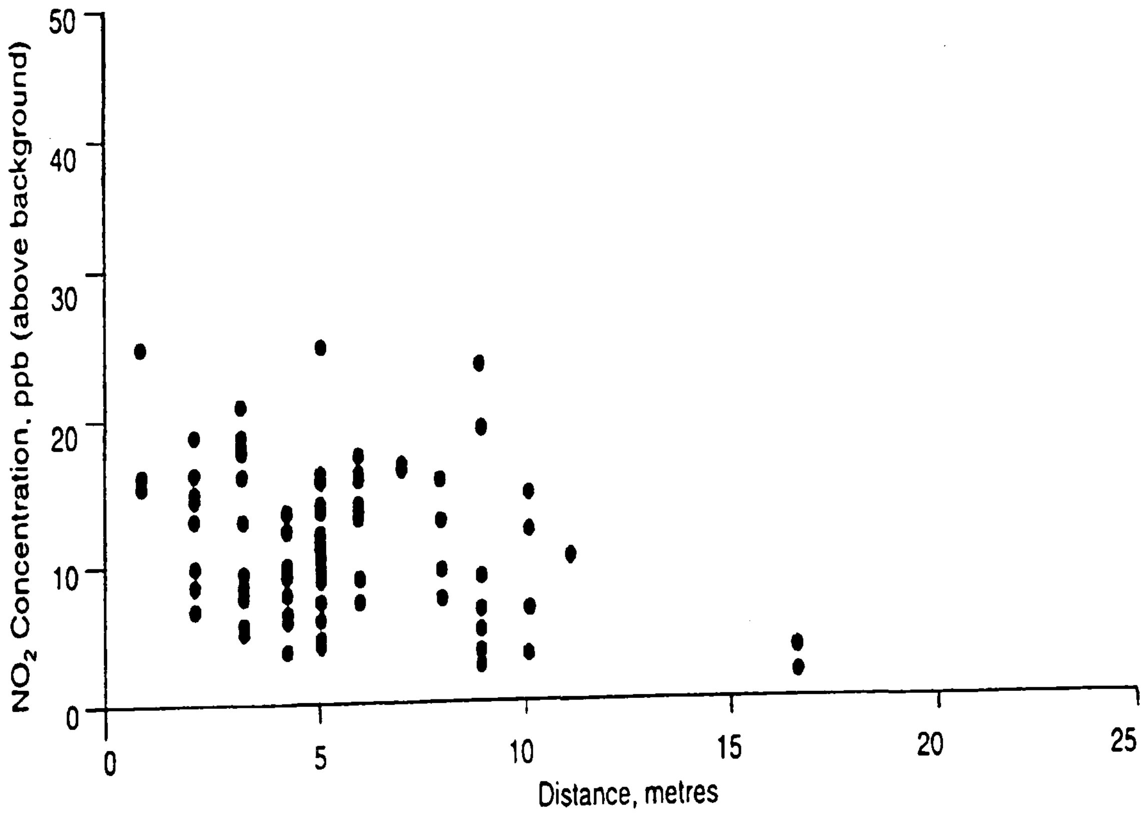


Figure 3.11 Variation of annual mean NO_2 concentrations (above background) with distance from kerb *(Laxen and Noordally 1989)*.

3.3.3.3 Air quality legislation

Limit and guide values *(Table 3.9)* have been specified by the European Union for NO_2 and incorporated into UK law by the Air Quality Standards Regulations *(DoE 1989)*. The limit value of 105 ppb (as a 98th percentile annual 1 hour average) is set at a level designed to protect human health and is binding on member states. If the limit values are exceeded, member states must then institute measures to reduce concentrations. Guide values are more restrictive than limit values but they are not mandatory. They have been set 'in order to improve the protection of human health and contribute to the long term protection of the environment' *(European Union 1985)*. They are set at 26 ppb, as a 50th percentile of hourly averages, and 71 ppb, as a 98th percentile of hourly averages. The World Health Organisation (WHO) have set their guidelines to take account of both acute and chronic effects *(WHO 1987)*. The one hour average guideline (400 $\mu g\ m^{-3}$) is based upon the lowest-observed-effect on asthmatics and although this is not necessarily deleterious to human health a guideline provides a further margin of protection. The 24-hour average guideline is based upon the premise that repeated exposure to the lowest-observed-effect level should be avoided to prevent chronic effects. The limit value is therefore set at 150 $\mu g\ m^{-3}$.

Table 3.9 Air quality limit values and guidelines for nitrogen dioxide.

Organisation	50th Percentile $\mu g\ m^{-3}$(ppb)	98th Percentile $\mu g\ m^{-3}$ (ppb)	24-hr Mean $\mu g\ m^{-3}$ (ppb)	1-hr Mean $\mu g\ m^{-3}$ (ppb)
Limit Value (EU)	-	200 (105)^B	-	-
Guide Value (EU)	50 (26)^A	135 (71)^B	-	-
World Health Organisation	-	-	150 (78)	400 (209)

^{A}guide value expressed as 50th percentile of hourly means over a year.

^{B}limit or guide value expressed as 98th percentile of hourly means over a year.

(To convert to ppb the following conversion factor, at standard atmospheric pressure and 20°C, is 1 $\mu g\ m^{-3} = 0.52$ ppb).

Measured exceedences of the EU directive limit value for NO_2 have been confined to London. These exceedences were monitored at Earls Court (1989), Cromwell Road (1989) and a central London roadside location of the London Borough's network (1988). The Earls Court exceedence was later shown to be due to excessive interference from local sources. Exceedence of EU guide values is much more frequent and has occurred in London for both the 50th percentile and the 98th percentile values in 1988, 1989, 1990, 1991, 1992 and 1993. Exceedence of EU guidelines occurs in other cities but is less frequent **(Table 3.10 A, B & C)**. The WHO guidelines, which are the most demanding, are exceeded even more frequently.

3.3.3.4 Health effects

Exposure to NO_2 in urban areas is thought to be most dangerous in periods of peak traffic flow. There is evidence that it is not the concentration and duration of exposure (integrated dose) that produces the greatest toxic impact, rather it is the peak concentrations that are most dangerous *(Hamilton & Duggan 1992)*.

Data from animals are only available for exposure/effect studies and therefore, those investigations showing the lowest-observed-effect in humans are used. The investigations have fallen broadly into two groups: epidemiological studies and controlled clinical exposure. The epidemiological studies provide some data concerning chronic effects that may result from long term exposure. Controlled clinical exposure provides data on the possible acute effects.

3.3.3.4.1 Chronic exposure

There are no epidemiological studies related to outdoor exposure from which any quantitative exposure/risk relationships can be drawn. Indoor exposure studies have shown that cooking with gas may have a very small effect - on either susceptibility to respiratory illness or lung function - especially with regard to children. However the

Table 3.10A Exceedences of limit values, guide values and guidelines for NO_2 in recent years *(modified after QUARG 1993).*

Site	Year	EC Directive			WHO	
		Limit (98)	Guide (98)	Guide (50)	Hourly	Daily
Central London, Victoria	1988	no	yes	yes	0	3
	1989	no	yes	yes	1	1
	1990	no	yes	yes	0	0
London, Bridge Place (A)	1990	no	yes	yes	0	5
	1991	no	yes	yes	23	9
	1992	no	yes	yes		7
	1993	no	yes	yes		
London, Cromwell Road (B)	1988	-	-	-	0	-
	1989	yes	yes	yes	11	15
	1990	no	yes	yes	0	12
	1991	yes	yes	yes	27	18
	1992	no	yes	yes	-	8
	1993	no	yes	yes	-	2
London, Earls Court (A)	1988	no	yes	yes	1	9
	1989	yes	yes	yes	25	9
	1990	no	yes	yes	0	4
	1991	no	yes	yes	16	5
Glasgow (A)	1988	no	no	yes	0	0
	1989	no	no	no	2	3
	1990	no	no	no	0	0
	1991	no	no	no	0	0

Table 3.10B Exceedences of limit values, guide values and guidelines for NO_2 in recent years *(modified after QUARG 1993).*

Site	Year	EC Directive			WHO	
		Limit (98)	Guide (98)	Guide (50)	Hourly	Daily
Manchester (A)	1988	no	no	no	8	5
	1989	no	no	no	2	1
	1990	no	yes	no	0	1
	1991	no	no	no	0	0
	1992	no	yes	yes	4	6
	1993	no	no	no	0	0
Walsall (A)	1988	no	no	no	0	0
	1989	no	no	yes	1	1
	1990	no	no	no	0	0
	1991	no	no	no	0	0
	1992	no	no	no	5	1
	1993	no	no	no	0	0
Birmingham (A)	1988	no	no	no	1	0
	1989	no	no	no	0	0
	1990	no	no	no	0	0
	1991	no	no	no	0	0
Central London (Background)(C)	1986	no	yes	yes	-	-
	1987	no	yes	yes	-	-
	1988	no	yes	yes	-	-
	1989	no	yes	yes	-	-

Table 3.10C Exceedences of limit values, guide values and guidelines for NO_2 in recent years *(modified after QUARG 1993)*.

Site	Year	EC Directive			WHO	
		Limit (98)	Guide (98)	Guide (50)	Hourly	Daily
Central London (Roadside) (C)	1986	yes	yes	yes	-	-
	1987	yes	yes	yes	-	-
	1988	yes	yes	yes	-	-
	1989	no	yes	yes	-	-
West London (C)	1986	no	no	no	-	-
	1987	no	yes	yes	-	-
	1988	no	no	no	-	-
	1989	no	yes	no	-	-
	1992?	no	yes	yes	0	0
	1993?	no	no	yes	0	0
Sheffield	1991	no	-	-	-	-
	1992	no	yes	yes	0	1
	1993	no	yes	no	1	0
Newcastle	1991	no	-	-	-	-
	1992	no	no	no	1	0
	1993	no	no	yes	1	0

effect (if it exists) disappears as the children age. Moreover, there is very little objective data of change in lung function with exposure.

3.3.3.4.2 Acute exposure

The most sensitive indicators of effects, from short term exposure, are increased airway resistance and decrements in lung function. Asthmatics are probably the most susceptible individuals, but bronchitics seem to be no more susceptible than the average person. Asthmatics showed pulmonary function effects when exposed for 30 minutes to a NO_2 concentration of 560 $\mu g\ m^{-3}$ (300 ppb). However, asthmatics exposed to 200 $\mu g\ m^{-3}$ (100 ppb) showed no significant change in lung function. Normal subjects showed pronounced decrements in lung function after being exposed to NO_2 concentrations above 5000 $\mu g\ m^{-3}$; exposure being at rest or after light exercise. The determination of the lowest observed effect level has proved problematic despite numerous studies having been undertaken. One study showed no effect after a 75 minute exposure to 7500 $\mu g\ m^{-3}$ while another showed an increase in airways resistance after a 20 minute exposure to 500 $\mu g\ m^{-3}$. There are inconsistencies in the data and no clear dose response relationship has been found.

3.4 Summary

Carbon monoxide concentrations were monitored using a NDIR gas absorption analyser and the levels of nitrogen oxides were determined using a chemiluminescent gas analyser. Both monitors allowed continuous real time measurement and data was recorded using a Campbell Scientific 21X Micrologger.

The major pollutants associated with road transport are sulphur dioxide, black smoke/particulates, nitrogen oxides, carbon monoxide and volatile organic compounds. Carbon monoxide is produced by the incomplete combustion of fossil fuels with the predominant anthropogenic source being the motor vehicle. However, emissions from petrol engined vehicles can be expected to decline with the introduction of three way catalysts to all new vehicles although the large increase in the number of vehicles on the road and the increase in the distance those vehicles are travelling will counteract the decline in emissions.

Work conducted by London Scientific Services showed roadside ambient CO air concentrations to be 2-3 times higher than local background with FoE claiming that concentrations at kerbside could be up to double the concentrations recorded 10 feet from kerbside.

There are no limit or guide values specified by the European Union for CO in ambient air. The EPAQS set an air quality standard for the UK of 10 ppm, measured as a running average 8-hour, which is very similar to the WHO guideline of 10 ppm over a set 8-hour period. The WHO guideline has been exceeded in the past, but, in recent years exceedences have been rare. Carbon monoxide, by combining with haemoglobin to form carboxyhaemoglobin, interferes with the supply of oxygen to cells. Those tissues most sensitive to hypoxia (the central nervous system and the cardiovascular system) are of most concern with regard to high exposures to CO.

The most significant nitrogen compounds related to road transport are nitric oxide (NO) and nitrogen dioxide (NO_2), known collectively as NO_x. Nitrogen oxides are primarily formed during high temperature combustion processes by the oxidation of nitrogen in the air. The chief anthropogenic sources are from the combustion of fossil fuels both in power generation and in the internal combustion engine. Power generation emissions have declined since 1970 while emissions from road transport have steadily grown. However, as with CO, NO_x emissions from petrol engined vehicles can be expected to decline with the introduction of the three way catalyst.

NO and NO_2 concentrations have been continuously monitored at two kerbside sites in London. The sites were located at Cromwell Road and a central London location. The Cromwell Road site, half a metre from the kerb, has recorded the highest concentrations of NO and NO_2 in the UK at 2200 ppb and 1817 ppb, respectively.

Limit and guide values have been specified by the European Union for NO_2 and incorporated into UK law by the Air Quality Standards Regulations *(DoE 1989)*. The limit value of 105 ppb (as a 98th percentile annual 1 hour average) is set at a level designed to protect human health. Guide values are set at 26 ppb, as a 50th percentile of hourly averages, and 71 ppb, as a 98th percentile of hourly averages.

NO_2 exposure is thought to be most dangerous to human health in periods of peak traffic flow. Evidence suggests that peak concentrations and not the duration of exposure are the most important factor to toxicity *(Hamilton & Duggan 1992)*.

CHAPTER 4. REMOTE VEHICLE EMISSION MEASUREMENT

4.1 Introduction

Chapter 4 primarily discusses the results from the FEAT system. The chapter begins with a consideration and description of the monitoring sites with regard to FEAT measurement and is followed by a discussion of the pilot survey and its implications for the subsequent monitoring programme. The third and final section presents the carbon monoxide and hydrocarbon results gained using the remote sensor. The discussion of the results focuses on the fleet emissions, the effect that the model year of a vehicle has on emissions and the contribution that model year makes to fleet emissions.

4.2 Site selection

A significant degree of introductory work was necessary before the commencement of each monitoring programme. Appropriate organisations needed to be contacted to gain permission to carry out the sampling including approval from the appropriate Local Authority for specific monitoring site location. This entailed liaison with Haringey, Middlesborough, Southwark and Leicester Borough Councils, as to the practicalities of sampling in their borough. Details such as power supply availability and which roadside location would cause the least disruption but fulfil the sampling criteria needed to be resolved before sampling could be undertaken.

Liaison was also necessary with the Universities of Greenwich and Nottingham to ensure the collaborative targets of the research were met. The police needed to be contacted to ensure that no infringement of local byelaws took place and local residents approached to ensure any disturbance caused was minimal. However, the availability of the FEAT equipment from the Transport Research Laboratory (TRL), from whom the equipment was borrowed, was the limiting factor as to when sampling could be undertaken. The final choice of location was therefore dependent to some extent upon the cooperation of the above organisations and was not solely determined by scientific criteria.

Three main criteria were considered when selecting monitoring sites *(Smith 1993; Lyons & Stedman 1991)*. Firstly, the location must have a good traffic flow with a suitable site having an average vehicle flow rate of more than 200 vehicles per hour. Secondly, the site must regulate the operating mode of most vehicles monitored to allow an accurate comparison of emissions from different vehicles (an ideal site would have the vehicles operating under slight load and an unsuitable site would have the vehicles operating under hard acceleration or in coasting mode).

Thirdly, the site must be adjacent to a single lane of unidirectional traffic with sufficient pavement area for the source, detector, video camera and data acquisition equipment (the data acquisition equipment was stored in either a van (plate 4.5) or a trailer (plate 4.7) during monitoring periods). When a two laned highway is used, as was the situation for each monitoring site, a unidirectional traffic flow was achieved by locating the source in the middle of the road and monitoring only one lane *(plate 4.1, plate 4.3, plate 4.6 and plate 4.8)*. In this situation, if two vehicles are passing at the same time, the measurement will be rejected as invalid. However, FEAT can operate across a dual carriageway, if the traffic volume is light enough to minimize the possibility of simultaneous multiple vehicle passes. This was initially attempted on the Old Kent Road but proved to be unsuccessful due to the traffic flow being too great resulting in large numbers of invalid readings caused by multiple passes. Therefore, the use of this site was discontinued and no results from the Old Kent Road will be reported.

Consideration must also be given to the impact that cold starting and hot starting vehicles have upon emissions. A hot start is when a vehicle is restarted only a short time after stopping; defined as a restart after ten minutes in the American Urban Dynamometer Driving Cycle *(Cadle & Stephens 1994)*. In a cold start, the emissions during the first few minutes are variable and generally high, as a result of the rich air to fuel ratios required during starting and because a catalyst system, if fitted, does not function until hot. Emissions are also elevated for a short time from a hot start. However, the catalyst warms quickly in a hot start thereby operating efficiently and reducing emissions rapidly.

Clearly, if a vehicle's emissions are measured by FEAT while in either a cold or hot start mode the possibility that the vehicle will be flagged as a high emitter regardless of its average emissions is greatly increased. Therefore, care should be taken to exclude cold and hot starting vehicles when selecting a site.

4.3 Site descriptions

The experimental sites **(Table 4.1)** are all single lane highways. Sites A, C and D are slow speed sites, with vehicle speeds no greater than 62 kmph and on average only 20 kmph. Site B, however, had average speeds of 40 kmph and top speeds approaching 80 kmph. At each site the FEAT system was used to measure carbon monoxide, hydrocarbons and carbon dioxide (not reported) emissions from passing vehicles.

Table 4.1 Descriptions of site locations used for FEAT monitoring programmes

Site	Name	Ordnance Survey Grid Reference	Site Description
Pilot	Bounds Green Road, Haringey	TQ 29 SE 294 918	Single lane carriageway, located in the Borough of Haringey, north London
A	Bounds Green Road, Haringey	TQ 29 SE 294 918	Single lane carriageway, located in the Borough of Haringey, north London
B	Dixons Bank, Middlesborough	NZ 51 SW 529 145	Single lane carriageway, located in the County of Cleveland, south of Middlesborough
C	Abbey Road, Southwark	TQ 37 NW 337 795	Single lane carriageway, located in the Borough of Southwark, south east London
D	Uppingham Road, Leicester	SK 140 614 053	Single lane carriageway, located in the County of Leicestershire, north Leicester

4.3.1 Bounds Green Road, Haringey (sampling site A)

The survey was undertaken between 25 May and 1 June 1994 on the A109, at a residential/industrial location in north London *(Plates 4.1 and 4.2)*. Due to the nature of the site the impact of cold/hot starting vehicle emissions is difficult to quantify. The primary source of cold/hot starting vehicles was thought to be those vehicles turning right from junction A *(Figure 4.1)*. Junction A is used as an exit from the university car parks and an industrial estate. However, this proved to be a minor problem. The use of junction A was very low during the day with elevated flows only occurring between the hours of 8am-9am and 5pm-6pm. Therefore, FEAT was not operated during these hours. Those vehicles exiting junction A during the day turned predominantly to the left towards the A406 (North Circular Road) with only light traffic flows turning to the right.

The FEAT unit was installed on the southeast bound carriageway 140 metres south of the junction with the North Circular Road (junction B). Vehicles approached the remote monitoring site up a slight incline although this was not thought to affect vehicle emissions significantly. FEAT measurement was not undertaken in the opposite lane because of the potential for stationary traffic when queuing at junction B *(Figure 4.1)*. A further problem with this direction is that vehicles tend to treat it as a two laned highway at busy times which could lead to large invalid readings. At the sampling point the road has an orientation of approximately 113 degrees from north.

Valid FEAT data was obtained from 11887 (91%) of the 13035 vehicles sampled for carbon monoxide over the four day monitoring period. The hydrocarbon channel provided 11099 (85%) valid readings from the same number of sampled vehicles. Vehicle flow rates on the south east bound carriageway varied between 511 and 780 vehicles per hour. Bounds Green Road represented a slow speed environment, for a single lane of traffic, with vehicle speeds ranging between 8 and 32 kmph.

Plate 4.1 Sampling site A (Bounds Green Road, Haringey) viewed from the north west.

Plate 4.2 Sampling site A (Bounds Green Road, Haringey) viewed from the south.

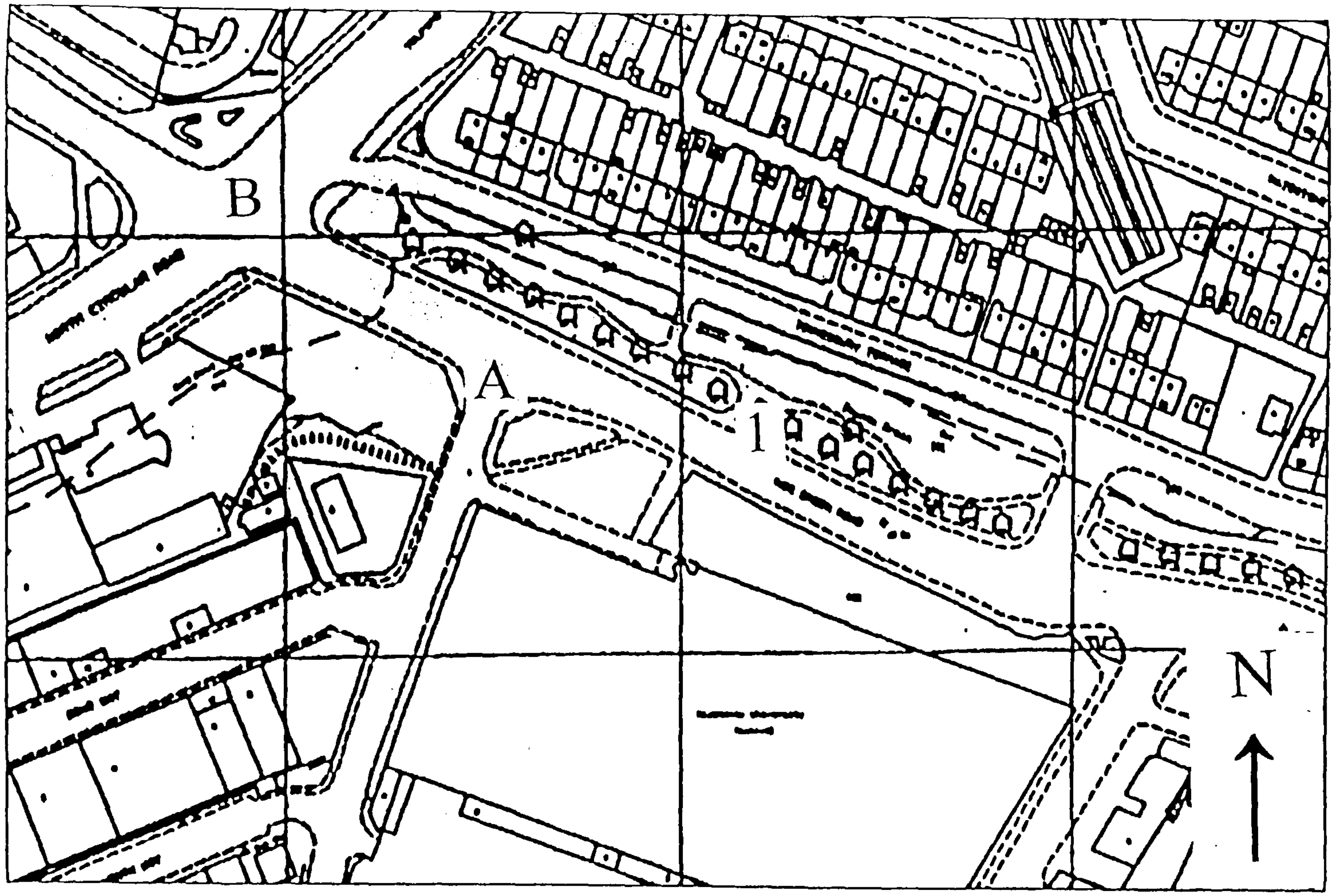


Figure 4.1 Map showing the locality of the Bounds Green Road monitoring site (scale 1:2500; *OS superplan*)

Key:
1. Position of the FEAT unit.
A. Junction with the industrial estate
B. Junction with the A406 (North Circular Road)

4.3.2 Dixons Bank, Middlesborough (sampling site B)

The survey was undertaken during the week of 7-11 June 1994 on the A172 *(Plates 4.3 & 4.4)* in a semi-residential location bordered by open fields. The residential nature of the site suggested that cold starting vehicles, with their elevated emission levels, could have biased the survey. Access to the monitoring site from the residential property was only possible by using junction A in *Figure 4.2.* During the monitoring period very low traffic volumes used this junction to exit from the residential property. It was assumed therefore, that cold starting vehicles were not a significant component of the vehicle fleet.

The FEAT unit was installed on the north west bound carriageway. Sampling was undertaken in this direction so that relatively high speed traffic could be monitored. The approach to the site was flat and straight for some distance to help facilitate higher speeds and constant driving conditions. However, this was not entirely successful as drivers inevitably slow to observe what is occurring and often mistake the emissions sensor for a speed trap. The opposite lane, heading in a south easterly direction, was not used because there was a fairly long uphill approach to the site. The uphill approach would have meant that the vehicle engines were working hard and therefore emissions would be higher than average for a given speed and could not have been said to be typical. At the sampling point the road has an orientation of approximately 310 degrees from the north *(Figure 4.2)*.

Valid FEAT data was obtained from 6509 (63%) of the 10255 vehicles sampled for carbon monoxide over the four day monitoring period. The hydrocarbon channel provided 3979 (53%) valid readings from a total of 7559 vehicles sampled. The difference in the number of vehicles sampled between the carbon monoxide and hydrocarbon channels is explained by the loss of one days data from the hydrocarbon channel. Vehicle flow rates on the northwest bound carriageway varied between 465 and 650 vehicles per hour. Dixons Bank represented a medium speed environment, for a single lane of traffic, with vehicle speeds ranging between 40 and 80 kmph.

Plate 4.3 Sampling site B (Dixons Bank, Middlesborough) viewed from the north west.

Plate 4.4 Sampling site B (Dixons Bank, Middlesborough) viewed from the south east.

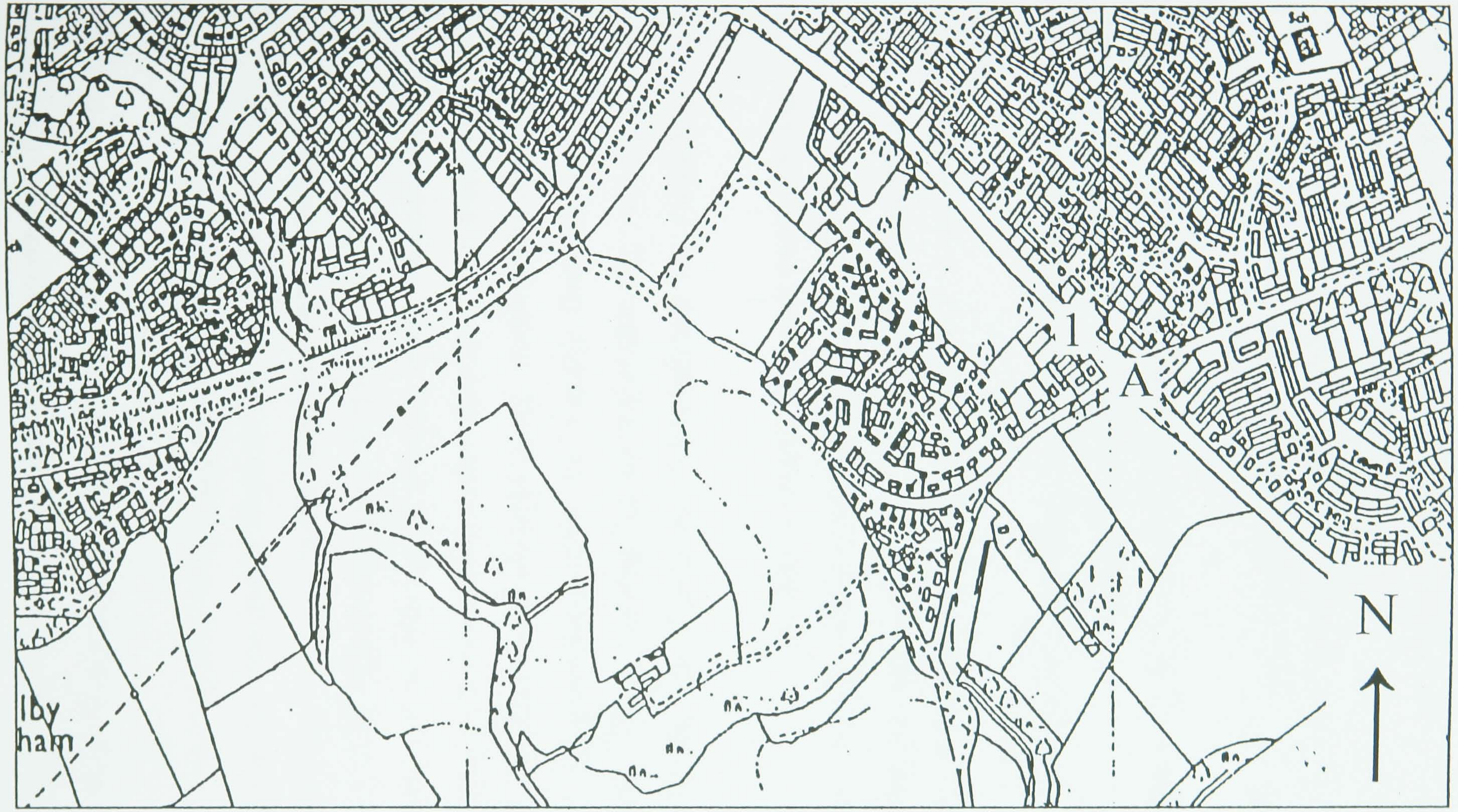


Figure 4.2 Map showing the locality of the Dixons Bank monitoring site (scale 1:10000; *OS NZ 51 SW*)

Key: 1. Position of the FEAT unit.
A. Junction with the residential estate

4.3.3 Abbey Street, Southwark (sampling site C)

Surveys were undertaken between December 1994 and August 1995. Monitoring was undertaken infrequently with on average three days sampling each month. However, sampling was dogged by problems and only the data collected on successfully completed days have been presented. This corresponds to the surveys carried out on 8 March and 8/9 August 1995. Some examples of the problems encountered were:

a) the stealing of generators, which occurred on two separate occasions
b) the breakdown of the FEAT equipment
c) steady rain for a week which meant that conditions were unsuitable for monitoring.

As with site A, quantification of cold start emissions is difficult. On a qualitative basis the site did not appear to have much local traffic with motorists using Abbey Street as a through route when travelling out of town. However, because the site is in a heavily urbanized area cold starts cannot be ruled out.

The FEAT unit was installed on the south west bound carriageway of the B202 *(Plates 4.5 & 4.6)*. The approach to the site *(Figure 4.3)*, as with Middlesborough, was flat and straight for some distance but unlike Middlesborough this site was not designed to facilitate higher speeds. The road layout allowed approaching drivers to be warned that an emissions test was underway by using traffic signs stating "Exhaust monitoring ahead drive through normally" *(plate 4.5)*. The opposite lane was not used due to the possible interference caused by a large number of vehicles entering a petrol filling station. Some queueing was caused on the southwest bound carriageway by vehicles turning into the petrol station but this was of minor significance. At the sampling point the road has an orientation of approximately 255 degrees from the north.

Valid FEAT data was obtained from 7414 (74%) of the 10052 vehicles sampled for carbon monoxide over the three days of monitoring. The hydrocarbon channel provided 6649 (66%) valid readings from the same number of sampled vehicles. Vehicle flow rates on the west bound carriageway varied between 127 and 376 vehicles per hour. Abbey

Plate 4.5 Sampling site C (Abbey Street, Southwark) viewed from the east.

Plate 4.6 Sampling site C (Abbey Street, Southwark) viewed from the west.

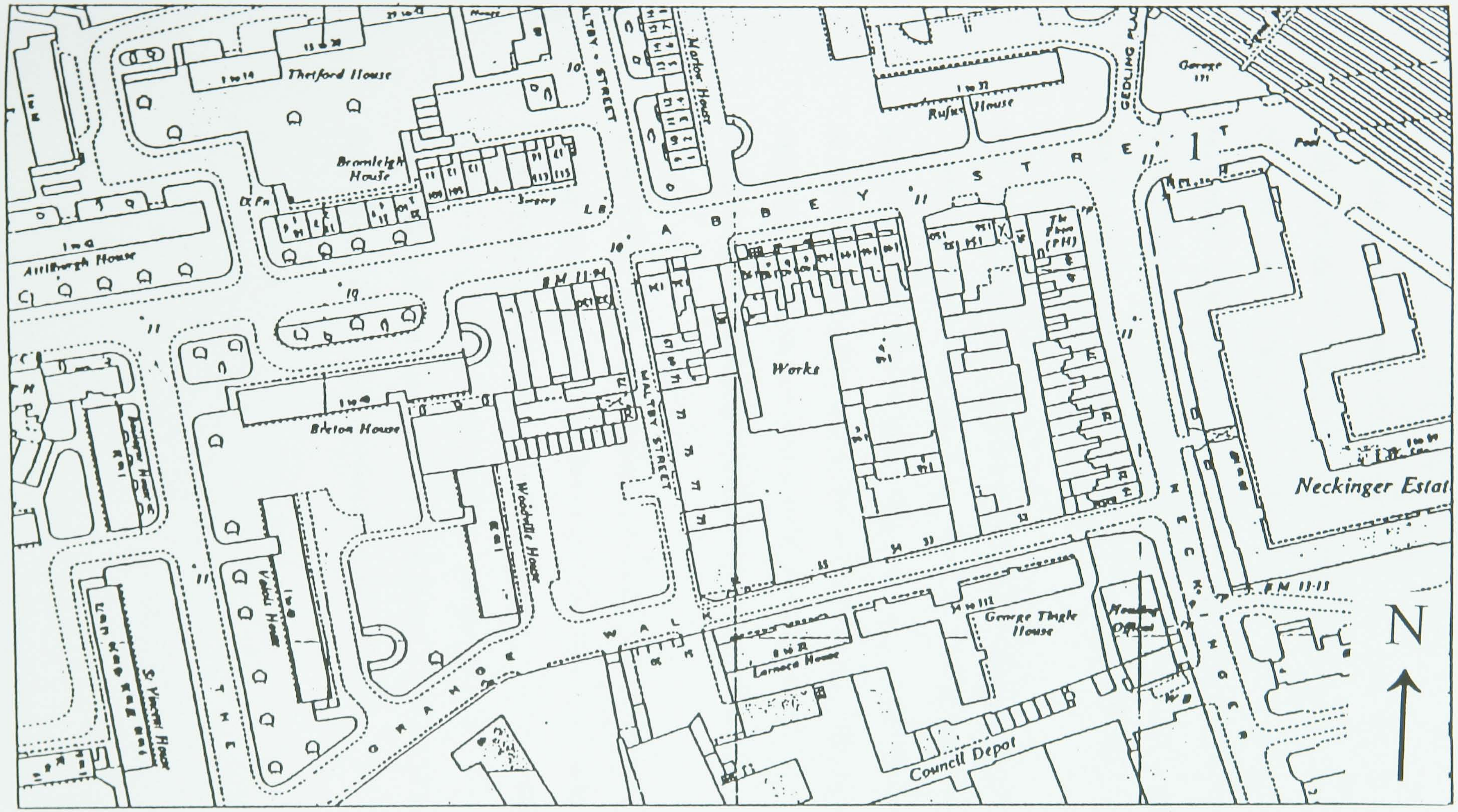


Figure 4.3 Map showing the locality of the Abbey Street monitoring site (*Sadler 1995*)

Key: 1. Position of the FEAT unit.

Street represented a slow speed environment, for a single lane of traffic, with vehicle speeds ranging between 4 and 50 kmph.

4.3.4 Uppingham Road, Leicester (sampling site D)

The survey was undertaken between 19 and 22 October 1995 on the A47 *(Plates 4.7 & 4.8)*, a residential/commercial location in Leicester. As with sites A and C quantification of cold start emissions is difficult and it is highly probable that the surrounding residential area does provide some cold starting vehicles. However, the A47 is a major route in and out of Leicester city centre and therefore the majority of the traffic should not be local and should be running at normal operating temperatures.

The FEAT unit was installed on the west bound carriageway *(Figure 4.4)* approximately 70 metres west of the junction (A) with the Leicester inner ring road (B667). Vehicles approached the remote monitoring site up a slight incline although this was not thought to affect vehicle emissions significantly. FEAT measurement was not undertaken in the opposite lane because of the potential for stationary traffic when queuing at junction A. At the sampling point the road has an orientation of approximately 270° from the north.

Valid FEAT data was obtained from 12,223 (81%) of the 15,176 vehicles sampled for carbon monoxide over the four days of monitoring. The hydrocarbon channel provided 7190 (48%) valid readings from the same number of sampled vehicles. Vehicle flow rates on the west bound carriageway varied between 315 and 701 vehicles per hour. Uppingham Road represented a slow speed environment, for a single lane of traffic, with vehicle speeds ranging between 8 and 30 kmph.

Plate 4.7 Sampling site D (Uppingham Road, Leicester) viewed from the east.

Plate 4.8 Sampling site D (Uppingham Road, Leicester) viewed from the north.

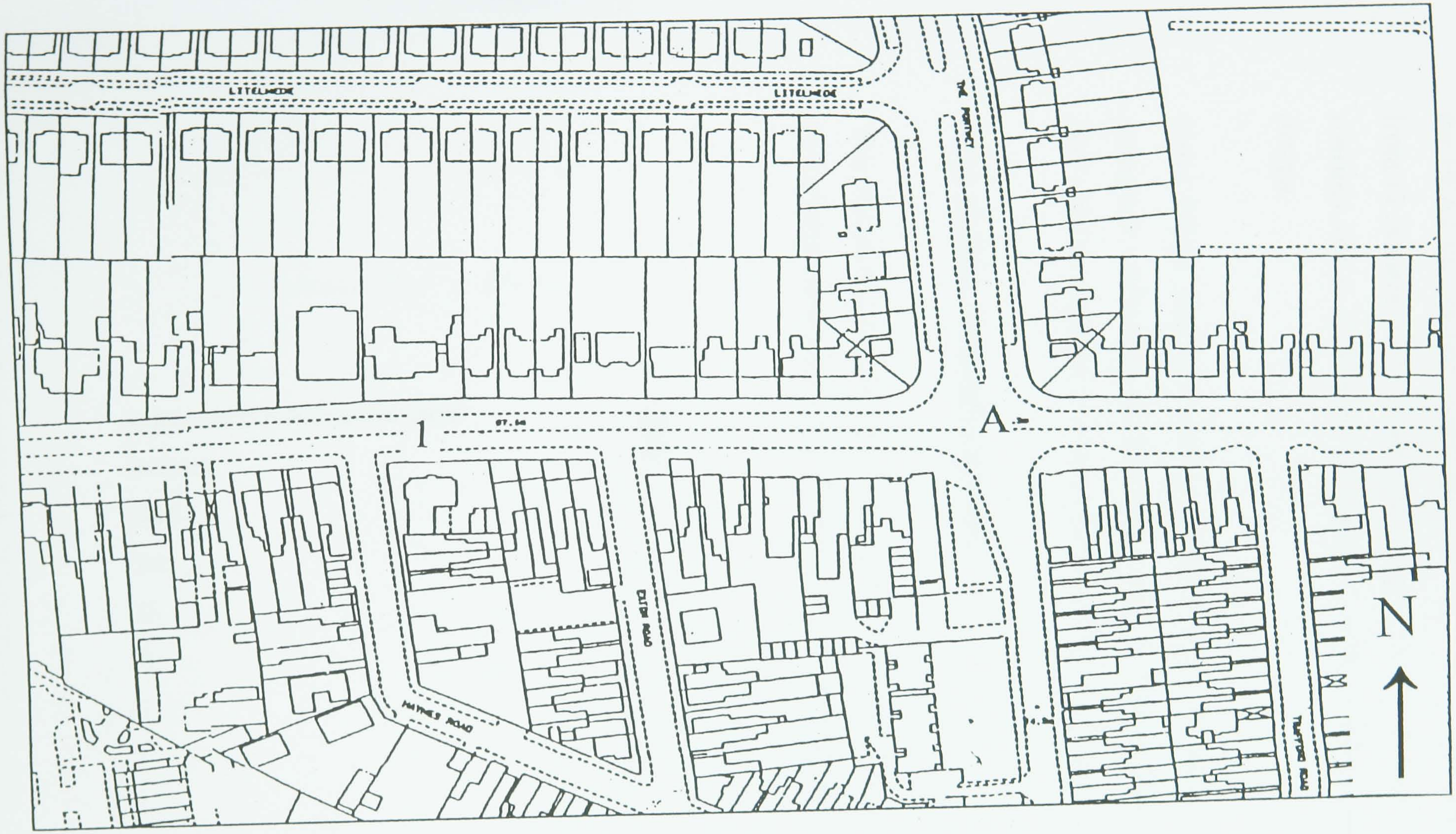


Figure 4.4 Map showing the locality of the Uppingham Road monitoring site (*Brookes 1995*)

Key: 1. Position of the FEAT unit.
A. Junction with the Leicester inner ring road (B667)

4.4 Pilot Study: Bounds Green Road, London 1993.

Prior to the main experimental stage of the project, a pilot survey was undertaken with the FEAT unit installed on Bounds Green Road, North London, for a three day period from 12 January to 14 January 1993. This was carried out to assess the sampling strategy routines and to familiarise the operators with equipment procedures and experimental design.

Valid FEAT data was obtained from 4535 of the 4834 vehicles sampled for carbon monoxide, carbon dioxide and hydrocarbons. Vehicle flow rates averaged approximately 600 vehicles per hour at a mean vehicle speed of 20 kmph.

4.4.1 Carbon monoxide emissions

The measured carbon monoxide emissions have been analysed according to their percentage contribution to the fleet and to the total fleet emissions (Figure 4.5).

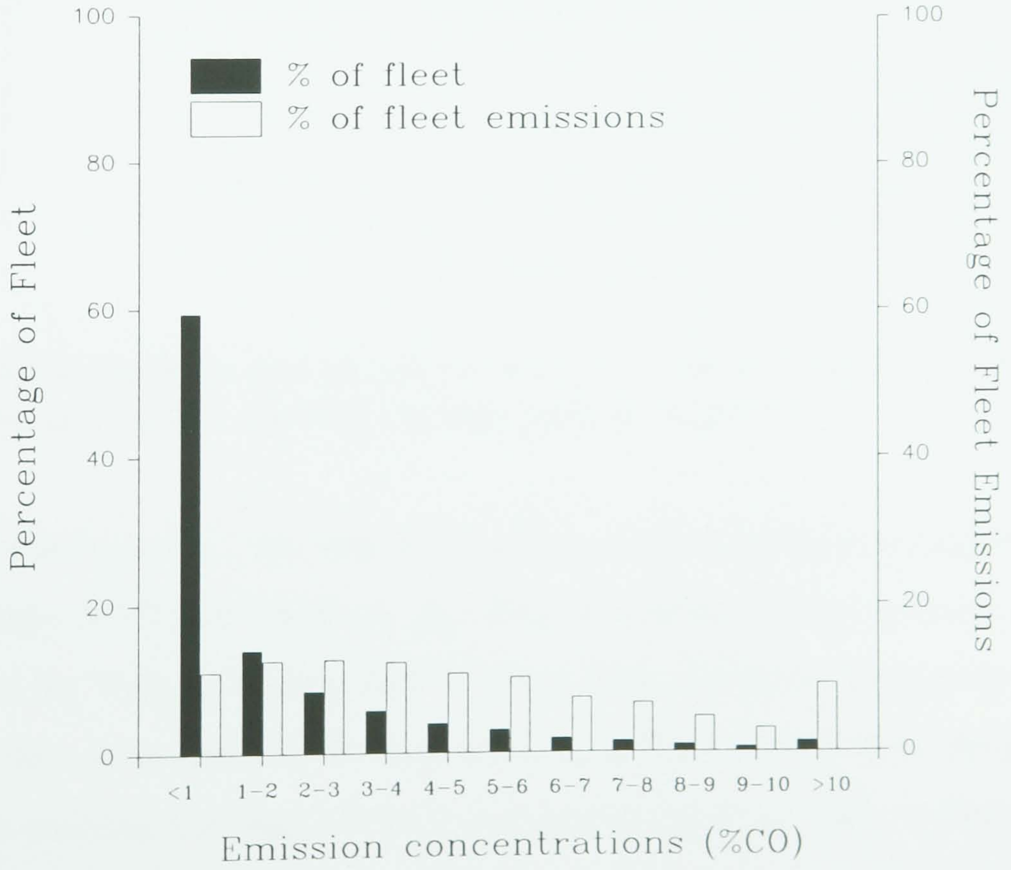


Figure 4.5 Carbon monoxide emissions for the pilot study (Bounds Green Road, Haringey) represented as percentage contribution to the fleet and the fleet emissions.

The fleet is defined as those motor vehicles which passed the remote sensor during the sampling exercise. The mean exhaust carbon monoxide level was 1.62%. However, the median measurement was considerably lower at 0.62%. The discrepancy can be explained by the high emitter component of the fleet, which although small in actual numbers provides a relatively large contribution to total fleet emissions. A high emitter as defined here, is any vehicle which falls within the highest 10% of polluters when the fleet is rank ordered *(Figure 4.6)*. Therefore, the median is a better test of the centre than the mean because the data is skewed by the high emitters.

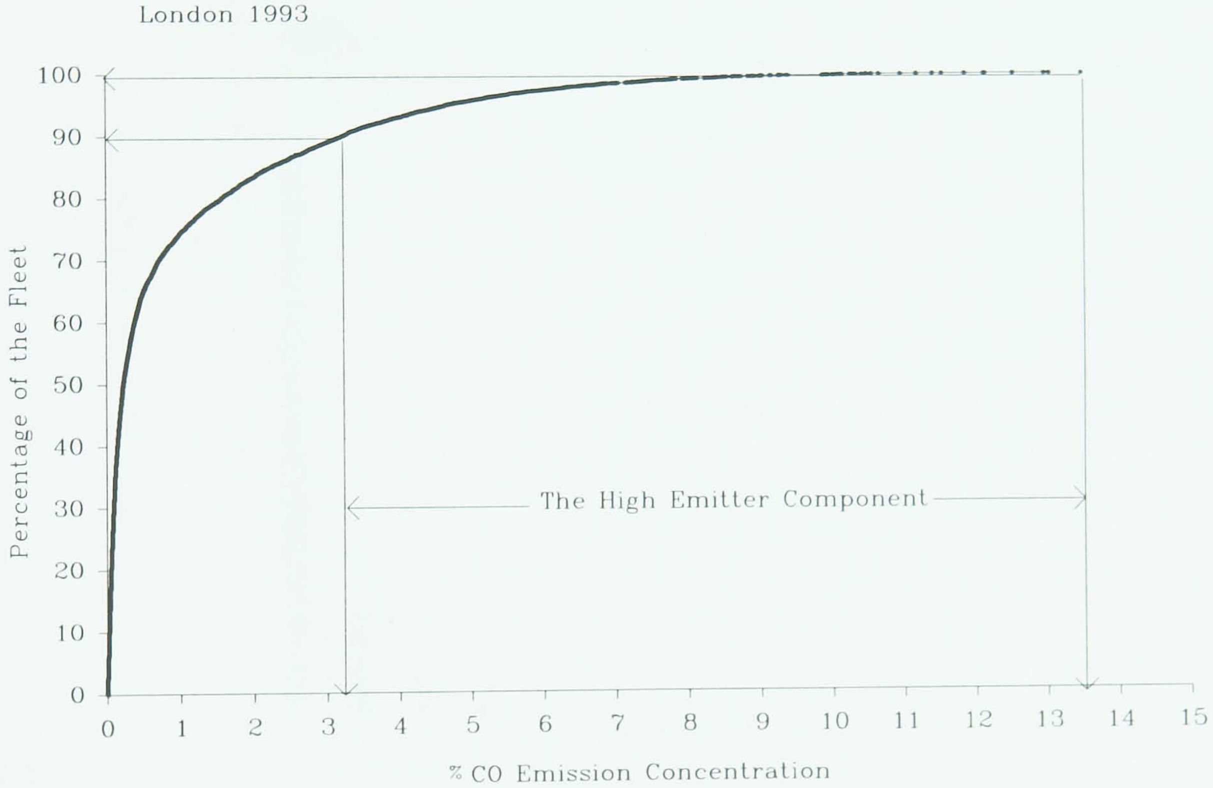


Figure 4.6 Cumulative plot of carbon monoxide emission concentrations against percentage of the fleet, showing the high emitter component.

It is clear from *Figure 4.5* that there is a small minority of highly polluting vehicles (CO emission range >7%) contributing a significant proportion to fleet emissions; with less than 5% of the fleet producing 24% of total fleet emissions. The converse is also apparent with a large majority of clean vehicles (a clean vehicle being defined here as any vehicle emitting less than 1% CO) contributing relatively little to fleet emissions. This is demonstrated explicitly by the fact that where 59% of the vehicles (CO emission range <1%) are producing only 11% of total fleet emissions (*Figure 4.5*).

4.4.2 Hydrocarbon emissions

The measured hydrocarbon emissions, as for carbon monoxide, have been analysed according to percentage contribution to the fleet and to total fleet emissions *(Figure 4.7)*. The mean exhaust hydrocarbon level was 767 ppm. However, as with carbon monoxide, the median value (400 ppm) was considerably less than the mean. The discrepancy is again explained by the high emitter component of the fleet. Therefore, the median is a much better test of the centre for both CO and HC emissions because the data is skewed by high emitters.

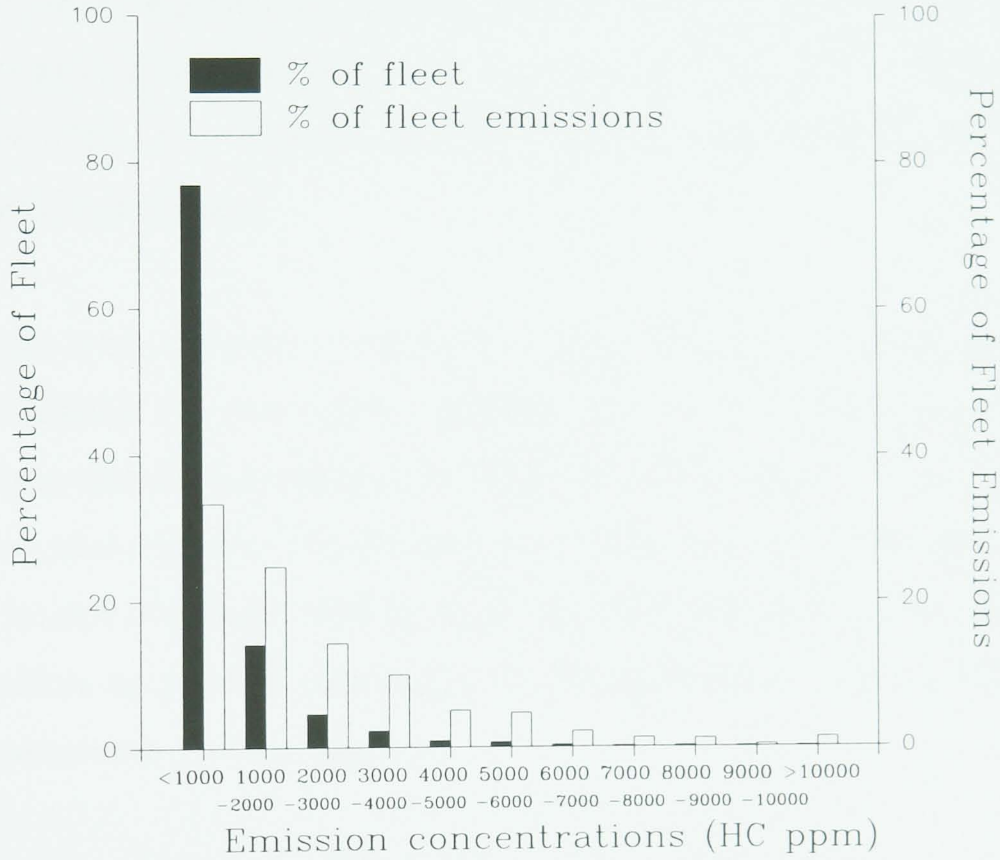


Figure 4.7 Hydrocarbon emissions for the pilot study (Bounds Green Road, Haringey) represented as percentage contribution to the fleet and the fleet emissions.

It is apparent that there is a large majority of clean vehicles (a clean vehicle being defined as any vehicle emitting less than 1000 ppm) contributing relatively little to fleet emissions. This is demonstrated explicitly in *Figure 4.7*, where 77% of the vehicles (HC emission range < 1000 ppm) are contributing only 33% of fleet emissions. The converse of this is also apparent with a small minority of highly polluting vehicles (HC emission range > 5000 ppm) contributing a significant proportion to fleet emissions; with less than 2% of the fleet producing 11% of total fleet emissions.

4.4.3 Implications of the pilot survey

During the pilot study a number of situations and problems became obvious which required alteration or re-consideration with regard to the initial experimental design. To ensure the best and most efficient operation of future sampling exercises, the following recommendations/observations were made:

- Sampling should only be conducted in dry weather/light showers and not during periods of heavy rain. Under such conditions the spray generated from tyres on a wet road interferes with some measurements, although particles of this size are not expected to greatly affect the transmission of infrared light *(Cadle & Stephens 1994)*. Thus, as a consequence of the interference, the efficiencies of the emission channels are reduced.

- Power was provided to the FEAT unit via the use of generators. This was problematic on two counts. Firstly, the use of generators could provide interference to the ambient air monitors. Therefore, the use of generators should be avoided whenever feasible. However, when their use is necessary, they should be located downwind with as much distance between them and the air quality monitors as possible. Secondly, on a practical basis a filling station had be convenient to provide fuel.

- From a practical viewpoint, the setup and calibration of the equipment needed more than one individual for successful completion. The setup of the FEAT unit takes approximately one hour with a speedy and accurate alignment of the source and detector requiring two people working in tandem. This is also the case when calibrating the system, with one person required to operate the computer software and another to operate the mechanics of the calibration. Therefore, two people are required when monitoring.

- Driving patterns were altered because of the presence of the FEAT unit as drivers mistook it for a speed trap. Therefore, to facilitate normal driving patterns signs should be placed on the approach to the monitoring site informing drivers that this is an emissions test and they should drive through normally. If this is not possible, an attempt should be made to mask the presence of the equipment and associated users.

- The FEAT equipment functioned correctly during the pilot survey but a blurred picture was produced on the monitor. This occurred because the high speed shutter on the video camera was not set to 1/1000 of a second. If the camera loses power all functions revert to the default settings and therefore the high speed shutter needs to be reset.

- The FEAT equipment may return a low voltage error due to too much CO_2 or a misaligned path between the source and detector *(Bishop & Stedman 1991)*. The misalignment of the source and detector proved to be the primary reason for the low voltage error. After initial installation the position of the detector and source were marked by spray painting their feet. Therefore, if the equipment moved subsequently, which occasionally occurred because of the vibrations caused by heavy traffic, the equipment could be quickly realigned.

- The FEAT unit may return a high voltage error because of an elevated signal or too short a pathlength between source and detector *(Bishop & Stedman 1991)*. An excessively high signal proved to be the primary reason for the high voltage error. This did not occur during the pilot survey but was a problem during the monitoring exercise in Leicester. After the equipment had been sent to the University of Denver for a service it returned with its signal strength boosted. This was overcome by placing a piece of gauze over the aperture on the detector, which received the infrared beam, thereby reducing signal strength.

4.5 Carbon monoxide emissions

4.5.1 Fleet emissions

The measured carbon monoxide emissions have been analysed according to their percentage contribution to the fleet and to the total fleet emissions *(Figures 4.8, 4.9, 4.10, & 4.11)*. It is clear that there is a large majority of clean vehicles contributing little to fleet emissions at all sites. This is demonstrated for site A (Haringey) in *Figure 4.8*, where 70% of the vehicles (CO emission range <1%) are providing only 12% of fleet emissions. Sites B, C and D produced similar results **(Table 4.2)** supplying evidence that a large proportion of the fleets are low emitting and contribute little to fleet emissions. Those vehicles in the emission category representing <1% CO concentrations in Middlesborough (site B), Southwark (site C) and Leicester (site D), accounted for 77%, 64% and 61% of the fleets respectively but contributed only 16%, 8% and 9% respectively of fleet emissions.

Further analysis of the Haringey data shows that those vehicles emitting less than 2% carbon monoxide accounted for 82% of the fleet, but contributed only 25% of total fleet emissions. Supplementary analyses of sites B, C and D produced similar figures to those of site A. Those vehicles in the emission category representing <2% CO concentrations in Middlesborough (site B), Southwark (site C) and Leicester (site D), accounted for 87%, 75% and 74% of the fleets respectively but contributed only 33%, 19% and 21% of fleet emissions.

In contrast, it is also apparent that a small minority of highly polluting vehicles (CO emission concentrations >7%) contribute a significant proportion to fleet emissions, with approximately 5% of the fleet at site C producing 30% of total fleet emissions. Sites A, B and D produced similar results **(Table 4.2)**. Those vehicles with emission concentrations >7%, accounted for only 3%, 1% and 5% of the fleet at the Haringey, Middlesborough and Leicester sites respectively but contributed to 27%, 13% and 29% of fleet emissions.

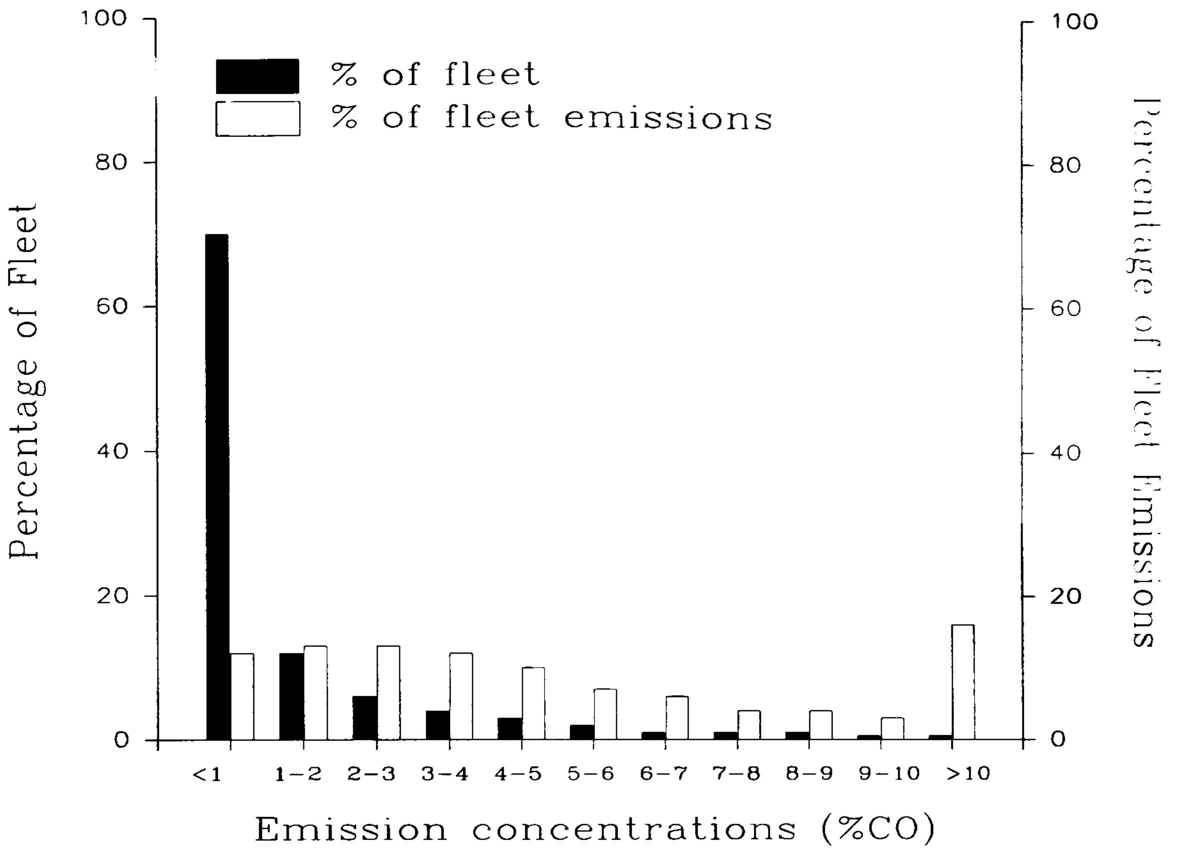


Figure 4.8 Carbon monoxide emissions for site A (Bounds Green Road, Haringey) represented as percentage contribution to the fleet and the fleet emissions.

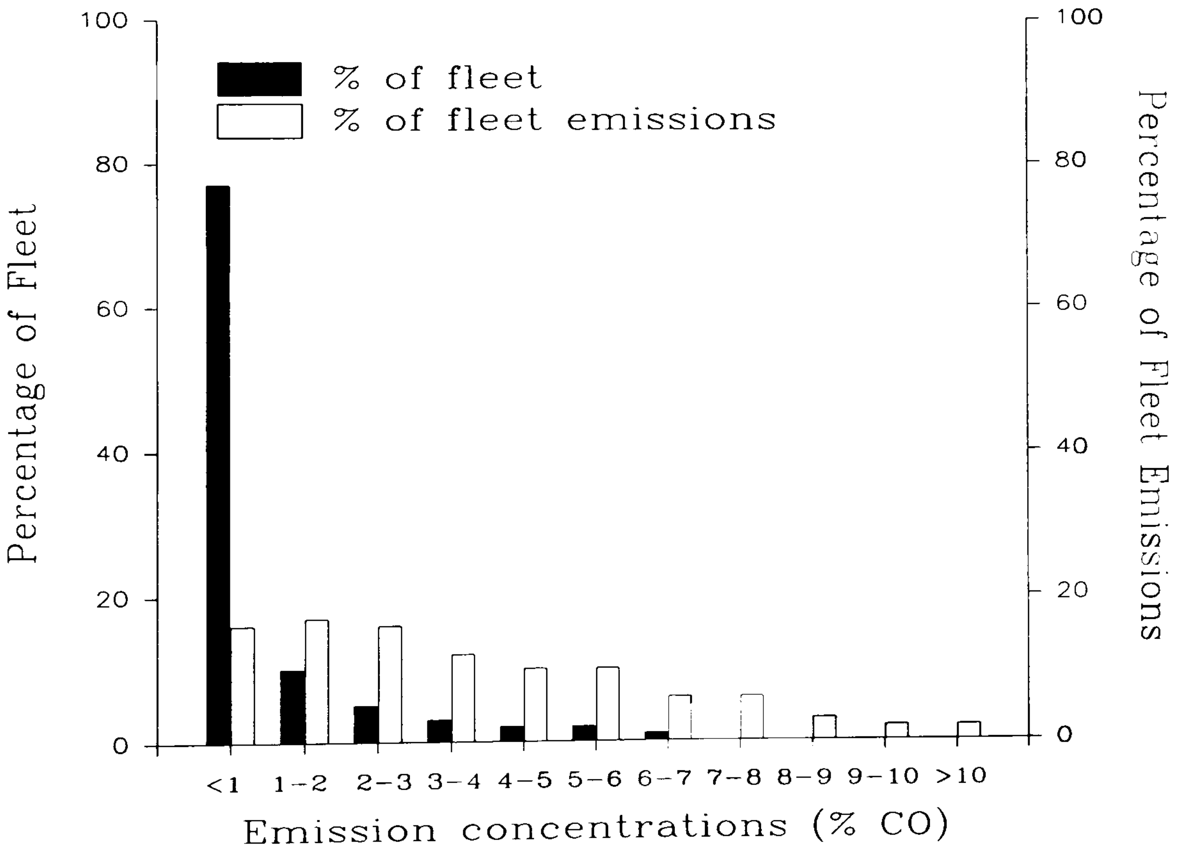


Figure 4.9 Carbon monoxide emissions for site B (Dixons Bank, Middlesborough) represented as percentage contribution to the fleet and the fleet emissions.

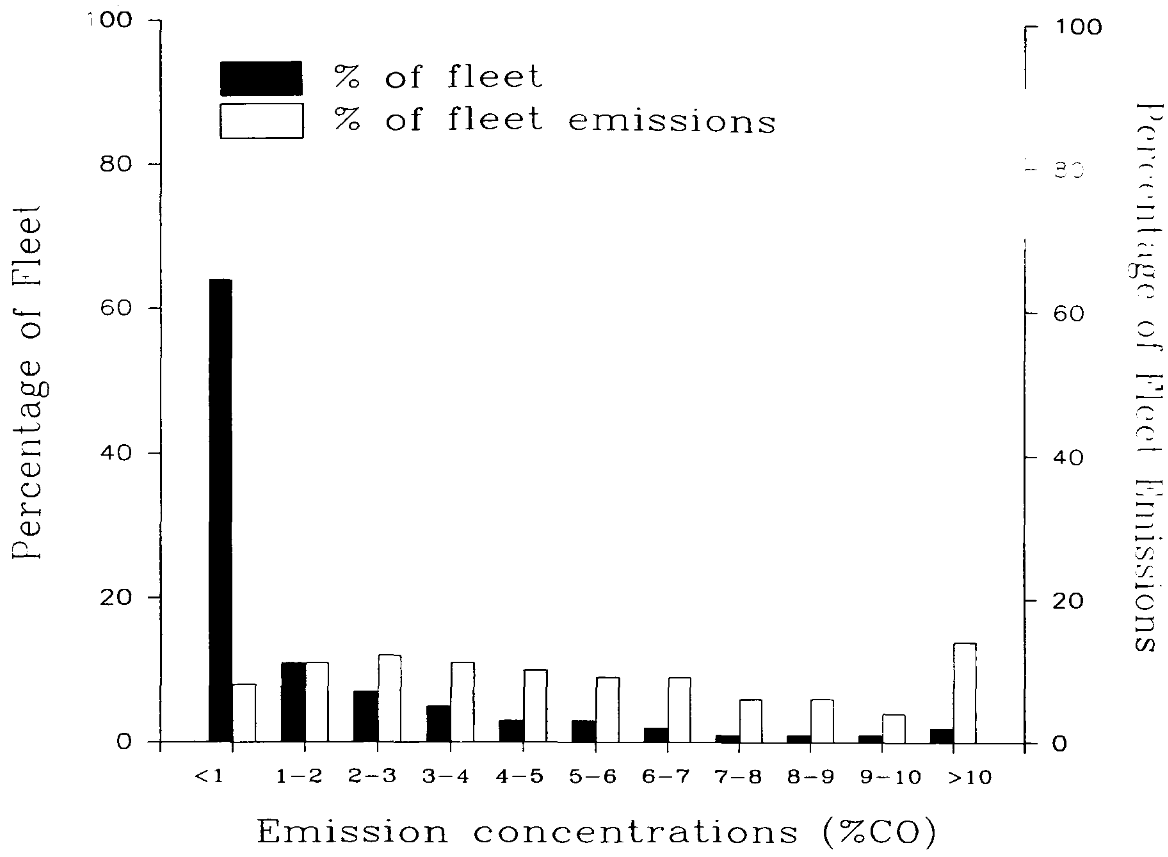


Figure 4.10 Carbon monoxide emissions for site C (Abbey Street, Southwark) represented as percentage contribution to the fleet and the fleet emissions.

