IMPERIAL COLLEGE LONDON
(University of London)

Faculty of Life Sciences

Centre for Environmental Policy

PARTICULATE MATTER EMISSION FROM PAVED ROAD SURFACES

By

Aditya Kumar Patra

A thesis submitted for the degree of Doctor of Philosophy of the University of London and for the Diploma of Imperial College London

January 2006

To my parents

DECLARATION OF OWN WORK

I declare that this thesis

PARTICULATE MATTER EMISSION FROM PAVED ROAD SURFACES

is entirely my own work and that where any material could be construed as the work of others, it is fully cited and referenced, and/or with appropriate acknowledgement given.

Signature of student: 

Name of student: Aditya Kumar Patra

Name of supervisor: Dr. Roy Norman Colvile

Abstract

The thesis presents the outcome of the research aimed at enhancing our understanding of the action of traffic-induced material movement processes that result in temporal and spatial variability of concentrations of particulate matter in the roadside environment, through a combination of measurement and modelling. A field experiment, that involved gritting a segment of road and then quantifying the movement of the grit along and across the road and to the air, is conducted at Gloucester Place in Central London. The modelling process consists of having an inventory of the important processes that contribute to particulate matter fluxes to and from the road surface, mathematical formulation of these processes and numerical integration of the equations, using Stella software. Measurements at Manor Road, where the material carry-over and subsequent deposition on road surface by the vehicles from a nearby waste transfer station is suspected to be the reason for non-compliance of the EU PM_{10} limit value, are used to evaluate the transferability of the model.

The field results show that, on a paved road with no major local source, road abrasion is the largest source of surface loading, followed closely by material carried over from the upstream road segment. With a single vehicle pass the average amount of material removed from a road segment in different directions are as follows: 0.04 % along the road, 0.005 % across the road and 0.03% into the air. The silt loading is the result of the interplay of replenishing and depleting fluxes and is therefore dynamic in nature.

The experiment and modelling exercise confirmed that a model for paved road particulate matter emission needs to be two-dimensional and dynamic, including material build-up and removal processes on the road surface and their characteristic times. It is proposed that this will not only increase the physical realism of the model but also reduce the gap between observation and model prediction, two main drawbacks of the empirical models.

Acknowledgements

I acknowledge Dr. Roy Colvile for his thorough supervision throughout the durations of PhD. Since my initial contact with him from India, which dates back to July 2000, he has been a constant support to me in every moment of need, both personal and academic. I appreciate his approach of ensuring enough freedom to have my thought ingrained to my work. I acknowledge Prof. Helen ApSimon for her vital inputs all along the course of PhD. I acknowledge Prof. Sevket Durucan for presiding over my review meetings and advising me on research presentation style.

I am grateful to the Greater London Authority for funding the field experiment, the Westminster City Council Department of Highways Management and Cleansing Management, Steve Neville of the Westminster City Council Department of Environmental Health and Ms Emma Bowen of Imperial College for the gritting event. I also thank the DAPPLE consortium, a multidisciplinary research group from six UK universities, and especially Dr Samanth Arnold, the DAPPLE field-work coordinator at Imperial College, for their contribution in field study and data analysis. My heartfelt thanks go to Jon Fox of Bexley Council, London for his prompt response to all my queries and his unreserved support in providing me the required data.

I gratefully acknowledge the Commonwealth Scholarships Commission for the financial support and the Council of Scientific and Industrial Research, my employer in India, for granting necessary leave to pursue research here. Many thanks to Imperial College London for providing me facilities to carry out the research.

Many thanks to friends in the research group (Fernando, Hongbin, Tom, Tim, Jorge, Surbjit, Martyn, Keith, Sirinath, James, Marios, ...) and Mrs Jennifer Eastwood with whom I shared not only the academic matter, but also many lighter moments.

My wife Minakshi, who has been with me throughout the duration of PhD and my family at India, especially my father who expired mid-way through my research, were the constant support all along the duration of PhD. Last, but not the least, I am grateful to Barnet Overseas Students Housing Association for providing shelter and friendship to me and my family during our stay at London.

Contents

List of Figures

List of Tables

Glossary of terms and abbreviations

Term	Description
Accuracy	A statistical method for measuring how well a set of data fits the true value.
ADB	Asian Development Bank.
ADT	Average Daily Traffic.
ADMS	Atmospheric Dispersion Modelling System. Computer modelling program produced by Cambridge Environmental Research Consultants Ltd, UK.
AH	Ampere Hour.
Air Quality Objective	Policy targets generally expressed as a maximum ambient concentration to be achieved, either without exception or with a permitted number of exceedances within a specified timescale.
Air Quality Standard	The concentration of a pollutant, and associated averaging period, which is without significant effect on human health at a population level.
Al	Aluminium.
Al_2O_3	Aluminium Oxide.
Ambient air	Outdoor air in the troposphere, excluding workplace air.
APEG	Airborne Particles Expert Group, UK.
AQDD	Air Quality Daughter Directive.
AQEG	Air Quality Expert Group, UK.

AQFD	Air Quality Framework Directive.
AQMA	Air Quality Management Area; an area which a local authority has designated for action, based upon predicted exceedances of Air Quality Objectives.
AQS	Air Quality Strategy.
ASTM	The American Society of Testing Materials.
Atmospheric Dispersion Models	A mathematical, often computer based, method for calculating pollutant concentrations from emissions data and specified meteorological conditions.
AURN	Automatic Urban and Rural Network of air pollution measurement sites, managed by contractors on behalf of DEFRA and the devolved administrators.
BADC	British Atmospheric Data Centre.
Brownian motion	Constant small movement of suspended particles due to bombardment by surrounding particles.
BST	British Summer Time.
Ca	Calcium.
CAFE	Clean Air for Europe.
CARB	California Air Resources Board.
Carcinogenic	Known or believed to cause cancer in human beings.
CASAC	Clean Air Scientific Advisory Committee, EPA, USA.
Cd	Cadmium.
CEPMEIP	Co-ordinated European Programme on Particulate Matter Emission Inventories, Projections and Guidance.
CERC	Cambridge Environmental Research Consultants, UK.
Cl^-	Chloride.
CLEA	Contaminated Land Exposure Assessment.
CMB	Chemical Mass Balance.

CNG	Compressed Natural Gas.
CO	Carbon monoxide; A gaseous pollution formed during incomplete combustion of a hydrocarbon fuel. Lethal at high concentrations.
CO_2	Carbon dioxide.
Cohort study	Study in which a group or cohort of people are followed over time to see whether they develop a disease in response to exposure to the factor of interest.
COMEAP	Committee on the Medical Effects of the Air Pollutants, Department of Health, UK.
Concentration	The amount of a (polluting) substance in a volume (of air), typically expressed as a mass of pollutant per unit volume of air at a standard conditions of temperature and pressure (e.g., micrograms per cubic metre) or as the ratio of the number of molecules of the pollutant to the total number of molecules in the volume of air (e.g., parts per billion, parts per million).
Cu	Copper.
CWS	Canada-Wide Standards.
DAPPLE	Dispersion of Air Pollution and Penetration into the Local Environment, an EPSRC funded project comprising of multidisciplinary research groups from six U.K. universities.
DETR	Department of the Environment, Transport and the Regions, UK.
DEFRA	Department of the Environment, Food and Rural Affairs, UK.
Dispersion coefficient	Dispersion relation between emission from a section of a road and concentration at the receptor.

EC	Elemental Carbon; European Commission.
ECD	Electron Capture Detection.
EEA	European Environment Agency.
EGTS	Erith Group Transfer Station.
Emission factor	The amount of a (polluting) substance emitted in a certain amount of time, typically expressed as a mass of pollutant emitted per unit time (e.g., grams per second, tonnes per year) or per unit area of an urban area (e.g., tonnes per annum per square kilometre). For traffic emissions, it is usually expressed as mass emitted per unit length of a road by a single vehicle (e.g., grams per kilometre per vehicle).
Emissions Inventory	A quantification and compilation of emission sources by geography and time, usually including data covering one or several years.
Emission flux	Mass emitted from unit area of road per unit time.
EPAQS	Expert Panel on Air Quality Standards, UK.
Epidemiology	Statistical studies of health effects in populations (DEFRA, 2000); A branch of medicine that deals with incidence and control of disease in a population; The study of the distribution and determinants of health and disease in populations (AQEG, 2004).
EPSRC	Engineering and Physical Sciences Research Council, UK.
ERG	Environmental Research Group at King's College London, UK.
EU	European Union.

Euro I, II, III, IV & V	Progressively stricter Europe-wide vehicle emissions standards requiring vehicles manufactured after 1992, 1996, 2000, 2006 and 2008 respectively to achieve set emissions limits.
Exceedance	A period of time where the concentration of a pollutant is greater than the appropriate air quality objective.
Fe	Iron.
FID	Flame Ionisation Detection.
Fugitive emission	Emissions that occur directly from a process or source into the workplace or outside atmosphere without first being collected in ducting or other container and then being vented to atmosphere. Therefore, these emissions are difficult both to quantify and to abate (AQEG, 2004).
GC	Gas Chromatograph.
GIS	Geographical Information Systems.
GPS	Global Positioning System.
GVW	Gross Vehicle Weight– Combined weight of the vehicles and the goods (AQEG, 2004).
HDD	Heavy Duty Diesel.
HDV	Heavy Duty Vehicles– Road vehicles greater than 3.5 tonnes of GVW (AQEG, 2004).
HGV	Heavy Goods Vehicle– Road vehicles greater than 7.5 tonnes of GVW (AQEG, 2004).
Hz	Hertz; samples per second.
HPS	High Performance Systems, Inc., USA.
ID	Inner Diameter.

Industrial augmentation factor	A dimensionless quantity that accounts for higher emissions from industrial roads than from urban roads (varies between 1 and 7).
In-vitro	Taking place in isolation from a living organism.
In-vivo	Taking place within a living biological organism.
IR	Infra-red.
ITS	Institute for Transport Studies, University of Leeds, UK.
K	Potassium.
KCl	Potassium Chloride.
KCL	King's College London, UK.
Kerbside location	A roadside location, measured at within 0-5 m from the kerb of a busy road. Site classification is done as per the guidance given by the DETR in Technical Guidance TGI(98) and to the classes defined by Stanger Science and Environment in relation to the London-wide surveys.
kt	Kilotonne.
L	Litre.
LAQM	Local Air Quality Management.
LDV	Light Duty Vehicles - Road vehicles less than 3.5 tonnes of weight.
LDSI	Light - Duty Spark Ignition.
LGV	Light Goods Vehicles - Goods vehicles less than 3.5 tonnes weight.
μg m^{-3}	Micrograms per cubic metre.
μm	1 μm = 1 micron = 1 millionth of a metre.
London Congestion Charging zone	Some parts of the Central London area the access to which by motorist incurs a fee.

MATLAB	Matrix Laboratory.
Mg	Magnesium.
MMD	Mass Median Diameter.
Na	Sodium.
NAAQS	National Ambient Air Quality Standards, USA.
NAEI	National Atmospheric Emissions Inventory, UK.
NDIR	Non Dispersive Infra Red.
NEPC	National Environment Protection Council, Australia.
NERC	National Environment Research Council, UK.
NETCEN	National Environmental Technology Centre, UK (part of AEA Technology plc, UK).
nm	Nanometre; 1 nm = 1 billionth of a metre.
NH_4^-	Ammonium.
NO_x	Generic name for all oxides of nitrogen.
N_2O	Nitrous oxide.
NO	Nitric oxide; The main oxide of nitrogen which is created during high temperature combustion.
NO_2	Nitrogen dioxide; Pollutant which is created during high temperature combustion and as the result of oxidation of nitric oxide in the presence of ozone.
NO_3^-	Nitrate.
NPL	National Physical Laboratory (The UK's National Measurement Laboratory).
NYSERDA	New York State Energy Research and Development Authority.
O_3	Ozone.
OC	Organic Carbon.
OD	Outer diameter.
OEG	Onyx Environmental Group plc, UK.

PAH	Polycyclic Aromatic Hydrocarbon.
Pb	Lead.
$PM_{0.1}$	Ultra-fine particles with a mean aerodynamic diameter upto 0.1 μm.
$PM_{2.5}$	Particulate matter with a mean aerodynamic diameter upto 2.5 μm.
PM_3	Particulate matter with a mean aerodynamic diameter upto 3 μm.
PM_{10}	Particulate matter with a mean aerodynamic diameter upto 10 μm.
PM_{15}	Particulate matter with a mean aerodynamic diameter upto 15 μm. It is also referred to as inhalable particles (IP) (USEPA, 1993c).
PM_{coarse}	Coarse fraction of PM_{10} that is larger than 2.5 μm and smaller than 10 μm.
PMCH	Perfluoromethylcyclohexane, $C_6F_{11}CF_3$.
PO_4^{2-}	Phosphate.
ppm	Parts per million.
ppb	Parts per billion.
Precision	A statistical definition of how closely a range of readings are to one another; the degree to which multiple measurements of the same parameter repeat.
PTFE	Polytetrafluoroethylene.
QA/QC	Quality assurance and Quality Control.
QUARG	Quality of Urban Air Review Group, UK.
Resuspension factor	Ratio of airborne concentration of the material and the surface loading of the material.
Resuspension flux	Mass emitted from unit area of road surface in unit time.

Resuspension fraction	Ratio of the amount of material resuspended from a road segment by one vehicle to that available on a road segment.
Resuspension rate	Fraction of surface deposit removed per unit time.
Roadside location	A roadside location, measured at within 0 - 20 m from the kerbside of a busy road.
SCOOT	Split, Cycle and Offset Optimisation Technique.
SD	System Dynamics.
SEM	Scanning Electron Microscope.
SF_6	Sulphur hexafluoride.
Si	Silicon.
Silt loading	Silt loading is defined as the mass of silt size ($\leq 75\,\mu$m) material present on unit area of the road surface. It is obtained by multiplying surface material loading with silt fraction.
SIA	Secondary Inorganic Aerosols; Nitrate, sulphate and ammonium are summarised as SIA.
Silt	Particulate matter of size $< 75\,\mu$m in diameter.
SO_2	Sulphur dioxide; a gas, mainly produced from the burning of coal and oil with a small contribution from diesel.
SO_4^{2-}	Sulphate.
Source area	Area of a section of a road, emissions from which contribute to the receptor.
STANGER	Stanger Science and Environment, UK.
Surface loading	Surface material loading is defined as the mass of particulate material (irrespective of size) present on unit area of road surface. It is estimated by collecting materials from a section of road surface by broom sweeping and vacuuming (USEPA, 1993a).

TEOM	Tapered Element Oscillating Microbalance.
Ti	Titanium.
Toxicology	Study of compounds that are poisonous to humans.
TRAKER	Testing Re-entrained Aerosol Kinetic Emissions from Roads.
TRAMAQ	Traffic Management and Air Quality research programme, UK; `http://www.roads.dft.gov.uk/roadnetwork/tramaq/`.
TROWS	Tyre and Road Wear and Skip Assessment.
TSP	Total Suspended Particulate Matter.
UK	United Kingdom.
UKAURN	UK Automatic Urban and Rural (air quality monitoring) Network.
Urban background	An urban location distanced $>40\,\mathrm{m}$ from busy road sources and therefore broadly representative of city-wide background conditions.
US	United States; same as USA.
USA	United States of America.
USEPA	United States Environmental Protection Agency.
UTC	Urban Traffic Control.
V	Volt.
VOC	Volatile Organic Compounds, (like benzene and other hydrocarbons that evaporate readily).
WCC	Westminster City Council.
WHO	World Health Organisation.
WRAP	Western Regional Air Partnership, USA.
Zn	Zinc.
ZnS	Zinc Sulphide.

Symbols

Symbol	Definition	Unit
A	Source area	m^2
A_{wh}	Average wheel surface area	m^2 wh^{-1} [1]
C	PM_{10} air concentration	μg m^{-3}
C_b	Background concentration of PM_{10}	μg m^{-3}
C_e	Contribution of exhaust emission to roadside PM_{10} concentration	g m^{-3}
C_{br}	Contribution of brake wear emission to roadside PM_{10} concentration	g m^{-3}
C_r	Wind-induced resuspended PM_{coarse} concentration	μg m^{-3}
C_z	Contribution of resuspension to roadside PM_{10} concentration	g m^{-3}
C_t	Concentration of SF_6 released	%
d_y	Depth of the material build-up along the kerb	m
D	Deposition flux on surface	g m^{-2} h^{-1}
e	Particulate matter emission factor	g km^{-1} v^{-1} [2]
e_e	Average exhaust emission factor for a fleet of vehicle	g m^{-1} v^{-1}
e_{br}	Average brake wear emission factor for a fleet of vehicle	g m^{-1} v^{-1}
e_{ei}	Exhaust emission factor of each type of vehicle	g km^{-1} v^{-1}
e_{bri}	Brake wear emission factor of each type of vehicle	g km^{-1} v^{-1}
e'_e	Exhaust emission strength	g m^{-1} s^{-1}

[1] wh^{-1} stands for "per wheel".
[2] v^{-1} stands for "per vehicle".

e'_{e}	Brake wear emission strength	g m^{-1} s^{-1}
f_{wh}	Amount of material carried onto Manor Road by each wheel	g wh^{-1}
f_x	Fraction of material removed from a road segment along the road by one vehicle	v^{-1}
f_y	Fraction of material removed from a road segment across the road by one vehicle	v^{-1}
f_z	Fraction of material removed from a road segment to air by one vehicle	v^{-1}
f'_{w}	Amount of material carried onto Manor Road by each vehicle	g v^{-1}
f'_x	Amount of material removed from a road segment along the road by one vehicle	g v^{-1}
f'_y	Amount of material removed from a road segment across the road by one vehicle	g v^{-1}
f'_z	Amount of material removed from a road segment to air by one vehicle	g v^{-1}
f''_{r}	Fraction of material removed per unit time due to rain	h^{-1}
F_{a}	Rate of road abrasion deposition	g s^{-1}
F_{d}	Rate of dry deposition	g s^{-1}
F_{g}	Rate of grit deposition at Gloucester Place	g s^{-1}
F_{in}	Sum of accumulating fluxes	g s^{-1}
F_{out}	Sum of draining fluxes	g s^{-1}
F_{r}	Rate of material removal by rain (i.e., amount of material removed from a road segment in unit time due to rain)	g s^{-1}
F_{t}	Rate of tyre wear deposition	g s^{-1}

F_{w}	Rate of material input from EGTS onto Manor Road	g s^{-1}
F_x	Rate of material removal along the road (i.e., amount of material removed from a road segment along the road in unit time by the whole fleet of vehicles)	g s^{-1}
F_y	Rate of material removal across the road (i.e., amount of material removed from a road segment across the road in unit time by the whole fleet of vehicles)	g s^{-1}
F_z	Rate of material emission to air (i.e., amount of material removed from a road segment to air in unit time by the whole fleet of vehicles)	g s^{-1}
I	Industrial augmentation factor	dimensionless
I_{r}	Intensity of rain	mm h^{-1}
K	Resuspension factor	m^{-1}
l	Length of the road segment	m
m_0	Amount of material on road segment at the end of the gritting	g
m_{t}	Amount of material on road segment at time t	g
M_{w}	Molecular weight of SF_6	
n	Number of traffic lanes	
n_{hdv}	Flow rate of HDVs	v s^{-1}
n_{ldv}	Flow rate of LDVs	v s^{-1}
n_{ti}	Number of tyres/wheels in each vehicle	tyres/wheels per vehicle
n_{v}	Traffic flow rate	v s^{-1}
n_{w}	Traffic flow rate from EGTS to Manor Road	v s^{-1}
n_{wh}	Number of wheels per vehicle	wh v^{-1}

N	Number of days/hours in the averaging period used for moisture correction factor	days/hours
p	Pressure drop in tracer release equipment	bar
P	Number of days/hours with at least 0.254 mm (0.01 in) of rain during the averaging period used for moisture correction factor	days/hours
Q	SF_6 release rate at 20 °C	kg s^{-1}
r_t	Concentration-emission ratio obtained from the tracer release	s m^{-3}
R	Resuspension flux	g m^{-2} s^{-1}
s	Silt fraction	%
S	Surface loading	g m^{-2}
sL	Silt loading	g m^{-2}
S_g	Surface loading contributed by gritting	g m^{-2}
t_g	Duration of gritting	s
t_r	Tracer release time	min
t_{xavg}	Time elapsed between end of gritting and time when general level of grit in air at Site 1 exceeded that at Site 2	s
t_y	Time taken for the kerb-side build-up to take place	s
T_0	Time taken for the material on the road to decay by a factor of e^{-1}	s
u	Wind speed	m s^{-1}
u_*	Friction velocity of the particles	m s^{-1}
u_s	Street level wind speed	m s^{-1}
u_{xmax}	Maximum speed of material movement along the road by the whole fleet of vehicles	m s^{-1}
u_{xavg}	Average speed of material movement along the road by the whole fleet of vehicles	m s^{-1}

u'_x	Average distance of material movement along the road by one vehicle	m v^{-1}
$u_{y\max}$	Maximum speed of material movement across the road by the whole fleet of vehicles	m s^{-1}
$u_{y\text{avg}}$	Average speed of material movement across the road by the whole fleet of vehicles	m s^{-1}
u'_y	Average distance of material movement across the road by one vehicle	m v^{-1}
v	Volume of tracer release equipment	ml
V_a	Volume of road abrasion per unit time	m^3 s^{-1}
v_d	Deposition velocity of particulate matter	m s^{-1}
V_m	Molar volume at 20 °C	L
V_s	Settling velocity of the particles	m s^{-1}
V_w	Volume of material carried onto Manor Road by each wheel	m^3 wh^{-1}
w	Width of the road segment	m
W	Average vehicle weight	tonne
w_wh	Average wheel width of the vehicles entering from EGTS to Manor Road	m
w_y	Width of the material build-up along the kerb	m
x_{1-2}	Distance between Site 1 and Site 2 at Gloucester Place	m
z_i	Proportion of each type of vehicle	%
z_j	Proportion of each type of road material	%
β	Angle between Geographical North and North of the rotated coordinate system at Gloucester Place	degree
θ	Direction of material movement relative to vehicle movement i.e., angle of spread	degree
Λ	Total resuspension rate	s^{-1}

Λ_{air}	Resuspension rate for particles moving in true airborne suspension	s^{-1}
Λ_s	Scavenging coefficient	s^{-1}
Λ_{salt}	Resuspension rate for particles moving in saltation	s^{-1}
Λ_{sc}	Resuspension rate for particles moving in surface creep	s^{-1}
ρ	Average density of road material	g m^{-3}
ρ_j	Density of each type of road material	g m^{-3}
ρ_w	Density of material entering from EGTS to Manor Road	g m^{-3}
σ_x	Along-street spread of a patch of material on road surface	m
σ_y	Across-street spread of a patch of material on road surface	m
τ	Thickness of material sticking around the wheel at the entry to Manor Road	m
τ_a	Thickness of abrasion per vehicle on a good road	m v^{-1}
τ'_a	Thickness of abrasion per vehicle with road condition considered	m v^{-1}
τ''_a	Thickness of abrasion per unit time	m s^{-1}
ϕ	Average wheel diameter of the vehicles entering from EGTS to Manor Road	m
χ	Dispersion coefficient	m^{-2} h;m^{-1} s
ψ	Road condition factor	dimensionless
ω_t	Average tyre wear rate for a fleet of vehicle	g m^{-1} v^{-1}
ω'_t	Tyre wear deposition rate per unit length of road surface	g m^{-1} s^{-1}
ω''_t	Tyre wear rate	g km^{-1} per tyre

Chapter 1

Introduction

1.1 Background

Effects of airborne particulate matter alone and in conjunction with gaseous pollutants not only have adverse impacts on plant, animal and non-living materials (Painter, 1974), but also have been partly responsible for several global, regional and local impacts, e.g., change of radiative balance (Haywood and Boucher, 2000; Kalognomou and Moussiopoulos, 2005; Rosen and Novakov, 1977), modification of cloud property, the global climate change (Kalognomou and Moussiopoulos, 2005; Saxena, 1997), acid rain (Angstorm, 1964; Charlson and Rodhe, 1982; Colvile et al., 2001; Sanhueza, 2001), stratospheric ozone depletion (Colvile et al., 2001), reduction in visibility and soiling of monuments (Kalognomou and Moussiopoulos, 2005). In human beings, inhalation of particulate matter can cause chronic pulmonary diseases like lung cancer, chronic bronchitis, bronchial asthma and emphysema (Carlton et al., 1999; Hatch and Gross, 1964; Holgate et al., 1999; Kagawa, 1984; Painter, 1974; Stern, 1962; Wilson and Spengler, 1996). The sizes of the particles that are harmful to health are PM_{10} (Particulate matter having aerodynamic diameter below $10\,\mu$m) (AQEG, 2004; DEFRA, 2000; EPAQS, 1998, 2001) and this includes $PM_{2.5}$ (Particulate matter having aerodynamic diameter below $2.5\,\mu$m). Some studies suggest $PM_{2.5}$ to be more harmful to health than PM_{10} (Anderson et al., 1992; Lippmann, 1998; Pillai et al., 2002; Seaton et al., 1995; Schwartz et al., 1996). These particles can remain suspended in air for a long time,

can transport to long distances and can enter via the respiratory tract deeper parts of lungs (ApSimon et al., 2001; Brain and Valdberg, 1979; Stern, 1962). While the debate on properties (e.g., chemical and mineralogical composition, number, surface area and mass) of particulate matter that cause adverse health effects is still ongoing (Anderson et al., 1992; Brown et al., 2002; Englert, 2004; Harrison and Yin, 2000; Hauck et al., 2004; Laden et al., 2000; Lippmann, 1998; Schwartz et al., 1996; Maynard and Maynard, 2002; Seaton et al., 1995; WHO, 2003; Wilson and Suh, 1997), it has been proved beyond doubt that particulate matter has adverse impacts on health (Carlton et al., 1999; COMEAP, 1995, 1998, 2001; Dockery et al., 1993; Dockery and Pope III, 1994; Hoek and Brunekreef, 1993; Hoek et al., 2002; Katsouyanni et al., 1995, 1997; Maheswaran et al., 2002; Nyberg et al., 2000; Ostro et al., 1999; Pope III et al., 1995a,b; Pope III and Dockery, 1999; Salter and Parsons, 1999; Spix et al., 1998; Takano et al., 1997; van Wijnen and van der Zee, 1998; Vedal, 1995; Wilson and Spengler, 1996).

During the last three decades the consumption of coal in domestic burning has reduced significantly. Oil, natural gas and nuclear power are gradually replacing coal as an industrial fuel. On the contrary rapid urbanisation with very fast growth of traffic in cities across globe is resulting in the emissions from roads as the most important source of particulate matter in urban air. Several studies have established particulate matter emissions associated with traffic activity on paved roads as a major source of urban airborne particulate matter (ADB, 1992; APEG, 1999; AQEG, 2004; Barrowcliffe et al., 2002; Benitez et al., 2001; Buckingham et al., 1997; CARB, 1998; Chow et al., 1996b; D'Alessandro et al., 2003; DEFRA, 2000, 2003; Dorsey et al., 2002; Gaffney et al., 1995; Gertler et al., 2000; Harrison et al., 1997b; Johansson et al., 2004; Manoli et al., 2002; Zimmer et al., 1992). Urban traffic emissions consist of two components having different emission mechanisms. These are "exhaust emissions", which consist of the by-products of fuel combustion in the vehicle engine and the unburnt fuel, and are therefore directly linked with fuel and engine quality of the vehicle although driving pattern, congestion, etc. partly influence it; and "non-exhaust emissions", which are mostly comprised of the materials of the road surface that become suspended due to traffic movement, and are therefore called "paved road surface emissions". Often

paved road surface emission is referred to as "resuspension". The term "resuspension" is usually used for the process where surface materials, previously deposited due to atmospheric process or any other natural process or due to any anthropogenic activity, become airborne due to natural (e.g., wind action, raindrop impact, etc.) or anthropogenic activity (e.g., mechanical disturbance by pedestrians and vehicles, agricultural operations, etc.) (Nicholson, 1988; Nicholson and Branson, 1990). To be precise, the term suspension or entrainment should be used for surface materials which were not previously deposited (e.g., material produced out of vehicle-induced road abrasion) and resuspension or re-entrainment should be used for those previously deposited (e.g., atmospheric depositions, material from a nearby unpaved road carried along the traffic wheel onto paved road). Even in case of road abrasion, if all the material produced due to a vehicle become suspended due to that vehicle itself, then it will not be a case of resuspension because as these are produced they get suspended. However if some of it is left on the road, which is expected to be the case, and become airborne by other vehicles then it should be treated as resuspension. In this case it is the residence time of material on the road that dictates whether it should be termed either of the two. Clearly, it is difficult to distinguish between resuspension/re-entrainment and suspension/entrainment. Therefore, as discussed by Sehmel (1980b) for wind-induced resuspension, the term resuspension usually includes suspension/entrainment. Materials available for resuspension include any primary or secondary material that is moved in or deposited onto the road surface and therefore include redeposition of exhaust emission and other material produced at the road/vehicle interaction (e.g, particulate matter produced out of road abrasion, tyre wear). As such, resuspension, to some extent, double counts the emissions.

Exhaust emissions of particulate matter are much better quantified and documented than non-exhaust emissions (AQEG, 2004). With introduction of cleaner fuels (e.g, fuel quality improvement with regard to sulphur and aromatic content (Ntziachristos and Samaras, 2003), unleaded petrol replacing leaded petrol and use of Compressed Natural Gas (CNG) instead of petrol and diesel), better emission control technology (e.g., use of particulate matter filters and catalytic converters) and tighter legislation

(e.g., Euro standards), exhaust emissions have decreased appreciably despite an overall increase in traffic (AQEG, 2004; Goodwin et al., 1999; Kathuria, 2005). However, technological measures aimed at reducing exhaust emissions are not generally effective in reducing resuspension. Resuspension is believed to depend on some factors that influence exhaust emission as well as several other factors such as road condition, local meteorology and weather (Artíñano et al., 2004; Harrison et al., 1997a, 2001; Johansson et al., 2004; QUARG, 1996). Most combustion emissions, traffic exhaust emissions included, of PM_{coarse} (Coarse fraction of PM_{10} that is larger than $2.5\,\mu$m and smaller than $10\,\mu$m) were eliminated in the 20^{th} century. The control of exhaust emissions[1] at present is aimed at reducing the emissions of $PM_{2.5}$ because of recent evidence of its increased health risk (Anderson et al., 1992; Lippmann, 1998; Pillai et al., 2002; Seaton et al., 1995; Schwartz et al., 1996). Therefore, the emissions of PM_{coarse} associated with paved road surface emissions which are primarily produced from resuspension of soil and road surface dust by traffic-induced turbulence (Abu-Allaban et al., 2003a,b; AQEG, 2004; Artíñano et al., 2004; Braaten et al., 1990; Hall, 1989; Harrison et al., 2001; Kristensson et al., 2004; Kulmala et al., 1986) as well as perhaps by the wind (Triantafyllou et al., 2002), remain uncontrolled (Harrison et al., 2004a). Triantafyllou et al. (2002) have shown that wind induced resuspension is a major source at urban and industrial locations where, depending on weather condition, it can contribute substantially to general pollution level as well as to pollution episodes. While wind-induced resuspension is a concern, particularly in the aftermath of any accidental spill of hazardous contaminants like that of Chernobyl nuclear spill, contribution of traffic-induced resuspension to roadside particulate matter concentration and local and national emission level, which is about 10% of PM_{10} emissions in UK (Goodwin et al., 1999), is of significant importance, especially in urban areas. Further, while resuspension due to normal wind action causes redistribution of contaminants in urban and regional scale (Anspaugh et al., 1975; Garland and Playford, 1992), resus-

[1]Exhaust aerosol is emitted primarily in sub-micron particle sizes centered on particles of $0.1\,\mu$m to $0.2\,\mu$m in diameter (Hildemann et al., 1991b). The typical size ranges of diesel exhaust from a bus and petrol exhaust from a car are $0.02\,\mu$m to $0.13\,\mu$m and $0.04\,\mu$m to $0.06\,\mu$m respectively (Morawska et al., 1998; Ristovski et al., 1998).

pension due to traffic results in exposure of very high level of particulate pollution to commuters and people living close to urban streets. Laboratory studies indicate that the more toxic components such as quartz are associated with the resuspended road dust (Murphy et al., 1998). Therefore resuspended dust may be a cause of more adverse health impact than exhaust emissions. With studies indicating the entrainment of dust from the road surface as either a comparable or greater source than exhaust emission (Abu-Allaban et al., 2003b; APEG, 1999; AQEG, 2004; Ball and Caswell, 1983; DEFRA, 2000; Düring et al., 2002a; Johansson et al., 2004; Schulze, 2001), it is a cause of concern for compliance of the present and more importantly, the stricter future air quality standards such as the European Commission (EC) indicative stage 2 limit values of 50 μg m^{-3} PM_{10} as a 24-hour mean and 20 μg m^{-3} PM_{10} as an annual mean, to be achieved by 1 January 2010.

This has led to the investigation of paved road particulate matter emission since the early 1970s. To date, most of the studies undertaken to estimate the particulate matter emission due to traffic have essentially applied regression analysis between emission, obtained from measured concentration, and other probable road and vehicle related parameters to derive an empirical equation that can be used to quantify emission rate in units of mass per unit distance travelled by a vehicle, called "emission factor" equation. The most detailed and widely used United States Environmental Protection Agency (USEPA) AP-42[2] model for paved road emissions, which is based on an empirical approach that relates vehicle weight and silt loading ("silt" is particulate matter less than 75 μm in size; "loading" is amount of silt on the road surface per unit area of road surface) to emission rate of particulate matter (USEPA, 1997), has been widely criticised for its lack of mechanistic basis (Countess et al., 2001; Nicholson, 2001; Venkatram, 2000, 2001) and poor predictivity, reported from several field studies (Fitz and Bufalino, 2002; Kantamaneni et al., 1996; Venkatram and Fitz, 1998; Venkatram et al., 1999; Zimmer et al., 1992). The main criticism is that the formulation does not take into account sources that contribute to the particulate matter on the road and it also does not consider the removal mechanisms. This is a purely statistical model

[2]AP-42 model is described in Section 2.12.

which can give average results over an area with large number of roads. But it fails to predict the emission factor at local scale, e.g., for a certain road or section of road (Countess et al., 2001; Venkatram et al., 1999; Venkatram, 2000) and therefore, should be applied with extreme caution (Nicholson, 2001).

1.2 Research motivation

For emission estimation to be credible, a model should include various natural and anthropogenic processes through which particulate matter enters to a road/vehicle environment, moves within various compartments of the environment and ultimately escapes from it i.e., emission factors should be based on real-life processes involved in particulate matter emission, rather than statistically significant variables. A dynamic mass balance model that accounts for the sources of particulate matter on a road surface, its movement and ultimately its removal from the road surface could be a better way of understanding and estimating the paved road emission, a view shared by other researchers (Countess et al., 2001; Kuhns et al., 2001a). Therefore, the main motivation behind this research is to understand the particle emission process from paved roads, the need of which has been emphasised by many researchers (Nicholson, 2001; Venkatram, 2001).

1.3 Aim

The aim of this thesis is to present research conducted to enhance understanding of the interplay of different processes that are involved in traffic-induced resuspension of particulate matter. The principal aim of the research was to identify and quantify the processes that cause movement of particulate pollutants in the urban road environment, and parameterise them for a model that explains the emission and translocation of particulate matter from road due to traffic and importance of resuspension in urban air pollution.

1.4 Key research questions

The following key research questions were framed, the answers to which were expected to meet the aim stated above.

1. What are the sources and sinks of particulate matter in the road/vehicle environment?

2. What are the different traffic-induced processes through which particulate matter build-up, movement and loss take place on a paved road surface?

3. What are the variables that control the build-up, movement and depletion of particulate matter on a road surface and what are their relative importances?

4. How can these variables be parameterised for a simple-mass balance model? The parameterisation is expected to be achieved by finding answers to the following questions:

   (a) What is the rate of material input to a road surface from different sources?

   (b) How much time does it take for material to move from one segment of the road to the other, from carriageway to kerbs/pavements/drains, from road surface to air and finally to disappear from the road surface?

   (c) How far does material move in a given time along and across the road?

   (d) How much material is removed in different directions from a road segment in a given time, i.e., what are the material removal rates in different directions?

1.5 Objectives

The research to answer the above questions involves the following steps:

1. Undertaking a literature survey to review earlier works associated with paved road traffic emissions.

2. Carrying out a field experiment to observe the movement of particulate matter on a road surface.

3. Identification of different processes associated with material movement and emissions from paved road surfaces.

4. Development of quantitative estimates for the parameters that control the above processes.

5. Development of a model which includes mathematical formulation of these processes.

6. Checking the theoretical and scientific correctness of the model and evaluating the model against independent measurements.

1.6 Scope

This modelling exercise is carried out with the assumption that traffic movement causes particles on a road surface to move and become airborne and the intention is to formulate and parameterise a model that includes these movement and resuspension processes. This involves finding answers to the research questions 4a - 4d mentioned in Section 1.4. The model therefore does not deal with the mechanisms behind particle movement and resuspension such as the forces acting at the wheel/road interface, different aerodynamic forces that cause particle movement and uplift from a surface, relative importance of tyre stress and traffic-induced turbulence and relative importance between traffic-induced turbulence and wind-induced turbulence, although these are briefly reviewed in Chapter 2 to gain some understanding. The model proposed in this thesis will attempt to quantify the contribution of several sources that replenish the suspendible dust on road surface and rate of removal of these suspendible dust by traffic induced material removal processes. None of these are addressed in AP-42 model. Therefore description, parameterisation and quantification of traffic-induced material replenishment processes and material removal processes on paved road surfaces are the core contribution of this research.

1.7 Thesis outline

The thesis chapters, outlined below, reflect the objectives set out in Section 1.5.

Chapter 2 is a literature review. It starts with a review of association of airborne particulate matter with adverse health effects so that the importance of different properties of particulate matter that are likely to determine the effects are appreciated. Major sources of particulate matter in urban air are then discussed to emphasise the emergence of paved road emissions as an important source. Previous studies that highlight traffic as a major source of urban airborne particulate matter are reviewed. The growing importance of traffic-induced resuspension is discussed. General theory behind wind and traffic-induced resuspension is described. Results of some earlier studies that reported estimates for wind and traffic-induced resuspension are presented. Prevailing approaches for estimation of traffic-induced resuspension and its limitations are critically reviewed. Finally, the key issues that emerge from the above review are summarised, and the importance and novelty of this research are described.

Chapter 3 describes the on-field measurement at two sites. At Gloucester Place, as a part of this project, data are generated through a field campaign and at Manor Road, data generated earlier by several agencies have been compiled and further analysed for using in this research. The description of field measurements includes site selection criteria, site description, instrumentation and measurement protocol.

Chapter 4 presents results of the field measurements. The field results include meteorology, traffic flow and composition, dispersion condition, surface loading, PM_{10} level, qualitative observation and quantitative analysis of material movement processes.

Chapter 5 describes the model development process, the main output of this research. The model development process consists of a description of model structure and assumptions, a description of the software used for model development and nu-

merical integration of the equations, mathematical formulation of model components and finally, evaluation of parameter values. It also includes a brief note on model evaluation strategy.

Chapter 6 presents the model results for both sites. The scientific correctness of the model is assessed from the model result for Gloucester Place and model results for Manor Road are used to evaluate the model.

Chapter 7 starts with a brief summary of the thesis content. New understandings gained from the research that include comparison of measurement and model results with previous studies and the difference between present modelling approach and the earlier ones are described. Uncertainties associated with this study are enumerated. Relevance of this research findings to the urban air quality assessment are discussed. The thesis ends with a short conclusion and a summary of limitations and provides an outlook on future studies necessary in this area.

Chapter 2

Literature review

2.1 Introduction

Epidemiological studies have indicated a convincing relationship between the adverse health effects and PM_{10} (Bascom et al., 1996; Carlton et al., 1999; COMEAP, 1995; Dockery et al., 1993; Lippmann et al., 1998; Mukerjee, 1998; Ostro et al., 1998; Pope III and Dockery, 1992; Pope III et al., 1995b; Spix et al., 1998). Irrespective of places of study, that include different countries having widely varying pollution mix, the studies show consistent and coherent association of increased mortality, hospital admissions, respiratory symptoms and reduced pulmonary function with short-term exposure to PM_{10} (Dockery and Pope III, 1994; Hoek and Brunekreef, 1993; Hoek et al., 2002; Katsouyanni et al., 1995, 1997; Ostro et al., 1999; Salter and Parsons, 1999; Spix et al., 1998; Vedal, 1995; Wilson and Spengler, 1996; Wyzga, 2002). Association between long-term exposure to PM_{10} and increased mortality from lung cancer, respiratory and cardiovascular disease has also been reported by several studies (Abbey et al., 1998, 1999; AckermannLiebrich et al., 1997; Berico et al., 1997; Braun-Fahrländer et al., 1997; Dockery et al., 1993, 1996; Maheswaran et al., 2002; Nyberg et al., 2000; Pope III et al., 1995a,b; Wilson and Spengler, 1996). Some studies have reported association of chronic respiratory disease in children with differences in the level of air pollution within cities, and especially with distance to or living in busy streets (Edwards et al., 1994; Weiland et al., 1994; Wjst et al., 1993).

Estimates by the UK Department of Health's Committee on the Medical Effects of Air Pollutants (COMEAP) for the UK show short-term (acute) exposure to PM_{10} is associated with about 8100 deaths and 10500 hospital admissions brought forward in 1996 (COMEAP, 1998) and 6500 deaths and 6400 hospital admissions brought forward in 2002 (COMEAP, 2001). Pope III and Dockery (1999) estimated that an increase of PM_{10} by $10\,\mu g\ m^{-3}$ is associated with 0.8 % increase in mortality. Epidemiological studies by Wilson and Spengler (1996) show that a $10\,\mu g\ m^{-3}$ increase in PM_{10} concentration results an increase of 0.5 % to 1.5 % in premature total mortality in short term (episodic) exposure and up to 5 % in premature total mortality in long term (life long) exposure. An estimate based on several health studies is that, as a whole, there is a 1 % increase in total mortality rate per $10\,\mu g\ m^{-3}$ increase in PM_{10} concentration (Harrison and Yin, 2000; Katsouyanni et al., 1997; Vedal, 1995). The COMEAP estimates that a reduction of $1\,\mu g\ m^{-3}$ of $PM_{2.5}$ over the lifetime of the current population of England and Wales could lead to a gain of 0.2 million life-years to 0.5 million life-years, equivalent, on average, to between 1.5 days and 3.5 days per individual (COMEAP, 2001). Vedal (1995) reported an estimate that a $10\,\mu g\ m^{-3}$ exposure increment of PM_{10} above a base level of $20\,\mu g\ m^{-3}$ (although one cannot be sure yet that there exists a firm threshold concentration for PM_{10} below which no impacts are detectable) results in a 3.4 % increase of daily respiratory mortality and a 1.4 % increase of cardiac mortality. For each $10\,\mu g\ m^{-3}$ increase in PM_{10} concentration, Vedal (1995) estimated a 0.8% increase in hospitalisations, a 1.0 % increase in emergency room visits for respiratory illnesses, a 9.5 % increase in days of restricted activity due to respiratory symptoms, a 4.1 % increase in school absenteeism and a 1.2 % increase in reporting of cough. A European assessment suggested that air pollution caused 6 % of total mortality or more than 40,000 attributable cases per year and half of these air pollution induced mortalities was attributed to motorised traffic (Künzli et al., 2000). A review by Vedal (1997) gives a good summary of acute health effects due to particulate air pollution.

As a result, countries across the globe, including the UK and Europe, have specified limit values for PM_{10} (Goodwin et al., 1999). On the other hand, emerging evidence from several epidemiological and toxicological laboratory studies indicates $PM_{2.5}$ or

even finer particles such as $PM_{0.1}$ (ultra-fine particles with diameter less than 0.1 μm) a better metric for health effects than PM_{10} (Anderson et al., 2001; Dockery et al., 1992, 1993; Donaldson et al., 1998; Ferin et al., 1991; Ibald-Mulli et al., 2002; Laden et al., 2000; Li et al., 1999; Oberdörster et al., 1992, 1995; Peters et al., 1997b; Pope III et al., 2002; Pillai et al., 2002; Samet et al., 2000a,b; Schwartz et al., 1996; Seaton et al., 1995; Wichmann and Peters, 2000; Wyzga, 2002). Probable reasons for this include the ability of these small particles to remain suspended and move long distance in air, penetrate deep into lungs (Brain and Valdberg, 1979; Seaton et al., 1995; Stern, 1962) and get enriched with toxic elements (Anderson et al., 1992; Infante et al., 1990; Lippmann, 1998; Schwartz et al., 1996; Wilson and Suh, 1997). This has brought the concerns about the environmental impacts of tiny particles, especially from traffic, into sharper focus (Bagley et al., 1996; Künzli et al., 2000; Ruellan and Cachier, 2001; Shi et al., 2001). Therefore masses or numbers of fine particles such as $PM_{2.5}$, PM_1, $PM_{0.1}$ are expected to be the basis of future legislation. Table 2.1 shows the $PM_{2.5}$ standard of USA, Canada, Australia and the EU.

However several studies that have highlighted the growing importance of $PM_{2.5}$ have not necessarily ruled out any adverse health outcome from PM_{10}; rather these studies have shown that the likelihood of association of $PM_{2.5}$ with health effects are comparatively more than the association between PM_{10} and adverse health effects (Anderson et al., 1992; Berico et al., 1997; Dockery et al., 1993; Ibald-Mulli et al., 2002; Laden et al., 2000; Lippmann, 1998; Schwartz et al., 1996; Wyzga, 2002). Therefore there remains the risk on health from PM_{coarse}. And then the debate on mechanisms behind the adverse health effect of particulate matter is still inconclusive. Whether the important variable that explains the toxic effect of particulate matter is particle number, particle surface area, particle mass or chemical composition of particle is yet not fully understood (Adamson et al., 1999; Beverland et al., 2002; Brown et al., 2002; Englert, 2004; Harrison et al., 1999; Harrison and Yin, 2000; Hauck et al., 2004; Hoek et al., 2002; Maynard and Maynard, 2002; Nemmar et al., 2002; Ntziachristos and Samaras, 2003; Peters et al., 1995, 1997a,b; Phupinyokul and Harrison, 1999; Schlesinger and Cassee, 2003; Seaton et al., 1995; WHO, 2003).

Table 2.1: Air quality standards for $PM_{2.5}$

Country	Standard	Averaging time
USA	15 μg m^{-3}	Annual arithmetic mean[a]
	65 μg m^{-3}	24-hour average[b]
California (USA)[c]	12 μg m^{-3}	Annual arithmetic mean[a]
Canada[d]	30 μg m^{-3}	24-hour average[e]
Australia[f]	8 μg m^{-3}	Annual arithmetic mean
	15 μg m^{-3}	24-hour average
EU	25 μg m^{-3}	Annual arithmetic mean[g]

[a]To attain this standard, the 3-year average of the weighted annual mean $PM_{2.5}$ concentrations from single or multiple community-oriented monitors must not exceed 15.0 μg m^{-3}.

[b]To attain this standard, the 3-year average of the 98th percentile of 24-hour concentrations at each population-oriented monitor within an area must not exceed 65 μg m^{-3}.

[c]California has even stricter $PM_{2.5}$ standard than the US federal standard (American Lung Association, 2005).

[d]The $PM_{2.5}$ Canada-Wide Standards (CWS) had been established for the interim period prior to the planned review of the standard to be completed by 2005, which will incorporate advancements in scientific, technical and economic information and analysis (Sawicki et al., 2000).

[e]24 hour standard at the 98th percentile averaged over 3 years with a compliance date of 2010 (Sawicki et al., 2000).

[f]These are Advisory Reporting Standards. The goal is to gather sufficient data nationally to facilitate a review of the Advisory Reporting Standards as part of the review of this measure scheduled to commence in 2005 (NEPC, 2003).

[g]20 % on the entry into force of this Directive, reducing on the next 1 January following and every 12 months there after by equal annual percentages to reach 0 % by 1 January 2010. It should be assessed as a 3-calendar year running annual mean concentration averaged over all sampling points established. The reference year 2010 shall be the mean concentration of the years 2008, 2009 and 2010 (Commission of the European Communities, 2005).

To date in the UK, $PM_{2.5}$ and PM_{10} have been very closely correlated wherever they have been measured and their associations with the health effects have thus been inseparable (APEG, 1999; Harrison et al., 2001). It is possible that a long time-series comparing the associations of these and other metrics with health outcomes will resolve these difficulties which is not available at present. The Expert Panel on Air Quality Standards (EPAQS), in considering the most appropriate particle fraction on which to develop air quality standards, has reaffirmed its view that both fine and coarse fractions represent a health-risk, although the precise method of their actions and outcomes may differ, and therefore, measurement of particulate air pollution as the metric of PM_{10} mass concentration, which includes essentially all respirable particles, provides the most appropriate basis for an air quality standard for protection of public

health in the UK. It may be that further research will lead to additional or alternative metrics, for example, $PM_{2.5}$ or counts of ultra-fine particles, but the currently available data do not allow satisfactory derivation of a standard based on these (AQEG, 2004; DEFRA, 2000; EPAQS, 1998, 2001).

$PM_{2.5}$ standards recommended by the USEPA (Table 2.1) were included, for the first time, in the US National Ambient Air Quality Standards in July 1997 (Hitchins et al., 2000; Kittelson, 2001; Pope III, 2000). After several litigations, the US Court of Appeals finally cleared the way to implement these standards in March 2002 (NAAQS, 2002; Pope III, 2000). The main technical argument against the $PM_{2.5}$ standard is that implementation of standards for $PM_{2.5}$ and PM_{10} simultaneously double regulates the $PM_{2.5}$ fraction. Because at present there exists no evidence to suggest that all of the action of PM_{10} reported in the past was due to $PM_{2.5}$, we must have a standard for PM_{coarse}, which in combination with $PM_{2.5}$ standard regulates PM_{10}[1]. Therefore, until the time a PM_{coarse} standard is implemented, PM_{10} standards should be the basis for air pollution control.

2.2 Emergence of traffic as the major source of urban airborne particulate matter

In urban atmosphere, PM_{10} may be regarded as having three predominant source types: primary particles, emitted directly from sources involving combustion process (e.g., road traffic, power generation and other industrial combustion processes), which are mostly small in size, often less than 1 μm, and dominate $PM_{2.5}$; primary particles which are natural in origin and dominate PM_{coarse} (mainly formed from a variety of primarily non-combustible sources such as wind-blown suspended soils and dusts of crustal origin, forest and other natural fires, resuspended dust, sea-salts, biological particles, particles

[1]The USEPA is now in consultation stage to set up standards for PM_{coarse}. It is expected that PM_{10} standards will be phased out with implementation of new standards for PM_{coarse}. The EPA Clean Air Scientific Advisory Committee (CASAC) feels that the $PM_{2.5}$ standard should be even stricter i.e., 35 μg m^{-3} to 30 μg m^{-3} for 24-hour period and 14 μg m^{-3} to 13 μg m^{-3} on an annual average basis. In addition the CASAC endorses establishment of a 24-hour average PM_{coarse} standard, mainly for urban areas (CASAC, 2005). Sawicki et al. (2000) have also expressed such a standard for Canada.

from construction work, mining, agricultural harvesting and quarrying); and secondary particles (mainly sulphate and nitrate formed by chemical reactions such as gas-to-particle conversion and photochemically formed particles in the atmosphere) which are also generally less than 2.5 μm, but the size could vary depending on humidity (APEG, 1999; DEFRA, 2000, 2003; Goodwin et al., 1999; Le Bihan et al., 2004; QUARG, 1996). The important distinction between primary particles and secondary particles is that while primary particles are directly emitted from a source into air, secondary particles are not emitted directly from source, be it either natural or anthropogenic. Secondary particles are formed in the atmosphere as a result of chemical reactions leading to the formation of substances of low volatility which consequently condense into the solid or liquid phase, thereby constituting particulate matter. Such particles are generally the result of atmospheric oxidation processes and the substances oxidised may be either natural or anthropogenic in origin (AQEG, 2004).

Activity-wise, three main sources of primary PM_{10} are road transport, stationary or domestic combustion and industrial processes. Industrial emissions in UK are falling over the years due to better environment-friendly practices in mining, quarrying, construction and bulk handling of materials. Despite a significant growth of electricity generation between 1970 and 1999, emissions from this sector are declining due to fuel sources changing from coal to natural gas and nuclear power and improved performance of electrostatic precipitators at coal-fired power plants. Emissions from domestic coal combustion have significantly declined, from 223 tonnes (42% of total UK emission) in 1970 to 38 tonnes (20 % of the total UK emission) in 1999, after implementation of the Clean Air Acts that eliminated domestic coal combustion in many urban areas (Goodwin et al., 1999), although it can still be a significant source in smaller towns and villages. A similar trend is seen in other countries with the exception of a few emerging economies such as China and India where, because of high demand of cheap fuel and increasing requirement of housing, emissions from mining, coal burning and construction activities are still among the major sources of urban airborne particulate matter.

This leaves the transport sector as a major source of urban air particulate pollution.

Although emissions from road, air, rail and water contribute towards the deterioration of the environment (Colvile et al., 2001), emissions from road traffic have been the cause of much concern due to their effects on urban air quality. Because of the increasing trend of people living in urban areas in all parts of world, it has a pronounced impact on human health. Emissions from road transport have shown significant growth, 13 % of total UK emission in 1970 to 20 % of total UK particulate matter emission in 1999. In urban areas with little industrial activity, where public power and industrial processes do not make a significant contribution, the contribution of road transport to urban emissions is even higher; for example, as much as 80 % of primary emissions in London are contributed by road transport (Buckingham et al., 1997). Diesel powered vehicles emit a greater mass of particulate matter than petrol engine vehicles (Harrison et al., 2005; Kittleson et al., 2004; Le Bihan et al., 2004), roughly one order of magnitude higher per vehicle kilometre (Berdowski et al., 1996) or per unit of fuel used (Johnson et al., 2005). Emissions from diesel vehicles have been growing due to the growth in heavy-duty vehicles and the move towards more diesel cars (diesel cars have increased from 3 % to 12 % of the UK vehicle fleet between 1990 and 1999) (Goodwin et al., 1999).

2.3 Contribution of traffic to urban airborne particulate matter : the evidence

It has long been studied and proved that traffic is a major source of urban particulate matter. The contributions of traffic to urban air particulate matter level result from a combination of exhaust emission and resuspension of road dust caused by passing traffic. On a data based on a combination of both Gaffney et al. (1995) estimated that traffic on paved roads contributed about 30 % of PM_{10} emissions in California. In the Californian South Coast Air Basin, paved road dust accounted for 65 % of the PM_{10} emissions (CARB, 1998). Using a chemical mass balance (CMB) receptor model, Chow et al. (1996b) found motor vehicles as the largest sources of PM_{10} in Santa Barbara

County, California. They also found that contribution of motor vehicle to PM_{10} was greater than the marine contribution in the urban areas, but smaller at the non-urban background sites. Zimmer et al. (1992) found that emissions of particulate matter from paved road accounted for between 40 % and 70 % of total PM_{10} emission in the Denver city area. A study along a high traffic area in Los Angeles showed paved road particulate matter emission to be about 30 % of the total PM_{10} (Venkatram et al., 1999). The emission inventory of primary PM_{10} for Switzerland for the year 1997 suggested road traffic to be the dominant source of national PM_{10} emission, contributing as high as 51 %. Emissions from industry, agriculture/forestry and domestic sector contributed 28.5 %, 16.7 % and 3.7 % respectively (Heldstab et al., 1999). The spatial average values of road traffic PM_{10} contribution in Switzerland varied from 30 % (equivalent to 6 μg m^{-3} of annual mean) at rural sites to 46 % (equivalent to 14 μg m^{-3} of annual mean) at urban town centres with maximum value of 60 % at some urban sites (Heldstab et al., 1999). In the UK, road transport was the largest single source of national emissions of $PM_{2.5}$ in 1999, being responsible for 28 kilotonnes (kt) out of the total emissions of about 109 kt. Of these, diesel vehicles produced about 22 kt (DEFRA, 2003). In London, road traffic contributes 77 % to total PM_{10} emissions and in the UK, this contribution is 25 % (Buckingham et al., 1997). The UK Department of the Environment, Food and Rural Affairs reports these figures as 78 % and 26 % respectively (DEFRA, 2000). The Airborne Particles Expert Group (APEG) report shows that, in city centres, traffic contributes typically 30 % to 40 % to the average annual PM_{10} concentration (APEG, 1999; DEFRA, 2000). Findings by Barrowcliffe et al. (2002) show that the traffic on principal roads and primary routes in the UK urban areas does indeed cause concentrations of $PM_{2.5}$ to be elevated appreciably above urban background concentrations. They reported that, broadly speaking, a doubling of traffic from 50,000 vehicles per day to 100,000 vehicles per day implies an increase of 15 μg m^{-3} in PM_{10} concentration at the kerbside. Limited data indicate roadside elevation of PM_{10} of about 30 % above the background concentrations near a major highway (AQEG, 2004). Average roadside increments of PM_{10} at heavily trafficked sites in London and Birmingham were between 8.7 μg m^{-3} and 15.6 μg m^{-3} (AQEG,

2004). Road dust has been reported as a major local source for exceedance of PM_{10} limit values in Sweden. PM_{10} concentrations reaching $400\,\mu g\ m^{-3}$ to $500\,\mu g\ m^{-3}$ were recorded at the kerbsides in Stockholm, Göteborg and Umeå (Johansson et al., 2004). Studies by Manoli et al. (2002) at Thessaloniki, Greece show that traffic contributes 38 % to fine-sized aerosol ($<3\,\mu m$) in urban air concentration. An Asian Development Bank (ADB) assisted study attributes two-third of the suspended particulate matter emission in Metro Manila to diesel powered jeepneys, taxis and buses (ADB, 1992). Dorsey et al. (2002) have found a positive correlation between traffic flow and aerosol fluxes above the city of Edinburgh. Review by Gertler et al. (2000) gives an assessment of contribution of mobile sources to PM_{10} and $PM_{2.5}$ in several US cities and highlights the importance of traffic contribution to urban particulate matter level. Several other studies have reported traffic as an important contributor to airborne particulate matter, especially in urban areas (AQEG, 2004; Benitez et al., 2001; D'Alessandro et al., 2003; Harrison et al., 1997b).

2.4 Increase in importance of traffic-induced resuspension

The contributions of traffic to urban air particulate matter level may either be from exhaust emission or from resuspension of road dust caused by passing traffic. Exhaust emissions are known reasonably well, although data on emissions from heavy-duty vehicles are still based on relatively few measurements (AQEG, 2004). With introduction of cleaner fuels (e.g, fuel quality improvement with regard to sulphur and aromatic content (Ntziachristos and Samaras, 2003), unleaded petrol replacing leaded petrol[2] and use of Compressed Natural Gas (CNG) instead of petrol and diesel) and better emission control technology (e.g., use of particulate matter filters and catalytic converters), exhaust emissions have decreased appreciably in spite of an overall increase in traffic (Goodwin et al., 1999; Kathuria, 2005). In the UK, emissions of PM_{10} from road transport have decreased considerably because of tighter EU vehicle emission and

[2]General marketing of leaded petrol has been banned from 1 January 2000.

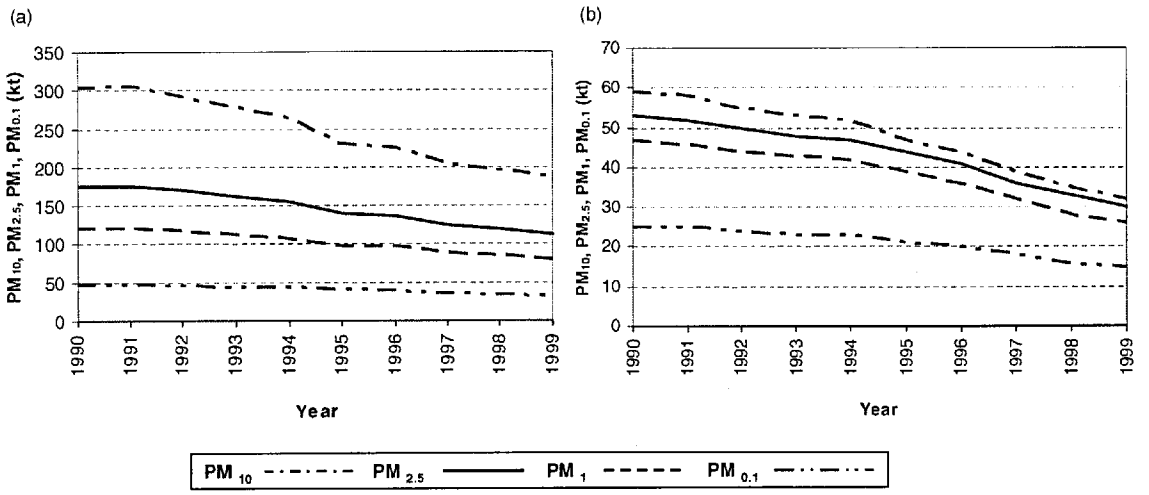


Fig. 2.1: UK emissions of particulate matter (1990–1999): (a) Particulate matter emissions from all sources; and (b) Particulate matter emissions from road transport (Goodwin et al., 1999).

fuel quality standards, known as "Euro standards", despite an overall increase in traffic of 15 % since 1990 (DEFRA, 2000, 2003).

Fig. 2.1a shows, between 1990 and 1999, UK emissions of PM_{10} from all sources fell by 39 % from 304 kt to 187 kt, emissions of $PM_{2.5}$ fell by 36 % from 176 kt to 112 kt, emissions of PM_1 fell by 33 % from 121 kt to 81 kt and emissions of $PM_{0.1}$ fell by 33 % from 48 kt to 32 kt. Corresponding figures for road transport are 46 %, 43 %, 45 % and 40 % (Fig. 2.1b). In all size ranges the decrease of emissions due to road transport was more than the corresponding decrease of emissions from all sources, indicating effect of tighter legislation, better technology and cleaner fuel on emissions from road transport. Less decrease in smaller size fraction is due to increased contribution of traffic exhaust emissions to the finer particles[3]. In 1999, road traffic accounted for 20 % of PM_{10} emissions, 27 % of $PM_{2.5}$ emissions, 32 % of PM_1 emissions and 47 % of UK $PM_{0.1}$ emissions (Fig. 2.2). These figures, based on analysis of data from the UK National Atmospheric Emission Inventory (NAEI), for 2001 are 18 %, 28 %, 35 % and

[3]In general, combustion processes such as in vehicles emit a higher proportion of $PM_{2.5}$ than mechanical sources such as quarrying, construction activities, fugitive road dust, etc. which produce more PM_{coarse} (Abu-Allaban et al., 2003b; Chow et al., 1996a; Dockery et al., 1993; Goodwin et al., 1999; Pope III, 2000; Wei et al., 1999). Therefore, exhaust aerosols deposit very slowly (Seinfeld and Pandis, 1998) and mechanically generated particles, because of the gravitational sedimentation/impaction, deposit more readily

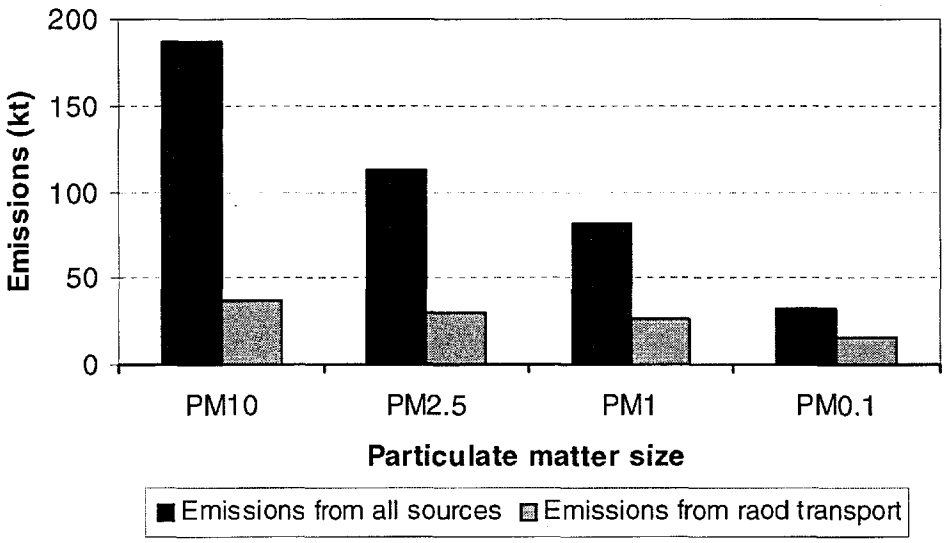


Fig. 2.2: Contribution of road transport emissions to UK emissions of particulate matter in 1999 (Goodwin et al., 1999).

48 % respectively (AQEG, 2004).

Table 2.2 summarises the NAEI estimates of contribution of different source sectors of transport to different size fractions of particulate matter in the UK atmosphere. Clearly, the contribution of petrol and diesel vehicles increases as particle size decreases. The maximum permitted emissions of particles from a new diesel car meeting Euro III standards, applicable to all new vehicles sold from 1 January 2001, are five times less than maximum permitted emissions from one that meets pre-Euro I standard. Euro IV standards, which will be mandatory for new diesel vehicles from 1 January 2006 (Euro V for heavy duty vehicles), will lead to further reduction of emissions of partic-

Table 2.2: Contribution, in different size ranges, of various source sectors of transport to UK emission of particulate matter in 2001 (AQEG, 2004)

	% of the UK emissions			
Source sector	PM_{10}	$PM_{2.5}$	PM_1	$PM_{0.1}$
Road transport - petrol	1.7	2.4	2.9	3.2
Road transport - diesel	15.2	24.1	31.4	44.5
Road transport - brake/tyre wear	4.9	4.8	1.2	2.3
Off-road vehicles	4.0	5.5	6.8	3.6
Other transport	1.0	1.7	2.1	0.9

ulate matter (AQEG, 2004). Some diesel car manufacturers are introducing particle filters to their latest models, thereby effectively eliminating particulate matter emissions from these vehicles. Under Auto-Oil programme directives, from the year 2005, almost all heavy-duty diesel vehicles are effectively required to be fitted with particulate matter traps which can reduce PM_{10} emissions by up to 90 % relative to emissions standards of year 2000. The sulphur content of the diesel which influences the amount of particles emitted by vehicles, has also been progressively reduced. All diesel vehicles and most petrol vehicles now meet the 2005 limit of 50 parts per million (ppm) compared to maximum sulphur content of 2000 ppm, prior to 1996 (DEFRA, 2000, 2003). The UK urban exhaust emissions have declined by 51 % from 1990 to 2000 and by 2010 and 2020 are predicted to decrease by 44 % and 59 % of the 2000 levels respectively (AQEG, 2004).

While exhaust emission shows a downward trend with time, the same is not the case with resuspension. Goodwin et al. (1999) have estimated that, between 1990 and 1999, resuspension of PM_{10} from the UK road transport has increased by 12.5 %, from 16.8 kt in 1990 to 18.9 kt in 1999. In London Brent site, during 1990-2002 period, particulate iron shows an upward trend of 4 % per year indicating steady rise of resuspended particulate matter due to traffic (AQEG, 2004). Its origin can be brake wear, corrosion products of vehicles and, to a lesser extent, soil; although its relative amounts of direct emission and resuspension from the road surface are not known (Harrison et al., 2003, 2004b). In any case, because the coarse component is associated with resuspension of road dust by traffic (Chan et al., 1997; Harrison et al., 2001; Johansson et al., 2004; Manoli et al., 2002; Thurston and Spengler, 1985), and the majority of it is generated along roads, it is likely to increase in future as traffic flow increases (Harrison et al., 2004a). Therefore, exhaust emission control measures have little effect on compliance of present and future air quality standards based on PM_{10} metric, especially in urban areas. While control of exhaust emission reduces the emissions of $PM_{2.5}$, emissions of PM_{coarse}, which are primarily produced from resuspension of soil and road surface dust by traffic-induced turbulence (Abu-Allaban et al., 2003a; AQEG, 2004; Artíñano et al., 2004; Braaten et al., 1990; Hall, 1989; Harrison et al., 2001; Kristensson et al., 2004;

Kulmala et al., 1986; Thurston and Spengler, 1985) as well as perhaps by the wind (Triantafyllou et al., 2002), remain uncontrolled (Harrison et al., 2004a). Therefore, measures to control resuspension are required as well. If the entrainment of dust from road surfaces is either a comparable or greater source than exhaust emission, which has been reported in many cases (Abu-Allaban et al., 2003b; APEG, 1999; AQEG, 2004; Ball and Caswell, 1983; DEFRA, 2000; Düring et al., 2002a; Johansson et al., 2004; Schulze, 2001), it will cause more areas to fail in complying with the air quality objectives. It is therefore important to investigate the contribution of traffic-induced resuspension to urban PM_{10} so that compliance with air quality standards can be achieved without concentrating efforts only on reducing exhaust emission, but by developing strategies to reduce resuspension. Particulate matter emitted from tailpipes should not be blamed in isolation when resuspended dust is equally responsible for failure of urban areas to meet air quality standards (Colvile et al., 2001).

2.5 Traffic-induced resuspension: the evidence

Removal of material, which can vary from a very small amount to about 40% depending on the vehicle type and material on road and environmental parameters, from roads by vehicle movement is visually observed and experimentally established by several researchers (Nicholson et al., 1989; Nicholson and Branson, 1990). It is probable that some of these are redeposited to road and some are deposited at kerb and hence can get resuspended in vehicle wakes. Under influence of local meteorology, these may get redistributed throughout a city network.

Using London as a case study, Ball and Caswell (1983) identified that while exhaust emissions from diesel vehicles are the main source of dark carbonaceous, possibly oily, deposits, the lighter, predominantly siliceous material is re-entrained from road surfaces by passing vehicles. Kittelson (1998) has reported that both vehicle exhaust particles and resuspended road dust are the principal sources of primary particulate matter in the urban environment. Road dust has been reported as a major local source for exceedance of PM_{10} limit values in Sweden (Johansson et al., 2004). Jaecker-Voirol and Pelt (2000)

reported resuspension by traffic contributing 3 to 7 times more PM_{10} than the exhaust gas emissions. A study in several European cities indicates contribution of road dust to PM_{10} between $9\,\mu g\ m^{-3}$ and $24\,\mu g\ m^{-3}$ in Sweden, $6\,\mu g\ m^{-3}$ in Spain and between $1\,\mu g\ m^{-3}$ and $5\,\mu g\ m^{-3}$ in other cities such as Berlin, London, Birmingham, Vienna and Bern (Querol et al., 2004). For $PM_{2.5}$ these values are between $2\,\mu g\ m^{-3}$ and $3\,\mu g\ m^{-3}$ in Sweden and Spain and $0.5\,\mu g\ m^{-3}$ in other cities. The higher resuspension in southern European Union (EU) countries is attributed to resuspension of soil during dry conditions and the higher proportion of resuspension in PM_{10} than $PM_{2.5}$ is attributed to easier resuspendibility of PM_{coarse}. High road dust emission in Sweden is due to use of studded tyres, frequent sanding and salting of roads during the winter and spring. Using NO_x as tracer for kerbside particulate matter emission, Johansson et al. (2004), for roads of Sweden, estimated typical annual mean values for $PM_{2.5}$ at rural background, urban background and kerbside as $5\,\mu g\ m^{-3}$ to $10\,\mu g\ m^{-3}$, $6\,\mu g\ m^{-3}$ to $11\,\mu g\ m^{-3}$ and $10\,\mu g\ m^{-3}$ to $15\,\mu g\ m^{-3}$ respectively. For PM_{10} these values were $7\,\mu g\ m^{-3}$ to $13\,\mu g\ m^{-3}$, $12\,\mu g\ m^{-3}$ to $18\,\mu g\ m^{-3}$ and $20\,\mu g\ m^{-3}$ to $30\,\mu g\ m^{-3}$, making the $PM_{2.5}/PM_{10}$ ratios approximately 0.8, 0.6 to 0.7 and 0.4 to 0.6. This decreasing ratio indicates the increasing importance of local traffic-induced resuspension of PM_{coarse}. In this case resuspension of road dust gave an additional kerbside concentration of $12\,\mu g\ m^{-3}$ to $15\,\mu g\ m^{-3}$.

It has been found in Thessaloniki, Greece, that while road dust contributes 28 % to the fine fraction ($< 3\,\mu m$) of particulate matter in urban air, it is a dominant source, contributing as much as 57 %, to the coarse fraction (between $3\,\mu m$ and $10\,\mu m$) of airborne particles (Manoli et al., 2002). Therefore it indicates PM_{coarse} is more easily resuspended than $PM_{2.5}$. Johansson et al. (2004) reported total non-exhaust emission factor for PM_{10} 9 times larger than PM_{10} exhaust emission factor. However, the same study found that in case of $PM_{2.5}$ non-exhaust emission factor is equal to the exhaust emission factor. This shows PM_{coarse} dominates the non-exhaust emission, largely composed of resuspended road dust. Gehrig and Buchmann (2003) reported the long-term average of the $PM_{2.5}/PM_{10}$ ratios of the daily values at kerbside site of Bern to be 0.59, significantly lower than 0.75, reported from other locations in Switzerland. This

indicates strong influence of coarse particles from traffic-induced resuspension on daily PM_{10} level in Bern.

Using a CMB receptor model, Chow et al. (1996b) estimated that in Santa Barbara County, California, the road dust accounted for 25 % to 27 % of PM_{10} mass, more than the contribution from marine aerosols (18 % to 23 %) and comparable with exhaust emissions (30 % to 42 %). Studies showing chemical and mineralogical signatures of road dust in atmospheric particulate matter have confirmed the presence of road dust in air (Abu-Allaban et al., 2003b; Harrison et al., 1997b; Janssen et al., 1997; Jansz, 2002; Jansz et al., 2004; Manoli et al., 2002; Schneider et al., 2004). While assessing source contribution to inhalable particles in metropolitan Boston, Thurston and Spengler (1985) found that a high percentage of coarse materials were of crustal origin and indicated coarse road dust suspended by city traffic as the source. A further meteorological analysis on the same data set indicated that the coarse component correlated with the effects of precipitation and season while the fine fractions did not (Thurston and Spengler, 1985). This indicated that exhaust emission was not the source of coarse fractions; resuspended road dust was the most plausible source. In a street canyon measurement at Manchester, Longley et al. (2004) reported that coarser aerosols were behaving differently from the finer ones with concentrations of coarse aerosols increasing with wind speed. They indicated the wind-driven resuspension of road dust as the possible source. Harrison et al. (1997a) found that PM_{coarse} at the roadside air is predominantly derived from natural and resuspended road dust. The use of chemical composition data has shown that while the roadside increment in $PM_{2.5}$ is comprised almost wholly of elemental carbon (EC) and organic carbon (OC) derived from exhaust emissions, increment in PM_{coarse} is contributed from a number of sources, with resuspended particulate matter being the most important (AQEG, 2004; Barrowcliffe et al., 2002; Schneider et al., 2004). Janssen et al. (1997) reported, from studies involving two cities in the Netherlands, that levels of Fe and Si, elements associated with mainly natural sources, are higher at the kerbside of a busy road than the nearby background location. Further, the amount of these elements were significantly higher in PM_{10} than in $PM_{2.5}$ in roadside air. This indicates higher contribution of traffic-induced resus-

pended road dust to PM_{coarse} than $PM_{2.5}$. Composition of particles measured at an urban background site at London showed the abundance of resuspended mineral matter and sea salt (sodium chloride) in PM_{coarse} (Harrison, 2004). The emission inventory in 1988 for the South Coast Air Basin, California attributed 73 % of fugitive dust emissions to resuspension from paved roads, 6 % to resuspension from unpaved roads, and 20% to construction and demolition (Chow et al., 1990). Studies in Vienna and Illmitz showed that, at traffic sites, EC, OC and Secondary Inorganic Aerosols (SIA) (nitrate, sulphate and ammonium are summarised as Secondary Inorganic Aerosols) each contributed 20 % to PM_{10} and 30 % was contributed by resuspension of road dust, dominated by PM_{coarse} and enriched with Fe, Ca, Mg, Na, chlorides and other compounds that form soil (Schneider et al., 2004). This study showed that on 10 December 2001 when Vienna had experienced an adverse dispersion condition due to a low stable inversion at ground level with very low mixing heights and small wind velocities, strong re-suspension by early morning heavy-duty vehicle (HDV) traffic resulted in resuspension of road dust to contribute 120 μg m^{-3} of the total 182 μg m^{-3} PM_{10}. Reconstruction of major components from the observed elemental composition by van der Gon and Visschedijk (2004) indicated that, in the Netherlands, the average contribution of soil material to PM_{coarse} varied from 2.5 μg m^{-3} at coastal locations to 6.5 μg m^{-3} in inland locations. Alastuey et al. (2004) reported road dust contribution of 6.5 μg m^{-3} (45 %) to kerb side PM_{10} level in Spain, the proportion of which decreased to 2 μg m^{-3} (20 %) in case of $PM_{2.5}$. Chemical characterisation of road side PM_{10} sample in Madrid shows that proportion of materials of crustal origin is high (38 %) in PM_{10} and less (20 %) in $PM_{2.5}$ (Artíñano et al., 2004).

The study at Pasadena, California compared the chemical signature of road dust to tyre tread debris, vehicular exhaust and vegetative detritus and estimated their contribution, by mass, about 1.6 %, 7.6 % and 2.2 % respectively, to the fine fraction (diameter $< 2\,\mu$m) of road dust (Rogge et al., 1993). Scanning electron microscope (SEM) examination by Abu-Allaban et al. (2003b) revealed abundance of silicon-aluminium and silicon rich particles in PM_{coarse} which are indicative of crustal material, such as soil. Detection of minor to trace amounts of iron-rich and zinc species from most of the

samples from different roadside locations indicated a common source of brake and tyre wear (Abu-Allaban et al., 2003b). Using CMB receptor modelling, Abu-Allaban et al. (2003b) have reported, for eight locations and four driving conditions (e.g., low-speed light-duty traffic, high-speed light-duty traffic, heavy-duty traffic and traffic causing significant brake wear) at North Carolina, USA, road dust emission to be as high as 100 (the average figure is 10) times of tailpipe emission. They estimated tailpipe emission, emissions with vegetative origins, brake and tyre wear emissions and resuspension of road dust to be, by mass, between $<5\,\%$ and $77\,\%$ (majority of them are between $5\,\%$ and $40\,\%$), $0\,\%$ and $40\,\%$ (majority of them are between $5\,\%$ and 20%), $0\,\%$ and $20\,\%$ (majority of them are less than 5%), $4.5\,\%$ and $96.7\,\%$ (majority of them are close to $40\,\%$) respectively. Goodwin et al. (1999) have reported contribution of exhaust emission and resuspension from road transport to PM_{10} emissions in UK in 1999 as 32 kt and 18.9 kt respectively, thereby indicating resuspension can be as high as $60\,\%$ of exhaust emission. A study in Berlin attributed additional street concentration to $52\,\%$ mineral components (mostly resuspension and abrasion), $7\,\%$ tyre wear and $41\,\%$ exhaust emission (Düring et al., 2002a). Harrison et al. (1997b) reported large differences between the cities in source contributions for PM_{10} and total suspended particulate matter (TSP), with soil dust estimated to contribute $62\,\%$ of TSP in the atmosphere of Lahore (Pakistan), but contributing much less in Birmingham (U.K.) and Coimbra (Portugal) where road traffic emissions comprise a substantial percentage of the total. Kim et al. (1990), while investigating the contribution of tyre treads to particulate matter level in urban air, showed evidence of traffic-induced scattering of deposited particulate matter near road at a height of 16 m above ground.

Studies in the US and UK show a strong correlation between $PM_{2.5}$ and PM_{coarse} concentrations with a higher proportion of PM_{coarse} in the dryer months (AQEG, 2004; Chow et al., 1994; Harrison et al., 1997a). The relative contribution of $PM_{2.5}$ and PM_{coarse} to the PM_{10} is approximately in the ratio of 2:1 although it varies from site to site and from season to season (Barrowcliffe et al., 2002). The relationship for the UK urban atmosphere, derived from the hourly measurement data from 1994 to 1997 (Harrison et al., 2001), is presented in Table 2.3.

Table 2.3: Relationship of $PM_{2.5}$ and PM_{coarse} with PM_{10} in urban UK atmosphere (Harrison et al., 2001)

Season	$PM_{2.5}$	PM_{coarse}
Autumn	$PM_{2.5} = 0.78\ PM_{10} - 2.57$	$PM_{coarse} = 0.22\ PM_{10} + 2.58$
Winter	$PM_{2.5} = 0.72\ PM_{10} - 0.72$	$PM_{coarse} = 0.27\ PM_{10} + 0.74$
Spring	$PM_{2.5} = 0.69\ PM_{10} - 0.87$	$PM_{coarse} = 0.31\ PM_{10} + 0.87$
Summer	$PM_{2.5} = 0.47\ PM_{10} + 2.54$	$PM_{coarse} = 0.53\ PM_{10} - 2.55$

A street canyon study by Harrison et al. (2001) shows that $PM_{2.5}$ is diluted with increasing wind speed. However, a component of PM_{coarse} increases with wind speed and hence can be attributed to wind-induced resuspension. Lin et al. (1999) observed similar effect of wind speed on particulate matter resuspension in Chicago, i.e., the low-speed winds raised the $PM_{2.5}$, whereas high-speed winds raised the PM_{coarse}. Therefore, coarse fractions are expected to be more easily resuspended by the vehicle-induced turbulence, because vehicle-induced turbulence has a similar effect to high-speed wind. Indeed, studies in London and Birmingham show that PM_{coarse} increment is closely related to total traffic volume when site dispersion characteristics are taken into account (Ball and Caswell, 1983). Easier resuspendibility of large particles than smaller ones has also been reported from earlier studies (Corn and Stein, 1965; Garland, 1983; Nicholson, 1993). Sehmel (1976a) has reported association of finer particles with larger host particles such as soil, for their easy resuspendibility. Abu-Allaban et al. (2003b) reported that tailpipe emissions and emissions having vegetative origin mainly contributed to $PM_{2.5}$ where as emissions of road dust and brake and tyre wear material dominated PM_{10}. For different vehicle categories (Light-duty trucks, cars, heavy-duty diesel vehicles and construction vehicles), Abu-Allaban et al. (2003a) reported significantly higher variability of PM_{10} than $PM_{2.5}$ with variations of speed and acceleration. This happened due to high variability of PM_{coarse}, which constitutes a major part of PM_{10} and therefore, they attributed it to the contribution from resuspended road dust. Harrison et al. (2001) observed appreciably higher PM_{coarse} concentration during weekdays than weekends, during day than night at urban sites and at the urban sites than rural sites and indicated anthropogenic activities (mainly traffic) to be its source. In

London, they observed a substantial increment of coarse particle concentrations at a kerbside location over the urban background station and therefore, suggested it as a strong indication of traffic-induced resusupension. Chan et al. (1997), from a study in Brisbane, implied resuspension of road dust to be one of the sources of higher PM_{coarse} concentration observed during weekdays than weekends.

In the absence of recent data on surface loadings in UK, Harrison et al. (2001) considered the surface silt loading on London roads to be $0.1\,kg\,km^{-1}$, an estimate reported in earlier studies (Ball and Caswell, 1983). With assumption of average road width to be 10 m, the average surface loading came to about $10^{-2}\,g\,m^{-2}$. With this value of surface loading and resuspension factors due to wind between $10^{-8}\,m^{-1}$ to $10^{-5}\,m^{-1}$, a range reported by Nicholson (1988), they found resuspended particulate matter concentration in the vicinity of the roadway of the order of $10^{-4}\,\mu g\,m^{-3}$ to $10^{-1}\,\mu g\,m^{-3}$. This value was much lower than the estimates they obtained by applying regression analysis on data obtained from 5 locations in the UK, suggesting resuspension near the road is not entirely due to wind action and therefore traffic-induced resuspension has contribution to this air concentration. Injection of road dust particles, especially the coarse fractions of road dust, into the atmosphere through resuspension by wind and vehicle-induced turbulence has also been reported by several other studies (AQEG, 2004; Artíñano et al., 2004; Braaten et al., 1990; Hall, 1989; Kleeman and Cass, 1988; Kristensson et al., 2004; Kulmala et al., 1986; Sapkota, 2002; Vermette et al., 1992; Watson et al., 1994).

2.6 Importance of study of processes behind resuspension

Technological measures aimed at reducing exhaust emissions are not generally effective in reducing resuspension of particulate matter. Unlike exhaust emission which is primarily dependent on fuel quality and engine characteristics, resuspension is governed by many other parameters such as meteorology (wind speed, humidity, temperature,

precipitation amount, duration between precipitation events and type of precipitation e.g., snow/rain), amount of dust available on road surface, type of street surface e.g., paved/unpaved, concrete/bitumen top, plain/blocks and street geometry e.g., building geometry on either side of the road. On a given street, the meteorological conditions are of greatest importance. A study in Zurich, which was principally based on chemical analyses of average daily values of dust samples and included receptor and dispersion modelling, indicated that while traffic load variables do indeed have a significant statistical influence on PM_{10} concentrations, it is far less than the influence of meteorological variables (Weigand et al., 2004). For example, if the street surface is wet due to precipitation, the amount of resuspension may be practically zero while dry conditions with high wind speed increases the particulate matter concentrations to its maximum. A Swedish study shows that resuspension is more pronounced during early spring season when the roads are dry, and before the winter salt and studded tyres have been removed (Johansson et al., 2004). The UK studies show distinct seasonality with summer maximum and winter minimum (QUARG, 1996). Measurements show that proportion of PM_{coarse} is greatest in the spring and summer, and least in the wet autumn and winter (Artíñano et al., 2004; Harrison et al., 1997a, 2001). This may be due to more effective wind-driven suspension of PM_{coarse} in the dryer months. Chan et al. (1997) have reported similar observation of dependence of resuspension on season where PM_{coarse} was on an average 58.5 % of PM_{10} in the summer season, the highest value in a year for Brisbane, Australia. Abu-Allaban et al. (2003b) have reported, for a study covering eight locations at North Carolina, USA, while tailpipe emissions for all sites were within 30 % of each other, the non-exhaust particles, mainly comprised of resuspended road dust, varied widely from site to site.

Therefore, control of resuspension can be achieved by reduction of the source of particulate matter on road surface and by better understanding of the processes behind resuspension. Limited studies on sources of PM_{coarse} are available (Harrison et al., 2001). Source reduction may include proper vehicle maintenance (i.e., less use of studded tyre), effective road maintenance (plain roads without potholes will reduce road abrasion as well as brake and tyre wear emission), covering loads carried by lor-

ries, paving of unpaved roads, vacuum sweeping and water flushing of road surface in hotspot areas, traffic flow alteration and road-charging schemes[4] (AQEG, 2004). While control of wind induced resuspension may be impossible, measures mentioned above may be adopted to control traffic-induced resuspension if estimates can be made for the amount of resuspension and therefore, a cost-benefit analysis for the implementation of one or more of the above measures can be carried out. However, very little is known about the processes that control the depletion and replenishment of particulate matter on road surface (Harrison, 2004). In conjunction with significant site-specific emissions from activities like construction, waste transfer, industrial activities, etc., at critical urban locations, undertaking pollution abatement strategies becomes more complex (AQEG, 2004; Bexley, 2001, 2003).

Therefore, as long as air quality standards are expressed in terms of PM_{10}, reduction of PM_{coarse} and therefore reduction of resuspended dust will be expected to gain focus to meet the standards. On the other hand, lack of data on resuspension and methods to estimate it have been observed by several researchers (AQEG, 2004; CAFE, 2004; Harrison et al., 2001; Karppinen et al., 2004). The relationship of traffic related (e.g., traffic volume, traffic speed, etc.,) and road related (e.g., road surface loading, road surface condition, etc.,) parameters with resuspension and the the process of resuspension itself is very poorly investigated. This thesis presents research carried out to enhance our understanding on the above areas with emphasis on the resuspension process.