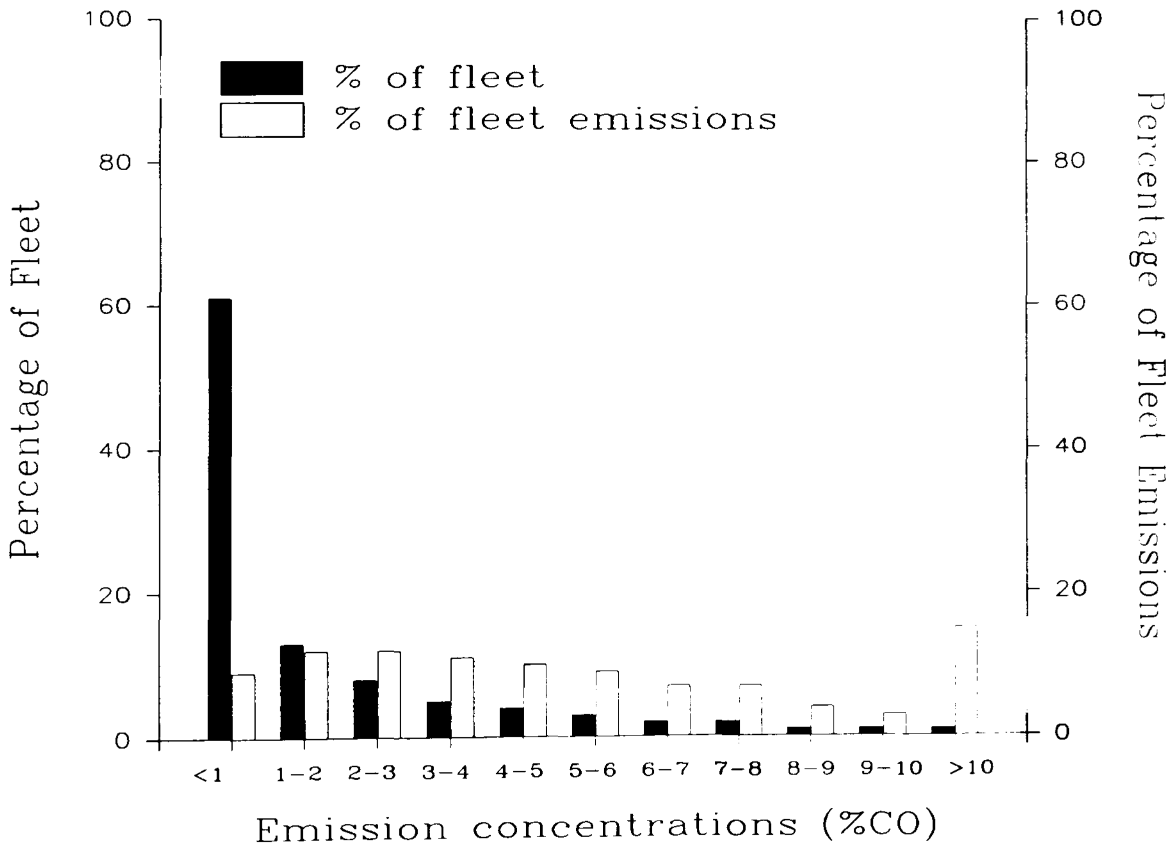


Figure 4.11 Carbon monoxide emissions for site D (Uppingham Road, Leicester) represented as percentage contribution to the fleet and the fleet emissions.

The mean emissions for sites A, B, C and D are: 1.26%, 0.84%, 1.46% and 1.63% respectively. The corresponding median values at each site are: 0.27%, 0.14%, 0.35% and 0.53% respectively **(Table 4.2)**. The differences between the mean and median values reflect the very skewed nature of the emission distributions shown in *Figures 4.8, 4.9, 4.10 and 4.11*. Calculation of skewness values for the emission distributions shows site D to have the most skewed distribution with a skewness value of 2.46 although similar values were recorded at all sites (site A, 2.35; site B, 2.39; site C, 2.35). The emission distributions are similar to those seen in other remote sensing studies carried out in the UK *(Muncaster 1995; Muncaster et al. 1994; Vanke & Bidgood 1992)* and overseas *(Sjodin 1993; Stephens & Cadle 1991)*. However, those studies undertaken in Mexico are quite different, with vehicle emissions being more normally distributed *(Beaton et al. 1992; 1991)*. The explanation given for the more normal distribution was that there was an absence of emissions control equipment and a deliberate tuning of vehicles for peak power.

Table 4.2 Summary of carbon monoxide statistics

Site	Mean (%)	Median (%)	Emission category <1%		Emission category <2%		Emission category >7%	
			Vehicles (%)	Fleet emissions (%)	Vehicles (%)	Fleet emissions (%)	Vehicles (%)	Fleet emissions (%)
A	1.26	0.27	70	12	82	25	3	27
B	0.84	0.14	77	16	87	33	1	13
C	1.46	0.35	64	8	75	19	5	30
D	1.63	0.53	61	9	74	21	5	29

The mean emission level found in Leicester is substantially lower than the 2.32% value recorded by *Zhang et al. (1995)* **(Table 2.2)** for the same city but similar to the 1.34% found by *Vanke & Bidgood (1992)*. The 2.32% value is the highest published mean emission level recorded in the UK and should be treated with some care as it is not likely to be representative of the Leicester fleet as a whole. Such a high value would place Leicester in cluster Z *(Figure 2.16)* in the hierarchial cluster analysis performed by *Zhang et al. (1995)* along with Mexico City, Kathmandu and Bangkok. Leicester seems out of place in cluster Z, where the fleet is composed of older vehicles that are less well maintained and are not subjected to emission control legislation or new emission standards, and if a value of 1.63% as found at site D or 1.34% as found by *Vanke & Bidgood (1992)* were used in place of 2.32% then a more realistic placing in cluster Y would occur. The fleet in cluster Y are newer than those of cluster Z, more well maintained and are subject to emission control legislation.

The mean emission levels for the two London sites are similar but higher than the 0.96% found by *Zhang et al. (1995)*. *Vanke & Bidgood (1992)* obtained a mean value for London of 1.471%, very similar to that found in Haringey and Southwark. A possible reason for the lower value found by *Zhang et al.* is that the sampling site used, had a high proportion of new and diesel powered vehicles which effectively lower the mean value. The London fleet is placed by *Zhang et al.* in cluster X, where the fleet is predominantly new and strict inspection programmes are enforced, but a more logical placement for the whole London fleet would be cluster Y. No comparable data is available for site B but if it had been included in the cluster analysis its mean value would place it approximately between Seoul and Lyons in cluster X among the new well controlled vehicles. However, the fleet at site B was not 'new' and has the same emission control legislation as sites A, C and D. A possible reason for the lower mean emission value at site B is that the vehicles are better maintained in comparison to sites A, C and D.

4.5.2 Model year carbon monoxide emissions

In the following analyses, model year is determined by the prefixed or suffixed letter on the number plate so that model year 1984 corresponds to any vehicle registered between 1 August 1984 and 31 July 1985.

Mean carbon monoxide emissions have been plotted against model year to determine if there is a relationship between them and the age of the vehicle. Error bars have been calculated, using a 95% confidence interval, for the mean values to indicate the degree of uncertainty in the data. The error bars define a range within which it may be reasonably assumed the true value lies. The extreme values of the range define the confidence limits.

The results initially showed a large degree of scatter *(Figures 4.12, 4.13, 4.14 & 4.15)*. The Middlesborough and Leicester fleets showed the lowest association between the model year of a vehicle and mean pollutant emissions returning r^2 values of 0.19 and 0.31 respectively. The relationship was stronger for the Haringey and Southwark fleets with r^2 values of 0.44 and 0.56 respectively. The divergence from a 1:1 agreement can be largely attributed to the older vehicles in the fleet, which clearly result in increased scatter at all sites. This is reflected by the error bars from the older vehicles with, in general, older model years having larger error bars and therefore there being greater uncertainty in the data. As the number of samples of a fixed size decreases the observed distribution of the mean moves further away from the underlying theoretical distribution. In other words, the fewer the cars sampled the larger the error bars are. This is what occurs for the older model year vehicles.

An examination of the Haringey emissions data showed that, from a total of 7397 valid vehicle measurements (a valid vehicle measurement being one where the year of first registration is visible upon the number plate), only 949 measurements (13% of the fleet) correspond to vehicles produced prior to 1983. An examination of the Middlesborough emissions data showed a similar scenario with only 310 valid vehicle measurements (5% of the fleet) recorded for those vehicles produced before 1983. The Southwark and Leicester fleets provided comparable figures with 7% (from a total of 4453 vehicle measurements) and 9% (from a total of 11363 vehicle measurements) of the respective fleets being registered before 1983.

Further analysis of the Middlesborough data on an annual basis, showed that for 1964, 1970 and 1973 there was only one vehicle measured for each model year and this increased to only two for 1968, 1971 and 1972 (there was no data available for vehicles produced in 1965, 1966, 1967 and 1969). For those model years with only one vehicle measured no error bars have been calculated. Those model years with only two vehicles sampled demonstrate the effect a small sample size has on error bars with enormous error

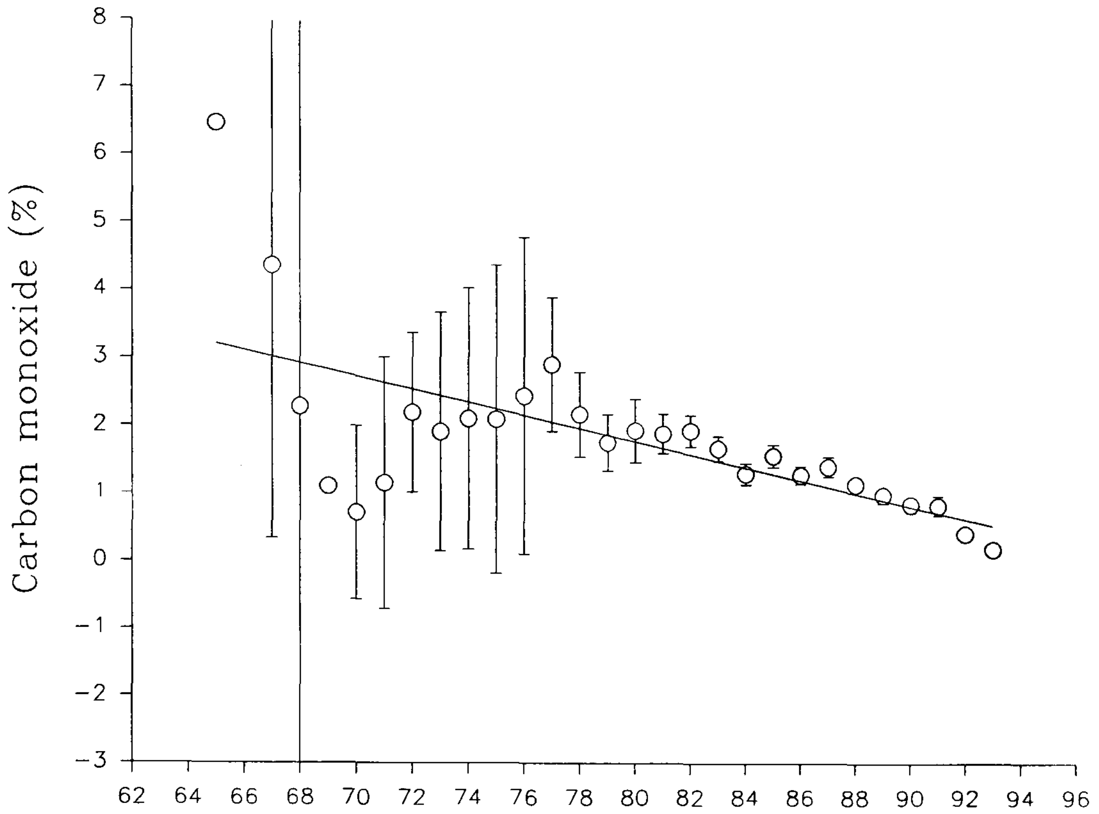


Figure 4.12 Relationship between mean annual carbon monoxide emissions, with error bars calculated using a 95% confidence interval, and model year for site A (Bounds Green Road, Haringey) (r^2 = 0.44).

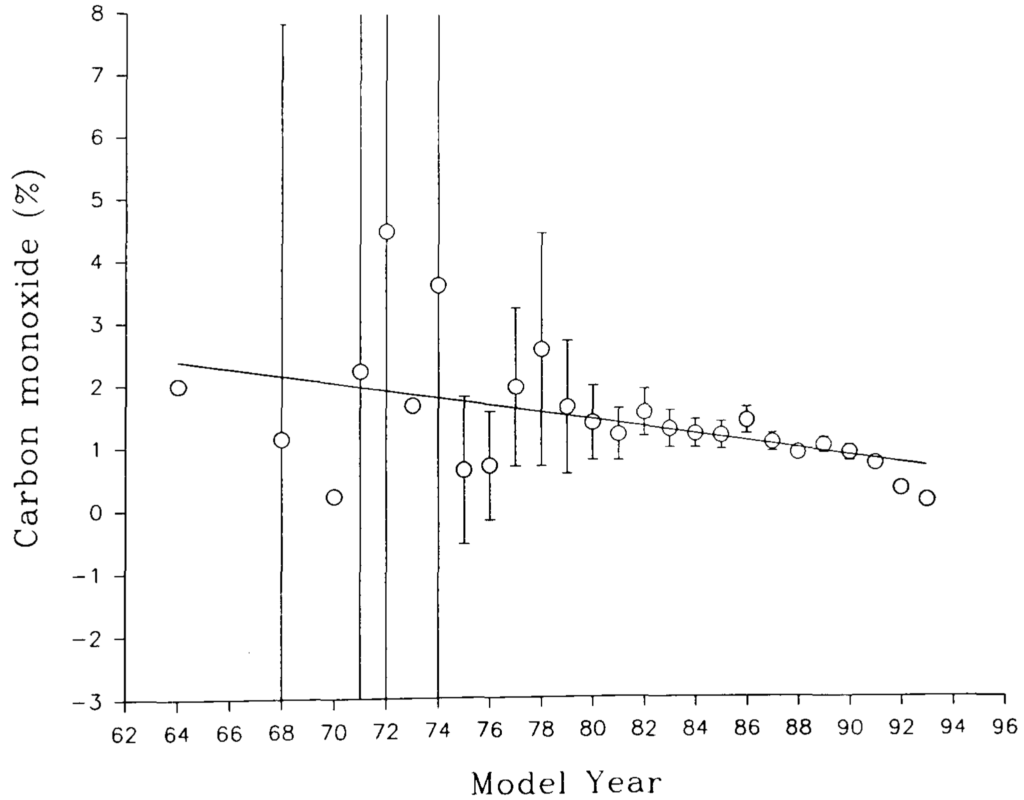


Figure 4.13 Relationship between mean annual carbon monoxide emissions, with error bars calculated using a 95% confidence interval, and model year for site B (Dixons Bank, Middlesborough) (r^2 = 0.19).

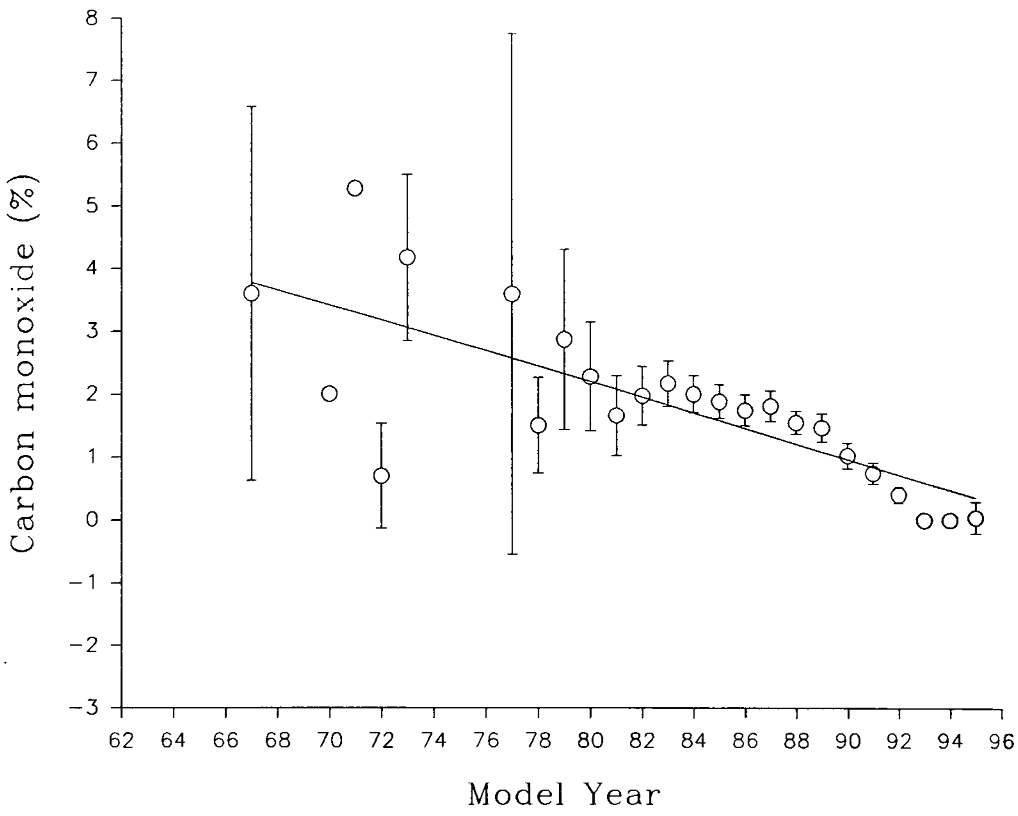


Figure 4.14 Relationship between mean annual carbon monoxide emissions, with error bars calculated using a 95% confidence interval, and model year for site C (Abbey Street, Southwark) (r^2 = 0.56).

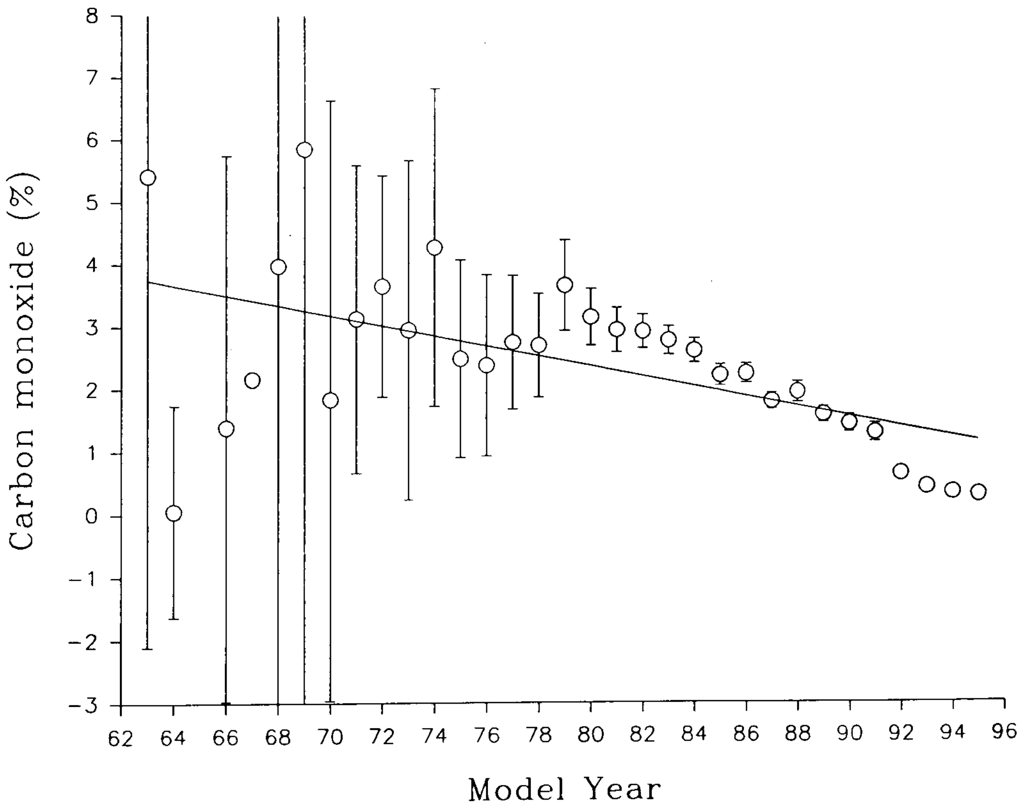


Figure 4.15 Relationship between mean annual carbon monoxide emissions, with error bars calculated using a 95% confidence interval, and model year for site D (Uppingham Road, Leicester) (r^2 =0.31).

bounds for 1968, 1971 and 1972. Due to the size of the confidence limits full error bars cannot be included as they would make interpretation of the remainder of the graph almost meaningless. Analysis of the confidence limits for 1971 demonstrate this with values of 9.6% and -5.2%. This situation is repeated across all sites for model years with small sample sizes. The extreme error bounds have therefore not been included for those model years with extremely small sample sizes. Small sample sizes are found at site A, to a greater or lesser extent, until 1980 when the sample size increases to 47 vehicles. In this situation, the observed distribution of the mean is closer to the underlying theoretical distribution, thereby reducing uncertainty in the mean and the size of the error bars. The Southwark site, displayed a comparable data set with only two vehicles measured for model years 1970 and 1971 and only three vehicles measured for 1975 (there was no data available for vehicles produced in 1968, 1969, 1974, 1975 and 1976) with the associated large error bounds. This situation improves with the mean becoming more accurate by 1981 as the model year sample size increased to 58 vehicles. The Haringey and Leicester fleets had similar distributions to those of Middlesborough and Southwark.

Thus, the vehicle pool is not large enough to support statistical analysis for older vehicles. However, even if there were sufficient vehicles for an analysis, a straight line relationship would be unlikely. It is probable that the increases in emissions with age would not continue at the same rate and a levelling off would occur. A possible reason for this is that those vehicles which were high emitting due to poor maintenance have become unroadworthy and only those vehicles which are well maintained, and therefore low emitting, remain in the fleet.

4.5.2.1 Model year emissions 1983-1995

In view of the uncertainity of the data from older vehicles and the associated large error bounds, only those vehicles registered between 1983-1993 for sites A and B and 1985-1995 for sites C and D were included in an additional regression analysis of the variation of carbon monoxide emissions with model year. The large number of vehicles sampled for the newer model years (1983-1995) means that the ranges within which the true mean values lie are comparatively small for all sites. Due to monitoring having been conducted at sites A and B approximately 15 months prior to sites C and D, the most recent data available for sites A and B were from model year 1993 not 1995. However, the newest ten model years were analysed from each location.Sites A & B and C & D are not strictly comparable when evaluating model year emissions. This is because although vehicles were registered in the same model year, those vehicles measured at sites A and B are two years older than those measured at sites C and D. For example those vehicles registered in model year 1993 at sites A and B were new whereas those vehicles registered in model year 1993 at sites C and D were two years old.

Scatter plots showing the relationships between mean carbon monoxide levels and model year are displayed in *Figures 4.16, 4.17, 4.18 and 4.19.* As a result of omitting the older vehicles from the additional regression analyses much stronger relationships were found at all sites. The Southwark and Leicester fleets showed the greatest association between the model year of a vehicle and mean pollutant emissions returning similar r^2 values of 0.94 and 0.95 respectively. The regression equations for the relationships found at sites C and D are shown in Equations 4.3 and 4.4, respectively. Sites C and D have similar large negative regression coefficient values of -0.233 and -0.220, respectively. This indicates that at both sites mean pollutant emissions change in a similar pattern such that a small increase in model year is accompanied by a large decrease in pollutant level.

The relationship between the model year of a vehicle and mean pollutant emissions was only a little weaker for the Haringey and Middlesborough fleets where r^2 values of 0.88 and 0.77, respectively were found. The regression equations for the relationship found at sites A and B are shown in Equations 4.1 and 4.2, respectively. These sites have similar regression coefficient values of -0.131 and -0.106, respectively, which indicate that at both sites mean pollutant emissions change in a similar pattern. However, whereas at sites C and D a small increase in model year was accompanied by a large decrease in pollutant level, at sites A and B because the regression coefficients are smaller and therefore, the slope of the regression line is shallower, a larger increase in model year is required for same decrease in pollutant level.

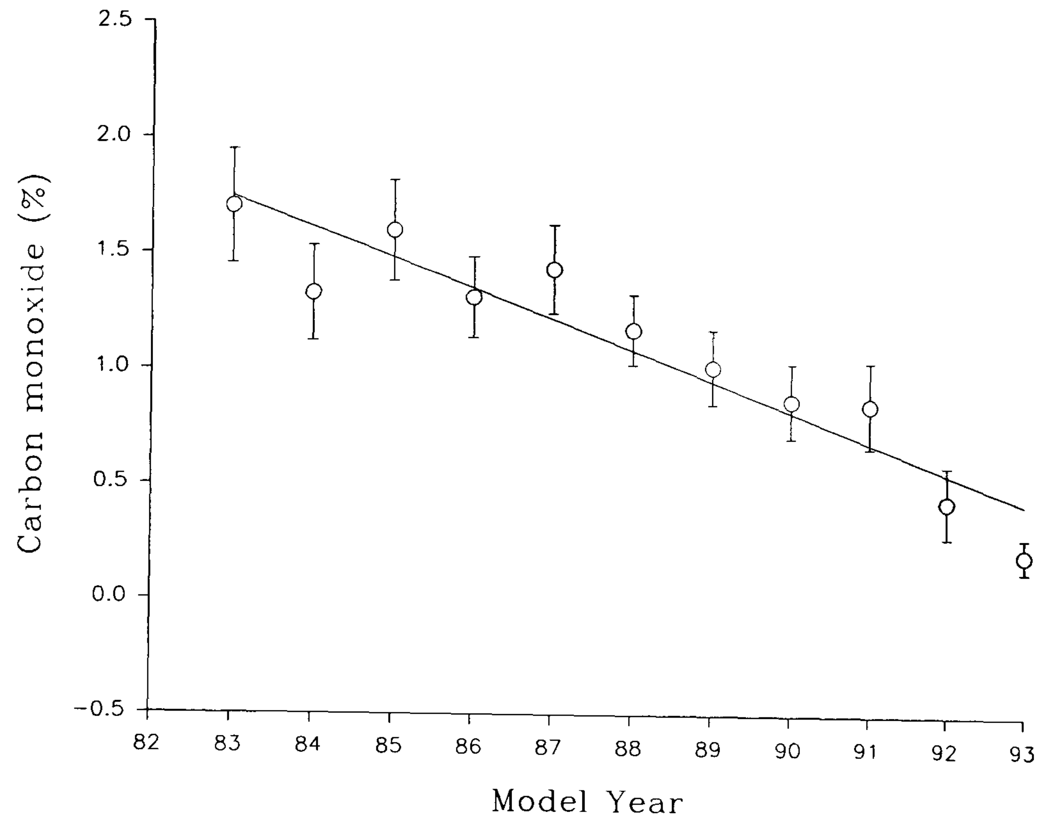


Figure 4.16 Relationship between mean annual carbon monoxide emissions, with error bars calculated using a 95% confidence interval, and model years, 1983-1993, for site A (Bounds Green Road, Haringey) (r^2 = 0.88).

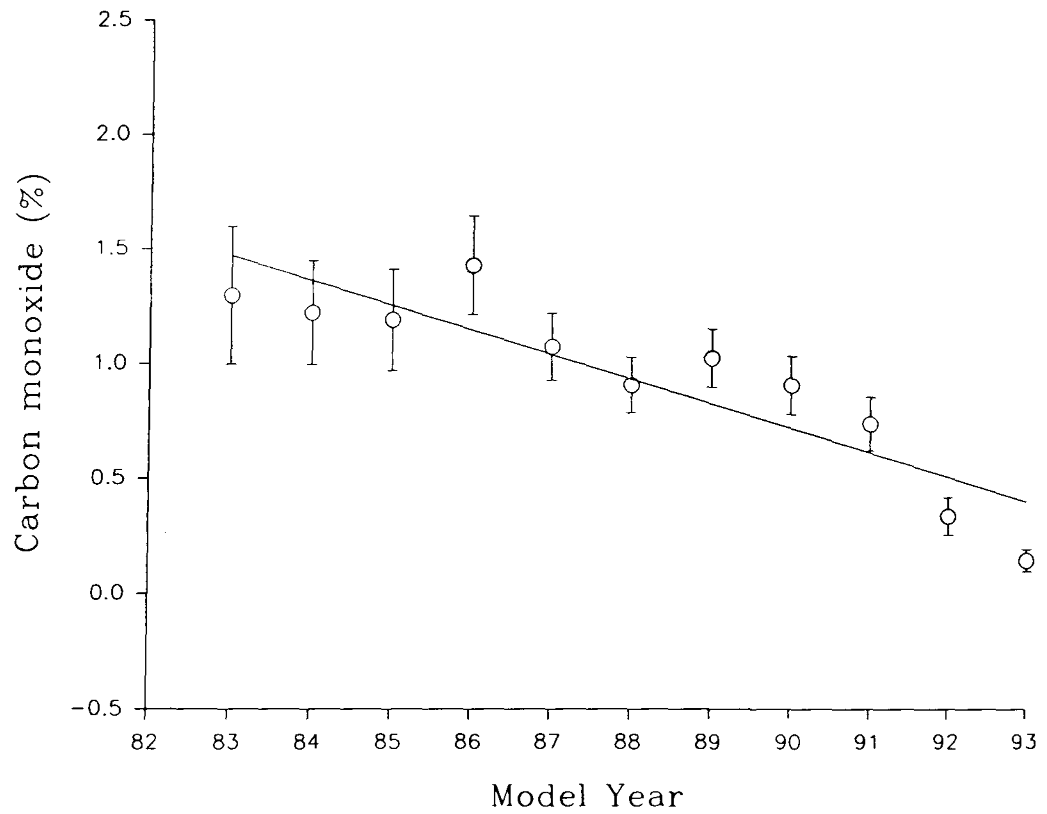


Figure 4.17 Relationship between mean annual carbon monoxide emissions, with error bars calculated using a 95% confidence interval, and model years, 1983-1993, for site B (Dixons Bank, Middlesborough) (r^2 = 0.77).

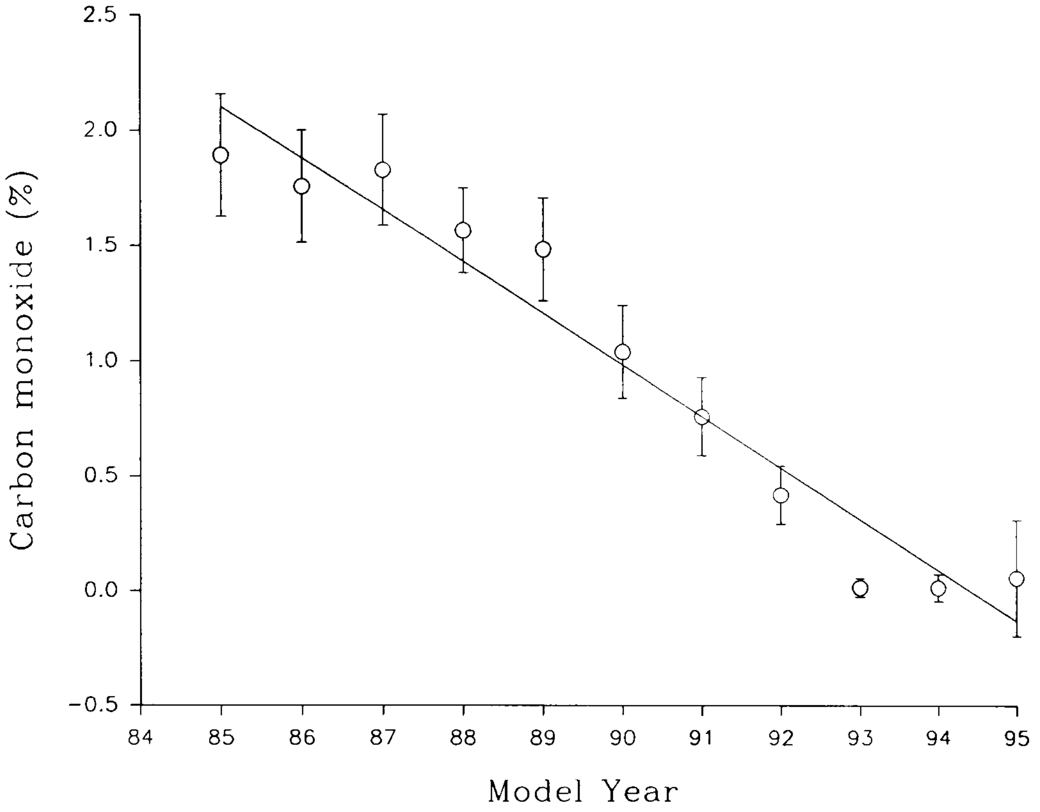


Figure 4.18 Relationship between mean annual carbon monoxide emissions, with error bars calculated using a 95% confidence interval, and model years, 1985-1995, for site C (Abbey Street, Southwark) (r^2 = 0.94).

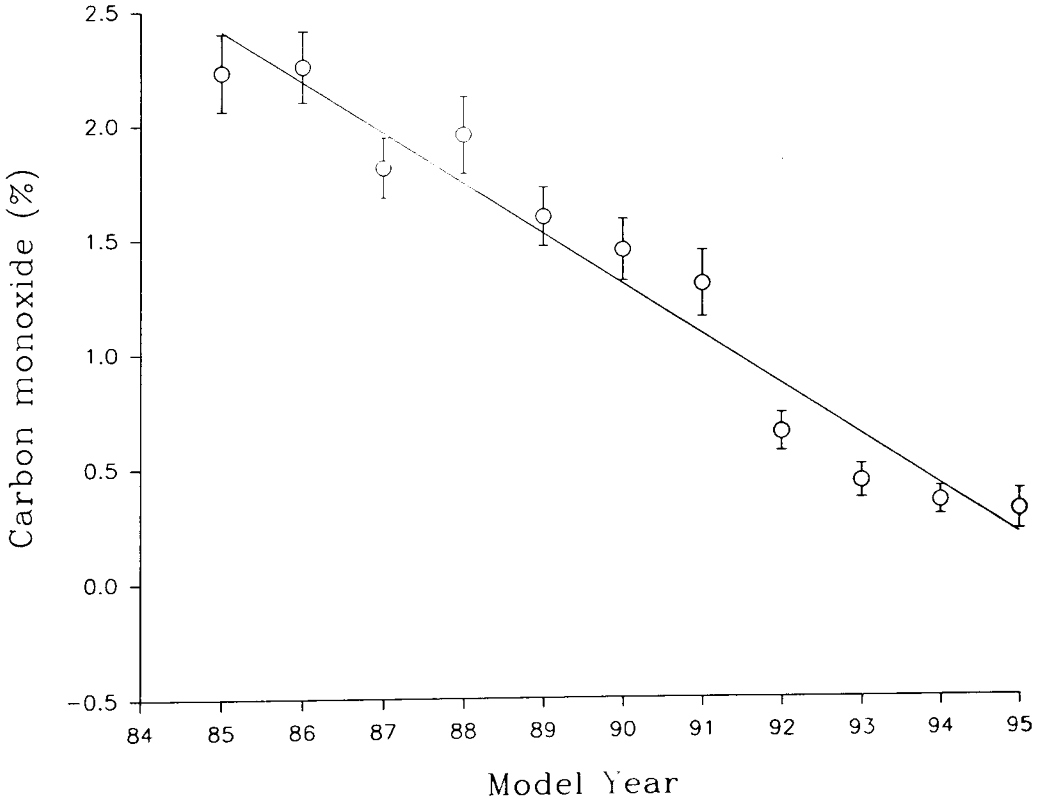


Figure 4.19 Relationship between mean annual carbon monoxide emissions, with error bars calculated using a 95% confidence interval, and model years, 1985-1995, for site D (Uppingham Road, Leicester) (r^2 = 0.95).

This difference may be due to the additional catalyst equipped model years at sites C and D compared to sites A and B. These additional catalyst equipped model years increase the slope of the regression line by accentuating the contrast in emissions between the older non-catalyst equipped model years and the newer catalyst equipped vehicles such that a small increase in model year is accompanied by a large decrease in pollutant level. Catalyst technology is further discussed on page 113.

The regression equation for the relationship at site A was calculated to be:

$$CO = 12.66 - (0.131 *yr) \qquad (r^2=0.88) \qquad \text{Eq. 4.1}$$

Where: CO = predicted mean annual emission (%).

-0.131 = regression coefficient (negative in this case as emissions increase with age).

yr = model year (1983.....1993) and '83' is used to represent 1983 etc.

The regression equation for the relationship at site B was calculated to be:

$$CO = 10.3 - (0.106 *yr) \qquad (r^2=0.77) \qquad \text{Eq. 4.2}$$

Where: CO = predicted mean annual emission (%).

-0.106 = regression coefficient (negative in this case as emissions increase with age).

yr = model year (1983.....1993) and '83' is used to represent 1983 etc.

The regression equation for the relationship at site C was calculated to be:

$$CO = 21.07 - (0.223 *yr) \qquad (r^2=0.94) \qquad \text{Eq. 4.3}$$

Where: CO = predicted mean annual emission (%).

-0.223 = regression coefficient (negative in this case as emissions increase with age).

yr = model year (1985.....1995) and '85' is used to represent

1985 etc.

The regression equation for the relationship at site D was calculated to be:

$$CO = 21.16 - (0.220 *yr) \qquad (r^2=0.95) \qquad \text{Eq. 4.4}$$

Where: CO = predicted mean annual emission (%).

-0.220 = regression coefficient (negative in this case as emissions increase with age).

yr = model year (1985.....1995) and '85' is used to represent 1985 etc.

(All relationships are statistically significant at a 99% level)

Table 4.3 shows the observed and predicted mean carbon monoxide emissions by model year. It is apparent for 1993, that there is an over prediction of mean emissions. Analysis of model year 1993, shows site C to have the largest negative residual with a value of -0.31%. Sites A, B and D had similar negative residuals with values of -0.27%, -0.29% and -0.26% respectively.

Since January 1993 all new cars in the UK have had to meet European Union emission standards set out in EC Directive 91/441/EEC. This effectively mandated the fitting of Three Way Catalysts (TWC) to new petrol cars to meet these standards. Therefore, a proportion of the carbon monoxide emissions not explained by the regression equations can be attributed to the introduction of catalyst technology to the United Kingdom fleet. Mean emissions for model year 1992 are also lower than predicted and are explained by the early introduction of some catalyst equipped vehicles ahead of Directive 91/41/EEC.

Analysis of model year 1994 at sites C and D also shows the effects of catalyst introduction with an over prediction of mean emissions. Analysis of model year 1995 at sites C and D however, shows an under prediction of mean emissions giving positive residuals of 0.18% and 0.06% respectively. This under prediction occurs because the steep decline in emissions from older non-catalyst equipped vehicles to new catalyst equipped vehicles is not sustainable. Due to all post 1992 model year vehicles having catalysts there is a plateauing of emissions at a very low base level and not a reduction, hence the under prediction of emissions.

It is important to note that EC Directive 91/441/EEC is only the latest of many vehicle emission regulations beginning with EC Directive 70/220/EEC (instituted to take measures against air pollution by gases from positive ignition engines of motor vehicles) for which the effect has been to reduce emissions from motor vehicles over time.

Table 4.3 Observed and predicted mean carbon monoxide emissions (%) by model year

Model Year	Site A		Site B		Site C		Site D	
	observed emissions	predicted emission	observed emissions	predicted emission	observed emissions	predicted emission	observed emissions	predicted emission
1983	1.70	1.79	1.30	1.50	-	-	-	-
1984	1.33	1.66	1.22	1.40	-	-	-	-
1985	1.60	1.53	1.19	1.29	1.89	2.12	2.23	2.46
1986	1.31	1.39	1.44	1.18	1.76	1.89	2.26	2.24
1987	1.44	1.26	1.08	1.08	1.83	1.67	1.82	2.02
1988	1.18	1.13	0.91	0.97	1.57	1.45	1.97	1.80
1989	1.02	1.00	1.03	0.87	1.49	1.22	1.61	1.58
1990	0.88	0.87	0.92	0.76	1.05	1.00	1.46	1.36
1991	0.86	0.74	0.75	0.65	0.77	0.78	1.31	1.14
1992	0.44	0.61	0.34	0.55	0.42	0.55	0.66	0.92
1993*	0.21	0.48	0.15	0.44	0.02	0.33	0.44	0.70
1994*	-	-	-	-	0.02	0.11	0.36	0.48
1995*	-	-	-	-	0.06	-0.12	0.32	0.26

* mandatory fitting of catalysts.

4.5.3 Model year contribution to carbon monoxide fleet emissions

A popular misconception is that old vehicles are highly polluting and must therefore contribute significantly to fleet emissions. This is not wholly incorrect in that emissions do increase with age as shown in *Figures 4.16, 4.17, 4.18 & 4.19* although this does not mean that they fall in the category of high emitters with similar findings being reported by *Anilovich & Hakkert (1996)*.

However, older vehicles do not contribute significantly to fleet emissions with the primary reason being the small number of old vehicles on the road. *Holman (1991)* has reported that most old vehicles are low emitters with more than 70% of the measured vehicles belonging to model year 1975 or earlier having exhaust CO concentrations of less than 1%. Analysis of vehicle numbers concerning site A fleet emissions shows that 949 pre-1983 registered vehicles (13% of the fleet) contributed only 21% of total fleet emissions. Site B fleet emissions demonstrate a similar scenario with 310 pre-1983 registered vehicles (5% of the fleet) contributing only 9% to fleet emissions. Scrutiny of vehicle numbers and fleet emissions at sites C and D displays a similar pattern with only 11% and 15%, respectively, of total fleet emissions contributed by pre-1983 manufactured vehicles.

A further misconception is that new vehicles are low emitters and therefore do not contribute significantly to total fleet emissions. This is not wholly incorrect in that newer vehicles do have lower mean emissions *(Figures 4.16, 4.17, 4.18 & 4.19)*. However, they do contribute significantly to total fleet emissions with 79% of fleet emissions, at site A, coming from vehicles registered during and after 1983. The emissions data for site B reveals a similar pattern with vehicles registered during and after 1983 contributing 91% of total fleet emissions. Emissions data for sites C and D show a comparable situation with 89% and 85% of total fleet emissions respectively being produced by vehicles registered during and after 1983.

However, reduction of on-road emissions by blanket targeting of this more modern age group would not necessarily produce the desired result. The vast majority of these vehicles are low emitters and it is the gross polluter component that contributes significantly to total fleet emissions and therefore needs specific targeting. A gross polluter is defined as any vehicle that falls within the top 20% of emitters (quintile 5) when the model year is rank ordered *(Figures 4.20, 4.21, 4.22 & 4.23)*. Therefore, the absolute value that defines a gross polluter changes from year to year, with a gross polluter at site C in 1988, emitting between 2.97% and 8.78% CO and in 1987 emitting between 3.26% and 11.88% CO.

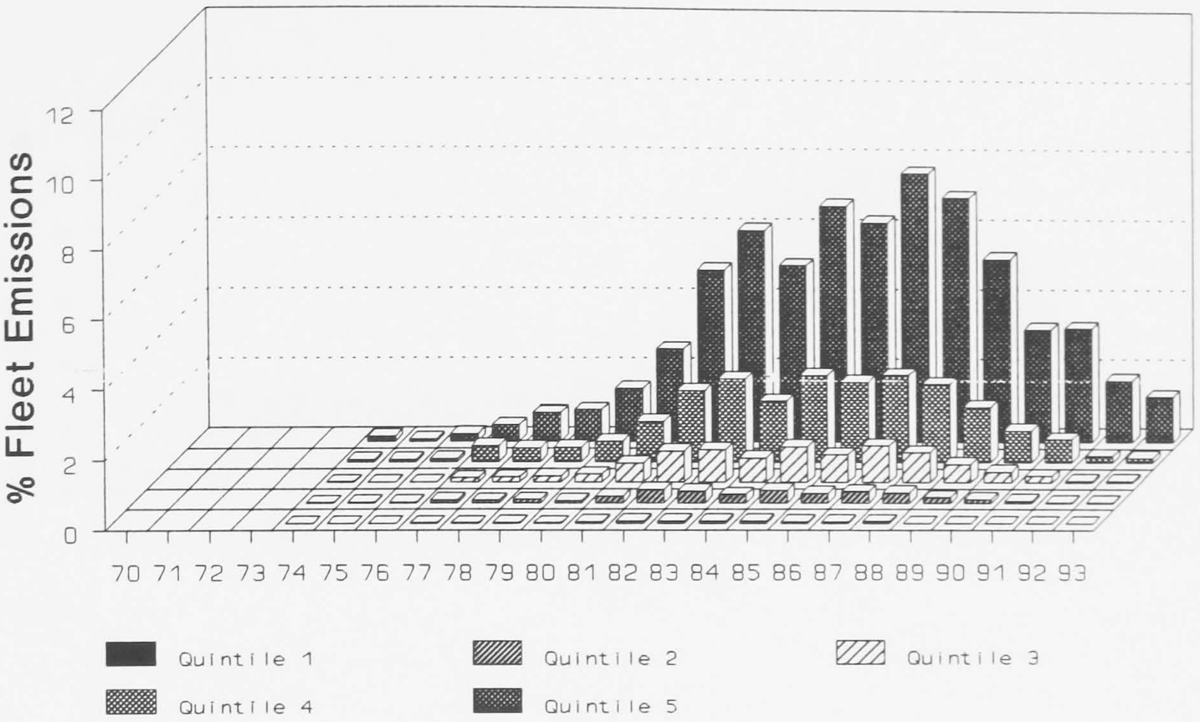


Figure 4.20 Carbon monoxide quintile emissions for site A (Bounds Green Road, Haringey).

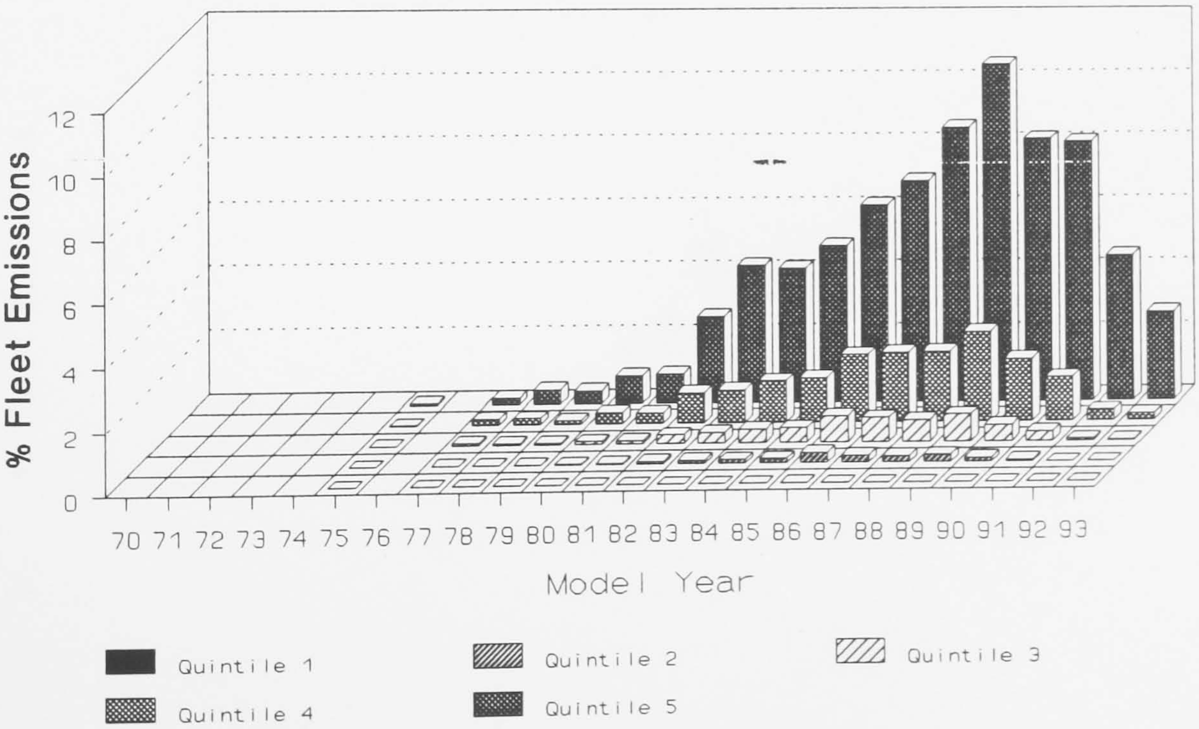


Figure 4.21 Carbon monoxide quintile emissions for site B (Dixons Bank, Middlesborough).

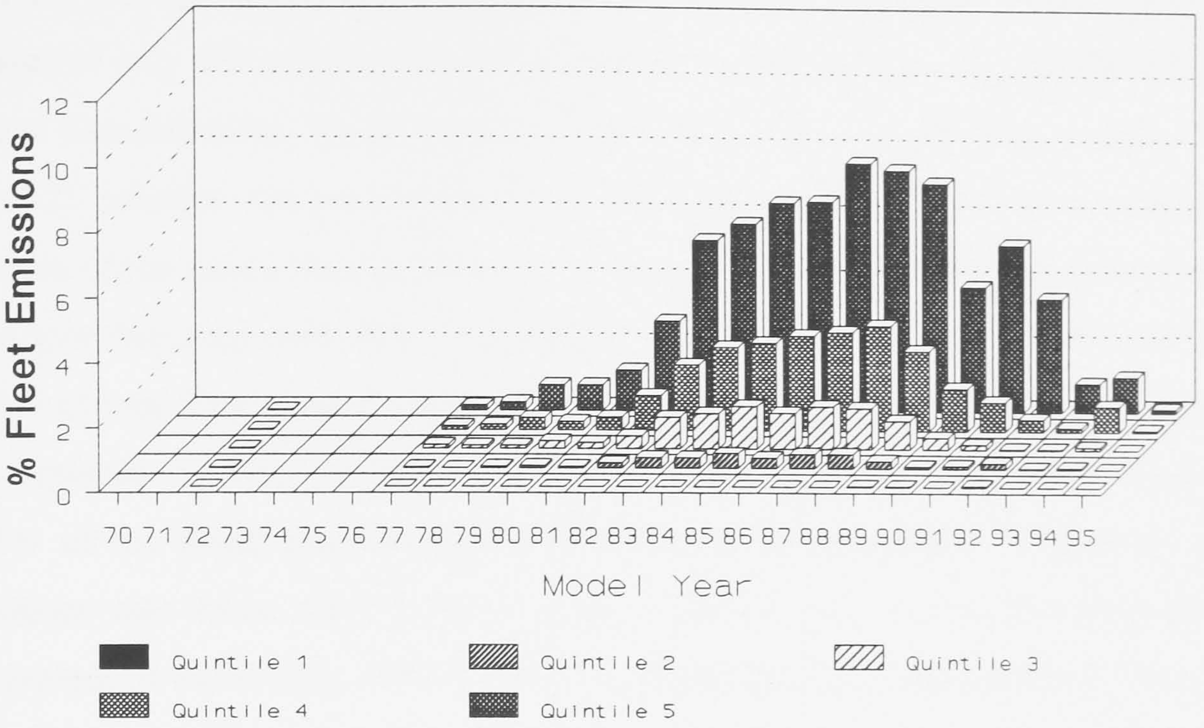


Figure 4.22 Carbon monoxide quintile emissions for site C (Abbey Street, Southwark).

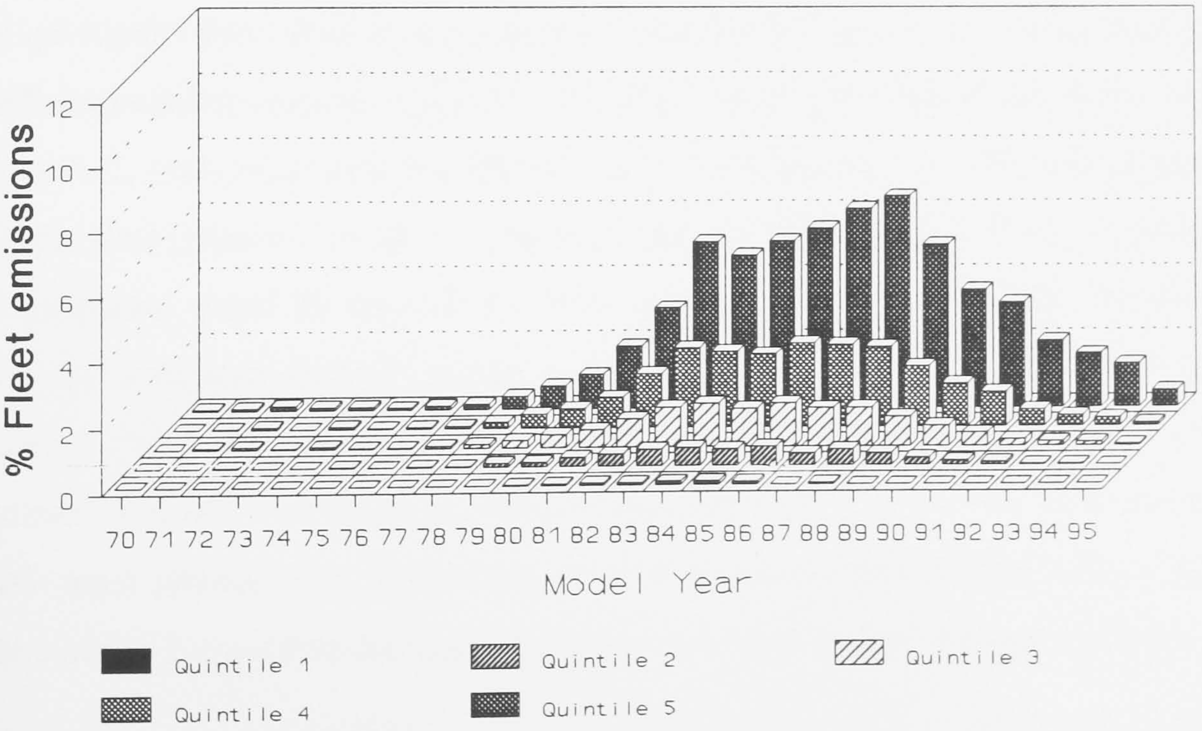


Figure 4.23 Carbon monoxide quintile emissions for site D (Uppingham Road, Leicester).

An investigation of the emissions data from site A clearly demonstrates the effect that targeting gross polluters could have. The gross polluter component of model year 1987 consists of only 147 vehicles out of a fleet total of 7397 vehicles and contributed 7.7% of total fleet emissions. Those vehicles registered during 1985 provided a similar result with 121 vehicles, the gross polluters, contributing 6.8% of total fleet emissions. An analysis of the emissions data from site B further illustrates the impact gross polluters have upon fleet emissions. The gross polluter component of model year 1989 contributed 10.6% of total fleet emissions but accounted for less than 3% of the fleet. This pattern was repeated for those vehicles registered in 1990 with the gross polluter component (1.8% of the fleet) contributing 8.2% of total fleet emissions. An analysis of the emissions data from sites C and D gives a similar picture with the gross polluter component of model year 1987 at site C (1.6% of the fleet) contributing 7.7% of total fleet emissions. This pattern is repeated for those vehicles registered in 1988 at site D, with the gross polluter component (1.8% of the fleet) contributing 6.5% of total fleet emissions.

Therefore, fleet emissions could be significantly reduced if those vehicles in quintile 5 received regular servicing or were removed from the highway by the withdrawal of their vehicle registration. An investigation of the site A data shows that if only those vehicles in quintile 5, registered between 1985 and 1991, were targeted then there is the potential to reduce fleet emissions by 40%. The same analysis at the sites B, C and D shows that fleet emissions could be reduced by 54%, 44% and 35% respectively. *Rueff (1992)* conducted a remote sensing survey to see if emission reduction was possible and concluded 'that the repair of high-emitting vehicles identified via remote sensing constitutes a viable, cost-effective means of reducing certain types of vehicle emissions'. Similar work *(Bishop et al. 1993; Cadle et al 1993)* has shown that after repairs had been carried out emissions had decreased by more than 50% for the repaired vehicles.

Older gross polluters are less important because there are so few vehicles in this category and therefore they have a small impact on fleet emissions. An analysis of emissions data from site A shows the minimal importance of older gross polluters. If those vehicles registered in 1974 are examined, it can be shown that only two vehicles were gross polluters, and they contributed only 0.15% to total fleet emissions. An analysis of emissions from the other sites shows a similar trend. If those vehicles registered in 1971 from site D are examined, it can be shown that only two vehicles were gross polluters, and they contributed only 0.07% to total fleet emissions.

The quintile emissions distributions are similar in form to those seen in other remote sensing surveys *(Guenther et al. 1994; Stedman et al 1991a; Stedman et al. 1991b)* with the greater contribution to fleet emissions coming from the gross polluters in quintile 5 with little contribution to fleet emissions from old vehicles.

One implication of the research is that fleet emissions can be greatly reduced if only a small proportion of vehicles are repaired or removed from the road. One possible way of achieving this is to employ FEAT to screen the in-use fleet for high emitting/gross polluting vehicles, which would then be subject to further test and repair. This would lower MOT costs because only those vehicles with major emissions problems would be repaired and properly functioning vehicles would not undertake a MOT emissions test. Remote screening would have the additional advantage of being performed without warning, thereby identifying vehicles that may be tampered with before and after inspection and those drivers who avoid inspections completely.

4.6 Hydrocarbon emissions

4.6.1 Fleet emissions

The measured hydrocarbon emissions, as with CO, have been analysed according to their percentage contribution to the fleet and to the total fleet emissions *(Figures 4.24, 4.25, 4.26 & 4.27)*. The hydrocarbon emissions follow a similar pattern to those of CO, with the greater majority of vehicles being low emitters and contributing little to total fleet emissions. This is displayed in *Figure 4.24* for site A (Haringey) where 69% of vehicles (HC emission range <1000 ppm) are contributing only 24% of fleet emissions. Sites B (Middlesborough), C (Southwark) and D (Leicester) produced similar results **(Table 4.4)**. Those vehicles in the emission category representing <1000 ppm hydrocarbon concentrations in Middlesborough, Southwark and Leicester, accounted for 67%, 75% and 65% of the fleets respectively but contributed only 5%, 18% and 9% respectively of fleet emissions.

Further analysis of data from site A shows that those vehicles emitting less than 2000 ppm accounted for 89% of the fleet, but contributed only 50% to total fleet emissions. Further analyses of the data from sites B, C and D produced a similar trend to that of site A with a large proportion of the fleet contributing comparatively little to fleet emissions. However, the data do vary, in that the amounts contributed to fleet emissions at sites B, C and D, from the emission category representing <2000 ppm hydrocarbon concentrations, are considerably less than for site A **(Table 4.4)**.

The converse situation is that a small minority of highly polluting vehicles (HC emission concentrations > 10000 ppm) contribute a significant proportion to fleet emissions, with 3% of the fleet at site B producing 40% of fleet emissions. Site D produced a very similar result with 4% of the fleet producing 33% of total fleet emissions. However, sites A and C although agreeing with the general trend, only had 1% of the fleet in the emission concentration range greater than 10000 ppm and this contributed 14% and 19% of the fleet emissions, respectively. This situation, although agreeing with the general pattern found for carbon monoxide emissions, differs in that a greater proportion of the total fleet emissions are contributed by the highly polluting vehicles.

The mean emissions for sites A, B, C and D are 1000 ppm, 1510 ppm, 940 ppm and 1552 ppm respectively. The corresponding median values at each site are 590 ppm, 420 ppm, 350 ppm and 450 ppm **(Table 4.4)**. The differences between the mean and median values, as was the case for carbon monoxide, reflect the very skewed nature of the emission distributions shown in *Figures 4.24, 4.25, 4.26 and 4.27*. Calculation of the skewness values for the emission distributions shows site C to have the most skewed distribution with a skewness value of 2.52. However, as with carbon monoxide similar values were found at all sites (site A, 2.17; site B, 2.28; site D, 2.29).

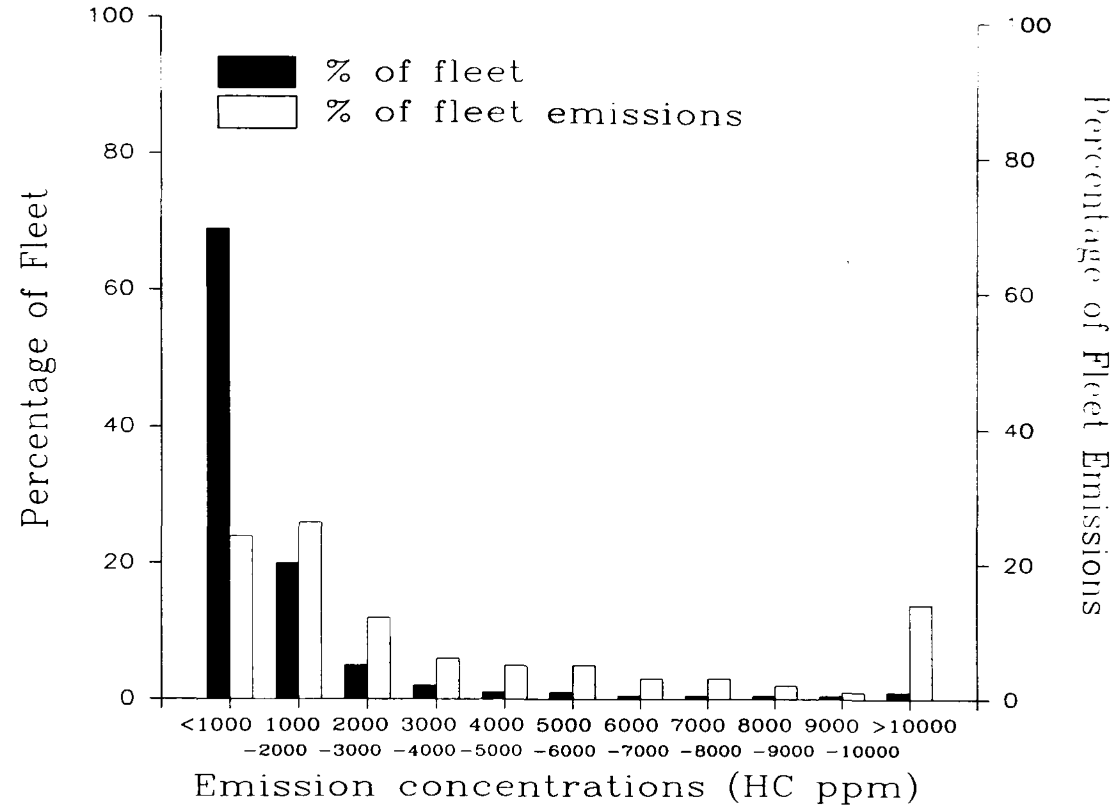


Figure 4.24 Hydrocarbon emissions for site A (Bounds Green Road, Haringey) represented as percentage contribution to the fleet and the fleet emissions.

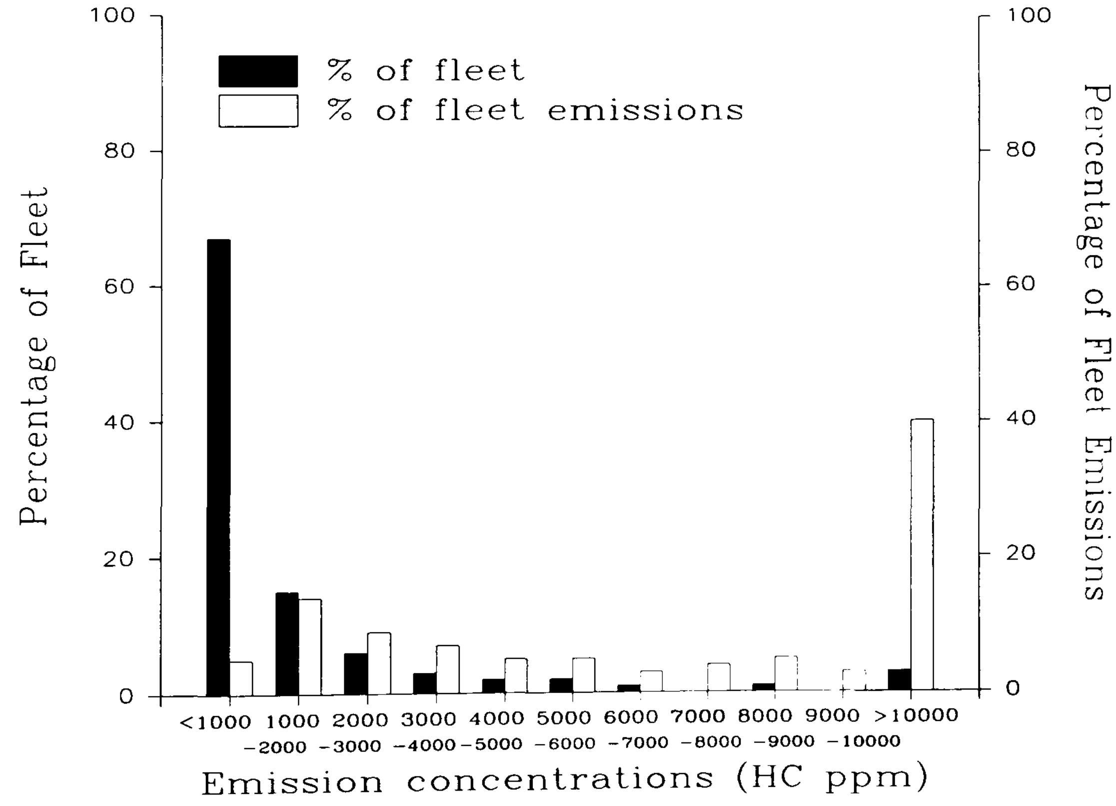


Figure 4.25 Hydrocarbon emissions for site B (Dixons Bank, Middlesborough) represented as percentage contribution to the fleet and the fleet emissions.

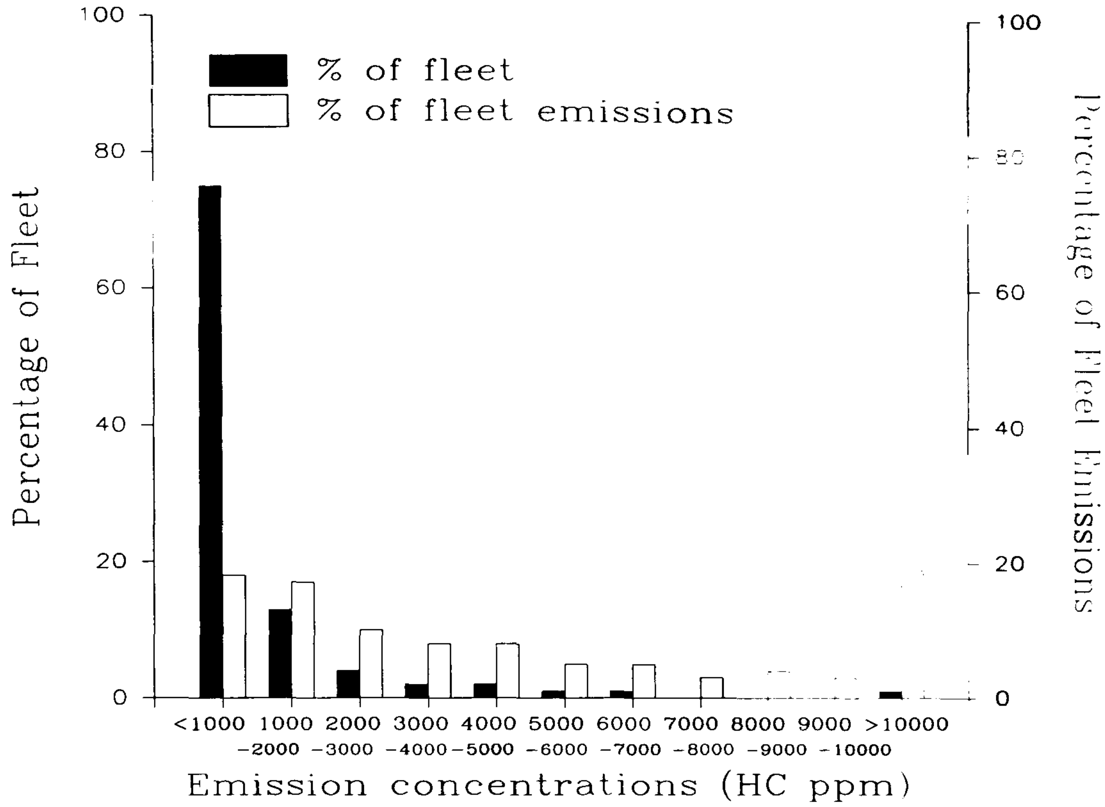


Figure 4.26 Hydrocarbon emissions for site C (Abbey Street, Southwark) represented as a percentage contribution to the fleet and the fleet emissions.