2.7 Theory of resuspension

Particulate matter may become entrained in the atmosphere either due to the shearing action of the wind or as a consequence of mechanical actions such as road sweeping, vehicle movement and other anthropogenic activities. Bagnold (1941) and Chepil (1945, 1959) observed that resuspension by wind, like any other aerodynamic removal, is initiated by airflows with speed exceeding a threshold value. "Threshold wind speed" is the

[4]The focus of the scheme should be on reduction of traffic growth.

wind speed, a function of particle properties (e.g., size, density, etc.) as well as surface properties (e.g., smooth vs irregular surface, paved vs unpaved road, etc.), below which no obvious particle movement (i.e.,creeping[5]) is observed (Gillette, 1978, 1983; Sehmel, 1973; Xuan, 1999). Resuspension is also influenced by environmental parameters such as wind, solar radiation, precipitation and humidity in air (Sehmel, 1973). Harrison et al. (2001) reported threshold wind speeds between 1 m s^{-1} and 3 m s^{-1} (majority of the values are between 2 m s^{-1} and 3 m s^{-1}) for six UK sites comprising of rural, urban and kerbside sites. Initiation of particle movement takes place when threshold speed is exceeded i.e., shear stress on the particle exceeds the frictional retarding force (Punjrath and Heldman, 1972). However, below threshold wind speed, saltation[6] may take place (Gillette et al., 1974) and saltating material can remove kinetic energy from free wind and transfer this to surface particles, initiating resuspension (Nicholson, 1993). Beyond the threshold wind speed, windblown dust emissions may be lofted vertically to great heights above the ground by the sustained energy provided by the vertical component of the wind and consequently may be transported much further from the source of emissions than mechanically generated fugitive road dust emissions because the initial vertical energy associated with the moving vehicles that suspended the particulate matter is short-lived and unsustained. In the absence of strong wind with a large vertical component to sustain the vertical motion of these mechanically resuspended particles, they disperse mainly by the daytime turbulent eddies caused by the solar heating of the ground (Countess et al., 2001). Consequently, this dispersion process may not be strong enough for the particles to escape tall buildings in big cities, and therefore, may lead to cumulative build-up of particulate matter level in ambient air.

Threshold wind speed for resuspending particles decreases with increasing particle size (Bagnold, 1941; Chepil, 1945, 1959). This trend has been explained in terms of a

[5]Creeping is a type of particle movement where particles roll along the ground and it is associated with comparatively larger particles of the size 500 μm to 1000 μm (Newman et al., 1974).

[6]Saltation is a type of particle movement where particles ejected vertically into an airstream gain considerable horizontal momentum but fall back to the ground a little later due to gravitational pull. During this skipping action, particle may also rotate at high speeds. This is observed with particles of the size 100 μm to 500 μm, i.e., particles of size between those move by creeping and those move in suspension (Sehmel, 1980b).

balance of the aerodynamic and adhesive forces; both forces increase with increasing particle size but the aerodynamic force, that causes detachment of particles from the road surface, has a greater size dependence (Phillips, 1980) than adhesive force, the inter-surface molecular force with which small particles are held onto the surface (Reeks et al., 1988). Reeks et al. (1988) recognised the influence of the transfer of turbulent energy to a particle on a surface from the resuspending flow and proposed that such a transfer takes place through aerodynamic lift force which fluctuates randomly in time. There can also be transfer of momentum from moving particles to stationary ones (Garland, 1979; Punjrath and Heldman, 1972). Therefore, a particle in contact with a surface is in a constant state of vibration, building up energy until this energy is sufficient to detach the particle from the surface causing resuspension to take place. With this proposition, Reeks et al. (1988) showed that the threshold flow velocity for detachment of particles can be significantly lower than that based on a balance of aerodynamic and adhesive forces.

2.8 Estimation of resuspension

Two terms are commonly used to express and estimate resuspension:

Resuspension rate, Λ, is defined as the fraction of surface deposit removed per unit time (Nicholson, 1988, 1993; Slinn, 1978), i.e.,

$$\Lambda = \frac{R}{S} \tag{2.1}$$

Where,

$\Lambda =$ resuspension rate (s^{-1})

$R =$ resuspension flux (g m^{-2} s^{-1})

$S =$ surface material loading (g m^{-2})

Resuspension factor, K, is defined as the ratio of airborne concentration of the deposited material and surface loading of the material (Nicholson, 1988, 1993; Stewart,

1967), i.e.,

$$K = \frac{C}{S} \tag{2.2}$$

Where,

K= resuspension factor (m^{-1})

C= airborne concentration of surface material (g m^{-3})

S = surface material loading (g m^{-2})

2.8.1 Limitations in estimating resuspension

As reviewed by Nicholson (1988), choice of either K or Λ for expressing resuspension, is not free from limitations. Both Λ and K are related with surface loading, S. However, what proportion of S is available for resuspension is not known. It should ideally be either in terms of depth of surface deposit or in terms of percentage of surface loading (Nicholson, 1988; Sehmel, 1973). While the former seems to be more realistic in case of surfaces with high surface loading (e.g., unpaved surface with less traffic density), the latter may be more applicable to public roads with low surface loading and high traffic density (Nicholson, 1988). Nicholson (1993) has expressed a similar view and has added that complex surfaces such as surfaces with soil and vegetation canopies make this estimate further unreliable. Further C is a function of height and depending on meteorological condition, C should be measured at different heights for its proper relation with resuspension (Sehmel, 1973). For a scenario where local resuspension dominates air concentration, K may be useful and can be easily determined. However, downwind air concentration can not be predicted from K (Nicholson, 1988; Sehmel, 1973). Therefore, for such a scenario, Horst (1976) advocates the use of Λ. However, R and therefore Λ cannot be directly measured in field experiments and can only be deduced by fitting the observed data to a numerical model. Nicholson (1988) suggests that Λ is a better parameter in predicting airborne concentration in varying scenarios.

2.9 Resuspension by wind action

Earlier studies, both in the field (Sehmel, 1983) and in a wind tunnel (Garland, 1983), have reported that resuspension increases with wind speed and a power relationship best fits the relation.

$$\Lambda \text{ or } K \propto u^a \tag{2.3}$$

Where,

$\Lambda =$ resuspension rate (s^{-1})

$K =$ resuspension factor (m^{-1})

$u =$ wind speed ($m\ s^{-1}$)

$a =$ coefficient derived from power curve fit

Sehmel (1984) reported that values of a lie within 1 and 6, most of them being ≥ 3, typically 3 or 4. Although it is probable that type of road surface and environmental conditions will affect the resuspension process, Nicholson (1988) found too large scatter in reported measurements to verify it. His review indicated that K, for wind-induced resuspension processes in urban type locations, lie typically within the range of $10^{-8}\,m^{-1}$ to $10^{-5}\,m^{-1}$. Reviewing relevant loss process from contaminated land, Thiessen et al. (1999) reported wind-induced resuspension factors in the range of approximately $10^{-7}\,m^{-1}$. Sehmel (1980b, 1984), summarising a wide range of reported results, reported values of K between $10^{-10}\,m^{-1}$ and $10^{-4}\,m^{-1}$ for wind-induced resuspension without any mechanical disturbance and a further wide range of $10^{-10}\,m^{-1}$ to $10^{-2}\,m^{-1}$ with mechanical disturbances. Harrison et al. (2001) broadly agree with the above observation although they found the value of a to be less than 3, mainly in autumn and winter and concluded that it was probably due to lesser efficiency of resuspension in wetter climates associated with autumn and winter in UK.

2.10 Resuspension by traffic

Resuspension of particulate matter due to traffic comprises, primarily, of two processes i.e., one due to effects of "shearing stress of the tyres" and the other due to "induced

mechanical and thermal (because of gas exhaust temperature) turbulence" (Ruellan and Cachier, 2001; Sehmel, 1973, 1976b). Sehmel (1973, 1976b) used ZnS tracer on an asphalt road and measured resuspension due to a car and a truck for particles with mass median diameter of $8\,\mu$m. From the downwind concentration of tracer, he estimated the amount of loss of material from the road and therefore, the resuspension rate. Nicholson and Branson (1990) conducted an experiment which comprised of laying porous silica particles (nominal size varying between $5\,\mu$m and $20\,\mu$m, density $\approx 1000\,\mathrm{kg\ m^{-3}}$), attached with fluorescent dye, on road surface and allowing a medium size car to pass over it. In one case, the wheels of the car passed on either side of the test area so that resuspension, in this case, was presumed to be mainly from vehicle induced turbulence. In the other case, the wheels of the car passed over the test area so that resuspension was attributed to both induced turbulence and tyre stress. Findings of both the studies are summarised below.

1. Sehmel (1973, 1976b) reported that fraction resuspended per vehicle pass was in the range of 10^{-5} to 10^{-2} ($10^{-3}\,\%$ to $1\,\%$) on the same day of tracer application. This fraction decreased with time and was two to three orders of magnitude less after 30 days. He attributed this to the effect of weathering[7].

2. The resuspension rate was higher if the vehicle was driven through the tracer test area than on the adjacent lane. Sehmel (1973, 1976b) attributed the effect of tyre stress, in addition to turbulence, as the reason for higher resuspension. Tyre stress is absent when the vehicle is driven on adjacent lane. Nicholson and Branson (1990) observed little difference between resuspension due to tyre stress and resuspension due to induced turbulence, except in case of particle size of $4.2\,\mu$m size indicating that whatever may be the mechanism, particulate matter from road surface becomes resuspended quite readily.

3. The resuspension rate increased with the square of vehicle speed when the car was driven through tracer indicating resuspension rate is proportional to car-

[7]The attachment of tracer particles to host particles and therefore, becoming less readily resuspended is termed as weathering (Sehmel, 1973, 1976b).

generated turbulence (Sehmel, 1973, 1976b). At very low speed, under 20 km h^{-1}, Nicholson and Branson (1990) observed little variation in amounts of resuspension with varying vehicle speed and suggested that at such lower speeds, variation in amount of turbulence is nearly insignificant.

4. The amounts of material removed, only due to turbulence, after a single pass of the vehicle at a speed of 64 km h^{-1}, were about 60 % for particles of 20 μm size, 50 % for 12 μm and 9.5 μm size and 20 % for 4.2 μm size. The majority of the resuspension took place in the first 3 or 4 passes of the vehicle, the highest being on the very fast pass of the vehicle. After 10 passes of the vehicle, the amounts of resuspension, for the above size ranges, were 75 %, 55 % and 35 % respectively. It is suggested that particles that remain on road surface after a number of vehicle passes may only be confined to the road depressions and this is likely to be a small proportion of the total and will decrease quickly with increasing vehicle speed (Nicholson and Branson, 1990).

5. Bigger particles were more easily resuspended. This is because smaller particles have smaller surface areas on which shear stress act. However, unlike resuspension due to turbulence, when smaller particles are less easily resuspended, resuspension of smaller particulate matter due to tyre stress takes place more easily (Nicholson and Branson, 1990).

6. Resuspension by wind forces was minimal compared to the resuspension caused by traffic (Sehmel, 1973, 1976b).

7. Lager vehicles (e.g., truck) caused more turbulence than smaller ones (e.g., car) and therefore, more resuspension (Sehmel, 1973, 1976b).

A regression analysis on data from London Marylebone Road indicates that contribution of heavy-duty vehicles (HDVs) is about 15 times greater per vehicle than for light-duty vehicles (LDVs) for coarse particle concentrations (AQEG, 2004). As mentioned above the principal reason for this is the substantial turbulence induced by heavy-duty traffic that is responsible for a greater magnitude of resuspension of

particles from the road than the light-duty traffic. The multiple regression equation and its results are presented in Equation 2.4 and Table 2.4 respectively (AQEG, 2004).

$$\mathrm{PM}_{\mathrm{coarse}} = c_1\, n_{\mathrm{ldv}} + c_2\, n_{\mathrm{hdv}} + c_3 \tag{2.4}$$

Where,

n_{ldv} = hourly mean flow of LDVs by day of the week (v h^{-1})

n_{hdv} = hourly mean flow of HDVs by day of the week (v h^{-1})

c_1, c_2 and c_3 = constants to be derived from regression

Table 2.4: Results from the multiple regression (AQEG, 2004)

Constants	Estimated values	Standard error	t statistics
c_1	0.000850	0.000112	7.6
c_2	0.012306	0.000534	23.0
c_3	3.932492	0.299579	13.1

The study shows that concentration of PM_{10}, $\mathrm{PM}_{2.5}$, $\mathrm{PM}_{\mathrm{coarse}}$ and particle number concentration are all higher on weekdays than on Sundays despite there being little difference in overall traffic volume. This appears to be explained by the reduction in heavy-duty traffic on Sundays which caused less exhaust emission as well as less resuspension (AQEG, 2004). However, whether road dust emissions are dominated by the mechanical interaction between the tyres and road or by the turbulent shear applied to road by the moving vehicle are yet not established without doubt. Kuhns et al. (2001b) have expressed similar views.

For a wet paved road surface, "tyre spray" may be the principal mechanism of resuspension (Smith, 1970). Unlike resuspension from dry roads where particulate matter has more likelihood to get airborne, materials from wet surfaces will more likely be pushed by the vehicles, as an effect of tyre spray, onto the edge of the road or get spread, by adhering to tyres, along the road in the direction of the traffic. Nicholson and Branson (1990) estimated an average resuspension factor of $1 \times 10^{-7}\,\mathrm{m}^{-1}$ to $4 \times 10^{-7}\,\mathrm{m}^{-1}$ for rock-salt application on urban roads.

Resuspension of particulate matter also depends on nature of the surface. Dry surfaces favour resuspension. Meteorological parameters in general, and precipitation in particular, have a significant effect on the resuspension process (Düring and Lohmeyer, 2001; Düring et al., 2002a; Gámez et al., 2001a; Muleski et al., 2001; Myers, 2001; Nicholson and Branson, 1990; Rauterberg-Wulff, 2000; USEPA, 2002). The effect of duration and intensity of precipitation on resuspension process is discussed in Section 5.6.3.4.

2.11 Estimation of traffic-induced particle resuspension: the current state of science

The most common approach that is widely used to estimate traffic-induced particle resuspension is to determine an emission factor for certain traffic and road conditions. An "emission factor" is a representative value that attempts to relate the quantity of a pollutant released to the atmosphere with an activity associated with the release of that pollutant. These factors are usually expressed as the weight of pollutant divided by a unit weight, volume, distance, or duration of the activity emitting the pollutant (e.g., kilograms of particulate matter emitted per ton of coal burned, grams of particulate matter emitted per kilometer of vehicle travel). Such factors facilitate estimation of emissions from various sources of air pollution. In most cases, these factors are simply averages of all available data of acceptable quality, and are generally assumed to be representative of long-term averages for all facilities in the source category i.e., a population average (Painter, 1974; USEPA, 1995). The European Environment Agency (EEA) defines emission factor as the estimated average emission rate of a given pollutant for a given source, relative to units of activity (EEA, 1999). Although these emission factors represent the best available data at any point of time, they are subject to continuous change with time and these factors should be used only if local data cannot be obtained from specific sources in the area of concern (Painter, 1974). For traffic-related emissions, an emission factor is defined as the amount of material emit-

ted per unit distance of vehicle travel and is usually expressed in units of gram per kilometre of vehicle travel (g km^{-1} v^{-1}).

The UK National Atmospheric Emissions Inventory (NAEI) assigns a value of 0.04 kt per billion vehicle kilometers (equivalent to 0.04 g km^{-1} v^{-1}) for resuspension for the year 2000 (NAEI, 2003). While this database gives emission factors from road transport for different types of pollutants (e.g., PM_{10}, nitrogen oxides as NO_2 and N_2O, SO_2, CO, CO_2 as C, methane, non-methane VOC, benzene, etc.) in terms of vehicle types (e.g., motorcycles, cars, light goods vehicles (LGVs), heavy goods vehicles (HGVs) , buses and coaches, etc.) and also in terms of drive types (e.g., rural driving, urban driving, motorway driving, etc.), it does not give any such detailed estimate in case of emission factor for resuspension. This is precisely due to the fact that there has not been enough study on this area and therefore, the data are unavailable. This raises doubt about the correctness of the use of NAEI recommended emission factor in estimating contribution of resuspension in a mixed traffic condition.

To date, most of the studies undertaken to estimate emission factors due to traffic have essentially followed an empirical approach wherein regression analysis between emission, obtained from measured concentration, and other probable road and vehicle related parameters is carried out to derive a equation that can be used to quantify emission rate in units of mass per unit distance travelled by a vehicle. Extensive studies in this area have been carried out by the United States Environmental Protection Agency (USEPA, 1993c, 1995, 1997) and results are documented as "AP-42, Compilation of Air Pollutant Emission Factors" (hereafter referred as AP-42). In recent times, a substantial amount of work with regards to emission factors, especially vehicle related, has been carried out in the UK (NAEI, 2003) and by the EEA (EEA, 1999). Studies in Germany have tried to estimate, separately, the contribution of exhaust and non-exhaust emission (Düring et al., 2002a,b). However, AP-42 emission factors are widely used for the estimation of road traffic resuspension. Therefore a detailed discussion on this is presented below.

2.12 AP-42: its background and evolution

AP-42 was first published by the US Public Health Service in 1968. Since 1971 it has been published by the USEPA. Supplements to AP-42 are routinely published to update existing emission factors and AP-42 is periodically updated to add new emission source categories. Emissions from the road are discussed in two sections: emissions from "paved roads" and emissions from "unpaved roads". Until the fourth edition of AP-42, the paved road emission factor models, presented in Equations 2.5, 2.6, 2.7, 2.8 & 2.9 were developed with assumption that, depending on the surface loading characteristics, emission levels, traffic characteristics, and viable control options, paved roads, for the purpose of estimating particulate emissions, are of two distinct categories i.e., "public paved roads" and "industrial paved roads". While the mechanisms of particle deposition and resuspension are largely the same, public roads generally tend to have lower surface loading, higher traffic volume and lower proportion of heavy vehicles than industrial roads. However, a surface loading figure above or below which a particular road is characterised as an industrial or public road is not mentioned in the AP-42 document (USEPA, 1993c). This subjective distinction led to the confusion in applying a equation appropriate to a paved road. Some industries, especially iron and steel, have far less surface loading to be treated as public paved roads rather than industrial paved roads. So, in such a situation, applying the industrial paved road equation resulted in more emissions estimate than the actual. Therefore, in a major review in September 1997, section 11.2.5 on "public paved road" and section 11.2.6 on "industrial paved roads" were replaced with one section, named "paved roads" (USEPA, 1997). The fifth and the latest edition of AP-42 document which first came into effect in January 1995, has 15 chapters for emission factors for various types of sources. In chapter 13, entitled "miscellaneous sources", sections 13.2.1 and 13.2.2 deal with emission factors for "paved roads" and "unpaved roads" respectively (USEPA, 1993c, 1995, 1997).

2.12.1 AP-42 emission factor model for paved road emission

Because the formal establishment of PM_{10} as the primary standard in USA occurred in 1987 and the formulae presented below in Equations 2.5, 2.6, 2.7, 2.8 & 2.9 to estimate emission were developed prior to that, these have been referenced to other particle size ranges, such as TSP[8] and PM_{15}[9]. Drafted first in 1984, section 11.2.5 of AP-42 on fugitive emission included Equation 2.5 to estimate emissions from "public paved roads" (USEPA, 1993c):

$$e = k\left(\frac{sL}{0.5}\right)^{a} \tag{2.5}$$

Where,

- e = particulate matter emission factor (unit matching with unit of k, e.g., g km^{-1} v^{-1})
- k = an empirical constant (also called base emission factor). Its value depends on size of the particle and units used to express emission factor. When unit of e is g km^{-1} v^{-1}, k has values 1.02 for $PM_{2.5}$, 2.28 for PM_{10}, 2.54 for PM_{15} and 5.87 for TSP.
- sL = road surface silt loading, i.e., mass of particulate material of size $< 75\,\mu m$ in diameter per unit area of the road surface (g m^{-2})
- a = a dimensionless empirical constant with values 0.6 for $PM_{2.5}$, 0.8 for PM_{10}, 0.8 for PM_{15} and 0.9 for TSP

Published first in 1983 and slightly modified in Supplement B (1988) to the fourth edition of AP-42, section 11.2.6 provided 3 sets of equations for emission factors for "industrial paved roads" (USEPA, 1993c). For TSP, Equation 2.6 was used to estimate the emission factor.

$$e = 0.022\,I\left(\frac{4}{n}\right)\left(\frac{s}{10}\right)\left(\frac{L}{280}\right)\left(\frac{W}{2.7}\right)^{0.7} \tag{2.6}$$

[8]As measured by the standard high-volume air sampler, total suspended particle (TSP) consists of a relatively coarse particle size fraction varying roughly from 25 μm to 50 μm (USEPA, 1993c).

[9]PM_{15} is defined as particulate matter no greater than 15 μm, also referred to as inhalable particles (IP) (USEPA, 1993c).

Where,

e = particulate matter emission factor (kg km^{-1} v^{-1})

I = industrial augmentation factor (dimensionless). I accounts for higher emissions from industrial roads than from urban roads (varies from 1-7).

n = number of traffic lanes

s = silt fraction (%)

L = surface material loading across all traffic lane (kg km^{-1})

W = average vehicle weight (ton)

For particles smaller than TSP, Equation 2.7 was used (USEPA, 1993c).

$$e = k \left(\frac{sL}{12}\right)^{0.3} \tag{2.7}$$

Where,

e = particulate matter emission factor (kg km^{-1} v^{-1})

k = an empirical constant (also called base emission factor). Its value depends on size of the particle e.g., 0.081 for $PM_{2.5}$, 0.22 for PM_{10}. and 0.28 for PM_{15}

sL = road surface silt loading (g m^{-2})

However, light duty traffic (<4 tons) on heavily loaded surfaces (silt loading >15 g m^{-2}) was considered neither a case of industrial paved road nor a public paved road. So a separate single-valued emission factor was derived as follows (USEPA, 1993c):

$$e = 0.093 \text{ for } PM_{10} \tag{2.8}$$

$$e = 0.12 \text{ for } PM_{15} \tag{2.9}$$

Where,

e = particulate matter emission factor (kg km^{-1} v^{-1})

The latest EPA model for paved road emission, which was formulated after two different sections on "public paved roads" and "industrial paved roads" of the fourth edition of

AP-42 were replaced with a single section named "paved roads" in the fifth edition of AP-42 documentation, is as follows (Fitz and Bufalino, 2002; USEPA, 1997):

$$e = k \left(\frac{sL}{2}\right)^{0.65} \left(\frac{W}{3}\right)^{1.5} \tag{2.10}$$

Where,

e = particulate matter emission factor (unit matching with unit of k, e.g., g km^{-1} v^{-1})

k = a constant, (also termed as "base emission factor" for particle size range and units of interest) e.g., 1.1 g km^{-1} v^{-1} for $PM_{2.5}$ and 4.6 g km^{-1} v^{-1} for PM_{10}

sL = road surface silt loading (g m^{-2})

W = average weight of the vehicle of a fleet (tons)

In Equation 2.10 "default" values of 2 g m^{-2} and 3 ton are divided to achieve nondimensionalisation so that the unit of k matches the unit of e. The above equation is based on a regression analysis of numerous emission tests, including 65 tests for PM_{10}. The equation is supposed to produce good prediction, if applied within the range of source conditions that were tested in developing the equation, such as silt loading between 0.02 g m^{-2} and 400 g m^{-2}, mean vehicle weight between 1.8 tonnes and 38 tonnes and mean vehicle speed between 16 km h^{-1} and 88 km h^{-1}. Further, the model is only applicable to free flowing traffic, i.e., it can not predict expected higher emission due to stop-and-go traffic.

Paved road silt loadings are dependent upon traffic characteristics (speed, ADT[10], and fraction of heavy vehicles); road characteristics (curbs, number of lanes, parking lanes); local land use (agriculture, new residential construction) and regional/seasonal factors (snow/ice controls, wind blown dust). The range of sL values in the data base for normal conditions is 0.01 g m^{-2} to 1.0 g m^{-2} for high-ADT[11] roads and 0.054 g m^{-2} to 6.8 g m^{-2} for low-ADT roads. The recommended default values, where site-specific sL values cannot be obtained, are 0.1 g m^{-2} and 0.5 g m^{-2} in normal and worst-case

[10]ADT stands for "average daily traffic".

[11]High-ADT roads refer to roads with at least 5000 vehicles per day (USEPA, 1997).

conditions[12] for high-ADT roads and 0.4 g m^{-2} and 3.0 g m^{-2} in normal and worst-case conditions for low-ADT roads (USEPA, 1997).

2.12.1.1 Criticism of AP-42 model for paved road emission

In spite of several modifications to the AP-42 algorithm in last three decades, all have shared a few common features. All are developed through multiple linear regression over a large data set generated by experiments conducted primarily in US and therefore, AP-42 models are empirical in nature. Further, all have considered silt loading as an essential parameter controlling emission. During an initial phase of model development, it was observed that there exists a high degree of interdependence between silt loading, emission factors, and speed; and low degree of interdependence between (a) silt loading and weight and (b) weight and speed. This suggested either of the two combinations may be used to derive an emission factor model. However, it was also recognised that models developed using only the independent variables weight and speed did not vary substantially during a year and therefore, could not explain seasonal variation or predict higher emission levels known to occur after road sanding, etc. Thus models incorporating surface loading values as an independent variable were pursued because loading represents a reasonable means of introducing seasonal variability (USEPA, 1997).

Use of the silt loading

Silt loading (sL) is defined as the mass of silt size material present on unit area of the road surface. It is obtained by multiplying surface material loading (S) with silt fraction and is expressed in the same unit as of surface material loading i.e., g m^{-2}. Surface material loading (total road surface dust loading), S is defined as the mass of particulate material (irrespective of size) per unit area of road surface. S is estimated by collecting materials from a section of road surface by broom sweeping and vacuuming (USEPA, 1993a). Although dry vacuuming is usually adopted (No wet vacuuming data was used to formulate the AP-42 algorithm), recent studies show that

[12]Worst-case conditions are defined as conditions such as post-winter storm or areas with substantial mud/dirt carry-out (USEPA, 1997).

wet vacuuming is a more efficient method than dry vacuuming for collection of the road surface material (Deletic et al., 2000; Jansz, 2002). However, whether the extra amount of road material collected by wet vacuuming contributes to the emissions from the road is yet not verified.

Silt size refers to a size $\leq 75\ \mu$m in diameter. The silt fraction, expressed as percentage of the material of this size present in the total mass of the road dust, is determined by measuring the proportion of the loose dry surface dust that passes through a 200-mesh screen, using the American Society of Testing Materials (ASTM)-C-136 standard (USEPA, 1993b). However, why this size is important for emissions calculation is not known. Kuhns et al. (2001b) have expressed similar view. Whether the particle size cut-off higher or lower than 75 μm can contribute to emissions is not investigated.

Several authors have raised doubts over the physical basis of AP-42 assumption of silt loading as a important parameter that influences paved road emission. Kuhns et al. (2001a,b) estimated that, in Las Vegas, Nevada, roads with vehicle speeds greater than 45 miles per hour were 20 times cleaner than roads where vehicle speeds were less and roads with ADT greater than 10,000 per lane were 12 times cleaner than roads with lower average daily traffic, suggesting a negative correlation between silt loading and particulate matter concentration in air when either the speed of vehicle is high or traffic density is high. This may be due to high traffic density and high speed causing more resuspension of particulate matter and thereby making silt loading low and particulate matter concentration in air high. Earlier studies have reported that, besides other factors such as driving pattern, gradient and altitude, emissions increase with vehicle speed (Countess et al., 2001; Nicholson and Branson, 1990; Sehmel, 1973, 1976b; Sturm et al., 1996) and therefore, support the above argument. However, AP-42 emission factor for paved roads does not consider vehicle speed (nor does it consider traffic density) as a parameter since it provided no additional improvement to the formulation in the overall correlation. This is an example of potential flaw in the use of statistical evaluations to determine real world parameterisations of physical processes[13].

[13]In the latest modifications of emission factors for unpaved roads, the USEPA has included vehicle speed as a parameter. However, the performance of model after inclusion of speed is yet to be thoroughly evaluated.

AP-42 emission factor for paved roads assumes that paved road itself is generating appreciable amount of loose material, almost inexhaustible, so that it is the prime source of emission and hence the silt loading is important (Nicholson, 2001; Venkatram, 2001). This may be true for roads where local sources continuously supply material on to the road or where traffic density is so low that not much material is removed by it. However this may not be true for emissions from roads where the silt residence time is much less or zero such as roads with high traffic density and low surface loading in urban city centres (Goodwin et al., 1999; Nicholson, 2001). This could be explained in terms of either exhaust emission dominating paved road emission, if each of these is estimated separately, or immediate resuspension of material generated on road surface, if processes that generate material on the road surface are described, neither of which is considered in present AP-42 formulation. Further, AP-42 states that the silt loading reaches an equilibrium value without the addition of fresh material. Fitz and Bufalino (2002) argue that if equilibrium is attained, then the emission rate should go to zero, although this is not what the equation predicts and therefore they doubt its universal application unless the material is continuously replaced, a phenomenon which, they suggest, for most public roads is not likely. Venkatram (2000) has discussed almost a similar logical dilemma with silt loading. He argues that if silt loading affects emission it can not be a stable parameter that characterises road surface as is implied by the AP-42 model and if it is a stable parameter, it cannot affect the emission factor. Therefore he suggests an instantaneous silt loading, not a default silt loading, for a particular type of road should be considered for estimation of the resuspension. And to estimate the instantaneous silt loading, the mass-balance calculation is required that can only be achieved by knowing the different processes that contribute the silt loading and their likely estimate.

However, if the sources are really available that make a paved road behave like an unpaved road (e.g. construction activity on a slice of road, regular movement of vehicles from unpaved roads to paved roads, etc.), the logical dilemma discussed by Venkatram (2000) could be explained. Because, in such a situation, either the silt loading is changing or is in dynamic equilibrium. Gámez et al. (2001b) has shown that

although a AP-42 default silt loading is not able to account for observed roadside PM_{10} concentration, it could be explained by considering some processes (e.g., road abrasion) that might be generating material on paved road and therefore replenishing the silt loading. Therefore, the lack of inclusion of processes that can replenish the silt reservoir on the road surface in AP-42 algorithm limits the scientific basis and the understanding of paved road emission. However for both the cases it is instantaneous silt loading, not the universal or averaged value as is used in AP-42, that should provide estimation of emission factor for that slice of time and location. This argument is valid even if there exists dynamic equilibrium between silt inflow-outflow amount. Venkatram (2000) has argued along similar lines. However this is further complicated by the fact that if there exists a large influx of material which is not dependent on vehicle density, the material on the road surface may be temporarily high but will gradually reduce and may reach an equilibrium. But then up until equilibrium, studies have shown that the resuspension fraction decreases with increasing number of vehicle passes over the road and therefore is not constant (Nicholson and Branson, 1990). Therefore even though the emission factor will depend on traffic density, it may not be directly proportional to traffic density or vehicle distance travelled. Even the above argument can be extended to a case where material available for resuspension is generated only due to vehicle movement (i.e. abrasion, brake and tyre wear erosion, etc,) and therefore, is proportional to the vehicle density and the distance travelled. Because the same traffic subsequently is responsible for its resuspension, expressing emissions in terms of vehicle distance travelled may be true, because there is a material replenishment process and depletion process, both dependent on vehicle density and the distance they travelled i.e., vehicle distance travelled. However, the balance of material may vary and it may or may not attain equilibrium. Even if it achieves equilibrium, it may be temporary and a constant resuspension fraction may not be the right approach. Some authors have argued along the same lines and have suggested that material migration processes should be included for each different location when comparison is made (Nicholson, 2001; Venkatram, 2000).

Venkatram (2000) has shown that the AP-42 model for paved road is not reducing

to a model of an unpaved road even if the silt loading is considered as high as that of an unpaved road and therefore suggested that the formulation is theoretically unsustainable. This is the shortcoming of an empirical model. Had the AP-42 algorithm been developed with processes involved in emission described, it would have necessitated only the change of parameter values of the processes or inclusion/exclusion of processes for it to be applicable in case of an unpaved road.

Studies in Germany indicate that the proportion of exhaust emission to other emissions is about 50:50 (Düring et al., 2002a; Schulze, 2001). Abu-Allaban et al. (2003b) found that for places like freeway exits, where a vehicle slows rapidly while exiting and thus maximising occurrence of brake wear, brake wear emission is more than tailpipe emission. However, such a separate estimate of exhaust and non-exhaust emission is not possible with AP-42 formulation. Each of these emissions may have taken place due to different processes and therefore quantifying them through a single relation may not be appropriate. Separate estimate of exhaust and non-exhaust emissions is particularly important when we estimate possible particulate matter reductions due to precipitation if we consider independency of exhaust pipe emissions to precipitation[14].

Emission factors developed using the upwind-downwind method by default treat emissions as a continuous process, since a continuous line source dispersion model is back-calculated to derive the emission factor. For paved roads, this assumption, especially for multi-lane, high average daily traffic volume, is probably reasonable for periods other than the late evening/early morning where non-continuous situations exist. However, when the traffic flow is almost nil at late evening/early morning hours, it does not qualify for continuous source (Countess et al., 2001). Therefore a no emission case at these hours can not be accounted for by the emission factors developed considering fugitive dust emissions as continuous process. This again highlights the effect of absence of physical basis in an empirical model.

It is expected that silt loading decreases by street sweeping and therefore, paved

[14]Very recently, EPA has made some effort to calculate exhaust emission separately from other emissions such as brake and tyre wear and re-entrained dust for various vehicle speeds by using MOBILE 6.2 model (USEPA, 2003). It came out with a set of values for these emission factors corresponding to varying silt loading and varying vehicle speed and vehicle type. These are yet to be evaluated.

road emissions will also decrease proportionally. Fitz (1998) observed that although sweeping decreases the silt loading, there is little impact of it on PM_{10} emission. This is consistent with the observation from Bexley Council of London that suggest there is no measurable effect of sweeping on PM_{10} emission rate at Manor Road (Bexley, 2003). No relationship between street sweeping and ambient PM_{10} concentrations has also been reported from study at Reno, Nevada (Chow et al., 1990). Several other US studies have reported either no reduction or statistically insignificant reduction of particulate matter after street sweeping on public roads (Seton, 1983a,b; PEDCo Environmental Inc., 1981a,b). This lack of relationship could be due to emissions created during sweeping cancelling the expected benefit of sweeping, particulate matter loading getting rapidly replaced after sweeping to an equilibrium level or may be that the decrease of PM_{10} emission rate due to sweeping is too low to be detected by measurement (Fitz and Bufalino, 2002). This again indicates silt loading is not a reliable parameter that explains paved road emission.

Therefore, the main criticism is that the AP-42 emission factor model is a purely statistical model which can give average results over an area with a large number of roads, but fails to predict the emission factor at a local scale, e.g., for a certain road or section of road (Countess et al., 2001; Venkatram et al., 1999; Venkatram, 2000). This model does not have a mechanistic approach that explains the processes involved in the emission and its formulation is highly dependent on the data set used to derive it (Countess et al., 2001; Venkatram, 2000). However, even using the same data sets that were used to derive the present AP-42 emission factor model (Equation 2.10), Venkatram (2000) showed that for a single predicted emission factor value of 0.5 g km^{-1} v^{-1}, the observed emission factor values varied from 0.1 g km^{-1} v^{-1} to 10 g km^{-1} v^{-1}, i.e., by a factor, as high as, 20 and therefore, concluded that the formulation is lacking in scientific basis (Venkatram, 2000, 2001).

Field applications have shown how silt loading is a less important parameter than is considered for AP-42 emission factor model. Zimmer et al. (1992), from their study at Denver metropolitan area, reported that silt loading only accounts for as little as 20 % of the variation of emission factor and its implied relationship with emission factor

is different from that of Equation 2.10. Kantamaneni et al. (1996) measured PM_{10} emissions from paved roads in Spokane, Washington. The results showed that, for a silt loading of 3.25 g m^{-2} with little variation for a period of 2 months, Equation 2.10 predicted a constant emission factor of 10.5 g km^{-1} v^{-1} when the actual measurements were within a range of 0.47 g km^{-1} v^{-1} to 1.71 g km^{-1} v^{-1} and therefore, reported no correlation of silt loading with emission factor. Using on-board particle sensors, a technique recommended by Venkatram et al. (1999) as a better one than upwind-downwind method of emission estimation, Fitz and Bufalino (2002) estimated emission factors for different types of road (i.e., freeways, arterial, collector and local)[15] in southern California to vary between 0.06 g km^{-1} v^{-1} and 0.13 g km^{-1} v^{-1}. In contrast AP-42 algorithm estimation for this case varies between 0.08 g km^{-1} v^{-1} and 0.53 g km^{-1} v^{-1}, a factor of 4 higher in case of upper limit. In a study to measure and model the PM_{10} emissions for a similar road and traffic condition in southern California, Venkatram and Fitz (1998) observed silt loadings generally lower than those suggested by default values in AP-42 document. They estimated emission factor varying between 0.1 g km^{-1} v^{-1} and 3.0 g km^{-1} v^{-1} and found little correlation between the measured emission factors and those predicted by AP-42. In the UK environment, AP-42 emission factor results in improbably high emissions (Buckingham et al., 1997) and therefore, is considered not to be applicable in the UK environment (APEG, 1999). These studies, therefore, indicate, in addition to lack of scientific basis, silt loading as a parameter is unreliable in real field measurements.

Use of the average vehicle weight

The use of average vehicle weight in the Equation 2.10 limits the calculation to only one emission factor that represents the whole fleet. The equation is not intended to be used to calculate a separate emission factor for each vehicle weight class. For example, if 99 % of traffic on the road are 2 ton cars while the remaining 1 % consists of 20 tonne trucks, then the W to be used in Equation 2.10 is 2.2 tonne. However, as

[15]Roadways are classified as follows: Freeway(four or more lanes with at least 150,000 cars in a day), Arterial(three, four or more lanes with 10,000-150,000 cars in a day), Collector(two lanes with 500-10,000 cars in a day) and Local(two lanes with 500 or fewer cars in a day).

discussed by Countess (2001), the fleet-averaged W may sometimes lead to a very wrong conclusion. He considered a case of one car weighing 3 tonnes and one loaded truck weighing 39 tonnes. Using the average weight of these two vehicles in Equation 2.10, the value of e for PM_{10} comes as 12.2 g km^{-1} v^{-1} i.e., 24.4 g km^{-1} for both vehicles. When it is calculated separately for car and truck and then added, the result is 31.4 g km^{-1} with 30.8 g km^{-1} contributed by the truck. By averaging the vehicle weights, the total emission rate is less than that of the truck by itself (in this case by about 25 %). Obviously, averaging the vehicle weights does not give an accurate account of the true situation.

2.13 Summary

The review indicates that the silt loading, the main variable in the widely used AP-42 model, does not seem to be a good parameter for emission estimation, both from theoretical and experimental viewpoints. For emission estimates to be credible, a model should include different sources (e.g., vehicle exhaust, road abrasion/pavement wear, brake and tyre wear debris, spillage from goods vehicle, atmospheric deposition, significant local source and some seasonal activities like winter sanding and salting) and different anthropogenic and natural processes through which particulate matter enters to a road/vehicle environment, moves within various compartments of the environment and ultimately escapes from it i.e., emission factors should be based on a physical model that accounts for mass-balance between input and output of material on road surface, rather than statistically significant variables. Even the AP-42 document encourages this "material balance approach"[16] and suggests this may provide a better estimate of emissions than emission tests would (USEPA, 1995). Several researchers have emphasised the need of such a kinetic mass-balance model that includes the sources and

[16]Material balances are appropriate for use in situations where a high percentage of material is lost to the atmosphere (e.g., sulfur in fuel, or solvent loss in an uncontrolled coating process). In contrast, material balances may be inappropriate where material is consumed or chemically combined in the process, or where losses to the atmosphere are a small portion of the total process throughput (USEPA, 1995). In the present model no chemical transformation of material is considered and resuspension is considered as a major process that causes loss of material from road surface to air. Therefore material balance approach can be applied.

processes involved in particle replenishment and depletion on road (Countess et al., 2001; Kuhns et al., 2001a; Nicholson, 2001; Venkatram, 2001).

With respect to traffic as a variable for estimation of resuspension, the ideal scenario would be to have a resuspension fraction[17] for each class of vehicle. With time series of traffic flow and composition, it would then be possible to estimate resuspension more precisely. No such vehicle-specific resuspension fractions are available at present and therefore, until such vehicle-specific resuspension fractions are available, an average vehicle resuspension factor could be a better variable than average vehicle weight used in the AP-42 algorithm. Therefore it is necessary that in addition to the inclusion of the important accumulating and draining fluxes, the model should be able to provide an estimate of material removal fraction[18] for each of the draining fluxes, including the above-mentioned resuspension fraction.

It is important that exhaust emissions and resuspension are calculated separately because, as discussed in Section 2.6 each of these are generated from different processes. With reasonably good information available on exhaust-emissions, the main emphasis should be to develop a model that includes processes that cause resuspension and material translocation and estimate their relative contribution to the ambient urban air.

[17]Resuspension fraction is defined as the ratio of the amount of material resuspended from a road segment by one vehicle to that available on a road segment.

[18]Material removal fraction is defined as the ratio of the amount of material removed from a road segment by one vehicle to that available on a road segment. It is expressed in the unit of "per vehicle" (v^{-1}). More detail about this including its model parameterisation is available in Section 5.6.3.

Chapter 3

Field sites and measurements

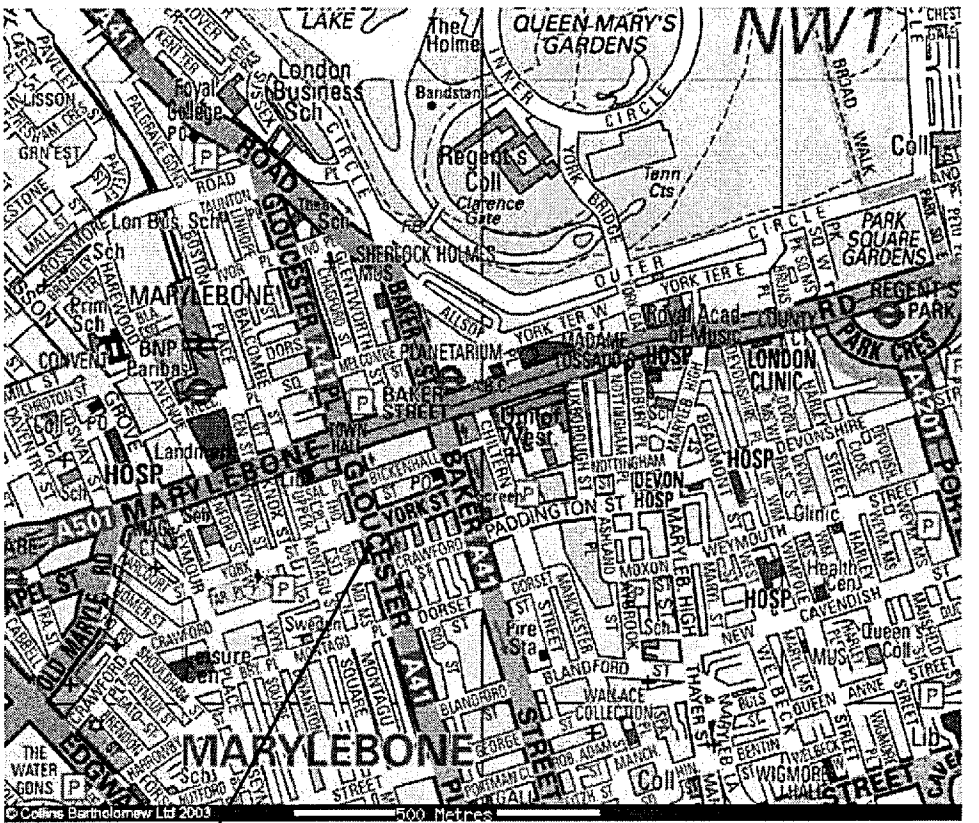
3.1 Introduction

The ideal scenario for a model will be to formulate it and estimate all the parameter values from physical principles. However, in reality, it is difficult to achieve. Therefore experimental data is often used to find the parameter values of the model. The input parameters to the model presented in this thesis have been taken from theory as well as from available literature. The experiment at Gloucester Place was designed to obtain parameter values for the draining fluxes of the model. Available data from Manor Road were used for evaluation of the model. This chapter describes both these field sites, experimental methodology at Gloucester Place and details of data available for Manor Road.

3.2 Site 1: Gloucester Place

3.2.1 Site description

The experiment was carried out at Gloucester Place, south of the intersection with Marylebone Road in Central London (Fig. 3.1). Gloucester Place is a paved road, aligned approximately North-South, joining Oxford Street to the south and Park Road to the north, intersecting Marylebone Road in between. Marylebone Road is a dual carriageway approximately 26 m wide and in places containing up to seven lanes in-

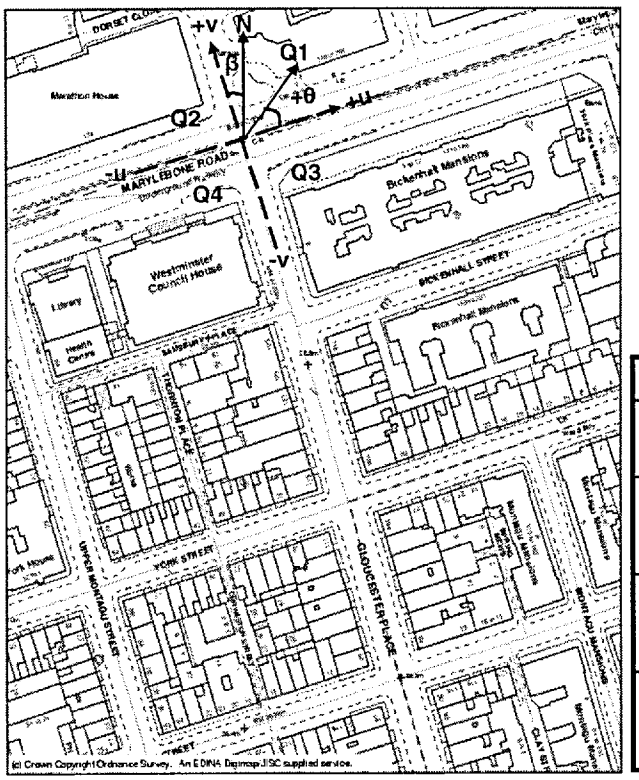


(Source: http:// www.Streetmap.co.uk/)

Fig. 3.1: Local street plan of the southern section of the Gloucester Place area.

cluding those for dedicated bus use and left/right turning traffic. Roads to the south of Marylebone Road come under the London Congestion Charging zone[1]. Gloucester Place is a three-lane road approximately 13 m wide. The first lane from the West is a designated bus/taxi lane. The one way traffic on this road moves from south to north. Directions of study area are based on a rotated Cartesian coordinate system with its North 20° west of the geographical North. This makes Marylebone Road running East-West and the Gloucester Place running North-South (Fig. 3.2). Arnold et al. (2004) have described more details of the study site. The measurements described here were confined to a length of Gloucester Place between Crawford Street to the south and Marylebone Road to the north, about 180 m in length.

[1]London Congestion Charging zone includes some parts of the Central London area the access to which by motorist incurs a fee. Details are available at http://www.biocrawler.com/ encyclopedia/London_Congestion _Charge (last accessed on 15 January 2006).



INDEX

N	Geographical North
Q1...Q4	Four quadrants of rotated Cartesian coordinate system
β = 20°	Angle between Geographical North and North of rotated coordinate system
+v	Direction of northerly component of wind under rotated coordinate system
+u	Direction of easterly component of wind under rotated coordinate system

Fig. 3.2: The Gloucester Place study area.

Buildings on both sides make Gloucester Place a street canyon with roads intersecting it perpendicularly at intervals. Buildings located on either side of the experimental site include the Westminster City Council (WCC) house at the north-west (NW) corner, Bickenhall Mansions at the north-east (NE) corner and at the east, Bickenhall hotel at the west and Montagu Mansions at the south-west (SW) and south-east (SE) corners. Dimensions of site features are presented in Table 3.1. The main criteria for site selection include the followings:

1. The site includes a busy public road at the centre of London that is representative of major urban roads in large cities.

2. The unidirectional flow of vehicles simplifies the determination of the rate of traffic-induced movement/dispersion of particulate matter.

3. There was considerable existing infrastructure for the field experiment from an

Table 3.1: Dimensions of site features

Site Feature	Dimension
Width of the Gloucester Place carriageway	13.0 m
Approximate width of the pavements	2.5 m - 3.0 m
Width of intersecting roads:	
Bickenhall Street	12.0 m
Salisbury Place	5.0 m
York Street	10.0 m
Crawford Street	12.0 m
General height of adjacent buildings:	
Westminster City Council house	15.0 m
Bickenhall Mansions	23.0 m
Bickenhall Hotel	18.0 m
Montagu Mansions (SW corner)	14.0 m
Montagu Mansions (SE corner)	17.6 m

on-going multidisciplinary research campaign "Dispersion of Air Pollution and Penetration into the Local Environment" (DAPPLE, http://www.dapple.org.uk).

4. The site itself is near symmetric on both sites of the road in terms of the dimensions of adjacent buildings, pavement widths and spatial characteristics of the intersecting roads over the test area.

5. The site afforded adequate space on both sides of the road to safely set out the experimental apparatus.

3.2.2 Application of grit

The on-street measurements were made on 26 May 2004 between 10:30 and 16:30[2]. In order to create a dusty road source of known initial dimensions and location, ordinary road surface gritting salt, of the type used to grit roads in winter, was applied to a section of the carriageway at the experimental site. The grit used was rock salt, supplied by Onyx Environmental Group plc (OEG, 2005). The selection criteria for gritting salt were easy availability, ease of obtaining necessary permissions to apply on

[2]All times in this thesis are in a 24 hr framework and in British Summer Time

road surface, minimum health and safety implications, heterogeneity of particle size fractions and known and existing method of application to the road surface. Particle size distribution was obtained by using graduated sieves, on a 100 g sample of grit collected from the gritting lorry (Fig. 3.3).

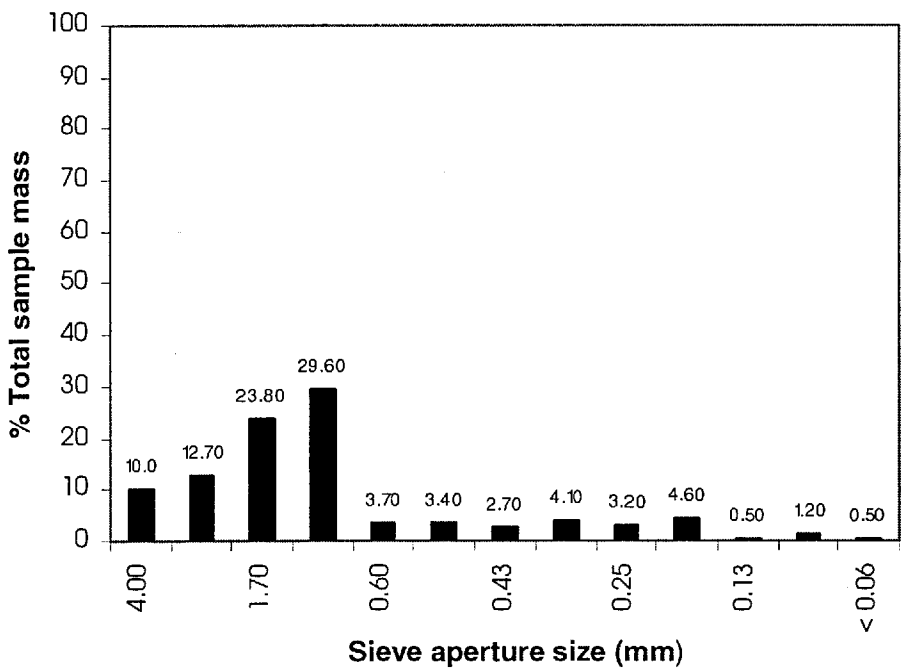
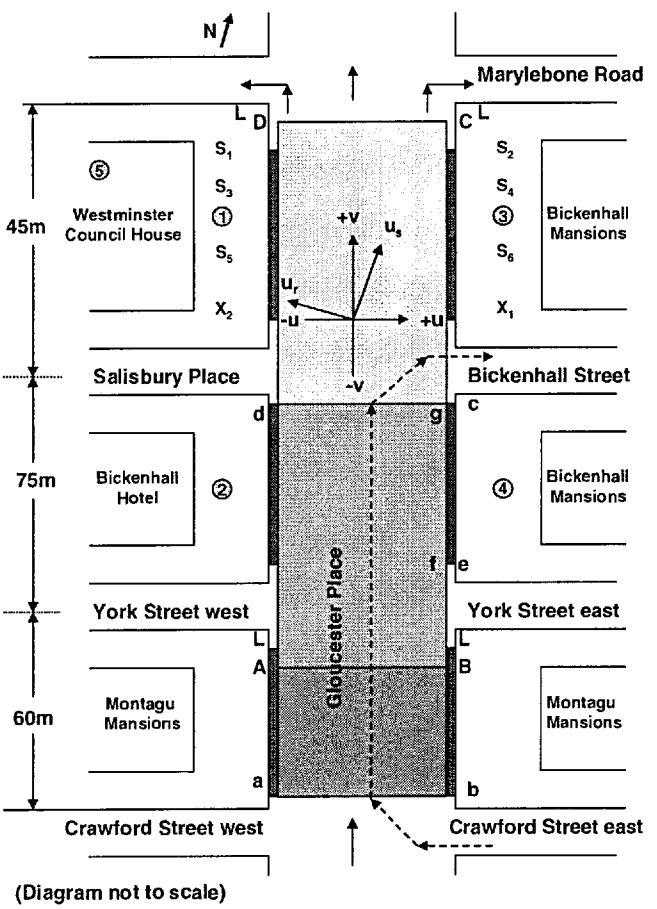


Fig. 3.3: Particle size distribution of grit.

The gritting was carried out in consultation with the Westminster City Council (WCC) Department of Highways Management and Cleansing Management. The grit was deposited by an OEG gritting vehicle, calibrated to deposit a known volume via a horizontally spinning wheel underneath the rear of the vehicle. This had the effect of spraying the grit onto the road from a height of approximately 1 m. Traffic was not stopped during the gritting procedure. The flow of traffic at the actual moment of gritting was light (< 10 vehicles passing the gritting lorry at any time) owing to the timings of the traffic-light system. It was also observed that, in order to avoid the spray of the gritting vehicle, other road users tended to avoid the gritted patch of road during gritting. In order to leave a defined transition between the source and receiving areas, the gritting lorry moved from the outside (Easterly) lane of the Gloucester Place into Bickenhall Street, where the vehicle was parked for the spinning delivery mechanism

to come to a halt (Fig. 3.4). This prevented residual grit from being deposited on the "receiving area" (stretch of the the Gloucester Place and adjacent pavements between Bickenhall Street and Marylebone Road, approaching the intersection with Marylebone Road). Gritting commenced at 11:15 and took approximately 5 minutes to complete. The grit acted as a tracer. Its movement, interpreted from data captured in roadside instruments (e.g., grimm and P-Trak), represents the movement of solid material on a paved road surface and the aim of the experiment was to observe its fate during the minutes to hours after application.



INDEX

abcd	Initially gritted section (Length = ad = 120 m, Width = ab = 13 m)
ABCD	Grit location at time t_1
cefg	Build-up of grit at kerb at time t_1 (Width = ef = 20 mm, Depth = 2mm)
L	Traffic signal lights
→ u_s	Predominant street- level wind direction
→ u_r	Predominant roof- level wind direction
→	Direction of traffic
- - - - - →	Route of gritting lorry
X_1, X_2	SF_6 release points
S_1...S_6	SF_6 sampling points
①	Site 1: Location of Grimm 1 (G1), P-Trak 2 (P2) and ultrasonic anemometer 1 (A1)
②	Site 2: Location of Grimm 2 (G2)
③	Site 3: Location of Grimm 3 (G3), P-Trak 4 (P4) and CO_2 monitor (C1)
④	Site 4: Location of Grimm 4 (G4)
⑤	Site 5: Location of ultrasonic anemometer 2 (A2)

Fig. 3.4: Schematic diagram of the experiment site showing gritted road surface and different sampling locations.

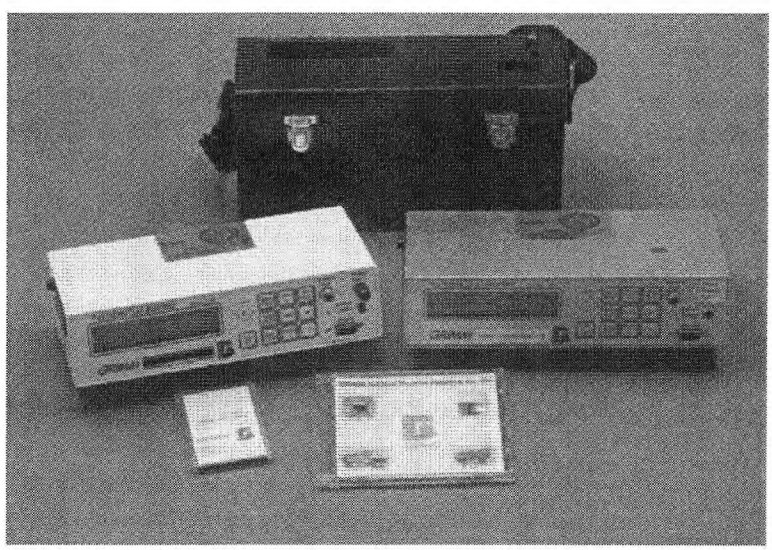
The area gritted covered a stretch of carriageway approximately 120 m in length, between Crawford Street and Bickenhall Street, and 13 m in width (same as width of the Gloucester Place). Gritting started from a-b and ended at c-d (Fig. 3.4). In order to obtain a visible coating of grit, the quantity applied to the road was 20 g m^{-2},

representative of WCC's "snow precautionary gritting" regime, making the approximate mass of grit deposited 31.2 kg. This is equivalent to an addition of silt loading[3] between 0.1 g m^{-2} and 0.34 g m^{-2}.

3.2.3 Roadside measurements

3.2.3.1 Particulate matter concentration in air

To allow quantitative observation of the movement of grit in the direction of traffic movement from an initially gritted section abcd to an initially ungritted section dcCD, particle concentrations in the air at near road-surface level were measured in eight particle size channels from 0.75 μm to $>15\,\mu$m every minute using four Grimm Model 1.105 version 5.60E (Fig. 3.5) light scattering particle counters, G1, G2, G3 and G4 located on the east and west pavements of Gloucester Place, at Sites 1, 2, 3 and 4 respectively.



(Source: manufacturer's web site)

Fig. 3.5: Grimm optical particle counter.

The monitoring stations were adjacent to but set back from the roadside and regularly spaced along the source and receiving area (Fig. 3.4 & Table 3.2). The Grimms

[3]From Fig. 3.3, the silt size of 75 μm lies somewhat between the two lowest size fractions and therefore the silt loading lies between 0.50% and 1.70% of the surface loading.

remained stationary in the monitoring position, and were switched on to record measurement for a period of up to 6 hours. The data from the whole experiment, stored to each instrument's memory card were quickly downloaded to a laptop loaded with the Grimm software, within and immediately after the monitoring period.

Table 3.2: Grimm locations

Site	Location	Grimm ID	Height above the pavement	Distance from the kerb
1	WCC ingress	G1	1.50 m	2.55 m
2	Steps leading up to door of 117 Bickenhall hotel	G2	0.45 m	2.60 m
3	Steps leading up to door of 8 Bickenhall Mansions	G3	0.50 m	2.60 m
4	Steps leading up to door of 110 Bickenhall Mansions	G4	0.45 m	2.80 m

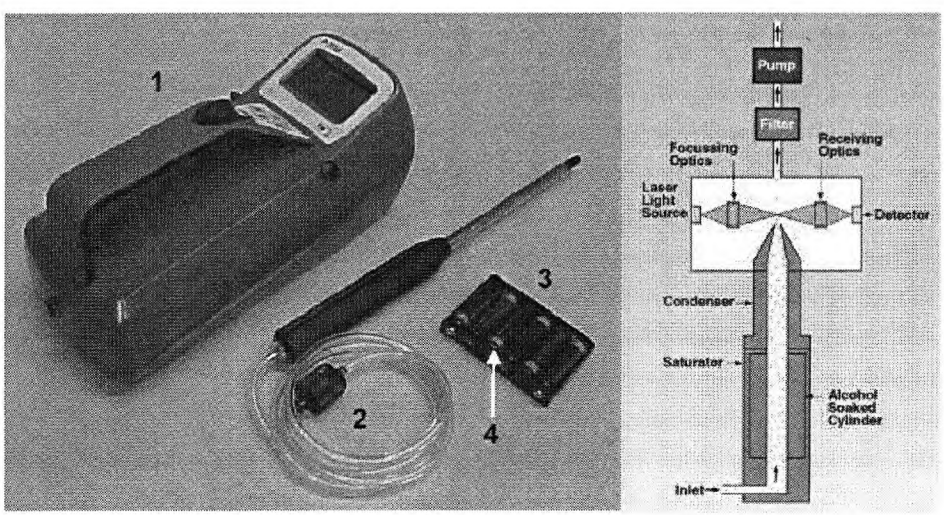
The Grimm uses an optical sensing technique, based on light scattering caused by particles entering the unit via a small vacuum pump. For this experiment, the instruments, that can be used in a variety of operational modes to record particle counts/concentration, were used in the particles per litre (pt L^{-1}) mode. The use of optical sensors to measure PM_{10} has been the subject of debate, but they provide a robust quantitative measurement of the number of particles in the air as a function of particle size, especially for particles in the size range 0.5 μm to 10 μm which are of most interest for road surface emissions. Their portable nature makes them especially suitable for assessment of spatial variation in concentrations and changes over short periods of time.

3.2.3.2 Ultrafine particle count

Two P-Trak Model 8525 and Serial nos. 05021012 and 03021004 butanol supersaturation ultrafine particle counters (Fig. 3.6) P2 and P4, deployed at Site 1 and Site 3 respectively measuring at 1 Hz, captured time-varying levels of particulate matter in the size range 0.02 μm to 1 μm which is expected to be dominated by exhaust emis-

sions[4]. Depending on the memory of P-Traks, the data were transferred to a computer using TRAKPRO software (NYSERDA, 2002; TSI, 2004).

A P-Trak is a hand-held portable instrument that measures concentration of ultrafine particles in air in units of particles per cubic centimeter (pt cm^{-3}). The instrument can record particle counts up to 5×10^5 pt cm^{-3}. Particles are drawn through the P-Trak using a built-in pump. The sampling flow rate is approximately 700 cm^3 min^{-1}. It is a condensation nuclei counter where, upon entering the instrument, particles pass through a saturator tube where they mix with an alcohol vapor. The particle/alcohol mixture then passes into a condenser tube where alcohol condenses onto the particles, causing them to grow into larger droplets. The droplets then pass through a focused laser beam, producing flashes of light which are sensed by a photo-detector. The particle concentration is determined by counting the light flashes (NYSERDA, 2002; TSI, 2004).



1. P-TRAK ultrafine particle counter
2. Probe assembly, consisting of telescoping probe, sample tube, and inlet screen assembly
3. Battery holder
4. AA alkaline battery

P-Trak: Theory of operation

(Source: Manufacturer's website http://www.tsi.com)

Fig. 3.6: Model 8525 P-Trak ultrafine particle counter.

[4]Exhaust aerosol is emitted primarily in sub-micron particle sizes centered on particles of 0.1 μm to 0.2 μm in diameter (Hildemann et al., 1991b). The typical size ranges of diesel exhaust from a bus and petrol exhaust from a car are 0.02 μm to 0.13 μm and 0.04 μm to 0.06 μm respectively (Morawska et al., 1998; Ristovski et al., 1998).

3.2.3.3 CO_2 level in roadside air

Concentrations of CO_2 above the local background, attributable to vehicle exhaust emissions, were measured every 2 s at Site 3 by infra-red absorption using a battery-powered sensor with 1/8 s response time and $\pm 2\%$ precision. This provided an additional indication on the temporal variability of vehicle exhaust concentrations in roadside air, along with the P-Trak. Changes in the ratio of dust to ultrafine particles and CO_2 in roadside air throughout the experiment may therefore be attributed to the rise and decay times of dust levels on the road after the deposition of the grit. The CO_2 monitor, chained to a lamp-post 20 cm above ground on the east pavement of Gloucester Place, was operated by experts from School of Chemistry, University of Bristol, UK. Therefore, the data used in this research were obtained from them.

The Gascard II plus instrument, used in this experiment (Fig. 3.7), was constructed in-house at Bristol University from components supplied by Edinburgh Instruments. The standard analogue output of the sensor was modified to provide a digital output that enables the CO_2 level to be measured and recorded using a computer with a serial interface. This CO_2 detector is smaller in size than older tuneable IR spectrometers (Fig. 3.7). As well as the obvious size and weight differences that increase its portability in the field, the new sensor can be powered by batteries. This enables the detectors to be discretely placed at the road side whilst not blocking pedestrian access.

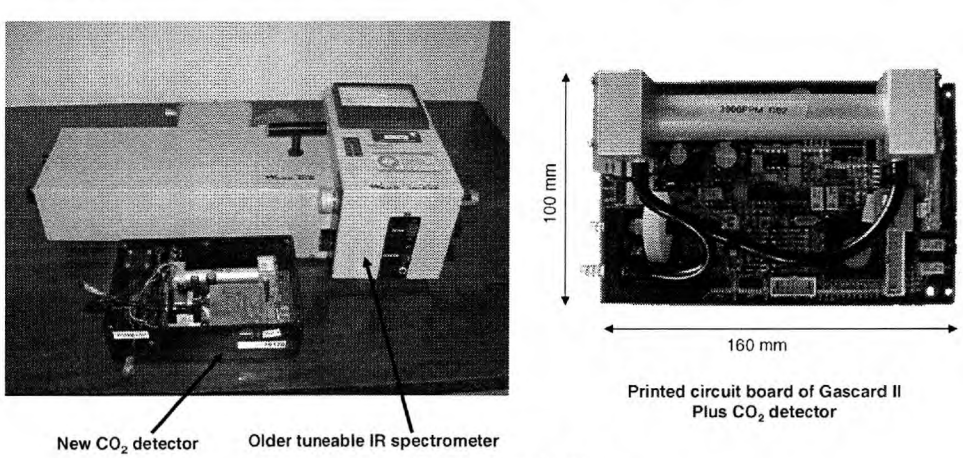


Fig. 3.7: CO_2 detector.

The detector specification allows it to measure a concentration between 0 - 2000 ppm, a range that covers the limits of atmospheric CO_2 (CO_2 in atmosphere is approximately 350 ppm). The detector is accurate to $\pm 2\%$ of the value of the CO_2 concentration and has a temporal resolution of 1/8 s, a resolution that can record short time-scale events that would be missed using slower methods. The detector has been fitted with a pump that allows the machine to work at its optimum efficiency. The system output data goes to a laptop which again is battery powered (Henshaw, 2003).

Method of detection

Molecules and atoms absorb and emit radiation. CO_2 shows a strong absorption at 2349 cm^{-1} (Fig. 3.8).

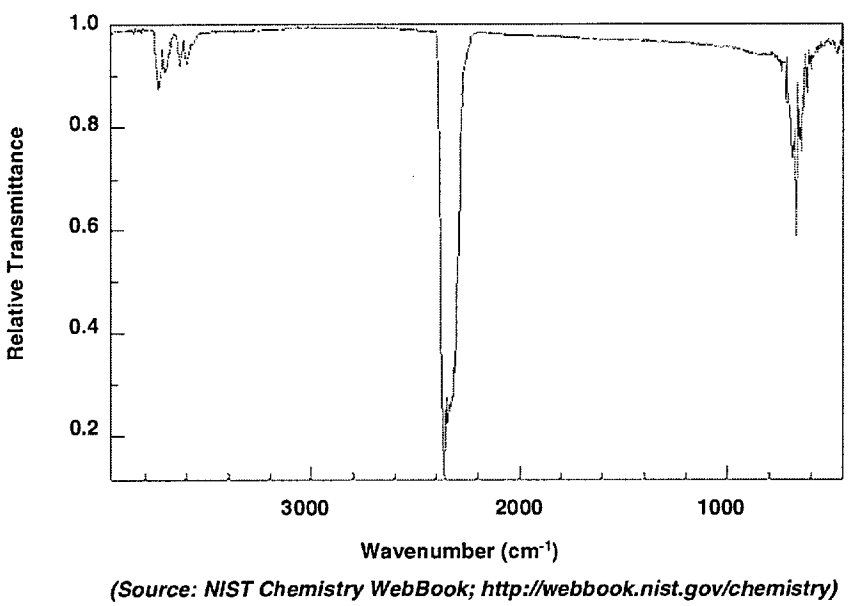


Fig. 3.8: CO_2 infra-red spectrum.

The CO_2 detector has an infra-red lamp at one end that emits radiations over a wide range of frequencies (Fig. 3.9). A detector at the other end of the cell has a filter placed across it. The filter is tuned to allow only radiation of 2349 cm^{-1} to pass through it. A second path of equal length has the same arrangement as before except that it does not have the filter. The difference between the two paths is the amount of light that has been absorbed at the frequency at which CO_2 absorbs IR radiation.

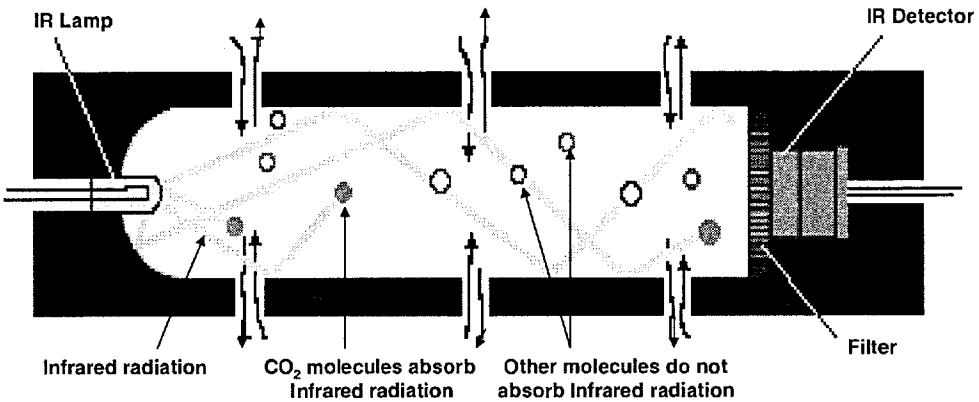


Fig. 3.9: Non Dispersive Infra Red (NDIR) CO_2 sensor: mechanism of operation.

The concentration of the CO_2 is calculated using the "Beer Lambert law".

$$\frac{I_1}{I_0} = e^{-\alpha L c} \tag{3.1}$$

Where,

I_0 = intensity of the incident light

I_1 = intensity of the transmitted light

α = absorption coefficient of CO_2

L = length of the sample

c = concentration of the sample

With absorption coefficient of CO_2 a known quantity, Equation 3.1 allows the concentration of the CO_2 to be calculated provided that the length of the cell is known. The calculations for this are made inside the machine using its on-board microprocessor. The program that stores the data output of the detector uses a binary data format for the data. This allows all of the data recorded to take up only 3 MB at full resolution for 1 day.

Calibration of Equipment

The Grimm, P-Trak, ultrasonic anemometer and CO_2 detector were factory calibrated before deployment. In addition, the four Grimms were run side-by-side in Gloucester

Place immediately at the end of the measurement period to ensure identical particle counts within the limit of counting statistics. All data-logging clocks were synchronised using Global positioning system (GPS) or radio controlled clocks. Further details of the quality control procedures are outlined by Arnold et al. (2004).

3.2.3.4 Traffic

Traffic flow on Gloucester Place was measured every 96 s by collecting data from the SCOOT (Split Cycle and Offset Optimisation Technique) induction loop counters (Arnold et al., 2004; Hunt et al., 1982; Tate and Bell, 2002), calibrated by local manual counts during the days prior to the dust release experiment, and supplemented by occasional approximate manual counts of numbers of vehicles passing per traffic signal cycle on the day. SCOOT data for this experiment were obtained from Institute for Transport Studies (ITS), University of Leeds, UK, who, being one of the experts in this area, was responsible for maintaining the network and retrieving data from it. These data were further processed to change it to a format that can be used in subsequent modelling exercises.

SCOOT uses a network of inductive loop detectors embedded in the road surface to measure basic traffic parameters such as flow and lane occupancy in order to optimise the traffic signals and to minimise the delay in response to fluctuating traffic levels. It, therefore, is a demand responsive Urban Traffic Control (UTC) System, a platform for an on-line integrated traffic monitoring, modelling and management system (Arnold et al., 2004; Hunt et al., 1982; Tate and Bell, 2002). SCOOT data are converted to usable traffic flow, congestion and queue-length parameters using algorithms developed by ITS, University of Leeds (Tate and Bell, 2002).

3.2.3.5 Meteorology

Two ultrasonic anemometers, A1 and A2 (Gill Scientific Instruments, Research Grade 3-axis type), were used for this experiment to record wind speed and direction at a frequency of 20.83 Hz. A2 was mounted on a 2 m mast on the rooftop of the WCC house at Site 5 to measure the rooftop wind direction and speed. Road-side sonic

anemometer A1, deployed to measure street-level wind conditions, was located on the west pavement of Gloucester Place in front of the WCC East door at Site 1 (Fig. 3.4), mounted on street furniture using brackets secured by steel band tie. The instrument was projected 0.4 m from the street furniture. Data was logged, using FASTCOM, to laptops at the base of the supporting mast. Anemometers were powered by batteries in boxes secured to the foot of the posts. Alignments were in a direction different from the coordinate system used for analysis and hence raw data needed rotation to conform to the Cartesian coordinate system used for interpretation of the experimental outcome.

3.2.3.6 Tracer study

To assist the interpretation of the measurements of the dust tracer after it became airborne, simultaneous to the measurements of continuous emissions of fine particles and carbon dioxide (CO_2) from the vehicle exhaust, gas tracer measurements were made in the same area using known releases of sulphur hexafluoride (SF_6). Dispersion function obtained by tracer method is essentially the ratio of concentration of the tracer and the known emission rate of the tracer (Venkatram et al., 1999). Although tracer study involves more field logistics than a dispersion model method, it is suggested to be more accurate than the latter in representing the dispersion condition (Fitz and Bufalino, 2002; Venkatram, 2000). Estimation of emission rates for sulphate (Cadle and Chock, 1977) and NO_x (Okamoto et al., 1990) are a few of the earliest cases where this method has been used in experiments to estimate emissions from road. Two recent studies used tracer method for estimation of PM_{10} emission from paved and unpaved road surfaces (Claiborn et al., 1995; Kantamaneni et al., 1996).

Tracers are gases or lightweight compounds and an ideal tracer must confirm to some criteria such as it must be inert and thus long lived; it should be non-depositing, non-toxic and harmless to environment; it should have a low background level and therefore, low level of detection; and cost of the tracer and its analysis should be low (Lovelock and Ferber, 1982). Over the course of research, many different compounds have been used as tracers including flourescin acid (Bultnyck and Malet, 1972), Zinc

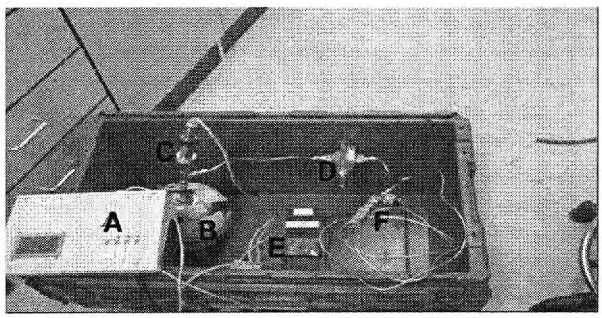
Sulphide (ZnS) (Sehmel, 1973), CO_2 (Henshaw, 2003) and perfluoromethylcyclohexane ($C_6F_{11}CF_3$) (PMCH) (Arnold et al., 2004). Spores of lycopodium are probably the most unusual material to be used to investigate the air flow; the particular colour of the spores was an aid in their detection. Work in the mid 1970's by Pasquill identified SF_6 as a good candidate for a tracer. It fulfills all of the criteria for a good tracer as defined by Lovelock and Ferber (1982) with the exception that it is now in commercial use, with output at 85700 tonnes in 1995 (Miass and Brenninkmeijer, 1998). It is also a greenhouse gas but only very small quantities are released for tracer studies, although rising background levels require larger release than in the past. So their contribution to climate change is insignificant. SF_6 is detectable at the part per trillion level.

The tracer study on 26 May 2004 was aimed at investigating near-source dispersion (< 100 m). Two independent SF_6 releases were performed to estimate the average dispersion relation. Each involved the constant flow release of SF_6 at point X_1 and X_2, upwind for the prevailing wind direction along the Gloucester Place, between Bickenhall Mansions and the WCC building off the Marylebone Road. A number of paired receptors, S_1 through S_6, were placed at distances away from the source (SF_6 release points) and time integrated samples were taken using Tedlar bags (Fig. 3.4). All the sampling stations were located on the street level. Samples were analysed in triplicate on site using a Hewlett Packard 6890 Gas Chromatograph (GC) with Electron Capture Detection (ECD). A dilution ratio at each of the receptor points was calculated and the crossroad gradients were determined.

The tracer study of the day was carried out under the supervision of experts from School of Chemistry, University of Bristol, UK. Consequently, the initial raw data were supplied by them which were further analysed to derive dispersion relations applicable for this experiment. The tracer study in detail is available elsewhere (Martin et al., 2004). The salient points are described, in brief, below.

Tracer release mechanism

The release equipment (Fig. 3.10) comprised of a 3 L stainless steel (BRC Rasmussen, Oregon, USA) canister filled with a certified 1% SF_6 in air mixture (Air Products Ltd.,



A: Data logger
B: 3 litre stainless steel canister
C: Pressure transmitter
D: Flow controller
E: 12 volt battery
F: Solenoid valve

(Source: Martin, et al., 2004)

Fig. 3.10: Experimental set up for the release equipment.

Cheshire, UK). This is connected via a 1/4" stainless steel T-piece to a pressure transmitter on one side and a constant flow controller on the other. The flow release was controlled by the use of a MNBS12 Flostat flow controller (Roxspur Measurement and Control Ltd., Hampshire, UK). This maintains constant flow when the supply pressure is varying and the discharge pressure is relatively constant. The pressure, in the range of 0 bar to 5 bar with a precision of 1 millibar or 0.05 % at full scale, is monitored by a pressure transmitter (Keller UK Ltd., Dorchester, UK). An in-house data logger monitored the pressure and temperature of the release apparatus. The output of the flow controller was connected via 1/8" stainless steel tubing to a 12 volt (V) solenoid valve (Precision Dynamics, Connecticut, USA). The solenoid is controlled by the use of a switch powered by a 12 V/7 ampere hour (AH) battery. The outlet nozzle from the release equipment is stainless steel (1/8" OD x 0.085" ID). This gives a typical release velocity of 0.45 m s^{-1} per 100 cm^3 of gas released. All the components were installed in a portable container.

Sampling and analysis

Sampling consists of a multiport ladder fitting with solenoid valves, onto which a sampling bag is attached. The bags used were 10 L Tedlar bags with a stainless steel fitting and have been shown to be non-contaminating for SF_6. Sample is drawn in from a height of 1.5 m through Nathgal tubing (3 mm ID) (Pnuemax ltd., UK) by using a small vacuum pump (Vacuum Pump Manufacturing Co. Ltd, Bucks, UK). This draws

in sample at a flow rate of approximately 1 L min^{-1}. The 6 V vacuum pumps are powered by 12 V/7 AH batteries through a voltage step-down converter.

During both releases , 8 minute integrated samples were taken. Before sampling, all sampling lines were purged for 2 minutes. The flushing time for the sampling apparatus in this configuration is approximately 0.6 s $\pm$ 0.2 s. This ran in synchronisation with the release, which for both experiments occurred over a 10 minute time period. Controller boxes, manufactured in-house, were operated by volunteers by switching into purge and sample modes at the desired times. Times were dictated by the use of radio-controlled clocks.

The analysis by a Hewlett Packard 6890 GC involved separation of SF_6 sample from atmospheric oxygen by a 10 m x 0.53 mm Ultimetal Column with a 50 m coating of Molecular Sieve 5 A and introducing these to the analytical system by a 50 microlitre (μl) fixed loop injection. Instrumental parameters are shown in Table 3.3.

Table 3.3: Instrumental parameters for SF_6 analyses

Parameter	Value
Column flow (N_2)	120 ml min^{-1}
Column temperature	400 °C
Detector temperature	3000 °C
Detector make up flow	30 ml min^{-1}

Tracer calibration

A 0.90 ppmv SF_6 Standard (BOC gases Ltd., Surrey, UK) was spiked with pure ethane (BOC gases Ltd., Surrey, UK). The propane concentration was analysed by GC using a KCl modified Al_2O_3 Porous Layer Open Tubular (PLOT) 50 m x 0.32 mm x 5 m. Detection was by flame ionisation (FID) and calibration was achieved by comparison with a 5 alkane certified mixture (Scotty Speciality Gases, PA, USA). A new standard value incorporating the volume change was ascertained. This sample was then serially diluted using 5.5 zero air (Air Products PLC, Cheshire, UK). The dilution ratio was then determined by analysis for propane and comparison with the parent standard.

3.2.3.7 Sampling of dust on the road surface

In order to provide a qualitative assessment of the extent of dispersion around the road, dust samples from the road surface were collected into a dustpan by brush-sweeping a length of 30 cm along the gutter and then transferred into sealable sample bags. Samples were taken at 3 time intervals during the monitoring period, from the specified sampling points using individual dustpan and brush sets for each sampling point. Furthermore, dust from the road surface was collected onto transparent adhesive tape (Selotape, "Diamond" brand) and stored in plastic wallets for later examination by microscope. All the samples were stored at ambient temperature for future analysis. Location of various sampling points are described in Table 3.4.

Table 3.4: Road surface dust sampling locations

Sampling location	Distance from the Gloucester Place
Crawford Street west	11.9 m
Crawford Street east	10.6 m
York Street west	9.0 m
York Street east	9.0 m
Bickenhall Street	4.2 m
Salisbury Place	8.4 m
Pelican crossing round the corner on Marylebone Road (outside the WCC building)	19.9 m

3.3 Site 2: Manor Road

3.3.1 Site description

Manor Road, Erith, is a narrow, single-carriageway road running east-west, parallel to the south bank of the River Thames. It is situated in the north of the London Borough of Bexley in East London and is about 400 m from the south bank of the River Thames. The westerly half of the road is residential, with terraced houses immediately adjacent to the road providing an opportunity for residents to suffer long-term exposure to air pollution emissions from the passing traffic. The easterly half of Manor Road passes

through an area of industrial development, including two sites handling cement, one metal recycling plant, and a waste transfer station (Fig. 3.11) (Bexley, 2001, 2003).

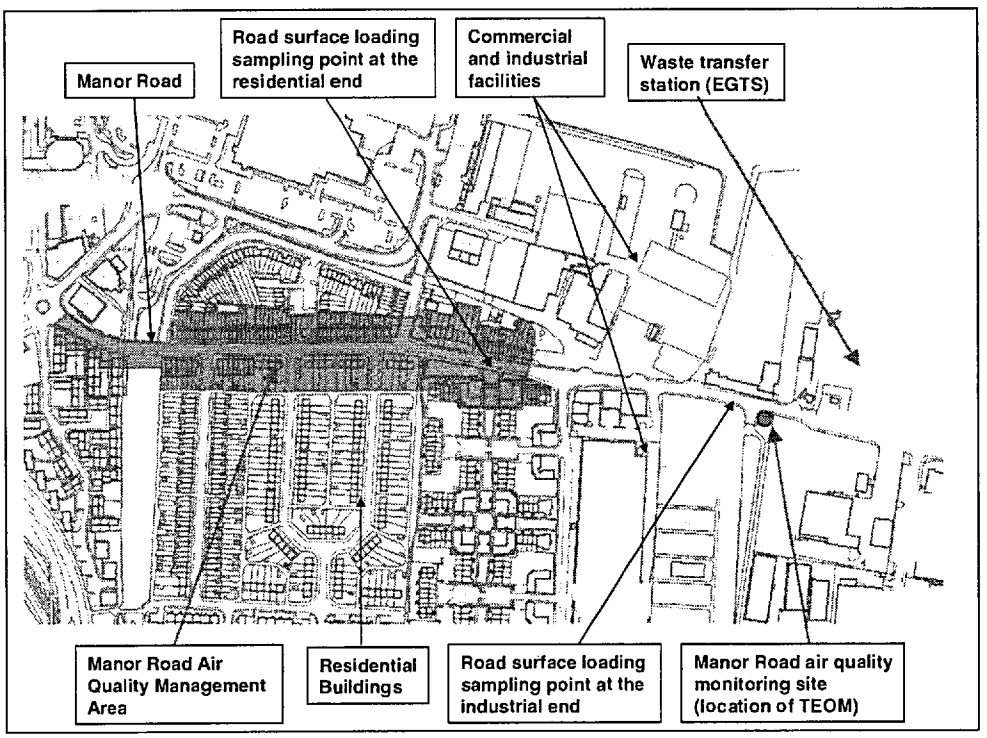


Fig. 3.11: Manor Road study area.

Manor Road is selected for model evaluation purpose because the residential part of it is an Air Quality Management Area[5] (AQMA) where resuspended coarse particles from road surface are identified as the main reason behind the non-compliance of the Standard (Section 3.3.1.1). The surface of the road is visibly soiled by mud carried onto it by traffic exiting from an adjacent waste transfer station, called Erith Group Transfer Station (EGTS). It is therefore suspected that this mud, being suspended into the air, is the reason for the elevated roadside concentrations of PM_{10}. Selecting it as

[5]Section IV of the UK Environment Act 1995 requires local authorities to periodically assess air pollution within their area against the UK Air Quality Standards (The standards for PM_{10}, to be achieved by 31 December 2004 are (a) $50\,\mu g\ m^{-3}$ as a 24 hr mean, not to be exceeded more than 35 times a calender year and (b) $40\,\mu g\ m^{-3}$ as an annual mean with no exceedances.) (Buckingham et al., 1997; DEFRA, 2000, 2003). Under this Act, where a local authority considers that one or more of the air quality objectives are unlikely to be met by the due dates, the authority must declare an (AQMA), covering the area where the problem has been identified (AQEG, 2004; DEFRA, 2000; NETCEN, 2005). On the basis of this guideline, following a 3-stage process of reviewing and assessing local air quality in the Bexley Council, it was found that the Council may not meet these Standards in respect of PM_{10} at the residential part of the Manor Road by the target date (DEFRA, 2000). Thus, on 22 August 2001, the Council declared the residential section of the Manor Road as an AQMA for PM_{10} (Bexley, 2001).

the second field site, the aim is to test this hypothesis, and to make progress towards improved capability in quantifying the rate of emission from this road and similar ones elsewhere, including those on which the level of soiling is not as high.

3.3.1.1 Information available from on-going investigations

Stage 3 report of review and assessment of air quality in Bexley identified that there is about 40 % of the particulate matter that is unaccounted for after taking care of the all possible local industrial sources. In 1999, the 90^{th} percentile of fixed 24 hr mean for PM_{10} at Manor Road was found to be 86.6 μg m^{-3}, an exceedance of air quality objective by 73 %. On the other hand, a monitoring site at Slade Green, which measures urban background level (Section 3.3.2.2), recorded PM_{10} level of 40.3 μg m^{-3}, thus implying that the excesses of particulate matter were due to local sources at Manor Road. Local source apportionment by wind direction showed that TEOM records highest PM_{10} levels when wind came from the direction between north-east and north-west and therefore travelled over Manor Road before reaching the monitoring station. Air from north-east will also travel across EGTS, a waste recycling and transfer site, before reaching the monitoring station. Modelling of PM_{10} distribution in and around Manor Road, carried out using relatively well known exhaust emission factors from the UK Atmospheric Emission Inventory (NAEI, 2003), revealed that the 90^{th} percentile of PM_{10} at Manor Road was 44.5 μg m^{-3}, half the monitored value of 86.6 μg m^{-3}. Thus it suggested that some major sources are missing and indicated that suspension of dust from Manor Road to be the most likely source, although local industrial activities can also be the reason. Subjective assessment of the area confirmed this hypothesis (Bexley, 2001).

Following this, as a part of stage 4 review and assessment by Bexley Council, samples of road dust were examined under Scanning Electron Microscope (SEM). The SEM examination showed that the composition of air samples collected at Manor Road resembles the chemical composition of road dust samples (Jansz, 2002; Jansz et al., 2004). The study also found positive correlation between silt loading and PM_{10} concentration in air. Further, a study by the Environmental Research Group (ERG), King's College

London (KCL) compared the spatial and temporal variation of NO_x with PM_{10} and concluded that there is excess PM_{10} concentration at Manor Road that cannot be attributed to exhaust emission (Bexley, 2003). The effective PM_{10}:NO_x ratio at Manor Road is over 10 times higher than the average factor at other sites in London (AQEG, 2004; Fuller and Green, 2004). It also suggested that fixed industrial units operating in the area are not significant non-exhaust PM_{10} contributors. Close relationship between non-exhaust PM_{10}, local exhaust PM_{10} and their diurnal variation with local NO_x suggested that these all originate from traffic along Manor Road. The report concluded that suspension and resuspension of material from the road surface is a significant source along Manor Road. Table 3.5 gives approximate relative contribution of various sources of particulate matter at Manor Road, estimated from a modelling exercise during the stage 4 review (Bexley, 2001, 2003).