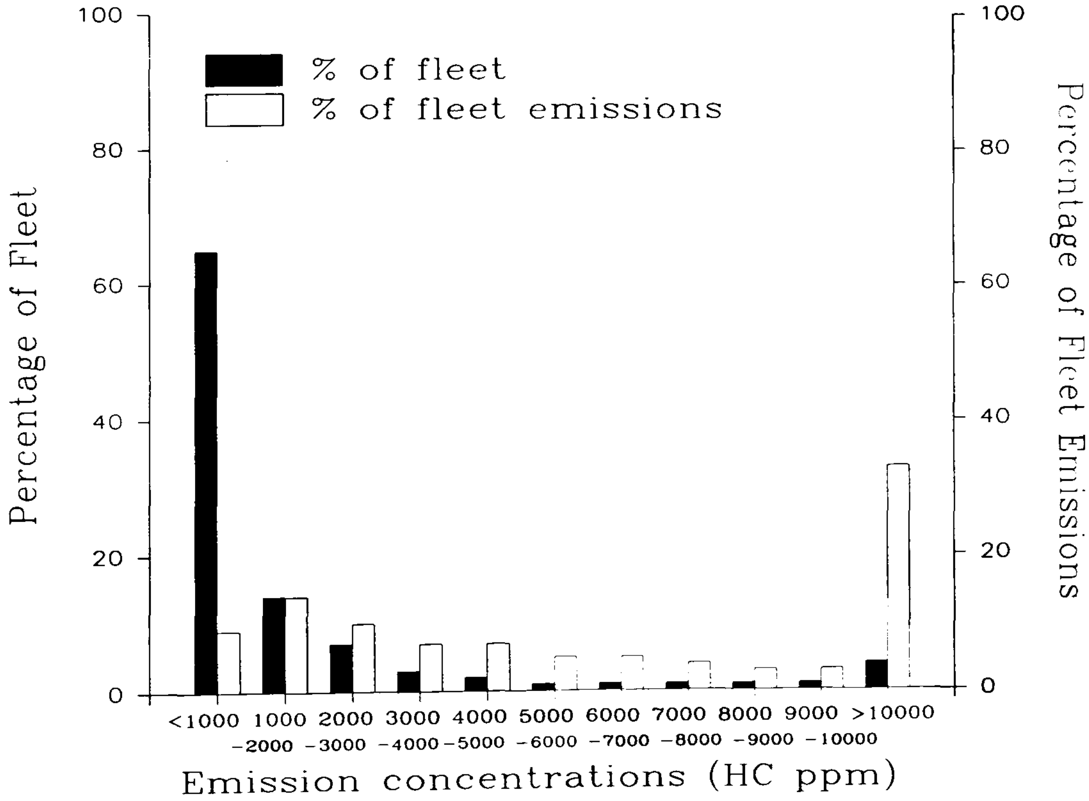


Figure 4.27 Hydrocarbon emissions for site D (Uppingham Road, Leicester) represented as percentage contribution to the fleet and the fleet emissions.

Table 4.4 Summary of hydrocarbon statistics

Site	Mean (ppm)	Median (ppm)	Emission category <1000 ppm		Emission category <2000 ppm		Emission category >10000 ppm	
			Vehicles (%)	Fleet emissions (%)	Vehicles (%)	Fleet emissions (%)	Vehicles (%)	Fleet emissions (%)
A	1000	590	69	24	89	50	1	14
B	1510	420	67	5	82	19	3	40
C	940	350	75	18	88	35	1	19
D	1552	450	65	9	79	23	4	33

The emission distributions are similar to those seen in other remote sensing studies carried out in the UK *(Muncaster et al. 1994;Vanke & Bidgood 1992)* and overseas *(Sjodin 1993; Stedman et al. 1991b).*

The mean emission level found in Leicester is lower than the 2100 ppm value recorded by *Zhang et al. (1995)* **(Table 2.3)** for the same city but similar to the 1650 ppm found by *Vanke & Bidgood (1992).* However, unlike the carbon monoxide value found by *Zhang et al.* the hydrocarbon value is within expectations although substantially higher than the other mean values found in Leicester. The 2100 ppm value placed Leicester in cluster Z *(Figure 2.17)* in the hierarchial cluster analysis performed by *Zhang et al. (1995)* along with Mexico City, Kathmandu and Bangkok. Leicester would seem to be out of place in cluster Z, where the fleets are composed of older vehicles that are less well maintained and not subject to emission control legislation or new emission standards, and if a value of 1552 ppm as found at site D or 1650 ppm as found by *Vanke & Bidgood (1992)* were used in place of 2100 ppm then a more realistic placing in cluster Y would occur *(Figure 2.17).* The fleets in cluster Y are newer than those of cluster Z, better maintained and are subject to emission control legislation.

The mean emission levels for the two London sites are similar but lower than the 1400 ppm found by *Zhang et al. (1995)* and the 1910 ppm obtained by *Vanke & Bidgood (1992).* The London fleet is placed by *Zhang et al.1995* in cluster Y *(Figure 2.17)* which agrees with the values provided by Haringey and Southwark. No comparable data is available for site B but if it had been included in the cluster analysis its mean value would place it approximately between London and Thessaloniki in cluster Y.

4.6.2 Model year hydrocarbon emissions

Mean hydrocarbon emissions have been plotted against model year to determine if there is a relationship between mean emissions and the age of the vehicle. Error bars were again calculated using a 95% confidence interval. The results initially showed a large degree of scatter *(Figures 4.28, 4.29, 4.30, & 4.31)* more so than for carbon monoxide with lower r^2 values calculated at all sites. The Leicester fleet (site D) showed the highest association between model year and mean hydrocarbon emissions returning an r^2 value of 0.26. The Haringey (site A), Middlesborough (site B) and Southwark (site C) fleets all gave r^2 values of 0.09, indicating that only 9% of the variation in mean emissions can be explained by the model year of the vehicle. A possible reason for this, as with carbon monoxide, is that the small number of older vehicles in the fleet are responsible for the low association between the variables with an increased scatter with age being apparent. As with carbon monoxide this is reflected by the larger error bounds on the older model years which are probably occurring because of the small sample sizes.

Examination of the emissions data from site A shows that out of a total of 7017 valid vehicle measurements, only 912 measurements (13% of the fleet) were recorded for those vehicles produced before 1983. An examination of site B emissions data provided a comparable situation with only 169 valid measurements (4% of the fleet) recorded for those vehicles manufactured before 1983. The emissions data from sites C and D displayed a similar pattern with 8% (from a total of 4373 vehicle measurements) and 10% (from a total of 6670 vehicle measurements) of the respective fleets registered before 1983.

Further analysis of the data from site A, shows that for 1965, 1968 and 1969 there was only one vehicle measured for each model year, with only two vehicles measured for years 1967 and 1970 (there were no data available for vehicles produced in 1964, 1966 and 1967). An example of the problem a small data set can produce is amply shown

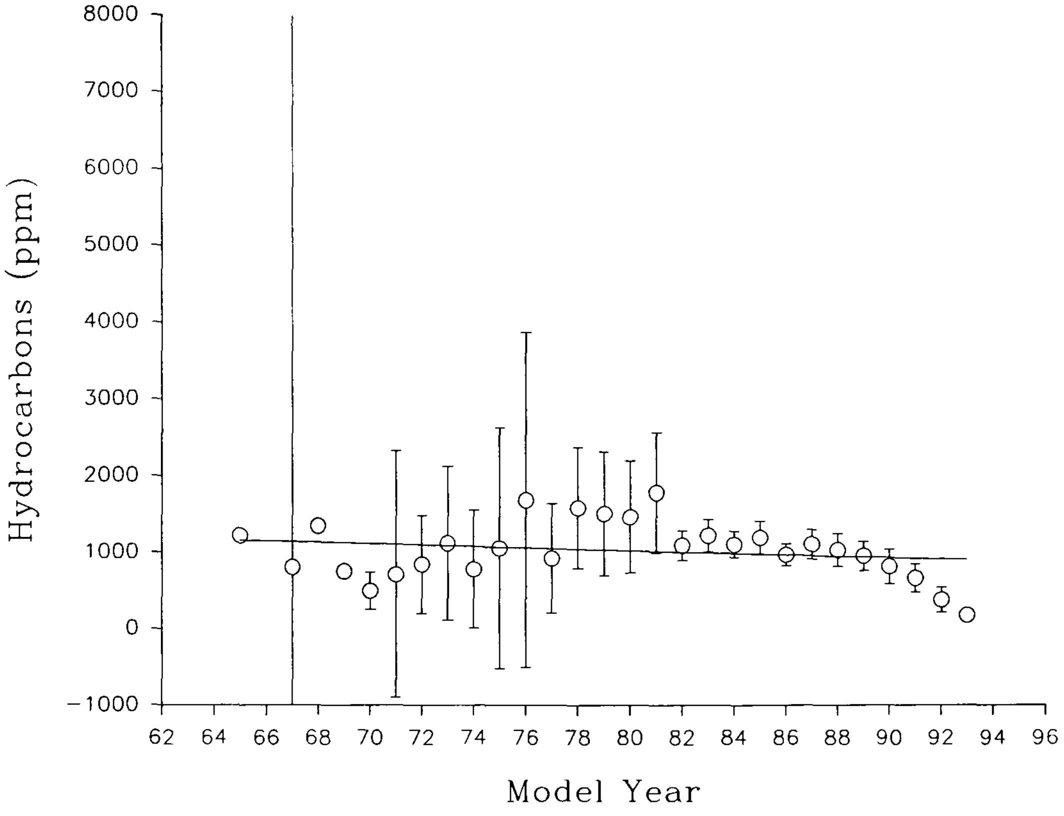


Figure 4.28 Relationship between mean annual hydrocarbon emissions, with error bars calculated using a 95% confidence interval, and model year for site A (Bounds Green Road, Haringey) (r^2 = 0.09)

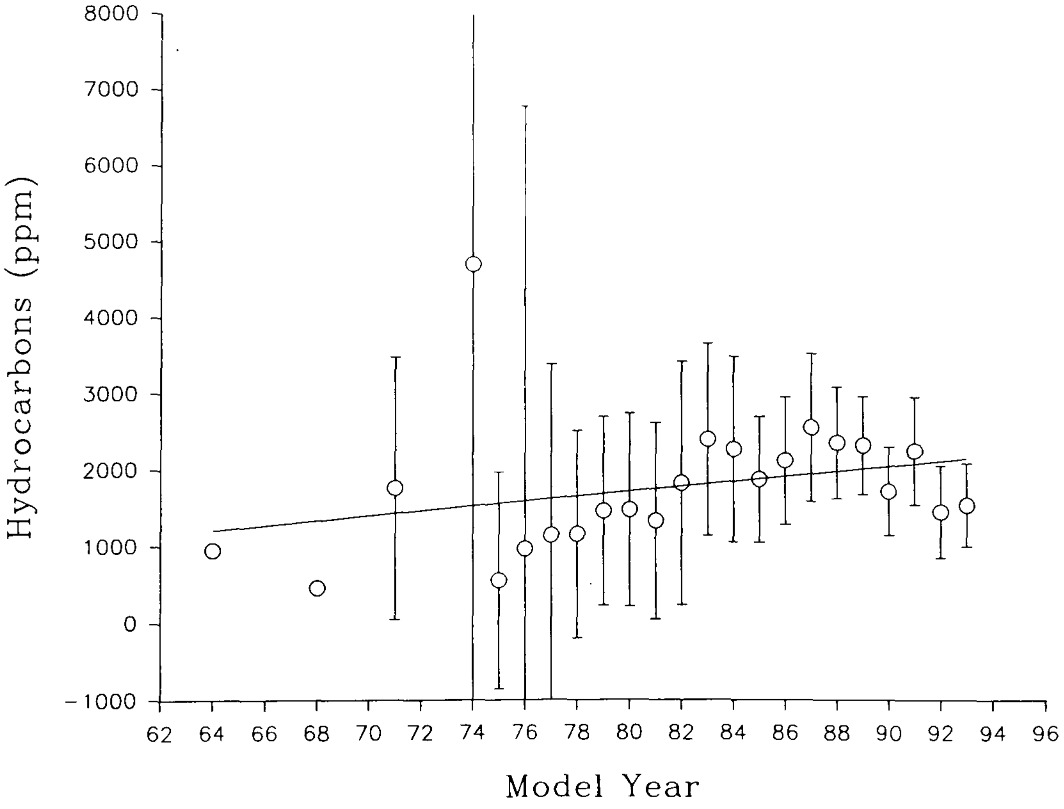


Figure 4.29 Relationship between mean annual hydrocarbon emissions, with error bars calculated using a 95% confidence interval, and model year for site B (Dixons Bank, Middlesborough) (r^2 =0.09).

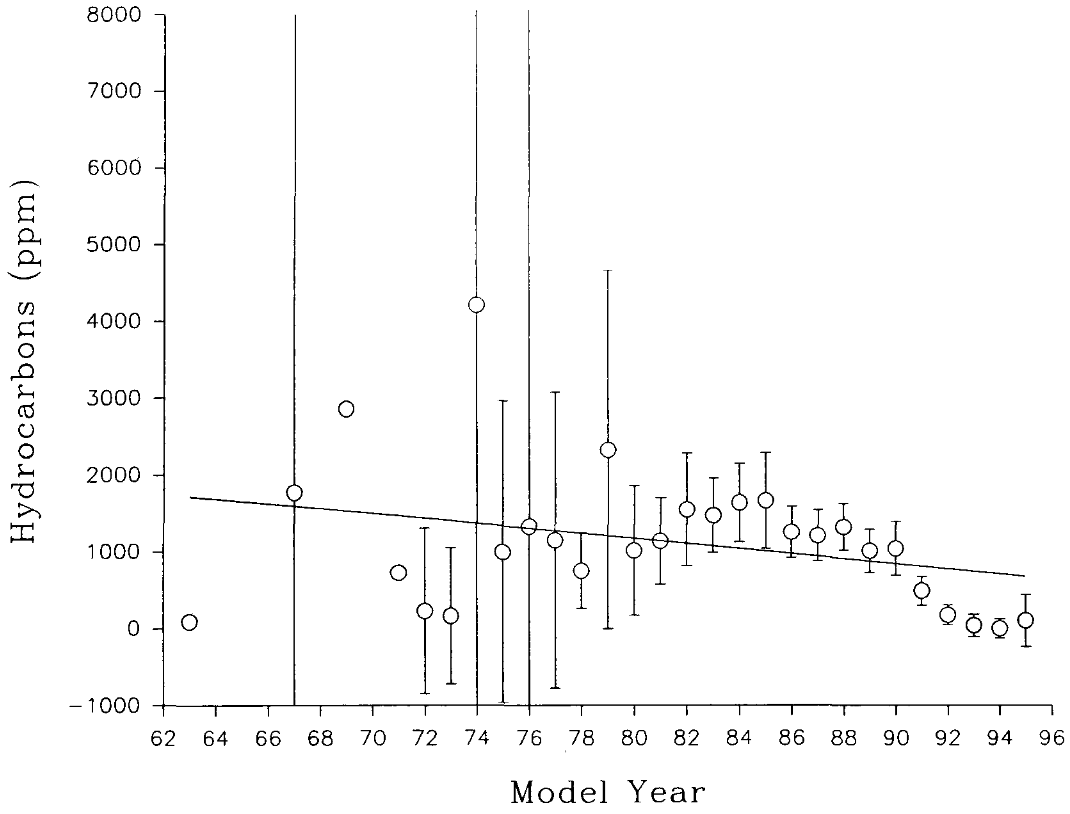


Figure 4.30 Relationship between mean annual hydrocarbon emissions, with error bars calculated using a 95% confidence interval, and model year for site C (Abbey Street, Southwark) (r^2 = 0.09)

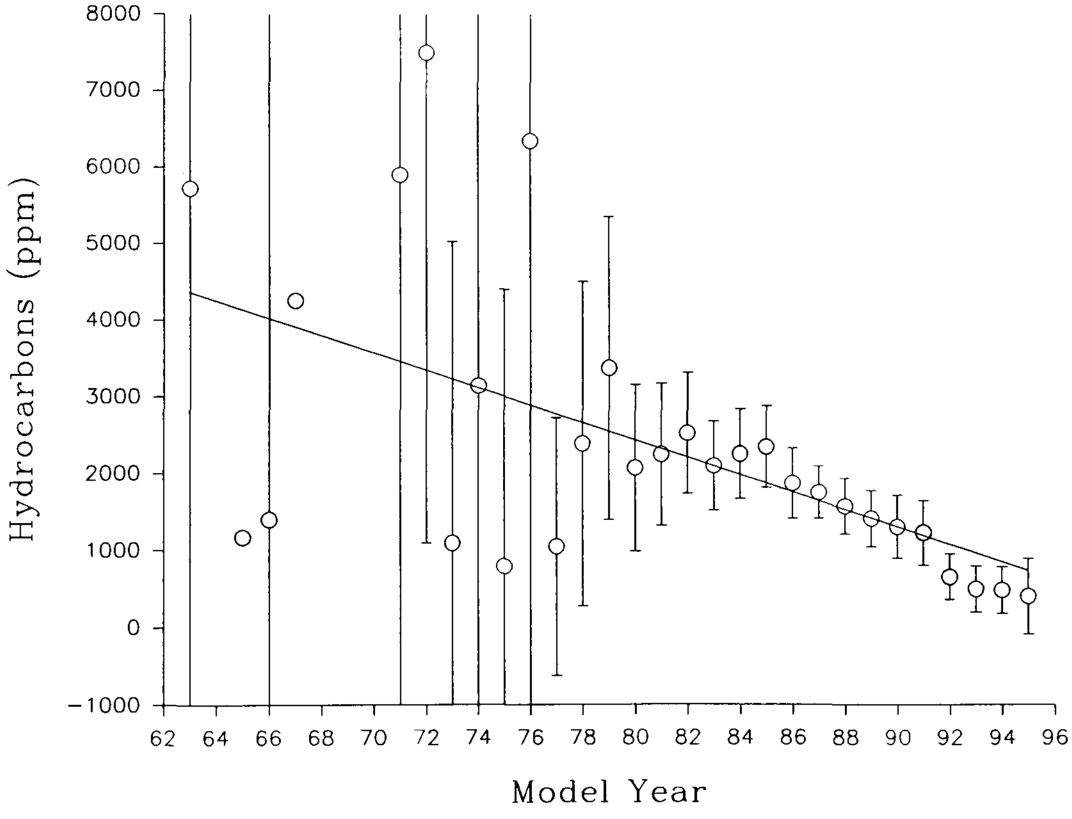


Figure 4.31 Relationship between mean annual hydrocarbon emissions, with error bars calculated using a 95% confidence interval, and model year for site D (Uppingham Road, Leicester) (r^2 = 0.26).

by the error bars for the mean value for model year 1967. The confidence limits (16630 ppm and -15020 ppm) are so large that they cannot be included on Figure 4.28 without rendering interpretation of the rest of the Figure 4.28 meaningless. This situation is repeated across all sites for model years with small sample sizes. The extreme error bounds have therefore not been included for those model years with extremely small sample sizes. The situation is repeated until 1978 at site A when the mean becomes less uncertain because the sample size increases to 56 vehicles with the associated decrease in error bounds. An additional investigation of the emissions data from site B gives similar results with only one vehicle measured in each of the following model years: 1964, 1968 and 1972. Sample size increases to 64 vehicles in 1982 increasing the confidence in the mean. There was no data available for vehicles produced in 1965, 1966, 1967, 1969, 1970 and 1973. Sites C and D had similar vehicle distributions. Thus, as with carbon monoxide, the vehicle pool is not large enough to support statistical analysis for older vehicles.

4.6.2.1 Model year emissions 1983-1995

In view of the uncertainity of the data from older vehicles and the associated large error bounds, only those vehicles registered between 1983-1993 for sites A and B and 1985-1995 for sites C and D were included in an additional investigation of the variation of hydrocarbon emissions with model year. As with carbon monoxide the larger number of vehicles sampled for the newer model years (1983-1995) leads to much smaller error bars at all sites with the exception of site B. One possible explanation for the larger error bars at site B could be the smaller sample size (4056 vehicles) in comparison to site A (6107 vehicles) and site D (6003 vehicles). However, the sample size is similar to that of site C (4023 vehicles) but the data have a much larger degree of uncertainty. A further possible explanation for the greater uncertainty at site B could be the more highly skewed emissions distribution found there (section 4.6.1) with the sample distribution at site B further from the underlying theoretical distribution than at other sites.

Due to monitoring being conducted at sites A and B approximately 15 months before sites C and D, the most recent data available for sites A and B were from model year

1993 not 1995. However, the newest ten model years were analysed from each location. Scatter plots displaying the relationships between mean hydrocarbon levels and model year are displayed in *Figures 4.32, 4.33, 4.34 and 4.35*.

As a result of omitting the older vehicles from the additional regression analyses much stronger relationships were found at all sites with the exception of site B. The Southwark and Leicester fleets, as they did for carbon monoxide, showed the greatest association between the model year of a vehicle and mean pollutant emissions returning similar r^2 values of 0.91 and 0.96, respectively. The regression equations for the relationships found at sites C and D are shown in Equations 4.7 and 4.8, respectively. Sites C and D have similar large regression coefficient values of -172.77 and -190.63, respectively. This indicates that at both sites mean pollutant emissions change in a similar pattern such that a small increase in model year is accompanied by a large decrease in pollutant level as was found for carbon monoxide.

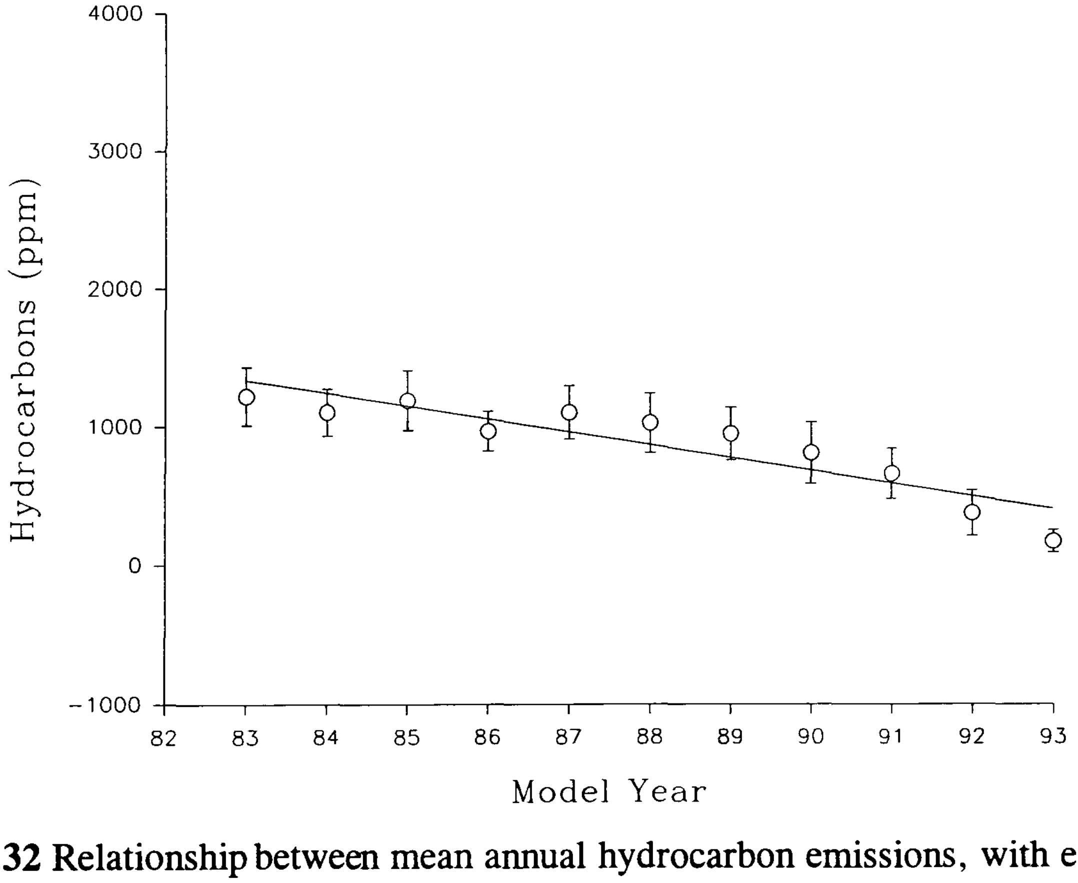


Figure 4.32 Relationship between mean annual hydrocarbon emissions, with error bars calculated using a 95% confidence interval, and model years, 1983-1993, for site A (Bounds Green Road, Haringey) (r^2 = 0.82)

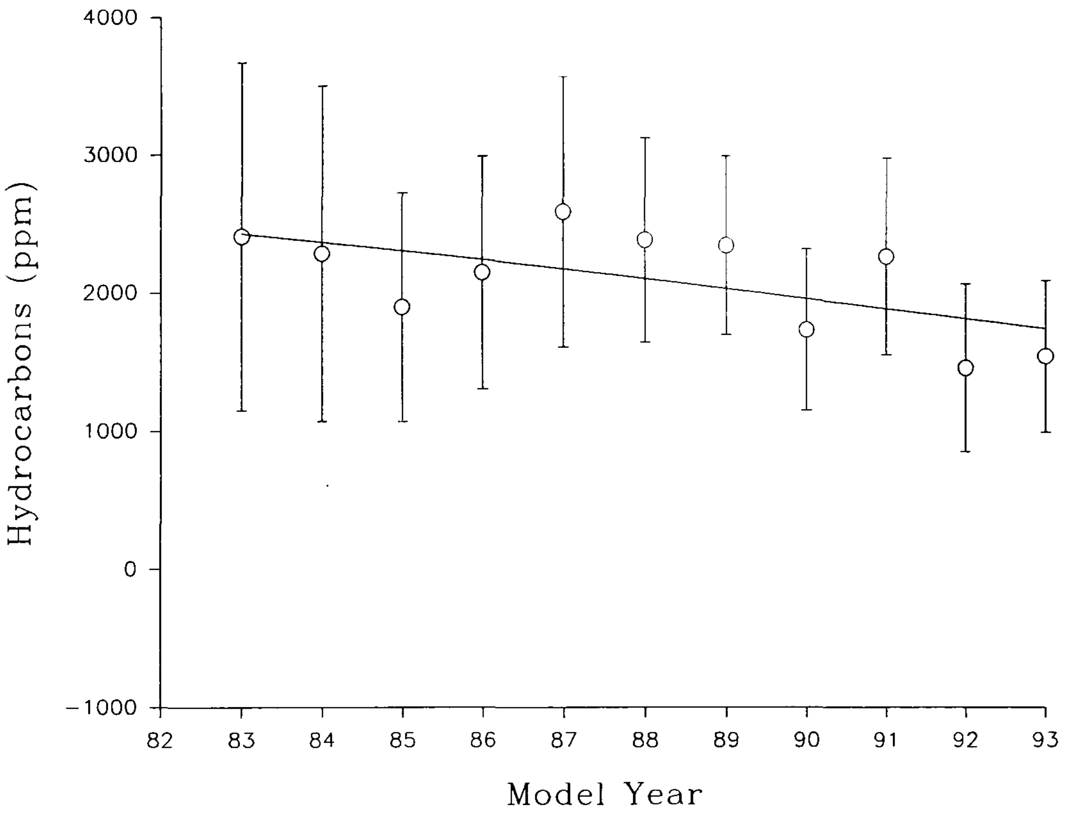


Figure 4.33 Relationship between mean annual hydrocarbon emissions, with error bars calculated using a 95% confidence interval, and model years, 1983-1993, for site B (Dixons Bank, Middlesborough) (r^2 = 0.30).

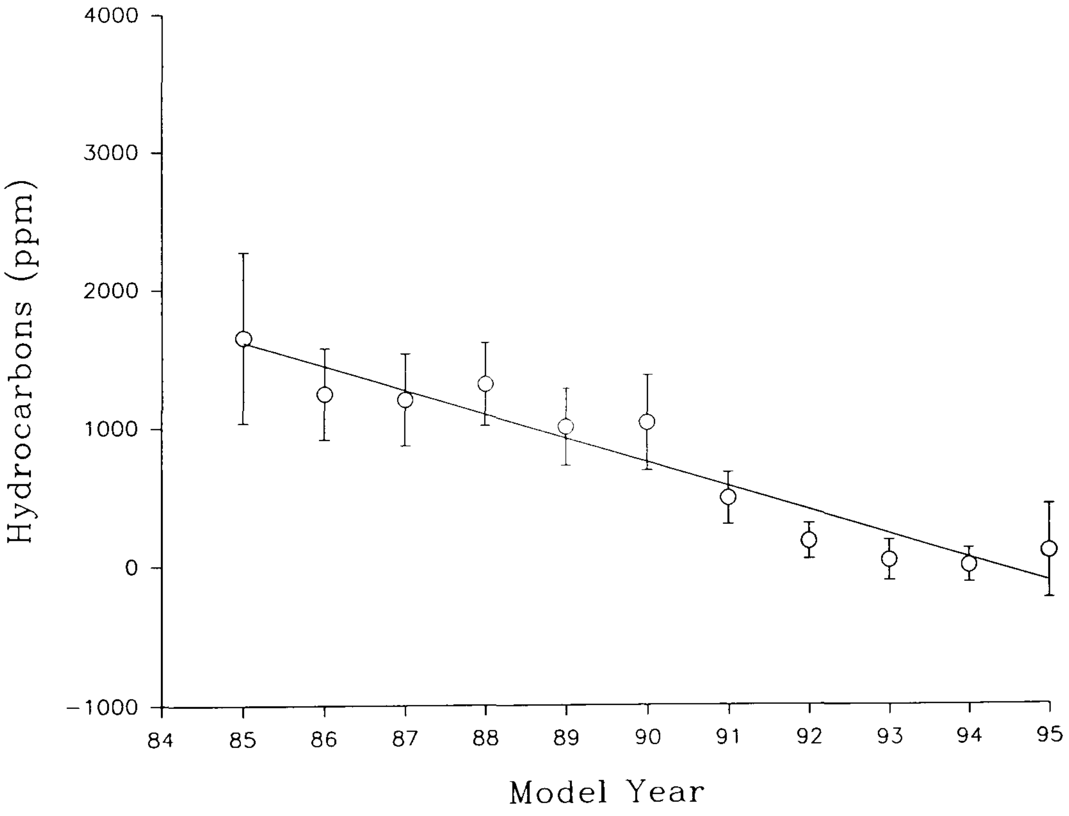


Figure 4.34 Relationship between mean annual hydrocarbon emissions, with error bars calculated using a 95% confidence interval, and model years, 1985-1995, for site C (Abbey Street, Southwark) (r^2 =0.91).

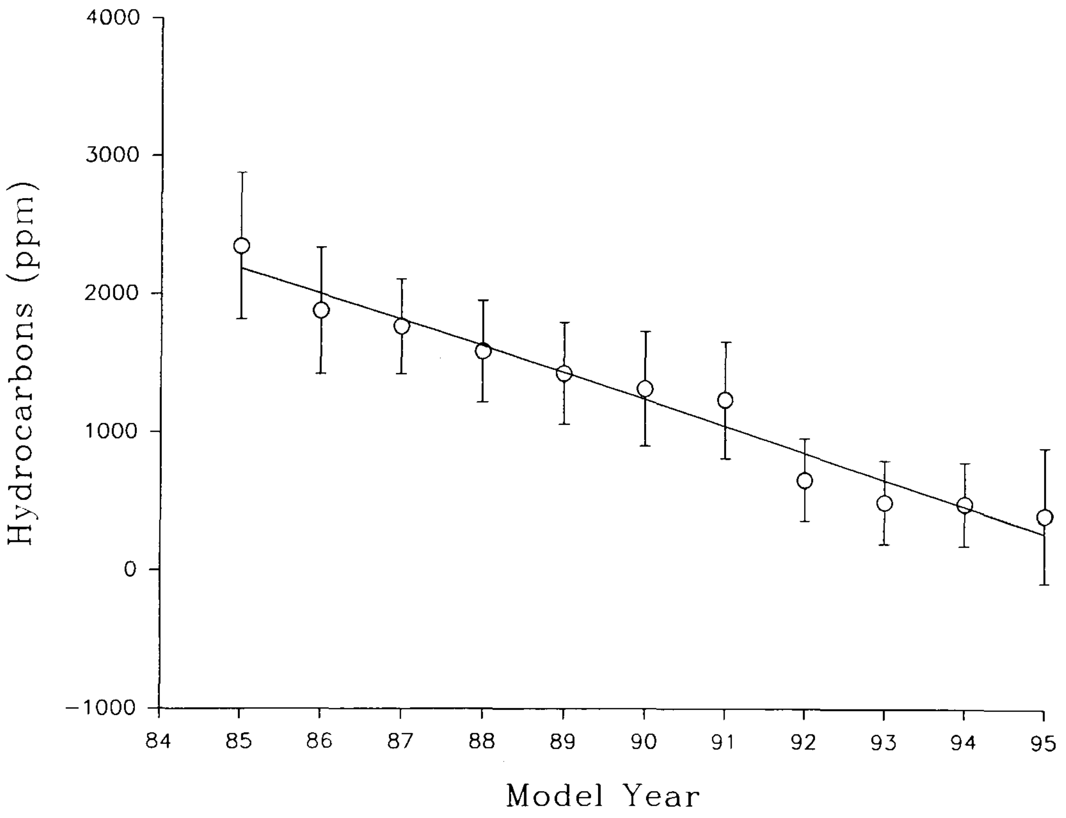


Figure 4.35 Relationship between mean annual hydrocarbon emissions, with error bars calculated using a 95% confidence interval, and model years, 1985-1995, for site D (Uppingham Road, Leicester) (r^2 = 0.96).

The relationship between the model year of a vehicle and mean pollutant emissions was only a little weaker for the Haringey fleet with an r^2 value of 0.82 which is similar to the 0.88 value found for carbon monoxide. The Middlesborough fleet however, returned an r^2 value of only 0.30 which is significantly lower than the 0.77 found for carbon monoxide at the same site. However, an explanation for this is unclear but it is probably linked to the greater uncertainty in the data from site B.

The regression equations for the relationships found at sites A and B are shown in Equations 4.5 and 4.6, respectively. These sites have similar regression coefficient figures of -92.52 and -68.8, respectively. This indicates that at both sites mean pollutant emissions change in a similar pattern. However, whereas at sites C and D a small increase in model year was accompanied by a large decrease in pollutant level, at sites A and B because the regression coefficients are smaller and the slope of the regression line shallower, a larger increase in model year is required for same decrease in pollutant level. As for CO (section 4.5.2.1) this difference may be due to the additional catalyst equipped model years at sites C and D compared to sites A and B and the influence they exert on the slope of the regression line.

The regression equation for the relationship at site A was calculated to be:

$$HC = 9018 - (92.52 * yr) \qquad (r^2=0.82) \qquad \text{Eq. 4.5}$$

Where: HC = mean annual emission (ppm).

-92.52 = regression coefficient (negative in this case as emissions increase with age).

yr = model year (83.....93) and '83' is used to represent 1983 etc.

The regression equation for the relationship at site B was calculated to be:

$$HC = 8133 - (68.8 * yr) \qquad (r^2=0.30) \qquad \text{Eq. 4.6}$$

Where: HC = mean annual emission (ppm).

-68.8 = regression coefficient (negative in this case as emissions increase with age).

yr = model year (83.....93) and '83' is used to represent 1983 etc.

The regression equation for the relationship at site C was calculated to be:

$$HC = 16304 - (172.77 * yr) \qquad (r^2=0.91) \qquad \text{Eq. 4.7}$$

Where: HC = mean annual emission (ppm).

-172.77 = regression coefficient (negative in this case as emissions increase with age).

yr = model year (85.....95) and '85' is used to represent 1985 etc.

The regression equation for the relationship at site D was calculated to be:

$$HC = 18386 - (190.63 * yr) \qquad (r^2=0.96) \qquad \text{Eq. 4.8}$$

Where: HC = mean annual emission (ppm).

-190.63 = regression coefficient (negative in this case as emissions increase with age).

yr = model year (85.....95) and '85' is used to represent 1985 etc.

(All relationships are statistically significant at a 99% level except for site B which is statistically significant at a 95% level)

Table 4.5 shows the observed and predicted mean hydrocarbon emissions for different model years. It is apparent, as it was for carbon monoxide, that there is an over prediction of mean emissions for model year 1993.The analysis for this year, shows site A to have the largest negative residual with a value of -237 ppm. Sites B, C, and D had similar negative residuals with values of -196 ppm, -103 ppm and -159 ppm respectively. This difference between the observed hydrocarbon emission values and the predicted emission values, as with carbon monoxide, is attributable to EC Directive 91/441/EEC introduced in January 1993.

The Directive effectively mandated the fitting of Three Way Catalysts (TWC) to meet new emission standards. Therefore, a proportion of the hydrocarbon emissions not explained by the regression equations can be attributed to the introduction of catalyst technology to the United Kingdom fleet. Mean emissions for model year 1992 **(Table 4.5)** are also lower than predicted, and are explained by the early introduction of some catalyst equipped vehicles ahead of Directive 91/441/EEC.

Analysis of model year 1994 at site C also shows the effects of catalyst introduction with an over prediction of mean emissions. However, analysis of model year 1995 at site C and of model years 1994 and 1995 at site D shows an under prediction of mean emissions giving positive residuals of 215 ppm, 19 ppm and 134 ppm, respectively. This under prediction occurs because the steep decline in emissions from older non-catalyst equipped vehicles to new catalyst equipped vehicles is not sustainable. Due to all post 1992 model year vehicles having catalysts there is a plateauing of emissions at a very low base level and not a reduction, hence the under prediction of emissions.

As with carbon monoxide it is important to note that EC Directive 91/441/EEC is only the latest of many vehicle emission regulations beginning with EC Directive 70/220/EEC (instituted to take measures against air pollution by gases from positive ignition engines of motor vehicles) for which the effect has been to reduce emissions from motor vehicles over time.

Table 4.5 Observed and predicted mean hydrocarbon emissions (ppm) from model year

Model Year	Site A		Site B		Site C		Site D	
	observed emissions	predicted emission	observed emissions	predicted emission	observed emissions	predicted emission	observed emissions	predicted emission
1983	1222	1339	2406	2423	-	-	-	-
1984	1106	1246	2276	2354	-	-	-	-
1985	1195	1154	1881	2285	1655	1619	2342	2182
1986	971	1061	2126	2216	1248	1446	1869	1992
1987	1109	967	2562	2147	1206	1273	1751	1801
1988	1033	876	2360	2079	1319	1100	1571	1611
1989	955	784	2324	2010	1010	927	1412	1420
1990	816	691	1723	1941	1040	755	1304	1229
1991	666	599	2251	1872	492	582	1232	1039
1992	381	506	1453	1803	179	409	652	848
1993*	177	414	1539	1735	43	236	498	657
1994*	-	-	-	-	4	64	486	467
1995*	-	-	-	-	106	-109	410	276

* mandatory fitting of catalysts.

4.6.3 Model year contribution to fleet hydrocarbon emissions

Older vehicles do not contribute significantly to fleet emissions due to the the small number of old vehicles on the road. Analysis of vehicle numbers and fleet emissions at site A shows that 1405 pre-1983 registered vehicles contribute only 18% of total fleet emissions. Site B fleet emissions demonstrate a similar scenario with 169, pre-1983 registered vehicles (4% of the fleet) contributing only 5% to fleet emissions. Scrutiny of vehicle numbers and fleet emissions at sites C and D displays a similar pattern with only 12% and 16% respectively, of total fleet emissions, contributed by pre-1983 manufactured vehicles.

New vehicles contribute significantly to total fleet emissions. Analysis of vehicle numbers in relation to fleet emissions at site A shows that 5610 vehicles (80% of the fleet) registered during and after 1983 contribute 82% of total fleet emissions. Site B emissions data reveals a similar pattern with vehicles registered during and after 1983 contributing 95% of total fleet emissions. Emissions data for sites C and D reveals a similar pattern with 88% and 84% respectively of total fleet emissions being produced by vehicles registered during and after 1983.

However, reduction of on road emissions by blanket targeting of this more modern age group would not necessarily produce the desired result. The vast majority of these vehicles are low emitters and it is the gross polluter component that contributes significantly to total fleet emissions and therefore needs specific targeting. A gross polluter as for carbon monoxide is defined as any vehicle that falls within the top 20% of emitters (quintile 5) when the model year is rank ordered *(Figures 4.36, 4.37, 4.38 & 4.39)*.

An analysis of the emissions data from site A illustrates this clearly. The gross polluter component of model year 1987, a mere 139 vehicles out of a fleet total of 7017 vehicles, contributed 7.7% of total fleet emissions. This pattern was repeated for those vehicles registered in 1988 with the gross polluter component (152 vehicles) contributing 8.1% of total fleet emissions. An analysis of the emissions data from site B further illustrates the impact gross polluters have on fleet emissions. The gross polluter component of model year 1989 contributed 10.7% of total fleet emissions but accounted for just 2.3% of the fleet. This pattern was repeated for those vehicles registered in 1990 with the gross polluter component (2% of the fleet) contributing 9.7% of total fleet emissions. An analysis of the emissions data from site C and D gives a similar picture with the gross polluter component of model year 1988 at site C (1.9% of the fleet) contributing 9.5% of total fleet emissions. This pattern was repeated for those vehicles registered in 1985 at site D, with the gross polluter component (1.5% of the fleet) contributing 8.9% of total fleet emissions.

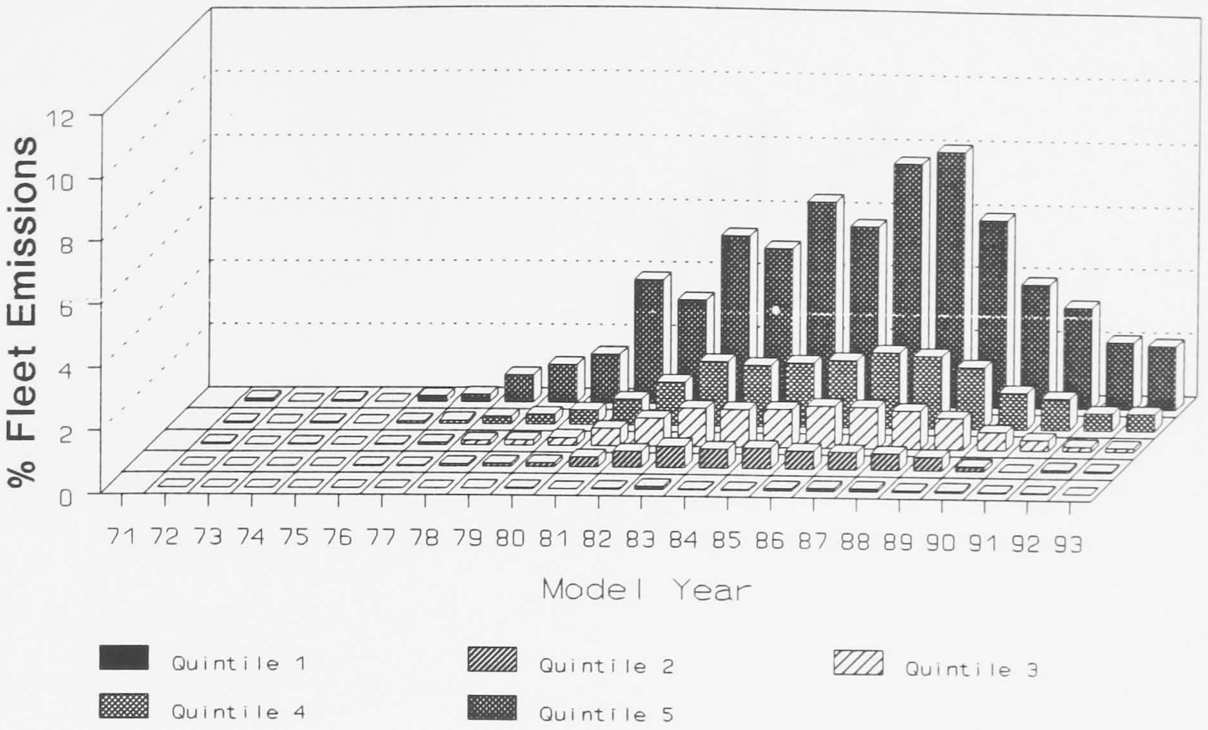


Figure 4.36 Hydrocarbon quintile emissions for site A (Bounds Green Road, Haringey).

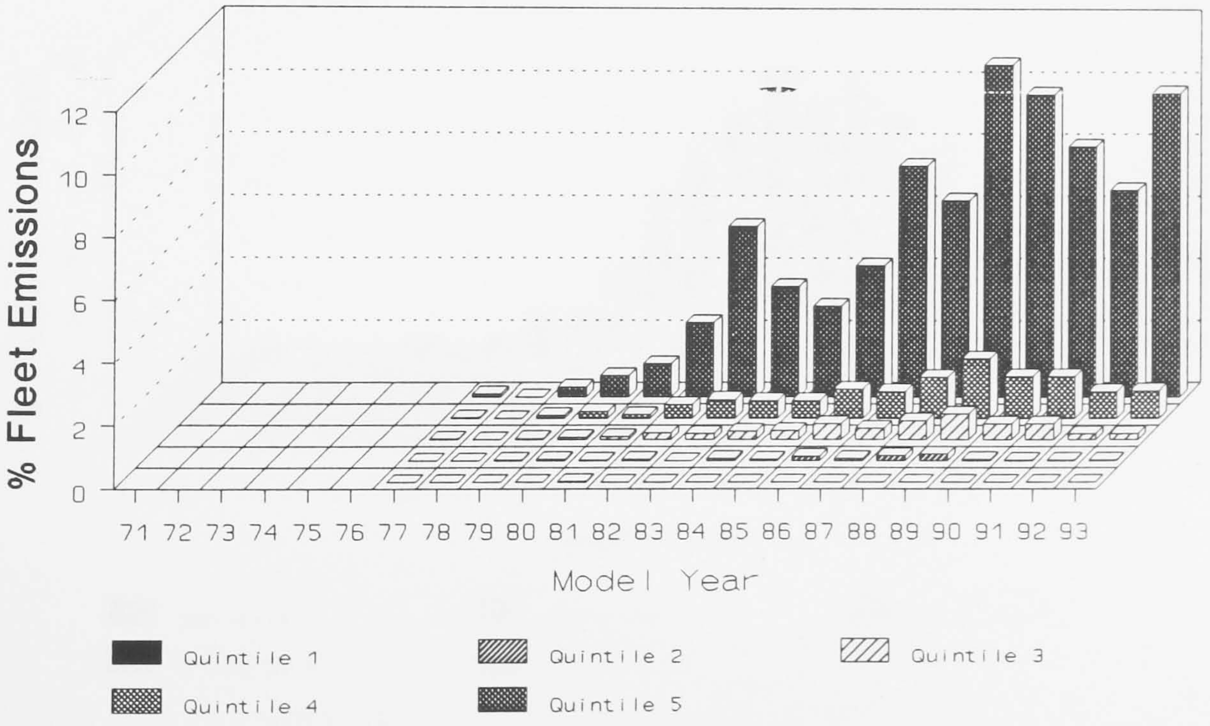


Figure 4.37 Hydrocarbon quintile emissions for site B (Dixons Bank, Middlesborough).

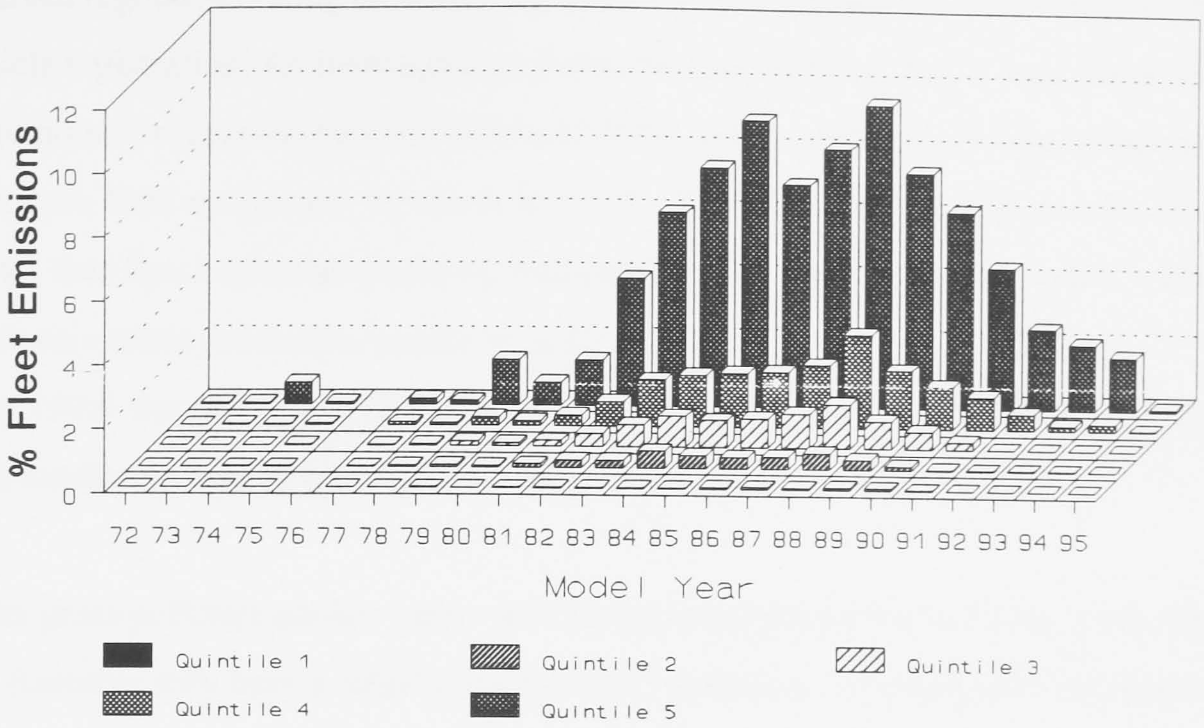


Figure 4.38 Hydrocarbon quintile emissions for site C (Abbey Street, Southwark).

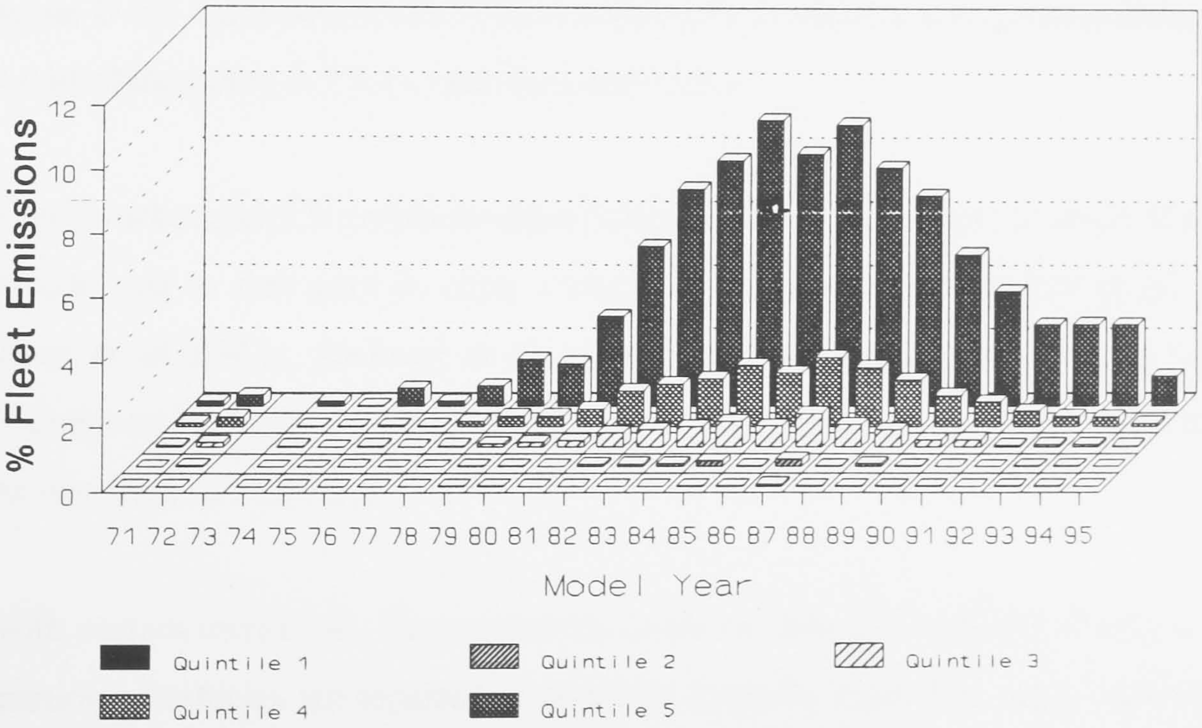


Figure 4.39 Hydrocarbon quintile emissions for site D (Uppingham Road, Leicester).

Therefore, fleet emissions could be significantly reduced if those vehicles in quintile 5 received regular servicing or were removed from the highway by the withdrawal of their vehicle registration. An investigation of the site A data shows that if only those vehicles in quintile 5, registered between 1985 and 1991, were targeted then there is the potential to reduce fleet emissions at site A by 41%. The same analysis at sites B, C and D shows that fleet emissions could be reduced by 66%, 51% and 48% respectively. The work on carbon monoxide emissions reduction *(Rueff 1992; Bishop et al. 1993; Cadle et al.1993)* discussed in section 4.5.3 was also undertaken for hydrocarbon emissions with similar emissions reductions noted.

Older gross polluters are less important because there are so few vehicles in this category and therefore they have a small impact on fleet emissions. An analysis of emissions from site A shows the minimal importance of older gross polluters *(Figure 4.36)*. If those vehicles registered in 1976 are examined, it can be shown, that only 2 vehicles were gross polluters, and they contributed only 0.2% to total fleet emissions. An analysis of emissions from the other sites shows a similar trend. If those vehicles registered in 1974 from site D are examined, it can be shown that only 2 vehicles were gross polluters, and they contributed only 0.2% to total fleet emissions.

The hydrocarbon quintile emissions distributions are similar in form to those of carbon monoxide and to that seen in other remote sensing surveys *(Guenther et al. 1994; Stedman et al 1991a; Stedman et al. 1991b)* with the greater contribution to fleet emissions coming from the gross polluters in quintile 5 with little contribution to fleet emissions from old vehicles.

As with carbon monoxide, fleet emissions could be reduced markedly if only a small proportion of vehicles are repaired or removed from the road. The steps suggested for the reduction of carbon monoxide fleet emissions could be equally applied to hydrocarbons with similar results.

4.7 Summary

Emission summaries for carbon monoxide and hydrocarbons are provided in **Tables 4.6 and 4.7**. It is clear that there is a large majority of low emitting vehicles contributing little to fleet emissions at all sites and for both pollutants. The converse is also true with a small minority of high emitting vehicles contributing significant proportions to fleet emissions at all sites and for both pollutants. However, the small minority of high emitting hydrocarbon vehicles in general contributes an even greater proportion to fleet emissions than does the comparable minority of carbon monoxide high emitters.

Table 4.6 Statistical interpretation of carbon monoxide emissions

Location	Mean (%)	Median (%)	High Emitters (dirtiest 10%)		Low Emitters (cleanest 70%)	
			% of fleet emissions	Mean vehicle conc (%)	% of total emissions	Mean vehicle conc (%)
Site A	1.26	0.27	57	7.2	12	0.2
Site B	0.84	0.14	59	4.9	8	0.1
Site C	1.46	0.35	50	7.5	13	0.3
Site D	1.63	0.53	50	8.2	16	0.4

Table 4.7 Statistical interpretation of hydrocarbon emissions

Location	Mean (ppm)	Median (ppm)	High Emitters (dirtiest 10%)		Low Emitters (cleanest 70%)	
			% of fleet emissions	Mean vehicle conc (ppm)	% of total emissions	Mean vehicle conc (ppm)
Site A	1000	590	48	5000	25	400
Site B	1510	420	67	10200	7	150
Site C	940	350	59	6000	14	200
Site D	1552	450	62	10420	11	260

A pattern of high mean values and lower median values is consistently observed at all locations and for both pollutants. This occurs because mean emissions are dominated by a small number of high emitters which skew the data to the right (positive skew). Thus, the median which is not biased by the high emitters returns a much lower value. Work conducted with remote sensing data sets and laboratory dynamometer emission measurements by *Zhang, Bishop and Stedman (1994)* showed a similar result, with carbon monoxide and hydrocarbon emissions well represented by a γ-distribution.

Analysis of model year data suggests a low association between vehicle age and mean emissions for vehicles produced prior to 1983, at all sites and for both pollutants. The relationship improves after 1983 with regression analyses returning average r^2 values of 0.89 for carbon monoxide and 0.75 for hydrocarbons.

Analysis of model year contribution to fleet emissions shows relatively new gross polluting vehicles to be the greatest contributors to fleet emissions with on average 43% of carbon monoxide and 52% of hydrocarbon fleet emissions produced by the gross polluters in model years 1985-1991. Therefore, fleet emissions could be significantly reduced if new gross polluting vehicles received regular servicing or were removed from the highway by the withdrawal of their vehicle registration. Older vehicles play a minor role in fleet emissions with on average only 14% of carbon monoxide and 13% of hydrocarbon fleet emissions produced by vehicles registered prior to 1983.

CHAPTER 5. ROADSIDE AIR QUALITY

5.1 Introduction

Differences in meteorology, topography, source characteristics, pollutant behaviour and legal and administrative constraints mean that air pollution monitoring programmes vary in scope, content and duration. However, ambient air sampling which is, considered to be the collection of air samples in any unconfined location exposed to the atmosphere *(Pio 1986)*, can be broadly divided into the following categories *(Hewitt & Allot 1992)*:

- source-oriented sampling for monitoring individual or small groups of emitters as part of a local survey
- sampling in a more extensive survey which may be located in areas of highest expected pollutant concentrations, high population density, or in rural areas to give a complete nationwide coverage
- base-line sampling to obtain background concentrations, usually in remote or rural areas with no anticipated changes in land-use.

The measurement of roadside air quality undertaken in this project is an example of source oriented sampling as part of a local survey. The sources in this case are motor vehicle emissions and the local survey area is the roadside.

5.2 Site description

Continuous pollutant monitors for CO and NO_x together with data acquisition equipment (see Chapter 3) were situated at kerbside (3 metres from the road/pavement interface), 7.5 metres and 15 metres from the road at sites A, B, C and D with an additional roadside (on the kerb) location at site D. However, because only one CO monitor and one NO_x monitor were available, measurements at different distances from the road were not carried out simultaneously but on different days.

Air pollutant concentrations were recorded every second and from this, five minute and hourly averages were calculated. An 8-hour average could only be estimated as the equipment was normally in situ for a maximum of seven hours. The collected data have been used to plot time series variations in real time (second by second variation) and as five minute means and hourly averages.

5.2.1 Site A (Bounds Green Road, Haringey)

Ambient air sampling was undertaken, in conjunction with the remote sensing measurements, between 25 May and 1 June 1994, adjacent to the north west bound carriageway of the A109 at locations directly opposite the FEAT unit *(Figure 5.1)*. Monitoring was conducted adjacent to the north west bound carriageway for two reasons. Firstly, although ambient air monitoring could have been undertaken adjacent to the south east bound carriageway, there would have been a greater possibility of pollutant interference due to emissions from the generators powering the FEAT system on this side of the road. Secondly, a power supply for the ambient air monitors, other than from a generator, would have been problematic adjacent to the south east bound carriageway whereas a mains connection from the university campus was readily available adjacent to the north west bound carriageway.

At the sampling point the vehicle flow towards Wood Green has an orientation of approximately 113 degrees from north with the vehicle flow towards the A406 having an orientation of approximately 293 degrees from north.

5.2.2 Site B (Dixons Bank, Middlesborough)

Ambient air sampling was undertaken, in conjunction with FEAT measurement, during the week 7-11 June 1994, adjacent to the north west bound carriageway of the A172 *(Figure 5.2)*. Ambient sampling adjacent to the FEAT equipment was possible because power was supplied to the latter via a mains connection to a lamp-post/traffic bollard and not by petrol generators. Therefore, any possible interference from the exhaust emissions of the generators to the ambient air monitors was removed. Power was also supplied to the ambient air monitors via a mains link to a lamp-post/traffic bollard.

At the sampling point the vehicle flow towards Middlesborough has an orientation of approximately 310 degrees from north with the vehicle flow away from Middlesborough having an orientation of approximately 130 degrees from north.

5.2.3 Site C (Abbey Street, Southwark)

Ambient air sampling was undertaken, in conjunction with FEAT measurement, on 8 March and 8 and 9 August 1995, adjacent to the south west bound carriageway of the B202 *(Figure 5.3)*. The primary reason for siting the ambient air monitors adjacent to the south west bound carriageway was the difficulty in gaining distance away from the road, near to the FEAT equipment, on the filling station side of the road. This was also a problem adjacent to the south west bound carriageway with a distance of 15 metres from the road only possible by moving the monitors down a lightly trafficked side street. Unfortunately, power had to be supplied to the ambient air monitors and the FEAT equipment by petrol generators as a mains connection to either a lamp-post or a traffic bollard was not possible. Therefore, the generators were positioned downwind when feasible. This proved impossible on certain occasions at which point extension cables were used to gain as much distance between the generators and the monitors as possible.

At the sampling point the vehicle flow away from the railway has an orientation of approximately 255 degrees from north with the vehicle flow towards the railway having an orientation of approximately 75 degrees from north.

5.2.4 Site D (Uppingham Road, Leicester)

Ambient air sampling was undertaken, in conjunction with remote sensing measurements, between 19-22 October 1995, adjacent to the west bound carriageway of the A47 *(Figure 5.4)*. The primary reason, as with site B, for sampling adjacent to the west bound carriageway was that, unlike sampling sites A and C, power was supplied to the FEAT equipment and ambient air monitors via a mains connection to a lamp-post/traffic bollard and not by petrol generators. Therefore, any possible interference from the exhaust emissions of the generators to the ambient air monitors was removed and they could be located nearer the FEAT system.

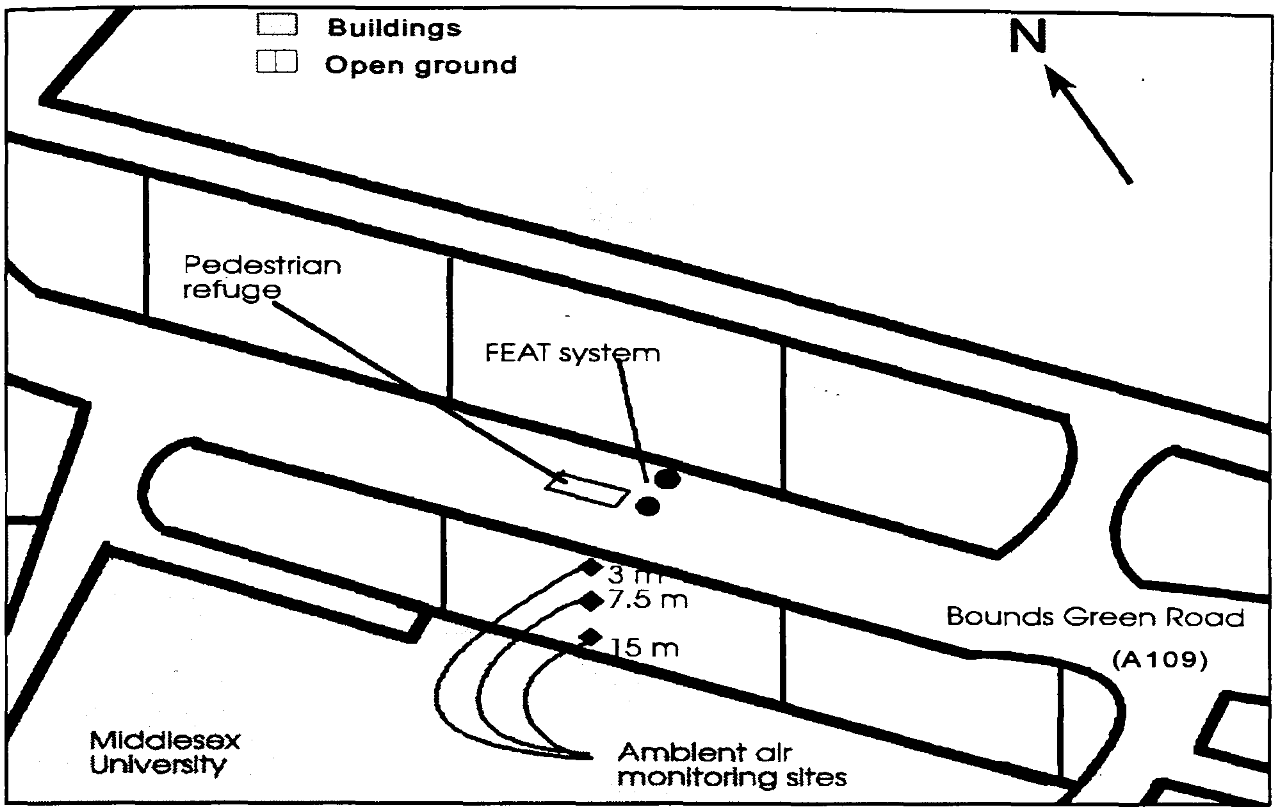


Figure 5.1 Plan view of sampling site A.

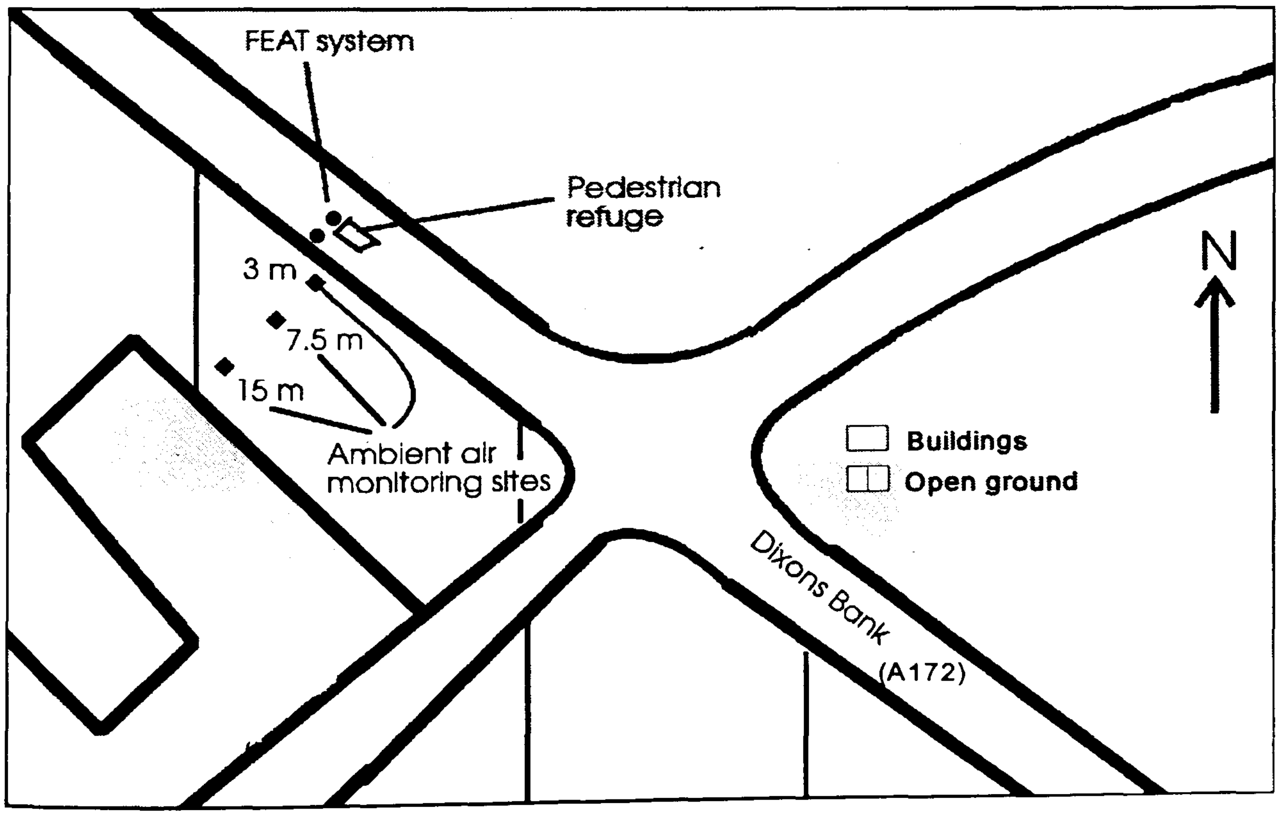


Figure 5.2 Plan view of sampling site B.