Table 3.5: Source contribution of PM_{10} at Manor Road

Source	Contribution
Exhaust emissions	1.5 %
Industrial emissions	3.5 %
Resuspension of road dust	32 %
Urban background	63 %

The large quantity of data available, and accumulation of evidence in favour of a hypothesis that the road is a major source of airborne PM_{10}, make Manor Road an ideal location to try and improve our understanding of the process by which sediment moves along a road and becomes entrained into the air during the passage of vehicles.

3.3.2 Particulate matter concentration in air

3.3.2.1 PM_{10} at Manor Road

A Tapered Element Oscillating Microbalance monitor (TEOM) sampling PM_{10} is located at a distance of 6 m from the kerb on the south side of Manor Road, a few metres to the west of the exit from the EGTS, to provide hourly measurements of roadside air pollution levels continuously. The equipment is run by the local authority and quality

check is carried out by AEA/NETCEN. Frequent heavy goods traffic exiting from the waste transfer station turns right and proceeds to the west along Manor Road, past the monitoring station and then on through the residential area (Fig. 3.11).

The TEOM is the most widely used equipment for measurement of airborne particulate matter in the UK which is capable of operating with a time resolution of, as high as, 0.5 s. This technology utilises a hollow tube with the wide end of the tapered tube fixed. The narrow end of the tube holds a filter cartridge, and a sample is passed through the filter and tube to a flow controller. The tube-filter unit acts as a simple harmonic oscillator with its oscillating frequency being a function of filter mass loading (Green et al., 2001; Moosmüller et al., 2001). The system can be calibrated by placing a calibration mass on the filter and recording the frequency change due to this mass. Air is sampled at 16.7 L min^{-1} through a USEPA approved PM_{10} size selective inlet. A sample splitter divides the flow, carrying 3 L min^{-1} to a 16 mm Polytetrafluoroethylene (PTFE)-coated quartz filter (AQEG, 2004; Green et al., 2001). The sample inlet is heated to 50 °C to cause evaporation of particle bound water that had condensed at the time of sampling. However, this aerosol conditioning leads to undesired losses of some volatile compounds i.e., hydrocarbons, nitrates (especially, ammonium nitrate), etc. Therefore, the TEOM technique may give lower concentrations than the gravimetric methods that are recommended as reference methods for measurement of PM_{10} and $PM_{2.5}$ in the EU and in USA (Allen et al., 1997; King et al., 2000). The APEG report in 1999 concluded that at concentrations around 50 $\mu g\ m^{-3}$, the TEOM underreads between 15 % and 30 % when compared with gravimetric measurements (APEG, 1999). In order to allow TEOM data to be compared against the air quality objectives[6], a correction factor of 1.3 has been applied as recommended in the Pollution Specific Guidance, although several studies indicate that a single correction factor does not reflect the site and season specificity (Allen et al., 1997; APEG, 1999; Green et al., 2001; Salter and Parsons, 1999; Smith et al., 1997) and that correction factors can vary depending upon the amount of volatile matter (AQEG, 2004; EPAQS, 1998, 2001).

[6]The air quality objectives are based upon measurements carried out using the European reference method, a filter-based gravimetric measurement.

3.3.2.2 Background PM_{10}

Monitoring stations that measure urban background pollution level in Bexley area are located at Bedonwell, Belvedere, Eltham, Thamesmead and Slade Green, out of which Slade Green, located about 1.3 km south of Manor Road, is the closest one to Manor Road. Therefore monitoring data of this station is used as background level for Manor Road. The monitoring station is located within a self-contained, air conditioned housing located approximately 40 m from the nearest road in a suburban background setting. The surrounding area is generally open and comprises residential dwellings and gardens. The manifold inlet is approximately 3 m above ground level. It is measuring O_3, CO, SO_2, PM_{10} and NO_x at 15 minute intervals since 1 May 1997. This monitoring site is a part of the UK Automatic Urban and Rural Network (UKAURN) (NETCEN, 2005; Stanger, 2005). Like the PM_{10} measurement at Manor Road, a correction factor of 1.3 has been applied to convert TEOM concentration to equivalent gravimetric concentration.

Quality assuarance/Quality control(QA/QC)

Day to day operation of the sites, bi-weekly site visits to undertake replacement of filters, zero and span checks and review of operation of equipment are undertaken by Bexley council staff trained for this operation. Independent data checking, adjustment and quality control checking are undertaken by the ERG, KCL. Additional independent quality assurance and control is undertaken by AEA/NETCEN. Half-yearly, staff from National Physical Laboratory (NPL) visit the site to carry out further independent audits of analysers and equipment (Bexley, 2003).

3.3.3 Traffic

From the average hourly NO_x measurement for a week, obtained from a study conducted by ERG, KCL for a 3-month period from 6 July 2001 to 18 October 2001 and hourly traffic count and composition for a single day, obtained from Babtie Group traffic survey on 24-25 July 2001 at Manor Road (Bexley, 2003), average hourly traffic flow on Manor Road for a week was estimated and the same weekly pattern has been

assumed for the whole modelling period.

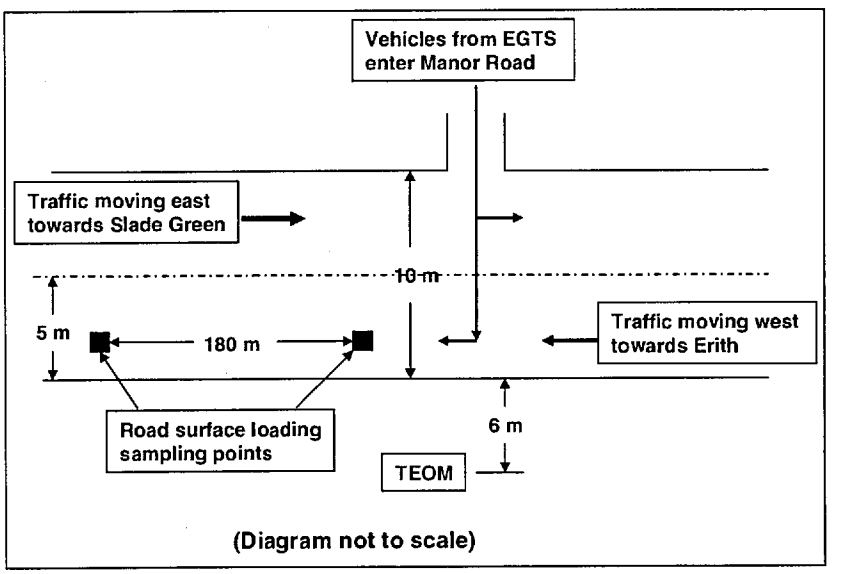


Fig. 3.12: Simplified diagram of traffic flow on Manor Road.

Vehicles from EGTS enter Manor Road at the industrial end of it from where a few vehicles move east towards Slade Green and the majority move west towards the residential end of the Manor Road (Fig. 3.12). Method similar to that of Manor Road is followed for estimating the vehicle count from EGTS to Manor Road except that, while NO_x study was available to extrapolate the traffic flow pattern on Manor Road for a week, no such study was available for vehicles coming from EGTS. So it is assumed that the same pattern of flow is maintained throughout all weekdays. On weekends, the EGTS operates between 8:00 and 13:00 and traffic flow for this period is assumed same as that of corresponding periods for the weekdays.

3.3.4 Road surface loading

Available data on road surface loading, generated from road dust samples collected and size-segregated by Jansz (2002) as a part of her MSc. fieldwork, have further been analysed and used in this research. Briefly, it involved collection of samples between 11:30 and 12:30 on Saturdays for five consecutive weeks beginning on 29 June 2002 and ending on 27 July 2002. On all days, samples were collected from a 500 mm $\times$ 1000 mm section of the road at the industrial end outside the TEOM

monitoring station (Fig. 3.12). On last two Saturdays, samples were also collected from the residential end of the road, located at about 180 m west of the first sampling point. In addition to usual dry vacuuming method, wet vacuuming methods were used which involved spraying of deionised water onto the road surface and collection using an industrial wet vacuum cleaner (Jansz et al., 2004). The wet vacuuming technique is believed to be much more efficient than dry vacuuming at collecting the smaller size of particles from the road which, if become airborne, likely to contribute to PM_{10} emissions (Deletic and Orr, 2005). Samples on the first four weeks were collected at each point at the industrial end from 3 adjacent positions representing three different tyre tracks and the average was calculated. Further, at one position, a second vacuuming was carried out to find out the vacuuming efficiency and to apply necessary corrections to road surface loading. Sample analysis consisted of estimating different size fractions following the basic protocol of separating, drying and weighing. Particulate matters were separated into five size groups: coarse ($> 425\,\mu$m), medium coarse ($150\,\mu$m to $425\,\mu$m), medium fine ($63\,\mu$m to $150\,\mu$m), fine ($< 63\,\mu$m) and PM_{10} ($< 10\,\mu$m). The first four size fractions were separated by sieving while the PM_{10} fractions are the materials that were left over the filter (Jansz, 2002).

3.3.5 Meteorology and Dispersion condition

A dispersion function that relates emission rate with air concentration can be obtained by any of the following three methods: mass balance method, dispersion model method and tracer methods (Fitz and Bufalino, 2002; Lohmeyer et al., 2002; Venkatram, 2000). Considering accuracy and field logistics, dispersion model method is suggested to be a compromise between mass balance method and tracer experiments (Fitz and Bufalino, 2002; Venkatram, 2000). A dispersion model is a mathematical description of the transport and dispersion process of airborne materials due to interplay of meteorological parameters during a particular period and for a particular emission inventory (Painter, 1974). The standard models for estimating the time and spatial distribution of point sources of contamination in the atmosphere are the Gaussian statistical solutions of the atmospheric diffusion equation. These models are obtained from solution

of the classical differential equation for time-dependent diffusion in three dimensions. Pasquill (1961) has discussed the physical basis, analytical solutions, and the use of these equations. Turner (1970) and Hanna et al. (1982) have compiled workbooks on applications of these solutions to air pollution problems, including the application of the Gaussian models to area and line sources.

Atmospheric Dispersion Modelling System (ADMS) Urban version 1.6 (ADMS, 2000) is used to derive the dispersion function at Manor Road because of its easy accessibility and familiarity. The model has been developed by Cambridge Environmental Research Consultants (CERC), UK, to simulate the dispersion of a wide range of buoyant and passive releases to atmosphere. The atmospheric dispersion algorithms used by the model are based on current understanding of the atmospheric boundary layer. A Gaussian distribution is used to model vertical concentrations in the plume under neutral or stable atmospheric conditions, while a skewed Gaussian distribution is used for convective conditions. Horizontal concentrations are modelled using a standard Gaussian distribution. Plume spread is calculated using local boundary layer variables (ADMS, 2000). The model has been validated against field data from a number of sites in the UK and elsewhere and is used extensively in the UK. The model contains a set of predefined chemical species such as SO_2, NO_2, CO, PM_{10} and NO_x with the option of user adding more substances. Averaging time can be 15 minute, 1 hour, 8 hours, 1 day or 1 year. It can take care of point, line, area and volume sources. Five sets of parameters that are required as inputs to the model are set up details (site details and modelling options), sources details (source types, source locations, source dimensions and release conditions), meteorology (wind speed, wind direction, temperature, solar radiation and cloud cover), grids (type and size of grids for spatial output data) and outputs (output options).

Temperature, wind speed and wind direction data are taken from the measurement carried out at Manor Road. Solar radiation and cloud cover data for Greenwich observatory, the nearest place to Manor Road for which such data are available, obtained from the British Atmospheric Data Centre (BADC[7]) website, were used.

[7]BADC is the UK Natural Environment Research Council's (NERC) designated data centre for

3.4 Summary

Table 3.6 presents a comparison between the two sites.

Table 3.6: Comparison between field sites

Parameters	Gloucester Place	Manor Road
Legislative interpretation	PM_{10} level is high	An AQMA for PM_{10}
Street geometry	Street canyon	Flat terrain on industrial end; residential end is a street canyon
Road surface condition	Paved and well-maintained	Paved, but poorly maintained; visible pot-holes on road surface
Source origin	Exhaust and non-exhaust emission plus short-period initial gritting	Exhaust and non-exhaust emission plus continuous input of material from EGTS
Source area	A gritted road segment of known dimension	A road segment of known dimension to which significant amount of material is added from EGTS
Traffic	High traffic flow rate, proportion of heavy-duty vehicles is 10 %	Low traffic flow rate, proportion of heavy-duty vehicles is 17 %
Measurements		
Surface material loading	Amount is known from calibration of gritting lorry	Amount is estimated from dry and wet vacuuming
Particulate matter level	Grimm and P-Trak	TEOM
Traffic	SCOOT data and manual traffic count	Manual traffic count
Meteorology	Data is generated at site by using ultrasonic anemometers	Some data are generated at site and the rest are taken from BADC website
Dispersion relation	Obtained from SF_6 tracer study and CO_2 emission factor	Estimated using ADMS

the atmospheric sciences (BADC, 2003).

Chapter 4

Results of field measurements

4.1 Site 1: Gloucester Place

4.1.1 General condition

At the start of experiment, the sky was slightly overcast. With brief sunny spells, the weather stayed dry until shortly after the experiment was finished. At the roof level, the predominant wind direction was from East, varying between NE and SE (across the Gloucester Place from the East to the West) with the mean wind direction of 174° (Fig. 4.1a). Wind directions at street level, measured close to the facade of WCC, were largely from South (mean wind direction of 91°), a channelled flow along Gloucester Place in the direction of traffic flow (Fig. 4.1b). The highest wind speed recorded at the roof level was 8 m s^{-1}. However, most of the times it varied between 0 and 4 m s^{-1} (Fig. 4.1c). The mean, mode and 90^{th} percentile of the roof-level wind speed were 1.8 m s^{-1}, 1.4 m s^{-1} and 3.2 m s^{-1} respectively. The corresponding values for street-level wind speed were 1.6 m s^{-1}, 1.6 m s^{-1} and 2.6 m s^{-1}. The highest speed recorded at street level was 5.5 m s^{-1}. Most of the time the street-level wind speed varied between 0 and 3 m s^{-1} (Fig. 4.1d). Wind speed at street-level was intermittent and was characterised by periods of low speed, especially when the traffic wasn't moving. On average street-level wind speed was 70 % of the roof-level wind speed.

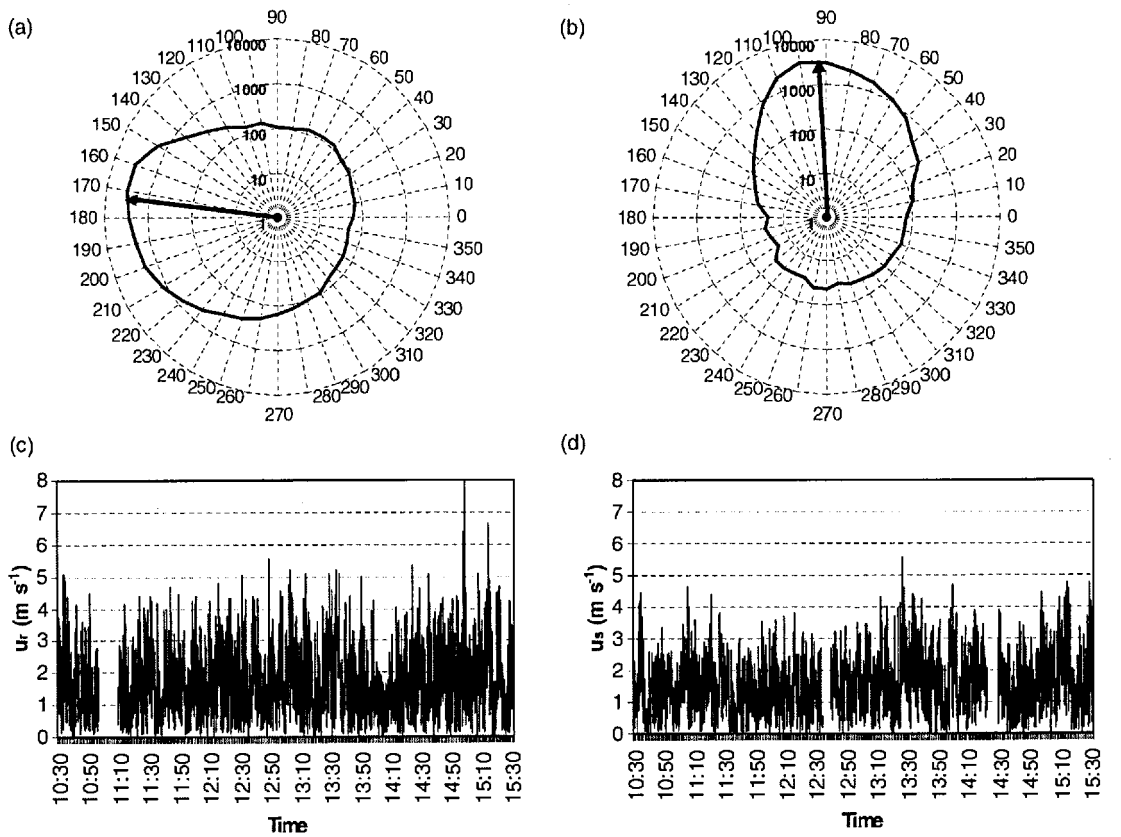


Fig. 4.1: Wind directions and speeds at Gloucester Place: (a) Wind direction at roof level; (b) Wind direction at street level (Arrow indicates mean wind direction. See Fig. 3.4 for orientation of u along 0° and v along 90°); (c) Wind speed at roof level; and (d) Wind speed at street level.

4.1.2 Traffic

Traffic on Gloucester Place consisted of a combination of motor cycles, cars, taxis, light goods vehicles (LGVs), heavy goods vehicles (HGVs), buses and coaches. The average traffic flow was 4800 vehicles per hour (i.e., 1600 per lane), buses/coaches and HGVs having a share of 10 % of the total flow. Hourly traffic flow and average traffic composition are presented in Fig. 4.2. Traffic flow was not continuous because of two traffic lights, marked by letter L in Fig. 3.4, that influenced the flow of traffic within the experimental site. The average durations of the traffic lights were 42 s of green phase and 47 s of red phase.

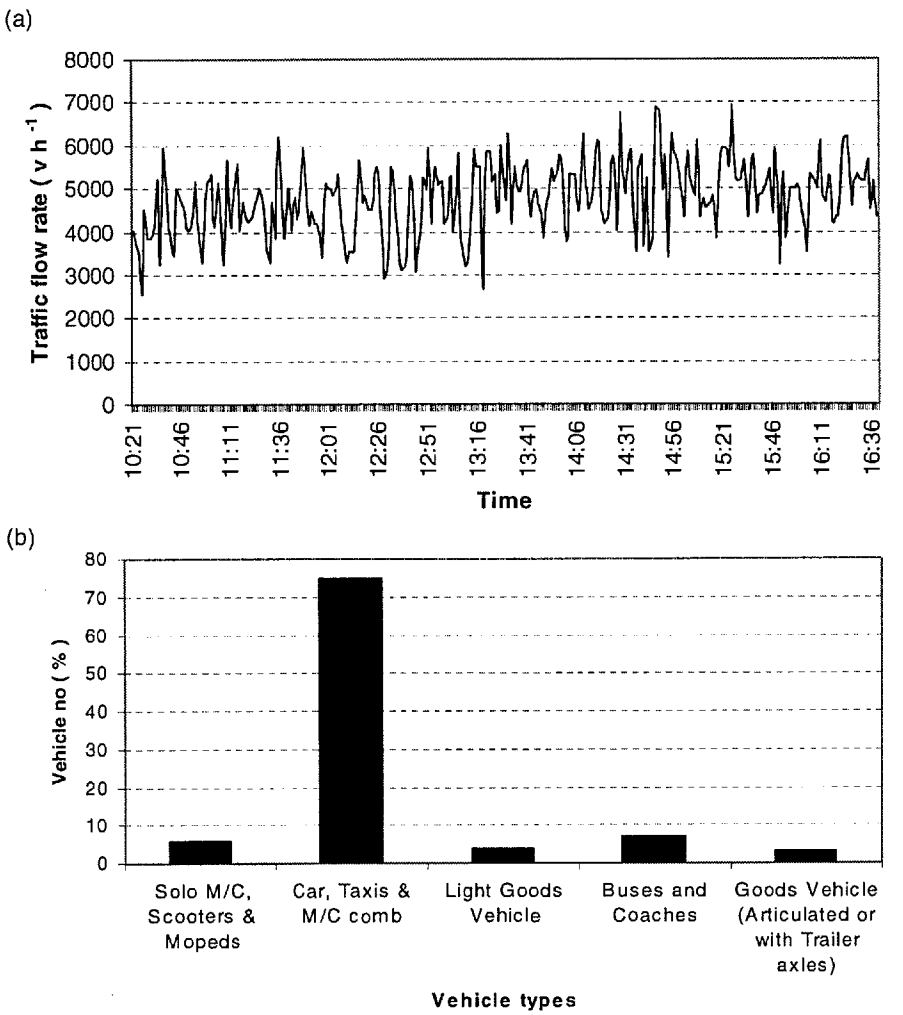


Fig. 4.2: Traffic at Gloucester Place: (a) Traffic flow rate; and (b) Traffic composition.

4.1.3 SF_6 tracer release

The first release of SF_6 took place at X_1, between 11:47 and 11:59 (Fig. 3.4). Bag samples were collected at each of the marked sampler positions (S_1, S_2, S_3 and S_4), on either side of the road, between 11:49 and 11:57 (i.e., an 8 min integrated sample). The second release of SF_6 took place at X_2, between 15:05 and 15:17, and samples were collected at S_1, S_2, S_3, S_4, S_5 and S_6, between 15:07 and 15:15 (i.e., an 8 min integrated sample). The release profiles are shown in Fig. 4.3.

The pressure drop across the release equipment corresponds to the amount of 1% SF_6 released over the integrated release period of 12 min. The pressure drop recorded

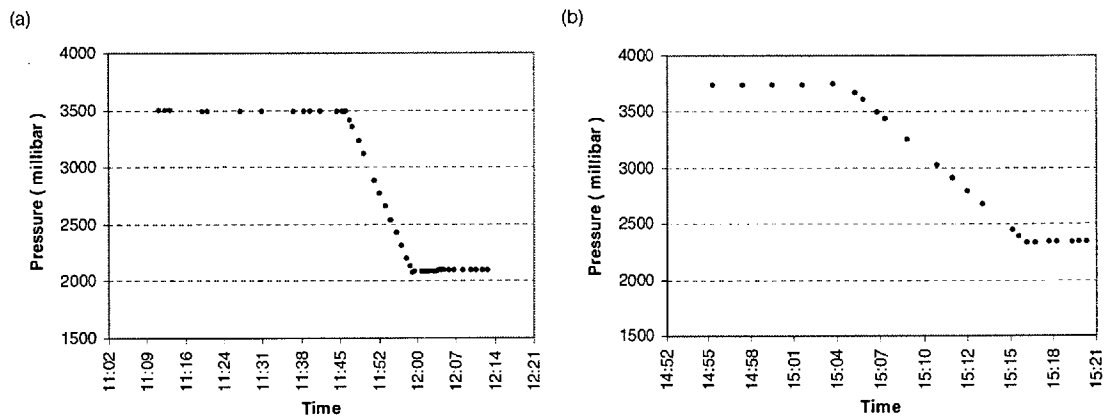


Fig. 4.3: Tracer release profiles: (a) First tracer release; and (b) Second tracer release.

across the 3000 ml release equipment was 1394 millibar for the first tracer release and 1406 millibar for the second tracer release (Fig. 4.3). At 20°C, the molar volume is 24.04 L. Molecular weight of SF_6 is 149.05. This equates to a release rate of 349 ml min^{-1} or 2.09×10^{-4} kg s^{-1} at 20 °C for the first tracer release and 352 ml min^{-1} or 2.11×10^{-4} kg s^{-1} at 20 °C for the second tracer release, calculated using Equation 4.1.

$$Q = \left(\frac{V_m M_w C_t}{60 \times 10^3 \times 10^3)}\right)\left(\frac{p\,v}{t_r}\right) \tag{4.1}$$

Where,

p = pressure drop in release equipment (bar)

v = volume of release equipment (ml)

t_r = tracer release time (min)

Q = SF_6 release rate at 20 °C (kg s^{-1})

V_m = molar volume at 20 °C (L)

M_w = molecular weight of SF_6

C_t = % concentration of SF_6 measured at locations S1 through S6

The results of tracer releases are summarised in Table 4.1.

Table 4.1: Results of tracer releases

Tracer release 1			
Sampling position	C_t (kg m^{-3})	Q (kg s^{-1})	$r_t = \frac{C_t}{Q}$ (m^{-3} s)
S_1	4.80×10^{-10}	2.09×10^{-4}	2.30×10^{-6}
S_2	6.65×10^{-9}	2.09×10^{-4}	3.18×10^{-5}
S_3	2.68×10^{-9}	2.09×10^{-4}	1.28×10^{-5}
S_4	9.22×10^{-8}	2.09×10^{-4}	4.41×10^{-4}
Tracer release 2			
Sampling position	C_t (kg m^{-3})	Q (kg s^{-1})	$r_t = \frac{C_t}{Q}$ (m^{-3} s)
S_1	1.21×10^{-9}	2.11×10^{-4}	5.73×10^{-6}
S_2	3.17×10^{-9}	2.11×10^{-4}	1.50×10^{-5}
S_3	3.85×10^{-9}	2.11×10^{-4}	1.82×10^{-5}
S_4	4.41×10^{-9}	2.11×10^{-4}	2.09×10^{-5}
S_5	7.64×10^{-9}	2.11×10^{-4}	3.62×10^{-5}
S_6	1.10×10^{-9}	2.11×10^{-4}	5.21×10^{-6}

The tracer measurements provide values of the dispersion relation between point source emissions and six point receptors. Assuming the dispersion over a 12-minute average is similar for parallel source-receptor vectors within a distance of about 10 m of the measured source-receptor positions, source-receptor relationships can be mapped for unit emission as a function of distance and direction, by interpolation between the six measured source-receptor vectors. The assumption of similar dispersion along parallel vectors is reasonable in this case because of the small angle between the wind direction and the orientation of the street. Integration of the resulting map then gives the contribution of an area source to concentrations at a single receptor.

The SF_6 measurements show that the majority of the concentration above background levels can be attributed to emissions within a distance of 5 m of each roadside measurement location (Fig. 4.4). Considering homogeneous area sources, Equation 4.2 has been used to estimate dispersion relation:

$$\chi = \int r_t \, dA \tag{4.2}$$

Where,

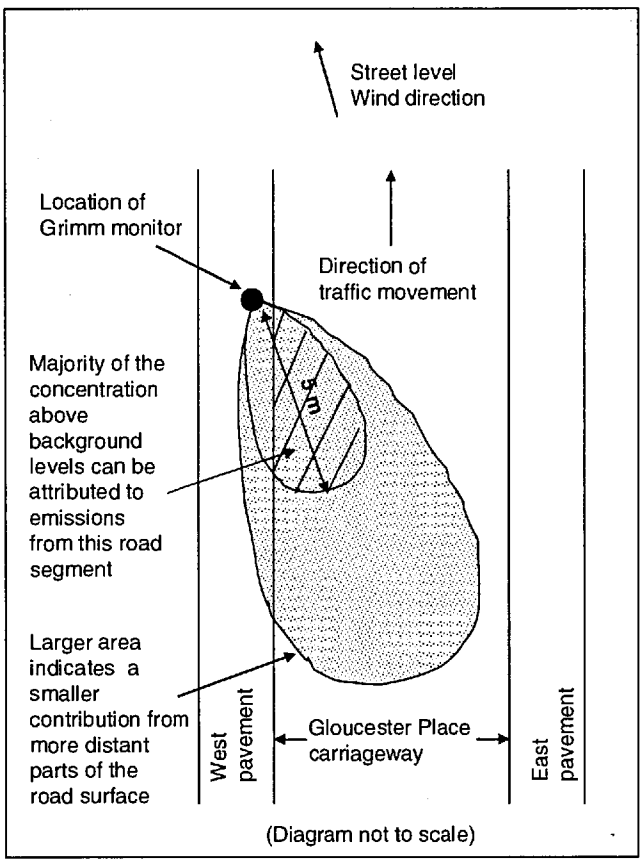


Fig. 4.4: Location of source area contributing to elevated particle concentration in air sampled by Grimm monitors.

χ = dispersion coefficient i.e., ratio between emission from a section of a road and concentration at the receptor (m^{-1} s)

r_t = concentration-emission ratio obtained from the tracer release (m^{-3} s)

A = source area (i.e., area of a section of a road, emissions from which contribute to the receptor) (m^2)

The estimates show that the dispersion conditions on both occasions of tracer release were not the same. For the first tracer release, χ values are 6.0×10^{-2} m^{-1} s and 7.1×10^{-2} m^{-1} s for receptors at Site 1 and Site 3 respectively. For the second release, the χ values are 5.7×10^{-3} m^{-1} s and 7.2×10^{-3} m^{-1} s for receptors at Site 1 and Site 3 respectively. It suggests that dilution of pollutants in air was 10 times greater during the second tracer release than the first one. This localised dispersion pattern is dictated by combined influence of flow of wind in street (which, in effect, is influenced

by the building geometry on both side of the street) and turbulence generated from traffic movement.

The dispersion relations derived from SF_6 tracer release measurements are found to be consistent with the one estimated from traffic data and expected emissions of carbon dioxide (CO_2), estimated by using CO_2 emission factor from the National Atmospheric Emissions Inventory (NAEI) database (http://www.naei.org.uk). The observed difference between the two is smaller for the second gas tracer release than the first. We, therefore, decided to use the dispersion coefficient obtained from second tracer release in subsequent calculations. Wind direction, tracer release point and location of receptor suggest that, out of the two χ values obtained for two receptors during second tracer release, the one between X2 and G1 ($\chi = 5.7 \times 10^{-3}$ m^{-1} s) is the appropriate one.

4.1.4 Qualitative observations of grit movement

The following on-field qualitative observations were made during the experiment:

1. The dust clearly moved in a way that was broadly similar to that expected in designing the experiment, but with some surprises. The initial movement was much faster than expected, followed by a slower decay and final accumulation of mostly coarser material along the east side of the carriageway. The dust provided visualisation of the large but very localised effect of bus exhaust gas exit velocity.

2. A visible plume of dust followed the vehicle. After about an hour, clear build-up of particulate at the edge of the road and deposition of grit on the pavements were seen. After one and half hours, no appreciable grit on road surface was seen, indicating most of resuspendible grit being removed by that time from the road surface. This was also corroborated by the fact that no visual plume was now seen after any light vehicle pass.

3. However when the buses and coaches passed over the gritted segment, visible plumes were still observed. This may be the grit locked within the interstices

on the road surfaces which were difficult to entrain into the air due to small turbulence created by the movement of light vehicles, but got entrained into the air by the movement of vehicles with more turbulent vehicle wake coupled with greater disturbance of the road surface by the weight of the vehicle. Similar observations have been reported from earlier studies (AQEG, 2004; Nicholson and Branson, 1990; Sehmel, 1973, 1976b). Larger vehicles also had tyre tracks closer to the kerb where accumulation of dust was most persistent and therefore, suspended this more frequently.

4. After about three hours, the road surface appeared clean, except for a small sand dune like feature along the windward facing kerb indicating a process that caused flow of grit across the road.

4.1.5 Roadside observation of exhaust and non-exhaust emissions

Fig. 4.5 shows the variability in wind speed and traffic flow compared with variability in roadside CO_2 and fine particle concentrations that are expected to be associated with exhaust emissions. Only the particles 0.75 μm to 1 μm measured in the smallest size channel of the Grimm show a response albeit weak to the application of grit at 11:15 (Fig. 4.5d).

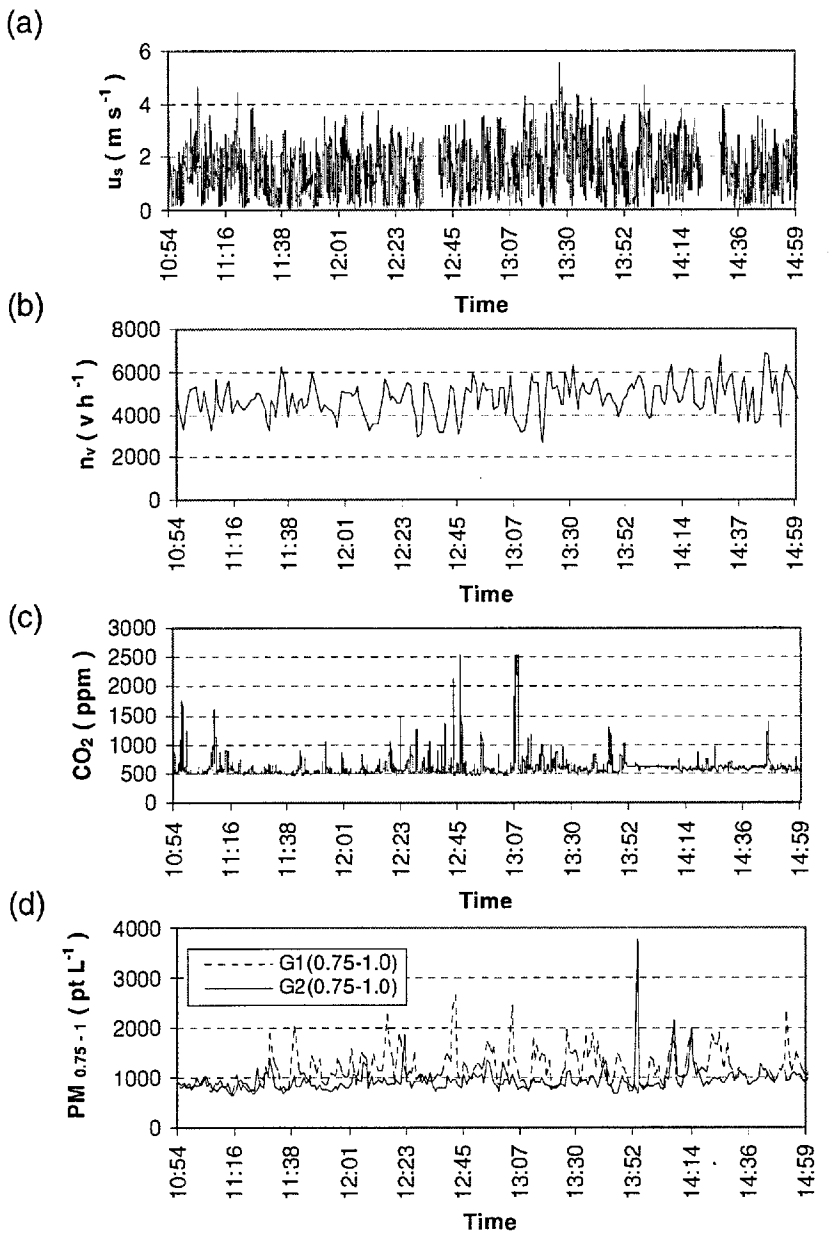


Fig. 4.5: (a) Wind speed; (b) Traffic flow rate; (c) CO_2 concentration; and (d) Concentration of $PM_{0.75-1}$.

Fig. 4.6 shows in more detail a period of time when the P-Trak, Grimm and CO_2 monitors were all working simultaneously, indicating how it is difficult to relate individual peaks to specific wind speed or traffic flow changes, but that the generally homogeneous noise in the data are all similar when the different averaging times of the various measurements are taken into account.

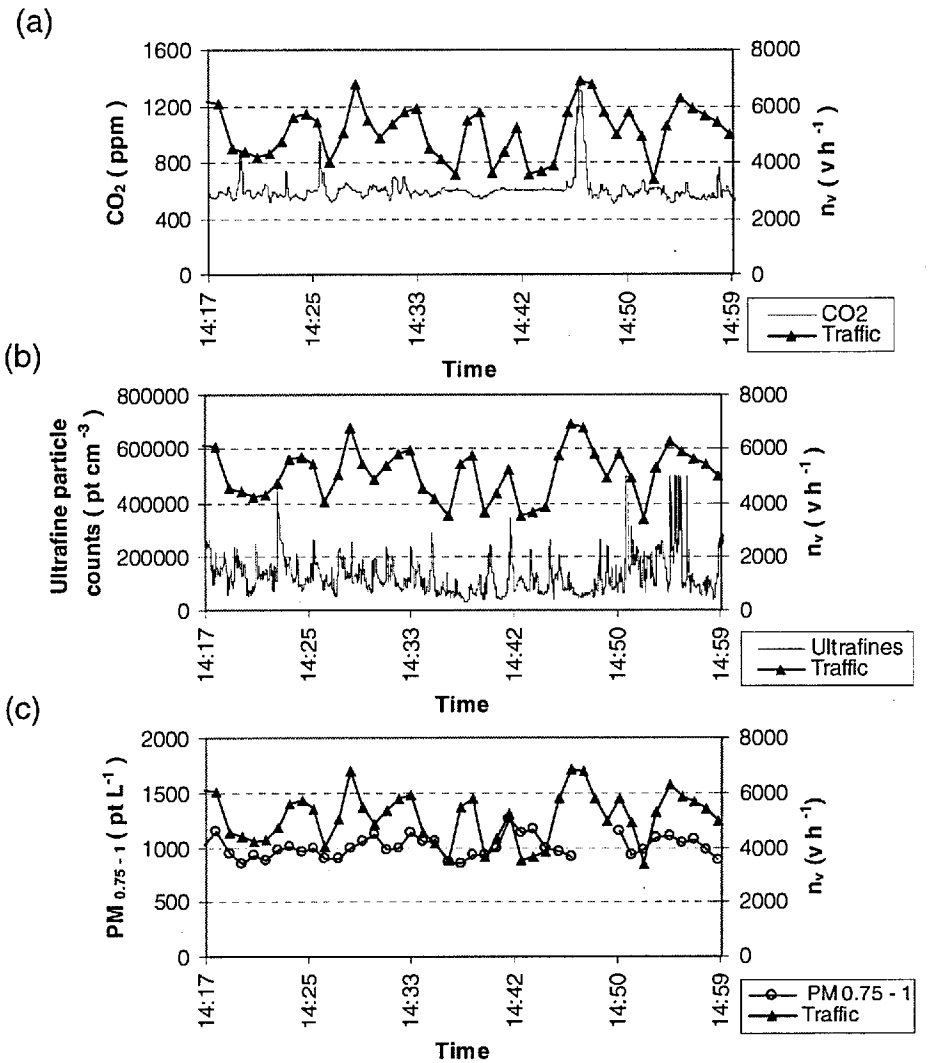


Fig. 4.6: (a) Traffic and CO_2; (b) Traffic and ultrafine particles; and (c) Traffic and $PM_{0.75-1}$.

Fig. 4.7 shows that the progressively coarser particle size fractions counted in the higher channels of the Grimm monitor show a very different pattern, with increasing response to the gritting event with increasing particle size. Most significantly, immediately after gritting, the levels of particles in air at Site 2 increased as follows: 0.75 μm to 1 μm by a factor of 1.5, 1 μm to 2 μm by a factor of 1.5, 2 μm to 3.5 μm by a factor of 2.5, 3.5 μm to 5 μm by a factor of 6.5, 5 μm to 7.5 μm by a factor of 9, 7.5 μm to 10 μm by a factor of 13, 10 μm to 15 μm by a factor of 15 and >15 μm by a factor of 30. After some time, it was observed that grit had moved along the road in the direction of traffic. The leading edge c-d had moved to a new position C-D, relatively more

distance than the movement of trailing edge, which moved from a-b to A-B (Fig. 3.4). Irrespective of particle sizes, the particulate matter level at Site 2 had come down to its pregritting level within the period of experiment. However, varying rates of decay for different particle sizes are observed. Particulate matter larger than $2\,\mu$m follows an exponential decay pattern (Figs. 4.7b, c, d, e & f). For very fine particles up to $2\,\mu$m, no clear decay pattern is observed at Site 2. However, for particles of this size range, differences in patterns between Site 1 and Site 2 are observed. While at Site 2, the particulate matter level came down to pregritting level within 2.5 hours, at Site 1, they were at a level higher than the pregritting level by a factor of 2 to 2.5 until the end of the experiment at 15:40 (Figs. 4.5d & 4.7a). This indicates addition of fine particles to road surface at Site 1 due to grinding of material by traffic as the material moves along the road from the gritted site to the ungritted site. With increase of particle size, the ratio of first peak particulate matter levels immediately after gritting at Site 1 and Site 2 decreases from >1 to <1, indicating little contribution of grinding to particles larger than $7.5\,\mu$m. These observations are from G1 and G2, following instrument failure of G3 and G4, although the measurements of ultrafine particles at Site 1 and Site 3 suggest concentration of G3 and G4 may be a factor of 2 to 5 higher than G1 and G2 respectively.

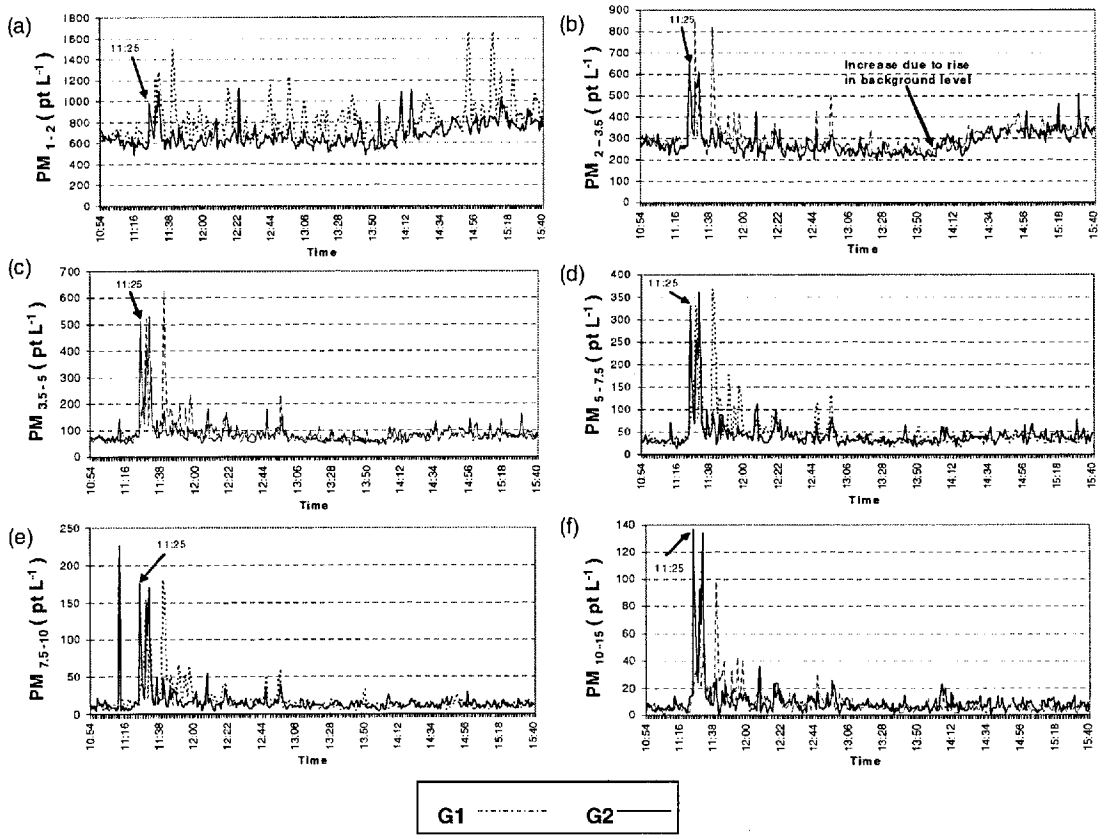


Fig. 4.7: Decay rates of particulate matter level as a function of particle size.

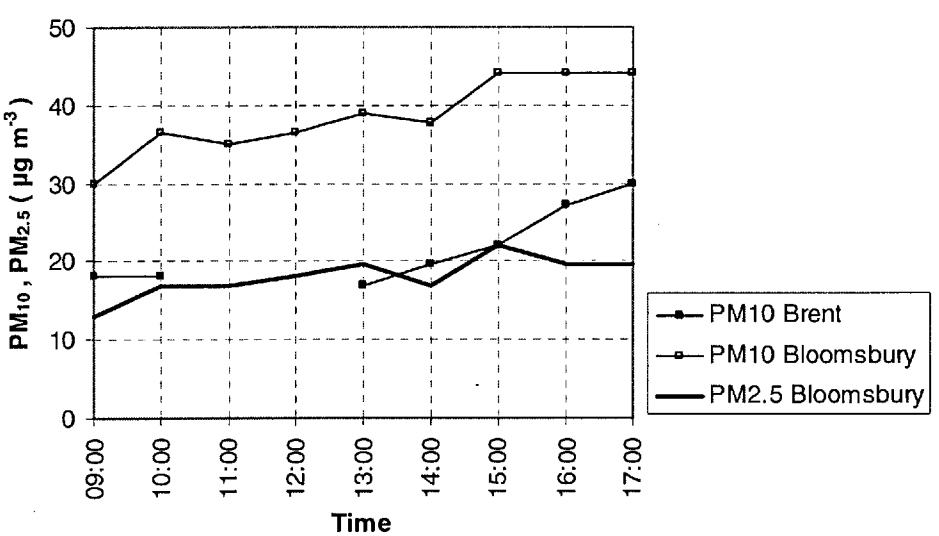


Fig. 4.8: PM_{10} and $PM_{2.5}$ levels on 26 May 2004 at urban background stations located close to Gloucester Place.

There was also a rise in background levels of mainly $PM_{2.5}$ fraction in the latter

part of the experiment that is largely obscured by the local exhaust contribution to the finest particles (Fig. 4.5d) but seen most clearly in the 1 μm to 3.5 μm range (Figs. 4.7a & b), consistent with data from two nearby London urban background monitoring sites (Fig. 4.8).

4.1.5.1 Roadside and background PM_{10} concentration in air

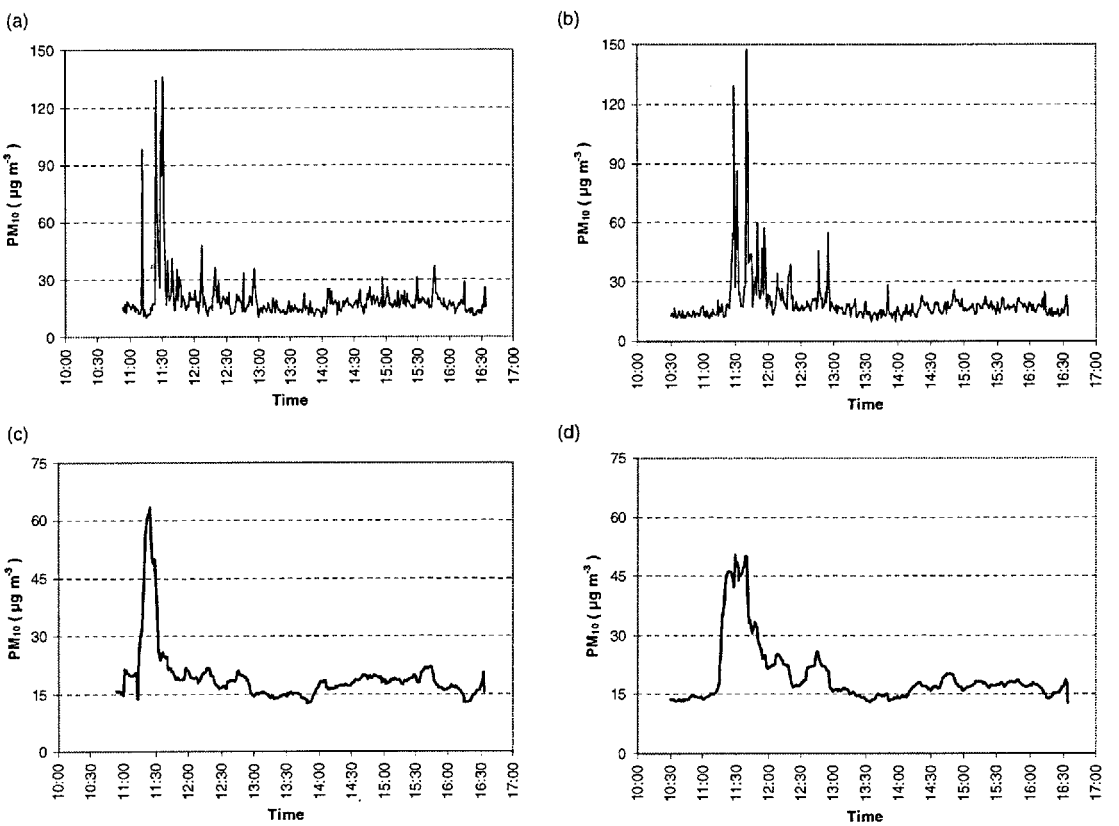


Fig. 4.9: (a) PM_{10} level at the gritted site; (b) PM_{10} level at the ungritted site; (c) 12-minute averaged PM_{10} level at the gritted site; and (d) 12-minute averaged PM_{10} level at the ungritted site.

In order to relate the mass of the material on road surface with the roadside measurement of particulate matter concentration and also to evaluate the performance of the model developed in Chapter 5, measured particle number concentration need to be converted to gravimetric units. For this, assumptions for "average diameter" and "effective density" of particle for each size are necessary. Median size of each particle size range is assumed as the average diameter for the particles in that size range; i.e.,

average diameters of 0.87 μm, 1.5 μm, 2.75 μm, 4.25 μm, 6.25 μm and 8.75 μm for particles with size 0.75 μm to 1 μm, 1 μm to 2 μm, 2 μm to 3.5 μm, 3.5 μm to 5 μm , 5 μm to 7.5 μm and 7.5 μm to 10 μm respectively. Effective density of the particle depends on particle composition and shape. Assuming spherical particles with effective density of 1×10^6 g m^{-3}, an assumption similar to that made during earlier experiments of this nature (Nicholson and Branson, 1990; Nicholson, 1993), the PM_{10} levels in gravimetric units at Site 1 and Site 2 are presented in Fig. 4.9.

Figs. 4.9a & b show the variation of PM_{10} in each minute. The 12-minute averaged PM_{10} time-series, shown in Figs. 4.9c & d, are generated to compare it with the model output that is obtained by using a 12-minute averaged dispersion relation obtained from tracer study (Section 4.1.3). The rise of PM_{10} levels from 13:54 until about the end of the experiment duration is caused by the rise of background level, discussed earlier in Section 4.1.5.

Data from two nearby London urban background monitoring sites (London Brent and London Bloomsbury) show that the PM_{10} levels at these sites are higher than the local background PM_{10} at Gloucester Place (Fig. 4.8). Therefore the lowest pre-gritting PM_{10} level of about 10 μg m^{-3} (Figs. 4.9a & b) is considered as the background level for Gloucester Place.

4.1.5.2 Particulate matter decay rate

From the tracer study it is observed that material on road surface within about 5 m of each roadside measurement location contributes to the majority of the concentration above the background levels (Section 4.1.3). Therefore assuming the excess airborne concentration at a location is proportional to the mass of the material on the road surface within about 5 m of each roadside measurement location, the exponential decay pattern of airborne particulate matter larger than 2 μm, discussed earlier in Section 4.1.5, can equally be applicable for the material decay on the road surface. Therefore the material decay on the road surface can be expressed as follows:

$$m_t = m_0 \, e^{-\left(\frac{t}{T_0}\right)} \tag{4.3}$$

Where,

m_t = amount of material on road at time t (g)

m_0 = amount of material on road at the end of the gritting (g)

T_0 = time taken for the material on the road to decay by a factor of e^{-1} (s)

It takes about an hour from the end of gritting for airborne particles with size >15 μm to come down to its pregritting level. This time increases with decreasing particle size until particles with size 2 μm to 3.5 μm taking about 1 hour and 30 minutes (Fig. 4.10a). Therefore, the rate of decay increases with increasing size of particulate matter ; T_0 decreases from 15 minutes for 2 μm to 3.5 μm particles to 9 minutes for >15 μm particles (Fig. 4.10b).

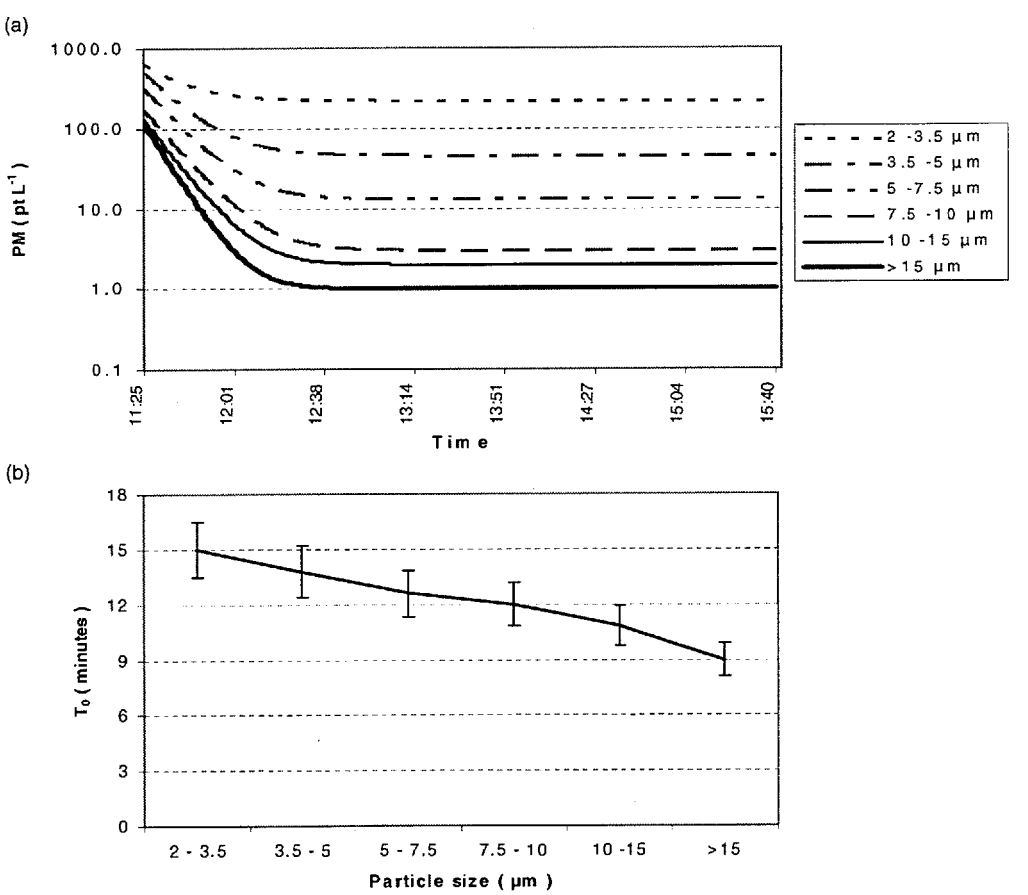


Fig. 4.10: (a) Decay of particulate matter level as a function of time; and (b) Decay time as a function of particle size.

4.1.6 Movement of material along the road

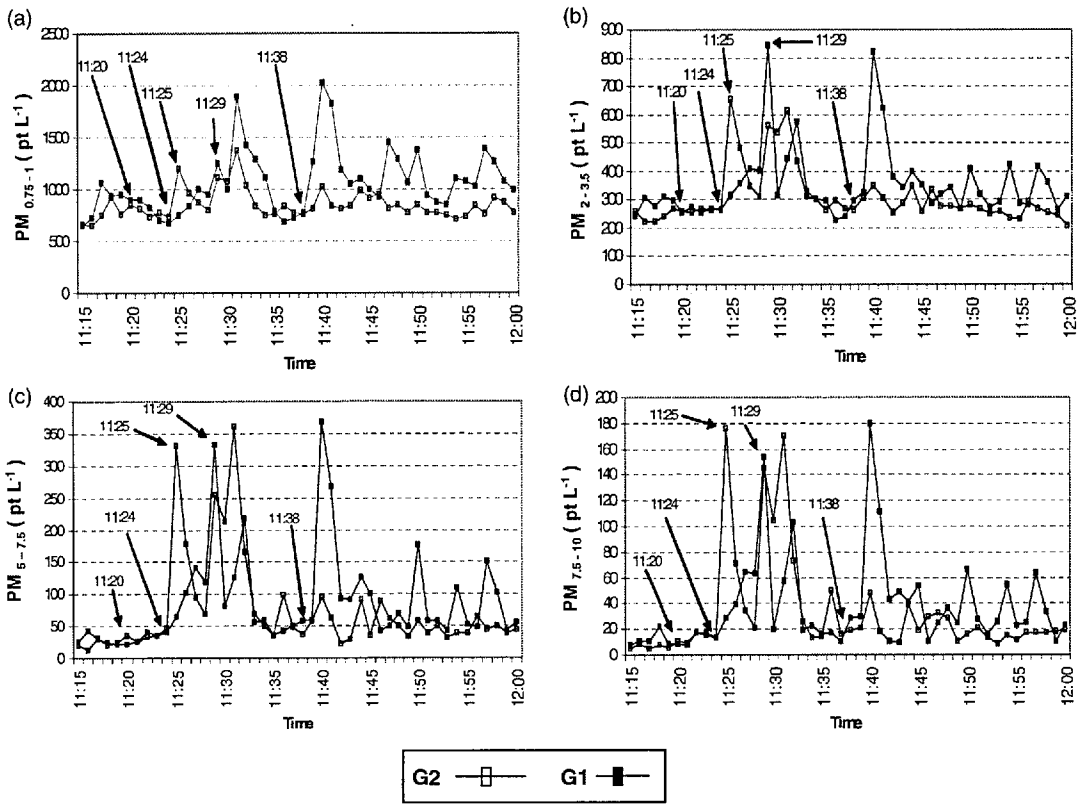


Fig. 4.11: Arrival times of airborne particulate matter downstream of initially gritted section of road.

Fig. 4.11 shows that, irrespective of particle sizes, the pattern of material movement along the road, during the first 30 minutes after gritting, was similar. Although gritting was over by 11:20, particulate matter levels recorded by both G2 and G1 were on the decline or at same general level until 11:24, suggesting no effect of gritting on particulate matter level at road side air until that time. Most of this delay is expected since the gritting was carried out during a point in the traffic signal sequence where only a few vehicles were travelling along Gloucester Place. At 11:24, G2 recorded a sharp rise of particulate matter level while particulate matter level at Site 1 was yet to be influenced by gritting. This indicates that it took less than 4 minutes for the grit to become airborne and reach the pavement right across the gritted area. It is likely that air emissions of grit commenced earlier but wind at that point of time was blowing away from the receptor causing some of this delay to happen. With a time

delay of about 1 minute, at 11:25, the effect of gritting was seen at Site 1 and at the same time the particulate matter level at Site 2 reached its maximum. Then, for the next 15 minutes since the first arrival of grit at Site 1, there followed a period of rapid material transfer along the road from gritted section (abcd) to an initially ungritted section (dcCD), between Salisbury Place and Marylebone Road, along the direction of traffic movement. At about 11:38, the particulate matter level of G1 exceeded the general particulate matter level of G2, thus indicating that much of material has now moved from an initially gritted section (abcd) and that material on the road in the new section (dcCD) is more than that of initially gritted section (abcd).

4.2 Site 2: Manor Road

4.2.1 Meteorology

During the study period of 9 weeks between 10 June 2002 and 10 August 2002[1], the highest wind speed on Manor Road was 5.2 m s^{-1}. However, most of the time it varied between 1 m s^{-1} and 3 m s^{-1} (Fig. 4.12a). The mean, mode and 90^{th} percentile of the wind speed were 2 m s^{-1}, 1.8 m s^{-1} and 3.4 m s^{-1} respectively. The predominant wind direction at Manor Road was from south-west (Fig. 4.12b). Events of rain are recorded in first, second, fourth, fifth, seventh, eighth and last weeks of the study period (Fig. 4.12d). The rainfall rate increased from the first week until the penultimate week which recorded highest rainfall of 11.8 mm h^{-1}. Effect of rainfall on reduction of PM_{10} is evident from Figs. 4.12d & e.

[1]Study period includes 3 weeks before and 1 week after the 5-week period for which the data on surface material loading are available from a previous study.

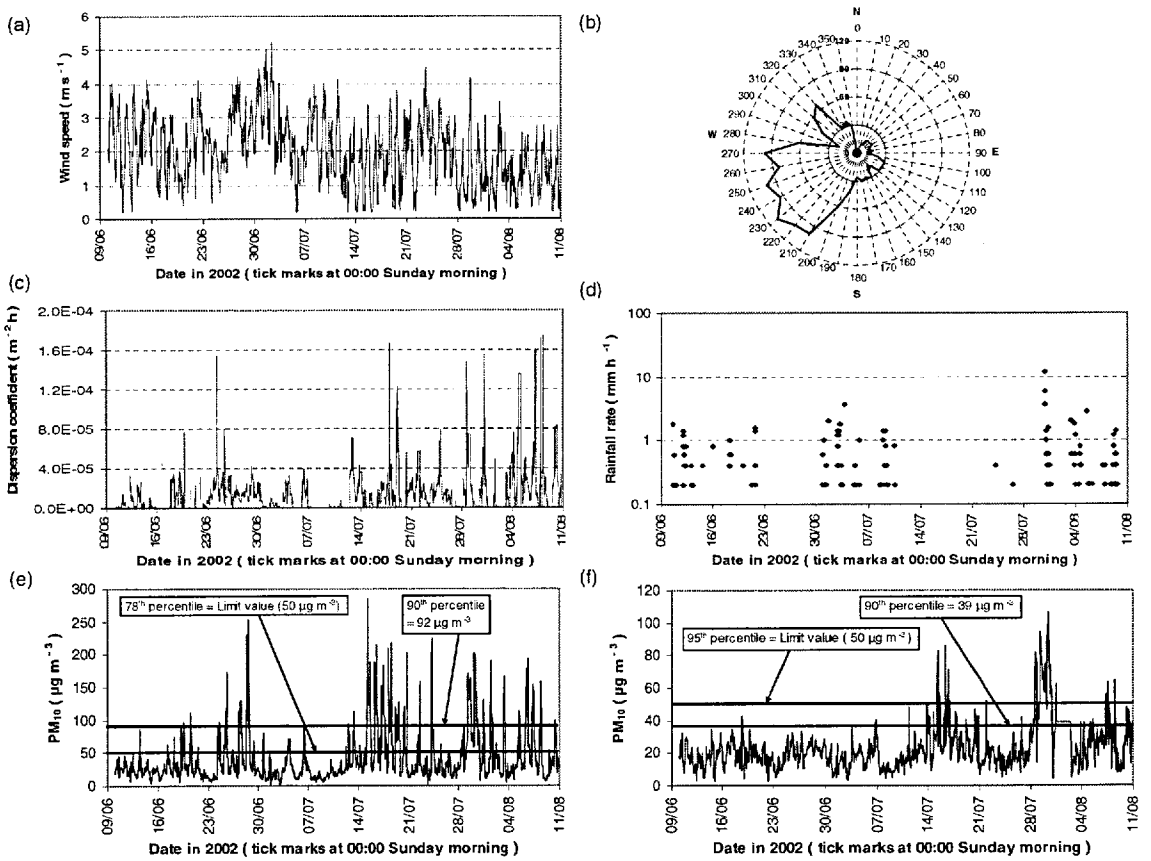


Fig. 4.12: Meteorology and PM_{10} at Manor Road: (a) Wind speed; (b) Wind direction (Arrow indicates mean wind direction); (c) Dispersion condition; (d) Rainfall; (e) PM_{10} at Manor Road; and (f) Background PM_{10} for Manor Road.

4.2.2 Dispersion relation

The dispersion coefficient is estimated from ADMS for each hour of the period studied. Data on solar radiation and cloud cover are unavailable for the model simulation period and in absence of this, cloud cover value is taken as 8 i.e., the sky is assumed to be completely covered with cloud and solar radiation playing no role in dispersion. So the dispersion process is largely influenced by local wind and associated atmospheric turbulence. There are no high buildings at the roadside at the industrial end of the road where the TEOM is located. Therefore a flat terrain is assumed for road geometry. Receptor height for ADMS is 3 m, same as the height of measuring probe of the TEOM.

The highest estimated dispersion coefficient value was $1.8 \times 10^{-4}\,m^{-2}\,h$. However,

on many occasions it was less than $4 \times 10^{-5}\,m^{-2}$ h (Fig. 4.12c) with mean of $1.3 \times 10^{-5}\,m^{-2}$ h and 90^{th} percentile of $3.3 \times 10^{-5}\,m^{-2}$ h. Association of high wind speed with low dispersion coefficient (indicating high dispersion) and low PM_{10} level shows the consistency of the meteorological and PM_{10} data (Figs. 4.12a, c & e).

4.2.3 Particulate matter concentration in air

The 90^{th} percentile of hourly PM_{10} at Manor Road was 1.8 times the limit value of $50\,\mu g\ m^{-3}$ prescribed in the UK Air Quality Standard (Fig. 4.12e). The highest level of PM_{10} recorded during the study duration was $285\,\mu g\ m^{-3}$. On 22% occasions the PM_{10} level exceeded the limit value $50\,\mu g\ m^{-3}$. This is equivalent to 47 exceedances in a 9-week period for which this study is conducted as against the permissible exceedances of 35 exceedances a calender year. In spite of the background level of PM_{10} recorded as high as $107\,\mu g\ m^{-3}$, the 95^{th} percentile of hourly PM_{10} was within the limit of $50\,\mu g\ m^{-3}$ (Fig. 4.12f).

4.2.4 Traffic

On a day, typically, 12000 vehicles travel on Manor Road. Hourly traffic flow for a week along half the carriageway of Manor Road, for which the modelling exercise is carried out, shows that traffic on Friday is the highest among all days (Fig. 4.13a). On each day a morning peak at 8:00 and an evening peak at 17:00 are observed. Traffic flow from EGTS to Manor Road is presented in Fig. 4.13b. On average 118 vehicles enter Manor Road from EGTS on a weekday, out of which 103 HGVs loaded with waste material move west towards Erith and 15 vehicles travel east towards Slade Green. On weekends, 51 HGVs carry waste towards Erith during the limited opening time of EGTS between 8:00 and 13:00. The HGVs comprise 17% of the traffic on Manor Road (Fig. 4.13c).

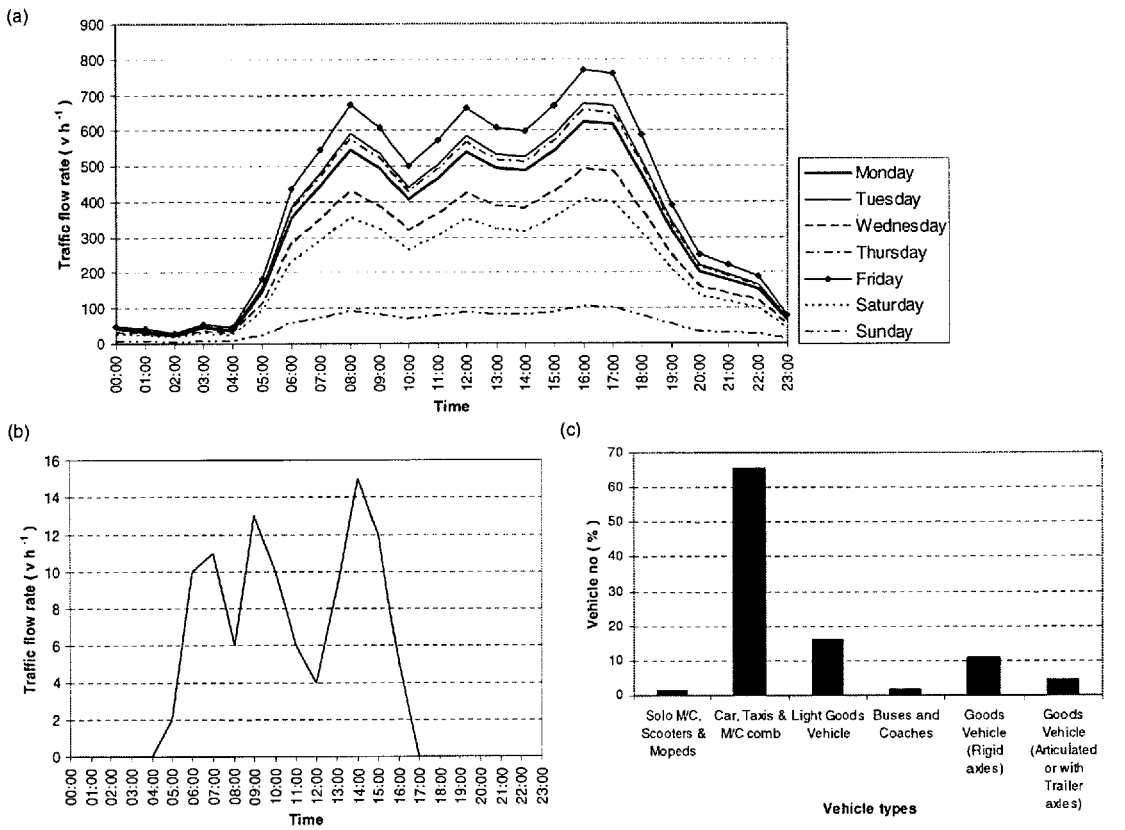


Fig. 4.13: (a) Traffic flow on Manor Road towards Erith; (b) Traffic flow from EGTS to Manor Road; and (c) Traffic composition on Manor Road.

4.2.5 Road surface loading

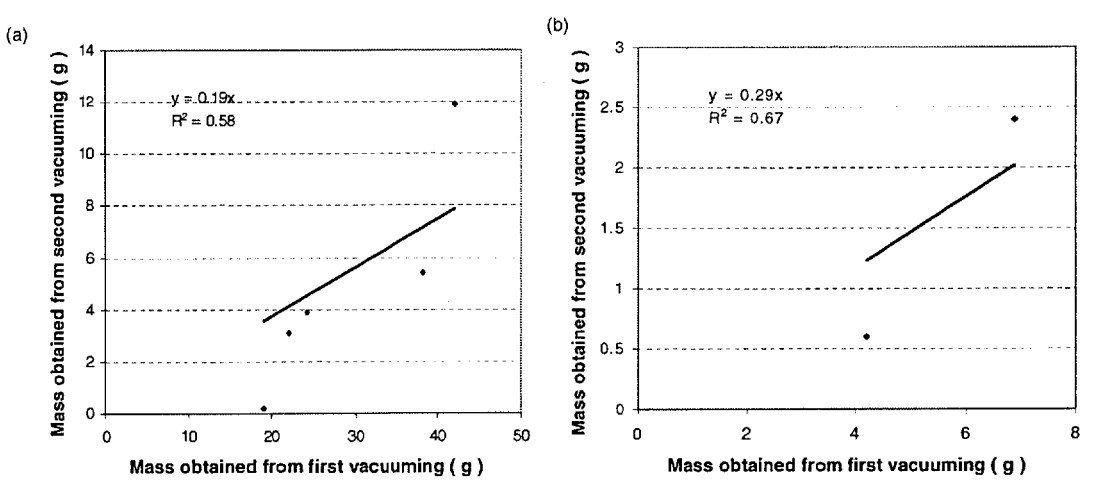


Fig. 4.14: Vacuuming efficiency of particles $< 63\,\mu$m at Manor Road: (a) Sampling location at industrial end; and (b) Sampling location at residential end.

Surface loadings, after correction for second vacuuming, from a road segment of 0.5 m^2 (Section 3.3.4) were obtained from an earlier study (Jansz, 2002). Out of the different size fractions to which the samples were segregated, particles below the size of 63 μm are assumed to contribute to PM_{10} emissions. This size is a little smaller than EPA defined silt size of 75 μm that is assumed to be proportional to PM_{10} emissions. With little knowledge of how grinding action by traffic changes the size of particles, this assumption appears to be reasonable. The second vacuuming accounts for 19 % and 29 % of additional mass for particles below the size of 63 μm at industrial and residential ends respectively (Fig. 4.14).

Table 4.2: Surface loading and silt loading on Manor Road

Date	Location	Surface loading ($g\ m^{-2}$)	Silt loading ($g\ m^{-2}$)	$\frac{\text{Silt loading}}{\text{Surface loading}}$
29 June 2002	Industrial end	304.8	84.1	0.28
06 July 2002	Industrial end	369.2	103.2	0.28
13 July 2002	Industrial end	245.0	34.7	0.14
20 July 2002	Industrial end	297.6	65.6	0.22
	Residential end	82.6	15.2	0.18
27 July 2002	Industrial end	253.6	57.5	0.23
	Residential end	60.2	8.8	0.15

The surface loading and silt loading after correction for second vacuuming are multiplied by 2 to estimate their values for a road segment of 1 m^2. Estimates of surface loading and silt loading are presented in Table 4.2. The estimates show that on average the silt loading is 0.23 times and 0.16 times of the surface loading at the industrial end and at the residential end respectively.

4.3 Quantitative analysis of dust movement

In this section various parameters that describe the particle movement and emission processes and, therefore, account for the results summarised in Sections 4.1.4, 4.1.5 & 4.1.6, have been estimated. The quantitative analysis is done prior to modelling only on Gloucester Place, since quantitative analysis on Manor Road had already been

carried out in previous works (Section 3.3.1.1).

4.3.1 Flux along the road

Section 4.1.6 suggests that there exists a process of material movement along the road that has caused material to move from the gritted section to the initially ungritted section in the direction of traffic. Visual observation indicates involvement of two processes. While the vehicle wake may be a dominant process for some movement of suspended particles, material sticking to the outer surface of the wheels and thus carried along the road is a more likely mechanism for movement of material that is not already airborne. From the along-street particle movement time measurements described in Section 4.1.6, different speeds with which material moves along the road are estimated as follows:

$$u_{x\max} = \frac{x_{1-2}}{t_{x\min}} \tag{4.4}$$

$$u_{x\text{avg}} = \frac{x_{1-2}}{t_{x\text{avg}}} \tag{4.5}$$

$$u'_x = \frac{u_{x\text{avg}}}{n_\text{v}} \tag{4.6}$$

Where,

$u_{x\max}$ = maximum speed of material movement along the road by the whole fleet of vehicles (m s^{-1})

$u_{x\text{avg}}$ = average speed of material movement along the road by the whole fleet of vehicles (m s^{-1})

u'_x = average distance of material movement along the road by one vehicle (m v^{-1})

x_{1-2} = distance between Site 1 and Site 2 (m)

$t_{x\min}$ = minimum time elapsed between end of gritting and first arrival of grit at Site 1 (indicated by first peak of G1) (s)

$t_{x\text{avg}}$ = time elapsed between end of gritting and time when general level of grit in air at Site 1 exceeded that at Site 2 (indicated by the peaks of G1 consistently higher than the peaks of G2) (s)

n_v = traffic flow rate (v s^{-1})

Substituting measured values into the Equations 4.4, 4.5 & 4.6 gives an estimate of 0.11 m s^{-1}, 0.06 m s^{-1} and 0.05 m v^{-1} for $u_{x\max}$, $u_{x\text{avg}}$ and u'_x respectively. Using the traffic flow variability along the Gloucester Place (Section 4.1.2), u'_x has a range of 0.03 m v^{-1} to 0.09 m v^{-1}. Assuming 5.6 m s^{-1} (equivalent to 20 km h^{-1}) as the average speed of vehicle along the Gloucester Place, the average speed of the material movement along the road is about 1.1 % of the vehicle speed and 3.8 % of the average street-level wind speed.

Therefore, material removal fraction along the road is estimated as follows:

$$\begin{aligned} f_x &= \frac{f'_x}{S\,l\,w} \\ &= \frac{F_x}{n_v\,S\,l\,w} \\ &= \frac{u_{x\text{avg}}}{n_v\,l} \\ &= \frac{u'_x}{l} \end{aligned} \tag{4.7}$$

Where,

f_x = fraction of material removed from a road segment along the road by one vehicle (v^{-1})

f'_x = amount of material removed from a road segment along the road by one vehicle (g v^{-1})

F_x = amount of material removed from a road segment along the road in unit time by the whole fleet of vehicles (g s^{-1})

$u_{x\text{avg}}$ = average speed of material movement along the road by the whole fleet of vehicles (m s^{-1})

u'_x = average distance of material movement along the road by one vehicle (m v^{-1})

n_v = traffic flow rate (v s^{-1})

S = surface material loading on the road segment (g m^{-2})

l = length of the road segment (m)

w = width of the road segment (m)

Substituting measured values into the Equations 4.7 gives an estimate of $3.9 \times 10^{-4}\,\mathrm{v}^{-1}$ for f_x. Using the time-series of traffic flow along the Gloucester Place, f_x has a range of $2.6 \times 10^{-4}\,\mathrm{v}^{-1}$ to $7.1 \times 10^{-4}\,\mathrm{v}^{-1}$.

4.3.2 Flux across the road

After about an hour of the gritting event, a clear build-up of material along the kerb was observed (Section 4.1.4). It stretched through the entire length of the gritted road segment. The build-up at the downwind kerb was sufficient enough to make an approximate visual estimate of dimensions as 20 mm wide and 2 mm deep. Without cross-street wind, build-up would have been seen on both sides of the road, but in the presence of cross-street wind all the material was observed on the downwind side. Therefore, material removal fraction across the road is estimated as follows:

$$\begin{aligned} f_y &= \frac{f'_y}{S\,l\,w} \\ &= \frac{F_y}{n_\mathrm{v}\,S\,l\,w} \\ &= \frac{\rho\,w_y\,d_y}{t_y\,n_\mathrm{v}\,S\,w} \end{aligned} \tag{4.8}$$

Where,

f_y = fraction of material removed from a road segment across the road by one vehicle (v^{-1})

f'_y = amount of material removed from a road segment across the road by one vehicle (g v^{-1})

F_y = amount of material removed from a road segment across the road in unit time by the whole fleet of vehicles (g s^{-1})

ρ = particle density (g m^{-3})

S = surface material loading on the road segment (g m^{-2})

l = length of the material build-up along the kerb (same as the length of gritted segment) (m)

w = width of the road segment (m)

w_y = width of the material build-up along the kerb (m)

d_y = depth of the material build-up along the kerb (m)

n_{v} = traffic flow rate (v s^{-1})

t_y = time taken for the kerb-side build-up to take place (s)

Substituting measured values into the Equations 4.8 gives an estimate of $5.0 \times 10^{-5}\ \mathrm{v}^{-1}$ for f_y. Using the time-series of traffic flow along the Gloucester Place, f_y has a range of $3.3 \times 10^{-5}\ \mathrm{v}^{-1}$ to $9.1 \times 10^{-5}\ \mathrm{v}^{-1}$. Larger cross-road fluxes may arise where there is a stronger cross-road component of the street-level wind.

Speed and direction of material movement across the road

Assuming speed of material movement along and across the road are proportional to the amount of material removed in those directions, the speed and direction of material movement across the road can be derived as follows:

$$u_{y\mathrm{max}} = \frac{f_y\, u_{x\mathrm{max}}}{f_x} \tag{4.9}$$

$$u_{y\mathrm{avg}} = \frac{f_y\, u_{x\mathrm{avg}}}{f_x} \tag{4.10}$$

$$u'_y = \frac{u_{y\mathrm{avg}}}{n_{\mathrm{v}}} \tag{4.11}$$

$$\theta = \tan^{-1}\frac{f_y}{f_x} \tag{4.12}$$

Where,

$u_{y\mathrm{max}}$ = maximum speed of material movement across the road by the whole fleet of vehicles (m s^{-1})

$u_{y\mathrm{avg}}$ = average speed of material movement across the road by the whole fleet of vehicles (m s^{-1})

u'_y = average distance of material movement across the road by one vehicle (m v^{-1})

θ = direction of material movement relative to the direction of vehicle movement i.e., angle of spread (degree) (Fig. 4.15)

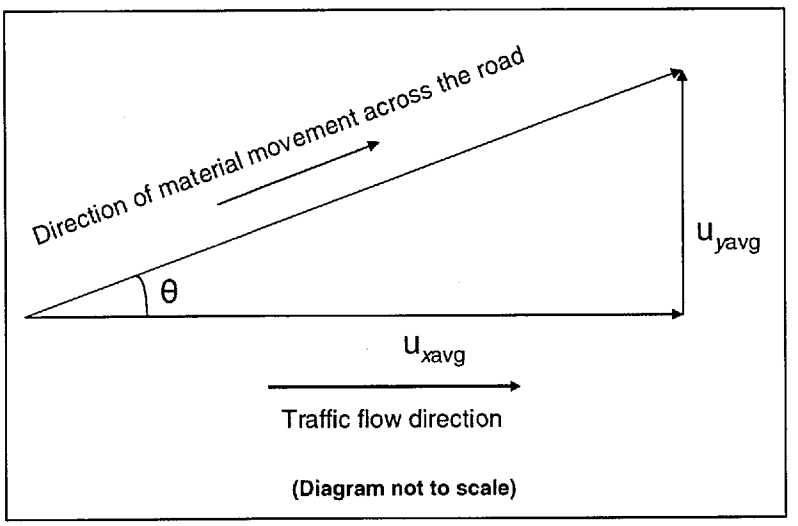


Fig. 4.15: Direction of the across-street material movement

Equations 4.9, 4.10 & 4.11 give an estimate of 0.014 m s^{-1}, 0.007 m s^{-1} and 0.006 m v^{-1} for u_{ymax}, u_{yavg} and u'_y respectively. Using the time-series of traffic flow along the Gloucester Place, u'_y has a range of 0.004 m v^{-1} to 0.011 m v^{-1}. From Equation 4.12, direction of the across-street component is estimated to be $\approx 8°$.

4.3.3 Flux to air

A fraction of the material is lost from the road surface due to emission to air. The varying levels of air concentration with time was recorded by the on-site Grimms (Section 4.1.5.1) and the dispersion coefficient is obtained from the tracer study (Section 4.1.3). Therefore, fraction of material emitted to air is estimated as follows:

$$\begin{aligned} f_z &= \frac{f'_z}{S\,l\,w} \\ &= \frac{F_z}{n_v S\,l\,w} \\ &= \frac{R}{n_v\,S} \\ &= \frac{C - C_b}{\chi\,n_v\,S} \end{aligned} \tag{4.13}$$

Where,

f_z = fraction of material removed from a road segment to air by one

vehicle (v^{-1})

f'_z = amount of material removed from a road segment to air by one vehicle (g v^{-1})

F_z = amount of material removed from a road segment to air in unit time by the whole fleet of vehicles (g s^{-1})

R = resuspension flux (i.e., mass emitted from unit area of road in unit time) (g m^{-2} s^{-1})

S = surface material loading on the road segment (g m^{-2})

n_v = traffic flow rate (v s^{-1})

C = concentration of particulate matter at the receptor (g m^{-3})

C_b = background concentration of particulate matter at the Gloucester Place (g m^{-3})

χ = dispersion coefficient (m^{-1} s)

l = length of the road segment (m)

w = width of the road segment (m)

Substituting measured values into the Equation 4.13 gives an estimate of 3.1×10^{-4} v^{-1} for f_z. Using the time-series of traffic flow along the Gloucester Place, f_z has a range of 2.1×10^{-4} v^{-1} to 5.4×10^{-4} v^{-1}.

4.3.4 Summary of flux estimates

Table 4.3.4 presents a summary of the flux estimates obtained from the experiment.

Table 4.3: Summary of flux estimates

Parameters	Average value	Range
f_x	$3.9 \times 10^{-4}\,\mathrm{v}^{-1}$	$2.6 \times 10^{-4}\,\mathrm{v}^{-1}$ to $7.1 \times 10^{-4}\,\mathrm{v}^{-1}$
f_y	$5.0 \times 10^{-5}\,\mathrm{v}^{-1}$	$3.3 \times 10^{-5}\,\mathrm{v}^{-1}$ to $9.1 \times 10^{-5}\,\mathrm{v}^{-1}$
f_z	$3.1 \times 10^{-4}\,\mathrm{v}^{-1}$	$2.1 \times 10^{-4}\,\mathrm{v}^{-1}$ to $5.4 \times 10^{-4}\,\mathrm{v}^{-1}$
f'_x	$1.2 \times 10^{1}\,\mathrm{g\,v}^{-1}$	$8.1\,\mathrm{g\,v}^{-1}$ to $2.2 \times 10^{1}\,\mathrm{g\,v}^{-1}$
f'_y	$1.6\,\mathrm{g\,v}^{-1}$	$1.0\,\mathrm{g\,v}^{-1}$ to $2.8\,\mathrm{g\,v}^{-1}$
f'_z	$9.6\,\mathrm{g\,v}^{-1}$	$6.5\,\mathrm{g\,v}^{-1}$ to $1.7 \times 10^{1}\,\mathrm{g\,v}^{-1}$
F_x	$1.6 \times 10^{1}\,\mathrm{g\,s}^{-1}$	-
F_y	$2.0\,\mathrm{g\,s}^{-1}$	-
F_z	$1.3 \times 10^{1}\,\mathrm{g\,s}^{-1}$	-

Estimates of fluxes show that with a single vehicle pass the average amount of material lost from a road segment in different directions are as follows: 0.04 % along the road, 0.005 % across the road and 0.03% into the air. Therefore, the total amount lost from a road segment after a single vehicle pass is 0.075 % of the material available on the road surface at that instant.

4.3.5 Relative contribution of different material removal processes

Equations 4.14, 4.15, 4.16, 4.17, 4.18 and 4.19 summarise the relationship between material emission to air and removal along and across the road.

$$\begin{aligned} F_z &= k_1 (F_x + F_y + F_z) \\ &= k_1 (F_x + k_2 F_x + k_3 F_x) \\ &= k_1 F_x (1 + k_2 + k_3) \end{aligned} \tag{4.14}$$

Where,

$$k_1 = \frac{f_z}{f_x + f_y + f_z} \tag{4.15}$$

$$k_2 = \frac{f_y}{f_x} \tag{4.16}$$

$$k_3 = \frac{f_z}{f_x} \tag{4.17}$$

Substituting values for f_x, f_y and f_z from Table 4.3.4 in Equations 4.15, 4.16 and 4.16, the average values of k_1, k_2 and k_3 are estimated as 0.4, 0.13 and 0.8. The estimate indicates 40 % of the material removed from the road surface is due to resuspension. Material removed across the road is about 13 % of that removed along the road. Amount of material removed through resuspension is 20 % less than the amount removed along the road.

Further, the amount of material removed through resuspension as a proportion of total material removed from the road surface is estimated by Equation 4.18:

$$\begin{aligned} F_z &= k_1 (F_x + F_y + F_z) \\ \Rightarrow F_z - k_1 F_z &= k_1 (F_x + F_y) \\ \Rightarrow F_z &= \frac{k_1}{1 - k_1} (F_x + F_y) \\ \Rightarrow F_z &= k_4 (F_x + F_y) \end{aligned} \tag{4.18}$$

Where,

$$k_4 = \frac{f_z}{f_x + f_y} \tag{4.19}$$

The average value of k_4 is 0.7, indicating the amount of material removed through resuspension is about 70 % that removed together through along-street and across-street movement.

4.3.5.1 Road surface dust dispersion formula

For a mass of material m on road, the following formula describes the road surface material dispersion at point (x, y) and at time t:

$$S(x, y, t) = \frac{m}{\sqrt{2\pi}\,\sigma_x\,\sigma_y} \; e^{-\left(\frac{x - t\,u_{\text{xavg}}}{\sigma_x}\right)^2} \; e^{-\left(\frac{y}{\sigma_y}\right)^2} \; e^{-\left(\frac{t}{T_0}\right)} \tag{4.20}$$

The tentative parameterisation is as follows:

$$T_0 = \frac{1}{f_z n_v} \tag{4.21}$$

$$\frac{\sigma_y}{\sigma_x} = \frac{f_y}{f_x} \tag{4.22}$$

$$\sigma_x = t(u_{\text{xmax}} - u_{\text{xavg}}) \tag{4.23}$$

Where,

$S(x, y, t)$ = road surface loading at point (x, y) and at time t (g m^{-2})

σ_x = along-street spread of a patch of material on road surface (m)

σ_y = across-street spread of a patch of material on road surface (m)

$u_{x\text{max}}$ = maximum speed of material movement along the road by the whole fleet of vehicles (m s^{-1})

$u_{x\text{avg}}$ = average speed of material movement along the road by the whole fleet of vehicles (m s^{-1})

T_0 = time taken for the total amount of material on the road to decay by a factor of e^{-1} (s)

4.4 Summary

This chapter presented results of the field measurements and associated quantitative analyses for its application to model parameterisation. Sections 4.1.4 to 4.1.6 provided the on-field evidence of the traffic-induced particle movement. For the model described in Chapter 5, meteorological data, traffic flow rate, dispersion relation and background PM_{10} serve as inputs to the model. Roadside PM_{10} measurement (Sections 4.1.5.1 and 4.2.3) as well as surface loading data (Section 4.2.5) are used to evaluate the model performance. Quantitative analysis of dust movement (Section 4.3) provided the parameter values for traffic-induced depleting fluxes.