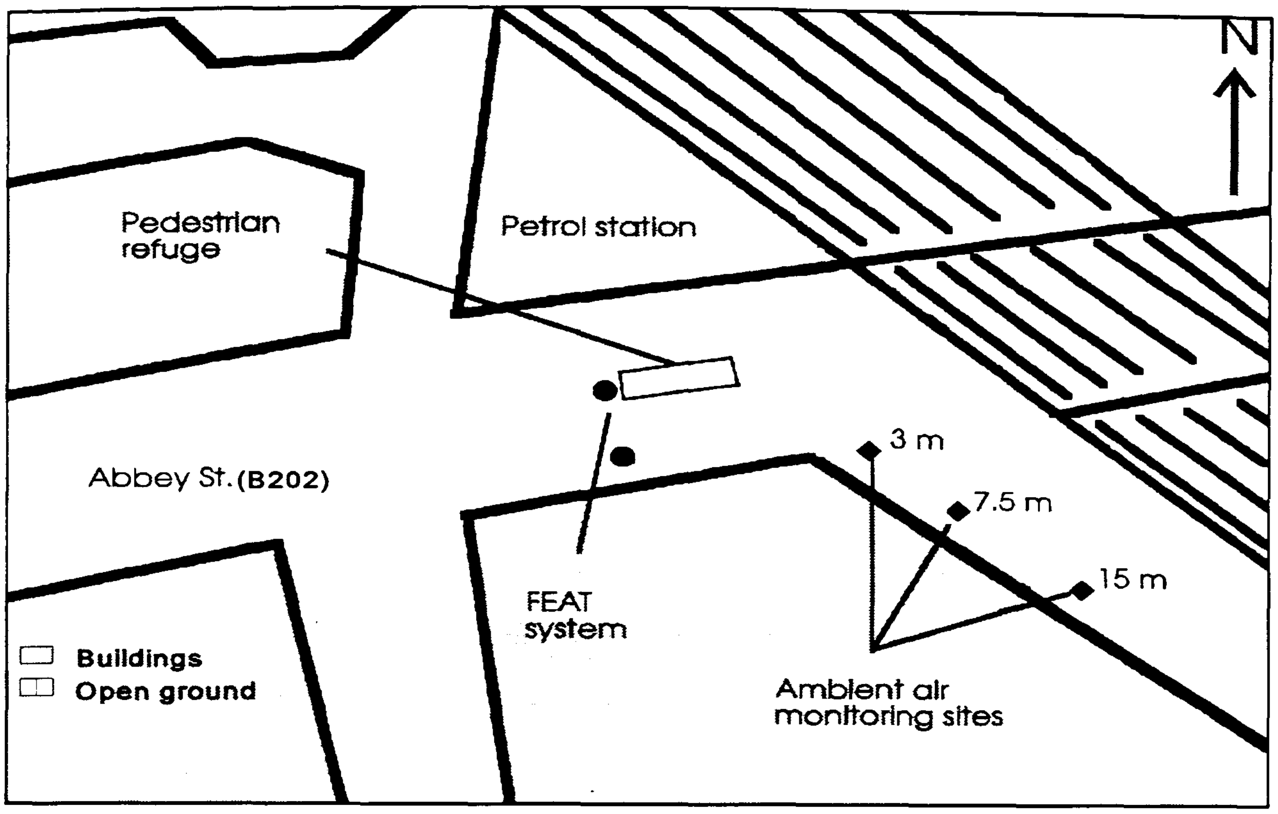


Figure 5.3 Plan view of sampling site C.

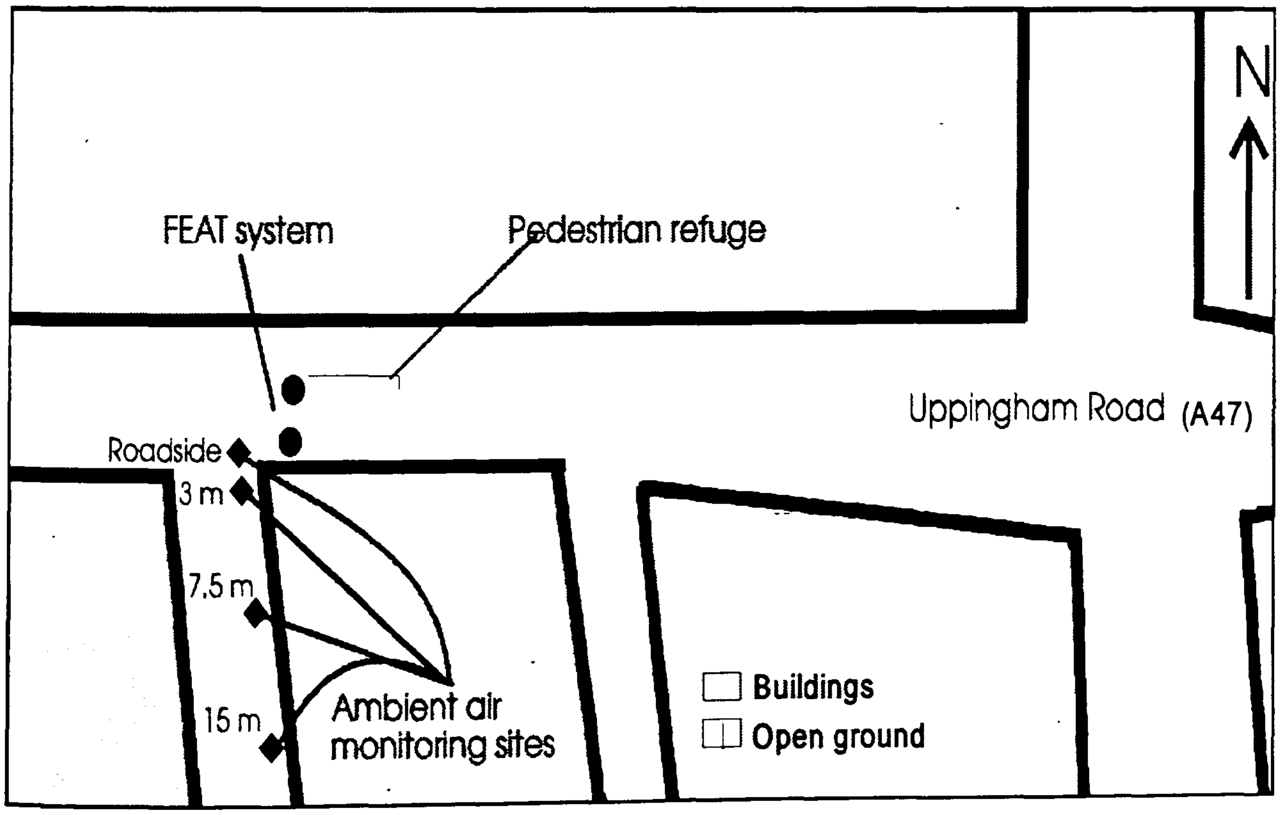


Figure 5.4 Plan view of sampling site D.

At the sampling point the vehicle flow towards Leicester city centre has an orientation of approximately 270 degrees from north with the vehicle flow away from Leicester city centre having an orientation of approximately 90 degrees from north.

5.3 Carbon monoxide air quality

The following sections include analyses of time series plots of carbon monoxide air quality from the four survey sites. Time series plots can be used to derive information concerning cycles and trends *(WHO; 1980)*. Cycles relate to short term effects which can be superimposed on longer term trends. Due to the short monitoring time span longer term trends cannot be examined. The data is compared to that obtained from the Automated Urban network and the carbon monoxide air quality section concludes with a discussion of the differences and similarities between sites.

5.3.1 Site A (Bounds Green Road, Haringey)

5.3.1.1 Time series plot of raw data from the carbon monoxide monitor

The highest value recorded was measured at kerbside on 25 May 1994 with a peak value of 13.9 ppm *(Figure 5.5 A)*. Measurements carried out at the same kerbside location on a different date (1 June 1994) produced a lower maximum peak value of 9.7 ppm *(Figure 5.5 B)*. The lowest CO concentrations were recorded 15 metres from the road where the highest recorded value was 2 ppm *(Figure 5.5 D)*. The site positioned 7.5 metres from the road produced an intermediate maximum value of 4.8 ppm *(Figure 5.5 C)*.

These peak values might be expected to coincide with peak traffic periods and therefore, peak emissions and this relationship will be discussed more fully in Chapter 6. However, an initial analysis shows that the highest concentrations on 25 May 1994 correspond to a possible early evening rush hour with the highest concentrations for 1 June 1994 corresponding to a possible midday rush hour. The highest peak values at the sites positioned 7.5 metres and 15 metres from the road are only marginally greater than the norm and appear not to reflect any influence from rush hour traffic. This may be because the extra dispersion that has taken place has muted the effects of the extra vehicles.

Daily mean concentrations, where daily is defined as the duration of the sampling day, follow the trend in reduction of concentration with distance with the highest average concentrations of 2.8 ppm (25 May 1994) and 1.9 ppm (1 June 1994) being recorded at kerbside and the lowest concentration recorded 15 metres from the road (0.6 ppm). The site 7.5 metres from the road exhibited a daily mean concentration of 1.5 ppm.

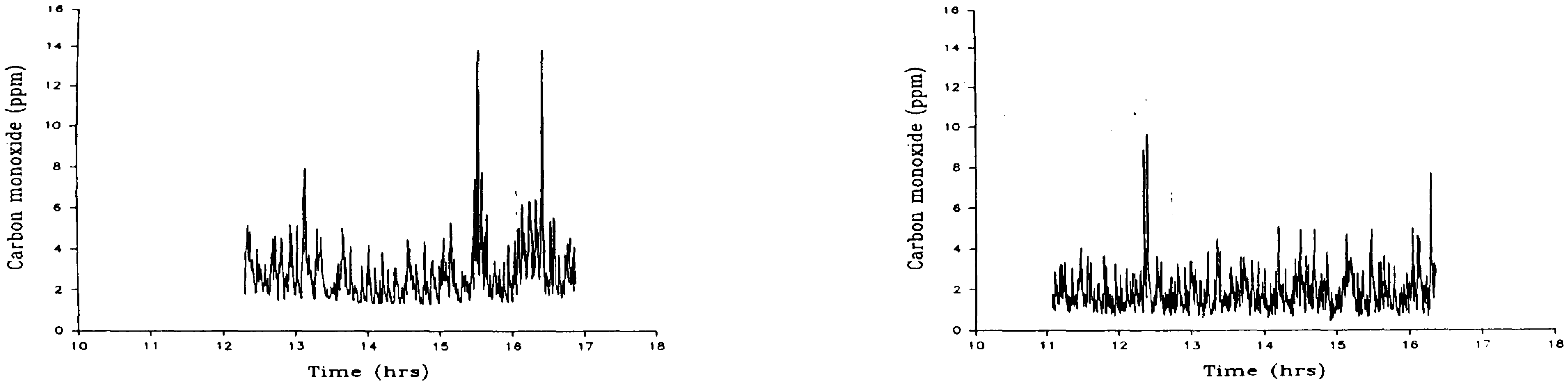
5.3.1.2 Time series plot of five minute mean concentrations

The pattern of reduction in pollutant levels with increasing distance away from the road is again demonstrated by the five minute mean concentrations *(Figure 5.6)* with the highest values - 5.8 ppm (25 May 1994) and 4.1 ppm (1 June 1994) - occurring at kerbside *(Figures 5.6 A & 5.6 B)* with correspondingly reduced values, at greater distances from the road - 3 ppm (7.5 m) & 1.2 ppm (15 m) *(Figures 5.6 C & 5.6 D)*.

5.3.1.3 Time series plot of hourly mean concentrations

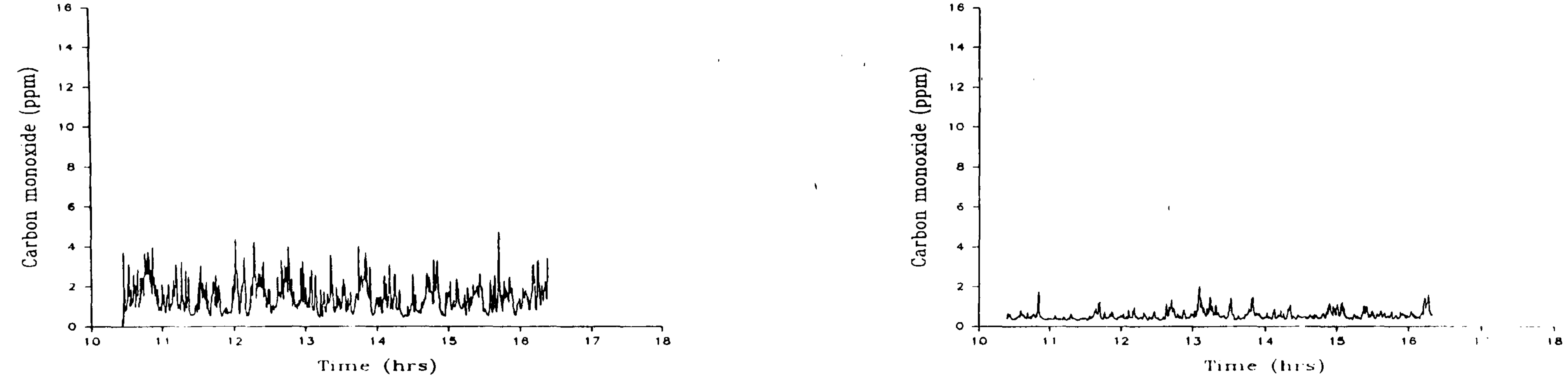
The hourly averaging period further smooths the data and both lowers the value of and reduces the significance of the highest concentrations. The plotted data *(Figure 5.7)* clearly demonstrates the reduction in CO concentration with distance away from the road.

The WHO has recommended a maximum carboxyhaemoglobin (COHb) level in blood of between 2.5% and 3% as a basis for protecting human health. To fulfil this commitment time weighted exposure limits have been established. The one hour exposure limit is 25 ppm *(WHO 1987)*. The maximum hourly peak value measured was only 2.9 ppm which is significantly lower than the WHO guideline *(Figure 5.7)*. The eight-hour exposure limit is 10 ppm and, although monitoring was not conducted for eight hours, it is reasonable to assume that this limit value would not have been approached had monitoring continued over the required period. There are at present no



A Kerbside (25 May 1994)

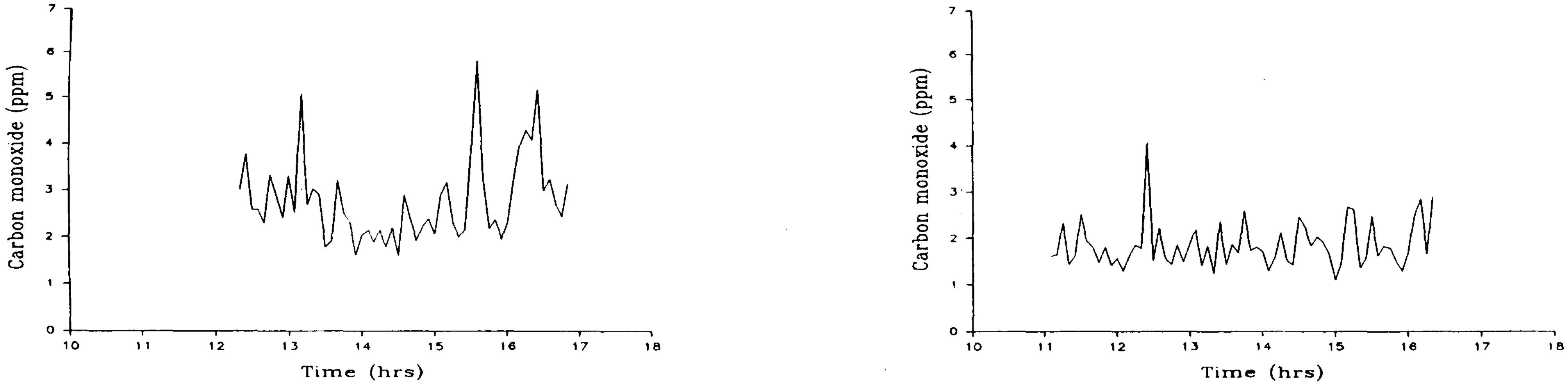
B Kerbside (1 June 1994)



C 7.5 metres (31 May 1994)

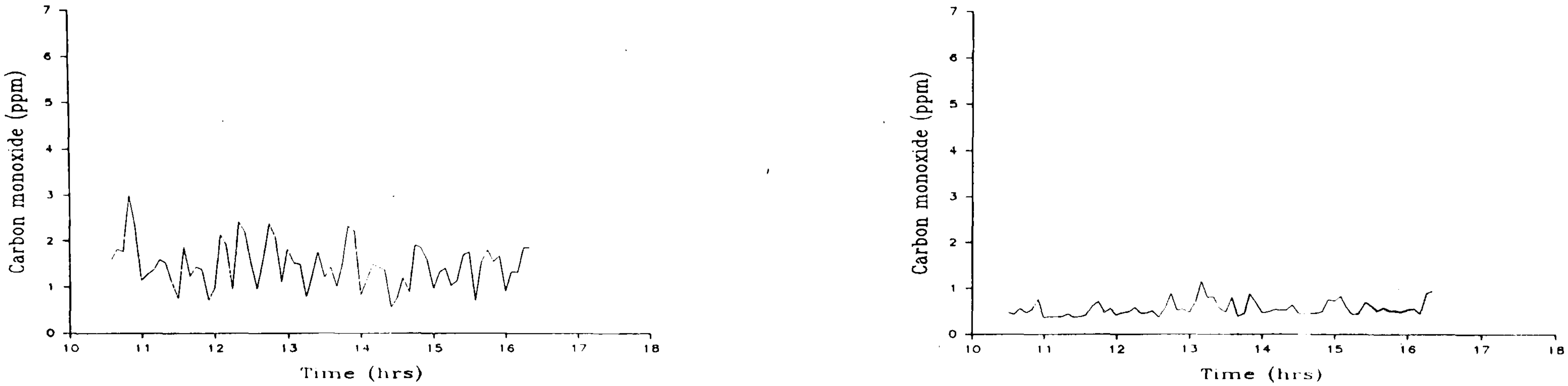
D 15 metres (27 May 1994)

Figure **5.5 (A-D)** Time series plots of raw data for CO at kerbside, 7.5 metres and 15 metres from the road at site A.



A Kerbside (25 May 1994)

B Kerbside (1 June 1994)



C 7.5 metres (31 May 1994)

D 15 metres (27 May 1994)

Figure 5.6 (A-D) Time series plots of five minute CO means at kerbside, 7.5 metres and 15 metres from the road at site A.

EU Directives regarding CO air quality, however, the UK Expert Panel on Air Quality Standards (EPAQS) has recommended a standard of 10 ppm measured as a running 8 hour average which is essentially the same standard as that set by the WHO.

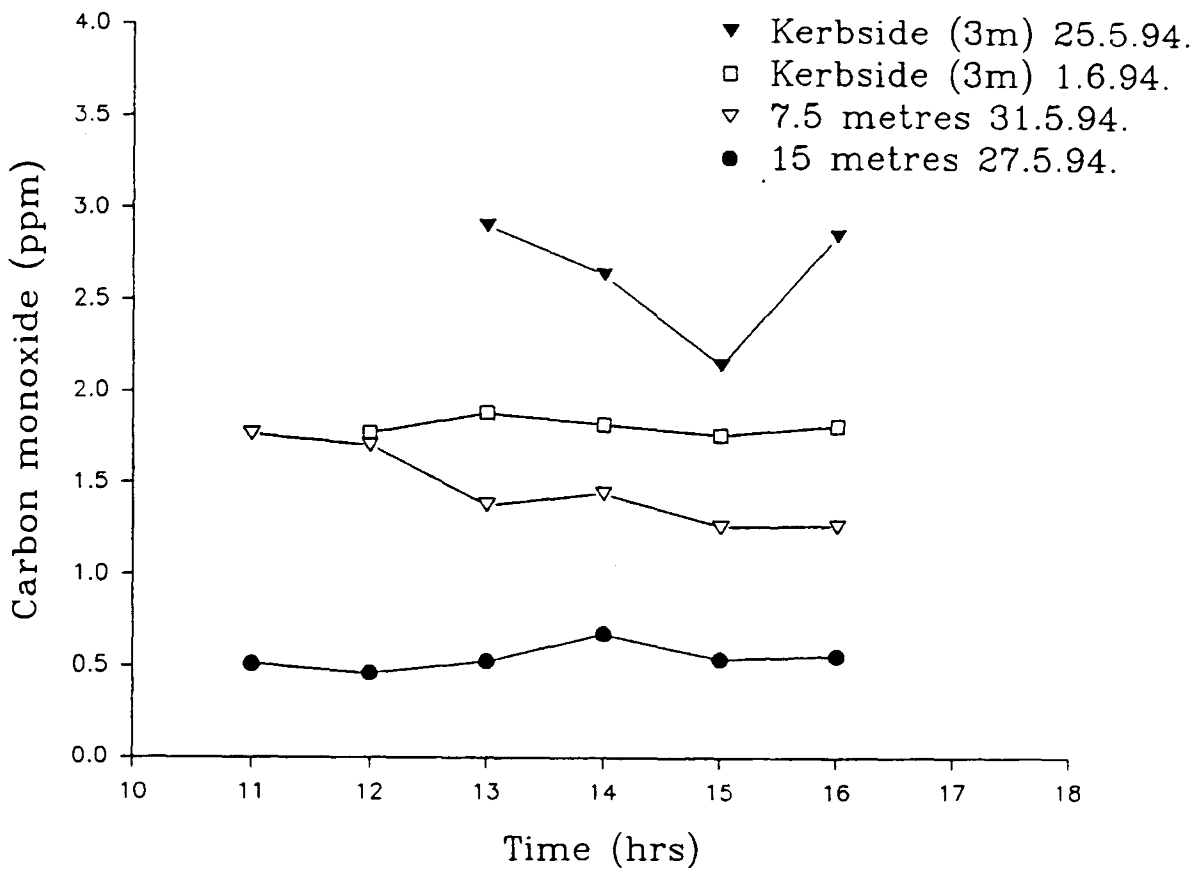


Figure 5.7 Hourly carbon monoxide concentrations at site A.

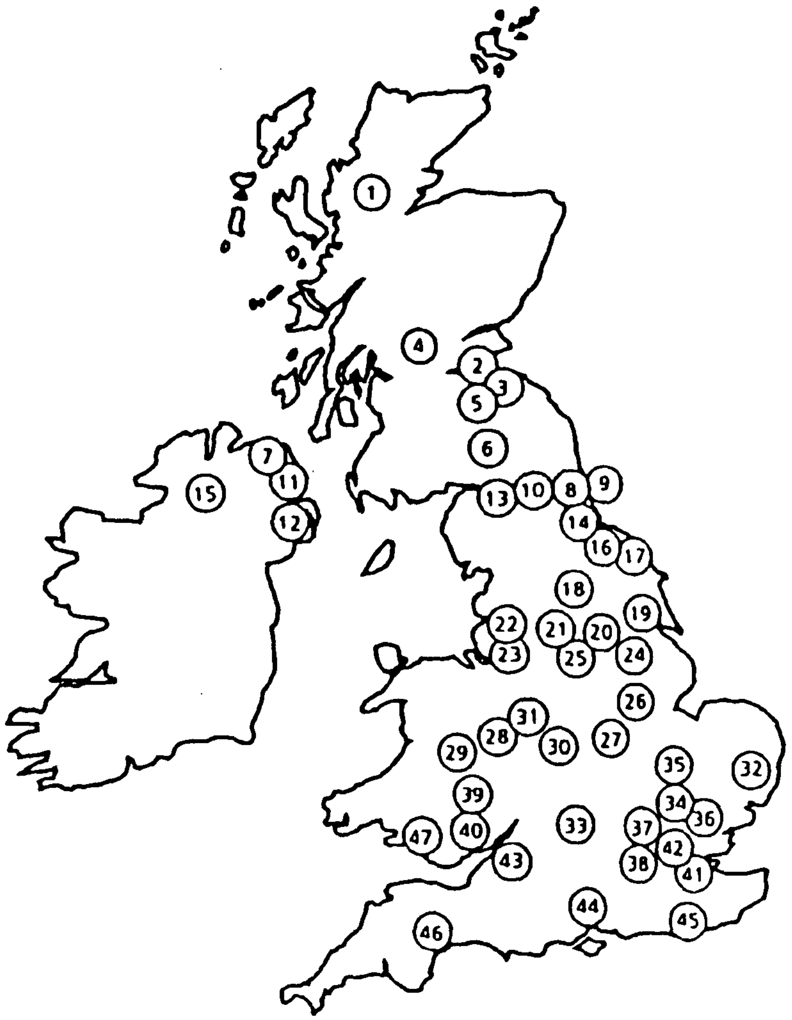
5.3.1.4 Comparison of the air quality at site A with Automated Urban Network data

In 1994/5 the DoE's automatic air quality monitoring was rationalised into two networks, the Automated Urban Network (AUN) and the Automated Rural Network (ARN). The AUN consists of those sites which formed the Enhanced Urban Network (EUN) and the Statutory Urban Network (SUN). Since 1994, this network has been used to assess UK compliance with the European Union SO_2 and Smoke, O_3, and NO_2 Directives *(Directives 80/779/EEC; 85/203/EEC & 92/72/EEC)*.

The AUN currently comprises 27 sites *(DoE 1995)* which monitor SO_2, NO_X, CO, O_3 and/or particulate (PM_{10}) concentrations at a variety of urban locations, ranging from kerbside monitoring, to monitoring of urban background areas representative of air pollution exposure for a large number of people. Following initial validation by TBV Science, the data are reported on an hourly basis to DoE's Data Dissemination Unit for transmission to CEEFAX, TELETEXT, a free phone air quality and health advice information service, and television weather bulletins. This network will be expanded by the end of 1996 to 36 sites and by the end of 1997 to include more than 50 sites *(DoE 1995)*. The ARN was set up in the early 1980s to provide information on photochemical pollution across the UK. The sites are not specifically in urban areas and therefore will not be used for comparative purposes in this discussion. Currently 15 stations are in operation measuring O_3 with three of these sites also monitoring SO_2 and NO_2. The location of existing automated monitoring sites in 1995 is shown in *Figure 5.8*.

Those sites most comparable to the Bounds Green site, within the AUN, include: West London (WL), Cromwell Road (CRD), London Bloomsbury (CLL2) and London Bexley (BEX). West London (WL) is an urban site within the Earls Court region of London (TQ 251 788). It is located in a municipal depot in a partly residential area bounded on all sides by major roads (90 metres from Warwick Road). Cromwell Road (CRD) is a kerbside site within London (TQ 264 789). The site is located at the kerbside of a busy arterial road in central London with a traffic density of approximately 60,000 vehicles per day. London Bloomsbury (CLL2) is classified as an urban site within London (TQ 302 820) and is located within Russell Square Gardens (SE quadrant), 35 metres from the kerbside. London Bexley (BEX) is an urban site within London (TQ 518 763) which is located within a community centre car park off Bridge Road in a residential area of North End.

The comparison of air quality data from site A with that monitored at the comparable AUN sites is shown in *Figure 5.9 A-D*. Comparison of the data gained from kerbside on 25 May 1994 *(Figure 5.9 A)* shows site A to have lower carbon monoxide concentrations than Cromwell Road but higher concentrations than those found at West London, London Bloomsbury and London Bexley. It was expected that the kerbside location would have lower carbon monoxide levels than Cromwell Road because of the generally higher traffic flows at the inner London location. It was also expected that site A would have higher carbon monoxide levels than the urban background sites as it is



Site No.	Site Name	Pollutants (see key)	Site No.	Site Name	Pollutants (see key)
1	Straith Vaich	★● ▲	26	Bottesford	★
2	Edinburgh Centre	★●■▲O	27	Leicester Centre	★●■▲O
3	Edinburgh Med S	⬟	28	Walsall	●
4	Glasgow	●■	29	Aston Hill	★
5	Bush	★	30	Birmingham Wardend	★●■▲O⬟
6	Eskdalemuir	★	31	Birmingham Centre	★●■▲O
7	Belfast Queens Uni	⬟	32	Sibton	★
8	Newcastle Centre	●■▲O	33	Harwell	★
9	Sunderland	▲	34	London UCL	⬟
10	Great Dun Fell	★	35	London Bloomsbury	★●■▲O
11	Belfast Centre	★●■▲O	36	Bridge Place	★●■▲
12	Belfast East	▲	37	Cromwell Road	●■▲
13	Wharley Croft	★	38	West London	●■
14	Billingham	●	39	Cardiff East	⬟
15	Lough Navar	★	40	Cardiff Centre	★●■▲O
16	Middlesbrough	⬟	41	London Bexley	★●■▲O
17	High Muffles	★	42	London Eltham	⬟
18	Leeds Centre	★●■▲O	43	Bristol Centre	★●■▲O
19	Hull Centre	★●■▲O	44	Southampton Centre	★●■▲O
20	Barnsley	▲	45	Lullington Heath	★● ▲
21	Manchester	●■	46	Yarner Wood	★
22	Glazebury	★	47	Swansea	★●■▲O
23	Liverpool Centre	★●■▲O			
24	Sheffield	●■			
25	Ladybower	★● ▲			

Key ★ O_3 ● NO_x ■ CO ▲ SO_2 O PM_{10} ⬟ HC

Figure 5.8 UK automated air quality monitoring sites: December 1995 *(DoE 1995)*.

closer to the primary source of carbon monoxide emissions. However, comparison of the data obtained from kerbside on the 1 June 1994 *(Figure 5.9 B)* shows a slightly different scenario with carbon monoxide levels at site A generally higher than those of Cromwell Road while also remaining predominantly higher than the urban background sites. This unusual situation could be caused by higher wind speeds at Cromwell Road leading to greater dispersion of pollutants compared to site A.

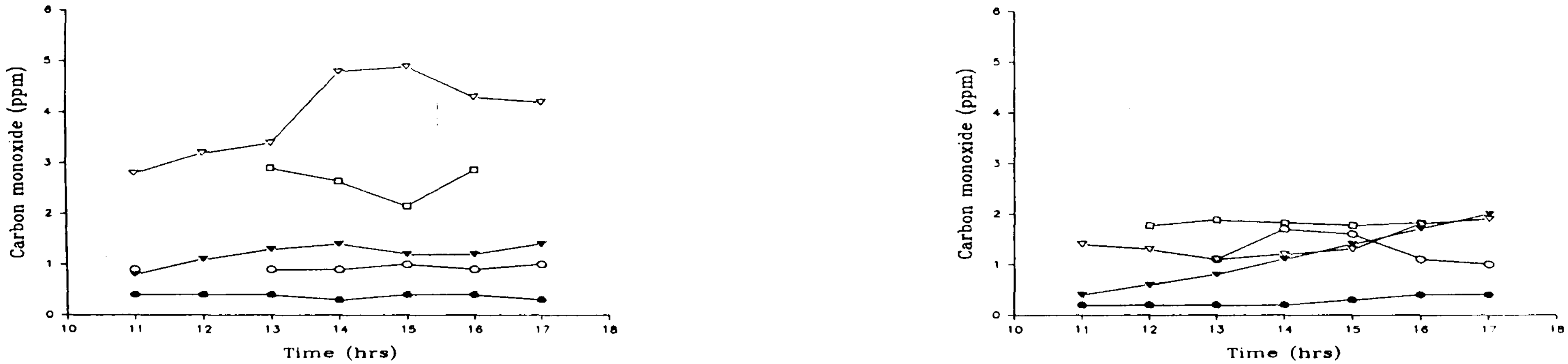
Comparison of the data gained from the site positioned 7.5 metres from the road with the AUN data is shown in *Figure 5.9 C.* This shows Cromwell Road to have higher carbon monoxide levels than those found at site A which in turn are higher than those found at the urban background sites. This is as expected with the two sites nearest to roads having the highest carbon monoxide concentrations. The comparison of the data gained from the site positioned 15 metres from the road with the AUN data is shown in *Figure 5.9 D* and shows the carbon monoxide levels at site A to have fallen to such an extent that they are indistinguishable from the concentrations found at the urban background sites. Therefore, at site A ambient levels can be considered to have reached urban background concentrations at a distance of 15 metres from the road. The Cromwell Road site has significantly higher carbon monoxide levels than all other locations.

5.3.2 Site B (Dixons Bank, Middlesborough)

5.3.2.1 Time series plot of raw data from the carbon monoxide monitor

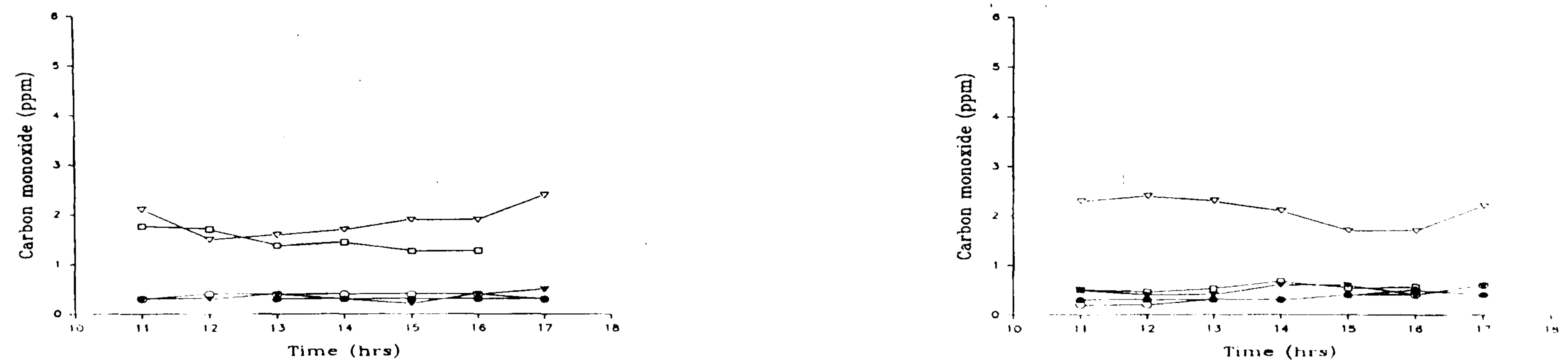
The highest carbon monoxide concentration was measured at kerbside on 10 June 1994 with a peak value of 3.4 ppm *(Figure 5.10 A)*. Measurements at kerbside on a preceding day (7 June 1994) had a lower maximum peak value of 2.1 ppm *(Figure 5.10 B)*. The site positioned 7.5 metres from the road had the lowest maximum value of 0.7 ppm *(Figure 5.10 C)*. No measurements were recorded 15 metres from the road because of a break in the cable connecting the data acquisition equipment and the ambient air monitors.

Daily mean values recorded ranged from 0.7 ppm at kerbside (10 June 1994) to 0.1 ppm at both 7.5 metres from the road (8 June 1994) and kerbside (7 June 1994).



A Comparison at kerbside (25 May 1994)

B Comparison at kerbside (1 June 1994)



C 7.5 metres (31 May 1994)

D 15 metres (27 May 1994)

Figure 5.9 (A-D) Comparison of one hour mean CO levels at Site A (□), at kerbside, 7.5 metres and 15 metres from the road, to those found at AUN sites WL (▼) BEX (●) CRD (▽) and CLL2 (○).

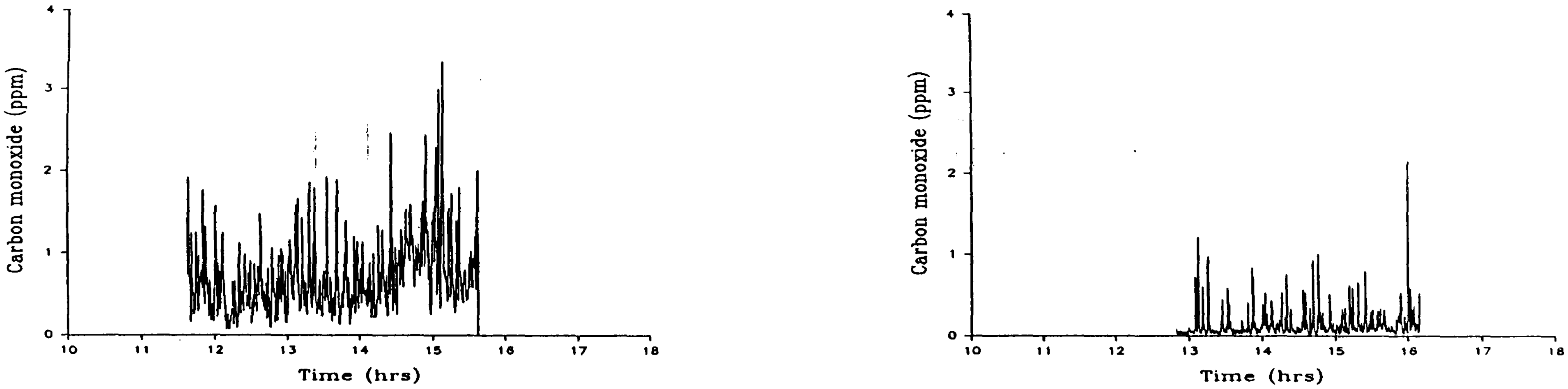
There is some similarity between the two kerbside days when the maximum peak measurements are compared, however, there is considerable dissimilarity when the mean measurements are compared reflecting the higher baseline values on the 10 June 1994.

One possible explanation for the dissimilarity in mean ambient levels is the difference in wind direction for the two kerbside days which is explored in more detail in Chapter 6. On the 7 June 1994 the prevailing wind direction was more parallel to the road whereas on the 10 June 1994 the prevailing wind direction was more from the road towards the ambient air monitors. Consequently, the 10 June 1994 has a higher mean ambient concentration.

A further contributing factor could be the change in wind speed between the two kerbside measurement days with a higher wind speed on 7 June 1994 possibly contributing to lower ambient concentrations. Unlike sampling site A the highest peak values do not appear to correspond to normal peak traffic periods such as morning, lunch or evening rush hours.

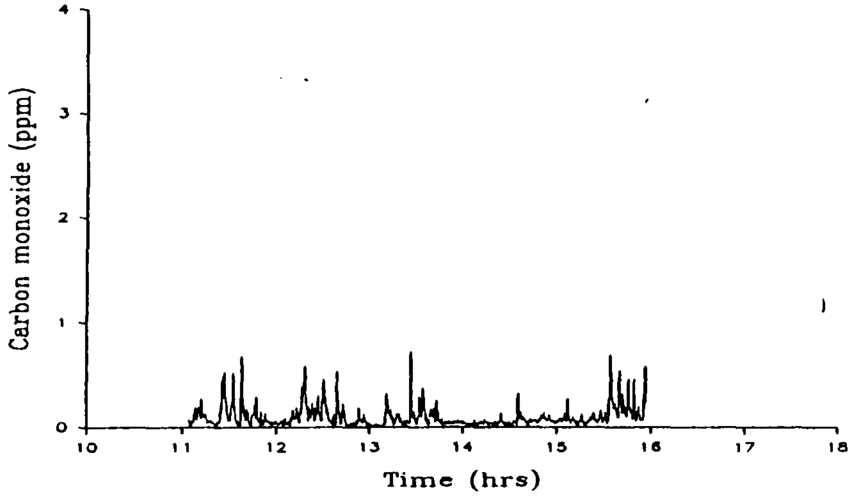
5.3.2.2 Time series plot of five minute mean concentrations

The pattern displayed in the second by second time series, of reduction in pollution levels with increasing distance, is again demonstrated although the maximum peak values are lower *(Figures 5.11 A-C)*. The five minute averaging period gave peak values of: 1.4 ppm (kerbside - 10 June 1994), 0.4 ppm (kerbside - 7 June 1994) and 0.3 ppm (7.5 metres - 8 June 1994).



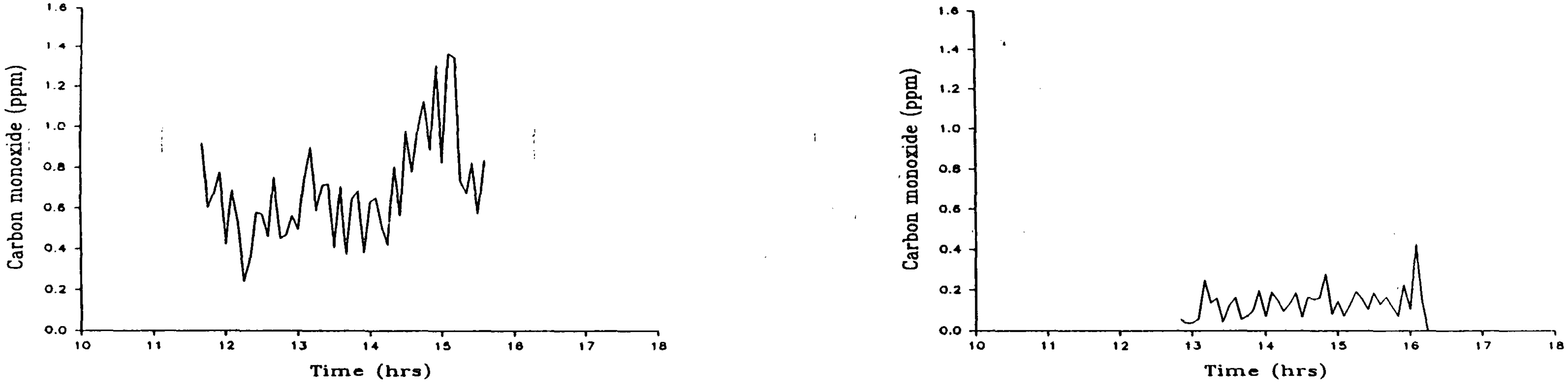
A Kerbside (10 June 1994)

B Kerbside (7 June 1994)



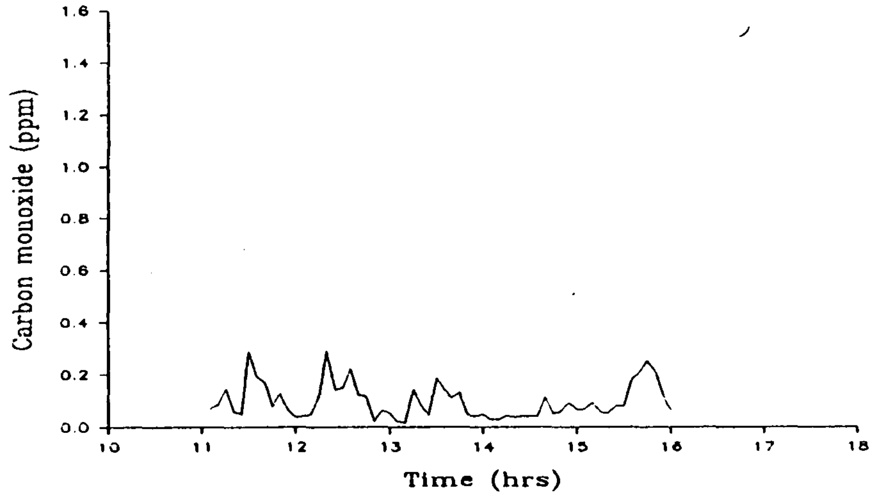
C 7.5 metres (8 June 1994)

Figure 5.10 (A-C) Time series plots of raw data for CO at kerbside and 7.5 metres from the road at site B.



A Kerbside (10 June 1994)

B Kerbside (7 June 1994)



C 7.5 metres (8 June 1994)

Figure 5.11 (A-C) Time series plots of five minute means for CO at kerbside and 7.5 metres from the road at site B.

5.3.2.3 Time series plot of hourly mean concentrations

The disparity between the two kerbside measurements observed in the time series plots of raw data and five minute means is also visible for the time series plot of hourly means *(Figure 5.12)*, thus reinforcing the need for a longer sampling period to accurately reflect ambient air concentrations at this location.

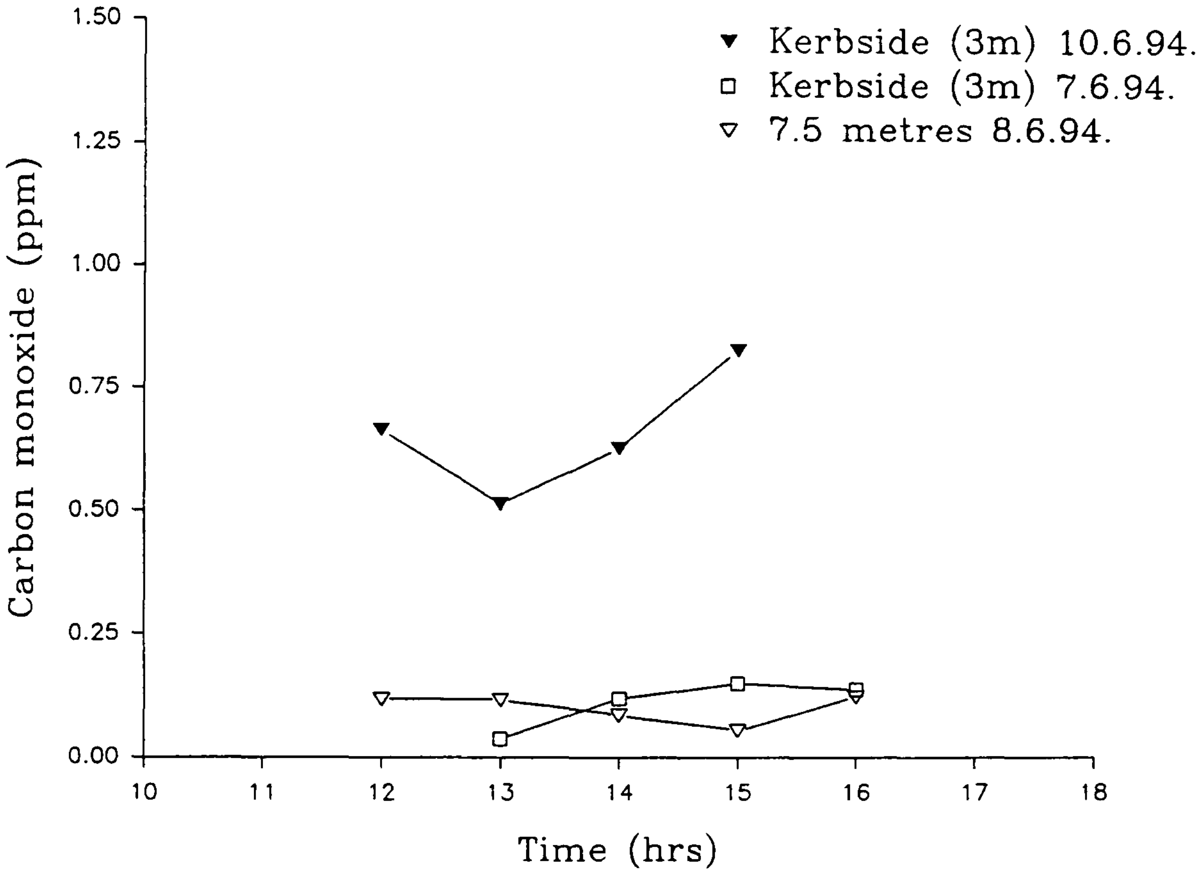


Figure 5.12 Hourly carbon monoxide concentrations at site B.

As with the Bounds Green site, the WHO one hour exposure limit of 25 ppm was not approached, with 0.8 ppm being the highest one hour peak value recorded at site B. An eight-hour exposure value could only be estimated but would fall far short of the 10 ppm exposure limit.

5.3.2.4 Comparison of the air quality at site B with Automated Urban Network data

A comparison with AUN data was not possible at sampling site B. The AUN includes an urban Middlesborough (MID) site, situated in a residential area near Longlands College of Further Education (NZ 505 194), however, the AUN site only became operational after sampling had been completed.

5.3.3 Site C (Abbey Street, Southwark)

5.3.3.1 Time series plot of raw data from the carbon monoxide monitor

The highest value recorded was measured 15 metres from the road with a maximum peak value of 13.9 ppm *(Figure 5.13 C)*. This high value was caused by the parking of taxis near to the monitors during the morning of the day when sampling was carried out. This problem was eradicated by parking the mobile laboratory in such a position as to prevent taxis waiting there. The peak value recorded after the removal of this interfering emission source was 2.4 ppm. Measurements carried out at the kerbside location produced a maximum peak value of 13.6 ppm *(Figure 5.13 A)*. The peak CO concentration recorded 7.5 metres from the road was 6.5 ppm *(Figure 5.13 B)*.

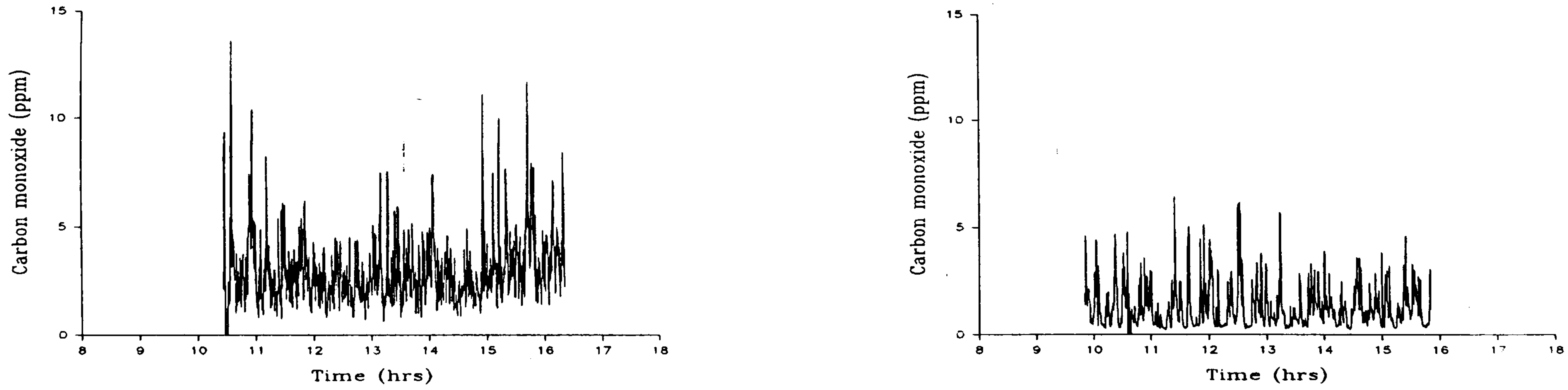
As with sampling site A, the kerbside location has peak carbon monoxide concentrations which appear to coincide with possible morning and early evening rush hours. Similarly the 7.5 and 15 metres locations do not appear to show any correlation between peak traffic times and high peak concentrations.

Daily mean concentrations followed the trend in reduction of concentration with distance from the road with the highest average concentration of 2.9 ppm being recorded at kerbside. The lowest mean value of 0.4 ppm (after anomalous peaks were removed) was found 15 metres from the road with a concentration of 1.3 ppm being recorded 7.5 metres from the road.

5.3.3.2 Time series plot of five minute mean concentrations

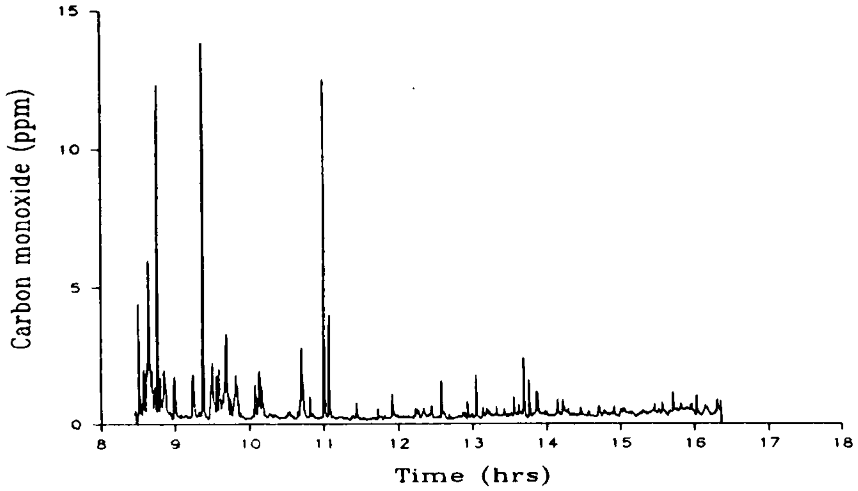
The pattern of reduction in 5 minute peak pollutant levels with increasing distance from the road is repeated *(Figure 5.13 A-C)*, with lower values, at greater distances from the road; 5.9 ppm (kerbside), 3.7 ppm (7.5 m) and 0.6 ppm (15 m) having ignored morning peak values due to interfering emissions from parked taxis.

Further analysis of the timing of the peak carbon monoxide concentrations again shows



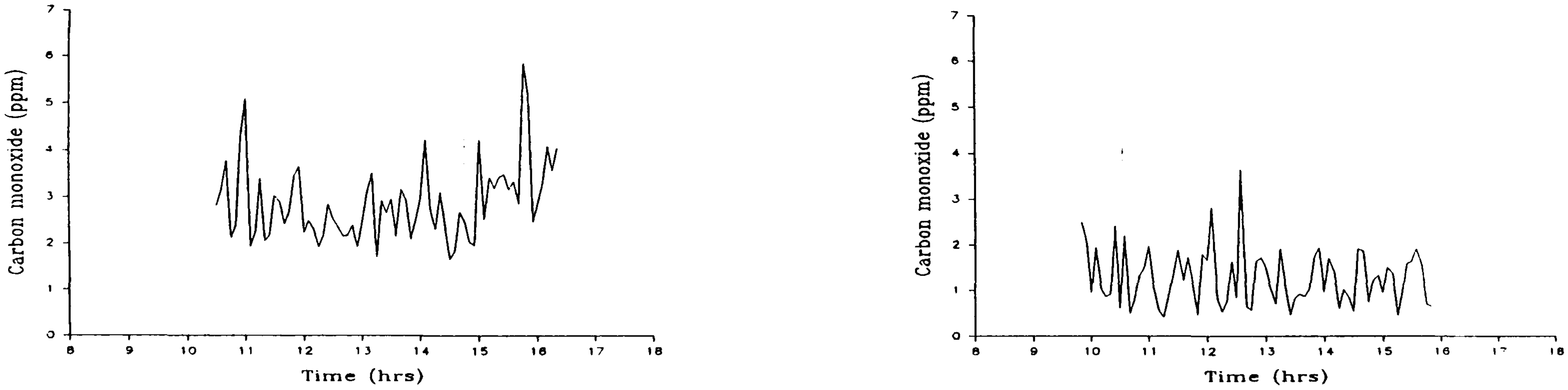
A Kerbside (8 March 1995)

B 7.5 metres (8 August 1995)



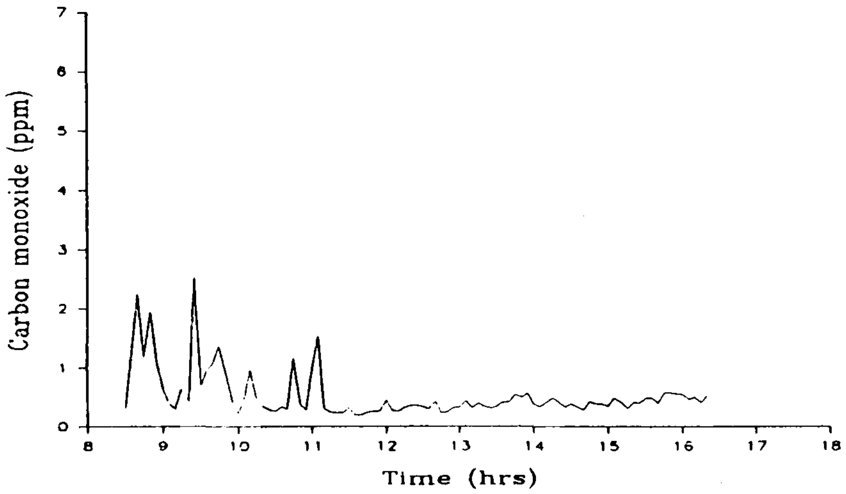
C 15 metres (9 August 1995)

Figure 5.13 (A-C) Time series plots of raw data for CO at kerbside, 7.5 metres and 15 metres from the road at site C.



A Kerbside (8 March 1995)

B 7.5 metres (8 August 1995)



C 15 metres (9 August 1995)

Figure 5.14 (A-C) Time series plots of five minute means for CO at kerbside, 7.5 metres and 15 metres from the road at site C.

the morning and early evening peaks at kerbside but also shows a lunch time peak level 7.5 metres from the road. The 15 metres site exhibits no exceptional deviations from a relatively low baseline after 11.30 am.

5.3.3.3 Time series plot of hourly mean concentrations

The plotted data *(Figure 5.15)* clearly demonstrate the reduction in CO concentration with distance away from the road. As for sampling sites A and B, the WHO one hour exposure guideline was not approached, with 3.5 ppm being the highest one hour peak value recorded at site C during the late afternoon. Although monitoring was not conducted for a continuous eight hour period, it is reasonable to estimate that the eight hour exposure guideline value would not have been approached.

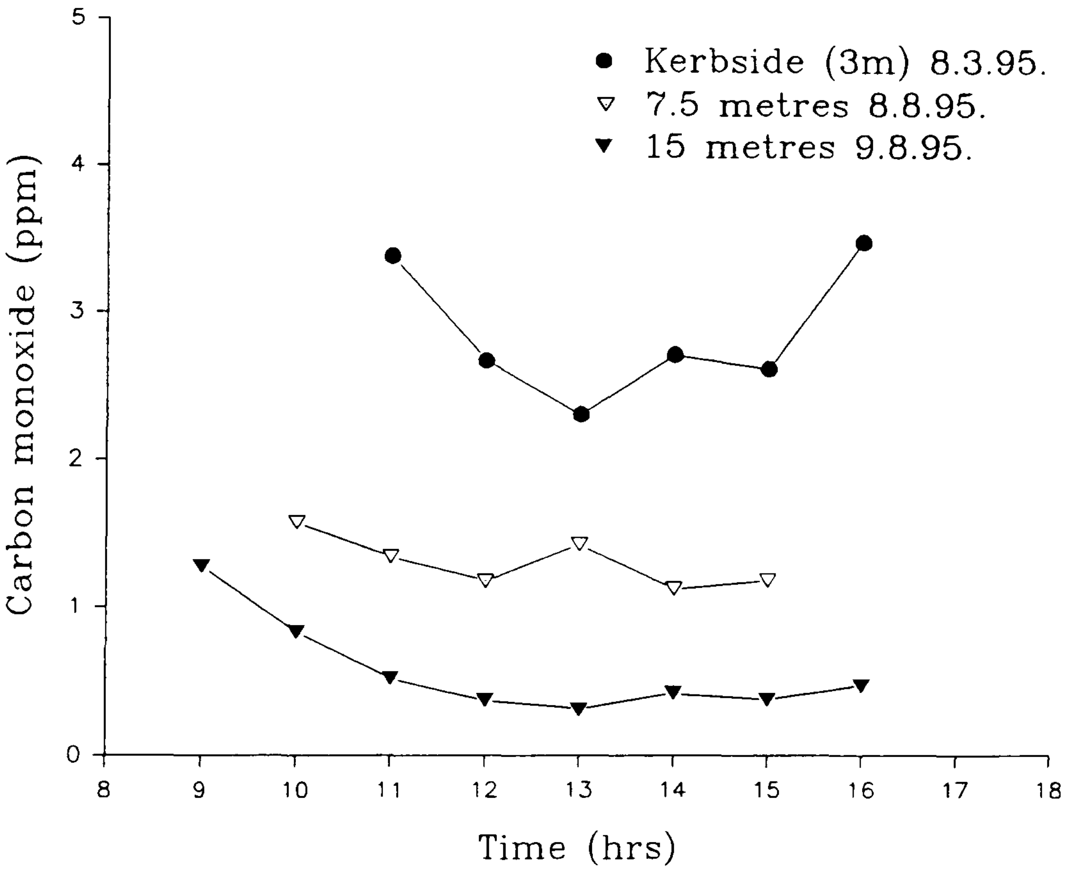


Figure 5.15 Hourly carbon monoxide concentrations at site C.

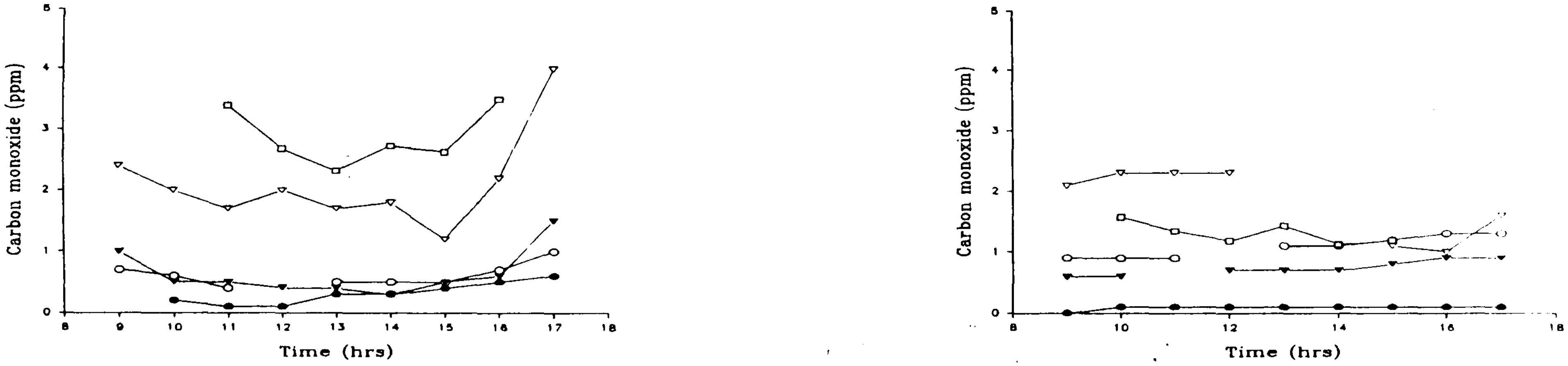
5.3.3.4 Comparison of the air quality at site C with Automated Urban Network data

Those sites (WL, CRD, CLL2, BEX - section 5.3.1.4) within the AUN, which were compared to sampling site A, are also relevant for comparing with the Abbey Street data. The comparison of site C air quality data with that gained at the comparable AUN sites is shown in *Figure 5.16 A-C.*

Comparison of the data gained from kerbside *(Figure 5.16 A)* shows site C to have the highest carbon monoxide concentrations. The values are similar to those found at Cromwell Road but significantly higher than the West London, London Bloomsbury and London Bexley sites. Predictions would have suggested that site C would have lower carbon monoxide levels than Cromwell Road because of the generally higher traffic flows found associated with latter site. However, the expectation that site C would have higher carbon monoxide levels than the urban background sites because it is closer to the primary source of carbon monoxide emissions was confirmed.

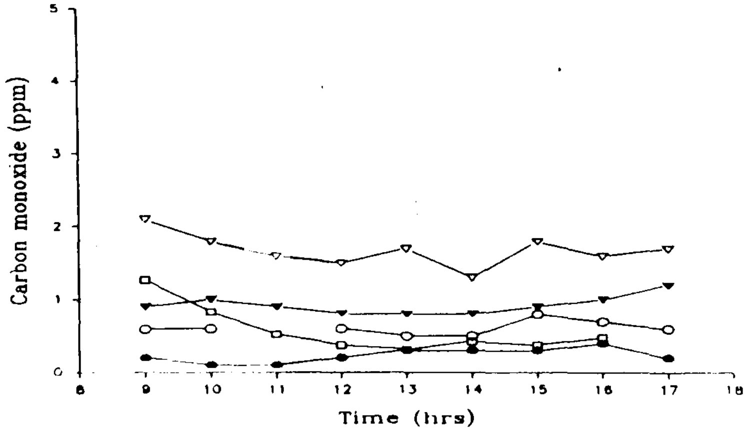
The comparison of the data gained from the site positioned 7.5 metres from the road with the available AUN data is shown in *Figure 5.16 B.* The AUN data was incomplete for Cromwell Road and London Bloomsbury. The network administrator could not be precise about the cause of the missing data, stating only that 'missing data are primarily caused by either calibration episodes or equipment malfunctions' *(Broughton 1996).*

Cromwell Road exhibits higher carbon monoxide levels than those found at site C with site C having predominantly higher carbon monoxide levels than those found at the urban background sites. This is as expected with the two sites nearest to roads having the highest carbon monoxide concentrations. The comparison of the data gained from the site positioned 15 metres from the road with the AUN data is shown in *Figure 5.16 C.* Carbon monoxide levels at site C have fallen to such an extent that they are analogous to the concentrations found at the urban background sites.



A Kerbside (8 March 1995)

B 7.5 metres (8 August 1995)



C 15 metres (9 August 1995)

Figure 5.16 (A-C) Comparison of one hour mean CO levels at Site C (□), at kerbside, 7.5 metres and 15 metres from the road, to those found at AUN sites WL (▾) BEX (●) CRD (▿) and CLL2 (○).

5.3.4 Site D (Uppingham road, Leicester)

5.3.4.1 Time series plot of raw data from carbon monoxide monitor

The highest value recorded was measured 7.5 metres from the road on 20 October 1995 with a maximum peak recorded value of 13.9 ppm *(Figure 5.17 C)*. The lowest maximum peak carbon monoxide concentration (6.1 ppm) was recorded at kerbside *(Figure 5.17 B)*. The site positioned 15 metres from the road produced a maximum value of 13.8 ppm *(Figure 5.17 D)* only marginally lower than the highest measured value. The roadside location produced a maximum peak value of 9.8 ppm *(Figure 5.17 A)*.

Daily mean concentrations followed a similar trend with the highest average concentration of 1.5 ppm recorded 7.5 metres from the road. The lowest mean value (0.7 ppm) was found at roadside with a concentration of 1.4 ppm being recorded 15 metres from the road. The kerbside location produced a mean value of 1.2 ppm.

This unusual situation is, in part, explained by the prevailing wind direction and wind speeds encountered during the different sampling days. The highest peak and mean values were both found on 20 October 1995, 7.5 metres from the road. The prevailing wind direction on 20 October 1995 was from the north east giving a wind direction, in crude terms, away from the road and towards the ambient air monitors. This was the only day this occurred with the prevailing wind direction on all other days being either from the south or south west, giving a wind flow away from the air quality monitors towards the road. Therefore, the 7.5 metres site recorded the highest carbon monoxide levels.

The variation in air quality between the 21 October 1995 (15 metres) and the 19 (kerbside) & 22 (roadside) October 1995 may be explained by the significantly lower wind speeds encountered on 21 October 1995. There was a period of calm wind conditions between 08:00 and 10:00 and very low wind speeds between 10:00 and 11:00 on 21 October 1995 coinciding with elevated background levels. The inference is that a build up of pollutants occurred and these were unable to disperse consequently giving rise to high background levels. Wind speeds increased after 11:00 but remained low and were significantly lower than on the 19 & 22 October 1995 thereby explaining the higher ambient levels 15 metres from the road.

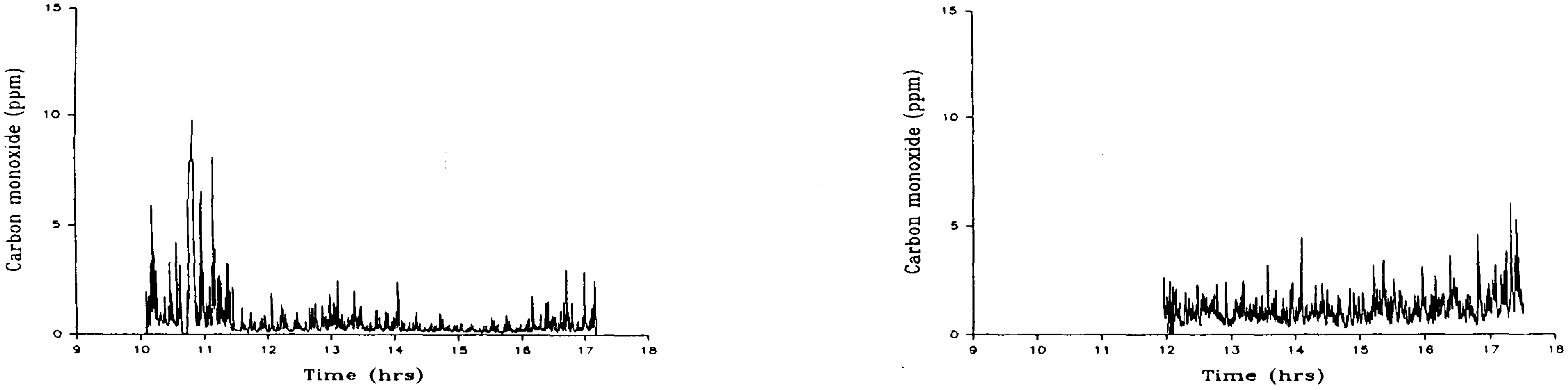
The variation in air quality for the 19 October 1995 and the 22 October 1995 cannot be explained by either the prevailing wind direction or wind speed as they were similar in direction and magnitude at both sites. A possible explanation is that because the monitoring at the roadside was conducted on a Sunday there were fewer vehicles on the road in comparison to Thursday 19 October 1995. Therefore, the kerbside measurement site had higher mean ambient levels. The highest peak measurement, however, was recorded in the morning at roadside, possibly reflecting people attending church.