Chapter 5

Model development

5.1 Introduction

A model is an abstraction of a real system, a simplification in which only those components which are seen to be significant to the problem at hand are represented. Thus, a model takes influence from aspects of the real system, aspects of the modeller's perception of the system and its importance to the problem at hand. Modelling supports in the conceptualisation and exploration of the behaviour of objects or processes and their interaction as a means of better understanding these and generating hypotheses concerning them. Modelling also supports the development of experiments in which hypotheses can be tested and outcomes predicted. A model provides a means of improving decision making capabilities as it helps in understanding the effect of changes on dynamic systems (most of the systems and problems are dynamic i.e., they vary with time) and verification of a model can identify deficiencies in both model structure and the information (i.e., data base or resources) (Baker, 1998).

Modelling is not an alternative to observation but, under certain circumstances, can be a powerful tool in understanding observations and in developing and testing theory. Observations will always be closer to truth and must remain the most important component of scientific investigation (Wainwright and Mulligan, 2004). The more complex a model, the more uncertainty is associated with the identification of model structure and parameters. The goal is a parsimonious model which nevertheless captures the

essential features of the underlying processes (Thiessen et al., 1999).

5.2 Objectives of the present modelling exercise

Decisions regarding the remedial measures for particulate pollution requires a two-prong knowledge; i.e., how much of it should be reduced and how should it be reduced? While the measurement of pollution level and air quality standards provide the answer to the former, the latter can only be addressed if knowledge on generation/propagation of particulate pollutants is available. The aim of this modelling exercise is to formulate a model that provides answers to the latter and therefore helps in identifying the appropriate control efforts. The goal of this model is to use an appropriate combination of measurements and modelling to identify the processes and estimate the process parameters that provide a sufficient level of accuracy and resolution in understanding the influence of traffic on roadside particulate pollution. Reasonably good information is available regarding the exhaust emission processes. Therefore the aim is to obtain information on processes behind resuspension. Estimation of resuspension alone without understanding of underlying processes has several limitations, discussed in detail, in Chapter 2. Therefore the objectives of this modelling exercise are to identify processes that cause resuspension as well as other modes of material movement in the road/vehicle environment and estimate parameter values for those processes so that a better understanding of the processes, the interplay of which determines the road dust resuspension, is obtained.

5.3 Modelling approach

The model includes description, parameterisation and estimation of parameter values for the process involved in road dust resuspension/movement. This modelling approach bears some similarity to studies in other branches of environmental science such as radioecological modelling (Whicker et al., 1999), where the need is to have more process-level studies which provide better insights into the fundamental mechanism which control the transport of particulate matter through various compartments of the

environment. Such studies can lead to better estimates of the pathways and parameters of individual processes and their variation with other factors (e.g., meteorology, traffic characteristics, sweeping, etc.). The better the key processes are understood and quantified, the more robust the model structure becomes. If such knowledge is adequate over the broad range of conditions and scenarios that occur across time and space, more generally applicable models can be constructed. These models require more input data, but they could provide much more informative predictions.

5.3.1 Model structure

Three dynamic processes, i.e., source, transport and sink, must be balanced in a model. In the present model the source terms are the processes that are believed to contribute to the traffic-induced particulate matter emissions. Besides exhaust emission, which as a source of traffic emission is well investigated (Abu-Allaban et al., 2003b; Goodwin et al., 1999; Harrison et al., 2005; Le Bihan et al., 2004; Kathuria, 2005; Kittleson et al., 2004), other sources and associated processes included in this model are either those indicated as a possible one in earlier studies, such as road abrasion (Gámez et al., 2001b) or site-specific sources which are perceived to be important from visual observation, such as carry-over from surrounding unpaved areas (Bexley, 2001, 2003). The estimates for these sources are obtained from available literature and field measurements. Sink in this case is roadside air, kerb/pavement and a section of road along the direction of traffic movement down the original road segment for which modelling is carried out. Roadside measurements provide estimates for these. The major gap is the knowledge of transport of particulate pollutants in road/vehicle environment and the main objective of this model is to narrow this knowledge-gap by quantifying these processes and understanding their interaction. Fig. 5.1 summarises these processes.

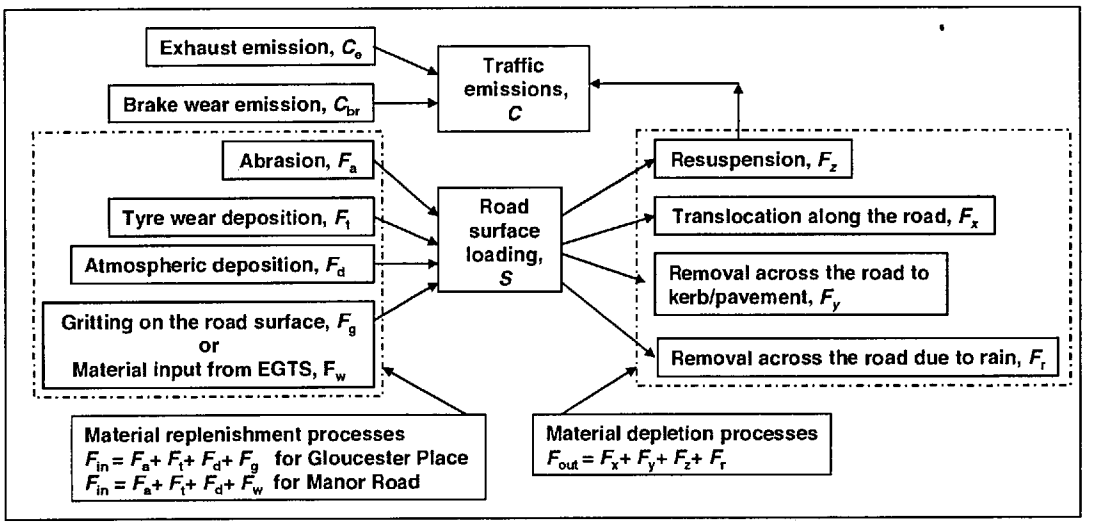


Fig. 5.1: Processes involved in traffic-induced particulate matter emissions.

The source terms of the model are represented by two sets of fluxes depending on their mode of contribution; the one that is emitted directly to the atmosphere and the other that contributes, to begin with, to road surface loading and subsequently to traffic emissions when it becomes resuspended. The processes which result in direct emission of particulate matter are exhaust emission and brake wear emission. Contributions of these two processes are represented by C_e and C_{br} respectively. Several sources contribute to the surface material loading such as abrasion of road surface, wear and tear of brake, tyres and other parts of vehicle, deposition from other nearby sources, carry over from surrounding unpaved areas, litter and spillage from vehicles, mud/dirt carry out from construction activities in the area and wind and/or water erosion of surrounding destabilised areas (USEPA, 1993c). However, modelling of the local sources such as deposition from other nearby sources and carry over from surrounding unpaved areas requires the site-specific information. Therefore the model structure is based on the premise that particulate matter enters the road surface as a result of the three permanent (i.e., process is occurring all the time) and universal (i.e., process is valid for all paved roads) processes. These are road abrasion, tyre wear deposition and dry deposition from atmosphere, represented by accumulating fluxes F_a, F_t and F_d respectively. Although these processes were recognised by earlier studies (Gámez et al., 2001a,b; Muschack, 1990), their likely contribution to resuspension was not thoroughly

estimated. Other site-specific sources, unique to a particular road/vehicle environment such as the ones mentioned before, can be included in the model depending on their nature and transport processes. For Gloucester Place, gritting is an additional local source and is represented by F_{g}. As discussed in Section 3.3.1, Erith Group Transfer Station (EGTS) is an additional local source for the road surface loading at Manor Road. Accumulating flux from EGTS is represented by F_{w}. Parameterisation of the source terms are described in Sections 5.6.1 & 5.6.2.

With vehicle movement, material on the road surface is moved and lost from the road surface through a number of transport processes. The quantity of material removed through any process depends on a number of factors such as moisture content of the material (e.g., dry material will be easily resuspended, material with a comparatively less moisture content will stick to the wheel and therefore will be more easily carried along the direction of movement, etc.), vehicle speed (e.g., higher vehicle speed will cause more turbulence and therefore more resuspension (Sehmel, 1973, 1976b)) and nature of the road surface (e.g., on a wet road removal of material to kerb and pavement through tyre-spray can be an important process (Smith, 1970)). The material removal processes, included in this model, are movement along the road, movement across the road, emission to air and removal by rain; represented by draining fluxes F_x[1], F_y, F_z and F_{r} respectively. These processes are conceptualised on the basis of on-road observation of material movement. F_x, F_y, F_z are parameterised on the basis of rate constants defined as the fraction of the material removed from road surface per vehicle pass and F_{r} parameterisation is linked to intensity of rain, defined as fraction of material removed in unit time. The relationship between removal fraction and intensity of rain, however, is non-linear in nature. While earlier studies are confined only in estimating the resuspended fraction of the road dust, the present model, for the first time, besides resuspension, attempts to identify other processes of material removal described above. Identification and estimation of these additional processes are required for the mass-balance of the particulate matter on the road and better understanding of the processes behind these. Section 5.6.3 illustrates parameterisation

[1] F_x therefore acts as a source flux for the downstream road segment

of these draining fluxes.

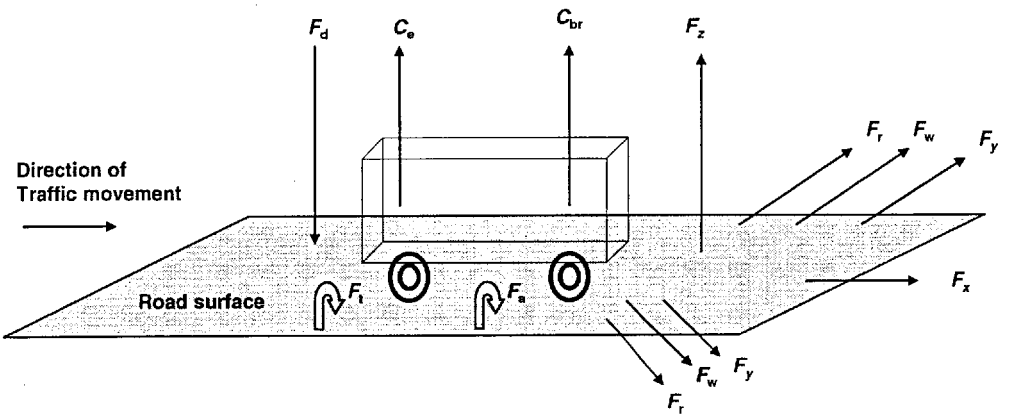


Fig. 5.2: Direction of accumulating and draining fluxes.

Fig. 5.2 shows the direction of accumulating and draining fluxes except those which are specific to a road surface as these can vary from one site to the other. C_e and C_{br}, the PM_{10} contributions due to emissions directly from the vehicle, are expressed in g m^{-3}, the units of airborne particulate matter concentration. Fluxes F_a, F_t, F_d, F_g, F_w, F_x, F_y, F_z and F_r are expressed in terms of their input and removal rate to and from the road surface respectively, with units of g s^{-1}.

A road segment is described, at any time, by the total mass of particulate matter on it. The dimension of a road segment is described by its surface area. Each road segment gains and loses particulate matter through a series of transport and emission processes that can be represented mathematically as first order losses i.e., the rate of loss is linearly proportional to the mass on road. Changes in quantity to the mass on road segment is therefore described by a mass-balance equation given below:

$$\frac{\mathrm{d}}{\mathrm{dt}} m_t = F_{\mathrm{in}} - F_{\mathrm{out}} \tag{5.1}$$

$$= (F_a + F_t + F_d + F_g) - (F_x + F_y + F_z + F_r) \tag{5.2}$$

for Gloucester Place

$$= (F_a + F_t + F_d + F_w) - (F_x + F_y + F_z + F_r) \tag{5.3}$$

for Manor Road

Where,

m_t = amount of material on road segment at any time t (g)

F_{in} = sum of accumulating fluxes (g s^{-1})

F_{out} = sum of draining fluxes (g s^{-1})

F_a = rate of road abrasion (g s^{-1})

F_t = rate of tyre wear deposition (g s^{-1})

F_d = rate of dry deposition (g s^{-1})

F_g = rate of gritting at Gloucester Place (g s^{-1})

F_w = rate of material input from EGTS onto Manor Road (g s^{-1})

F_x = rate of material removal along the road (g s^{-1})

F_y = rate of material removal across the road (g s^{-1})

F_z = rate of material emission to air (g s^{-1})

F_r = rate of material removal by rain (g s^{-1})

The types of data needed to construct this model include meteorological data such as wind speed, wind direction, deposition velocity and dispersion condition; hydrological data such as rainfall; traffic data such as traffic composition and traffic flow rate and parameter values for source terms and material transport and emission processes. Collection of these data and its quality depend on number of factors such as precision of field and laboratory instruments, accuracy of sampling technique and the computing techniques available. Data generation methodology, which includes QA/QC of data, is described in Chapter 3 and its quantitative estimation procedures, including the range of likely variability of the estimates for parameter values, are presented in Chapter 4.

5.3.2 Assumptions

Modelling requires some simplification of the complex real system through some assumptions. In the present modelling exercise following assumptions are made.

1. Except for the processes described, there is no other mode of material input and output from road surface.

2. Particles from various sources and of various sizes are thoroughly mixed.

3. There is no chemical reaction between various types of particles.

4. Due to the lack of any earlier study that provides an estimate of change of particle size on road surface due to traffic movement, it is assumed that there is no change in particle size between its arrival/generation on the road surface and its removal from the road surface.

5. All the material lying on road is available for resuspension. This may lead to a overprediction. However this assumption is reasonable due to the following two reasons:

   (a) Visual observations indicate that a paved road virtually gets cleaned after a certain time which besides meteorological and traffic condition depends on particle size, if there is no appreciable addition of material onto it. It is plausible that a certain proportion of material may remain locked within interstices of road surfaces. But it is certainly a very tiny fraction and over a period of time it ultimately gets resuspended due to heavy gusts of wind or high turbulence from heavy vehicles.

   (b) An overestimation is acceptable from a regulatory viewpoint, because it will necessitate better control regime.

6. Except for removal by rain and wind, the amount of material removed is directly proportional to the traffic volume.

7. Material from EGTS enters to Manor Road only by sticking to the outer surface of the wheels.

5.4 Justification of choice of modelling approach

The review in Chapter 2 highlights the lack of realism and accuracy associated with presently used empirical models for estimations of traffic emissions and suggests the

development of a model that includes the sources and the underlying processes involved in traffic emissions as an important option. Modelling the resuspension process separately from the exhaust emission process is identified to be the main necessity because in spite of its significant contribution to urban particulate matter level, processes behind resuspension are poorly understood. Gámez et al. (2001a) proposed to calculate PM_{10} emission as sum of emissions from exhaust pipe, vehicle component (e.g., tyre, brake and clutch wear), road abrasion and emissions resulting from material entering the road via external sources (e.g., dirt inputs, sanding process, etc.). Subsequent German studies by Düring et al. (2002a,b) tried to proceed in this direction although these studies also included some of the EPA assumptions and expressed lack of sufficient data as the constraint in understanding the resuspension process. Therefore, through this research, an attempt has been made to develop a mass-balance model that captures all the essential processes to explain the emission and translocation of particulate matter from road due to traffic and therefore to estimate importance of resuspension in urban air pollution.

The silt loading in this model is instantaneous and is always variable depending on inflow-outflow quantity and it contributes only to resuspension. When silt loading reaches equilibrium, i.e., material inflow on to the road is same as material outflow from the road, emission will still take place from the road which will consist of exhaust emission with resuspension component zero. Rogge et al. (1993) have expressed that resuspension, fallout, street sweeping, rain, and generation of new particles through vehicle exhaust drive a dynamic source and sink relationship contributing appreciable amounts of particulate matter to the atmosphere and hydrosphere. With both depletion and replenishment processes included and estimated in this model, such a dynamic source and sink relationship is therefore established. Therefore resuspension is a dynamic process, as indicated by other studies (Ruellan and Cachier, 2001). This eliminates the dilemma, discussed by earlier researchers (Fitz and Bufalino, 2002; Venkatram, 2000), over use of a single silt loading value (with no description of processes replenishing and depleting it) to deduce emission factor. Besides giving a better understanding of particle movement on road/vehicle environment, the model has the

potential to predict emission estimates that are based on realism that the earlier empirical approaches, including AP-42, were lacking. Necessity of such an approach has been advocated by many researchers (Countess et al., 2001; Karppinen et al., 2004; Kuhns et al., 2001b; Nicholson and Branson, 1990; Venkatram, 2000).

5.5 Modelling software

The model is based on a system dynamics (SD) approach where the the physical or abstract processes that define a system are first conceived and then these processes are transferred, through various building blocks in the model construction layer of the software, as components of the model. The basic building block of SD model is the "stock" that represents anything that accumulates. The second building block is the "flow" that represents the rate of accumulation or loss that causes change in magnitude of the stock. The third one, called "connector", transmits information that is used to regulate flows. The last building block is the "converter" that stores constant values or equations to generate output values for each time period. It often processes the initial input information to a form that can be used as a variable in the model. Converters, used in details, simplify the understanding of various processes of the model, although their presence itself increases the visual complexity of the model.

The systems thinking tool, Stella Research Version 8.0, developed by High Performance Systems, Inc., USA (HPS, 2003), is used to develop and simulate this SD model. Stella software executes the simulation model coded by visual programming based on SD for personal computer (Matsumoto, 1999). Basic tools for model building and model simulation are similar to several other softwares used to build SD model such as Simulink, the graphical modelling environment from the MathWorks, Inc., USA (MathWorks, 2004); Powersim from the Powersim Ltd., Norway (Powersim, 2003); Simile from the Simulistics Ltd., UK (Simulistics, 2004); and Vensim from the Ventana Systems, Inc., USA (Vensim, 2003). Stella software, which can be used by people of any discipline, e.g., humanities, engineering, life sciences, etc., has been used in population dynamics, financial planning, life cycle assessment of residential buildings, etc.

(Matsumoto, 1999). However no earlier studies could be found which has used this software for environmental problems.

In this model stock represents the particulate matter mass on a road segment. Inflows and outflows to the stock represent accumulating and draining fluxes that cause mass of particulate matter on road surface to vary with time. Connectors connect several converters, in a way defined by their connection pattern, to have the right calculation in steps until the final converter which connects it to a flow, the building block ultimately responsible for conveying the input to stock. On the model output side, connectors connect the outflows with a number of converters to present the output as per requirement. Converters have been used extensively for variety of purposes such as holding equations for calculations; time-series data, in graphical form, for traffic flow rate, PM_{10} concentration, dispersion function, etc and numerical values for road dimensions. So, in calculus parlance, flows represent time derivatives, stocks are the integrals of flows over time and converters contain the micro-logic of flows.

The software provides three algorithms for numerical integration of mathematical equations. They are Euler's method, 2^{nd} order Runge-Kutta method and 4^{th} order Runge-Kutta method.

5.6 Model formulation and estimation of parameter values

A model can be designed either around available measurements or can be fully developed first and then data collection follows as per the requirement of the developed model structure. For systems for which underlying processes are relatively unknown, development of a model on the basis of available measurement may be the only option to start with. However, many modellers would favour the latter approach, because it leads to the development of model structure (and thus the process underlying it) first and indicates which parameters need to be measured to validate the process. In such cases we may have to make reasonable guesses based on indirect evidence. The

modelling procedure then may be carried out iteratively to investigate which of a number of reconstructions may be the most feasible. This iterative approach to evaluate a correct model parameter is generally known as model calibration. Optimal model structure may also produce parameters which cannot be measured in the field setting, especially at the scales in which they are represented in the model. The limitations may be due to cost, or the lack of appropriate techniques. In such cases, it may be necessary to derive parameter values from (surrogate) parameters that are simpler to measure (Wainwright and Mulligan, 2004). Further, parameterisation is also costly. Work in the field requires considerable investment of time, and generally, also money. Therefore, investment in parameter measurement should be according to how great an effect the parameter has on the model output. Sensitivity analyses indicate the magnitude of the effect of parameters on model output (Wainwright and Mulligan, 2004).

The main approach followed here is to construct the basic structure of the model without considering any measurement. Therefore, the model formulation, described in this section, is based purely on the processes observed in the field and processes identified in earlier studies. Some of the parameter values for the processes are obtained from available literature and the rest are estimated from the data obtained from field measurements. However, no measurement is available for the process represented by flux F_w and therefore, this parameter is calibrated to fit with other available measurements at Manor Road.

5.6.1 Fluxes emitted directly to air

5.6.1.1 Exhaust emission, C_e

The assumption is that exhaust emissions are emitted to air at the point of release from vehicle and are therefore contribute directly to airborne particulate matter concentration.

In the model, exhaust emission is parameterised as follows:

$$\begin{aligned} C_e &= e'_e \chi \\ &= e_e n_v \chi \end{aligned} \tag{5.4}$$

Where,

C_e = contribution of exhaust emission to roadside PM_{10} concentration (g m^{-3})

e'_e = exhaust emission strength (g m^{-1} s^{-1})

χ = dispersion coefficient (m^{-2} s)

e_e = average exhaust emission factor for a fleet of vehicle (g m^{-1} v^{-1})

n_v = traffic flow rate (v s^{-1})

n_v and χ at Gloucester Place are estimated in Sections 4.1.2 & 4.1.3 (χ, obtained in units of m^{-1} s at Gloucester Place, is changed to units of m^{-2} s by dividing the width of the road). Estimates of n_v and χ at Manor Road are obtained from Sections 4.2.4 & 4.2.2. e_e for each site is estimated as follows:

$$e_e = \sum e_{ei} z_i \tag{5.5}$$

Where,

e_{ei} is the exhaust emission factor of each type of vehicle (g km^{-1} v^{-1}) and z_i is the respective proportion of each type of vehicle (%).

The traffic on Gloucester Place and Manor Road has been classed into following six types: (a) solo motorcycles and scooters; (b) cars, taxis and motorcycle combinations; (c) LGVs; (d) buses and coaches (e) HGVs (rigid); and (g) HGVs (articulated or with trailer axles). Exhaust emission factor for each type of vehicle has been taken from the "UK Emission Factor Database" (NAEI, 2003). The database contains emission factors for different driving conditions and different years, i.e., rural driving, urban driving and motorway driving for the year 2000, 2005 and 2010. In this model emission factors for urban driving condition for the year 2005, presented in Table 5.1, are used. These emission factor values are comparable with $PM_{2.5}$ tailpipe emission factors of 0.63×10^{-2} g km^{-1} v^{-1} for light-duty spark ignition (LDSI) vehicles and 3.5×10^{-1} g

km^{-1} v^{-1} for heavy-duty diesel (HDD) vehicles reported in recent studies (Baldauf et al., 2002).

Table 5.1: Exhaust emission factors for different types of vehicles for the year 2005 (NAEI, 2003)

Vehicle type	Exhaust emission factor ($g\ km^{-1}\ v^{-1}$)	
Solo M/C, scooters & mopeds	9.0×10^{-2}	
Cars, taxis & M/C comb	7.0×10^{-2}[a]	3.2×10^{-3}[b]
Light Goods Vehicle	1.3×10^{-1}[c]	4.8×10^{-3}[d]
Buses and Coaches	2.3×10^{-1}	
Heavy Goods Vehicle (rigid axles)	1.4×10^{-1}	
Heavy Goods Vehicle (Articulated or with Trailer axles)	3.3×10^{-1}	

[a]Diesel cars and taxis.
[b]Petrol cars and taxis.
[c]Diesel LGVs.
[d]Petrol LGVs.

Substituting emission factor values from Table 5.1 and traffic composition of Gloucester Place from Section 4.1.2 and that of Manor Road from Section 4.2.4 in Equation 5.5, the estimates for e_e are obtained as follows: 3.4×10^{-2} g km^{-1} v^{-1} to 8.9×10^{-2} g km^{-1} v^{-1} for Gloucester Place and 5.8×10^{-2} g km^{-2} v^{-1} to 1.0×10^{-1} g km^{-1} v^{-1} for Manor Road. The lower values correspond to petrol cars, taxis and LGVS and the higher values correspond to diesel cars, taxis and LGVs. These estimates are higher than the PM_{10} exhaust emission of 2.3×10^{-2} g km^{-1} v^{-1}, estimated for an urban kerbside location in Sweden (Johansson et al., 2004), mainly because of the high proportion of heavy duty vehicles (Sections 4.1.2 & 4.2.4) on Gloucester Place and Manor Road.

5.6.1.2 Brake wear emission, C_{br}

Debris from wear of brake linings have been identified as contributors to airborne particulate matter concentration in the vicinity of the busy roads and in ambient urban air (Abu-Allaban et al., 2003b; Alastuey et al., 2004; Kim et al., 1990; Pierson and Brachaczek, 1974; Rogge et al., 1993; Thompson et al., 1966). During application of the brake, motor vehicle brake linings and brake surfaces are subjected to frictional

heat generation and associated wear. This mechanically induced wear generates brake lining particles which under normal conditions are introduced into the environment (Rogge et al., 1993). Usually, passenger cars are equipped with front disc brakes and rear drum brakes. During the deceleration process, front brakes have to provide most of the braking power. In addition, front disc brakes have a smaller friction lining area when compared to rear drum brakes. Consequently, the friction force per unit area of brake lining material is 5 times to 6 times larger for front disc brakes. Because of the different physical demands (e.g., higher friction-induced temperature), brake lining materials for disc and drum brakes differ somewhat in composition. While basically all of the fine particulate brake dust from disc brakes is released to the environment, small amounts of brake dust (11 % on average) are retained in the drum brakes (Rogge et al., 1993).

Brake linings consist mainly of inorganic materials with appreciable amounts of copper, nickel, chromium and lead along with iron (Hewitt, 1990; Muschack, 1990). Strong correlation between particulate barium, a common element in brake linings, and heavy-duty traffic suggests that HDVs are responsible for larger brake wear particulate emissions than LDVs (Sternbeck et al., 2002). Garg et al. (2000) have suggested that higher brake wear emission rates by HDVs would include both mechanically generated coarse particles and the formation of finer particles due to high temperature at the brake/rotor interface. Metals like iron, manganese and copper show a distinct gradient in their concentration in airborne particles with highest concentration in roadside samples, moderate levels at background sites and lowest concentration at rural sites (AQEG, 2004), indicating traffic as its source.

Goodwin et al. (1999) have reported contribution of resuspension and emissions from brake and tyre wear to PM_{10} emissions in UK as 18.9 kt and 5.0 kt respectively. Therefore, brake and tyre wear contributes about 25 % to the resuspended material. Further, its increase by 25 % between 1990 and 1995 indicates the importance of brake and tyre wear as a source of resuspension. In 2001, brake and tyre wear emissions accounted for approximately 23 % of the total road transport emissions (AQEG, 2004). Typical PM_{10} emission factors from brake wear lie between 1×10^{-3} g km^{-1} v^{-1} and

9×10^{-3} g km^{-1} v^{-1} for LDV and between 2×10^{-2} g km^{-1} v^{-1} to 4×10^{-2} g km^{-1} v^{-1} for HDV (USEPA, 1995; TNO, 1997). Limited studies in this area also indicate emissions as PM_{10} are higher from brake wear than from tyre wear. The estimate for brake wear is around 1×10^{-2} g km^{-1} v^{-1} to 2×10^{-2} g km^{-1} v^{-1} and this figure is 3 or 4 times higher for HDVs. However, these studies conclude that a significant proportion, as high as 33 %, of brake wear emissions are less than 0.1 μm in diameter.

Based on average speeds on the UK roads, Table 5.2 presents tyre and brake wear emission factors for different vehicle types on different road types.

Table 5.2: Average PM_{10} emission factors for tyre and brake wear in the UK (AQEG, 2004)

Vehicle category	Road type	Tyre wear (g km^{-1} v^{-1})	Brake wear (g km^{-1} v^{-1})
Motorcycles	Urban	3.76×10^{-3}	5.8×10^{-3}
	Rural	2.92×10^{-3}	2.8×10^{-3}
	Motorway	2.49×10^{-3}	7.0×10^{-4}
Cars	Urban	8.74×10^{-3}	1.2×10^{-2}
	Rural	6.80×10^{-3}	5.5×10^{-3}
	Motorway	5.79×10^{-3}	1.4×10^{-3}
LGVs	Urban	1.38×10^{-2}	1.8×10^{-2}
	Rural	1.07×10^{-2}	8.6×10^{-3}
	Motorway	9.15×10^{-3}	2.1×10^{-3}
HGVs[a]	Urban	9.18×10^{-3}	5.1×10^{-2}
	Rural	7.37×10^{-3}	2.7×10^{-2}
	Motorway	6.08×10^{-3}	8.4×10^{-3}
Buses[a]	Urban	9.37×10^{-3}	5.4×10^{-2}
	Rural	7.37×10^{-3}	2.7×10^{-2}
	Motorway	6.08×10^{-3}	8.4×10^{-3}

[a] For HGVs and buses the unit of tyre wear is in gram per axle-km.

Based on data collected under Co-ordinated European Programme on Particulate Matter Emission Inventories, Projections and Guidance (CEPMEIP) project, Table 5.3 presents brake and tyre wear emission factors for different vehicle types in the EU.

The emission factors for brake and tyre wear emission from the "UK Emission Factor Database" is presented in Table 5.4. The database provides a combined emission factor for brake and tyre wear emission. For pollutants like PM_{10}, NO_x, CO, etc., the database contains emission factors for different driving conditions e.g., cold start, rural driving,

Table 5.3: Average PM_{10} emission factors for tyre and brake in the EU (Berdowski et al., 2001)

Vehicle category	Tyre wear ($g\ km^{-1}\ v^{-1}$)	Brake wear ($g\ km^{-1}\ v^{-1}$)
Motorcycles	1.72×10^{-3}	3.0×10^{-3}
Cars	3.45×10^{-3}	6.0×10^{-3}
LDVs	4.50×10^{-3}	7.5×10^{-3}
HDVs	1.86×10^{-2}	3.2×10^{-2}

urban driving and motorway driving and for different years e.g., year 2000, 2005 and 2010 (NAEI, 2003). However, for brake and tyre wear emissions, such drive-specific and year-specific data are not included in the database, indicating the lack of detailed study in quantifying it.

Table 5.4: Average PM_{10} emission factors for brake and tyre wear in the UK (NAEI, 2003)

Vehicle category	Brake & tyre wear ($g\ km^{-1}\ v^{-1}$)
Solo motorcycles, scooters and mopeds	4.57×10^{-3}
Cars, taxis and motorcycle combinations	9.15×10^{-3}
Light goods vehicles	9.15×10^{-3}
Buses and coaches	1.00×10^{-2}
Heavy goods vehicles (rigid)	1.00×10^{-2}
Heavy goods vehicles (articulated)	3.00×10^{-2}

These estimates for the UK, presented in Table 5.4, are consistent with the USEPA emission factors (USEPA 1985). For example, the USEPA base emission factor for PM_{10} emissions from passenger cars is 1.2×10^{-3} g km^{-1} v^{-1} for tyre wear and 7.95×10^{-3} g km^{-1} v^{-1} for brake wear. Taken together these give an emission factor of 9.15×10^{-3} g km^{-1} v^{-1} for brake and tyre wear, the same as the emission factor for cars, taxis and motorcycle combinations under NAEI (Table 5.4).

However, brake wear is still a factor of uncertainty. From a study at 8 locations with varying traffic composition at Nevada and North Carolina, US, Abu-Allaban et al. (2003b) found evidence of brake wear mainly confined to PM_{coarse} and from

only one location where vehicles exit from freeway and hence need to apply the brakes. Whether the brake wear emissions are instantly resuspended after their formation or are at first deposited on to the road and later become resuspended is not clear (Harrison et al., 2003, 2004b). In AP-42, brake wear is included in direct emission (USEPA, 1993c). Several authors suggest brake wear emission, like exhaust emission, is a direct emission and therefore quantify it in terms of certain percentages of exhaust emission, although some coarser fractions of it might be contributing to road surface loading. The assumption made in this model formulation is that all brake wear emissions are emitted to air immediately after their generation and therefore contribute directly to airborne particulate matter concentration.

In this model, brake wear emission is parameterised as follows:

$$\begin{aligned} C_{\mathrm{br}} &= e'_{\mathrm{br}}\,\chi \\ &= e_{\mathrm{br}}\,n_{\mathrm{v}}\,\chi \end{aligned} \tag{5.6}$$

Where,

C_{br} = contribution of brake wear emission to roadside PM_{10} concentration (g m^{-3})

e'_{br} = brake wear emission strength (g m^{-1} s^{-1})

χ = dispersion coefficient (m^{-2} s)

e_{br} = average brake wear emission factor for a fleet of vehicle (g m^{-1} v^{-1})

n_{v} = traffic flow rate (v s^{-1})

n_{v} and χ at Gloucester Place are estimated in Sections 4.1.2 & 4.1.3. Estimates of n_{v} and χ at Manor Road are obtained from Sections 4.2.4 & 4.2.2. e_{br} for each site is estimated as follows:

$$e_{\mathrm{br}} = \sum e_{\mathrm{bri}}\,z_{\mathrm{i}} \tag{5.7}$$

Where,

e_{bri} is the brake wear emission factor of each type of vehicle (g km^{-1} v^{-1}) and z_{i} is the respective proportion of each type of vehicle (%).

Brake wear emission factors for different types of vehicles have been taken from Ta-

bles 5.2 and 5.3. Traffic composition at Gloucester Place is obtained from Section 4.1.2 and that at Manor Road is obtained from Section 4.2.4. Using Equation 5.7, average brake wear emission factors are estimated to be between 8.2×10^{-3} g km^{-1} v^{-1} and 1.5×10^{-2} g km^{-1} v^{-1} for Gloucester Place and 1.1×10^{-2} g km^{-1} v^{-1} and 1.9×10^{-2} g km^{-1} v^{-1} for Manor Road.

5.6.2 Fluxes that replenish road surface loading, F_{in}

5.6.2.1 Road abrasion, F_a

Frictional force and thermal energy that act at the wheel/road interface during the movement of vehicles cause abrasion of the road surface. The abrasion rate will vary depending on the materials used for road construction and their relative proportion (e.g., asphalt has a larger abrasion rate than concrete). Use of lightweight studs and introduction of more wear resistant asphalt in Sweden resulted in reduction of average winter road wear from 24 g km^{-1} v^{-1} in 1996 to 0.2 g km^{-1} v^{-1} in 2000 (Lindgren, 1996). Rate of abrasion will also vary depending on the general road condition, especially the state of road surface (e.g., new, old, porous, smooth, rough, patched, cracked, etc.) (Gámez et al., 2001b) and weather. For the same traffic volume, a poorly maintained road having signs of significant weathering (e.g., Manor Road) will experience more abrasion than a well-maintained paved road (e.g., Gloucester Place). This is because eroded/damaged patches over a bad road allow more erosion to take place. In this model, the varying rate of road erosion has been accounted for through the parameter "road condition factor", ψ. Road condition is assessed visually and therefore, is subjective in nature.

Limited studies are available on quantification of abrasion from road surface. Investigations carried out in Northern Germany reported abrasion rates of 1.0×10^{-3} m per year for freeways and 8.0×10^{-4} m per year for highways (Muschack, 1990) to 10^{-3} m per 10^6 vehicles (Sieker and Grottker, 1988). Considering a typical traffic flow rate of 150,000 vehicles per day on a freeway and 75,000 vehicles per day on a highway, the abrasion rates reported by Muschack (1990) are equivalent to 1.8×10^{-11} m v^{-1} for free-

ways and 2.9×10^{-11} m v^{-1} for highways. Based on data collected under Co-ordinated European Programme on Particulate Matter Emission Inventories, Projections and Guidance (CEPMEIP) project, Table 5.5 presents emission factors for road abrasion for different vehicle types in the EU.

Table 5.5: Average PM_{10} emission factors for road abrasion in the EU (Berdowski et al., 2001)

Vehicle category	Road abrasion (g km^{-1} v^{-1})
Motorcycles	3.65×10^{-3}
Cars	7.25×10^{-3}
LDVs	9.50×10^{-3}
HDVs	2.69×10^{-2}

Road abrasion is parameterised as follows:

$$\begin{aligned} F_a &= V_a \rho \\ &= \tau_a'' l w \rho \\ &= \tau_a' n_v l w \rho \\ &= \tau_a \psi n_v l w \rho \end{aligned} \tag{5.8}$$

Where,

F_a = rate of road abrasion deposition (g s^{-1})

V_a = volume of road abrasion per unit time (m^3 s^{-1})

n_v = traffic flow rate (v s^{-1})

l = length of the road segment (m)

w = width of the road segment (m)

τ_a'' = thickness of road abrasion per unit time (m s^{-1})

τ_a' = thickness of road abrasion per vehicle with road condition considered (m v^{-1})

τ_a = thickness of road abrasion per vehicle on a good road (m v^{-1})

ψ = road condition factor

$$\psi = \begin{cases} 1, & \text{for good road (e.g., Gloucester Place).} \\ 2, & \text{for bad road (e.g., Manor Road).} \end{cases}$$

Düring et al. (2002a) have suggested that silt loading on a bad street surface is about twice that of a good surface. Therefore the above values for ψ are consistent with this suggestion.

ρ = average density of road material (g m^{-3})

In this model the range of τ_a values is taken from those reported by Muschack (1990) and Sieker and Grottker (1988). l and w, the dimensions of road segment, are estimated from the design of field experiment and application of model. For Gloucester Place, l and w are 120 m and 13 m respectively (Section 3.2.2) and for Manor Road the corresponding values are 180 m and 5 m respectively (Sections 3.3.4 & 5.6.2.5). n_v at Gloucester Place is estimated in Section 4.1.2. Estimate of n_v at Manor Road is obtained from Section 4.2.4. ρ is estimated as follow:

$$\rho = \sum \rho_j z_j \tag{5.9}$$

Where,

ρ_j is the density of each type of road material (g m^{-3}) and z_j is the respective proportion of each type of road material (%).

For an average UK road condition, which is made up of 60 % of aggregate, 20 % of sand and 20 % of asphalt with densities of 2.8×10^6 g m^{-3}, 2.8×10^6 g m^{-3} and 0.9×10^6 g m^{-3} respectively, the ρ estimate is 2.42×10^6 g m^{-3}. The above estimate of ρ broadly agrees with that considered by Gámez et al. (2001b) for a typical density of road material. In some cases the aggregate and sand accounts for 95 % of the total road material (Lindgren, 1996) and therefore, a road-specific ρ should be estimated.

The traffic flow at Gloucester Place is influenced by two traffic lights. Therefore, it is expected that braking and acceleration activities will be higher than normal. This will lead to increased erosion of tyre, increased wear of brake, increased abrasion of road surface and increased exhaust emission. While one explanation of higher resuspension by heavy-duty traffic is high turbulence generated by these vehicles, another explanation of higher dust emission may be due to higher tyre pressure of heavy-duty traffic that causes more abrasion of road. The abrasion can also lead to higher tyre

wear and brake linings (AQEG, 2004).

5.6.2.2 Tyre wear deposition, F_t

Several studies have identified particles generated from tyre wear as one of the sources of airborne particulate matter concentration in the vicinity of the busy roads and in urban areas (Abu-Allaban et al., 2003b; Alastuey et al., 2004; Kim et al., 1990; Pierson and Brachaczek, 1974; Rogge et al., 1993; Thompson et al., 1966). Tyre tread wear is a complex physicochemical process, driven by the frictional energy at the interface between the tread and the road surface (Veith, 1995). Tyre wear particles are generated during the rolling shear of the tyre tread against the road surface. Tyre wear and road wear are strongly related as the energy that wears the road is the energy that wears the tyre (Lupker et al., 2002). The rate of wear is dependent on a combination of factors including tyre characteristics (compound and design), vehicle characteristics (type and usage), road surface characteristics, environmental conditions (temperature and rainfall) and many others (AQEG, 2004; Lupker et al., 2002). Studies by Ahagon and Kaidou (1990) and Kaidou and Ahagon (1990) showed that tyre rubber is gradually altered during use due to changing operating temperatures, mechanical wear, oxidant and radical attack combined with the loss of protective rubber ingredients. Therefore, in addition to the material composition of tyre, age of tyre and the physical and chemical stress conditions (stress condition is a function of road conditions e.g., asphalt vs concrete; driving conditions e.g., smooth vs abrupt acceleration/deceleration, continuous running vs frequent braking, etc.; tyre conditions e.g., tyre type, tyre pressure, retreaded vs new tyres, etc.; and vehicle loading condition e.g., normal vehicle load vs vehicle overloading) to which the tyre is subjected determine the size and chemical composition of tyre wear particles. Veith (1992) has identified different kinds of tyre wear mechanism such as adhesive wear (caused by high transient adhesion), abrasive wear (cutting-rupture action of sharp angular asperities on the sliding counterface or as third bodies), erosive wear (cutting-rupture action of particles in a liquid/fluid stream), corrosive wear (action of chemical on surface) and fatigue wear (rapid or gradual changes of material property leading to cracks and material loss). Grosch (1992)

suggests that, out of the above mechanisms, fatigue wear dominates the tyre wear. Depending on the turbulence (traffic density and wind speed determine the strength of the turbulence) within the air column above the street, some of the tyre debris are entrained into the atmosphere where its removal rate depends among other factors on the particle size, which influences the particle settling velocity (Rogge et al., 1993). Typical figures for tyre wear emission as PM_{10} are between 1 % and 15 % by mass of passenger car tyre wear (AQEG, 2004). A detailed environmental study by Pierson and Brachaczek (1974), however, concluded that much of the tyre wear is in the form of non-suspendible particles deposited on or near the road and that, depending on driving conditions, the amount of airborne tyre-particulate averages just 20 % of the amount of the exhaust particulate matter from petrol engines burning leaded fuel. With unleaded petrol gradually replacing the leaded petrol worldwide, importance of tyre debris as a component of resuspension becomes more prominent.

Particles from tyre wear are mostly of organic origin with slow biological degradation rate. Inorganic materials that are associated with tyre wear particles are mainly the heavy metal oxides that are used as an additive in tyre manufacturing such as zinc, lead, chromium, copper and nickel (Hewitt, 1990; Muschack, 1990). Detection of minor to trace amounts of iron-rich (often contaminated with barium, sulphur and silicon) and zinc species from most of the samples from different road side locations indicated tyre wear, possibly with some brake wear, as the source (Abu-Allaban et al., 2003b). Kim et al. (1990) found that the concentrations and percentages of tyre tread in PM_{10} show seasonal variation. They reported weight percentage of tyre tread in PM_{10} varying between 1.3 % and 3 % in winter and between 0.5 % and 1.5 % in spring. The higher percentage in winter was primarily due to typical wind direction in winter that resulted in wind passing over the dense traffic area of Tokyo before reaching the receiver. Further, day and night profiles of tyre tread at ground level and at a height of 16 m followed the pattern of traffic and therefore, indicated traffic to be prime source of emission of tyre tread in the urban area. Close relationship of tyre tread particles with traffic, indicated by hydrocarbon concentration peaks following traffic flow, has also been reported by Siddiqi and Woeley (1977). Abu-Allaban et al. (2003b) esti-

mated, for 8 locations at North Carolina, US, brake and tyre wear emissions, by mass, between 0 % to 20 %. The majority of these were less than 5 % and were mainly contributing to PM_{10}, unlike tailpipe emissions and emissions of vegetative origin, which contribute primarily to $PM_{2.5}$. Alastuey et al. (2004) reported exhaust and tyre erosion contribution of 6 μg m^{-3} (40 %) to kerbside PM_{10} level in Spain. Kim et al. (1990) showed that 30 % of the mass concentration of tyre wear particles at the ground can still be found at a height of 86 m during the daytime. They concluded that absence of mixing of air due to higher wind speed and unobstructed flow of air at more height did not allow coarse particles of tyre emissions to diffuse to such a height. Pierson and Brachaczek (1974) have reported that only 10 % of the tyre tread particles (by mass) are smaller than 3 μm. This suggests that a large part of the tyre wear emissions is made up of coarse particles which become easily resuspended.