5.3.4.2 Time series plot of five minute mean concentrations

The pattern of reduction in pollutant levels with increasing distance away from the road is again not visible at site D with the highest value (8 ppm) recorded 15 metres from the road *(Figure 5.18 D)*. The roadside location produces the next highest peak value (7.4 ppm) *(Figure 5.18 A)* followed by the 7.5 metres site (4 ppm) *(Figure 5.18 C)*. The kerbside location *(Figure 5.18 C)* produced the lowest peak level with a value of 2.5 ppm.

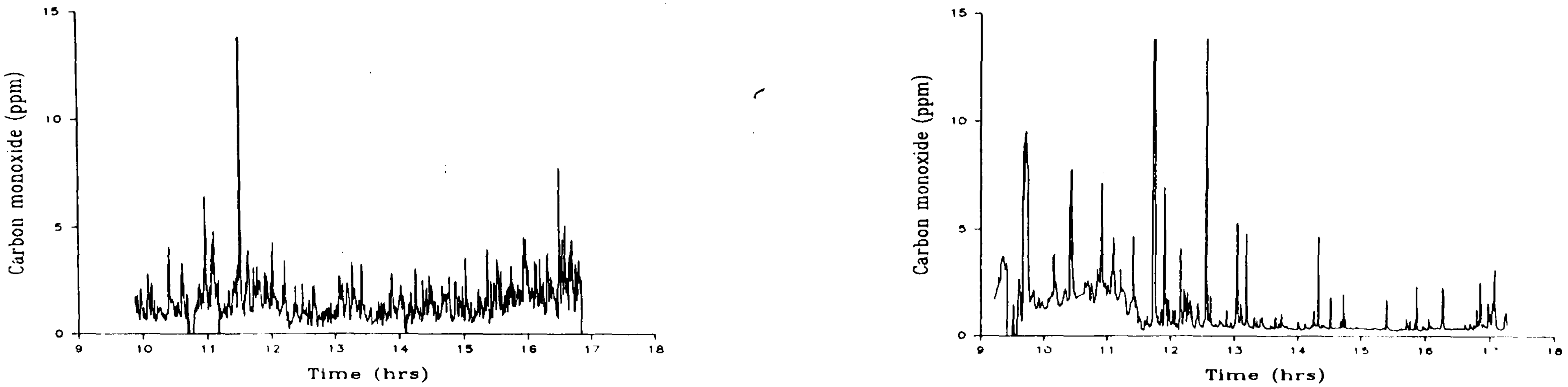
5.3.4.3 Time series plot of hourly mean concentrations

The hourly data *(Figure 5.19)* further demonstrates that there is no reduction in CO concentration with distance away from the road, as was apparent at the other sites. However, site D is similar to those previously discussed in that the WHO one hour exposure limit is not threatened, with 2.5 ppm being the highest one hour peak value recorded in Leicester. Moreover, any estimation of an eight-hour exposure value would be significantly less than the 10 ppm exposure guideline.



A Roadside (22 October 1995)

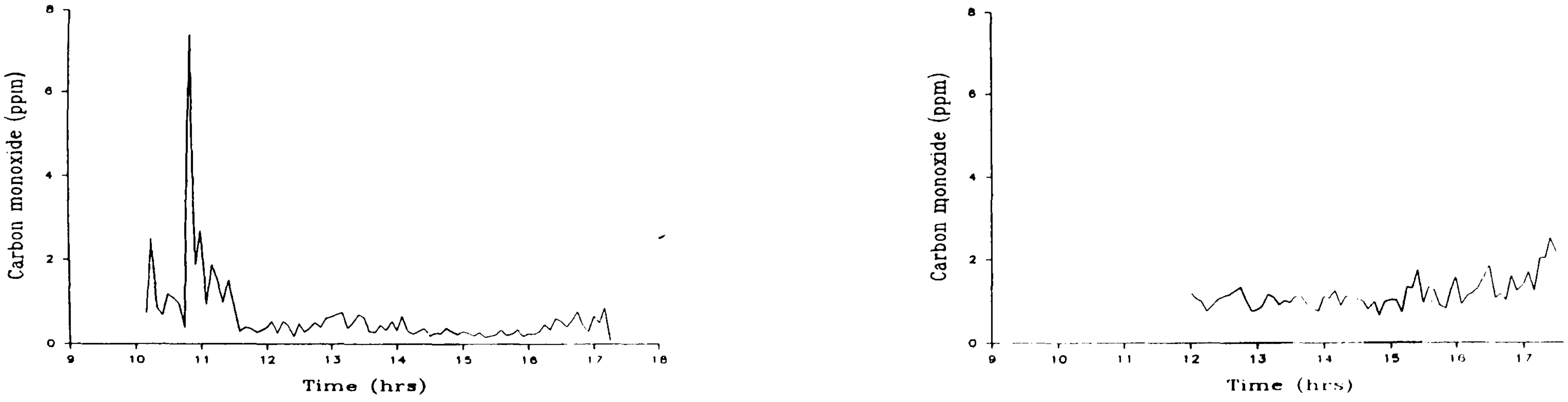
B Kerbside (19 October 1995)



C 7.5 metres (20 October 1995)

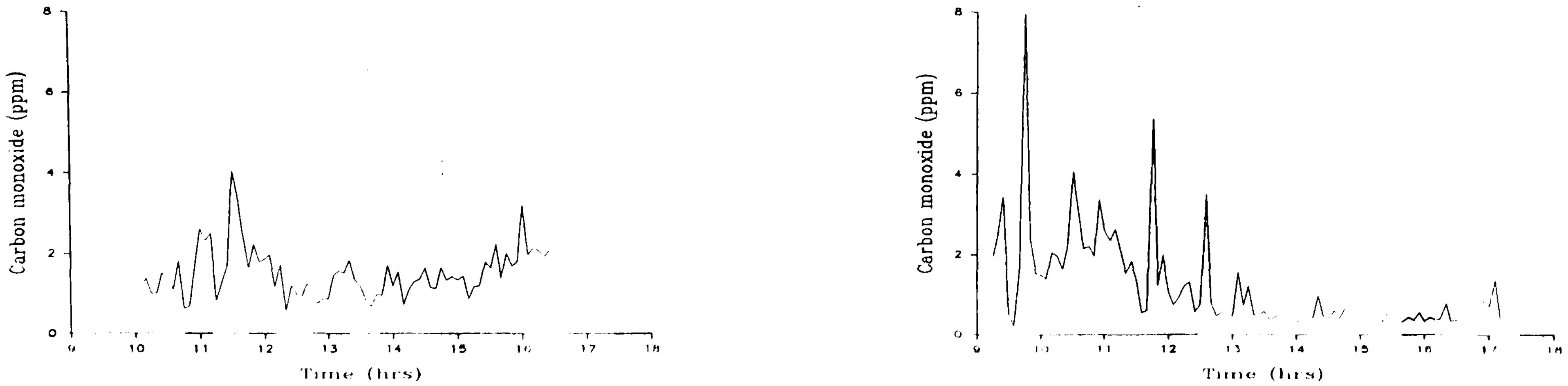
D 15 metres (21 October 1995)

Figure **5.17 (A-D)** Time series plots of raw CO data at roadside, kerbside, 7.5 metres and 15 metres from the road at site D.



A Roadside (22 October 1995)

B Kerbside (19 October 1995)



C 7.5 metres (20 October 1995)

D 15 metres (21 October 1995)

Figure 5.18 (A-D) Time series plots of five minute CO means at roadside, kerbside, 7.5 metres and 15 metres from the road at site D.

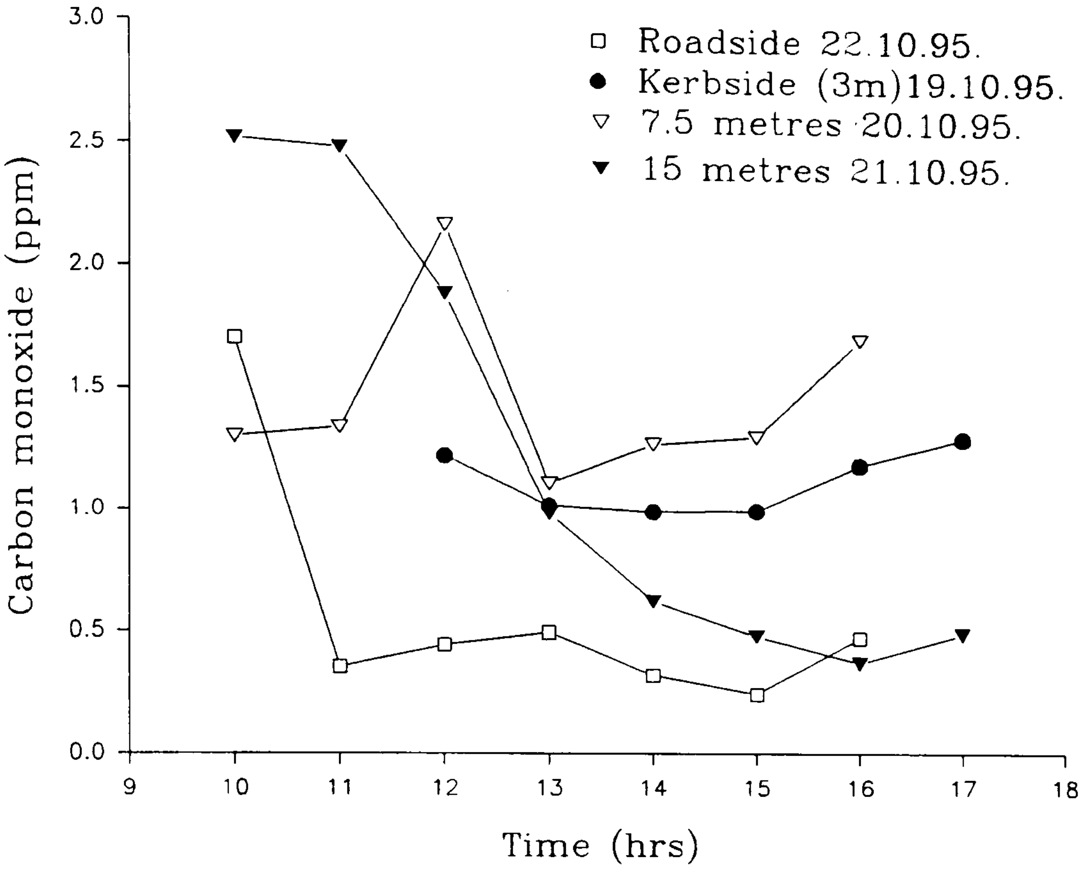
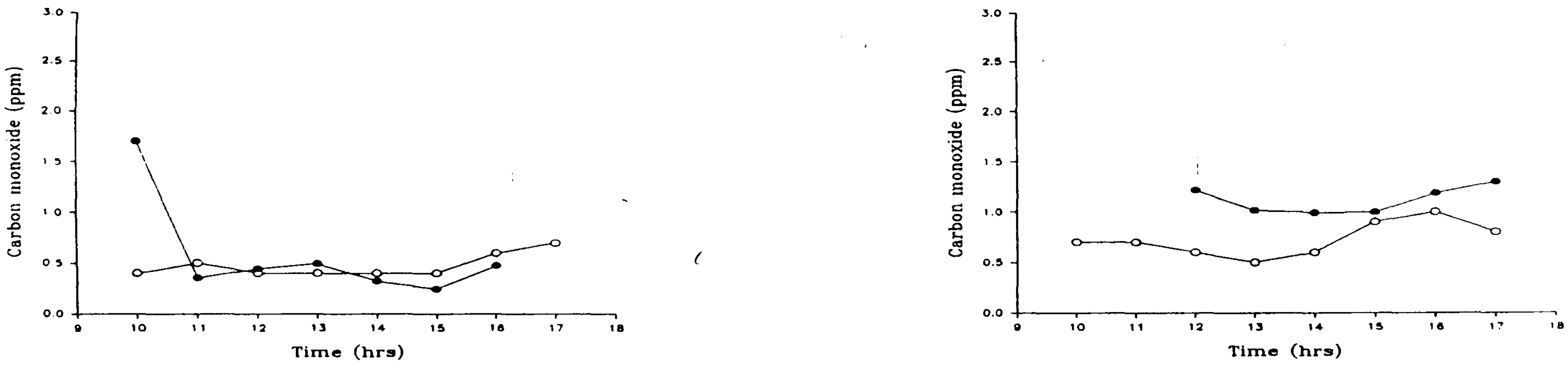


Figure 5.19 Hourly carbon monoxide concentrations at site D.

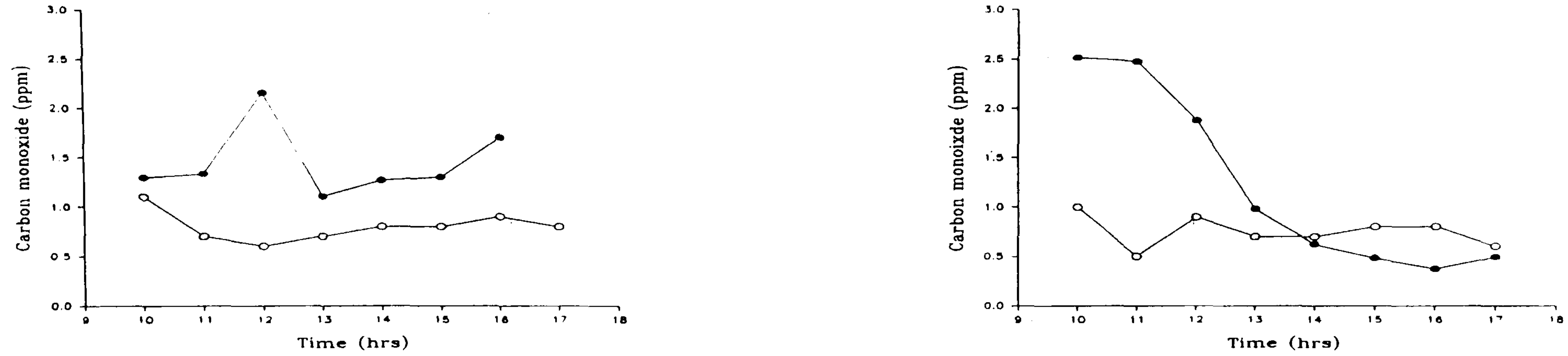
5.3.4.4 Comparison of the air quality at site D with Automated Urban Network data

Comparison can be made with the AUN site, identified as Leicester Centre (LEIC). This is an urban site (SK 587 040) located in a pedestrianised area of New Walk Centre, Welford Place, Leicester. The comparison of site D air quality data with that monitored at the AUN site is shown in *Figure 5.20 A-D*. The comparison of the data obtained at roadside with the AUN site *(Figure 5.20 A)* initially produces surprising results with very similar concentrations at low CO levels. One possible reason for the anomalously low concentrations at roadside is the very low traffic volumes encountered on Sunday 22 October and hence the similarity with the AUN urban background station. On all other days site D had predominantly higher values than the AUN site *(Figure 5.20 B-D)*, a slight exception occurring 15 metres from the road *(Figure 5.20 D)* when after 13:00 hours the two sites have similar values, possibly reflecting that at 15 metres from the road, air quality is approaching an urban background.



A Roadside (22 October 1995)

B Kerbside (19 October 1995)



C 7.5 metres (20 October 1995)

D 15 metres (21 October 1995)

Figure **5.20 (A-D)** Comparison of one hour mean CO levels at Site D (●), at roadside, kerbside, 7.5 metres and 15 metres from the road, to those found at the Leicester AUN (O) site.

5.3.5 Inter-site comparison of carbon monoxide air quality

Inter-site comparisons of daily mean carbon monoxide levels are displayed in **Table 5.1**. It is evident that sites A and C have similar carbon monoxide levels at kerbside, 7.5 metres and 15 metres from the road. The similarity in reduction in concentration with distance at the two sites is equally marked. Sites A and C are both in London and the similarity between sites could be explained by the overall similarity of the vehicle fleet as confirmed in Chapter 4 by the mean and median vehicle emissions.

Site B has the lowest concentrations, with the difference being so pronounced, that kerbside levels are lower than those measured at a distance of 15 metres at all other sites with the exception of site C where the difference is marginal. Site D is unusual in that the typical decrease in concentration with distance from the road does not occur, although the kerbside carbon monoxide levels are lower than at sites A and C.

Table 5.1 Inter-site comparison of daily mean carbon monoxide levels with distance from the road

Distance from the road (metres)	Site A	Site B	Site C	Site D
3 (Kerbside)	2.3	0.4	2.9	1.7
7.5	1.5	0.1	1.3	1.5
15	0.6	-	0.4	1.4

5.4 Nitrogen oxides air quality

The following sections include analyses of time series plots of nitric oxide and nitrogen dioxide air quality from the four survey sites and comparison with data obtained from the appropriate automated urban network site. This section concludes with a discussion of the differences and similarities between the four sites.

5.4.1 Site A (Bounds Green Road, Haringey)

5.4.1.1 Time series plot of raw nitric oxide data

The highest value recorded was measured at kerbside on 25 May 1994 with a peak value of 1077 ppb *(Figure 5.21 A)*. Measurements carried out at the same kerbside location on a different date (1 June 1994) produced a considerably lower maximum peak value of 592 ppb *(Figure 5.21 B)*. The discrepancy between the two kerbside measurement days is repeated for mean daily concentrations (see below) and is difficult to explain. Two factors which may be discounted are wind speed and wind direction as both were similar on each of the sampling days. Therefore, a possible explanation, further analysed in Chapter 6, could be that vehicle emissions were greater on 25 May 1994 compared to 1 June 1994.

The lowest nitric oxide concentrations were recorded 15 metres from the road with a maximum peak value of 83 ppb *(Figure 5.21 D)*. The site positioned 7.5 metres from the road produced an isolated maximum value of 503 ppb *(Figure 5.21 C)* which is similar in magnitude to the peak measurement recorded at kerbside on 1 June 1994. However, daily mean concentrations are substantially different with a mean concentration of 128 ppb found at kerbside (1 June 1994) and a mean concentration of only 55 ppb found 7.5 metres from the road, reflecting the exceptional nature of the maximum peak value measured 7.5 metres from the road. The lowest daily mean concentration (15 ppb) was recorded 15 metres from the road and the highest (286 ppb) at kerbside (25 May 1994). Thus, the daily mean concentrations clearly identify the trend of reduction in pollution levels with distance from the road.

The expectation that the behaviour of nitric oxide would be similar to carbon monoxide, as both pollutants are primarily produced by the motor vehicle, was to some degree correct in that nitric oxide levels decline with distance from the road as did carbon monoxide levels. However, whereas the highest carbon monoxide levels at kerbside at site A may have corresponded to midday and early evening rush hour traffic flows this does not seem to be reflected so strongly by peak nitric oxide levels.

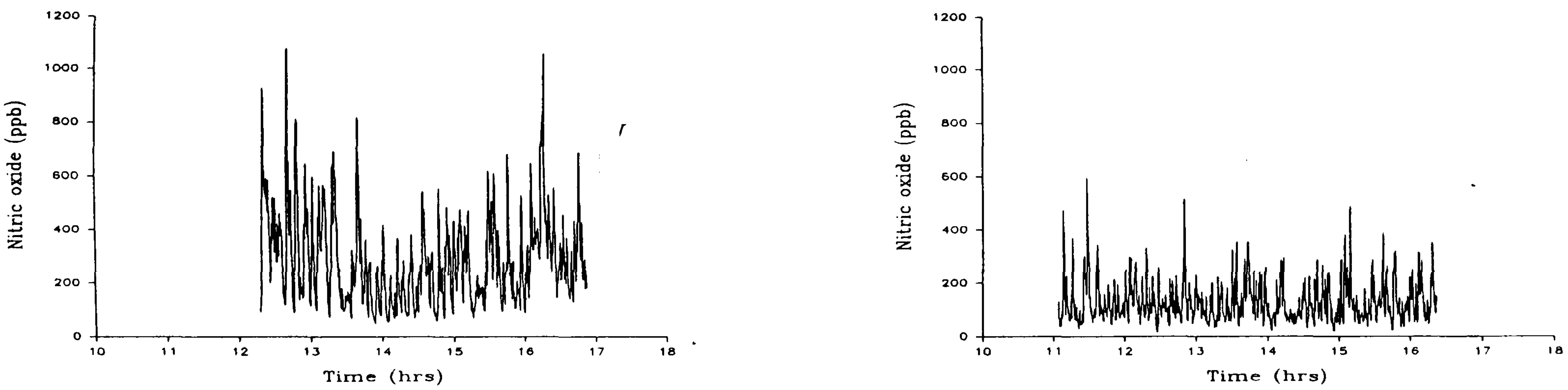
5.4.1.2 Time series plot of raw nitrogen dioxide data

The time series trace of the raw data for nitrogen dioxide differs in form from both carbon monoxide and nitric oxide by following a much flatter pattern, with less extreme peaks and troughs. Although as with carbon monoxide and nitric oxide the highest peak concentrations were recorded at kerbside *(Figure 5.22 A-B)* with values of 138 ppb (25 May 1994) and 92 ppb (1 June 1994) and increasingly lower values (7.5 metres - 64 ppb; 15 metres - 28 ppb) at greater distances from the road *(Figure 5.22 C-D)*.

The distance of the site from the road also appears to have less effect on NO_2 concentrations than was noticeable for CO and NO. The relative difference in daily mean ambient concentrations between the kerbside site (45 ppb), the site positioned 7.5 metres from the road (33 ppb) and the site positioned 15 metres from the road (11 ppb) is much smaller for NO_2 than for CO and NO. A possible explanation for this, which is further discussed in section 5.4.1.3, is that because NO_2 can be formed in the atmosphere by the oxidation of NO, unlike NO and CO which are primarily formed by the motor vehicle, the effect of dispersion in reducing pollutant concentrations with distance is somewhat counteracted producing an increase in NO_2 concentrations relative to CO and NO.

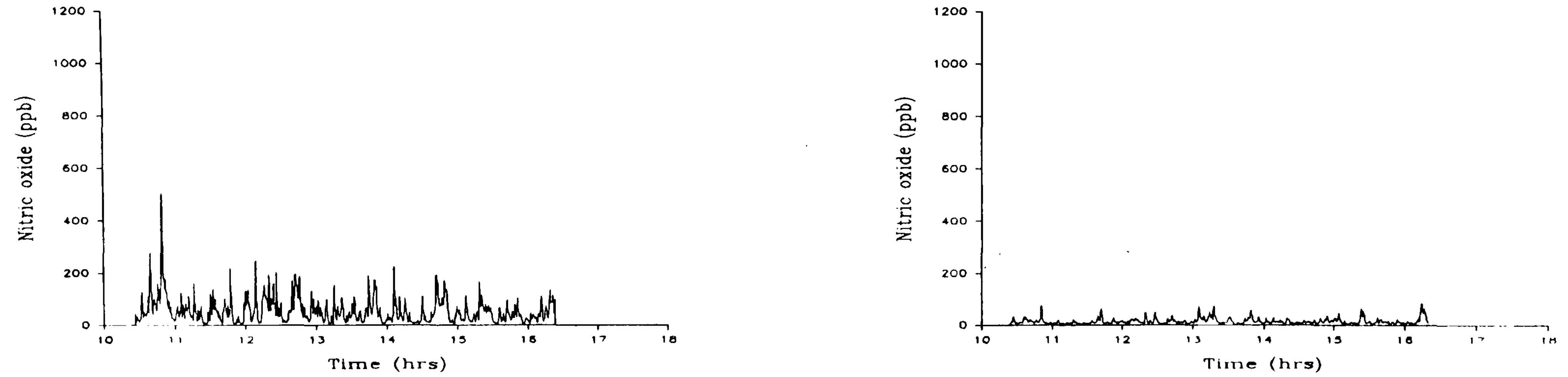
5.4.1.3 Time series plots of five minute mean nitric oxide and nitrogen dioxide concentrations

The pattern of reduction in pollutant levels with increasing distance away from the road has been demonstrated for raw data for both nitric oxide and nitrogen dioxide and is repeated for the five minute mean measurements. The highest nitric oxide concentrations were found at kerbside with values of 605 ppb *(Figure 5.23 A)* and 272 ppb *(Figure 5.23 B)* with correspondingly reduced values, at greater distances from the road - 175 ppb (7.5 metres) & 40 ppb (15 metres) *(Figure 5.23 C-D)*. The highest nitrogen dioxide concentrations were also found at kerbside with values of 90 ppb *(Figure 5.23 A)* and 68 ppb *(Figure 5.23 B)*. The site positioned 7.5 metres from the road had an intermediate maximum peak value of 51 ppb *(Figure 5.23 C)* and the site 15 metres from the road had the lowest peak value of 20 ppb *(Figure 5.23 D)*.



A Kerbside (25 May 1994)

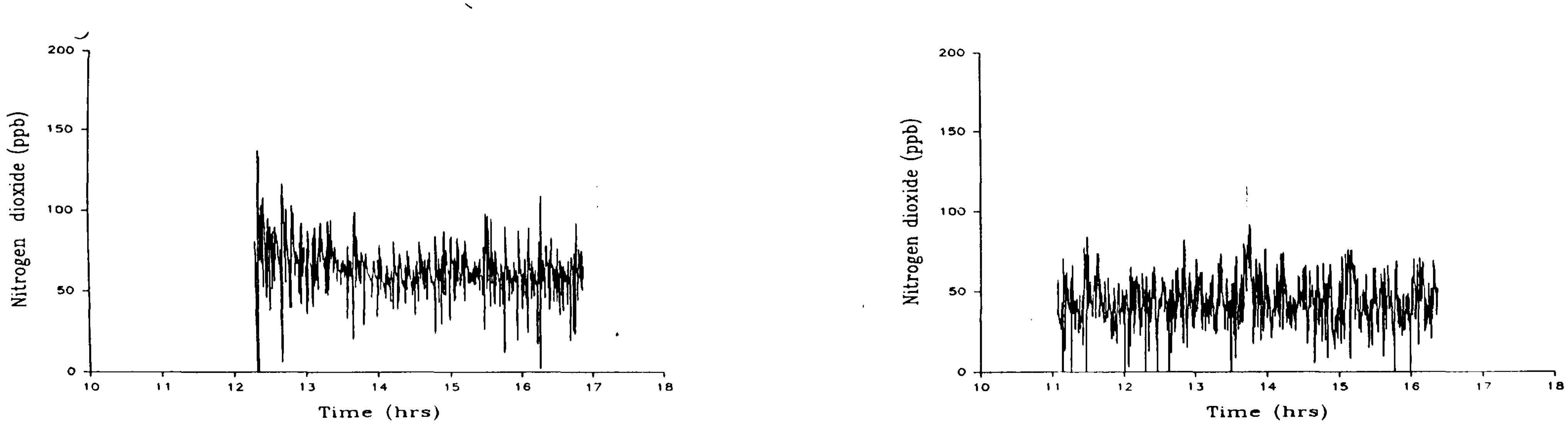
B Kerbside (1 June 1994)



C 7.5 metres (31 May 1994)

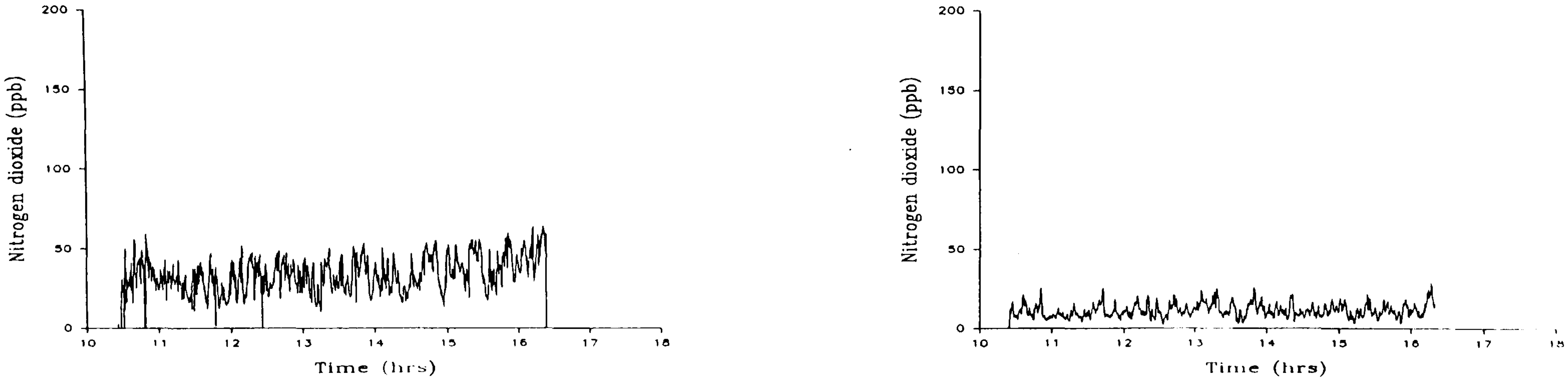
D 15 metres (27 May 1994)

Figure 5.21 (A-D) Time series plots of raw NO data at kerbside, 7.5 metres and 15 metres from the road at site A.



A Kerbside (25 May 1994)

B Kerbside (1 June 1994)



C 7.5 metres (31 May 1994)

D 15 metres (27 May 1994)

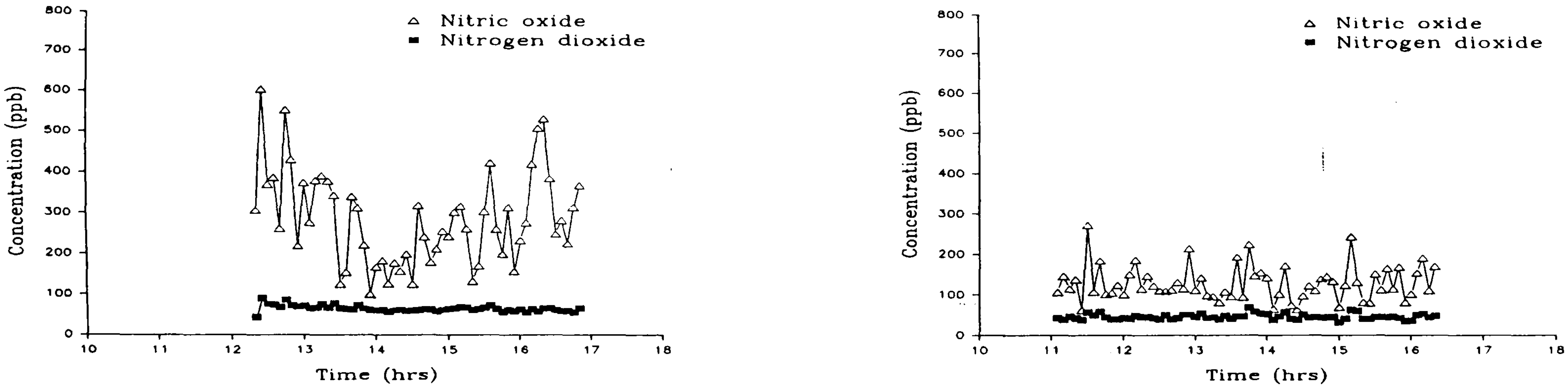
Figure 5.22 (A-D) Time series plots of raw NO_2 data at kerbside, 7.5 metres and 15 metres from the road at site A.

Nitrogen dioxide is formed primarily through the oxidation of nitric oxide to nitrogen dioxide in the atmosphere and therefore, the expectation is for higher concentrations of nitrogen dioxide, as nitric oxide is oxidized, at greater distances from the road. However, as can be seen from *Figures 5.23 A-D* and *Figures 5.22 A-D* this does not occur over a distance of 15 metres with both maximum peak measurements and daily mean concentrations of nitrogen dioxide lowest at this position and highest at the kerbside site. The most likely explanation for the high nitrogen dioxide concentrations at kerbside is that the emission of nitrogen dioxide by passing vehicles is greater than the production by the oxidation of nitric oxide in the atmosphere. Moreover, nitrogen dioxide concentrations are reduced at greater distances from the road because of the extra dispersion that has taken place.

However, when five minute mean nitrogen dioxide concentrations *(Figure 5.23 A-D)* are compared directly to nitric oxide concentrations, there is a relative rise in NO_2 concentrations with distance from the road that can be explained by the oxidation of nitric oxide to nitrogen dioxide. If the ratios of the two pollutants with distance are analysed *(Table 5.2)* the relative increase in nitrogen dioxide levels becomes even more apparent with a ratio of 4.5:1 at kerbside reducing to a ratio approaching 1:1 at the site positioned 15 metres from the road. The increase of NO_2 concentrations relative to NO helps to explain the decrease in influence that distance from the road has upon air quality for NO_2, as described in section 5.4.1.2.

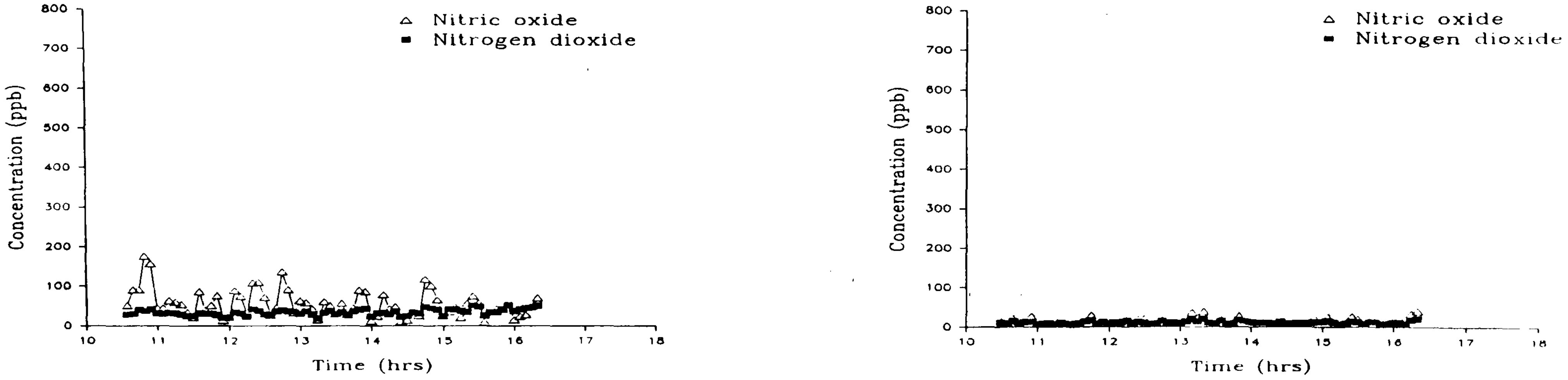
Table 5.2 Change in daily NO/NO_2 ratios with distance from the road at site A

Distance (metres)	NO/NO_2 Ratio
Kerbside (3 m)	4.5:1
Kerbside (3 m)	2.9:1
7.5	1.7:1
15	1.4:1



A Kerbside (25 May 1994)

B Kerbside (1 June 1994)



C 7.5 metres

D 15 metres

Figure 5.23 (A-D) Time series plots of five minute means for NO and NO_2 at kerbside, 7.5 and 15 metres from the road at site A.

5.4.1.4 Time series plots of hourly mean nitric oxide and nitrogen dioxide concentrations

The plotted data *(Figure 5.24 & 5.25)* again demonstrates the reduction in concentration of both pollutants with distance away from the road.

Unlike carbon monoxide the European Union have adopted limit and guide values for NO_2 (Chapter 3) as set out in *Directive 85/203/EEC*; there are no limit and guide values for NO because it is thought to be harmless to human health. The limit value for NO_2 is 105 ppb (as a 98th percentile annual 1 hour average) and is set at a level designed to protect human health. The guide values are set at 26 ppb, as 50th percentile of annual hourly averages, and 71 ppb, as a 98th percentile of annual hourly averages, with the intention of enhancing health prospects in the longer term. The WHO have set a 1-hour guideline value of 209 ppb and a 24-hour guideline value of 78 ppb which have been established to take into account both acute and chronic effects.

Although the air quality data are not strictly comparable to the European Union regulations, due to the annual percentile element of the regulations, it is nevertheless appropriate to see if the concentrations from site A exceed those concentrations specified as limit and guide values by the EU. No such problem exists when comparing site A data with the WHO guide values although no comparison can be made to the 24-hour guideline due to the limited time scale of the monitoring.

The maximum hourly peak concentration measured at site A was 73 ppb which is significantly less than both the EU 1-hour limit value and the WHO 1-hour guide value. If comparison is made to the EU guide values, the 71 ppb figure was exceeded only once, at kerbside on the 25 May 1994, although the 26 ppb figure was exceeded on all days except the 27 May 1994.

A daily summary of UK air quality, provided by the National Environmental Technology Centre *(Stedman & Willis 1994)*, has been available to the general public since 1990. Regional summaries are updated every hour together with a twice daily air quality forecast. At present, hourly average NO_2, O_3 and SO_2 concentrations are set against banded criteria (**Table 5.3**). The resulting air quality is defined as either 'very good', 'good', 'poor' or 'very poor'. A forecast is also issued for the next 24 hours together with health advice if levels become 'very poor'.

Table 5.3 Banded criteria for air quality *(Bower et al. 1995)*

Description	Nitrogen dioxide (ppb)	Ozone (ppb)	Sulphur dioxide (ppb)
very good	<50	<50	<60
good	50-99	50-89	60-124
poor	100-299	90-179	125-399
very poor	>= 300	>=180	>=400

If the measured NO_2 concentrations at site A are compared to these banded criteria, the air quality would be classified as 'very good' with the exception of the 25 May 1994 when it falls into the 'good' category.

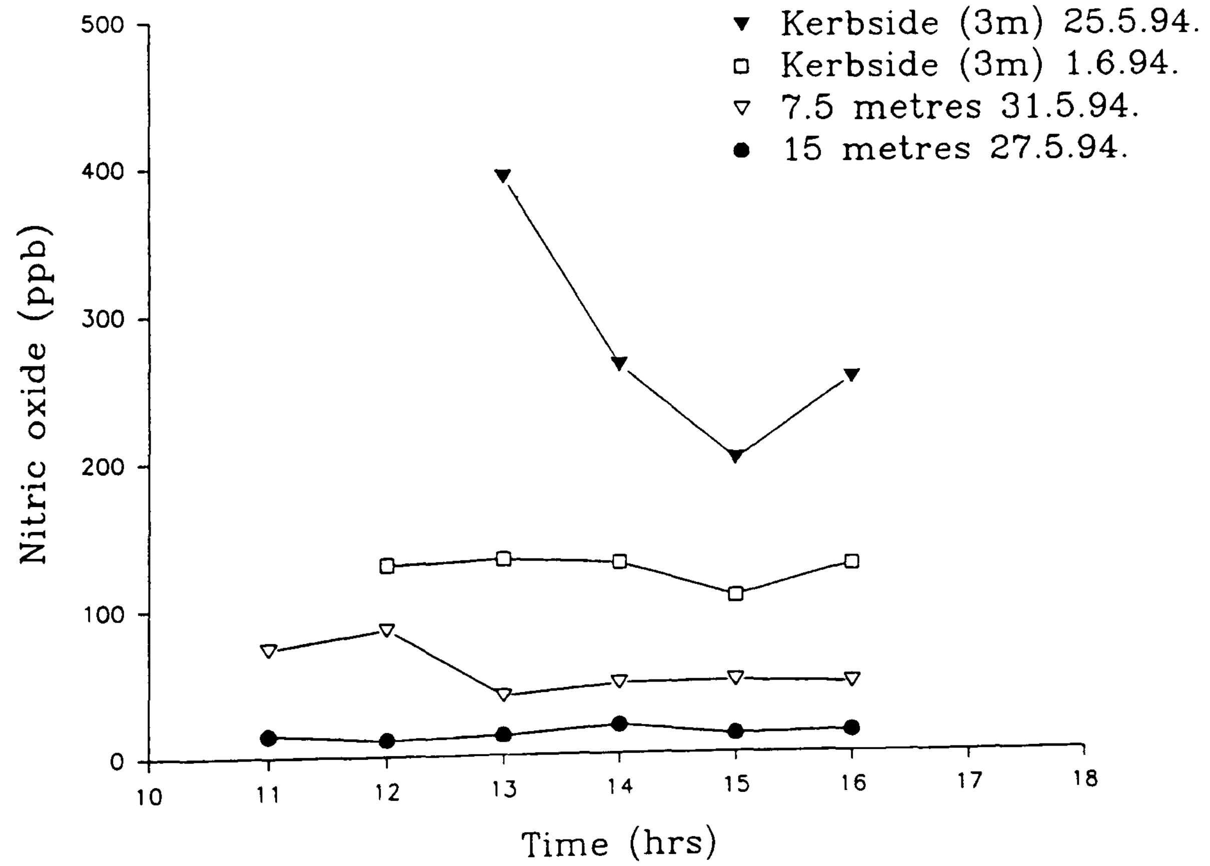


Figure 5.24 Hourly nitric oxide concentrations at site A.

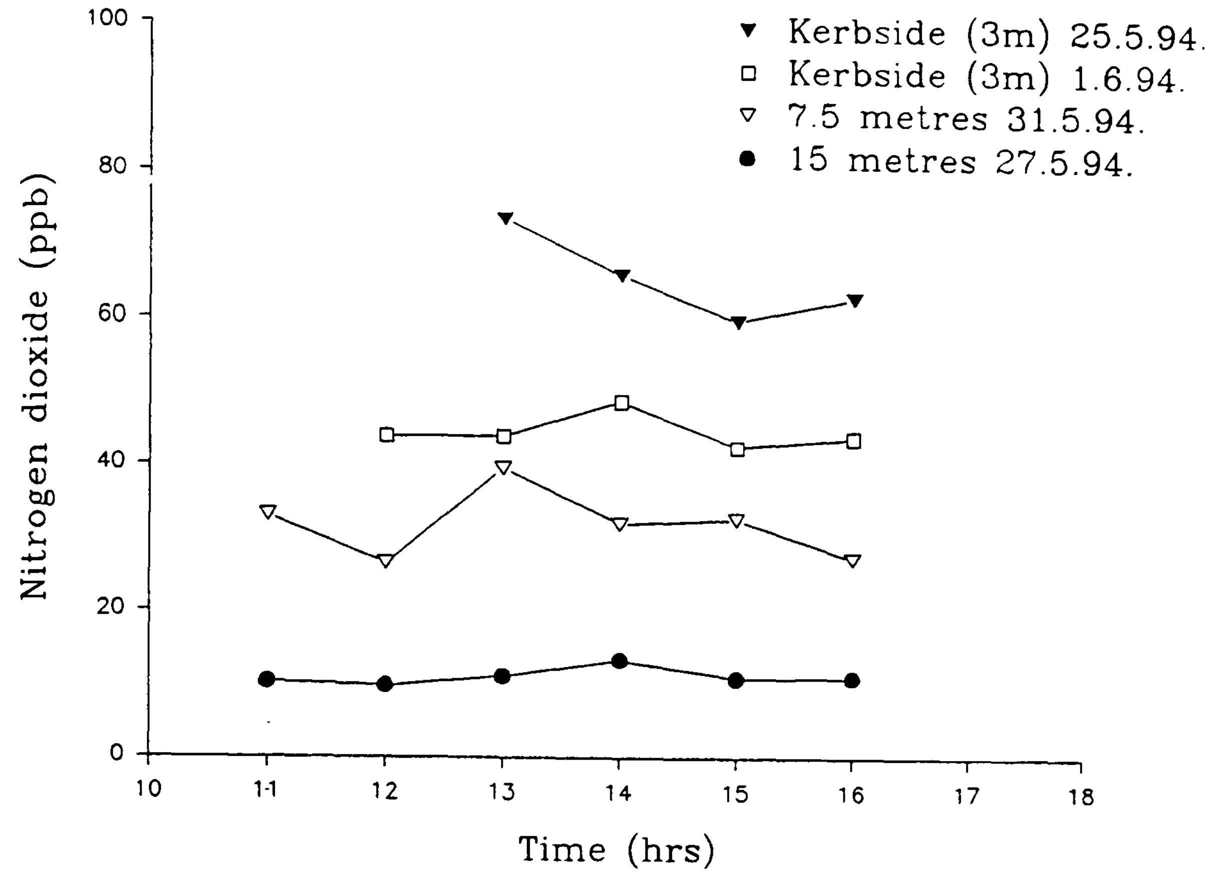


Figure 5.25 Hourly nitrogen dioxide concentrations at site A.

5.4.1.5 Comparison of the air quality at site A with Automated Urban Network data

Those sites (West London, Cromwell Road, London Bloomsbury and London Bexley) comparable to site A for CO air quality are also comparable for NO_x air quality.

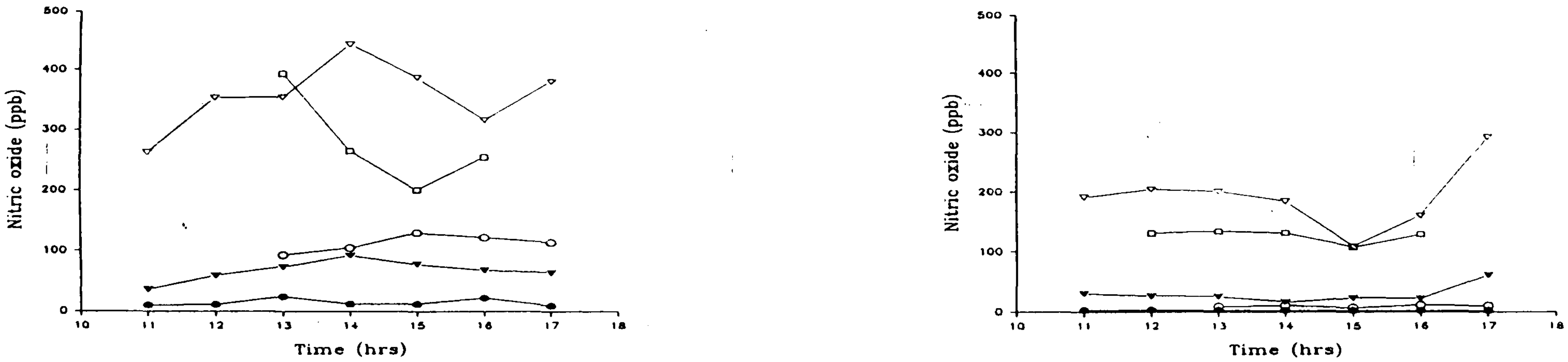
The comparison of site A nitric oxide air quality data with that obtained at the comparable AUN sites is shown in *Figure 5.26 A-D*. Comparison of the data gained from kerbside on 25 May 1994 *(Figure 5.26 A)* shows West London, London Bloomsbury and London Bexley to have lower nitric oxide concentrations, but Cromwell Road to have higher concentrations than site A. This pattern reflects that found for carbon monoxide with the sites further from a road and therefore an emission source, having better air quality than site A and the Cromwell Road site having worse air quality because of the generally higher traffic flows found there. Comparison of the data gained from kerbside on the 1 June 1994 *(Figure 5.26 B)* shows a similar scenario, with nitric oxide levels at site A generally lower than those at Cromwell Road while remaining higher than the urban background sites.

Nitric oxide data obtained from the site positioned 7.5 metres from the road has been compared to AUN data as shown in *Figure 5.26 C.* The trend once again parallels that found for carbon monoxide with the two sites nearest to a road, site A and Cromwell Road, having the highest ambient concentrations.

A comparison of the nitric oxide data monitored at the site positioned 15 metres from the road with the AUN data is shown in *Figure 5.26 D.* This shows the nitric oxide levels at site A to have reached an urban background concentration, as was the case for carbon monoxide levels at this location relative to the West London, London Bloomsbury and London Bexley sites.

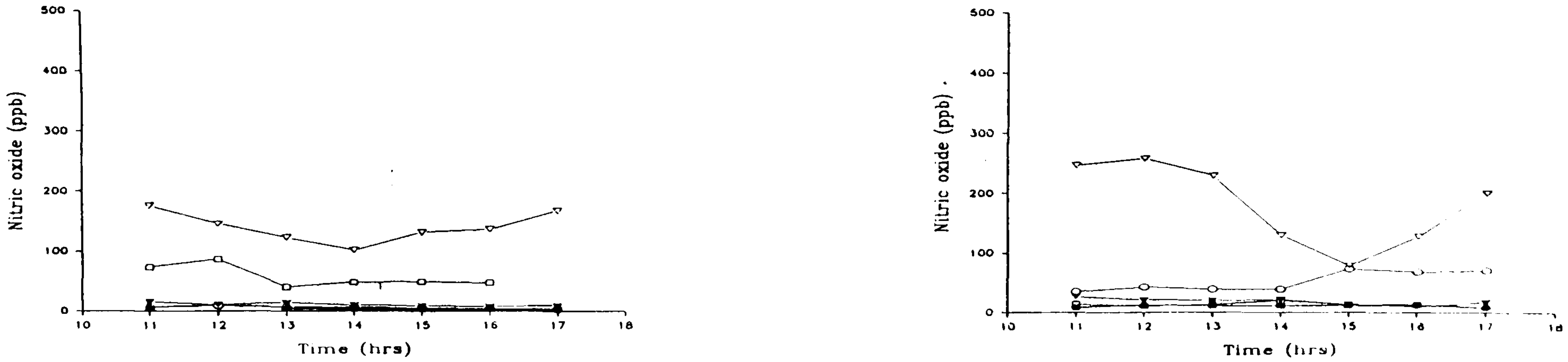
Nitrogen dioxide air quality data from site A have been compared to that obtained at the comparable AUN sites and are shown in *Figure 5.27 A-D.* Comparison of the data gained from kerbside *(Figure 5.27 A-B)* shows a markedly different situation to that found for nitric oxide and carbon monoxide. Unlike nitric oxide and carbon monoxide, the nitrogen dioxide concentrations at the West London, London Bloomsbury and London Bexley sites are much closer in value to site A and Cromwell Road. This may arise because of the increased oxidation of nitric oxide to nitrogen dioxide as the nitric oxide moves further away from its emission source.

The comparison of the nitrogen dioxide data obtained at the site positioned 7.5 metres from the road with the AUN data is shown in *Figure 5.27 C.* It demonstrates a similar trend to that seen at kerbside with the major difference being the reduction in nitrogen dioxide concentration at site A caused by dispersion. Nitrogen dioxide data obtained from the site positioned 15 metres from the road are compared to the AUN data as shown in *Figure 5.27 D.* This shows London Bloomsbury and Cromwell Road to have the highest concentrations, only marginally greater than the West London site, indicating that the production of nitrogen dioxide from the oxidation of nitric oxide can equal that from motor vehicles at 35 metres from the kerbside. However, the nitrogen dioxide concentration at site A continued to decrease, possibly because of increased dispersion or possibly a limitation on the amount of oxidants required to convert NO to NO_2.



A Kerbside (25 May 1994)

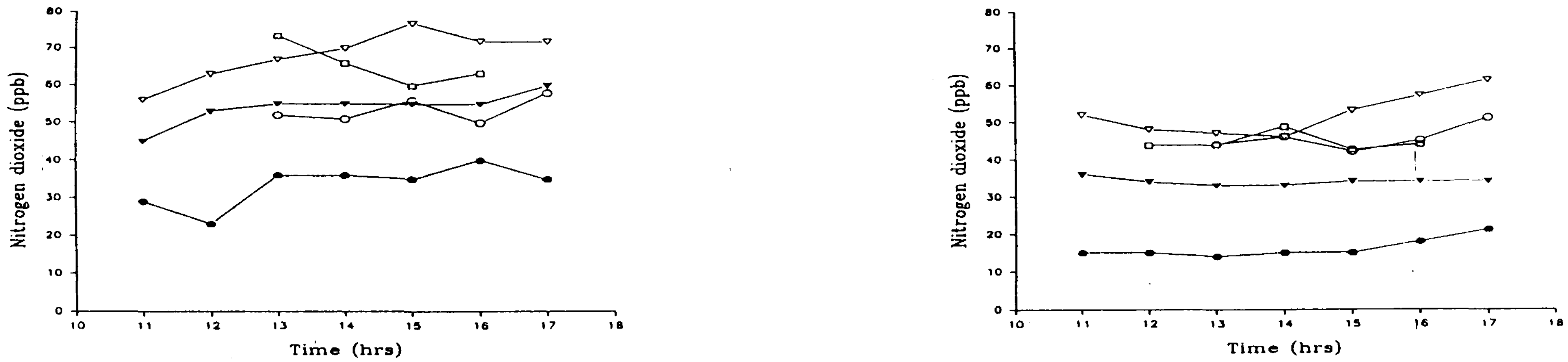
B Kerbside (1 June 1994)



C 7.5 metres (31 May 1994)

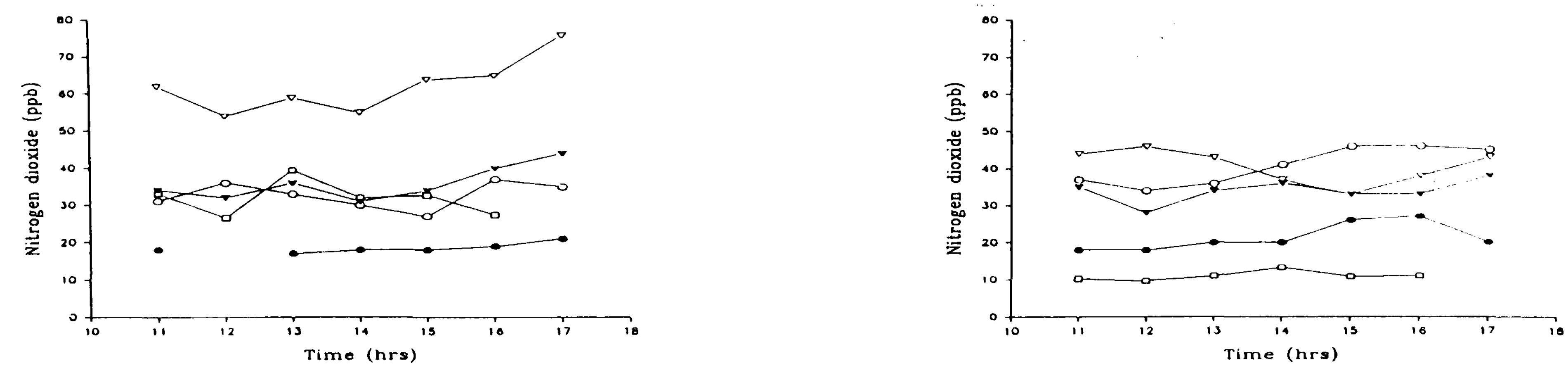
D 15 metres (27 May 1994)

Figure 5.26 (A-D) Comparison of one hour mean NO levels at Site A (□), at kerbside, 7.5 metres and 15 metres from the road, to those found at AUN sites WL (▾) BEX (●) CRD (▿) and CLL2 (○).



A Kerbside (25 May 1994)

B Kerbside (1 June 1994)



C 7.5 metres (31 May 1994)

D 15 metres (27 May 1994)

Figure 5.27 (A-D) Comparison of one hour mean NO_2 levels at Site A (□), at kerbside, 7.5 metres and 15 metres from the road, to those found at AUN sites WL (▾) BEX (●) CRD (▿) and CLL2 (○).

5.4.2 Site B (Dixons Bank, Middlesborough)

5.4.2.1 Time series plot of raw nitric oxide data

The highest value recorded was measured at kerbside on 10 June 1994 with a peak value of 226 ppb *(Figure 5.28 A)*. Measurements carried out at the same kerbside location on a different date (7 June 1994) produced a considerably lower maximum peak value of 92 ppb *(Figure 5.28 B)*. The difference in concentrations between kerbside measurement days becomes more extreme when daily mean levels are compared with a value of 77 ppb calculated for 10 June 1994 in contrast to a value of only 13 ppb for 7 June 1994. A similar scenario was recorded for CO at site B with 10 June 1994 having considerably higher ambient concentrations than 7 June 1994. An explanation for this was given in section 5.3.2.1, based on a shift in wind direction and wind speed between the two kerbside measurement days which appeared to be responsible for the changes in ambient levels. It is reasonable to assume that this is also the reason for the changes in ambient NO levels.

The lowest concentrations, as expected, were recorded 15 metres from the road with a highest value of 13 ppb *(Figure 5.28 D)* and a daily mean value of 2 ppb. The site positioned 7.5 metres from the road produced a maximum value of 88 ppb and a daily mean of 10 ppb *(Figure 5.28 C)* which are both similar in magnitude to those recorded on 7 June 1994. This similarity is unusual but could possibly be explained by the different prevailing wind directions on 7 June 1994 and 8 June 1994 (7.5 metres from the road). The prevailing wind direction on 7 June 1994 was from upwind of the source, relative to the ambient air monitors, thereby reducing ambient concentrations. However, on 8 June 1994 the prevailing wind direction was from downwind of the source, relative to the ambient air monitors, thereby increasing ambient concentrations and leading to a similarity in pollutant levels for the two sampling days.

5.4.2.2 Time series plot of raw nitrogen dioxide data

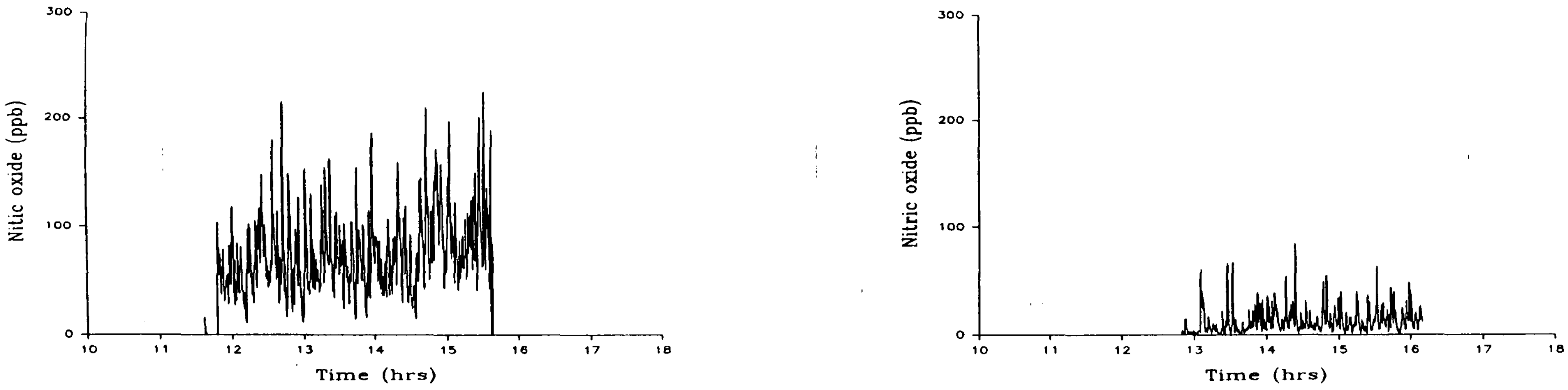
The time series trace of the raw data for NO_2 differs in form to that of both CO and NO at site B by following a flatter pattern with less extreme peaks and troughs *(Figure 5.29)*. However, it is similar to the NO_2 trace for site A but at considerably lower concentrations. The reduction in ambient concentrations with distance from the road, noted for CO and NO is again apparent with the highest value (30 ppb - *Figure 5.29 A)* recorded at kerbside on 10 June 1994. The lowest NO_2 concentrations were recorded 15 metres from the road with the highest value being 8 ppb *(Figure 5.29 C)*. The site positioned 7.5 metres from the road produced a maximum value of 18 ppb *(Figure 5.29 B)*. Mean daily values follow a similar trend with the highest average concentration of 15 ppb (10 June 1994) being measured at kerbside and the lowest concentration of 3 ppb, recorded 15 metres from the road. The intermediate distance of 7.5 metres had an intermediate concentration of 5 ppb.

Due to an equipment malfunction on 7 June 1994 (kerbside) no signal was received by the data acquisition equipment from the ambient air monitors for the NO_2 or NO_x channels although the CO and NO channels were functioning normally. The problem with the NO_2 and NO_x channels occurred because of breaks in the cable connecting the air quality monitor to the data acquisition equipment.

5.4.2.3 Time series plots of five minute mean nitric oxide and nitrogen dioxide concentrations

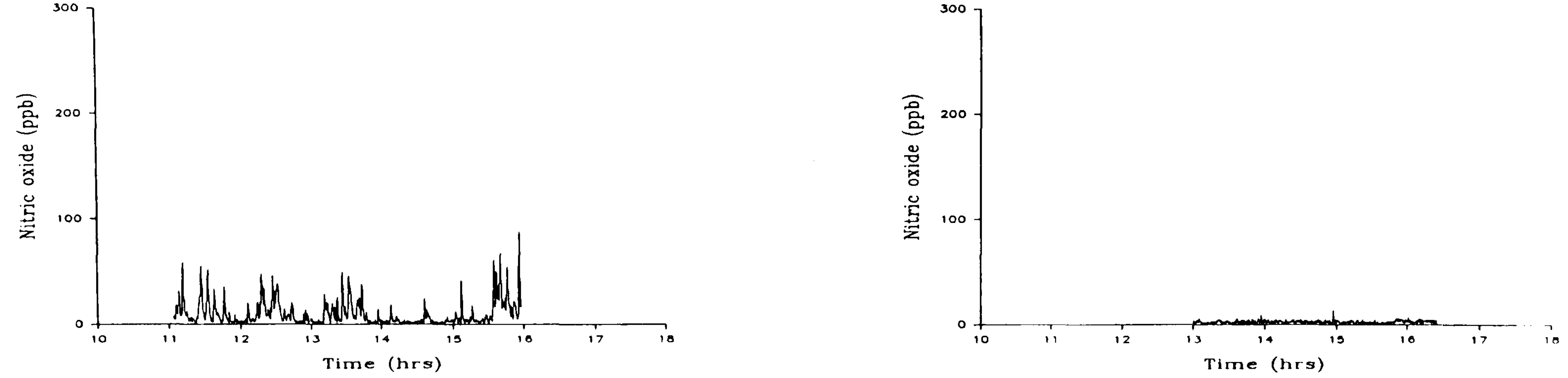
The pattern of reduction in pollutant levels with increasing distance away from the road is demonstrated for nitric oxide and to a lesser degree nitrogen dioxide. The highest nitric oxide concentration was found at kerbside with maximum peak value of 129 ppb *(Figure 5.30 A)*. The lowest daily maximum peak value (5 ppb) was found 15 metres from the road *(Figure 5.30 D)* with an intermediate peak value of 34 ppb found 7.5 metres from the road *(Figure 5.30 C)*.

However, the second kerbside measurement day *(Figure 5.30 B)* produced a maximum peak value of only 29 ppb very similar in value to that found 7.5 metres from the road. A possible explanation for this similarity was given in section 5.4.2.1 but to summarise it is thought that the change in the prevalent wind direction between the two sampling days was responsible for their similar pollutant levels. The highest nitrogen dioxide concentrations were found at kerbside with a maximum peak value of 24 ppb *(Figure*



A Kerbside (10 June 1994)

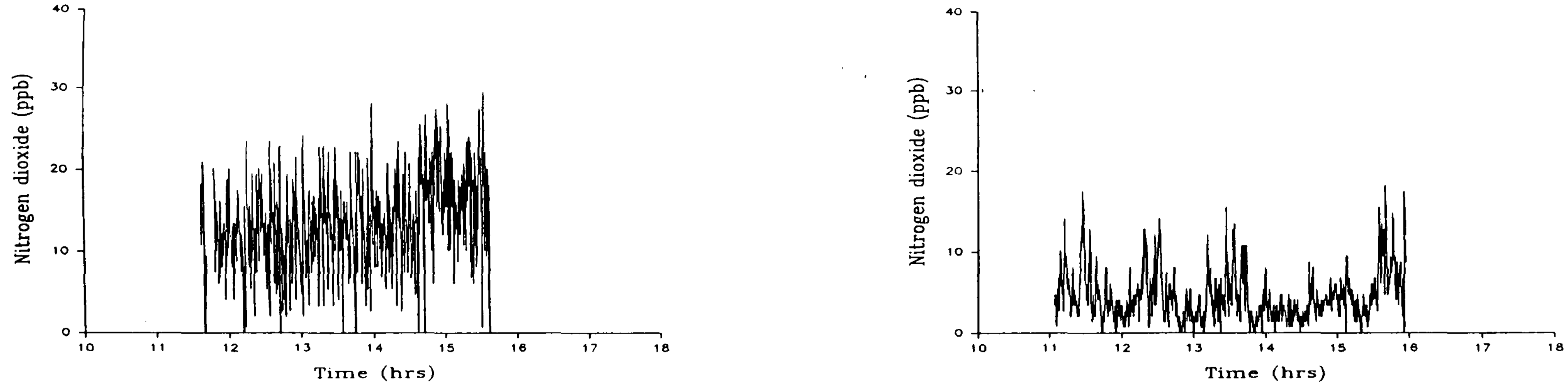
B Kerbside (7 June 1994)



C 7.5 metres (8 June 1994)

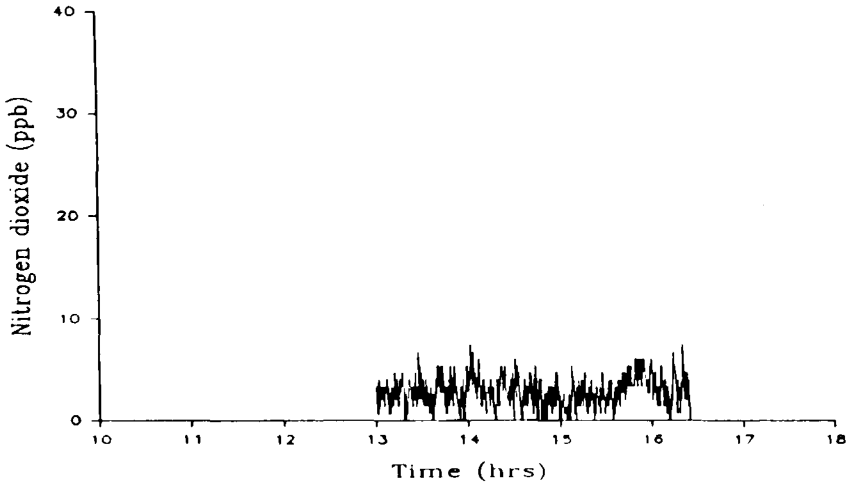
D 15 metres (9 June 1994)

Figure 5.28 (A-D) Time series plots of raw NO data at kerbside, 7.5 metres and 15 metres from the road at site B.



A Kerbside (10 June 1994)

B 7.5 metres (8 June 1994)



C 15 metres (9 June 1994)

Figure 5.29 (A-C) Time series plots of raw NO_2 data at kerbside, 7.5 metres and 15 metres from the road at site B.

5.30 A) with correspondingly reduced values, at greater distances from the road - 11 ppb (7.5 metres) and 5 ppb (15 metres) *(Figure 5.30 C-D)*.

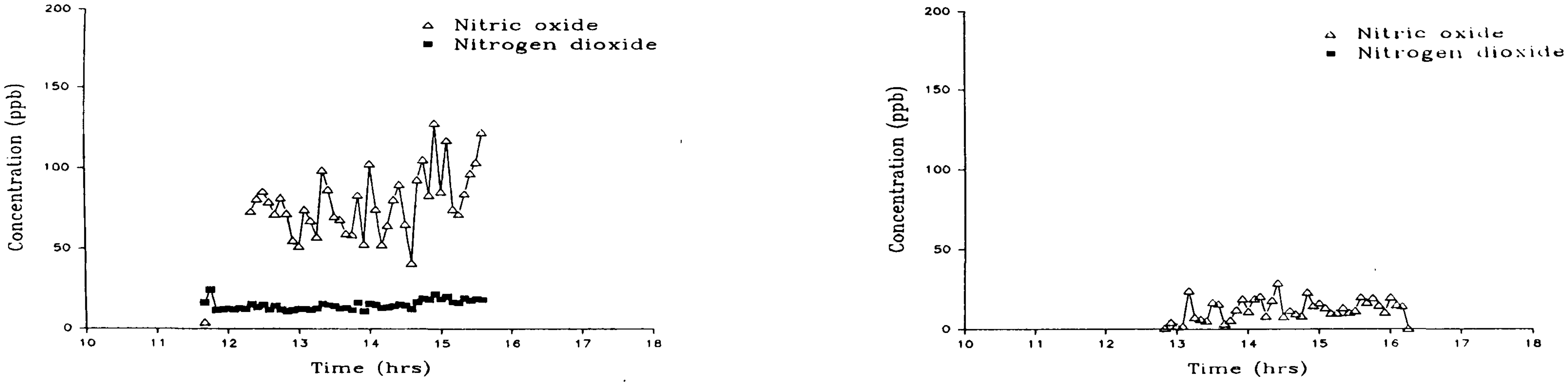
As was observed at site A, higher nitrogen dioxide concentrations are not found at greater distances from the road. The highest daily mean concentration of nitrogen dioxide was found at kerbside, with a value of 15 ppb and the lowest at 15 metres from the road, with a value of 3 ppb. However, as with site A (section 5.4.1.3), it is apparent that there is a rise in nitrogen dioxide concentrations relative to nitric oxide at greater distances from the road *(Figure 5.30 A-D)*. If the ratios of the two pollutants with distance are analysed *(Table 5.4)* this increase becomes even more apparent such that at 15 metres from the road the ambient concentration of NO_2 is greater than that of NO.

Table 5.4 Change in daily NO/NO_2 ratios with distance from the road at site B

Distance (metres)	NO/NO_2 Ratio
Kerbside (3 m)	5.1:1
7.5	2.0:1
15	0.67:1

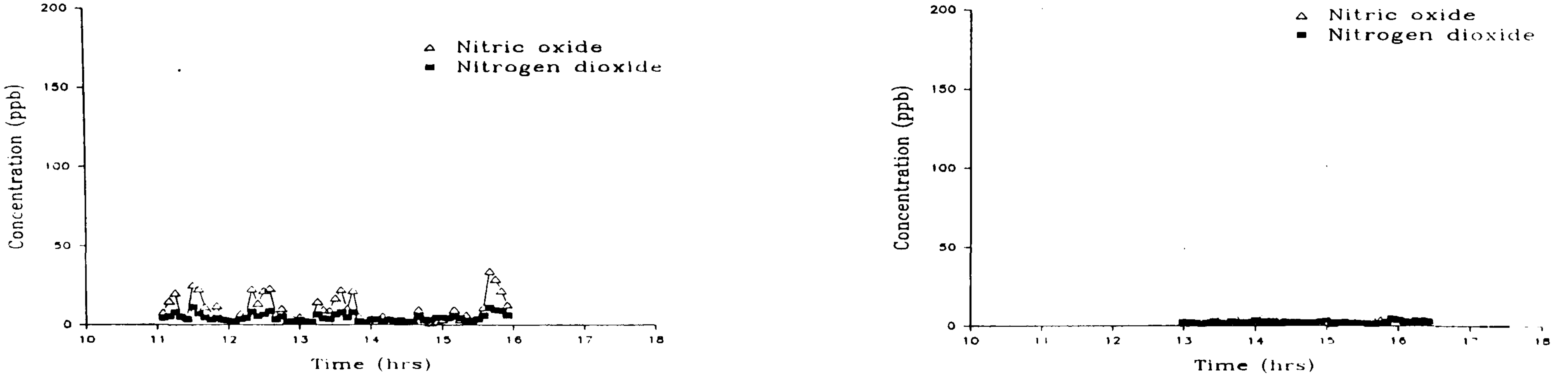
5.4.2.4 Time series plot of hourly mean nitric oxide and nitrogen dioxide concentrations

The plotted data *(Figures 5.31 & 5.32)* differ significantly from the hourly data for site A with site B having much lower ambient levels of both NO and NO_2. The reduction in pollutant levels with distance from the road is particularly rapid at site B such that there is little difference between pollutant levels at distances of 7.5 metres and 15 metres from the road. This is possibly related to low vehicle emissions which rapidly disperse to an urban background level before 15 metres from the road is reached, and therefore the typical decrease in concentration with distance is not apparent.



A Kerbside (10 June 1994)

B Kerbside (7 June 1994)



C 7.5 metres (8 June 1994)

D 15 metres (9 June 1994)

Figure 5.30 (A-D) Time series plots of five minute NO and NO_2 means at kerbside, 7.5 and 15 metres from the road at site B.

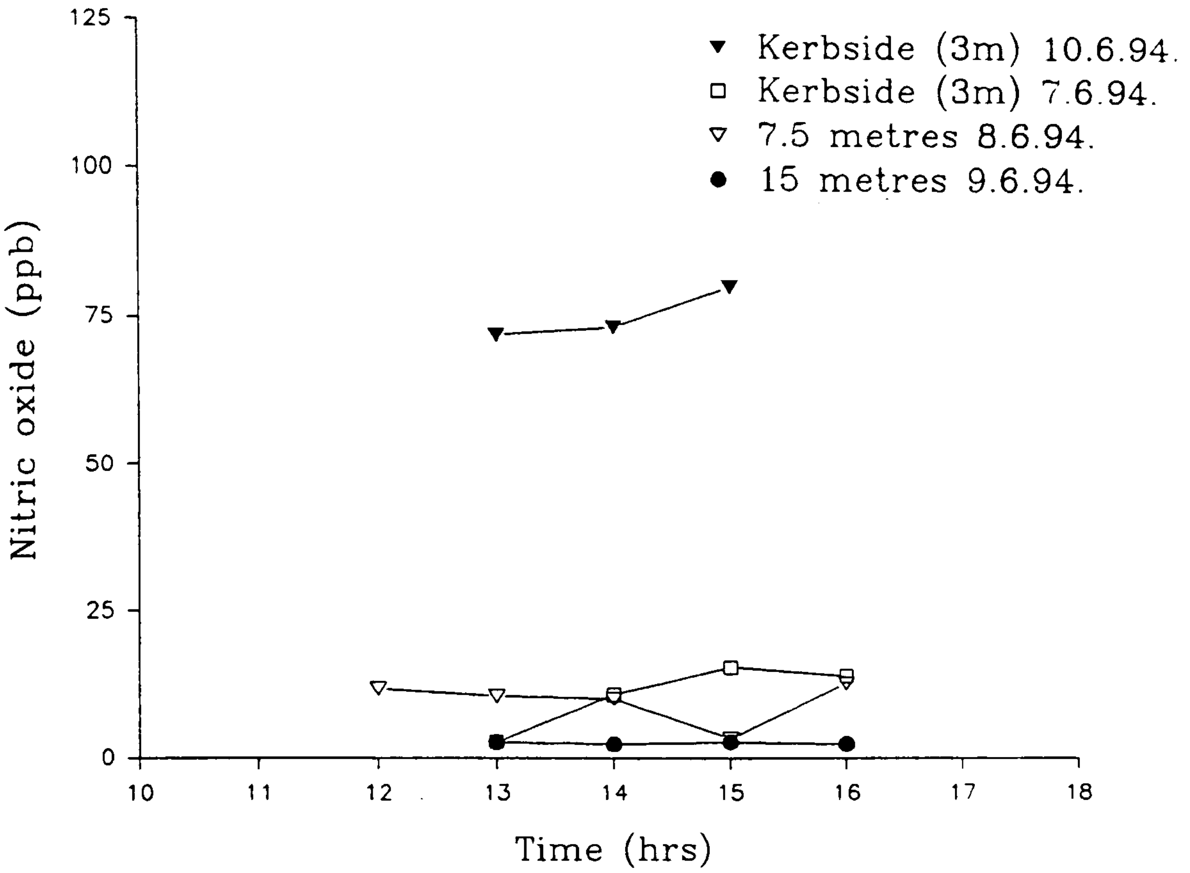


Figure 5.31 Hourly nitric oxide concentrations at site B.

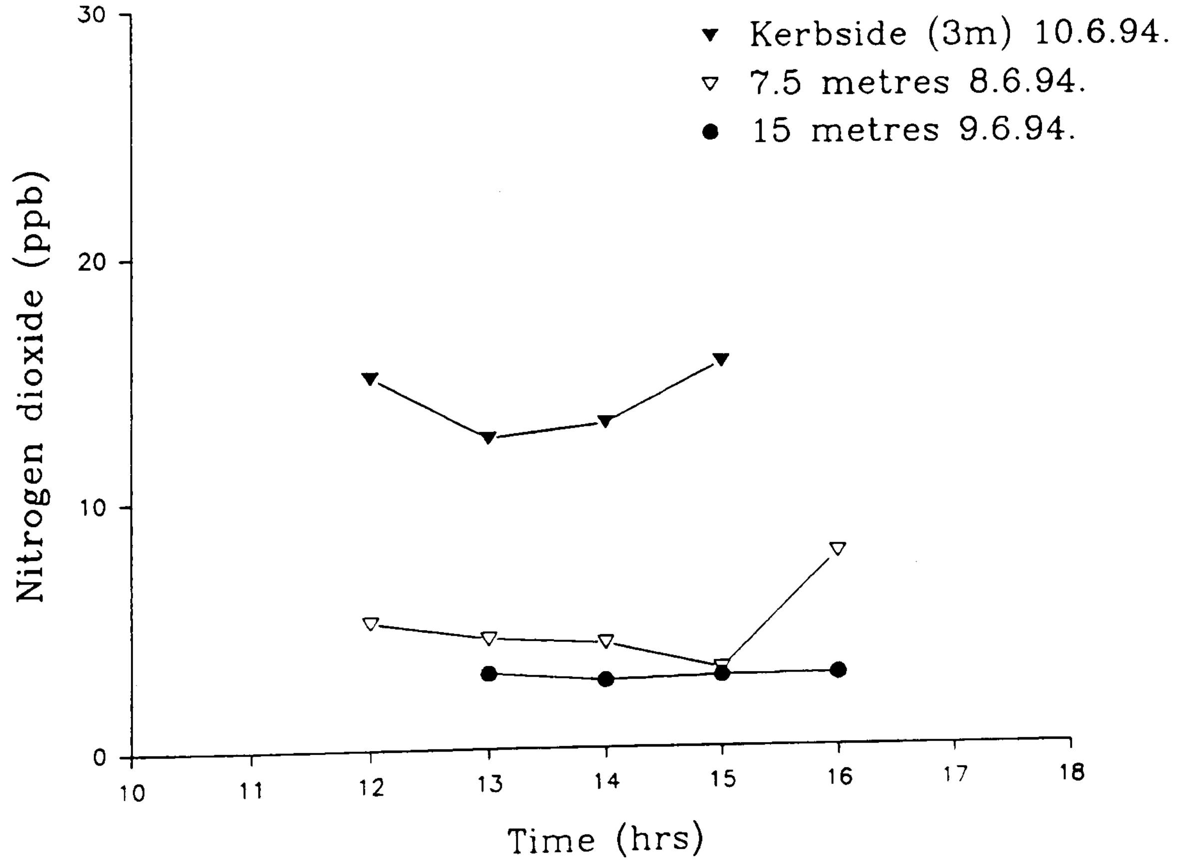


Figure 5.32 Hourly nitrogen dioxide concentrations at site B.

As for site A, the maximum nitrogen dioxide hourly peak value (16 ppb) was significantly less than both the EU 1-hour limit value and the WHO 1-hour guide value. If comparison is made with EU guide values neither the 71 ppb nor the 26 ppb guide values were exceeded at any time. If the NO_2 air quality is compared to the DoE air quality criteria, the air quality would be classified as 'very good' on each day.

5.4.2.5 Comparison of the air quality at site B with Automated Urban Network data

As with carbon monoxide no comparison was possible with AUN data because the AUN site only became operational after monitoring had been completed.

5.4.3 Site C (Abbey Street, Southwark)

5.4.3.1 Time series plot of raw nitric oxide data

Daily mean concentrations follow the pattern of reduction in pollution level with distance from the road displayed for sites A and B, with the highest concentration (86 ppb) recorded at kerbside and the lowest concentration (4 ppb) recorded 15 metres from the road (after correction has been made for parked taxis). The site positioned 7.5 metres from the road produced an intermediate mean value of 53 ppb. The peak values however, do not follow this trend with the highest peak value (526 ppb - *Figure 5.33B)* being recorded 7.5 metres from the road. The highest values recorded at kerbside and 15 metres from the road were 292 ppb *(Figure 5.33 A)* and 80 ppb (after screening the data for taxi emissions) *(Figure 5.33 C)*. The peculiarly high peak measurement recorded 7.5 metres from the road, which disrupts the pattern of peak pollution level with distance from the road, could be caused by particularly high emitting vehicle/s or perhaps an instantaneous change in meteorological conditions, such as a drop in wind speed.

5.4.3.2 Time series plot of raw nitrogen dioxide data

The data for the 9 August 1995 (15 metres site) are erroneously high and are either from an unidentified non traffic source or an equipment/logging error. Therefore, the data have been discounted in terms of roadside air quality.

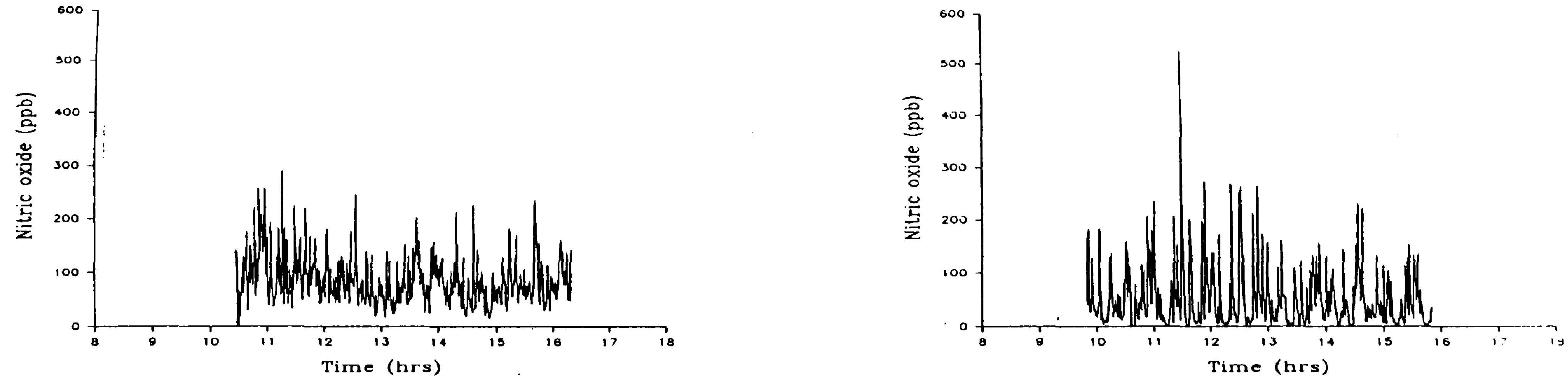
The highest value of 77 ppb was measured at kerbside *(Figure 5.34 A)* with the site positioned 7.5 metres from the road producing a similar maximum value of 74 ppb *(Figure 5.34 B)*. Daily mean values follow a similar trend with a mean concentration of 29 ppb being measured at kerbside and the site positioned 7.5 metres from the road having a slightly lower value of 24 ppb. The similarity between sites is probably due to NO_2 not only being a primary pollutant produced by motor vehicles but also a secondary pollutant formed by the oxidation of NO.

5.4.3.3 Time series plots of five minute mean nitric oxide and nitrogen dioxide concentrations

The pattern of reduction in pollutant levels with increasing distance away from the road noted for daily mean nitric oxide levels is not demonstrated for five minute maximum peak values with the highest five minute mean value of 176 ppb occurring 7.5 metres from the road *(Figure 5.35 B)*. The maximum peak kerbside value was measured at 174 ppb *(Figure 5.35 A)* with the 15 metres location giving a value of 33 ppb - excluding taxi influenced data *(Figure 5.35 C)*.

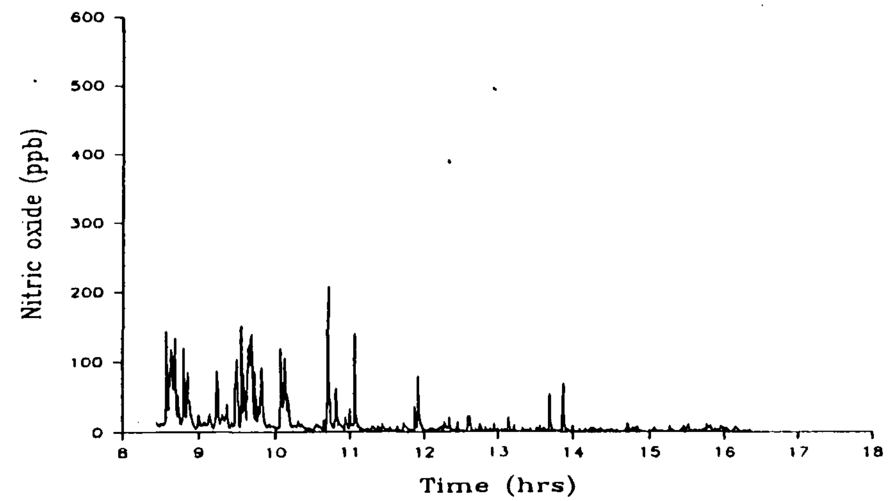
The similarity in concentrations between nitrogen dioxide maximum peak measurements noted for the raw data is again demonstrated for the five minute mean peak values with a maximum peak value of 37 ppb occurring at kerbside *(Figure 5.35 A)* and a maximum peak value of 33 ppb occurring 7.5 metres from the road *(Figure 5.35 B)*. This situation differs from both site A and site B which both showed larger relative differences in concentration between the kerbside site and the site positioned 7.5 metres from the road.

Unlike sites A and B, there were no reliable nitrogen dioxide concentrations recorded 15 metres from the road. Therefore, although it is apparent at sites A and B that there is an increase in nitrogen dioxide concentrations with distance, in comparison to nitric oxide concentrations, culminating in the highest NO_2:NO ratios 15 metres from the road, this is less clear for site C. However, if the ratios of the two pollutants at kerbside



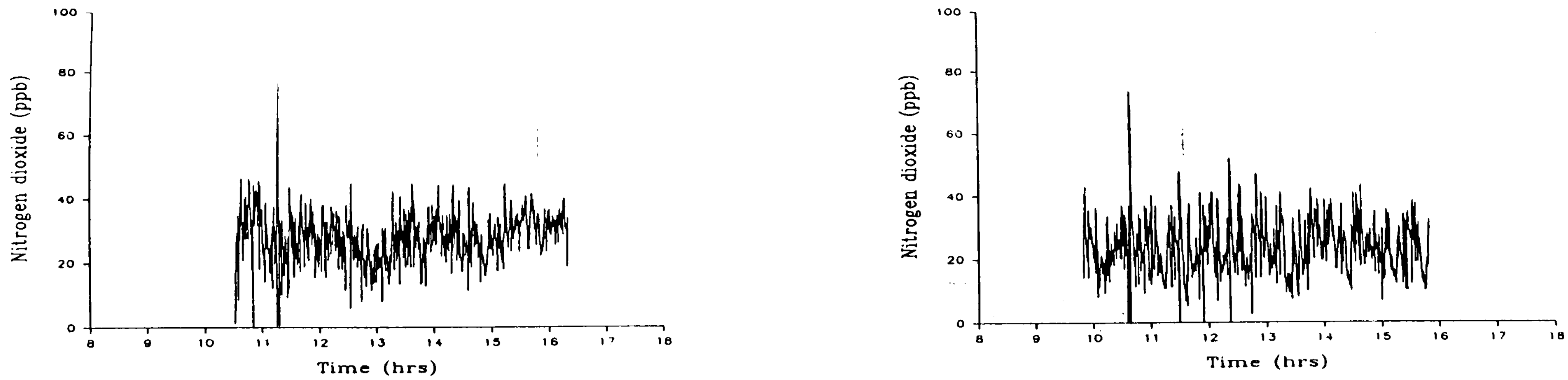
A Kerbside (8 March 1995)

B 7.5 metres (8 August 1995)



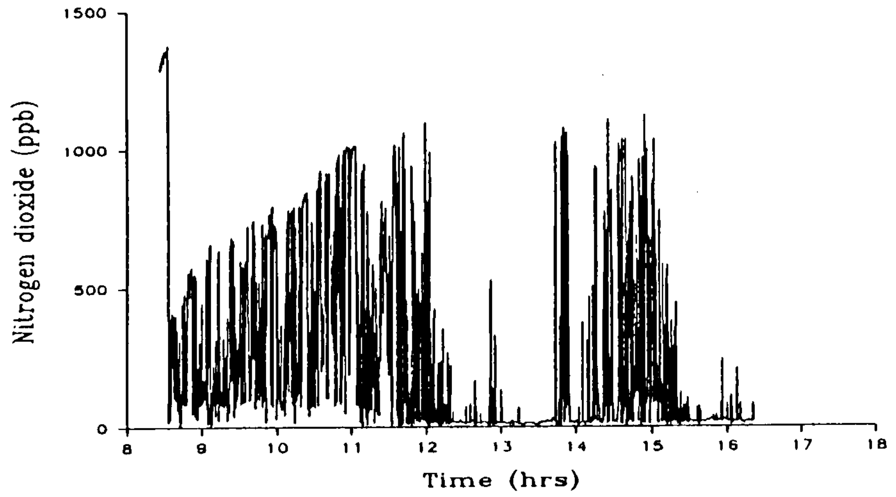
C 15 metres (9 August 1995)

Figure 5.33 (A-C) Time series plots of raw NO data at kerbside, 7.5 metres and 15 metres from the road at site C.



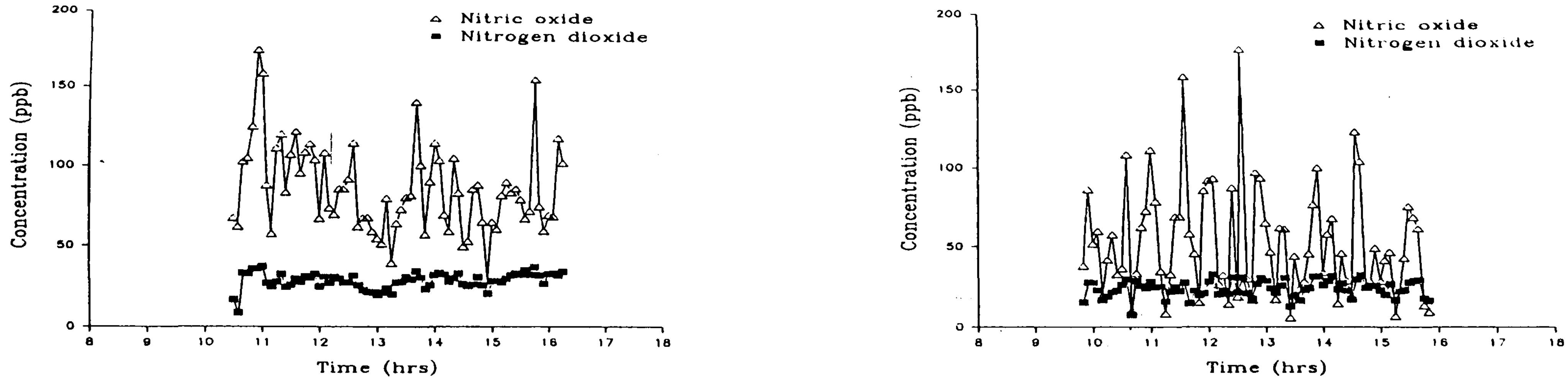
A Kerbside (8 March 1995)

B 7.5 metres (8 August 1995)



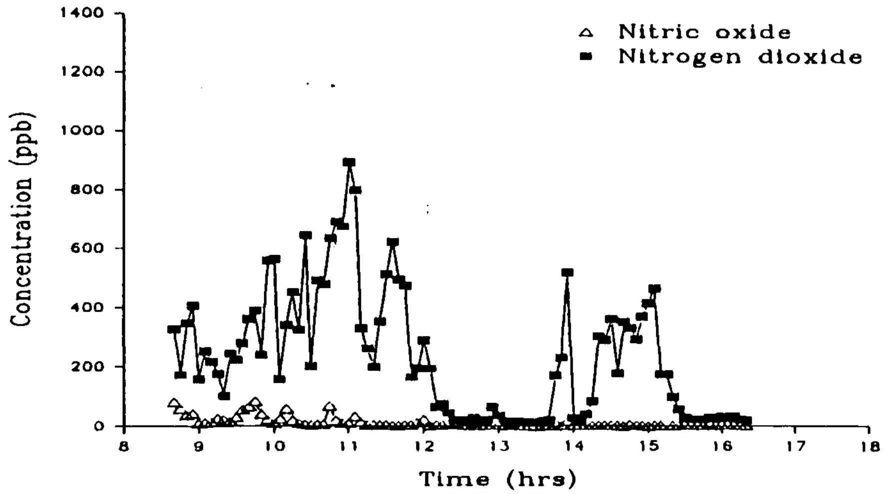
C 15 metres (9 August 1995)

Figure 5.34 **(A-C)** Time series plots of raw NO_2 data at kerbside, 7.5 metres and 15 metres from the road at site C.



A Kerbside (8 March 1995)

B 7.5 metres (8 August 1995)



C 15 metres (9 August 1995)

Figure 5.35 (A-C) Time series plots of five minute NO and NO_2 means at kerbside, 7.5 metres and 15 metres from the road at site C.

and 7.5 metres from the road are analysed *(Table 5.3)* a small rise in nitrogen dioxide levels relative to nitric oxide levels is noticeable.

Table 5.5 Change in daily NO/NO_2 ratios with distance from the road at site C

Distance (metres)	NO/NO_2 Ratio
Kerbside (3 m)	3:1
7.5	2.2:1
15	-

5.4.3.4 Time series plot of hourly mean nitric oxide and nitrogen dioxide concentrations

The plotted data for NO *(Figure 5.36)* demonstrates a strong relationship between reduction in pollutant concentration and increasing distance away from the road. This is different to site B where the relationship was weaker but similar to site A which also displayed a strong relationship between distance and concentration. The plotted data for NO_2 *(Figure 5.37)* display the similarity between the two NO_2 measurement days although due to the lack of data from 15 metres from the road no relationship between pollutant concentration and distance can be made.

As with both sites A and B, the maximum NO_2 hourly peak value (31 ppb) was significantly less than both the EU 1-hour limit value and the WHO 1-hour guide value. If comparison is made with EU guide values, the 71 ppb figure was not exceeded although the 26 ppb guide value was exceeded on all monitoring days. If the nitrogen dioxide air quality is compared to the Department of the Environment banded air quality criteria, the air quality would be classified as 'very good' on each day.

5.4.3.5 Comparison of the air quality at site C with AUN data

Those sites (West London, Cromwell Road, London Bloomsbury, London Bexley) within the AUN which were comparable to sampling site A are also useful when

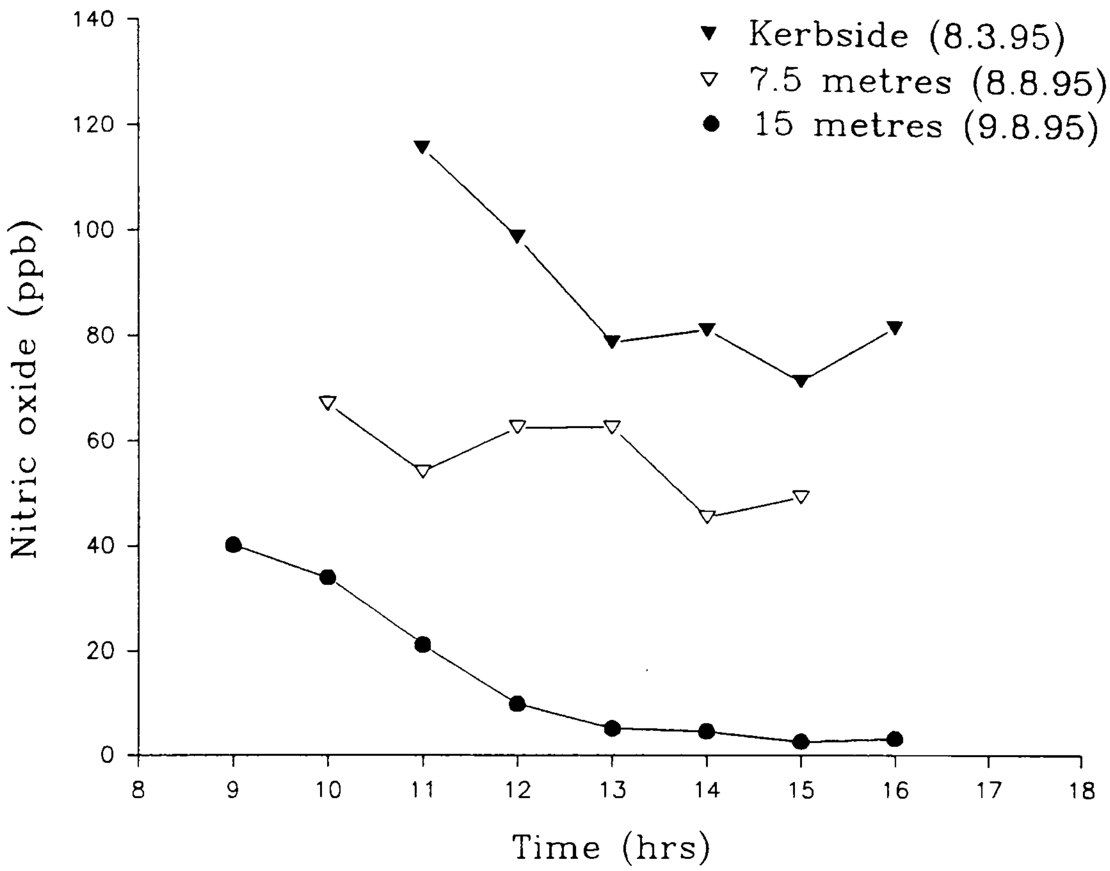


Figure 5.36 Hourly nitric oxide concentrations at site C.

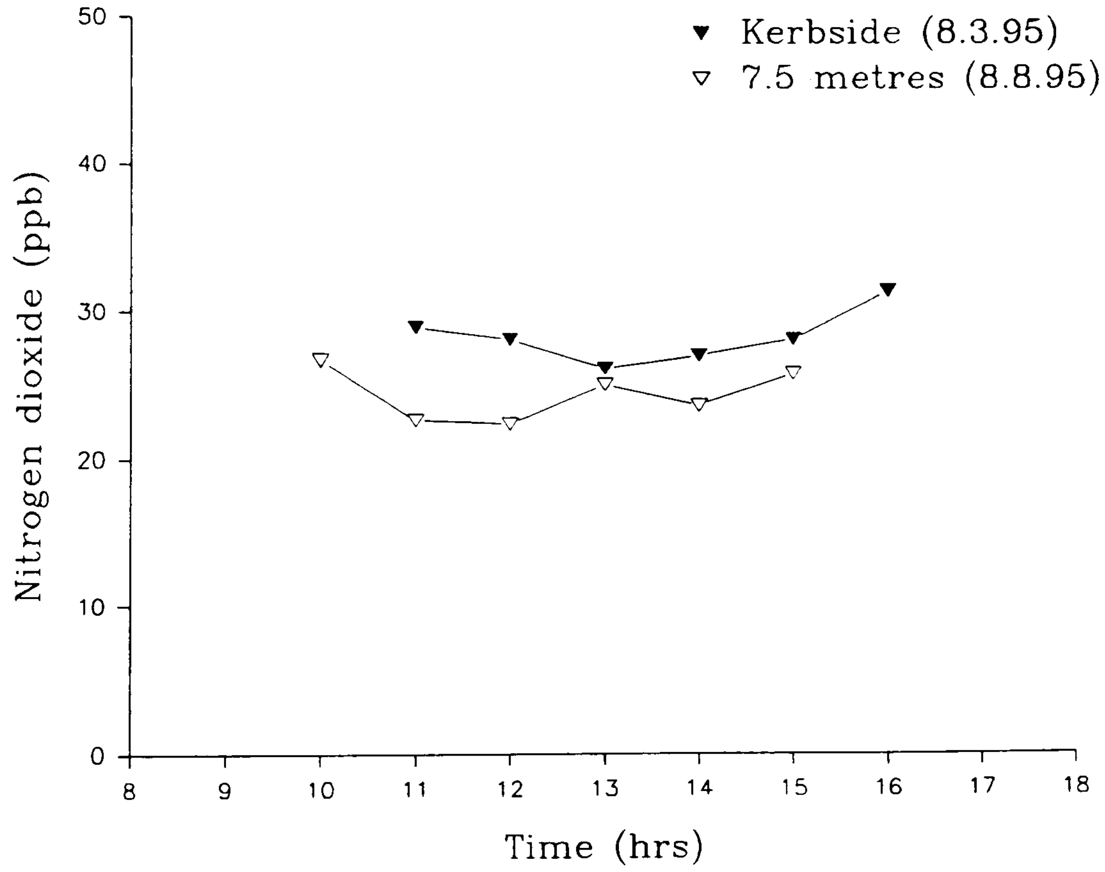


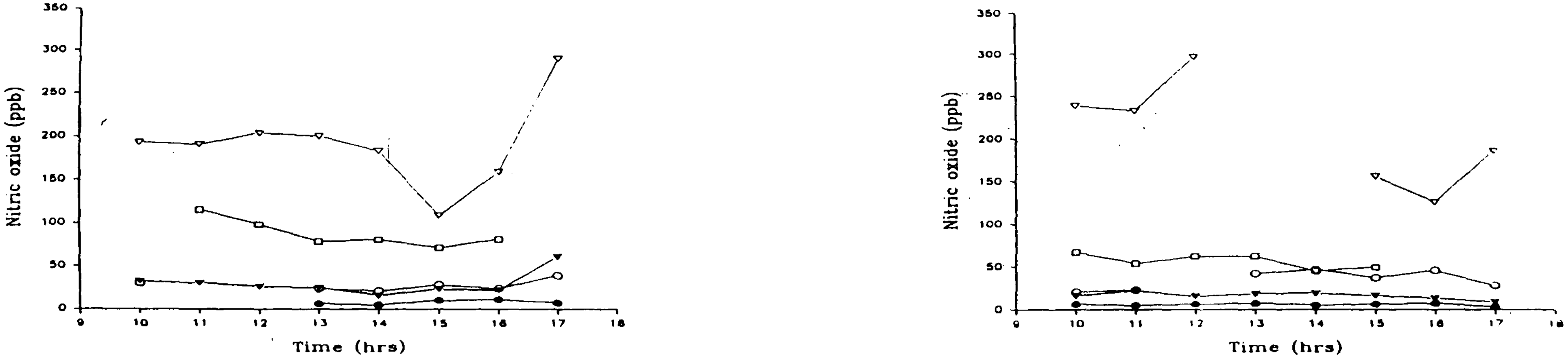
Figure 5.37 Hourly nitrogen dioxide concentrations at site C.

comparison is made with data from site C.

A comparison of site C nitric oxide air quality data with that collected at the comparable AUN sites is shown in *Figure 5.38 A-C.* The kerbside location at site C *(Figure 5.38 A)* exhibits lower nitric oxide concentrations than Cromwell Road but higher concentrations than those found at West London, London Bloomsbury and London Bexley. This pattern reflects that found for carbon monoxide with the sites further from the road and, therefore the emission source, having better air quality than site C which is, in turn, less polluted than Cromwell Road where generally higher traffic flows are found.

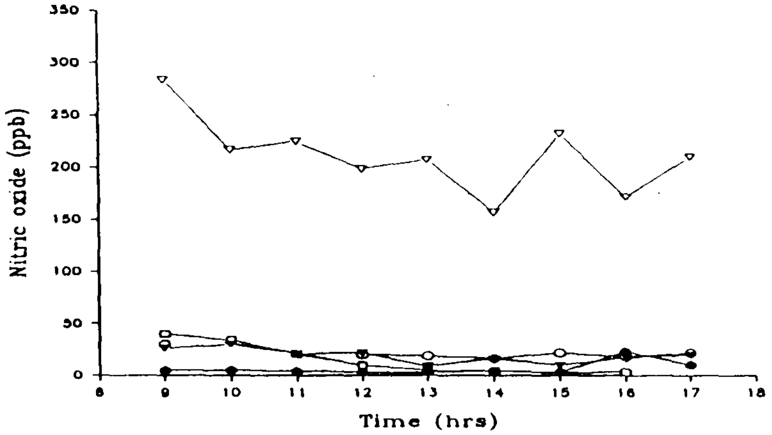
Comparison of the data gained from the site positioned 7.5 metres from the road with the AUN data is shown in *Figure 5.38 B.* As expected, Cromwell Road is shown to have higher nitric oxide levels than those found at site C, and site C to have higher nitric oxide levels than those found at the urban background sites although the difference with London Bloomsbury is minimal. The data obtained from the site positioned 15 metres from the road is compared with the AUN data in *Figure 5.38 C.* The nitric oxide levels at site C are shown to have decreased to an urban background concentration similar to that found at the West London, London Bloomsbury and London Bexley sites. The Cromwell Road site has significantly higher nitric oxide levels than all other locations.

Nitrogen dioxide air quality data from site C is assessed against comparable AUN sites in *Figure 5.39 A-C.* Comparison of the nitrogen dioxide data from kerbside *(Figure 5.39 A)* shows a markedly different situation to that found for nitric oxide and carbon monoxide. However, the situation is similar to the nitrogen dioxide comparison with AUN data at site A in that the urban background sites have much higher relative nitrogen dioxide levels which is because of the increased oxidation of nitric oxide to nitrogen dioxide at greater distances from the road. However, site C differs from site A (section 5.4.1.5) in that it does not have predominantly higher nitrogen dioxide levels at kerbside but comparable concentrations to those found at West London and London Bexley and lower concentrations than those found at Cromwell Road and London Bloomsbury.



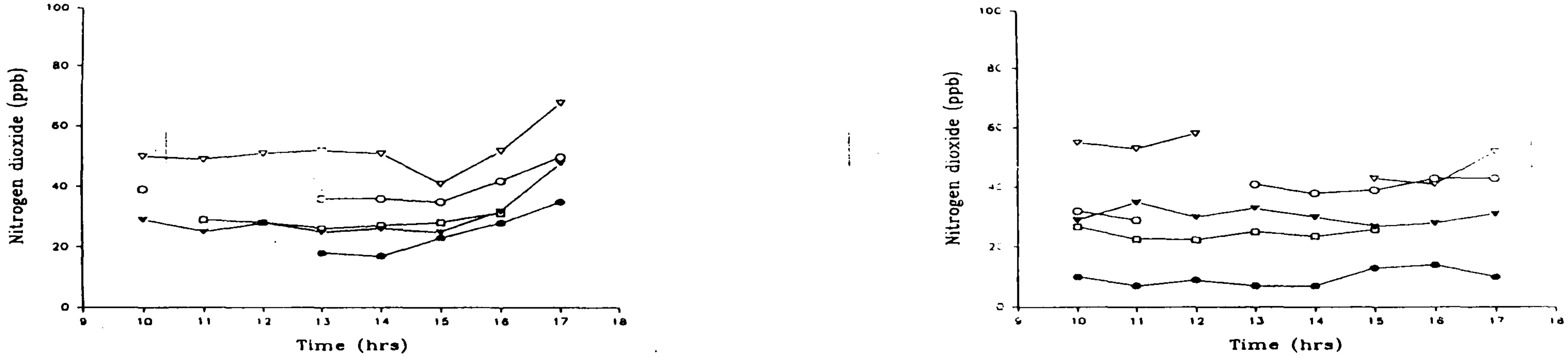
A Kerbside (8 March 1995)

B 7.5 metres (8 August 1995)



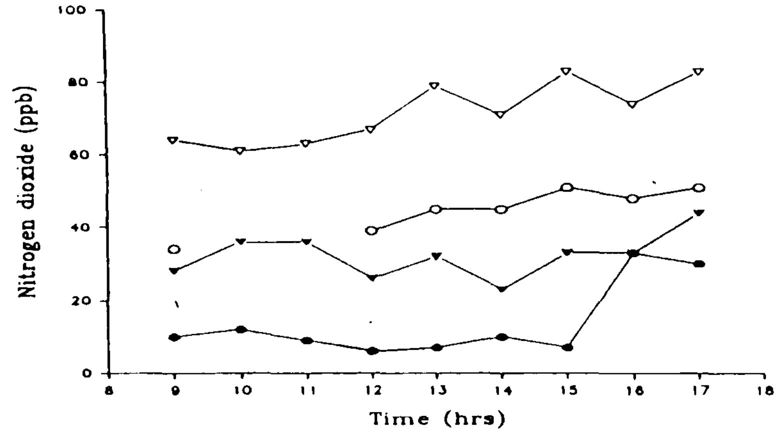
C 15 metres (9 August 1995)

Figure 5.38 **(A-C)** Comparison of one hour mean NO levels at Site C (□), at kerbside, 7.5 metres and 15 metres from the road, to those found at AUN sites WL (▾) BEX (●) CRD (▿) and CLL2 (○).



A Kerbside (8 March 1995)

B 7.5 metres (8 August 1995)



C 15 metres (9 August 1995)

Figure 5.39 (A-C) Comparison of one hour mean NO_2 levels at Site C (□), at kerbside, 7.5 metres and 15 metres from the road, to those found at AUN sites WL (▾) BEX (●) CRD (▿) and CLL2 (○).

Comparison of the data gained from the site positioned 7.5 metres from the road with the AUN data is shown in *Figure 5.39 B*, and indicates a similar scenario to that seen at kerbside with the only principal variation being the reduction in concentration at London Bexley. No comparison of the data gained from the site positioned 15 metres from the road was possible due to the unreliabilty of the data.

5.4.4 Site D (Uppingham Road, Leicester)

5.4.4.1 Time series plot of raw nitric oxide data

The pattern of reduction in peak and mean NO concentrations with distance from the road exhibited to varying degrees at sites A, B and C is absent from site D. However, the NO data are similar in form to the CO data for the same site.

The highest daily mean NO value (99 ppb) was recorded 7.5 metres from the road as was the highest mean CO value. This was previously discussed (section 5.3.4.1) in terms of the 20 October 1995 (7.5 metres) being the only sampling day when the prevailing wind direction was from the road towards the air quality monitors and therefore, produced the highest mean levels. The site positioned 15 metres from the road (21 October 1995) produced the next highest daily mean NO level (60 ppb) which is similar to the situation for CO. This was explained for CO by the very low wind speeds encountered on 21 October 1995 in comparison to the other sampling days and is probably responsible for the elevated NO levels. There was a period of calm wind conditions on 21 October 1995 between 08:00 and 10:00 which coincide with the maximum peak NO value (1217 ppb) recorded at site D *(Figure 5.40 D)*.

The sites positioned at roadside and kerbside produced daily mean levels of 33 and 31 ppb, respectively. The similarity between the two sites cannot be explained by either a change in the prevailing wind direction or wind speed as they were similar in direction and magnitude during both sampling days. A possible explanation for the similarity is that because the monitoring at roadside was conducted on a Sunday there were less vehicles in comparison to a weekday and therefore, less emissions and lower ambient levels than expected at roadside.

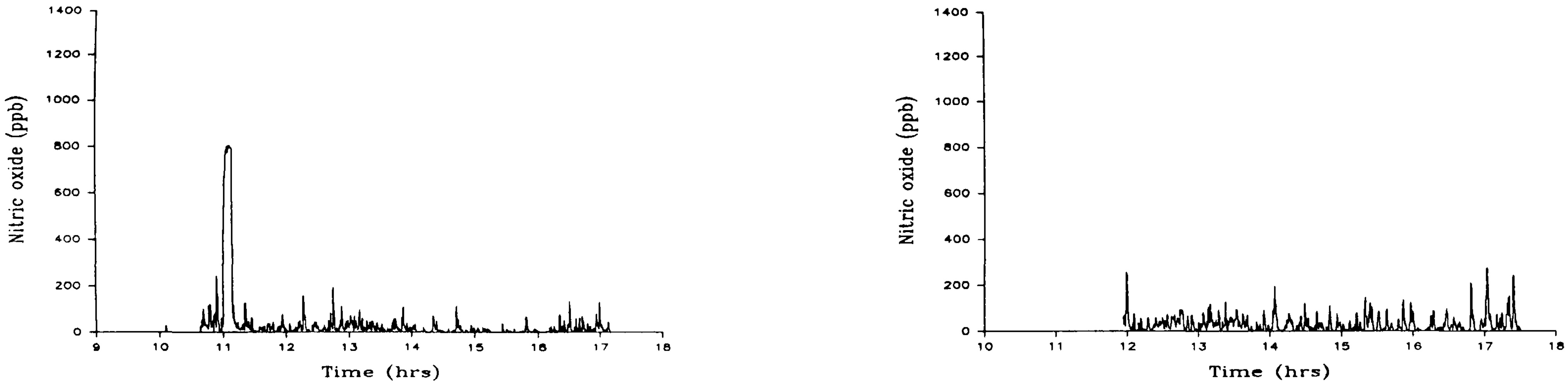
The maximum peak values do not follow the pattern of the daily mean values with, as reported above, the site positioned 15 metres from the road producing the highest peak value *(Figure 5.40 D)*. Measurements carried out at the roadside location produced a lower maximum peak value of 808 ppb *(Figure 5.40 A)* with the kerbside site having the lowest peak value of 278 ppb *(Figure 5.40 B)*. The site positioned 7.5 metres from the road produced a maximum value of 367 ppb *(Figure 5.40 C)*.

5.4.4.2 Time series plot of raw nitrogen dioxide data

The time series traces of the raw data for nitrogen dioxide are similar to nitric oxide in that they are influenced by high peak values. However, they differ in that the NO_2 traces have, in general, a flatter structure.

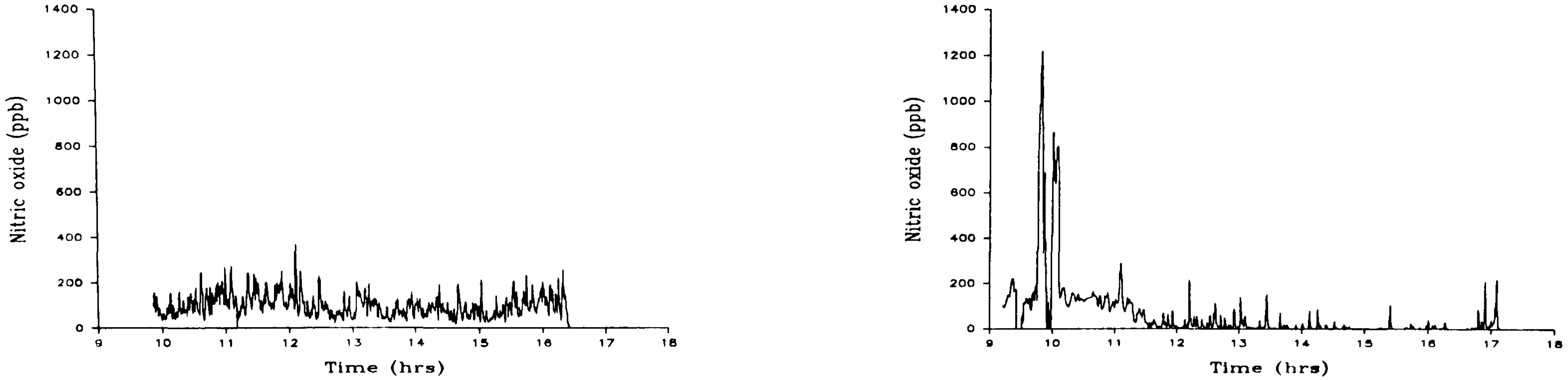
Analysis of daily mean levels shows that, as for NO, the highest average concentration (32 ppb) was measured 7.5 metres from the road. The kerbside site had a mean concentration of 29 ppb and the site positioned 15 metres from the road a mean value of 22 ppb. The roadside site had the lowest concentration at 14 ppb. Analysis of the daily means shows no reduction in pollutant concentration with distance from the road as was observed at sites A and B. Moreover, there is a similarity in daily mean levels at site D such that there is only 18 ppb difference between the highest and lowest daily mean concentrations.

As with nitric oxide, the maximum peak values do not follow the pattern of the daily mean values with the highest value (311 ppb) being recorded 15 metres from the road *(Figure 5.41 D)*. Measurements carried out at the kerbside location produced a somewhat lower maximum peak value of 166 ppb *(Figure 5.41 B)*. The lowest NO concentrations were recorded 7.5 metres from the road with the highest value being 76 ppb *(Figure 5.41 C)*. The site positioned at roadside produced a maximum value of 121 ppb *(Figure 5.41 A)*.



A Roadside (22 October 1995)

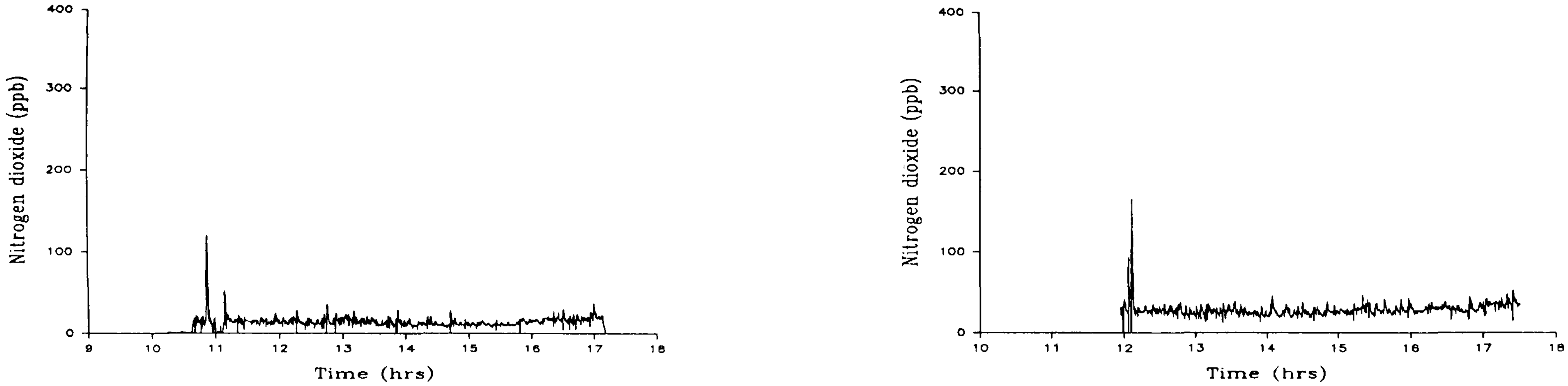
B Kerbside (19 October 1995)



C 7.5 metres (20 October 1995)

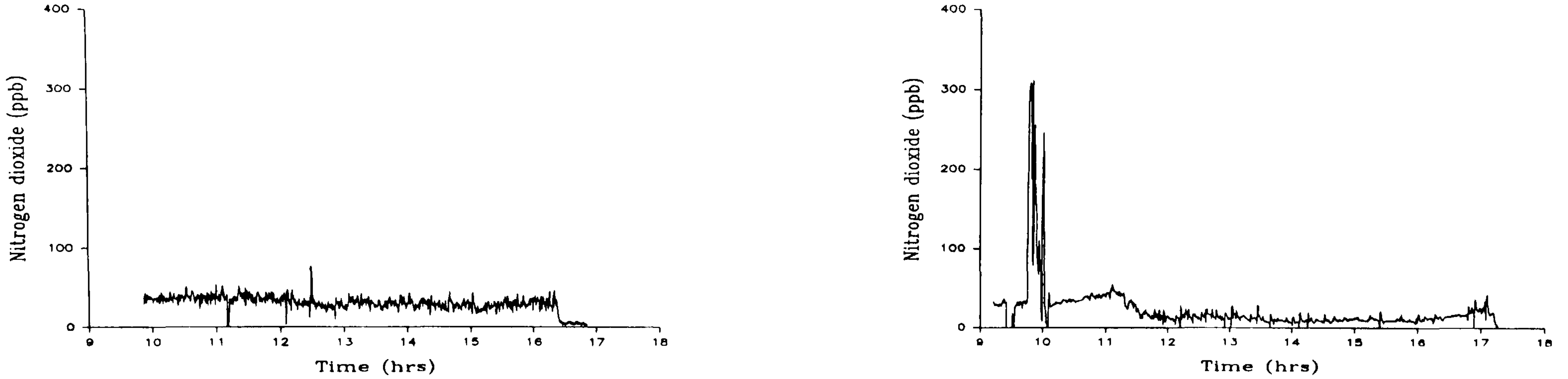
D 15 metres (21 October 1995)

Figure 5.40 (A-D) Time series plots of raw NO data at roadside, kerbside, 7.5 metres and 15 metres from the road at site D.



A Roadside (22 October 1995)

B Kerbside (19 October 1995)



C 7.5 metres (20 October 1995)

D 15 metres (21 October 1995)

Figure 5.41 (A-D) Time series plots of raw NO_2 data at roadside, kerbside, 7.5 metres and 15 metres from the road at site D.

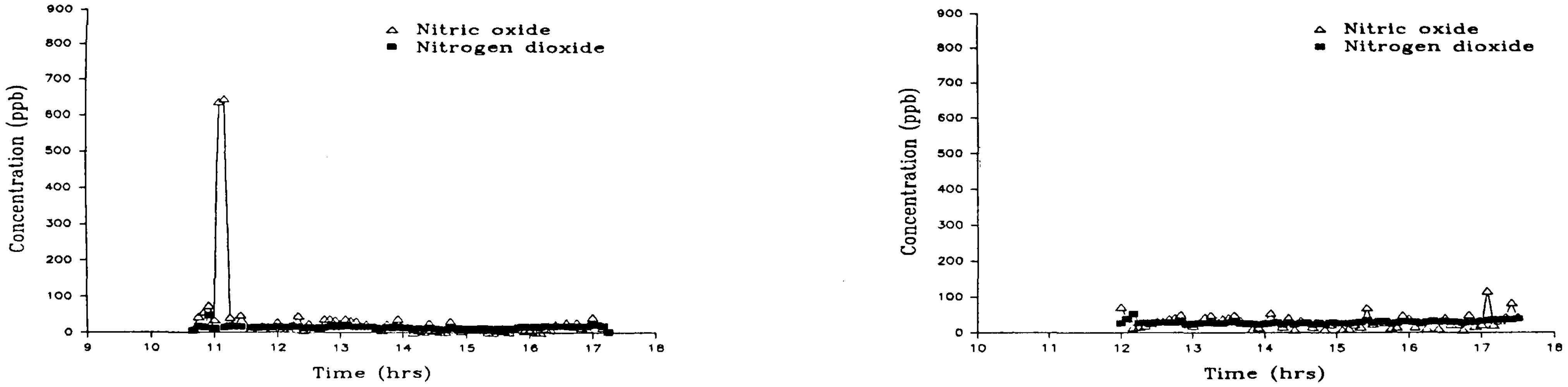
5.4.4.3 Time series plots of five minute mean nitric oxide and nitrogen dioxide concentrations

The pattern of reduction in nitric oxide and nitrogen dioxide five minute average levels with increasing distance away from the road, visible at the other sites, is not demonstrated at site D. The highest nitric oxide concentration was recorded 15 metres from the road with a value of 799 ppb *(Figure 5.42 D)*. The roadside site produced the next highest peak with a value of 647 ppb *(Figure 5.42 A)*. The kerbside site and the site positioned 7.5 metres from the road had similar peak values of 116 ppb and 177 ppb respectively *(Figures 5.42 B-C)*. The highest nitrogen dioxide concentration was recorded 15 metres from the road with a value of 226 ppb *(Figure 5.42 D)*. The kerbside site produced the next highest peak with a value of 52 ppb *(Figure 5.42 B)*. The roadside site *(Figure 5.42 A)* and the site positioned 7.5 metres from the road *(Figure 5.42 C)* had similar peak values of 47 ppb and 42 ppb respectively.

It is apparent from *Figure 5.42 A-D* that, unlike sites A, B and C, there is no relative rise in nitrogen dioxide concentration, in comparison to nitric oxide, with distance. An analysis of the ratios of the two pollutants *(Table 5.6)* shows no consistent pattern with the highest concentration of nitrogen dioxide relative to nitric oxide occurring at kerbside and the lowest 7.5 metres from the road.

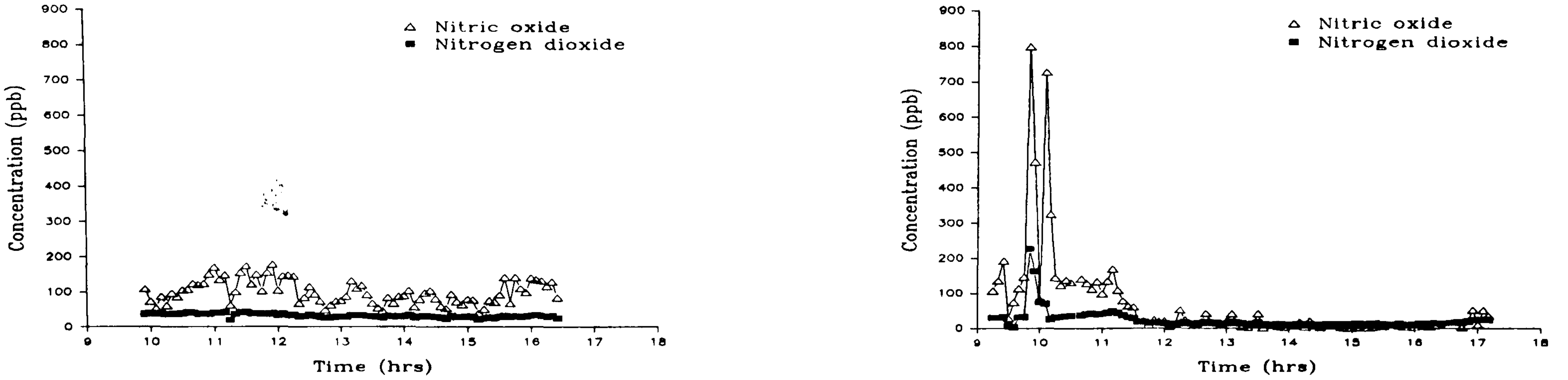
Table 5.6 Change in daily NO/NO_2 ratios with distance from the road at site D

Distance (metres)	NO/NO_2 Ratio
Roadside	2.4:1
Kerbside (3m)	1.1:1
7.5	3.1:1
15	2.7:1



A Roadside (22 October 1995)

B Kerbside (19 October 1995)



C 7.5 metres (20 October 1995)

D 15 metres (21 October 1995)

Figure 5.42 (A-D) Time series plots of five minute NO and NO_2 means at roadside, kerbside, 7.5 and 15 metres from the road at site D.

5.4.4.4 Time series plot of hourly mean nitric oxide and nitrogen dioxide concentrations

The plotted data *(Figures 5.43 & 5.44)* again do not demonstrate the reduction in pollution levels with distance which was visible at the other sites and show random fluctuations in pollutant concentration with distance away from the road.

As for sites A, B and C, the maximum hourly nitrogen dioxide peak value (57 ppb) was significantly less than both the EU 1-hour limit value and the WHO 1-hour guide value. If comparison is made with EU guide values, the 71 ppb figure was not exceeded although the 26 ppb guide value was exceeded at all monitoring locations except roadside. If the nitrogen dioxide air quality is compared to the Department of the Environment banded air quality criteria, the air quality would be classified as 'very good' with the exception of the 21 October 1995 when the air quality would be classified as 'good'.

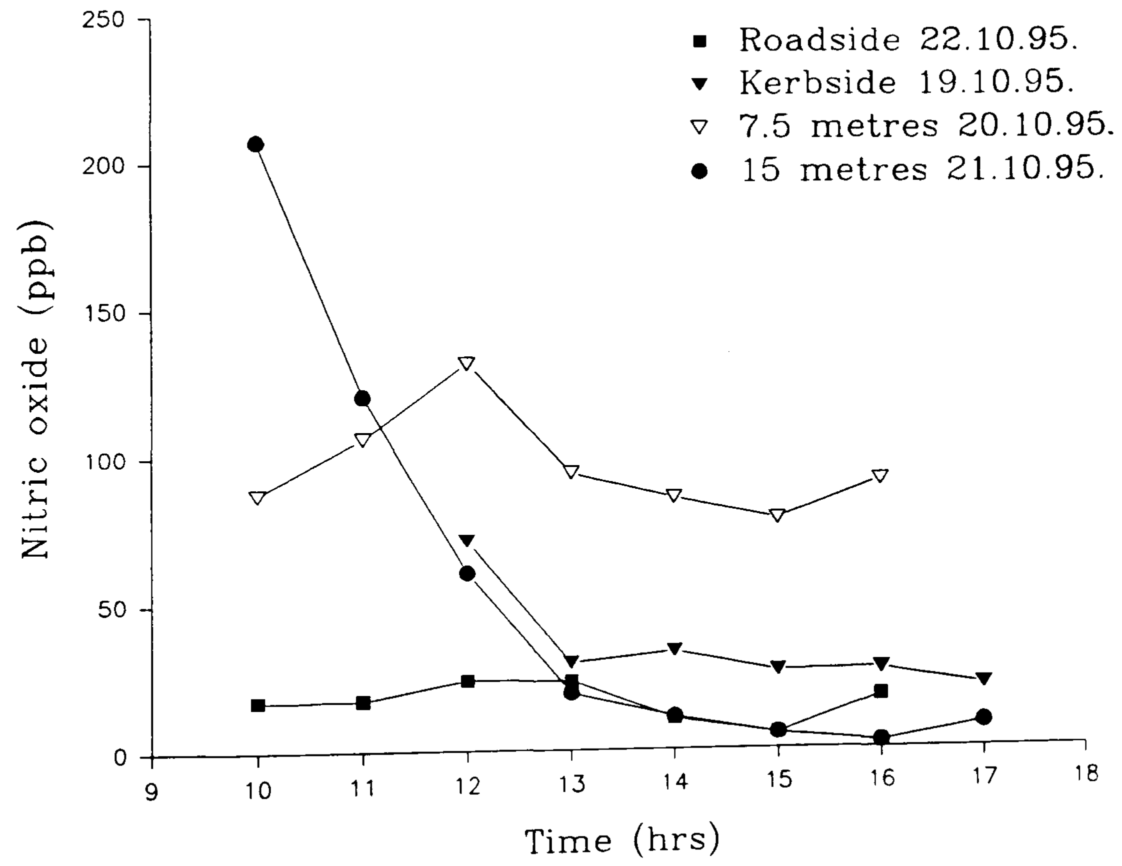


Figure 5.43 Hourly nitric oxide concentrations at site D.

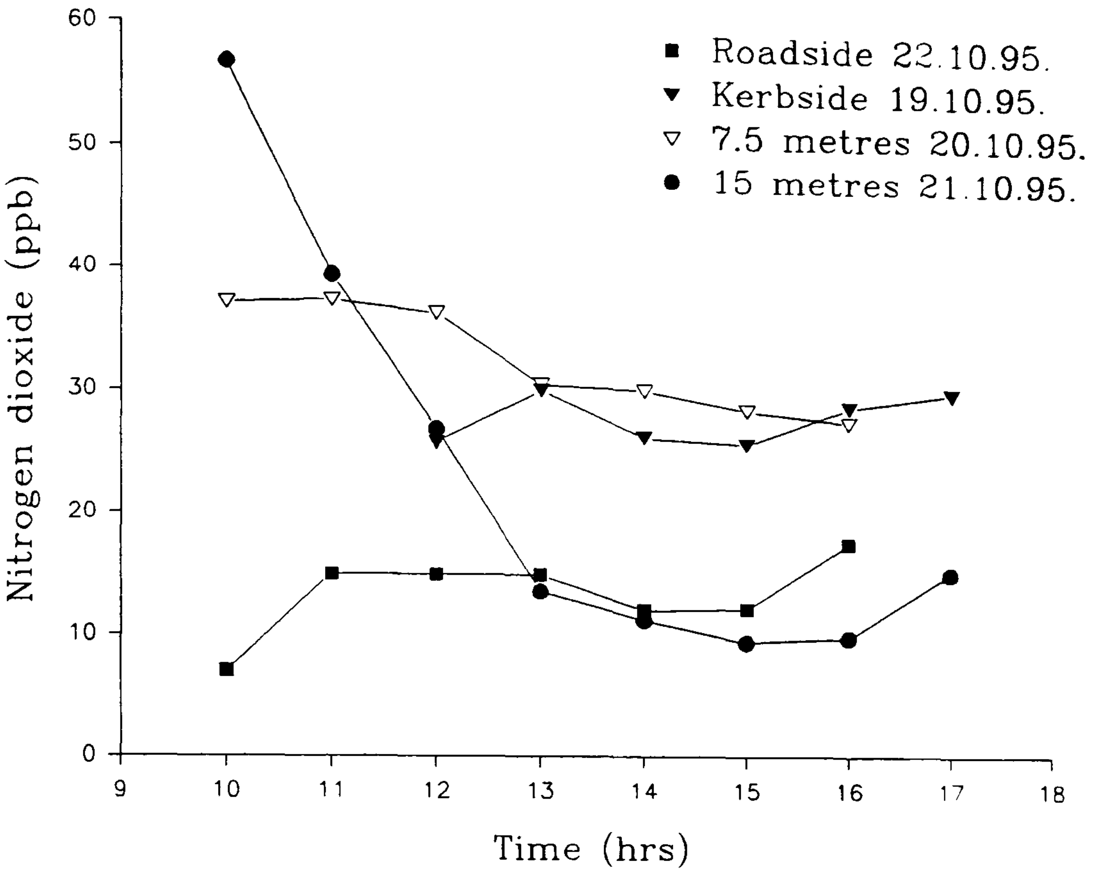


Figure 5.44 Hourly nitrogen dioxide concentrations at site D.

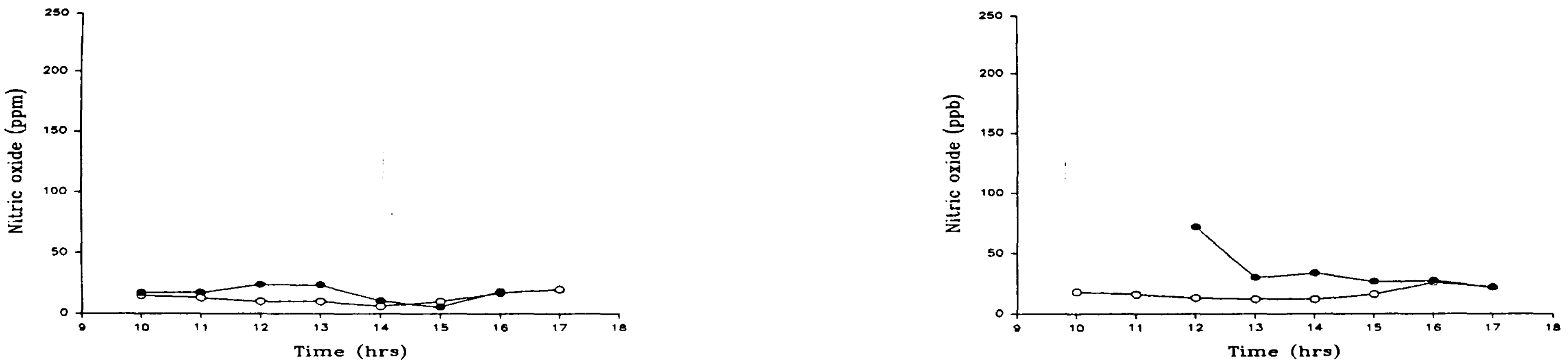
5.4.4.5 Comparison of the air quality at site D with Automated Urban Network data

A comparison of site D nitric oxide air quality data with that obtained from the Leicester centre AUN site is shown in *Figure 5.45 A-D*. Comparison of the roadside data to the AUN data *(Figure 5.45 A)* produced surprising results with very similar concentrations, at consistently low levels. One possible reason for the anomalously low concentrations at roadside is the very low traffic volumes encountered on Sunday 22 October and hence, the similarity with the AUN urban background station.

Comparison of the kerbside data with the AUN data is shown in *Figure 5.45 B* and is similar to that obtained for carbon monoxide with site D having only marginally higher ambient levels compared to the AUN site. This is because of the low ambient levels at kerbside, explained in section 5.4.4.1, and not by high levels at the AUN site. Site D has significantly higher NO concentrations at 7.5 metres from the road when compared to the AUN site *(Figure 5.45 C)* which in part reflects its proximity to the road but also the prevailing wind direction during measurement at this location. Comparison of the data obtained 15 metres from the road with AUN data *(Figure 5.45 D)* may be divided into two distinct parts. The first part occurred prior to 13:00 when a period of calm wind conditions at site D produced elevated concentrations thereby producing worse air quality in the morning than at the AUN site. After 13:00 wind speeds increased at site D such that concentrations were fractionally lower than at the AUN site, showing that dependent on wind conditions, nitric oxide concentrations can attain an urban background level 15 metres from the road.

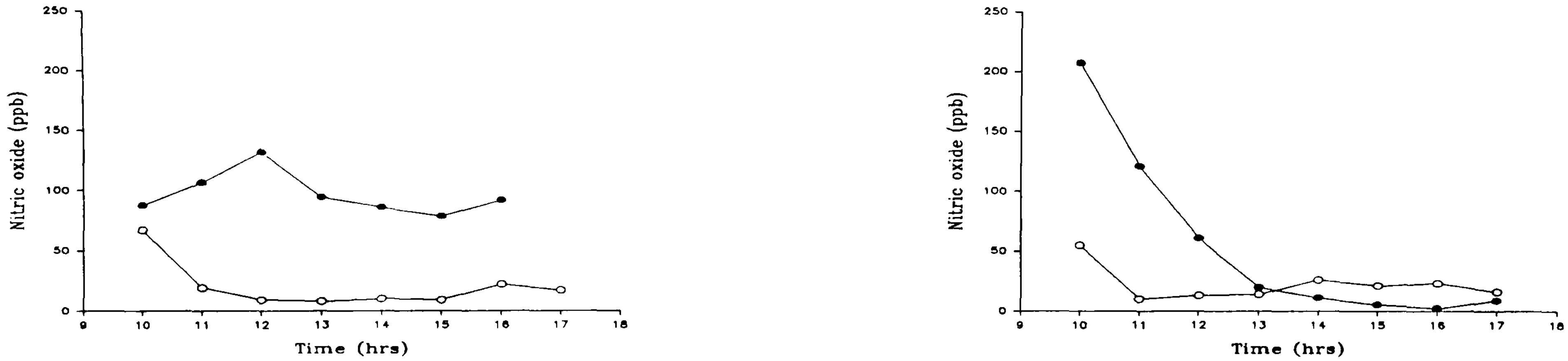
The nitrogen dioxide air quality data is compared with that obained at the comparable AUN site in *Figure 5.46 A-D*. Roadside levels at site D *(Figure 5.46 A)* have lower nitrogen dioxide concentrations than the AUN site possibly due to more oxidation of NO to NO_2 having occurred at the more distant AUN site relative to a road. The kerbside data *(Figure 5.46 B)* shows a different scenario with nitrogen dioxide levels at site D being very similar in magnitude to those found at the AUN site. This may be a consequence of more oxidation of NO to NO_2 at kerbside compared to roadside or because the primary emission of NO_2 was greater on the kerbside sampling day due to greater vehicle numbers.

A comparison of the data gained from the site positioned 7.5 metres from the road with the AUN data is shown in *Figure 5.46 C* and indicates predominantly higher nitrogen dioxide levels than those found at the AUN site reversing the trend found at roadside and kerbside. However, the result is similar to that seen for CO and NO and reflects the prevailing wind direction during sampling 7.5 metres from the road. The comparison of the data gained from the site positioned 15 metres from the road with the AUN data is shown in *Figure 5.46 D*. As with NO the comparison can be divided into two parts. Prior to 13:00, dispersion of the emissions was restricted by the calm conditions at site D and therefore ambient levels increased leading to higher NO_2 concentrations than the AUN site. After 13:00, increased wind speeds assisted dispersion of the pollutants at site D and resulted in a decrease in NO_2 concentrations below those recorded at the AUN site.



A Roadside (22 October 1995)

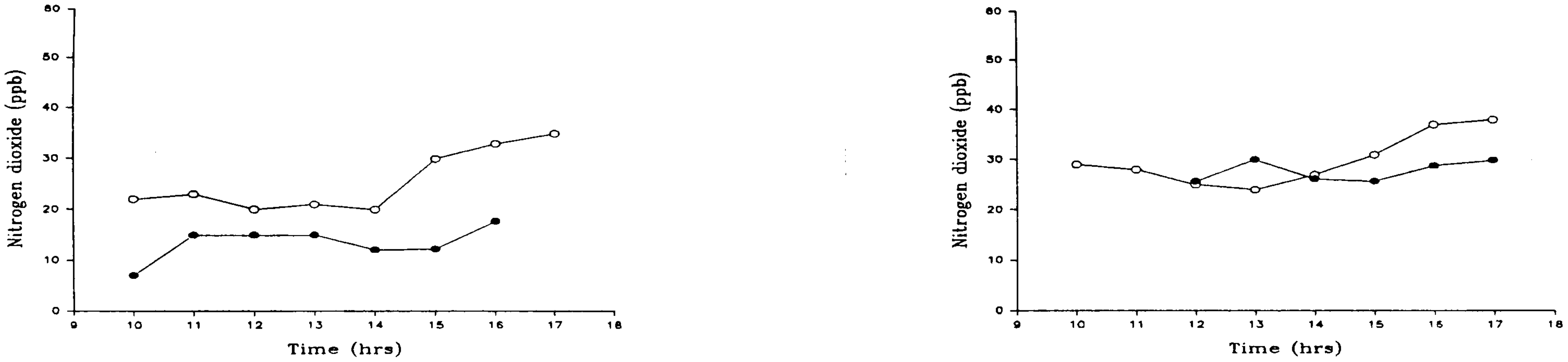
B Kerbside (19 October 1995)



C 7.5 metres (20 October 1995)

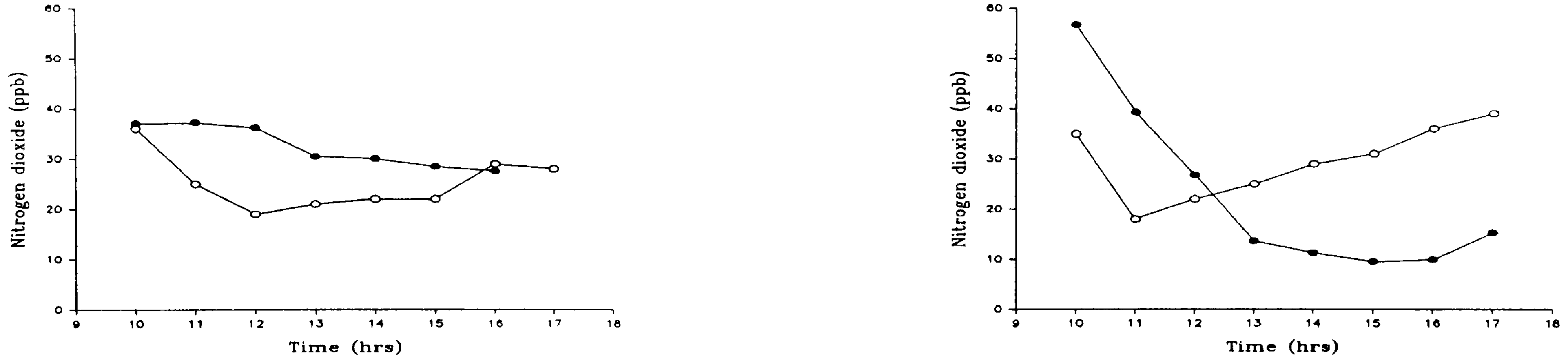
D 15 metres (21 October 1995)

Figure 5.45 (A-D) Comparison of one hour mean NO levels at Site D (●), at roadside, kerbside, 7.5 metres and 15 metres from the road, to those found at the Leicester AUN (O) site.



A Roadside (22 October 1995)

B Kerbside (19 October 1995)



C 7.5 metres (20 October 1995)

D 15 metres (21 October 1995)

Figure 5.46 (A-D) Comparison of one hour mean NO_2 levels at Site D (●), at roadside, kerbside, 7.5 metres and 15 metres from the road, to those found at the Leicester AUN (O) site.

5.4.5 Inter-site comparison of nitric oxide air quality

Inter-site comparisons of daily mean nitric oxide levels are displayed in **Table 5.7**. The similarity between sites A and C, evident for carbon monoxide air quality, is less obvious for nitric oxide air quality. The kerbside nitric oxide levels at sites A and C were considerably different with the concentration at site A being elevated by a factor of 2.4 compared to that at site C. However, the nitric oxide levels at the sites positioned 7.5 metres and 15 metres from the road were similar. Site B exhibited the lowest carbon monoxide concentrations and also the lowest nitric oxide concentrations. However, the difference in nitric oxide levels in comparison to other sites is not as pronounced as the difference was for carbon monoxide. The nitric oxide levels at the kerbside location of site B were significantly higher than the levels found 15 metres from the road at sites A and C, which was not the case for carbon monoxide. Sites A, B and C all show a reduction in concentration with increasing distance from the highway which is not apparent at site D. The highest concentration at site D was found 7.5 metres from the road and the lowest concentration at kerbside. An intermediate value was recorded 15 metres from the road.

Table 5.7 Inter-site comparison of daily mean nitric oxide levels with distance from the road

Distance from road (metres)	Site A (ppb)	Site B (ppb)	Site C (ppb)	Site D (ppb)
3 (Kerbside)	207	45	86	31
7.5	55	10	53	99
15	15	2	4	60

5.4.6 Inter-site comparison of nitrogen dioxide air quality

Inter-site comparisons of daily mean nitrogen dioxide levels are displayed in **Table 5.8.** The similarity between sites A and C, evident for both carbon monoxide air quality and to a lesser extent nitric oxide air quality, is not observable for nitrogen dioxide. Site B had the lowest carbon monoxide and nitric oxide concentrations and also has the lowest nitrogen dioxide concentrations.

Table 5.8 Inter-site comparison of daily mean nitrogen dioxide levels with distance from the road

Distance from road (metres)	Site A (ppb)	Site B (ppb)	Site C (ppb)	Site D (ppb)
3 (Kerbside)	64	15	29	29
7.5	33	5	24	32
15	11	3	-	22

Sites A and B show a reduction in concentration with increasing distance from the highway as does site C in a limited fashion but this is again not apparent for site D. The highest concentration at site D was found 7.5 metres from the road, with the lowest concentration recorded 15 metres from the road, and an intermediate value being recorded at kerbside.

5.5 Summary

A decrease of carbon monoxide and nitric oxide concentrations with distance from the road was noted for all sites with the exception of site D. The reduction in carbon monoxide concentrations was most marked for sites A and C, with similar size decreases in ambient concentrations noted at both locations. Site B had only a slight decrease in carbon monoxide concentrations with distance; no carbon monoxide data was available from the site positioned 15 metres from the road. Site D did not display any reduction in ambient levels of carbon monoxide or nitric oxide as meteorological parameters exerted greater influence upon air quality than did distance from the road.

A decrease of nitrogen dioxide concentrations with distance from the road was noted for sites A, B and C although it was not expected. The reduction was most distinct for site A followed by site B. The reduction in concentration, however, is generally much less than was noted for CO and NO. Site C is the best example of this with little difference in concentration between the kerbside measurement site and the site positioned 7.5 metres from the road; no data was available for the site positioned 15 metres from the road. Site D did not display any pattern in ambient levels with distance from the road.

The initial expectation was for nitrogen dioxide concentrations to increase at greater distances from the road as nitric oxide was oxidised to nitrogen dioxide. However, it appears that, over the distances from the road monitored in these field studies, the production of nitrogen dioxide as a primary pollutant by motor vehicles was more important than from the oxidation of nitric oxide. Therefore, any production of nitrogen dioxide due to oxidation was being masked by the large amounts produced by vehicle emissions. However, the production of nitrogen dioxide by oxidation can be inferred in two ways. The first is the much more gradual decline in NO_2 concentration away from the road when compared to CO and NO which is probably caused by the production of nitrogen dioxide by oxidation counteracting the reduction in concentration caused by dispersion. The second is an analysis of the change of ratio between nitrogen dioxide and nitric oxide with distance from the road which reveals an increase of NO_2 relative to NO with distance, presumably caused by the oxidation of NO to NO_2.

Carbon monoxide air quality was worst at the two London sites (A and C) with the two highest daily mean levels of 2.3 ppm and 2.9 ppm respectively, being recorded. However, nitric oxide concentrations were highest at sites A and D, with peak daily mean concentrations of 207 ppb and 99 ppb, recorded respectively. The air quality was best at site B with a maximum carbon monoxide mean daily value of only 0.4 ppm and a maximum nitric oxide mean daily value of only 45 ppb. Site D generated a maximum carbon monoxide daily value of 1.5 ppm and site C a maximum nitric oxide daily value of 86 ppb.

Comparison of the maximum carbon monoxide one hour concentrations from the respective sites (site A - 2.9 ppm; site B - 0.8 ppm; site C - 3.5 ppm; site D - 2.5 ppm) with the WHO one hour exposure guideline (25 ppm) indicates that the air quality at all sites is very good. There are no EU limit or guide values for nitric oxide as it is not considered to be harmful to human health.

As with nitric oxide the worst nitrogen dioxide air quality was found at sites A and D with peak daily mean concentrations of 64 ppb and 32 ppb, recorded respectively. The lowest concentrations were found at site B with a peak daily mean of 15 ppb. Site C generated a maximum nitrogen dioxide daily value of 29 ppb. The nitrogen dioxide air quality can be described as very good with the maximum one hour concentrations from the respective sites (site A - 73 ppb; site B - 16 ppb; site C - 31 ppb; site D - 57 ppb) considerably below the WHO one hour exposure guideline of 209 ppb.

CHAPTER 6. THE RELATIONSHIP BETWEEN REMOTELY MEASURED VEHICLE EMISSIONS AND ROADSIDE AIR QUALITY

6.1 Introduction

One of the primary objectives of this research project was to determine what relationship, if any, could be found between remotely sensed on-road vehicle emissions and roadside air quality. In order to accomplish this objective the roadside air quality data (presented in Chapter 5) have been compared with the remotely measured vehicle emission data (presented in Chapter 4). Meteorological data, in particular wind speed and direction, from the nearest weather station to the sampling site has been included in the development of any established relationship. The absence of remotely sensed NO_x measurements meant that no direct comparison could be made with NO_x roadside air quality. Therefore, NO_x vehicle emissions have had to be estimated and then compared to NO_x roadside air quality. In this chapter an examination of the data using graphical and statistical methods is presented.

6.2 Meteorological influences and parameters recorded

Atmospheric phenomena play an important role in the determination of ambient air quality. Meteorological factors have varying degrees of influence upon ambient air quality with the parameters of major significance being wind speed and wind direction, atmospheric stability/turbulence and precipitation *(Hoschele 1987; Pio 1986)*.

Wind speed and direction will directly affect the movement and dispersion of pollutants from emission sources within a given area of study. *Brief & Confer (1972)* have reported that ambient pollution levels, to a first approximation, are inversely related to wind speed. Wind, however, never blows perfectly uniformly, it is turbulent. Wind speed fluctuates in an irregular sequence of gusts and lulls; the wind direction makes correspondingly irregular excursions on either side of the mean, which are effectively cross wind gusts *(Forsdyke 1970)*. There are also variations in wind speed in the vertical direction. These are all manifestations of turbulence, a characteristic of all natural air motion. Turbulence arises from two main causes. The first is the presence of obstacles such as trees and buildings over which the wind blows. These give rise to wind shear and to eddies. The greater surface roughness encountered in urban areas generally increases turbulence and hence, enhances dispersion. The principal effect of the increased turbulence in urban areas is to reduce ground level pollutant concentrations.

The second cause of turbulence is the vertical instability of the air *(Forsdyke 1970)*. Atmospheric instability determines how rapidly a parcel of polluted air is dispersed as it moves away from the source. Dispersion is lower under stable conditions, which occur largely at night, compared to unstable conditions, which are most common during the middle of the day when heating of the ground causes thermal turbulence. Neutral stability, which is the most frequent condition, falls in between the two extremes.

Sometimes, the air temperature, instead of decreasing with height, increases in a layer of limited depth. There is then said to be an inversion of temperature. A temperature inversion is a layer of extreme stability and acts as a lid to the atmosphere, allowing pollutants to build up underneath. Temperature inversions occur at night when there is rapid cooling of the ground, and also when warm air moves in over cold ground. In rural areas the inversion may lie only a few tens of metres above the ground. However, in large urban areas the heat island effect produces inversions which are more typically 100 to 200 metres above the ground.

The conditions that favour high pollution episodes in urban areas involve low wind speeds, stable atmospheric conditions and low mixing heights. These meteorological conditions normally occur during anticyclonic weather and mainly at night. During the day the temperature inversion is likely to break up as the sun warms the ground. However, during the winter months, it is possible for inversions to persist throughout the day and they may survive for several days before breaking up.

Precipitation can take the form of rain, sleet, snow, hail and various combinations thereof. The general effect of precipitation is the scavenging of particulates and gases from the atmosphere *(Hales 1972; Slinn 1977; Marsh 1978)*. The net result of precipitation may be the removal of pollutants from the atmosphere before dispersion takes place.

Meteorological data was collected from the nearest Meteorological Office weather station to each of the monitoring sites. These were the London Weather Centre, for sites A and C, Hartlepool power station for site B and RAF Cottesmore for site D. Due to the research dealing with micro scale pollution problems it was thought that wind speed and wind direction were the most important meteorological parameters. In order not to swamp the statistical analyses with a multitude of independent variables only the above two meteorological parameters have been used.

6.2.1 Modification of wind direction for statistical analysis

For the purposes of statistical interpretation, the wind direction is divided into two components based upon whether the wind direction is upwind or downwind of the source in relation to the monitoring equipment. If the road is assumed to run east/west *(Figure 6.1)* and the air quality monitor is located to the south of the road, then wind directions from north of the road would mean the monitor was downwind of the source and are denoted by positive figures. Wind directions from south of the road would mean the monitor was upwind of the source and are denoted by negative figures.

However, a further transformation of wind direction is necessary to enable its use in statistical analyses. If the downwind hemisphere is considered, a direction of 5° from north and 355° from north are numerically very different but are similar in terms of their effect upon air quality. If for the purposes of the research it is assumed that the effect upon air quality is the same, then both 5° and 355° from north can be given the same value as can all mirrored angles eg 40° and 320°.

The values for the mirrored wind directions are determined by the angle, in degrees, from the line of the road. The line of the road is assigned a value of 0°, as the mirrored wind directions from the east (90° from the north) and west (270° from the north) are considered to have minimal effect on air quality. Therefore, mirrored wind directions of 40° and 320° from north would be assigned a transformed wind direction of 50° and mirrored wind directions of 5° and 355° from north would be assigned a transformed wind direction 85° *(Figure 6.1)*. Upwind is treated in the same fashion but with the directions denoted by minus figures.

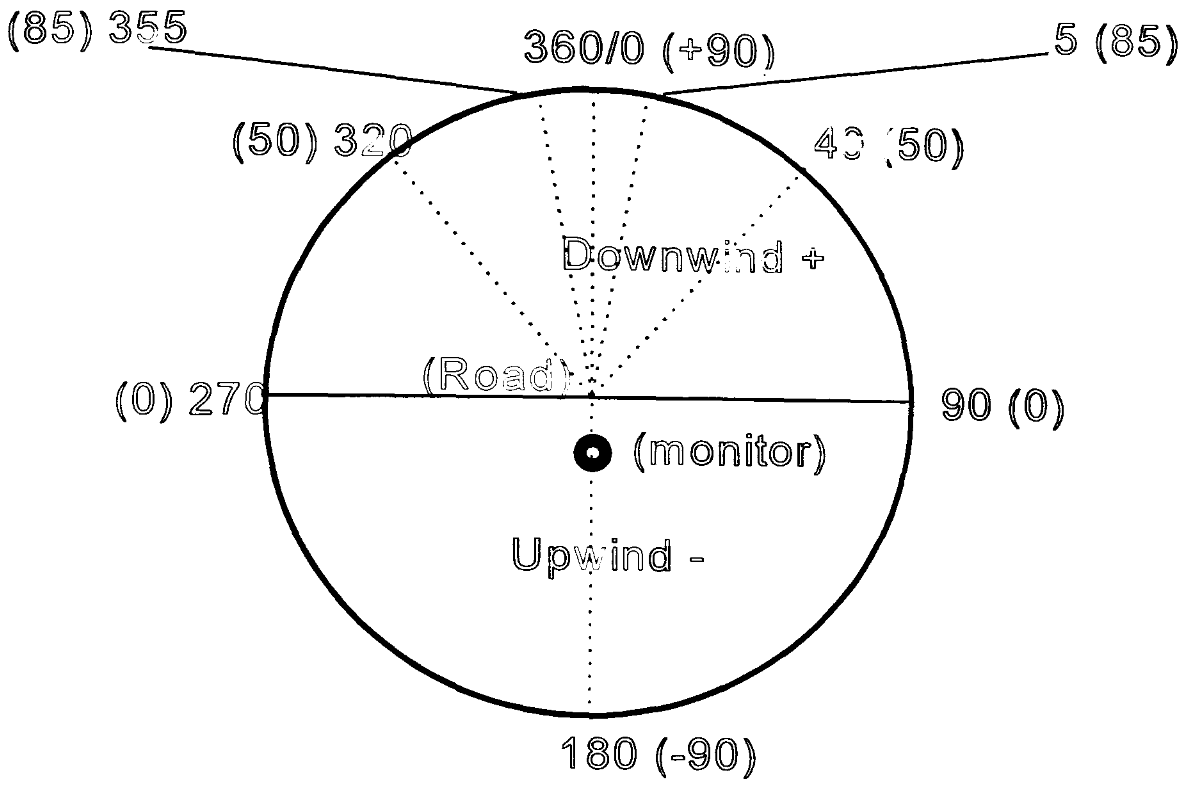


Figure 6.1 Schematic diagram of a site, showing the modified wind directions in brackets, which have been used in the statistical analysis.

6.3 The use of carbon monoxide emissions as an indicator of NO_x emissions

The absence of remotely sensed NO_x measurements prevents direct comparison with NO_x roadside air quality. However, the NO_x vehicle emissions can be estimated from a knowledge of the relationship between NO_X emissions and CO emissions at different vehicle speeds. The estimated emissions are given as total oxides of nitrogen and not NO and NO_2. This is because to obtain an estimated NO or NO_2 emission concentration an assumption about their relative proportions in vehicle exhaust emissions needs to be made. This adds further uncertainty to the calculations and was therefore, not undertaken.

Hickman & Colwill (1982) have postulated the use of the predictive equation:

$$P = CR \qquad \text{Eq. 6.1}$$

Where:
P is the concentration of NO_x emissions emitted from the exhaust pipe
C is the emitted concentration of carbon monoxide from the same source, obtained from remote sensing measurements
R is the ratio of the emission rate of NO_x to that of CO at a known vehicle speed. This value has been obtained from the emission factors calculated by *Latham (1996)*.

The emission ratios (R) between NO_x and CO *(Latham 1996)* are shown in **Table 6.1**. These indicate the relationships between average vehicle speed and the ratios of oxides of nitrogen to that of carbon monoxide. The ratio varies with respect to vehicle operating mode such that the proportion of oxides of nitrogen is small compared to carbon monoxide at low vehicle speeds where the engine is tending to operate inefficiently. At higher speeds, where the combustion process is more efficient, more oxides of nitrogen are formed, and less carbon monoxide.

To enable the calculation of emission ratios appropriate to each site the following assumptions regarding vehicle speeds have been made:

- the mean speed for vehicles at sites A and D was 20 kmph
- the mean speed for vehicles at site B was 50 kmph
- the mean speed for vehicles at site C was 30 kmph

The ratios of the emission factors of NO_x to that of CO at 20, 30 and 50 kmph are:

- 0.10 at 20 kmph
- 0.15 at 30 kmph
- 0.25 at 50 kmph

These ratios have been used to estimate individual vehicle emissions of NO_x at sites A and D, B and C, respectively. They have then been used in the development of any relationship between NO_x emissions and NO_x air quality.

Table 6.1 Performance-related emission factors for non-catalyst petrol cars, as a function of average speed *(Latham 1996)*

Speed (kmph)	Emission factors (g/km)		Ratio
	CO	NO_x	
5	78.5	3.55	0.05
10	39.7	2.17	0.05
15	26.7	2.03	0.08
20	20.3	2.01	0.10
25	16.4	2.01	0.12
30	13.8	2.02	0.15
35	12.0	2.04	0.17
40	10.6	2.07	0.20
45	9.5	2.12	0.22
50	8.6	2.17	0.25
55	7.9	2.24	0.28
60	7.3	2.31	0.32
65	6.9	2.41	0.35
70	6.4	2.52	0.39
75	6.1	2.64	0.43
80	5.7	2.78	0.49

6.4 Investigation of the relationship between remotely measured on-road emissions and roadside air quality

The initial statistical analyses were undertaken using the hourly mean results for both carbon monoxide and nitrogen oxides. However, due to the small amount of hourly data available, the significance levels were not sufficient to provide the confidence that the established relationships were not through random chance alone and were able to reject the null hypothesis. Therefore, the correlation analyses used to measure the strength of the relationships between roadside air quality and remotely measured vehicle emissions, wind speed and wind direction were conducted on half-hourly averaged data. Unfortunately, meteorological data was provided only on an hourly basis and, therefore, it was assumed that both wind speed and wind direction were the same in both half-hourly segments. Multiple regression analysis was then used to extend any developed relationships.

Further statistical analyses were undertaken on the five minute mean averaged data with the intention of a more finely resolved examination of the effect of vehicle emissions on roadside air quality. No weather data was available from the meteorological office over this shorter time period. In order to provide meteorological data an R & D Electronics automatic weather station was purchased so that data collection could be undertaken at site D. However, the automatic weather station malfunctioned and no measurements were obtained. Unfortunately, this meant that wind speed and wind direction could not be included in any of the five minute statistical analyses.

6.4.1 Half-hourly statistical analyses for carbon monoxide

Correlation analysis (section 6.4.1.1) was undertaken using SPSS for Windows to examine the relationships between carbon monoxide air quality and on-road vehicle emissions, carbon monoxide air quality and wind speed and carbon monoxide air quality and wind direction. An essential first step in studying the association between two variables is to plot the data using a scatterplot which gives a qualitative assessment of any relationship. This was undertaken for all sampling days and all independent variables but only a sample of the best and worst relationships are recorded in the thesis. It is more useful to quantify the strength of the association by calculating a summary index. One commonly used measure is the Pearson correlation coefficient, denoted by r, which has been reported for all sampling days and all independent variables. The Spearman correlation coefficient was not used because the Pearson correlation coefficient is a more powerful statistical measure if the variables under investigation are assumed to be approximately normally distributed. This assumption of normality for the variables under investigation is supported by the central limit theorem which shows that the sample mean may be approximately normal even if the parent population has a quite different distribution.

Attempts were then made to develop mathematical models, using multiple linear regression analysis (section 6.4.1.2), to explain carbon monoxide air quality in terms of on-road vehicle emissions, wind speed and wind direction. The goodness of fit of the model is given by the coefficient of determination (r^2), which gives the proportion of the variation of the dependent variable (carbon monoxide air quality) explained by the independent variables (on-road vehicle emissions, wind speed and wind direction).

6.4.1.1 Correlation analysis

The correlation coefficients between carbon monoxide air quality and the three independent variables (remotely measured vehicle emissions, wind speed and wind direction) for all sites and locations are shown in **Table 6.2**. This table also includes the confidence levels for each correlation. Thus, for example, at site A (1 June 1994) there is a 74% confidence level that the association between carbon monoxide air quality and on-road vehicle emissions did not occur through random chance.

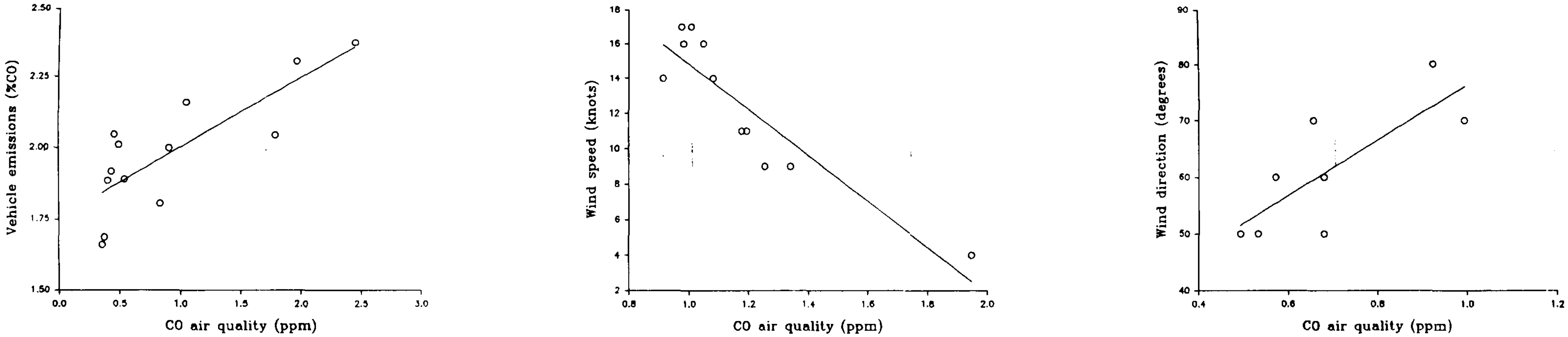
In general, on-road vehicle emissions can be seen to demonstrate positive correlations with carbon monoxide air quality, with the data for 21 October 1995 (15 metres from the road - *Figure 6.2A)* showing the highest positive correlation ($r = 0.81$) and the results for 8 August 1995 (7.5 metres from the road - *Figure 6.2B)* showing the lowest positive correlation ($r = 0.17$). However, the reverse is also apparent with negative correlations for carbon monoxide air quality and on-road vehicle emissions existing for all locations at site B and for 20 October 1995 (7.5 metres from the road) at site D. It is very unlikely that these negative correlations are a consequence of a causal relationship in which increasing vehicle emissions lead to decreasing ambient levels.

A possible explanation for the negative correlation occurring on 20 October 1995 is that

Table 6.2 Half-hourly inter-correlation coefficients between carbon monoxide air quality and the independent variables at each site.

Site	Location/distance from the road (metres)	Date	Vehicle emissions (Confidence levels)	Wind speed (Confidence levels)	Wind direction (Confidence levels)
A	kerbside	1 June 1994	0.25(74)	-0.61(96)	-0.34(81)$^{-}$
A	7.5	31 May 1994	0.34(84)	-0.41(89)	0.60(97)$^{-}$
A	15	27 May 1994	0.56(96)	-0.73(99)	-0.16(68)$^{+}$
B	kerbside	10 June 1994	-0.44(86)	-0.69(97)	0.78(99)$^{+}$
B	kerbside	7 June 1994	-0.55(87)	const	-0.17(63)$^{+}$
B	7.5	8 June 1994	-0.29(79)	-0.14(65)	-0.19(70)$^{-}$
C	kerbside	8 March 1995	0.63(98)	-0.56(96)	-0.72(99)$^{-}$
C	7.5	8 August 1995	0.17(70)	0.29(82)	0.07(58)$^{+}$
C	15	9 August 1995	0.44(93)	-0.74(99)	0.55(97)$^{+}$
D	roadside	22 October 1995	0.68(99)	-0.30(82)	0.40(89)$^{-}$
D	kerbside	19 October 1995	0.65(99)	-0.90(99)	0.08(59)$^{-}$
D	7.5	20 October 1995	-0.26(79)	-0.67(99)	-0.60(98)$^{+}$
D	15	21 October 1995	0.81(99)	-0.76(99)	0.61(99)$^{-}$

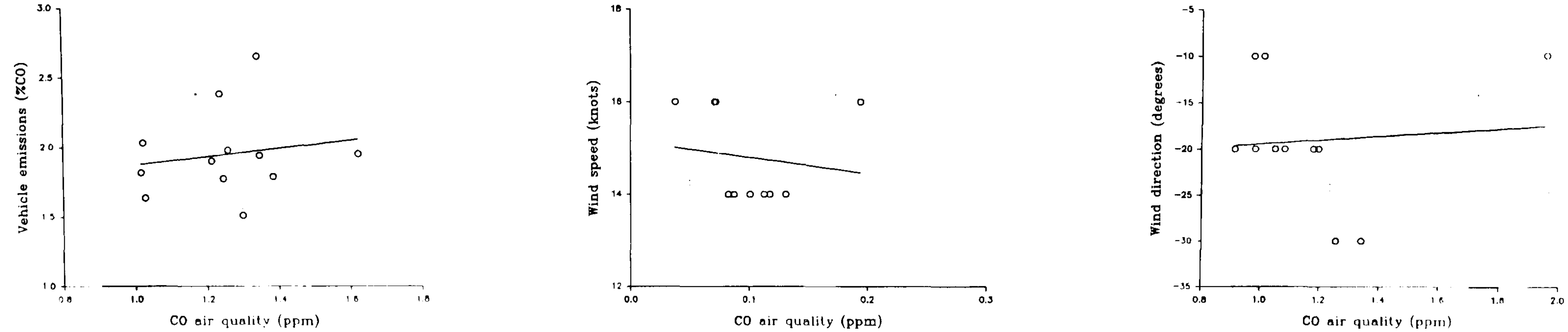
$^{-}$ = wind direction upwind of the source(road) $^{+}$ = wind direction downwind of the source(road)



A Vehicle emissions (r = 0.81) for 21 October 1995 (site D).

C Wind speed (r = -0.90) for 19 October 1995 (site D)

E Wind direction (r = 0.78) for 10 June 1994 (site B)



B Vehicle emissions (r = 0.17) for 31 May 1994 (site A)

D Wind speed (r = -0.14) for 8 June 1994 (site B)

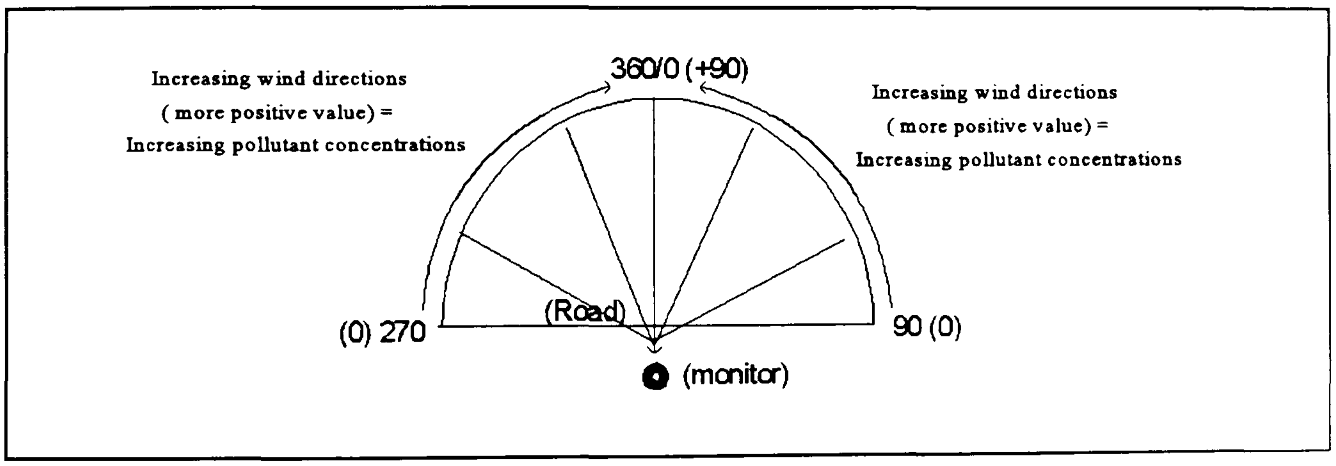
F Wind direction (r = 0.08) for 19 October 1995 at site D.

Figure 6.2 Scatter plots of the independent variables (on-road vehicle emissions, A & B; wind speed, C & D; wind direction, E & F) against carbon monoxide air quality the opposite carriageway, which is assumed to have the same characteristics as the monitored carriageway, had stationary traffic for most of the afternoon. Therefore, the emissions from these stationary vehicles could have been masking the effect of the emissions from the free flowing monitored highway with respect to influence on CO air quality. The consistently negative correlations at site B could be due to the wind strengths encountered during sampling as these were, on average, higher than at all other sites. The highest average daily wind speed across all sites was 16 knots, recorded for 7 June 1994, with the second highest of 14.8 knots recorded for 8 June 1994. The wind speed for 10 June, although somewhat lower, remained high at a daily average value of 9.2 knots. It is postulated that these strong winds could have been dispersing vehicle emissions so quickly that their reference to air quality is made worthless and hence, negative correlations arise. In other words wind speed was the predominant influence on the determination of air quality with the impact from on-road vehicle emissions being effectively screened out.

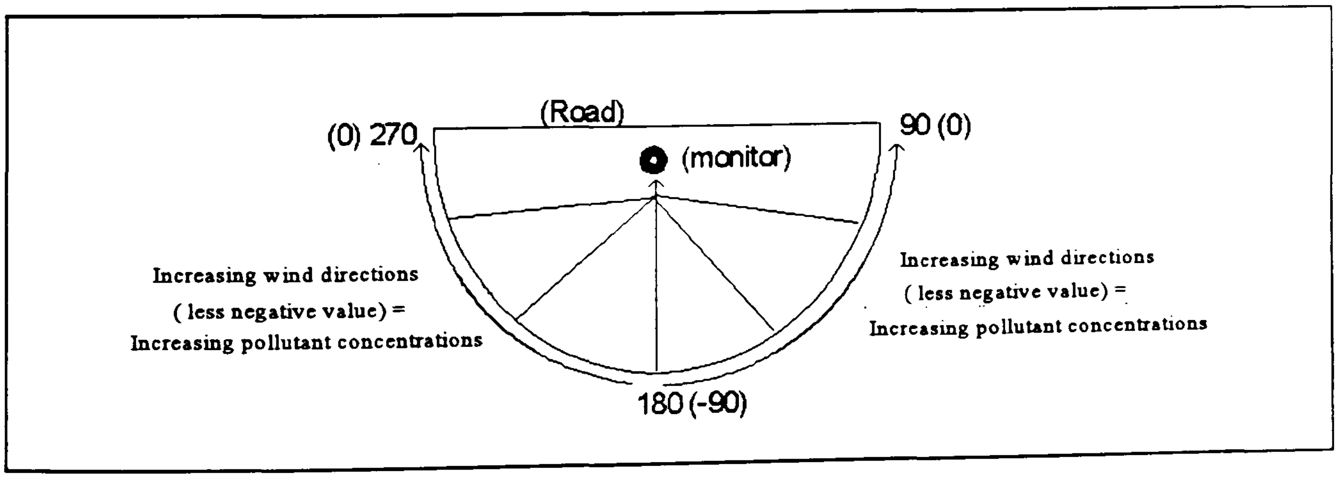
It might be expected that the association between air quality and vehicle emissions would decline with distance from the road as vehicle emissions are dispersed. However, no such relationship was found at any site. Site A was the only site which displayed any consistent pattern and this was contrary to expectations with the association between air quality and vehicle emissions increasing with distance from the road. The absence of the expected relationship could be due to sampling having been conducted at different distances on different days. If sampling had been conducted at the different distances simultaneously a relationship may have been found.

The correlations between carbon monoxide air quality and wind speed are predominantly negative **(Table 6.2)** as would be expected. Standard Gaussian theory states that pollutant concentrations are inversely proportional to wind speed *(Brief and Confer 1972)*, such that, at higher wind speeds there will be greater dispersion and therefore, lower concentrations. This is best demonstrated on 19 October 1995 (kerbside - *Figure 6.2C)* which gave the highest negative correlation (r = -0.90) of CO air quality with wind speed although even on 8 June 1994 (7.5 metres from the road - *Figure 6.2D)*, which has the lowest negative correlation (r = -0.14), the relationship remains valid.

The correlations displayed in **Table 6.2** between carbon monoxide air quality and wind direction do not follow either of the patterns set for vehicle emissions or wind speed. An almost equal number of negative and positive correlations can be seen to exist with no distinct pattern between the individual sites. The initial expectation was for positive correlations with wind direction to be found, such that, when the air quality monitors were downwind of the road *(Figure 6.3A)*, a more perpendicular wind direction to the road would lead to increased concentrations as was found for 10 June 1994 (kerbside - *Figure 6.2E)* which gave the highest correlation coefficient (r = 0.78). In situations where the air quality monitors were upwind of the road *(Figure 6.3B)* a wind direction more parallel to the road would lead to increased concentrations as was found for 19 October 1995 (kerbside - *Figure 6.2F)* which gave a correlation coefficient of 0.08.



A Downwind



B Upwind

Figure 6.3 Schematic diagram of a site showing the effect of increasing wind direction on air quality for **A**, downwind and **B**, upwind

Positive correlations between wind direction and air quality, with increasing wind directions leading to increasing concentrations, were not found for each sampling day. However, if the CO regression analysis section (6.4.1.2) is examined, those sampling days where there is a 90% confidence level in a linear relationship between the dependent variable and all the independent variables, show wind direction to have a positive effect upon ambient concentrations on all but one of these sampling days.

6.4.1.2 Regression analysis

Multiple linear regression analyses were undertaken in an attempt to develop mathematical expressions explaining carbon monoxide air quality in terms of on-road vehicle emissions, wind speed and wind direction. Multiple linear regression extends bivariate regression, as used in Chapter 4, by incorporating multiple independent variables. The model can be expressed as:

$$Y_i - B_0 + B_1 X_1 i + B_2 X_2 i + \ldots\ldots + B_l X_{li} \qquad \text{Eq. 6.2}$$

Where : B_0 is the intercept of the plotted data

$B_1, B_2\ldots$ are the partial regression coefficients

X_{li} indicates the value of the lth independent variable for case i.

Unlike the correlation analyses in which the r value was calculated for all data sets with a specified confidence level, the regression analyses were only conducted on the data from those days where there was a 90% confidence level (**Table 6.3**) that a linear relationship between the dependent variable and the entire set of independent variables existed. These are marked with an asterisk in **Table 6.3**. The 90% significance level was applied to the regression analyses and not the correlation analyses because it was extending the statistical interpretation and therefore, a higher level of confidence in the data was necessary.

Regression analysis was performed on the data from eight sampling days which are: 1 June 1994, 31 May 1994, 27 May 1994, 8 March 1995, 9 August 1995, 22 October 1995, 19 October 1995 and 21 October 1995. The data for 27 May 1994 are included in the analysis as it fails the confidence threshold by only 1% (**Table 6.3**). However, the data for 20 October 1995 are not included in the analysis even though the 90% threshold is achieved because a Type 1 error (ie, the null hypothesis has been rejected when it should have been accepted) has occurred. The null hypothesis should have been accepted because it is apparent from the correlation analysis for 20 October 1995 **(Table 6.2)** that there is a spurious relationship between air quality and on-road vehicle emissions, thereby invalidating the regression model.

Table 6.3 Confidence levels for the relationships between CO air quality and the independent variables on each monitoring day

Site	Location/distance (metres)	Date	Confidence level
A	kerbside	1 June 1994	95*
A	7.5	31 May 1994	91*
A	15	27 May 1994	89*
B	kerbside	10 June 1994	79
B	kerbside	7 June 1994	61
B	7.5	8 June 1994	14
C	kerbside	8 March 1995	97*
C	7.5	8 August 1995	25
C	15	9 August 1995	97*
D	roadside	22 October 1995	91*
D	kerbside	19 October 1995	99*
D	7.5	20 October 1995	99*
D	15	21 October 1995	99*

* Data sets used in regression analysis

The regression equations for the relationships between carbon monoxide air quality and mean on-road vehicle emissions, wind speed and wind direction for the different sites and dates are given below:

Site A; 1 June 1994 (kerbside)($r^2 = 0.76$)

$$CO = 0.762A - 0.076B + 0.005C + 1.816 \qquad \text{Eq. 6.3}$$

Site A; 31 May 1994 (7.5 metres from the road)($r^2 = 0.59$)

$$CO = 0.410A - 0.148B + 0.005C + 2.056 \qquad \text{Eq. 6.4}$$

Site A; 27 May 1994 (15 metres from the road)($r^2 = 0.55$)

$$CO = 0.101A - 0.144B + 0.0004C + 1.146 \qquad \text{Eq. 6.5}$$

Site C; 8 March 1995 (kerbside)($r^2 = 0.69$)

$$CO = 0.409A - 0.178B - 0.033C + 2.343 \qquad \text{Eq. 6.6}$$

Site C; 9 August 1995 (15 metres from the road)($r^2 = 0.60$)

$$CO = 0.147A - 0.310B + 0.004C + 2.943 \qquad \text{Eq. 6.7}$$

Site D; 22 October 1995 (roadside)($r^2 = 0.57$)

$$CO = 0.657A - 0.004B + 0.007C - 0.480 \qquad \text{Eq. 6.8}$$

Site D; 19 October 1995 (kerbside)($r^2 = 0.93$)

$$CO = 0.263A - 0.064B + 0.012C + 1.815 \qquad \text{Eq. 6.9}$$

Site D; 21 October 1995 (15 metres from the road)($r^2 = 0.78$)

$$CO = 1.811A - 0.235B + 0.012C - 0.327 \qquad \text{Eq. 6.10}$$

Where:
CO = predicted carbon monoxide air quality (ppm)
A = mean on-road vehicle emissions (%)
B = wind speed (knots)
C = wind direction (degrees from perpendicular)

The regression equations can be used to predict ambient concentrations of carbon monoxide on the sampling days. The r^2 value is interpreted as the percentage of variability in the dependent variable that is explained by the regression model and therefore, the r^2 value gives a good indication of how accurate the prediction will be.

Figures 6.4, 6.5 and 6.6 display the predicted and observed concentrations at site A, as a time series, for each monitoring distance from the road. The kerbside location *(Figure 6.4)* has the best fit between the predicted and observed concentrations with an r^2 value of 0.76. The changes in observed concentrations are mirrored well by the changes in predicted concentrations such that there is a high degree of agreement in the location of peaks and troughs in air quality.

This is in contrast to the site located 7.5 metres from the road *(Figure 6.5)* where the agreement between observed and predicted concentrations is less good ($r^2 = 0.59$). There is a pronounced smoothing of the predicted concentrations relative to the observed values between 13:00 - 15:00 hours. A possible explanation for this could be that although it has been assumed that meteorological conditions remain the same throughout an hour this could be incorrect between 13:00 and 15:00 hours, with shifting wind directions or speeds leading to the fluctuations in observed values which can not be accurately reflected in the predicted concentrations.

The predicted concentrations for the site located 15 metres from the road (*Figure 6.6* - $r^2 = 0.55$) are also unable to accurately follow some of the variations of the observed values. However, this smoothing is markedly less than occurred 7.5 metres from the road. A further point for consideration is the very low ambient concentrations recorded at this site. This means that even a large differential between observed and predicted values in *Figure 6.6* is a small difference in concentration, such that, the largest residual between observed and predicted values is less than 0.2 ppm.

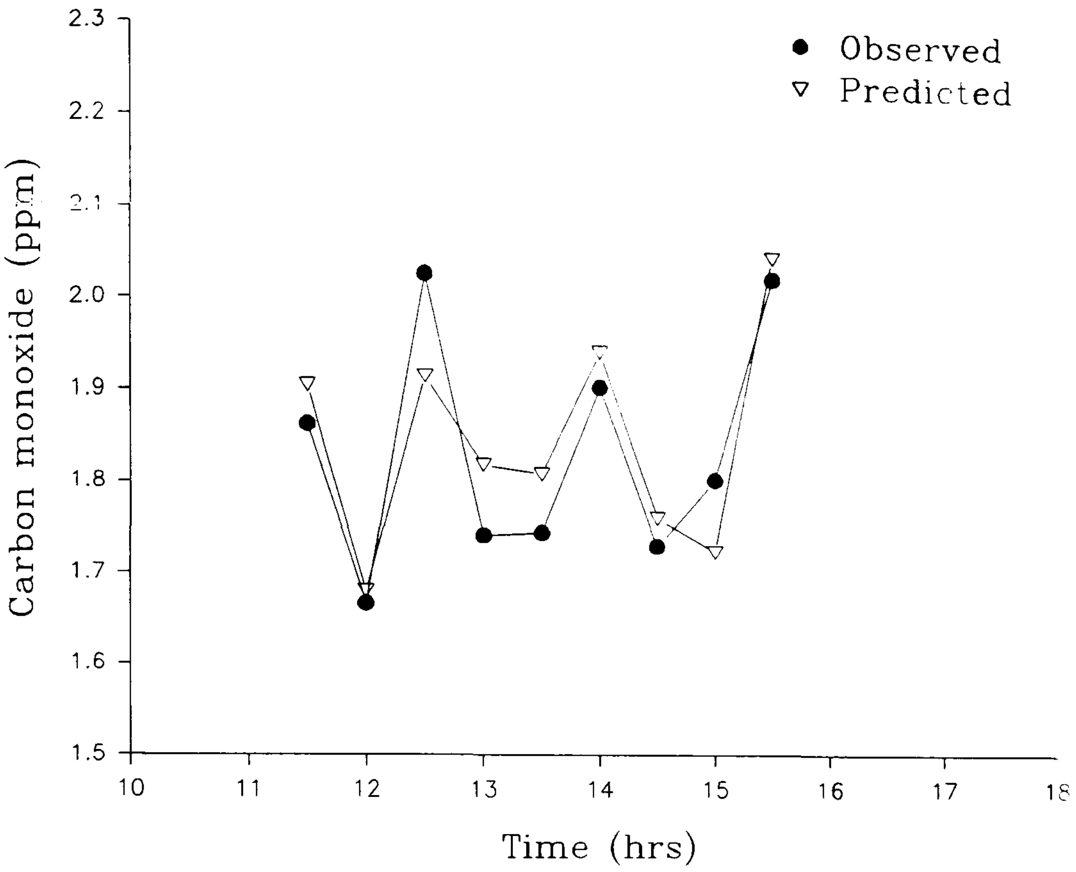


Figure 6.4 Plot of predicted and observed carbon monoxide air quality for 1 June 1994 at site A (kerbside) ($r^2 = 0.76$).

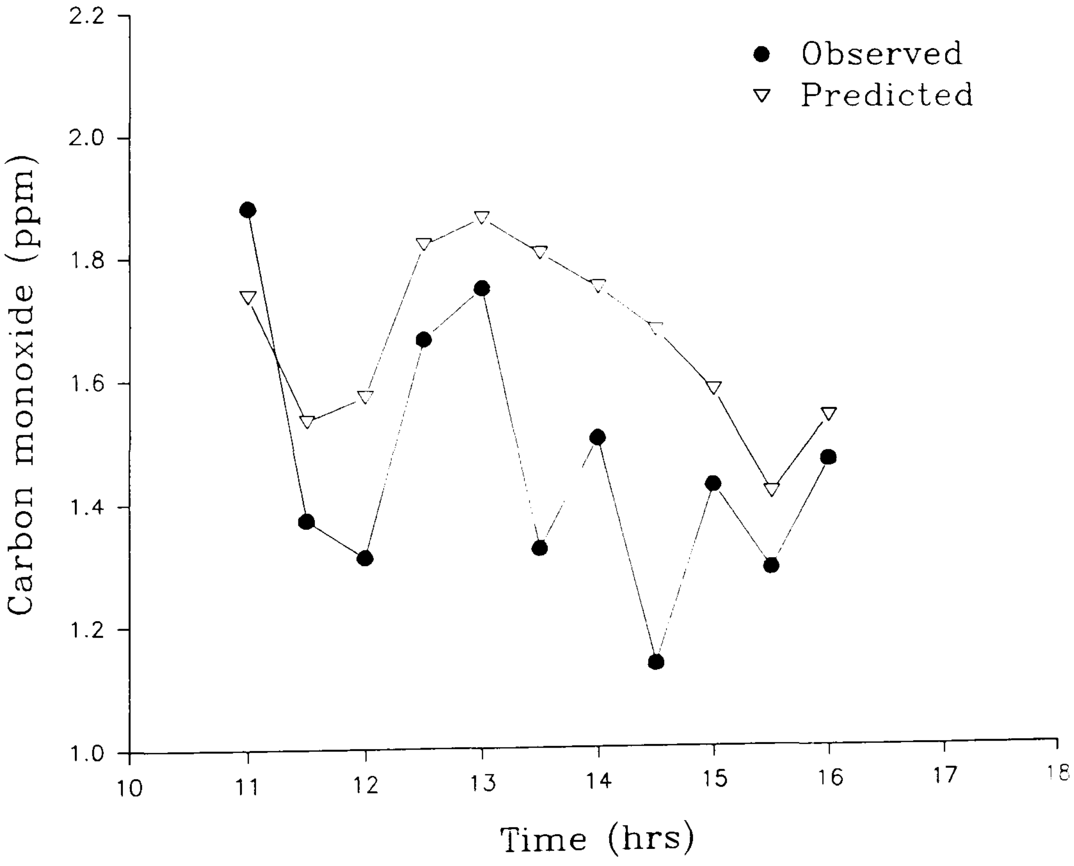


Figure 6.5 Plot of predicted and observed carbon monoxide air quality for 31 May 1994 at site A (7.5 metres from the road) ($r^2 = 0.59$)

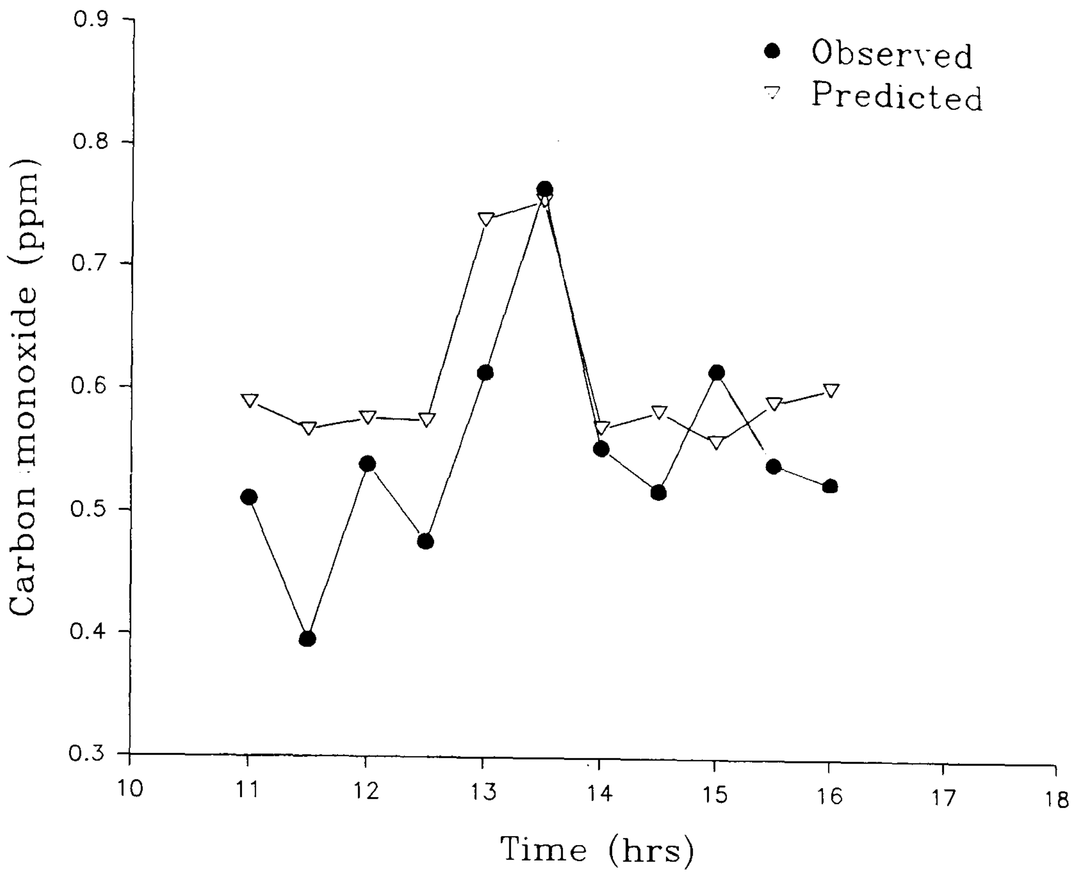


Figure 6.6 Plot of predicted and observed carbon monoxide air quality for 27 May 1994 at site A (15 metres from the road) (r^2 = 0.55)

Site B is not represented in the regression analysis due to the low statistical significance displayed for the sampling days (**Table 6.3**). Site C is represented by the kerbside monitoring location (*Figure 6.7* - r^2 = 0.69) and the monitoring position located 15 metres from the road (*Figure 6.8* - r^2 = 0.60). As for site A, the kerbside location produces a better fit between predicted and observed concentrations. The changes in observed concentrations are mirrored well by the changes in predicted concentrations with the exception of the large decrease in observed concentrations between 11:00 and 11:30 which is less emphatic for the predicted values. Predicted values are missing from *Figure 6.8* for the period 9:30 to 10:00 due to an absence of meteorological data for that time period. The most pronounced difference between predicted and observed values in *Figure 6.8* occurs at lunchtime (12:00 -13:30) with sizeable reductions in predicted concentrations relative to the observed values. The reason for this is unclear although the reductions do correspond to a change in wind direction, with the wind directions between 12:00 - 13:30 being more parallel to the road.

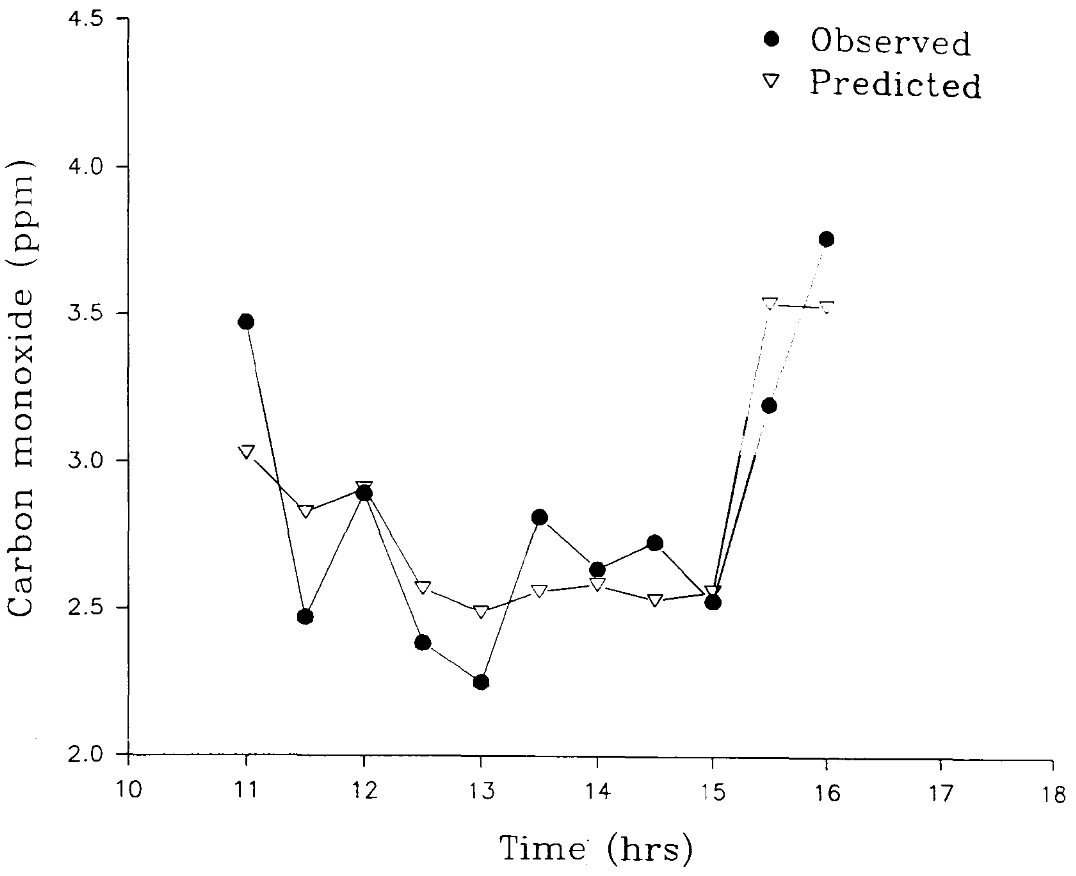


Figure 6.7 Plot of predicted and observed carbon monoxide air quality for 8 March 1995 at site C (kerbside) (r^2 = 0.69).

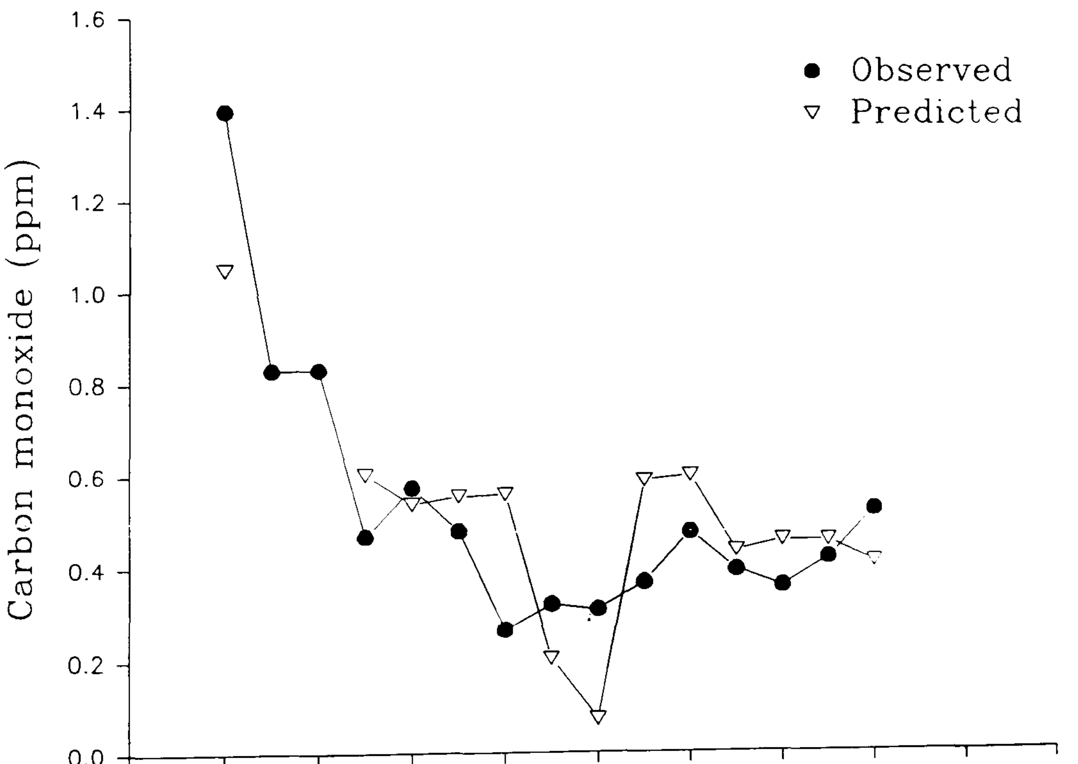


Figure 6.8 Plot of predicted and observed carbon monoxide air quality for 9 August 1995 at site C (15 metres from the road) (r^2 = 0.60)

Site D displays, on average, the best agreement between observed and predicted concentrations *(Figures 6.9, 6.10 and 6.11)* with calculated r^2 values of 0.57, 0.93 and 0.78 for the monitoring positions at roadside, kerbside and 15 metres from the road, respectively. *Figure 6.10* shows the best agreement between predicted and observed values at site D with little deviation between the two over the full sampling day. The worst agreement between predicted and observed values at site D occurs for the 22 October when the CO values were considerably reduced and consequently the largest residual is less than 0.2 ppm. *Figure 6.11* shows good agreement between predicted and observed values between 11:00 and 14:00, however, after this point the observed values remained reasonably constant whereas the predicted values are seen to fluctuate.

A similarity for all sites is that the kerbside location generated the best regression model thereby producing the closest fit between the observed and predicted concentrations and the highest r^2 values for each site. One inference that could be drawn from this, is that the regression modelling worked best the nearer the sampling site was to the road. However, perhaps a more correct interpretation would be that the regression modelling worked best on those sampling days with the highest CO concentrations at the respective sites. The reduction in efficiency of the regression modelling on the other sampling days could be related to the low CO concentrations encountered. These concentrations could be considered on some days to be at a background level and therefore beyond the scope of the regression models being used.

Site D, the monitoring position 15 metres from the road *(Figure 6.11)* produced the highest peak concentrations and not the kerbside location *(Figure 6.10)* although this produced the best agreement between the observed and predicted values. This could be interpreted as being contrary to the inference that the regression modelling worked best on those sampling days with the highest CO concentrations. However, the mean concentrations support the inference, with the kerbside location (1.2 ppm) having a higher mean value than the site positioned 15 metres from the road (0.93 ppm) during the regression modelling period.

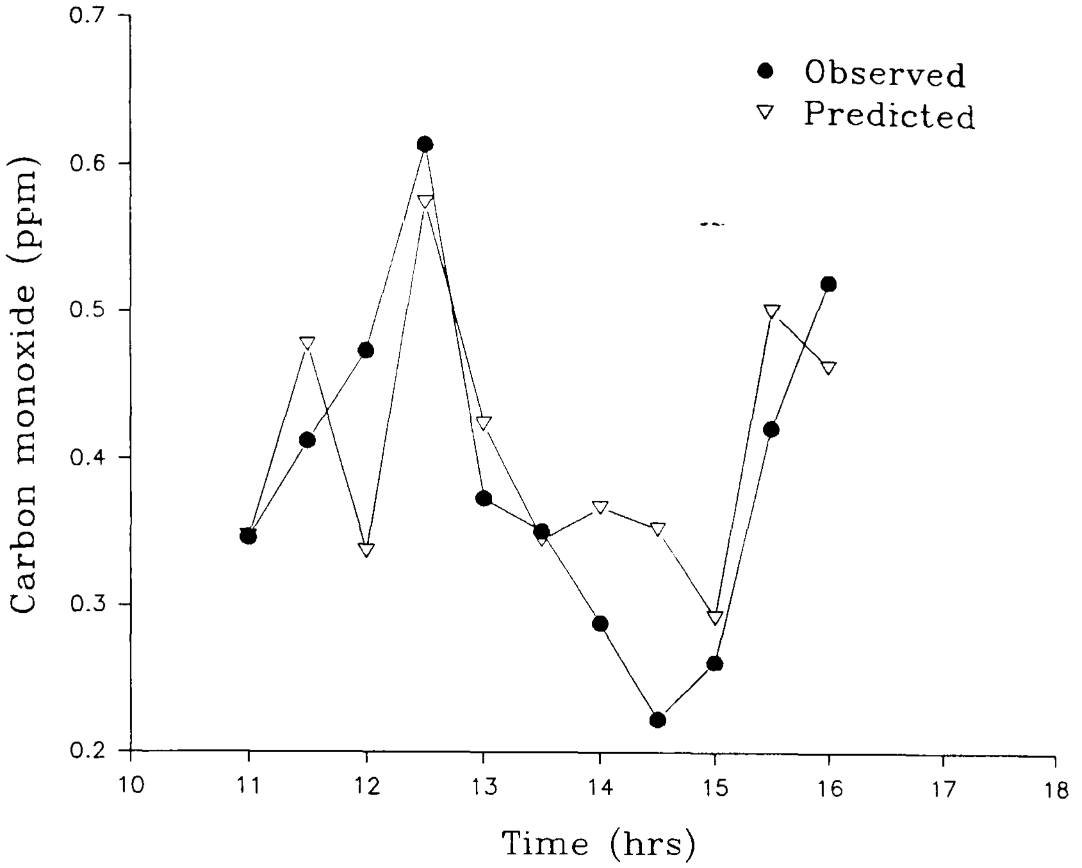


Figure 6.9 Plot of predicted and observed carbon monoxide air quality for 22 October 1995 at site D (roadside) (r^2 = 0.57).

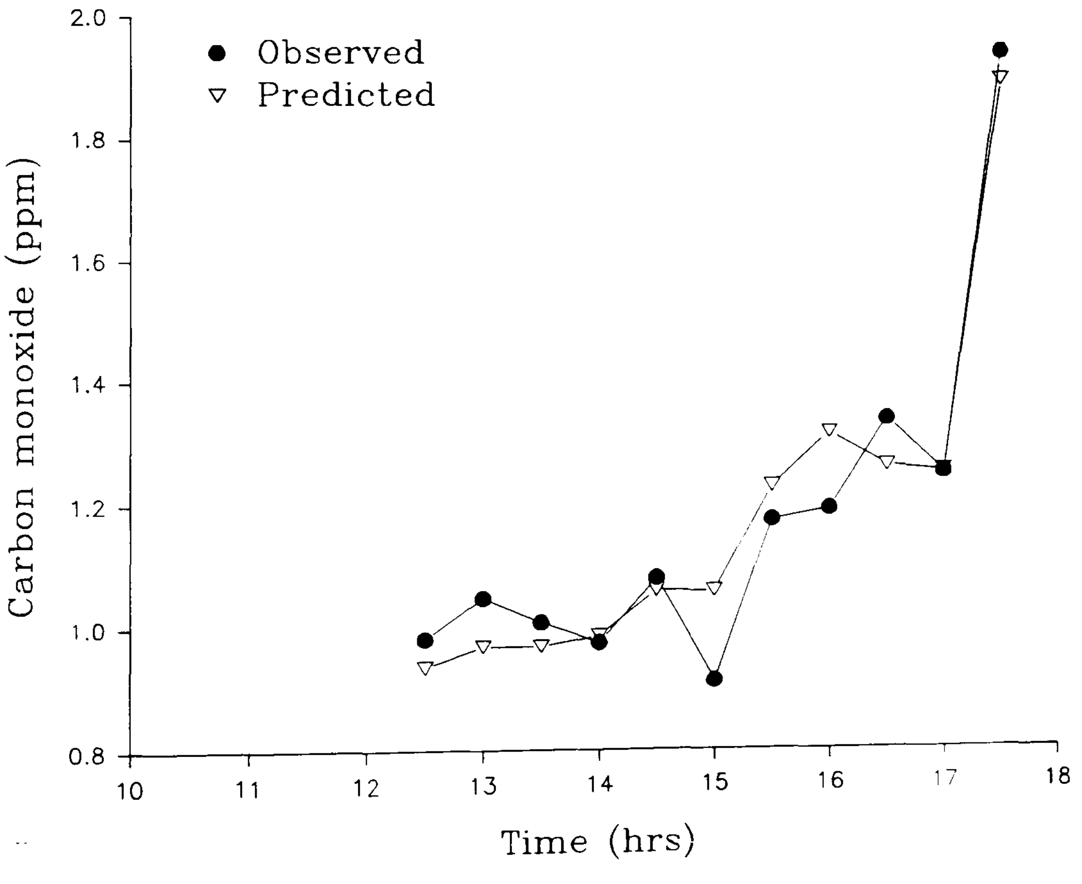


Figure 6.10 Plot of predicted and observed carbon monoxide air quality for 19 October 1995 at site D (kerbside) (r^2 = 0.93).

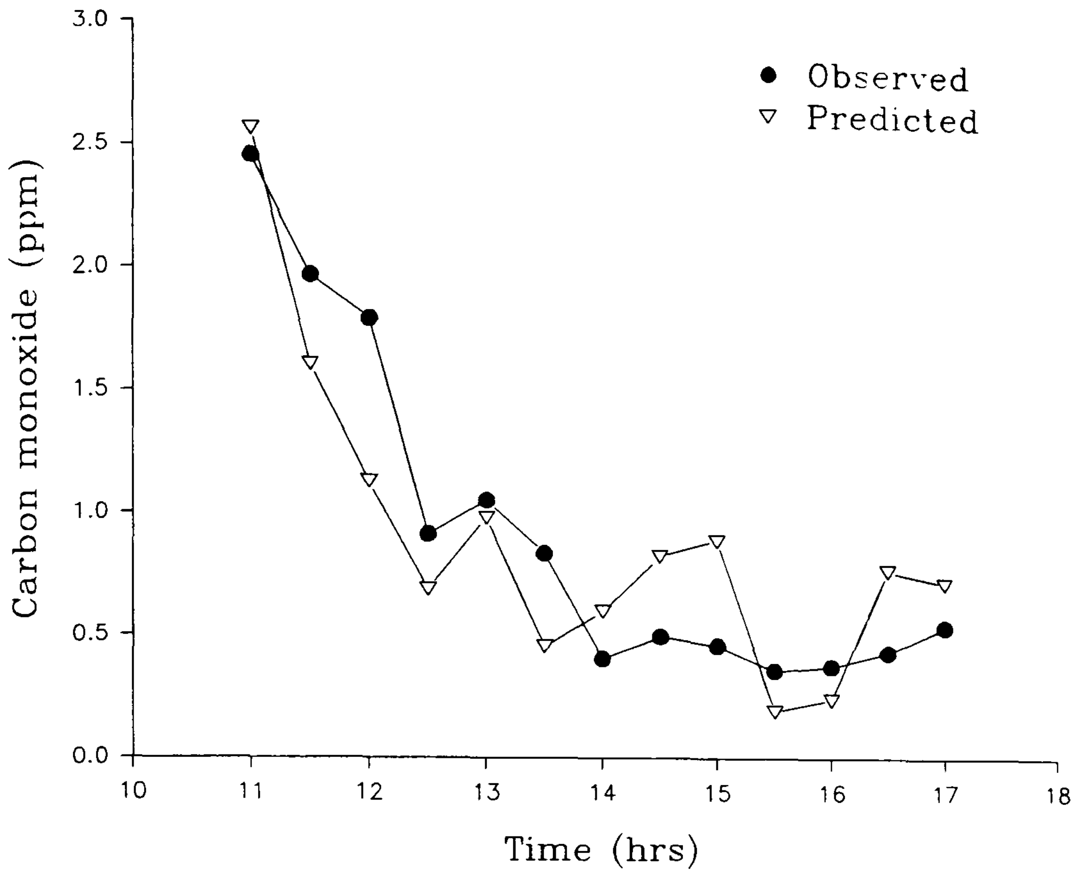


Figure 6.11 Plot of predicted and observed carbon monoxide air quality for 21 October 1995 at site D (15 metres from the road) (r^2 = 0.78).

All sampling days included in the regression analysis recorded r^2 values of greater than 0.5, such that for each sampling day at least half the variation in CO air quality is explained by the variation in, on-road vehicle emissions, wind speed and wind direction.

It is important to note that prediction is always difficult and can be misleading if it is attempted outside the range of the data available. Any extrapolation of the predicted CO concentrations into the evening/night or into the early morning would be unwise as the meteorological conditions and vehicle emission characteristics would be very different to those monitored and thus, invalidate the regression model being used. Therefore, no extrapolation of the predicted concentrations beyond the monitoring periods has been attempted although prediction would be possible if the monitoring periods had been extended.

6.4.1.2.1 Interpretation of beta coefficients

When examining the regression equations it is inappropriate to interpret the partial regression coefficients as indicators of the relative importance of the independent variables. The actual magnitude of the coefficients depends on the units in which the variables are measured and as all variables were measured in different units (ppm for air quality; percent for vehicle emissions; knots for wind speed; degrees for wind direction) their coefficients are not directly comparable.

However, the regression coefficients can be made somewhat more comparable if their beta weights are calculated **(Table 6.4)**, which are the coefficients of the independent variables when all variables are expressed in standardised (Z score) form. This therefore removes the influence of the units in which the variables were measured from the coefficients. However, the beta coefficients like the regression coefficients are contingent on the other independent variables in the equation and are affected by the correlations of the independent variables and do not in any absolute sense reflect the importance of the various independent variables.

Table 6.4 Beta coefficients for the independent variables for specific sampling dates

Site	Location/distance (metres)	Date	Vehicle emissions	Wind speed	Wind direction
A	Kerbside	1 June 1994	0.738	-1.392	0.826
A	7.5	31 May 1994	0.346	-0.467	0.470
A	15	27 May 1994	0.166	-0.617	-0.042
C	Kerbside	8 March 1995	0.159	-0.390	-0.521
C	15	9 August 1995	0.151	-0.535	0.258
D	Roadside	22 October 1995	0.611	-0.671	0.338
D	Kerbside	19 October 1995	0.116	-0.920	0.294
D	15	21 October 1995	0.554	-0.295	0.171

Analysis of the beta coefficients on a daily basis shows wind speed to have the greatest influence upon carbon monoxide air quality on five of the sampling days (1 June 1994, 27 May 1994, 9 August 1995, 19 October 1995 and 22 October). This influence does not appear to be distance dependent with all distances from the road being represented, with the exception of 7.5 metres from the road although this distance may also be included as wind speed is only marginally less important than wind direction. Neither does the influence appear to be site specific with all sites in the sample represented. However, the relative importance of wind speed was expected as it has a primary role in dispersing vehicle emissions.

On-road vehicle emissions appear to have the least influence upon carbon monoxide air quality with only one day (21 October 1995) indicating that vehicle emissions are the predominant influence. This coincided with a relatively low mean daily wind speed (6.3 knots) with a consequent slower rate of dispersion and may be the reason for the relative importance of on-road vehicle emissions to air quality on 21 October 1995.

An analysis of the beta weights for wind direction shows it to be the most important factor in determining air quality on 31 May 1994 and 8 March 1995 with both sampling days having wind blowing from upwind of the source relative to the air quality monitors. However, the wind direction being upwind of the source may not be relevant to an explanation of its primary influence on air quality on these dates as other sampling days (1 June 1994, 31 May 1994, 9 August 1995, 22 October 1995, 19 October 1995 and 21 October 1995) also had winds blowing from a predominantly upwind direction but did not have a primary influence on air quality.

An objective of the research was to develop a regression model to explain carbon monoxide air quality by on-road vehicle emissions, wind speed and wind direction and to be able to apply this model to any location. In order for this to be possible it was necessary for similar relationships to have been found on the different sampling days. However, it is clear from equations 6.3 - 6.10 that there is little agreement between the different sampling days although the beta coefficients do provide a general similarity in trend. A supplementary investigation to determine if a site specific model could be developed also foundered on the lack of similarity in relationships within sites.

6.4.2 Half-hourly statistical analyses for nitrogen oxides

As with carbon monoxide, correlation analysis (section 6.4.2.1) was undertaken using SPSS for Windows to examine the relationships between NO_x air quality and on-road vehicle emissions, NO_x air quality and wind speed and NO_x air quality and wind direction. An essential first step, identified in the carbon monoxide analysis, when studying the association between two variables is to plot the data using a scatterplot. This was undertaken for all sampling days and all independent variables but only a sample of the best and worst relationships are recorded in the thesis. However, it is more useful to quantify the strength of the association by calculating the Pearson correlation coefficient, which has been reported for all sampling days and all independent variables. Attempts were then made to develop mathematical models, using multiple linear regression analysis (section 6.4.2.2), to explain NO_x air quality in terms of on-road vehicle emissions, wind speed and wind direction.

6.4.2.1 Correlation analysis

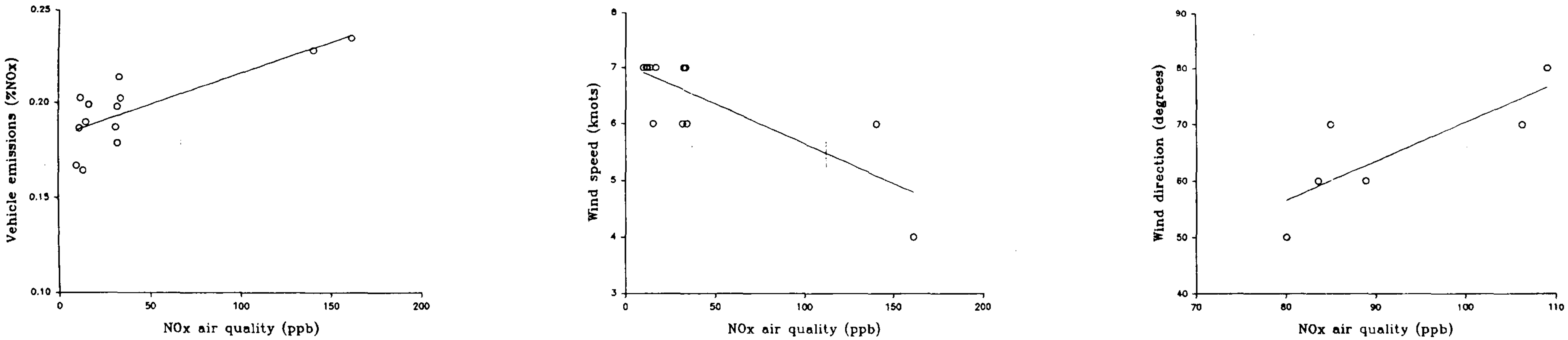
The correlation coefficients between NO_x air quality and the three independent variables (remotely measured vehicle emissions, wind speed and wind direction) together with their confidence levels are shown for all sites and locations in **Table 6.5**. Correlation analysis was not undertaken on 7 June 1994 due to the absence of any NO_2 data (see section 5.4.2.2) or 9 August 1995 due to dubious NO_2 data (see section 5.4.3.2). The correlation coefficients show a similar situation to that found for carbon monoxide with, in general, on-road vehicle emissions having positive correlations with air quality. The highest positive correlation ($r = 0.79$) was, as with CO, calculated for 21 October 1995 (15 metres from the road -*Figure 6.12A)* and the lowest positive correlation ($r = 0.10$) was calculated for 31 May 1994 (7.5 metres from the road - *Figure 6.12B)*.

However, as with CO, there are also negative correlations for on-road vehicle emissions and air quality. These negative correlations largely occur on the same sampling days for NO_x as they did for CO, that is, all locations at site B and on 20 October 1995 at site D. An exception is 8 March 1995 which gave a positive correlation for CO. Possible explanations of the negative correlations for CO at site B and on 20 October

Table 6.5 Half-hourly inter-correlation coefficients between NO_x air quality and the independent variables at each site.

Site	Location/distance from the road (metres)	Date	Vehicle emissions (Confidence levels)	Wind speed (Confidence levels)	Wind direction (Confidence levels)
A	kerbside	1 June 1994	0.60(96)	-0.30(78)	-0.35(82)$^-$
A	7.5	31 May 1994	0.10(62)	-0.48(93)	-0.23(75)$^-$
A	15	27 May 1994	0.47(93)	-0.54(96)	-0.19(71)$^+$
B	kerbside	10 June 1994	-0.35(75)	-0.74(95)	0.81(98)$^+$
B	kerbside	7 June 1994	-	-	-
B	7.5	8 June 1994	-0.50(93)	-0.17(68)	-0.04(55)$^-$
C	kerbside	8 March 1995	-0.08(60)	-0.47(93)	-0.20(72)$^-$
C	7.5	8 August 1995	0.42(91)	0.26(79)	0.49(94)$^+$
C	15	9 August 1995	-	-	-
D	roadside	22 October 1995	0.57(97)	-0.22(74)	0.28(80)$^-$
D	kerbside	19 October 1995	0.67(99)	-0.46(92)	0.58(97)$^-$
D	7.5	20 October 1995	-0.77(99)	-0.20(63)	-0.20(63)$^-$
D	15	21 October 1995	0.79(99)	-0.79(99)	0.67(99)$^-$

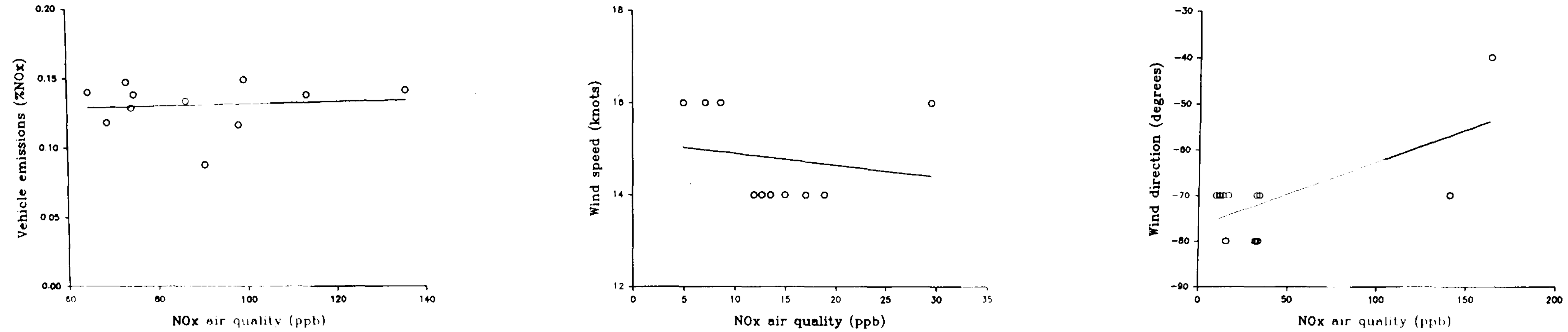
$^-$ = wind direction upwind of the source(road) $^+$ = wind direction downwind of the source(road)



A Vehicle emissions (r = 0.79) for 21 October 1995 (site D)

C Wind speed (r = -0.79) for 21 October 1995 (site D)

E Wind direction (r = 0.81) for 10 June 19945 (site B)



B Vehicle emissions (r = 0.10) for 31 May 1994 (site A)

D Wind speed (r = -0.17) for 8 June 1994 (site B)

F Wind direction (r = 0.67) for 21 October 1995 (site D)

Figure 6.12 Scatter plots of the independent variables (on-road vehicle emissions, A & B; wind speed, C & D; wind direction, E & F) against NO_x air quality

1995 at site D were given in section 6.4.1.1 and it is reasonable to assume that the same explanations would hold for NO_x. To summarise, strong winds could have been responsible for the negative correlations at site B and the negative correlation on 20 October 1995 may have been caused by emissions from stationary traffic in the opposite carriageway masking the effect of the emissions from the monitored lane of traffic. The negative correlation on 8 March 1995 is extremely small and can be considered to be showing no correlation between the variables. However, the explanation to why it is not showing a positive correlation is not clear.

As with CO it might be expected that the association between air quality and vehicle emissions would decline with distance from the road as vehicle emissions are dispersed. However, as with CO no such relationship was found.

The correlations between NO_x air quality and wind speed are predominantly negative **(Table 6.5)** as would be expected and as they are for carbon monoxide. The highest negative correlation (r = -0.79) of NO_x air quality with wind speed was obtained on 21 October 1995 (15 metres from the road - *Figure 6.12C)* with the lowest negative correlation (r = -0.17) being observed on 8 June 1994 (7.5 metres from the road - *Figure 6.12D)*. The latter situation is identical to that for CO.

Analysis of the correlation coefficients for NO_x air quality and wind direction **(Table 6.5)** show a similar number of positive and negative correlations with no distinct pattern between the individual sites. This situation is similar to that found for CO. The initial expectation, for both CO and NO_x, was for positive correlations with wind direction to be found, such that, when the air quality monitors were downwind of the road *(Figure 6.3A)*, a more perpendicular wind direction to the road would lead to increased concentrations as was found for 10 June 1994 (kerbside -*Figure 6.12E)* which gave the highest correlation coefficient (r = 0.81). In situations where the air quality monitors were upwind of the road *(Figure 6.3B)* a wind direction more parallel to the road would lead to increased concentrations as was found for 21 October 1995 (kerbside - *Figure 6.12F)* which gave a correlation coefficient of 0.67.

As with CO, positive correlations between wind direction and air quality were not universally calculated. However, if the regression analysis section (6.4.2.2) is examined on those sampling days where there is a 90% confidence level in a linear relationship, wind direction is consistently shown to have only a positive effect upon ambient concentrations although unlike CO, the regression analysis sample size is very small.

6.4.2.2 Regression analysis

Regression analysis were only conducted on the data from those sampling days where there was a 90% confidence level or above (as marked by an asterisk in **Table 6.6**) that a linear relationship between NO_x air quality and the entire set of independent variables existed.

However, unlike carbon monoxide this significantly reduces the number of valid sampling days, such that, regression analysis can only be performed on data from 19 October 1995 and 21 October 1995. As with CO regression analysis, the data for 20 October 1995 are not included even though the 90% threshold is achieved because a Type 1 error was identified. The null hypothesis should have been accepted because it is apparent from the correlation analysis for 20 October 1995 **(Table 6.5)** that there is a spurious relationship between NO_x air quality and on-road vehicle emissions, thereby invalidating the regression model.

The poorer confidence levels in the regression analysis for NO_x, in comparison with CO, may be explained by the range of emission sources of NO_x in urban areas. This range of sources could mean that any developed relationships between vehicle emissions and NO_x air quality may not be as strong as for CO with consequently lower confidence levels. However, Gaussian plume modelling studies undertaken by *Munday et al. 1989* have shown that non-vehicle emissions sources, with generally significantly greater emission heights and improved dispersion, contribute less significantly to ground level concentrations than do vehicle emission sources. Nevertheless emissions inventory work *(Derwent et al. 1995)* suggests that stationary combustion sources contribute up to 20% of low and medium level emissions in urban areas.

A further explanation could be that the estimated NO_x emissions do not fully reproduce the on-road vehicle emissions and therefore any developed relationship may suffer lower confidence levels.

Table 6.6 Confidence levels for the relationships between NO_x air quality and the independent variables on each monitoring day

Site	Location/distance (metres)	Date	Confidence level
A	kerbside	1 June 1994	84
A	7.5	31 May 1994	68
A	15	27 May 1994	61
B	kerbside	10 June 1994	54
B	kerbside	7 June 1994	76
B	7.5	8 June 1994	40
C	kerbside	8 March 1995	70
C	7.5	8 August 1995	77
C	15	9 August 1995	---------
D	roadside	22 October 1995	67
D	kerbside	19 October 1995	99*
D	7.5	20 October 1995	99*
D	15	21 October 1995	99*

* Data sets used in regression analysis

The regression equations for the relationships between NO_x air quality and mean on-road vehicle emissions, wind speed and wind direction for the different locations and dates at site D are given below:

19 October 1995 (kerbside)($r^2 = 0.77$)

$$NO_x = 285.429A - 1.413B + 1.128C + 55.677 \qquad \text{Eq. 6.11}$$

21 October 1995 (15 metres from the road)($r^2 = 0.80$)

$$NO_x = 1137.291A - 18.295B + 1.132C + 16.80 \qquad \text{Eq. 6.12}$$

Where: NO_x = predicted NO_x air quality (ppb)
A = mean on-road vehicle emissions (%)
B = wind speed (knots)
C = wind direction (degrees from perpendicular)

The predicted and observed NO_x concentrations for 19 October 1995 (kerbside) and 21 October 1995 (15 metres from the road) are shown in *Figures 6.13 and 6.14*, respectively. Good agreement is shown between observed and predicted values for both the kerbside location ($r^2 = 0.77$) and the site located 15 metres from the road ($r^2 = 0.80$). The predicted values for both sites are similar in concentration to the observed values and follow the changes in observed concentrations during the day although the predicted values at kerbside appear to lag the observed values between the period of 13:30 and 14:30.

6.4.2.2.1 Interpretation of beta coefficients

As it is inappropriate to interpret the partial regression coefficients as indicators of the relative importance of the independent variables, beta coefficients have again been calculated **(Table 6.7)**. As with CO (section 6.4.1.2.1) on-road vehicle emissions have a predominant influence on air quality on 21 October 1995. This may be explained by the relatively low mean daily wind speed of 6.3 knots and the consequent slow rate of dispersion. For the 19 October 1995, on-road vehicle emissions have the least influence on air quality. This is again partly explained by wind speed with a much higher mean daily wind speed (12.6 knots) leading to increased dispersion.

If the relative importance of wind direction is considered for the two sampling days it

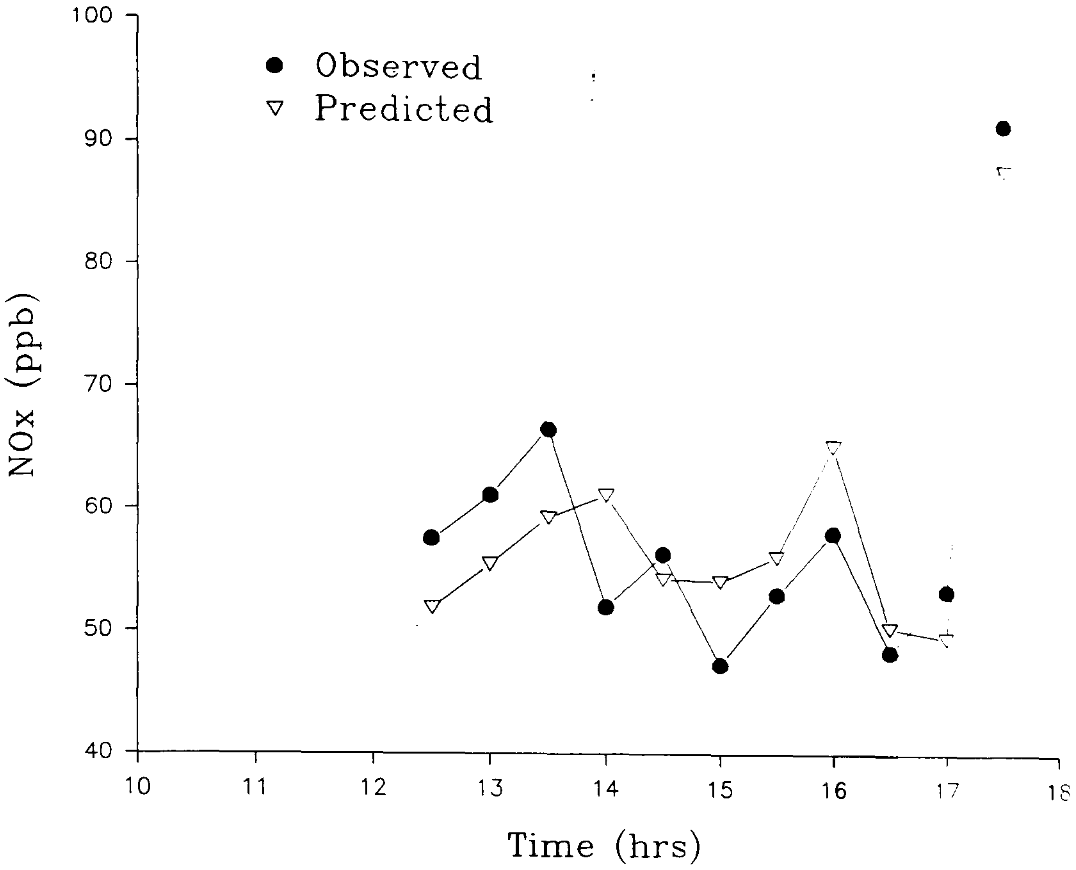


Figure 6.13 Plot of predicted and observed NO_x air quality for 19 October 1995 at site D (kerbside) (r^2 = 0.77)

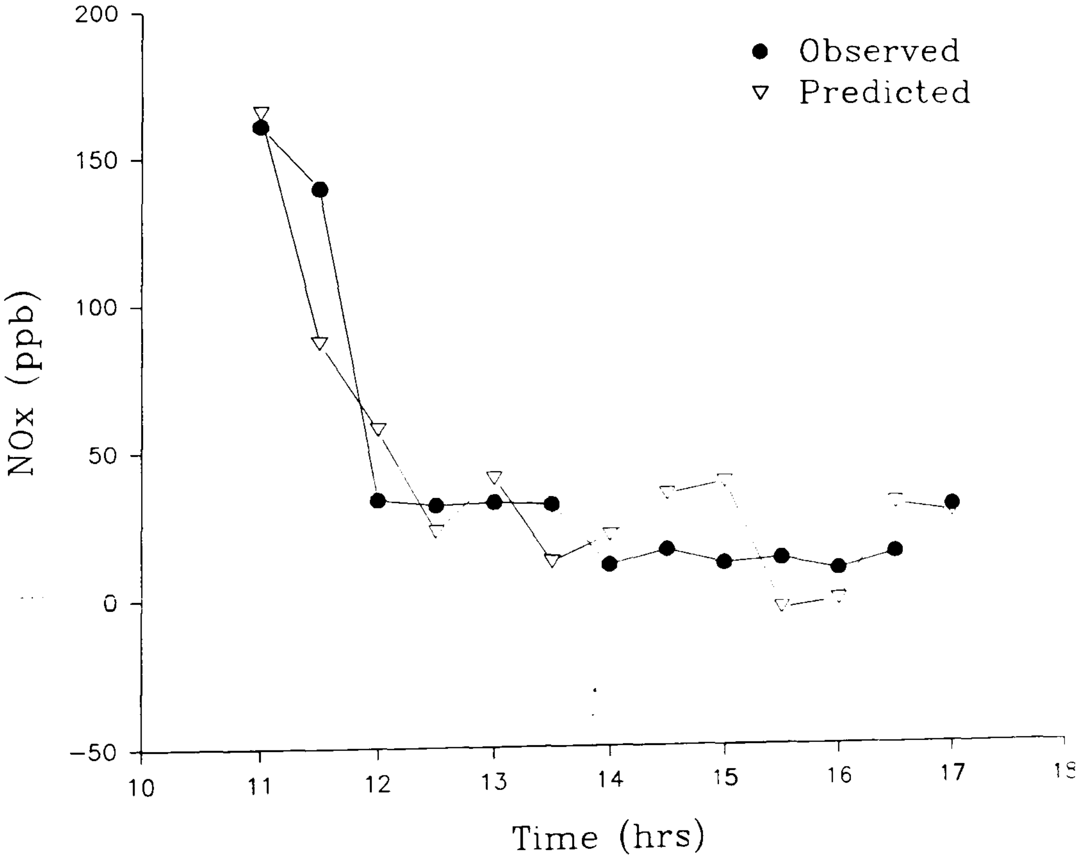


Figure 6.14 Plot of predicted and observed NO_x air quality for 21 October 1995 at site D (15 metres from the road) (r^2 = 0.80)

exerts the most predominant influence on 19 October 1995 and is the least important factor on 21 October 1995. This could be explained by the dominant wind direction on 21 October 1995 blowing from the monitor towards the road whereas on 19 October 1995 the dominant wind direction is more parallel to the road.

Table 6.7 Beta coefficients for independent variables for specific sampling dates

Site	Location/distance (metres)	Date	Vehicle emissions	Wind speed	Wind direction
D	Kerbside	19 October 1995	0.290	-0.470	0.636
D	15	21 October 1995	0.486	-0.324	0.237

No attempt was made to develop a unified regression model to explain NO_x air quality by on-road vehicle emissions, wind speed and wind direction as this would be of little merit with only two sampling days available for analysis.

6.4.3 Five minute statistical analyses for carbon monoxide

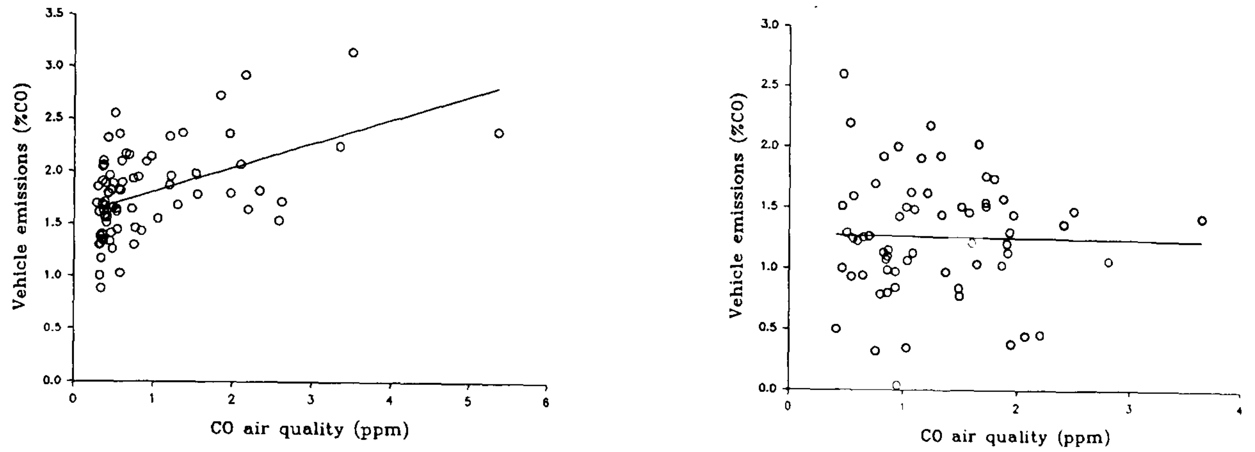
Unlike the half-hourly CO statistical analyses, five minute analyses could only be undertaken between CO air quality and on-road vehicle emissions due to the absence of meteorological office data on a five minute mean basis. Correlation analysis was therefore undertaken to study the variation between these two variables. The data were graphed using scatterplots to give a qualitative assessment of the relationships with only the best and worst relationships recorded in the thesis.

However, the Pearson correlation coefficients have been reported for all sampling days. Attempts were then made to develop mathematical models at the 0.1 significance level, using bivariate linear regression analysis, to explain carbon monoxide air quality in terms of on-road vehicle emissions.

6.4.3.1 Correlation analysis

The correlation coefficients between carbon monoxide air quality and on-road vehicle

CO emissions together with their confidence levels for all sites and locations are displayed in **Table 6.8**. The high degrees of association noted for the half-hourly data is absent for the five minute data with a highest r value of only 0.49 (calculated for 21 October 1995 at site D (15 metres from the road - *Figure 6.15A)*). The lowest r value (-0.01) (calculated for 8 August 1995 at site C (7.5 metres from the road - *Figure 6.15B)*) is more typical of the weak relationships found for the studied sites.



A 21 October 1995 (site D)(r= 0.49) **B** 8 August 1995 (site C)(r = -0.01)

Figure 6.15 Scatter plots of on-road vehicle emissions against CO air quality for 21 October 1995 **(A)** and 8 August 1995 **(B)**

A basic assumption in the analysis is that both carriageways have the same emission characteristics. This assumption may not be completely accurate over a short averaging period such as five minutes. A five minute averaging period may not provide sufficient time to smooth out extreme values and therefore, may not provide a perfect match in emission characteristics between both carriageways. This possible difference in emission characteristics may mean that the remotely sensed emissions do not provide a completely accurate picture of the emission characteristics of the whole road. Therefore, it is conceivable that the difference in emission characteristics between highways, over a five minute averaging period, may be responsible for the reduction in association between vehicle emissions and air quality compared to the half-hourly data.

Table 6.8 Inter-correlation coefficients between carbon monoxide air quality and on-road vehicle emissions at each site.

Site	Location/distance (metres)	Date	Correlation coefficients (Confidence levels)
A	kerbside	1 June 1994	0.12(81)
A	7.5	31 May 1994	-0.08(74)
A	15	27 May 1994	0.17(92)
B	kerbside	10 June 1994	0.09(73)
B	kerbside	7 June 1994	0.05(61)
B	7.5	8 June 1994	-0.02(56)
C	kerbside	8 March 1995	0.24(98)
C	7.5	8 August 1995	-0.01(53)
C	15	9 August 1995	0.05(67)
D	roadside	22 October 1995	0.06(68)
D	kerbside	19 October 1995	0.19(94)
D	7.5	20 October 1995	-0.05(66)
D	15	21 October 1995	0.49(99)

6.4.3.2 Regression analysis

As with the regression analyses applied to the CO half-hourly averaged data (section 6.4.1.2) only those sampling days where it was certain at a 90% level (**Table 6.9**) that the relationships found were not through random chance have been included in the regression analysis. The 90% significance level was applied to the regression analyses and not the correlation analyses because the former was extending the statistical interpretation and therefore, a higher level of confidence in the data was necessary. Regression analysis was therefore conducted on 8 March 1995 and 21 October 1995.

Table 6.9 Confidence levels for the relationship between CO air quality and on-road vehicle emissions on each monitoring day

Site	Location/distance (metres)	Date	Confidence level
A	kerbside	1 June 1994	62
A	7.5	31 May 1994	47
A	15	27 May 1994	87
B	kerbside	10 June 1994	46
B	kerbside	7 June 1994	21
B	7.5	8 June 1994	12
C	kerbside	8 March 1995	96
C	7.5	8 August 1995	7
C	15	9 August 1995	35
D	roadside	22 October 1995	36
D	kerbside	19 October 1995	87
D	7.5	20 October 1995	32
D	15	21 October 1995	99

The regression equations for the relationship between mean on-road emissions and carbon monoxide air quality for the different sites and dates are given below:

Site C; 8 March 1995 (kerbside) (r^2 = 0.06)

$$CO = 2.083 + 0.455A \qquad \text{Eq. 6.14}$$

Site D; 21 October 1995 (15 metres from the road) (r^2 = 0.24)

$$CO = -0.949 + 1.042A \qquad \text{Eq. 6.13}$$

Where: CO = predicted carbon monoxide air quality (ppm)

A = mean on-road vehicle emissions

The predicted and observed CO concentrations for 8 March 1995 (kerbside) and 21 October 1995 (15 metres from the road) are shown in *Figures 6.16 and 6.17*, respectively. The agreement between observed and predicted concentrations is not good for either site, with the kerbside location (8 March 1995) having an r^2 of only 0.06. The site located 15 metres from the road has a somewhat closer agreement between predicted and observed values with an r^2 of 0.24.

6.4.4 Five minute statistical analyses for nitrogen oxides

As for the five minute CO statistical analyses, the five minute NO_x statistical analyses could only be undertaken between NO_x air quality and on-road vehicle emissions due to the absence of meteorological office data on a five minute mean basis.

Correlation analysis was therefore undertaken to study the variation between the two variables. The data were graphed using scatterplots to give a qualitative assessment of the relationships with only the best and worst relationships being recorded in the thesis. However, the Pearson correlation coefficients have been reported for all sampling days. Attempts were then made to develop mathematical models, using bivariate linear regression analysis, to explain NO_x air quality in terms of on-road vehicle emissions.

6.4.4.1 Correlation analysis

The low degree of association noted between emissions and air quality for CO is also apparent for NO_x **(Table 6.10)** with the best correlation between emissions and air quality ($r = 0.31$) occurring on 21 October 1995 at site D *(Figure 6.18A)* as was the case for CO. The weakest correlation ($r = -0.003$) was calculated for 8 August 1995 at site C (7.5 metres from the road - *Figure 6.18B*). A possible explanation for the lack of association between the variables was given in section 6.4.3.1 but to summarise, the reduction in association could be caused by the difference in emission characteristics between the measured and unmeasured carriageways.

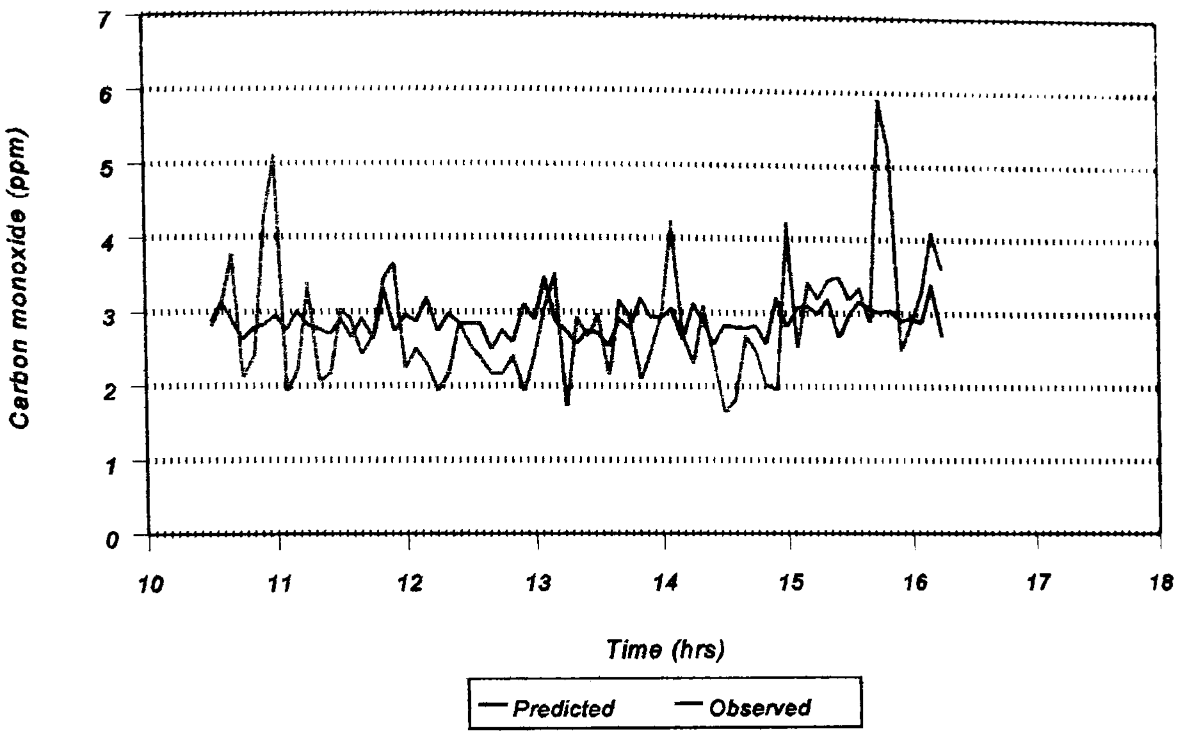


Figure 6.16 Plot of predicted and observed carbon monoxide air quality for 8 March 1995 at site C ($r^2 = 0.06$).

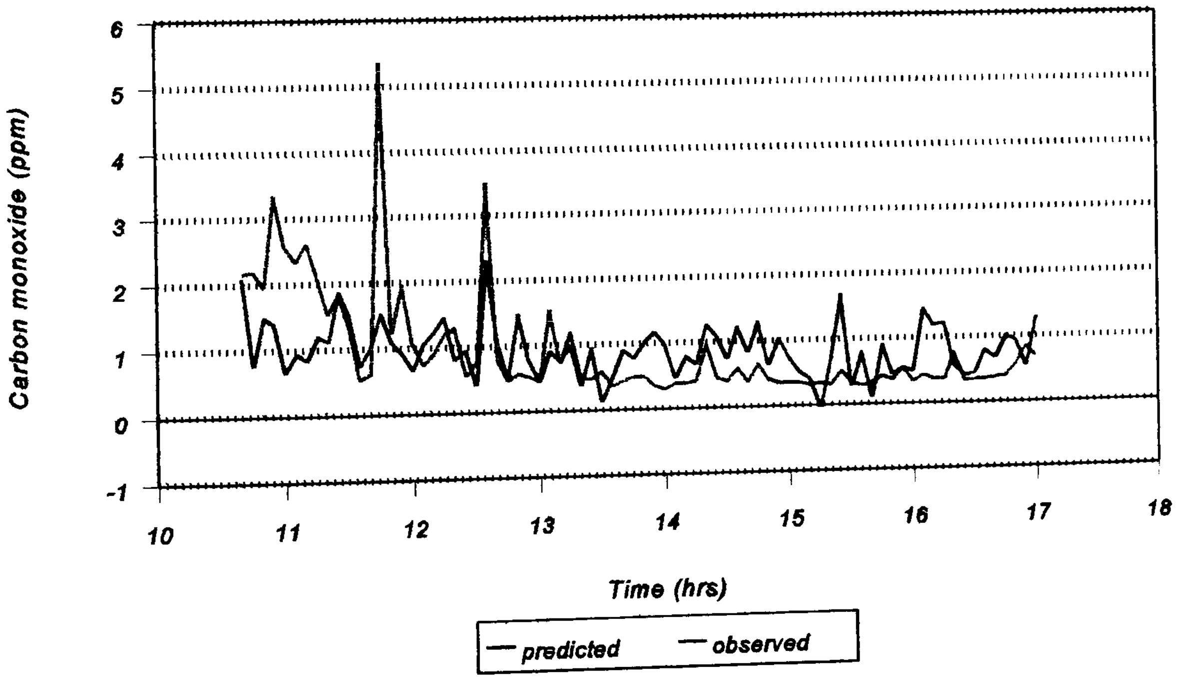
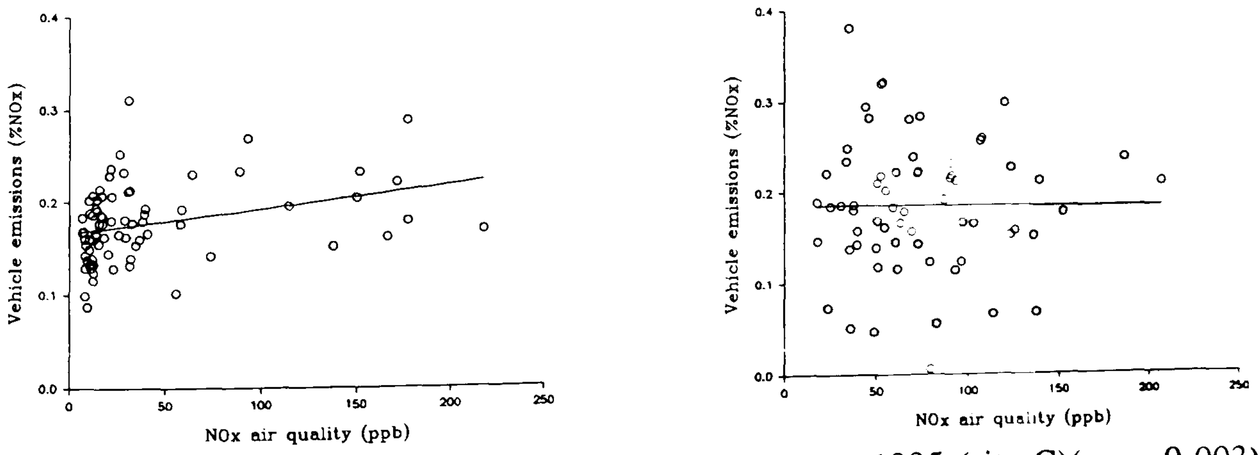


Figure 6.17 Plot of predicted and observed carbon monoxide air quality for 21 October 1995 at site D ($r^2 = 0.24$).

Table 6.10 Inter-correlation coefficients between NO_x air quality and on-road vehicle emissions at each site.

Site	Location/distance (metres)	Date	Correlation coefficients (Confid.limits)
A	kerbside	1 June 1994	-0.04(61)
A	7.5	31 May 1994	-0.15(88)
A	15	27 May 1994	0.14(88)
B	kerbside	10 June 1994	-0.17(86)
B	kerbside	7 June 1994	-0.03(55)
B	7.5	8 June 1994	-0.15(87)
C	kerbside	8 March 1995	-0.09(77)
C	7.5	8 August 1995	-0.003(51)
C	15	9 August 1995	------------
D	roadside	22 October 1995	-0.10(79)
D	kerbside	19 October 1995	0.02(56)
D	7.5	20 October 1995	-0.25(98)
D	15	21 October 1995	0.31(99)



A 21 October 1995 (site D)(r = 0.31)

B 8 August 1995 (site C)(r = -0.003)

Figure 6.18 Scatter plots of on-road vehicle emissions against NO_x air quality for 21 October 1995 (**A**) and 8 August 1995 (**B**)

6.4.4.2 Regression analysis

As with the regression analysis applied to the NO_x half-hourly averaged data (section 6.4.2.2) only those sampling days where it was certain, at a 90% level or above (**Table 6.11**), that the relationships found were not through random chance alone have been included in the regression analysis.

Unfortunately this reduced the sample to only 21 October 1995. The data for 20 October 1995 are not included in the analysis because it is apparent from the correlation analysis for 20 October 1995 (**Table 6.10**) that there is a spurious relationship between air quality and on-road vehicle emissions.

Table 6.11 Confidence levels for the relationship between NO_x air quality and on-road vehicle emissions on each monitoring day

Site	Location/distance (metres)	Date	Confidence level
A	kerbside	1 June 1994	22
A	7.5	31 May 1994	76
A	15	27 May 1994	76
B	kerbside	10 June 1994	72
B	kerbside	7 June 1994	10
B	7.5	8 June 1994	73
C	kerbside	8 March 1995	54
C	7.5	8 August 1995	2
C	15	9 August 1995	--
D	roadside	22 October 1995	57
D	kerbside	19 October 1995	11
D	7.5	20 October 1995	97
D	15	21 October 1995	99

The regression equation for the relationship between mean on-road emissions and NO_x air quality is given below:

Site D; 21 October 1995 (15 metres from the road)($r^2 = 0.10$)

$$NO_x = -25.184 + 370.277A \qquad \text{Eq. 6.15}$$

Where: NO_x = predicted NO_x air quality (ppb)
A = mean on-road vehicle emissions

The predicted (derived from equation 6.15) and observed NO_x concentrations for 21 October 1995 (15 metres form the road) are shown in *Figure 6.19*. There is little agreement between the predicted and observed concentrations ($r^2 = 0.10$) with predicted values failing to mirror the changes in observed concentrations. A further pronounced difference is the gulf between predicted and observed concentrations during the period 10:30 to 11:15 when predicted values were significantly lower than the observed concentrations.

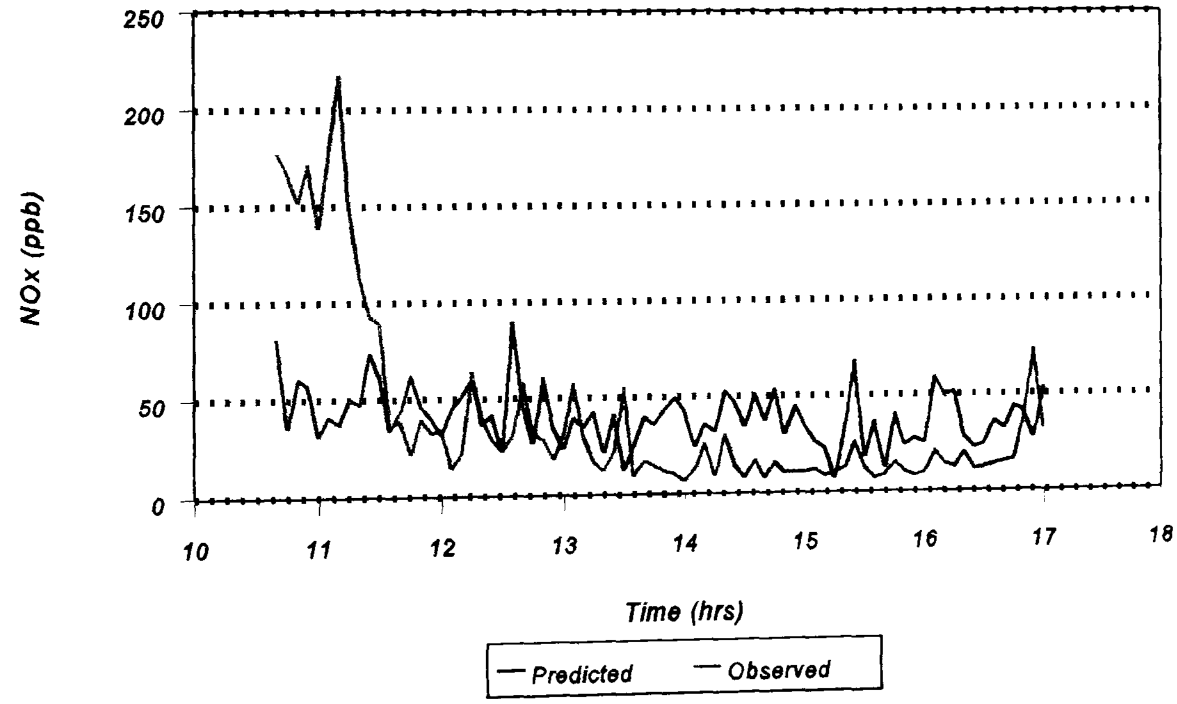


Figure 6.19 Plot of predicted and observed NO_x air quality for 21 October 1995 at site D (r^2 0.10).

6.5 Summary

For the half-hourly analyses, on-road vehicle emissions were found to have positive correlations with carbon monoxide and nitrogen oxides air quality, such that when on-road vehicle emissions increased so did ambient concentrations. The correlations of CO and NO_x air quality with wind speed were found to be predominantly negative. This was expected because at higher wind speeds there should be greater dispersion and therefore, lower pollutant concentrations. The correlations between CO and NO_x air quality and wind direction do not follow either pattern set for vehicle emissions or wind speed with both negative and positive correlations in almost equal measure.

Regression analysis was only undertaken on those monitoring days where the results were statistically significant at a 0.1 level. This reduced the sample size for CO to eight days and to only two days for NO_x. All the analysed CO sampling days recorded r^2 values of greater than 0.5, such that for each sampling day at least half the variation in CO air quality is explained by the variation in, on-road vehicle emissions, wind speed and wind direction. Supplementary analysis of the beta coefficients on a daily basis for CO shows wind speed to be the most important influence on air quality followed in importance by wind direction with vehicle emissions having the least influence.

The analysed NO_x sampling days recorded r^2 values of approximately 0.8. However, it is important to note that only two days were analysed and for this reason no attempt was made to identify trends in the beta coefficients.

An objective of the research was to develop regression models to explain CO and NO_x air quality by on-road vehicle emissions, wind speed and wind direction and be able to apply this model for any location. In order for this to be possible it was necessary for similar relationships to have been found on the different sampling days. However, it is clear for CO from equations 6.3 - 6.10 that there is little agreement between the different sampling days. Moreover, for NO_x, no attempt was made to develop a unifying regression model as this would be of little merit with only two sampling days available for analysis.

The high degree of association between CO and NO_x air quality and emissions noted for the half-hourly data is absent for the five minute data. Regression analysis was undertaken on those monitoring days where the results were statistically significant at 0.1 level. This reduced the sample size for CO to two days and to one day for NO_x returning r^2 values of no greater than 0.24.

CHAPTER 7. CONCLUSIONS AND RECOMMENDATIONS

7.1 Introduction

This chapter summarises the main findings of the research and lists possible avenues for further research. The research project had three major objectives, the first of which was to provide detailed information of the on-road emission characteristics of individual fleets in the UK. This was covered in Chapter 4 and is reviewed in section 7.2. The second major objective, the results of which are summarised in section 7.3, was to monitor the concentrations of carbon monoxide and nitrogen oxides at roadside and to assess how these concentrations change with distance from the road. The final objective was to investigate the relationship between remotely sensed on-road vehicle emissions (see section 7.4) and roadside air quality with the associated development of a model to explain air quality in terms of on-road vehicle emissions, wind speed and wind direction.

7.2 On-road vehicle emissions

The emission data presented in Chapter 4 for carbon monoxide and hydrocarbons clearly indicate that there is a large majority of low emitting vehicles contributing little to fleet emissions at all sites and for both pollutants. The converse is also true with a small minority of high emitting vehicles contributing significant proportions to fleet emissions at all sites and for both pollutants. However, the small minority of high emitting hydrocarbon vehicles in general contribute an even greater proportion to fleet emissions than do the comparable carbon monoxide high emitters.

A pattern of high mean emission values and lower median values is consistently observed for both pollutants at all locations. This occurs because mean emissions are dominated by a small number of high emitters which skew the data to the right (positive skew). Thus, the median which is not biased by the high emitters returns a much lower value. Work conducted with remote sensing data sets and laboratory dynamometer emission measurements by *Zhang, Bishop and Stedman (1994)* showed a similar result, with carbon monoxide and hydrocarbon emissions well represented by a γ-distribution.

Analysis of model year emissions data for both carbon monoxide and hydrocarbons suggests a low association between vehicle age and mean emissions for vehicles produced prior to 1983, at all sites. The relationship improves after 1983 with regression analyses returning average r^2 values of 0.89 for carbon monoxide and 0.75 for hydrocarbons.

Analysis of model year contribution to fleet emissions shows gross polluting vehicles to be the greatest contributors to fleet emissions with on average 43% of carbon monoxide and 52% of hydrocarbon fleet emissions being produced by the gross polluters in model years 1985-1991. Fleet emissions could therefore be significantly reduced if these gross polluting vehicles received regular servicing or were removed from the highway by the withdrawal of their vehicle registration. The introduction of three way catalyst technology to all new cars since 1993 has further accentuated the effect gross polluters have on fleet emissions. This has occurred because the well maintained low emitting vehicles have had their emissions lowered further and the gross polluting vehicles, which are now essentially only those vehicles where catalyst technology has failed, have had their emissions increase; emissions have increased because new cars without a catalyst have practically uncontrolled emissions. Therefore, the ability to reduce fleet emissions by targeting gross polluters in the new catalyst equipped fleet is even greater. Older vehicles play a minor role in overall fleet emissions with on average only 14% of carbon monoxide and 13% of hydrocarbon fleet emissions produced by vehicles registered prior to 1983.

7.3 Air quality

A decrease of carbon monoxide and nitric oxide concentrations with distance from the road was noted for all sites with the exception of site D. The reduction in carbon monoxide concentrations was most marked for sites A and C, with similar size decreases in ambient concentrations noted at both locations. Site B had only a slight decrease in carbon monoxide concentrations with distance from the road; no carbon monoxide data was available from the site positioned 15 metres from the road. Site D did not display any reduction in ambient levels of carbon monoxide or nitric oxide as meteorological parameters exerted greater influence upon air quality than did distance from the road at this site.

A decrease of nitrogen dioxide concentrations with distance from the road was noted for sites A, B and C although this was unexpected. The reduction was most distinct for site A, followed by site B. The reduction in concentration, however, is generally much less than was noted for CO and NO. Site C is the best example of this with little difference in concentration between the kerbside measurement site and the site positioned 7.5 metres from the road; no data was available for the site positioned 15 metres from the road. Site D did not display any pattern in ambient levels with increasing distance from the road.

The initial expectation was for nitrogen dioxide concentrations to increase at greater distances from the road as the emitted nitric oxide was oxidised to nitrogen dioxide. However, it appears that, over the relatively short distances from the road monitored in these field studies, the production of nitrogen dioxide as a primary pollutant by motor vehicles was more important than from the oxidation of nitric oxide. Therefore, any production of nitrogen dioxide due to oxidation was being masked by the large amounts produced by vehicle emissions. However, the production of nitrogen dioxide by oxidation can be inferred in two ways. The first, is the much more gradual decline in NO_2 concentrations away from the road when compared to CO and NO concentrations which is probably caused by the production of nitrogen dioxide by oxidation counteracting the reduction in concentration caused by dispersion. In addition, an analysis of the change of the ratio between nitrogen dioxide and nitric oxide with distance from the road reveals an increase of NO_2 relative to NO with distance, presumably caused by the oxidation of NO to NO_2.

Carbon monoxide air quality was worst at the two London sites (A and C) with the two highest daily mean levels of 2.34 ppm and 2.89 ppm respectively, being recorded. Nitric oxide concentrations were highest at site A with a peak daily mean concentrations of 207 ppb. The air quality was best at site B with a maximum carbon monoxide mean daily value of only 0.42 ppm and a maximum nitric oxide mean daily value of only 45 ppb. Site D generated a maximum carbon monoxide daily value of 1.51 ppm and site C a maximum nitric oxide daily value of 86 ppb.

Comparison of the maximum carbon monoxide one hour concentrations from the respective sites (site A - 2.90 ppm; site B - 0.83 ppm; site C - 3.50 ppm; site D - 2.51 ppm) with the WHO one hour exposure guideline (25 ppm) indicates that the air quality at all sites is very good.

As with nitric oxide the worst nitrogen dioxide air quality was found at sites A and D with peak daily mean recorded concentrations of 64 ppb and 32 ppb, respectively. The lowest concentrations were found at site B with a peak daily mean value of 15 ppb. Site C generated a maximum nitrogen dioxide daily value of 29 ppb. The nitrogen dioxide air quality can be described as `very good' with the maximum one hour concentrations from the respective sites (site A - 73 ppb; site B - 16 ppb; site C - 31 ppb; site D - 57 ppb) considerably below the WHO one hour exposure guideline of 209 ppb.

7.4 Modelling

Chapter 6 was an investigation of the relationship between remotely sensed on-road vehicle emissions, wind speed and wind direction with roadside air quality. On-road vehicle emissions were found to have positive correlations with carbon monoxide and nitrogen oxides air quality for the half-hourly analyses, such that when on-road vehicle emissions increased so did ambient concentrations. The correlations of CO and NO_x air quality with wind speed were found to be predominantly negative. This was expected because at higher wind speeds there should be greater dispersion and therefore, lower pollutant concentrations. The correlations between CO and NO_x air quality and wind direction do not follow either pattern set for vehicle emissions or wind speed with both negative and positive correlations being found in almost equal measure.

Regression analysis was only undertaken on those monitoring days where the results were statistically significant at a 0.1 level. This reduced the sample size for CO to eight days and to only two days for NO_x. All the analysed CO sampling days recorded r^2 values of greater than 0.5, such that for each sampling day at least half the variation in CO air quality is explained by the variation in, on-road vehicle emissions, wind speed and wind direction. Supplementary analysis of the beta coefficients on a daily basis for

CO shows wind speed to be the most important influence on air quality followed in importance by wind direction with vehicle emissions having the least influence.

The analysed NO_x sampling days recorded r^2 values of approximately 0.8. However, it is important to note that only two days were analysed and for this reason no attempt was made to identify trends in the beta coefficients.

An objective of the research was to develop regression models to explain CO and NO_x air quality with respect to on-road vehicle emissions, wind speed and wind direction and be able to apply this model for any location. In order for this to be possible it was necessary for similar relationships to have been found on the different sampling days which was not the case for CO. Moreover, for NO_x, no attempt was made to develop a unifying regression model as this would be of little merit with only two sampling days available for analysis.

The high degree of association between CO and NO_x air quality and emissions noted for the half-hourly data is absent for the five minute data. Regression analysis was undertaken on those monitoring days where the results were statistically significant at 0.1 level. This reduced the sample size for CO to two days and to one day for NO_x, with calculated r^2 values of no greater than 0.24.

7.5 Recommendations for further research

This research provided information for the on-road emission characteristics of CO and HC in the UK. At present there is limited information available relating to the on-road emission characteristics of NO_x and smoke but due to the development of remote sensing emissions channels for these pollutants valuable data could now be obtained through an emissions monitoring programme. This would also enable direct comparison of on-road NO_x emissions with NO_x air quality data.

The monitoring programme undertaken in this study concentrated on sites with low vehicle speeds with little emissions information being obtained from vehicles at higher speeds. A site targeted at high speed vehicles would provide valuable results on the change in emissions with speed and how these emissions affect air quality.

The sites employed in the research were designed to monitor vehicles in a cruising mode. A development of the research would be the selection of sites in which emissions were determined from vehicles either accelerating or decelerating to assess the impact which change in operating mode has on air quality. A further refinement of any future sampling programme would be the use of pneumatic tubes on the road surface, thereby enabling speed and operating mode of individual vehicles to be recorded and related to emissions.

A possible problem identified in the research was the difference in emissions characteristics between the opposite lanes of a highway. Therefore any subsequent work should seek to eliminate this problem by using two FEAT systems to monitor both lanes of a highway, thereby providing a clearer picture of on-road emissions.

Part IV of the Environment Act 1995 gave local authorities a duty to carry out regular reviews and assessments of air quality and to target areas which are likely to fail air quality standards. These 'hot-spots' of poor air quality are then declared Air Quality Management Areas (AQMAs). The FEAT system could be used by local authorities to identify specific sections of roads which are likely to produce high emissions and therefore 'hot-spots' of poor air quality. The FEAT unit could be used as a screening device to prevent high emitting vehicles using a section of road during periods of poor air quality, thereby ensuring that air quality standards are not exceeded.

During the air quality monitoring programmes pollutant concentrations were determined at different distances from the road but this was accomplished by using one monitor on different sampling days. If three air quality monitors were available then real time continuous comparison at different distances from the road could be undertaken. This would enable an examination of the change of concentrations with distance during one individual sampling day and prevent the situation encountered at site D where concentrations appear to increase with increasing distance from the road.

Recent research undertaken by *Moseholm et al. 1996* has used neural networks to predict roadside carbon monoxide concentrations (1 min average) from 1 minute average traffic and wind characteristics with reasonable accuracy ($r^2 = 0.69$). This technique could therefore be used with the on-road emissions data to improve modelling performance. Further improvement in modelling performance could be obtained by using meteorological data from an on-site weather station in place of regional data supplied by the meteorological office.

Appendix A: FEAT combustion equations

A.1 Introduction

The principles and validity of the following equations is not dependent on, or unique to, the FEAT equipment. Emissions data supplied by the Chrysler Corporation have been analysed using the FEAT equations *(Stedman et al. 1991c)* and then compared to the original data. The agreement between FEAT equation results and observed data proved to be excellent.

A.2 Determination of percentage gases in the exhaust

In the following equations all ratios are in molar units. The CO/CO_2 ratio by moles is expressed as Q. Moles are also directly proportional to volumes at constant temperature and pressure; therefore, emission percentages derived from Q are by volume.

An automobile can be considered as a device in which fuel containing carbon and hydrogen (formula CH_n) is burned with air (whose approximate composition is $0.21\ O_2 + 0.79\ N_2$) in a combustion chamber to derive power. The products are sometimes further burned on a catalyst or in the exhaust system. However, all combustion processes (power or catalytic) are governed by the same equation and its derivatives:

$$CH_n + m(0.21O_2 + 0.79N_2) \rightarrow \frac{n}{2}(H_2O) + aCO + bCO_2 + 0.79mN_2 \tag{1}$$

From equation 1, the following relationships are obtained:

$$a + b = 1 \tag{2}$$

(by carbon balance)

$$0.42m = \frac{n}{2} + a + 2b \tag{3}$$

(by oxygen balance)

$$Q = \frac{a}{b} \tag{4}$$

(by remote sensing)

When a and b are determined in terms of Q, the following equations are obtained:

$$a = \frac{Q}{1+Q} \tag{5}$$

$$b = \frac{1}{1+Q} \tag{6}$$

Thus, with a knowledge of n and Q, the molar coefficients in the balanced combustion equation are readily determined; these quantities can in turn be used to obtain other derived quantities. If CH_2 is used for the empirical formula of the fuel and Q is given a typical fleet value of 0.1, then the molar coefficients can be determined as follows:

$$a = \frac{Q}{1+Q} = \frac{0.1}{1+0.1} = 0.090909 \tag{7}$$

$$b = \frac{1}{1+Q} = \frac{1}{1+0.1} = 0.9090909 \tag{8}$$

$$m = \frac{1 + a + 2b}{0.42} = \frac{1 + 0.091 + (2*0.909)}{0.42} = 6.926 \tag{9}$$

The molar coefficients are then multiplied by 22.414 l (the volume 1 mole of an ideal gas occupies at 1 atm) to obtain the volume the emitted gas occupies in the exhaust:

$$CO = 0.091 * 22.414 = 2.0376l \tag{10}$$

$$CO_2 = 0.909 * 22.414 = 20.3763l \tag{11}$$

$$N_2 = (0.79 * 6.926) * 22.414 = 122.6391 \tag{12}$$

Therefore, individual component emission percentages can be derived from a knowledge of the total volume of emitted gases (145.053 l).

$$\%CO = \frac{2.0376}{145.053} * 100 = 1.40 \tag{13}$$

$$\%CO_2 = \frac{20.3763}{145.053} * 100 = 14.05 \tag{14}$$

$$\%N_2 = \frac{122.639}{145.053} * 100 = 84.55 \tag{15}$$

These equations are approximations and assume dried and undiluted exhaust. If these values are to be compared to conventional tailpipe measurements in which excess air is present, the tailpipe %CO values need to be adjusted upwards by a dilution factor of $21/(21 - \%O_2)$.

A.3 Determination of grams/gallon in CO emissions

The ratio of CO to ($CO + CO_2$) is a measure of CO emissions in moles of CO per mole of carbon in the fuel (assuming HC emissions to be negligible in comparison), thus:

$$\frac{CO}{(CO + CO_2)} = \frac{a}{(a + b)} = a \tag{16}$$

From Equation 5:

$$a = \frac{Q}{(1 + Q)} \tag{17}$$

For conversion of moles of CO to grams, this term is multiplied by the formula weight of CO, 28 g/mole. For conversion of moles of carbon in the fuel to gallons of fuel, the approximate fuel formula $CH_{1.85}$ (13.85 gm/mole) is assumed. The density of petrol is approximately 700 gm/litre, and there are 4.546 litres per gallon. A value of 0.1 has been assumed for Q.

Incorporation of these values and assumptions into equation 17 yields:

$$CO\ g/gallon = (\frac{4.546*700}{13.85}) * ((\frac{0.1}{1 + 0.1})* 28) = 584.85 \tag{18}$$

This equation is an approximation and neglects any contribution from exhaust HC.

If mass emissions in g/mile are required then g/gallon must be converted to g/mile by means of petrol mileage data. If we assume an average petrol mileage of 17 mpg, and a Stedman emission of 585 g/CO/gallon the equivalent CO g/mile value would be 34.

Values for hydrocarbons have not been derived but can be obtained using a similar set of combustion equations.

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