Subramani (1971) has reported an emission factor of 0.12 g km^{-1} v^{-1} for airborne particulate matter generated by wear of rubber tyres under U.S. driving conditions. Dannis (1974) reported tyre wear of 4.5 kg after 50,000 km driving and therefore a tyre wear rate of 0.09 g km^{-1}. A figure of as high as 120 g km^{-1} has been reported for the tyre abrasion particles for an average of 1000 vehicles per day on a German road (Göttle, 1979; Muschack, 1990). Average tread wear rate between 0.006 g km^{-1} and 0.09 g km^{-1} for a single passenger tyre under different driving condition, different road condition and different tyre condition has also been reported (Dannis, 1974; Fwa and Ang, 1991; Pierson and Brachaczek, 1974; Schuring and Clark, 1988). A recent study by Luhana et al. (2004) reports a typical figure of 0.074 g km^{-1} v^{-1} for tyre wear. From investigations carried out under the EC 5th framework project "Tyre and Road Wear and Skip Assessment" (TROWS), Lupker et al. (2002) reports tread wear of 1 mm per 10,000 km to 1 mm per 30,000 km for general truck tyres. Kristensson et al. (2004) reported emissions out of tyre and brake wear are about 100 times smaller compared to the resuspension of crustal materials.

However, tyre wear is still a factor of uncertainty. From a study at Nevada and North Carolina, US, Abu-Allaban et al. (2003b) found little evidence of tyre wear and attributed this to electrostatic charge carried by rubber particles that prevent it from

depositing and accumulating on the filter during the sampling process, as suggested by Hildemann et al. (1991a). Whether these materials are directly emitted after their formation or they first get deposited on to the road and then become resuspended is not clear (Harrison et al., 2003, 2004b). In AP-42, tyre wear emission is considered as a direct emission (USEPA, 1993c). Several authors suggest brake and tyre wear emission as a direct emission like exhaust emission and therefore quantify it in terms of certain percentages of exhaust emission. However, erosion of tyre takes places by the same process and at the same time as that of road abrasion. Therefore, in this model, like abrasion, the assumption is that all of the tyre wear materials are first deposited on the road surface and then become resuspended with movement of vehicle.

In the model, tyre wear deposition is parameterised as follows:

$$\begin{aligned} F_t &= \omega_t' \, l \\ &= \omega_t \, n_v \, l \end{aligned} \tag{5.10}$$

Where,

F_t = rate of tyre wear deposition on the road segment (g s^{-1})

ω_t' = tyre wear deposition rate per unit length of road surface (g m^{-1} s^{-1})

ω_t = average tyre wear rate for a fleet of vehicle (g m^{-1} v^{-1})

l = length of the road segment (m)

n_v = traffic flow rate (v s^{-1})

l for Gloucester Place is 120 m (Section 3.2.2) and for Manor Road is 180 m (Sections 3.3.4 & 5.6.2.5). n_v at Gloucester Place is estimated in Section 4.1.2. Estimate of n_v at Manor Road is obtained from Section 4.2.4. ω_t for each road is estimated as follows:

$$\omega_t = \omega_t'' \sum n_{ti} \, z_i \tag{5.11}$$

Where,

ω_t'' is the tyre wear rate (g km^{-1} per tyre), n_{ti} is the number of tyres/wheels in each vehicle (tyres/wheels per vehicle) and z_i is the respective proportion of each type of vehicle (%).

The wide range of tread wear rate for a single tyre between 0.006 g km^{-1} and 0.09 g km^{-1} is considered for the present model (Dannis, 1974; Fwa and Ang, 1991; Pierson and Brachaczek, 1974; Schuring and Clark, 1988). Traffic composition of Gloucester Place is obtained from Section 4.1.2 and that of Manor Road is obtained from Section 4.2.4. Using Equation 5.11, tyre wear rates are estimated to be between 2.4×10^{-2} g km^{-1} v^{-1} and 4.7×10^{-1} g km^{-1} v^{-1} at Gloucester Place and between 2.8×10^{-2} g km^{-1} v^{-1} and 5.5×10^{-1} g km^{-1} v^{-1} at Manor Road. As discussed earlier in this Section, besides road condition, the other factor that significantly influences the rate of tyre wear is vehicle speed. However, at present, this information is not available.

5.6.2.3 Dry deposition, F_d

Dry deposition is, broadly speaking, the transport of gaseous and particulate species from the atmosphere in the absence of precipitation (Seinfeld and Pandis, 1998). The dry deposition of particulate matter from the atmosphere to land and vegetation is quantified by "deposition velocity", v_d, defined as follows (Chamberlain and Chadwick, 1953; Seinfeld and Pandis, 1998):

$$v_d = \frac{D}{C} \tag{5.12}$$

Where,

v_d = deposition velocity of particulate matter (m s^{-1})

D = deposition flux on surface (g m^{-2} s^{-1})

C = concentration of particulate matter in air (g m^{-3})

Deposition velocity is an empirically determined quantity that depends on a variety of factors, such as particle size and density, interaction among meteorological parameters, chemical and physical characteristics of the particle, and the nature of surface itself (e.g., the rate of deposition in forests is greater than in the fields, deserts, and urban areas). The level of turbulence in the atmosphere, especially in the layer nearest the ground, governs the rate at which species are delivered down to the surface (Countess et al., 2001; Hofken et al., 1983; Schmel, 1980a; Seinfeld and Pandis, 1998; Thiessen et al., 1999).

The process of dry deposition of particles (and gases) is generally represented as consisting of three steps, each contributing to the v_d (Seinfeld and Pandis, 1998):

1. Aerodynamic transport down through the atmospheric surface layer to a very thin layer of stagnant air just adjacent to the surface.

2. Molecular (for gases) or Brownian (for particles) transport across this thin stagnant layer of air, called the quasi-laminar sublayer, to the surface itself.

3. Uptake at the surface.

For particles, one of the factors most strongly influencing the deposition velocity is the "particle size" (the other factor that significantly affects deposition velocity is "wind speed"). Rao et al. (1992) found that, by and large, the deposition velocity of a particle increases with the increase of its mass median diameter (MMD). However, velocities of dry deposition due to turbulent diffusion increase with decreasing size of particles (Gramotnev et al., 2003). The deposition velocity typically decreases with increasing distance from the release point, since the ratio of small to large particle concentrations increases due to the effective deposition of large particles (Thiessen et al., 1999). Some predominant mechanisms of dry deposition for particles of different size are described in Table 5.6.

Dry deposition is still a very important source of uncertainty. The majority of the emitted particles from a busy road is within the ultra-fine range with the median diameter $\leqslant 0.05\,\mu$m (Hitchins et al., 2000) and therefore increased rate of coagulation of such particles is expected (Jacobson, 1999). Moreover, a busy road is also a strong source of gaseous reactive pollutants, such as NO_x, which may lead to a possibility of nucleation processes, i.e., formation of new particles in the plume. However, there exists no clear understanding of these processes and their contributions near a busy road (Jacobson, 1999). A minimum deposition velocity is seen for particle between $0.1\,\mu$m and $1\,\mu$m and the typical estimate is $0.001\,\mathrm{m\ s^{-1}}$ with estimated uncertainty factor between 5 and 10 (Thiessen et al., 1999). Particles below $0.1\,\mu$m coagulate or attach to larger particles (Thiessen et al., 1999). Sehmel (1980a) reports the value of

Table 5.6: Dry deposition mechanism (Seinfeld and Pandis, 1998)

Particle size	Deposition mechanism	Description
$\approx 0.05\ \mu$m	Brownian diffusion	Small particles behave much like gases and are efficiently transported across the quasi-laminar layer by Brownian diffusion.
$0.05\ \mu$m to $2\ \mu$m	No effective mechanism	This is the size range for accumulation mode aerosols in the atmosphere and life-times of such particles by dry deposition may be many days unless they are scavenged by fog or precipitation.
$2\ \mu$m to $20\ \mu$m	Inertial impaction	These moderately large particles are efficiently transported across the laminar sublayer by inertial impaction.
$> 20\ \mu$m	Gravitational settling	These big particles settle by gravitational settling since the settling velocity, which increases with the square of the particle diameter, is appreciable in this case.

v_d, for a reference height of 1 m, to be 0.05 m s^{-1}. For a tropical climate like in India, Rao et al. (1992) reported mean dry deposition velocities for different ionic components in air varying between 0.0057 m s^{-1} for SO_4 of MMD 0.9 μm and 0.0229 m s^{-1} for Ca of MMD 3.3 μm. Other ionic compounds such as Cl, NO_3, NH_4, Na, K and Mg had MMDs and dry deposition velocities within the above range. Their observation of low standard deviations of seasonal values of dry deposition velocities indicated that dry deposition values are almost same throughout the year. A typical value of D for Germany is 1.16×10^{-6} g m^{-2} s^{-1} (Gámez et al., 2001b). For a PM_{10} level of 40 μg m^{-3}, this results in v_d value of 0.03 m s^{-1}. ApSimon et al. (2001) have reported dry deposition velocities of 0.001 m s^{-1} and 0.015 m s^{-1} for $PM_{2.5}$ and PM_{coarse} respectively. Findings of air quality experts panel, in association with US Western Regional Air Partnership (WRAP), show that the coarse fraction of PM_{10} with MMD of about 5 μm to 6 μm deposits relatively quickly at the rate of 0.005 m s^{-1} to 0.05 m s^{-1} compared with fine fractions with MMDs of about 0.3 μm to 0.5 μm which deposit at the rate of 0.0005 m s^{-1} to 0.05 m s^{-1} (Countess et al., 2001). As discussed in Table 5.6, the difference is due

to the fact that larger particles have higher gravitational settling velocity than small particles. Similarly particles are differently deposited by inertial impaction depending on the inertia of the individual particle; particles having higher inertia are deposited preferentially to particles having smaller inertia. Deposition by other mechanisms like thermophoresis and electrophoresis may also be important in particle removal. The two order of magnitude range in deposition rates (0.0005 m s^{-1} to 0.05 m s^{-1}) varies between two extremes: a maximum for dense forests and a minimum for barren desert areas, with a continuum of values between these two extremes for crop land, prairie land and urban (paved) surfaces (Countess et al., 2001).

The advantage of the deposition velocity representation is that all the complexities of the dry deposition process are bundled in a single parameter, v_d. The disadvantage is that, because v_d contains a variety of physical and chemical processes, it may be difficult to specify it properly. Because C is a function of height above ground, v_d is also function of height and must be related to a reference height (e.g., 10 m or less) at which C is specified (Seinfeld and Pandis, 1998).

The dry deposition is parameterised as follows:

$$\begin{aligned} F_d &= D\,l\,w \\ &= C\,v_d\,l\,w \end{aligned} \tag{5.13}$$

Where,

F_d = rate of dry deposition (g s^{-1})

D = deposition flux on surface (g m^{-2} s^{-1})

v_d = deposition velocity of particulate matter (m s^{-1})

C = concentration of particulate matter in air (g m^{-3})

l = length of the road segment (m)

w = width of the road segment (m)

In this model the background levels of PM_{10} (C_b), obtained from field measurements (Sections 4.1.5 & 4.2.3), are dry deposited. The wide range of v_d values from 5.0×10^{-4} m s^{-1} to 5.0×10^{-2} m s^{-1}, reported by Countess et al. (2001), is used. For Gloucester Place l and w are 120 m and 13 m respectively (Section 3.2.2) and for Manor

Road the corresponding values are 180 m and 5 m respectively (Sections 3.3.4 & 5.6.2.5).

Contribution from long-range transport of primary and secondary particulate matter is a part of background concentration. Local sources that contribute to airborne particulate matter, but not to the road surface loading, are part of local PM_{10}. Therefore both of these are dry deposited.

5.6.2.4 Local source at Gloucester Place: Material added onto the road surface by gritting, F_g

Grit is not a continuous source of road surface loading at Gloucester Place. Therefore, in the model, this source is considered to be active between 11:15 and 11:20, representing the start and the finish times of gritting process respectively (Section 3.2.2).

The grit application process is parameterised as follows:

$$F_g = \frac{S_g\, l\, w}{t_g} \tag{5.14}$$

Where,

F_g = rate of grit deposition (g s^{-1})

S_g = road surface loading contributed by gritting (g m^{-2})

l = length of the road segment (m)

w = width of the road segment (m)

t_g = duration of gritting (s)

S_g is 20 g m^{-2}, t_g is 300 s and l and w are 120 m and 13 m respectively, all of these values obtained from the design of the experiment (Section 3.2.2).

5.6.2.5 Local source at Manor Road: Material entering from EGTS to Manor Road, F_w

As discussed in Section 3.3.1, Erith Group Transfer Station (EGTS) has been suspected as the most important source and material carried from the unmade site of EGTS to Manor Road along the outer surface of the wheels of HGVs as the most plausible process that replenishes material on Manor Road (Bexley, 2003; Jansz et al., 2004). Visual observations of heavy soiling of half the width of the road on which loaded vehicles

from EGTS travel towards Erith strengthens this suspicion (Bexley, 2003). Therefore, in this model, EGTS has been considered as a significant source of Manor Road surface loading and material sticking to the outer surface of the wheels and thus carried along the road an important process. Material carry over from surrounding unpaved areas to nearby paved roads has also been recognised as a material replenishment process by earlier studies (AQEG, 2004; USEPA, 1993c). However, no information is available on either the amount replenished by such process or the procedure to quantify it. The AQEG (2004) has clearly expressed the lack of data in this area.

In this model the material input process from EGTS is parameterised in terms of thickness of material stuck on to the wheel surface and hence brought onto Manor Road. This may also include material carried in through other parts of the vehicle. This parameter value is finally arrived from the calibration of the model. Half of the carriageway of 5 m width on which loaded HGVs from EGTS travel towards Erith and which shows visible soiling, is considered for the modelling purpose (Fig. 3.12). Emissions from the other half of the carriageway, which are expected to be significantly less, are neglected. Therefore the model output is expected to be an underestimate of the actual PM_{10} emission from Manor Road.

The material input process is parameterised as follows:

$$\begin{aligned} F_{\mathrm{w}} &= f'_{\mathrm{w}}\, n_{\mathrm{w}} \\ &= f_{\mathrm{wh}}\, n_{\mathrm{wh}} n_{\mathrm{w}} \\ &= V_{\mathrm{w}}\, \rho_{\mathrm{w}} n_{\mathrm{wh}}\, n_{\mathrm{w}} \\ &= A_{\mathrm{wh}}\, \tau\, \rho_{\mathrm{w}}\, n_{\mathrm{wh}}\, n_{\mathrm{w}} \\ &= \pi\, \phi\, w_{\mathrm{wh}}\, \tau\, \rho_{\mathrm{w}}\, n_{\mathrm{wh}}\, n_{\mathrm{w}} \end{aligned} \tag{5.15}$$

Where,

F_{w} = rate of material input from EGTS onto Manor Road (g s^{-1})

f'_{w} = amount of material carried onto Manor Road by each vehicle (g v^{-1})

n_{w} = traffic flow rate from EGTS to Manor Road (v s^{-1})

f_{wh} = amount of material carried onto Manor Road by each wheel (g wh^{-1})

n_{wh} = number of wheels per vehicle (wh v^{-1})

V_w = volume of material carried onto Manor Road by each wheel (m^3 wh^{-1})

ρ_w = density of material entering from EGTS to Manor Road (g m^{-3})

A_{wh} = average wheel surface area (m^2 wh^{-1})

τ = thickness of material sticking onto wheel outer surface at the entry to Manor Road (m)

ϕ = average wheel diameter of the vehicles entering to Manor Road (m)

w_{wh} = average wheel width of the vehicles entering to Manor Road (m)

Estimate of n_w is obtained from Section 4.2.4. Value of ρ_w is assumed same as that of ρ (Section 5.6.2.1). From the on-field observation, ϕ, w_{wh} and n_{wh} are estimated to be 2 m, 0.4 m and 6 respectively. τ is obtained from model calibration, presented in Section 6.3.2.1.

5.6.3 Fluxes that deplete road surface loading, F_{out}

Gillette et al. (1974), while investigating the influence of wind velocity on wind erosion of soils, concluded that particles with settling velocity (V_s) sufficiently smaller than vertical velocities would remain suspended in air and be available for downwind transport. They found that threshold of suspension could be estimated in terms of friction velocity (u_*) and V_s. Particles for which V_s/u_* lie between 0.12 and 0.68 are likely to be suspended. Particles (typical size range is 100 μm to 500 μm) with greater values of V_s/u_* will tend to saltate (i.e., rise to a height of approximately less than 1 m and in absence of any appreciable vertical flux, will skip across the surface by horizontal flux (Anspaugh et al., 1975)), often associated with high speed rotation (Sehmel, 1980b) and massive ones (typical size range is 500 μm to 1000 μm) will tend to remain close to the surface and creep (i.e., roll along) across it (Newman et al., 1974). Therefore resuspension rate, Λ, defined as fraction (or percentage) of surface deposit removed per unit time can be expressed as (Sehmel, 1973):

$$\Lambda = \Lambda_{air} + \Lambda_{salt} + \Lambda_{sc} \tag{5.16}$$

Where,

Λ = total resuspension rate (s^{-1})

Λ_{air} = resuspension rate for particles moving in true airborne suspension (s^{-1})

Λ_{salt} = resuspension rate for particles moving in saltation (s^{-1})

Λ_{sc} = resuspension rate for particles moving in surface creep (s^{-1})

In this present model, unlike above where all the three processes are considered to contribute to resuspension, only the smaller particles that move in airborne suspension are considered to be contributing towards resuspension. Movement of larger particles across the road through saltation and creep is considered to be contributing primarily towards the across-street movement and, to some extent, the along-street on-wheel movement.

5.6.3.1 Flux along road, F_x

As discussed in Section 4.3.1, vehicle wake and material sticking to the outer surface of the wheel and hence carried along the direction of traffic are the two major processes that cause along-street movement of the material. Which mode will dominate depends primarily on moisture content of the material. Material with very high moisture content will be transported largely through sticking to the wheel whereas dry material will mostly be carried by vehicle wake.

The along-street movement of material is parameterised as follows:

$$F_x = f_x S l w n_v \tag{5.17}$$

Where,

F_x = amount of material removed from a road segment along the road in unit time by the whole fleet of vehicles (g s^{-1})

f_x = fraction of material removed from a road segment along the road by one vehicle (v^{-1})

S = surface material loading on the road segment (g m^{-2})

l = length of the road segment (m)

w = width of the road segment (m)

n_v = traffic flow rate (v s^{-1})

5.6.3.2 Flux across road, F_y

Two material movement processes, saltation and creep, described above, dominates the across-street movement of the material resulting in material build-up at the kerb. This is in agreement with the suggestion by Gámez et al. (2001a) that bigger particles are moved to kerbs. Previous researchers have attributed the higher silt loading values at kerbs than traffic lane to this across-street movement of material (Rauterberg-Wulff, 2000; Düring and Lohmeyer, 2001). Translocation of material to road parking strip has been reported by Sehmel (1973) who estimated that a visible material deposit on parking strip would happen after about 1000 vehicle passes.

Flux across the road is parameterised as follows:

$$F_y = f_y S l w n_v \tag{5.18}$$

Where,

F_y = amount of material removed from a road segment across the road in unit time by the whole fleet of vehicles (g s^{-1})

f_y = fraction of material removed from a road segment across the road by one vehicle (v^{-1})

S = surface material loading on the road segment (g m^{-2})

l = length of the road segment (m)

w = width of the road segment (m)

n_v = traffic flow rate (v s^{-1})

5.6.3.3 Emission to air/Resuspension, F_z

Particles smaller than 100 μm are largely removed by resuspension.

Resuspension of material is parameterised as follows:

$$F_z = f_z S l w n_v \tag{5.19}$$

Where,

F_z = amount of material removed from a road segment to air in unit time by the whole fleet of vehicles (g s^{-1})

f_z = fraction of material removed from a road segment to air by one vehicle (v^{-1})

S = surface material loading on the road segment (g m^{-2})

l = length of the road segment (m)

w = width of the road segment (m)

n_v = traffic flow rate (v s^{-1})

Estimates of parameter values for traffic-induced depleting fluxes

f_x, f_y and f_z are obtained from Sections 4.3.1, 4.3.2 & 4.3.3 respectively. S, for Gloucester Place, immediately after gritting is known from Section 3.2.2. At Manor Road, estimate of S is presented in Table 4.2. l and w, the dimensions of road segment, are estimated from the design of field experiment and application of model. For Gloucester Place, l and w are 120 m and 13 m respectively (Section 3.2.2) and for Manor Road the corresponding values are 180 m and 5 m respectively (Sections 3.3.4 & 5.6.2.5). n_v at Gloucester Place is estimated in Section 4.1.2. Estimate of n_v at Manor Road is obtained from Section 4.2.4.

5.6.3.4 Removal of material by rain, F_r

Precipitation, either in the form of rain or in the form of snow, significantly affects the material removal processes on road surface. While heavy rain causes removal of material in suspended form to the kerbs/gutters, snow acts as a barrier to road dust removal. When snow melts, it may have a similar effect as of rain. However, in both cases wet-material squeezed under the wheel and thrown across the road to kerbs/pavements will also be an important process. Although the EPA believes that snow would form more of a physical barrier to emission, they have no data to quantify additional reduction that can be attributed to snow over and above rain. Therefore they have not come up with any snow correction factor, and have opined that it will be a reasonable approach to assume zero emission during periods when the road surface is covered with snow

(Kuykendal, 2002; USEPA, 2002).

Not many studies are available on the combined influence of rain and traffic on material movement processes on paved roads. A German study shows that airborne particulate matter concentration reduction occurs with rainfall intensity as low as 0.1 mm h^{-1} (Düring et al., 2002a). The average reduction of emission factors during rain is 15 % to 20 % with the maximum reduction of 50 %. The study reported that the effect of rain in lowering the emission can continue until 7 hours after the rain has stopped. Initial AP-42 model assumed no emission to occur during days of rain. Muleski et al. (2001) found that introducing a moisture correction factor improved the predictability of the AP-42 emission factor model. After getting response to its October 2001 draft section 13.2.1 on "Paved roads", the EPA has recently updated the emission factor equations with a moisture correction term to take care of the rain event (Kuykendal, 2002; USEPA, 2002). It has included two correction factors, one for hourly basis and the other for daily basis, as follows:

$$\text{Hourly moisture correction factor} = 1 - 1.2P/N \tag{5.20}$$

$$\text{Daily moisture correction factor} = 1 - P/4N \tag{5.21}$$

Where,

P is the number of days/hours with at least 0.254 mm (0.01 in) of rain during the averaging period and N is the number of days/hours in the averaging period.

The above correction factors assume the following:

1. Long-term or annual average emissions are inversely proportional to the frequency of the measurable rain.

2. Measurable rain is rainfall of at least 0.254 mm (0.01 in).

3. When measurable rainfall occurs, the emissions are zero during the rain event.

4. Mitigative effect of 20 % takes place beyond the rain event.

Because PM_{10}, for regulatory purpose, is determined on a 24-hour average (and annual average), but not for a short duration of 1 hour, Chang (2001) advocated the use of

daily moisture correction factor. However, evaluating the influence of rain on emission of particulate matter is too complicated because, light and heavy rain may affect in different ways, for mitigation of the emission, as described below (Myers, 2001):

1. Light rain increases the availability of silt for suspension into the air as a result of the washing of the material out of crevices of the road. While the road surface is wet, water droplets suspended into the air will contain these suspended solids and if not deposited, will become airborne particulate emissions.

2. Light rain can increase silt levels as a result of the washing action of water on the undercarriage of the vehicles.

3. The timing of short duration rain events affects the reduction of silt during high traffic periods. Rain occurring late in the evening or at night is unlikely to significantly affect silt levels during high traffic periods since the undercarriage of the first vehicles to use the road will re-nourish the road surface due to washing by the residual water on the road surface.

4. Any rain can increase silt levels as a result of enhancing the track out of material from unpaved areas such as construction sites, unpaved roads, quarries and other industrial locations.

5. Heavy rain will remove most silt from the road surface. This effect will last the longest when rain occurs during rush hours.

6. Paved roads are designed to minimise the amount of water retained on the surface.

7. Normally, high speed and high traffic volume roadways will dry within an hour after rain has ceased.

8. Low humidity, high temperature and high solar insolation conditions will reduce drying times of a road.

For light rains of short duration, which appear to be the most prevalent, it is assumed that the times when silt content would be increased are balanced by the times

when the amount of rain is sufficient to reduce the silt content of the road. As a result, emissions are suppressed only during the period when it is raining. Myers (2001) suggested that since the meteorological data is usually recorded in one hour blocks, it may be reasonable to assume that emissions are suppressed for the entire hour. Gámez et al. (2001a) also express similar concerns about influence of precipitation on particulate matter emission from their studies in Germany.

Nicholson and Branson (1990) suggested that resuspension is likely to be controlled by the rate of incorporation of deposited material with soil or dust that may be translocated onto the road surface. Such incorporation may be most rapid when the surface is moist such that resuspension factors resulting from turbulence or tyre shear might attain the maximum value immediately after a road becomes dry. Rauterberg-Wulff (2000) estimated, from a three-months monitoring phase at Frankfurter Allee in Berlin, that the emission is reduced to one half during rainy days. Düring and Lohmeyer (2001) found, for a four-week monitoring phase in Schildhornstraße in Berlin, no reduction of PM_{10} emission on days with rain. These studies concluded that further measurements are needed for an understanding of the material movement processes associated with rain event.

In this model, however, removal of material by rain is parameterised with assumption that the amount of material removed is dependent on intensity of rain. No account has been taken of the after-rain mitigative effect.

Material removal by rain is parameterised as follows:

$$\begin{aligned} F_r &= f_r'' \, S \, l \, w \\ &= a \, (I_r)^b \, S \, l \, w \end{aligned} \tag{5.22}$$

Where,

F_r = amount of material removed from a road segment in unit time due to rain (g h^{-1})

f_r'' = fraction of material removed per unit time due to rain (h^{-1})

I_r = intensity of rain (mm h^{-1})

a & b = calibration coefficients

S = surface material loading on the road segment (g m^{-2})

l = length of the road segment (m)

w = width of the road segment (m)

The relation $f_r'' = a\,(I_r)^b$, used in Equation 5.22, is similar to the one reported from Tillydrone study. The study estimated values of a and b to be 0.015 and 1.5 respectively (Deletic et al., 1997; Deletic and Makismovic, 1998; Deletic et al., 2000).

5.6.4 Process summary

Table 5.7 and Figure 5.3 summarise various material movement processes included in this model and their mathematical formulations.

Table 5.7: Mathematical formulation of the processes

Process	From	To	Mathematical formulation
Exhaust emission	vehicle	air	$C_e = e_e\, n_v\, \chi$
Brake wear emission	vehicle	air	$C_{br} = e_{br}\, n_v\, \chi$
Road abrasion deposition	road	road	$F_a = \tau_a\, \psi\, n_v\, l\, w\, \rho$
Tyre wear deposition	vehicle	road	$F_t = \omega_t\, n_v\, l$
Dry deposition	air	road	$F_d = C_b\, v_d\, l\, w$
Gritting of road	-	road	$F_g = \frac{S_g\, l\, w}{t_g}$
Input from EGTS	road	road	$F_w = \pi\, \phi\, w_{wh}\, \tau\, \rho_w n_{wh} n_w$
Along-street movement	road	road	$F_x = f_x\, S\, l\, w\, n_v$
Across-street movement	road	kerb	$F_y = f_y\, S\, l\, w\, n_v$
Resuspension	road	air	$F_z = f_z\, S\, l\, w\, n_v$
Removal by rain	road	gutters	$F_r = a\,(I_r)^b\, S\, l\, w$

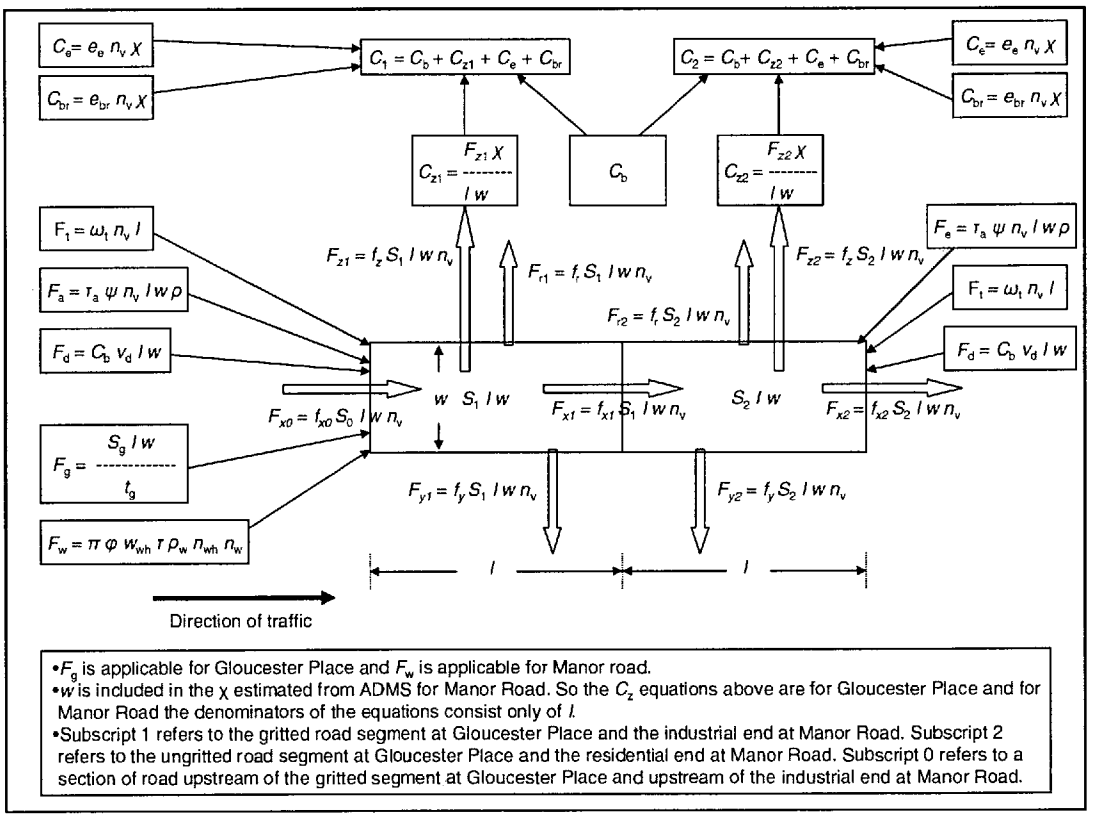


Fig. 5.3: Summary of process parameterisation.

5.6.5 Other processes

Some processes which are common to any urban paved road, but not considered in this model either due to lack of data or due to lack of understanding of the process, are described below.

5.6.5.1 Wet deposition

Removal of material from atmosphere by precipitation (i.e., rain or snow) is termed as wet deposition. Material may be incorporated into the precipitation either in-cloud or below-cloud (Nicholson et al., 1991). However for most of the urban area studies where measurements of particulate matter are done at a height below where clouds form, the below-cloud scavenging is the most important process. The efficiency of wet deposition is controlled exclusively by atmospheric processes (precipitation physics and chemistry) (Hofken et al., 1983). The wet deposition rates are proportional to the precipitation

rates and wet deposition will always be higher than the dry depositions (Barrie, 1988; Chan et al., 1986; Mosello et al., 1988). Rao et al. (1992) reported wet deposition fluxes varying between 1.5 and 2.0 times of dry deposition fluxes for different atmospheric ionic compounds. The rate of wet deposition is expressed in terms of "scavenging coefficient", defined as follows (Nicholson et al., 1991):

$$\frac{dC}{dt} = -\Lambda_s C \tag{5.23}$$

Where,

C = particulate matter concentration (μg m^{-3})

Λ_s = scavenging coefficient (s^{-1})

For a particle of 4.3 μm and collection efficiency of unity, Table 5.8 shows a summary of values for Λ_s. For particles bigger than this, the Λ_s value may go up by a factor of 3 to 4 of this estimate. For drizzle (i.e., rainfall rate < 0.1 mm h^{-1}), Nicholson et al. (1991) measured Λ_s between 1×10^{-4} s^{-1} and 6×10^{-4} s^{-1} for particles $9.0\,\mu\text{m} \pm 2.5\,\mu$m. For snow, Λ_s value is 4×10^{-3} s^{-1}, a factor of 10 higher than that for rainfall.

Table 5.8: Scavenging coefficients (Nicholson et al., 1991)

Rainfall intensity (mm h^{-1})	Λ_s (s^{-1})
0.1	7×10^{-5}
0.2	1×10^{-4}
0.4	2×10^{-4}
0.6	3×10^{-4}
0.8	3×10^{-4}
1.0	4×10^{-4}
1.5	5×10^{-4}
2.0	7×10^{-4}
2.5	8×10^{-4}

5.6.5.2 Removal of material due to road sweeping

Uncertainty associated with material removal due to road sweeping is discussed in Section 2.12.1.1. However, studies on industrial paved roads have shown the effect of sweeping in lowering the emission (Cowherd et al., 1982; Cuscino et al., 1983). Industrial paved roads are characterised by higher particulate matter loading than

public paved roads and therefore it indicates sweeping is effective where the particulate matter loading is high. Extending this argument to paved roads, especially where resuspension is several times that of exhaust emission, the effect of sweeping should be seen and a process could be added to the model. Information on timing of sweeping and sweeping efficiency, which depends on the sweeping machine and road surface characteristics, can help in modelling this as an independent process. At present, we may assume that it is implicitly included in across-street removal of the material.

5.7 Comparison of model structure with other environmental models

The structure of this model is similar to other environmental models, although those models are mainly confined to the area of radioactive decay, photolysis, biodegradation, environmental risk assessment and exposure analysis. Two environmental models that are based on process-level studies and therefore have a similar structure to this are CalToX[2] and CLEA [3] (CalTOX, 1993; CLEA, 2002).

Various compartments and transfer processes of material within and across the compartment of this new model are similar to CalTOX multimedia transport and transformation model with the difference that unlike CalTOX, which is a seven compartment model (compartments are air, plants, ground-surface soil, root-zone soil, vadose-zone soil, surface water and sediment layer), this model has only two compartments, i.e., air and road segment, making it much simpler. Further, unlike CalTOX, which includes chemical species and their transformation, this model does not include either any gaseous pollutant or any physical and chemical transformation process (CalTOX, 1993; McKone et al., 1997). Instead it is a model for particulate pollutants only. Re-

[2]CalTOX is a spreadsheet model, developed under the auspices of the University of California with funding provided by the Department of Toxic Substances Control of Californian Environmental Protection Agency to assist in assessing human exposures and health risk at uncontrolled hazardous waste sites.

[3]Contaminated Land Exposure Assessment (CLEA) model is developed by the Centre for Research into the Built Environment, Nottingham Trent University, on behalf of the Department for Environment, Food & Rural Affairs (DEFRA) and Environment Agency, UK for estimating child and adult exposure risks from soil contaminants for those potentially living, working and playing on contaminated sites over long periods of time.

suspension from road surface is analogous to the process of transfer of material from soil compartment to air in CLEA (CLEA, 2002). Both in CLEA and CalTOX model, the rate of loss of material from any compartment is defined as the product of the concentration of the material in the compartment and transfer rate constant. The transfer rate constants have their usual units, i.e., per unit time and expressed, like this model, as the fraction per unit time of the inventory (e.g., mass of particulate matter) of the compartment i (e.g., road segment) that is transferred to compartment j (e.g., air or to adjacent road segment or to the kerb). The only difference in defining these transfer processes is that the transfer and resuspension rate constants in this model are expressed in units of per vehicle (v^{-1}), which, after multiplication with vehicle flow rate ($v\ s^{-1}$), gives rise to the loss rate constants, the standard unit of which is per unit time (s^{-1}). To begin with, in CalTOX, the transfer and transformation rate constants were described in terms of landscape properties, chemical properties, and fugacity capacities (CalTOX, 1993). In its 1997 revision, however, McKone et al. (1997) have modified the units for source terms to mass per unit time, very much the same as this model. The parameterisation of resuspension and transfer rate constants in this model is also analogous to the deposition rate constant of molecules from a surface and chemical activation rate constant (Reeks et al., 1988), widely used by researchers working in the area of material resuspension.

5.8 Testing model for the scientific correctness and model validation

Subject to constraints of quantitative knowledge, simulation models should attempt to reproduce the reality. In practice, however, the environment and its inherent processes are so complex and variable in space and time that it will probably never be simulated perfectly. Representative sampling under a robust statistical design, followed by a well-controlled and calibrated laboratory analysis, is most likely to yield the best available (though not necessarily correct) estimate of the true value. Most scientists would prob-

ably tend to have more confidence in a quantitative estimate obtained through direct measurement than through a model prediction. It is for this reason that the comparison of model predictions to measured data has gained considerable acceptance as an important tool in the process of establishing model credibility (Kirchner and Whicker, 1984; Kirchner, 1991; Whicker et al., 1999). This process is often called model validation, even though one can always argue that a model can never be validated, only invalidated (Oreskes et al., 1994). The basic ground rule for the model validation is that data for model validation must be other than data used model construction or calibration (Nielsen et al., 1991). Model validations have their limitations also. No model will ever be tested under all possible combinations of times, places and conditions. Kirchner (1991) discusses this concept, referring to the "domain of applicability". The applicability of the model should clearly be described.

In this model, the estimates for traffic emission sources common to any road (i.e., C_e, C_{br}, F_a, F_t and F_d) are estimated from available data describing the road, the traffic on it and prevailing meteorological condition (Sections 5.6.1, 5.6.2.1, 5.6.2.2 & 5.6.2.3 respectively). F_g is obtained from the experimental design at Gloucester Place (Sections 3.2.2 & 5.6.2.4). F_w is calibrated to fit the measurements available at Manor Road (Section 5.6.2.5). Parameter values for F_x, F_y, F_z are estimated from the Gloucester Place experiment (Sections 3.2 & 4.3) and that for F_r is obtained from previous works (Deletic et al., 1997; Deletic and Makismovic, 1998; Deletic et al., 2000) (Sections 5.6.3.4).

In case of Gloucester Place, the model output is compared with a time-series of roadside PM_{10} near two road segments 120 m apart (Fig. 4.9), to check the scientific correctness of the model formulation. The same PM_{10} data is also one of the measurements that is used to derive the parameter values for some of the fluxes that deplete the road surface loading. However, the traffic-linked processes included in this model are independent of measurement and therefore, checking the scientific correctness of the model formulation consists of assessing whether the measurements used to derive parameter values can be reproduced by a model that considers these processes to be responsible for the traffic-induced resuspension. The model validation at Manor

Road consists of comparing the model output with roadside PM_{10} at one location (Section 4.2.3) and road surface loading at two road segments 180 m apart (Section 4.2.5), obtained from independent measurements.

Chapter 6

Model results

6.1 Introduction

Table 6.1 presents the ranges for the model input parameters for road and traffic conditions at Gloucester Place and Manor Road.

Table 6.1: Summary of input parameter values

Parameter	Unit	Estimated range	Field site
e_e [a]	g km^{-1} v^{-1}	3.4×10^{-2} to 8.9×10^{-2}	Gloucester Place
		5.8×10^{-2} to 1.0×10^{-1}	Manor Road
e_{br} [b]	g km^{-1} v^{-1}	8.2×10^{-3} to 1.5×10^{-2}	Gloucester Place
		1.1×10^{-2} to 1.9×10^{-2}	Manor Road
ω_t [c]	g km^{-1} v^{-1}	2.4×10^{-2} to 4.7×10^{-1}	Gloucester Place
		2.8×10^{-2} to 5.5×10^{-1}	Manor Road
v_d [d]	m s^{-1}	5.0×10^{-4} to 5.0×10^{-2}	
τ_a [e]	m v^{-1}	1.8×10^{-11} to 1.0×10^{-9}	
f_x [f]	v^{-1}	2.6×10^{-4} to 7.1×10^{-4}	
f_y	v^{-1}	3.3×10^{-5} to 9.1×10^{-5}	
f_z	v^{-1}	2.1×10^{-4} to 5.4×10^{-4}	

[a] Derived from exhaust emission factors given by NAEI (2003); the lower values are for traffic with petrol cars, taxis and LGVs and the higher values are for traffic with diesel cars, taxis and LGVs.

[b] Derived from brake wear emission factors given by AQEG (2004) and by Berdowski et al. (2001).

[c] Derived from the range of 6.0×10^{-3} g km^{-1} per tyre to 1.2×10^{-1} g km^{-1} per tyre (Dannis, 1974; Fwa and Ang, 1991; Pierson and Brachaczek, 1974; Schuring and Clark, 1988; Göttle, 1979; Muschack, 1990).

[d] This wide range is reported by Countess et al. (2001).

[e] The lower and higher limits are reported by Muschack (1990) and Sieker and Grottker (1988) respectively.

[f] f_x, f_y and f_z are estimated from the Gloucester Place experiment.

The parameter values for traffic-induced material removal processes (f_x, f_y and f_z) are obtained from the Gloucester Place experiment (Chapter 4) and those for universal processes (e_e, e_{br}, ω_t, v_d and τ_a) are obtained from reviews presented in Chapter 5. The range of e_e mentioned in the Table 6.1 is applicable for a traffic-flow with a speed of 20 km h^{-1}, averaged over a typical urban drive-cycle, that is expected to be the case at Gloucester Place taking into account of two traffic lights that control the traffic movement. However, the traffic-lights also cause a stop-and-go traffic situation and this results in higher exhaust emissions than the estimates obtained by using the NAEI emission factor[1] because acceleration and deceleration of vehicles produce more exhaust emissions than when they move without any obstruction with an average speed (USEPA, 1997). No attempt has been made to estimate this because of unavailability of emission factors for such a traffic-flow. Consequently, the exhaust emission estimates by the model are an underestimate of the actual emission.

Within the ranges of parameter values mentioned in Table 6.1, this chapter presents the model assessment consisting of two parts: (a) testing of the scientific correctness and the self-consistency of the model; and (b) applicability of the model in different field condition. Evaluation of the scientific correctness, presented in Section 6.2, is done by comparing model output with measurement for a location (i.e., Gloucester Place) and for a period that is also used to derive parameter values for the material removal processes. It however includes independent data for universal sources and the model formulation itself is independent of the measurements. The aim is to see whether the model formulation can explain the measurement and therefore be considered as representative of the processes occurring at the road-vehicle environment. The section also includes the source apportionment for universal processes to examine their importance in contributing to road surface loading and to estimate the more likely value for road abrasion which appears to be the dominant source in the absence of any local major source (Section 6.2.1.1). Section 6.3 examines the transferability of the parameter values and the model to another location (i.e., Manor Road) with different source and

[1]The NAEI database provides speed-dependent average emission factors for each category of the vehicle (NAEI, 2003).

traffic conditions and with different road geometry.

6.2 Site 1: Gloucester Place

6.2.1 Pregritting model performance

6.2.1.1 Calibration of parameter value for road abrasion

The aim of the calibration process, summarised in Table 6.2 and in Fig. 6.1, is to find the parameter value for road abrasion that causes the model to reproduce the PM_{10} measurement at Gloucester Place in the absence of any local source (e.g., gritting). To begin with, minimum values for parameters e_e, e_{br}, ω_t and v_d (Table 6.1) were used and a zero road abrasion case ($\tau_a = 0$) was assumed. Comparison of PM_{10} model prediction with the pre-gritting PM_{10} level, recorded for 30 minute at Site 1, indicated that the model with these parameter values is unable to produce enough material on the road segment to match the measurement.

Table 6.2: Determination of appropriate abrasion rate

Parameter value					PM_{10} level	Remark
e_e (g km^{-1} v^{-1})	e_{br} (g km^{-1} v^{-1})	ω_t (g km^{-1} v^{-1})	v_d (m s^{-1})	τ_a (m v^{-1})		
8.9×10^{-2}	1.5×10^{-2}	4.7×10^{-1}	5.0×10^{-2}	0	C_{2a}[a]	Too low
8.9×10^{-2}	1.5×10^{-2}	4.7×10^{-1}	5.0×10^{-2}	1.8×10^{-11}	C_{2b}	Too low
8.9×10^{-2}	1.5×10^{-2}	4.7×10^{-1}	5.0×10^{-2}	1.0×10^{-9}	C_{2c}	Too high
8.9×10^{-2}	1.5×10^{-2}	4.7×10^{-1}	5.0×10^{-2}	2.5×10^{-10}	C_{2d}	Optimum

[a]Subscripts '1' and '2' represent variables at the gritted and ungritted road segment at Gloucester Place and variables at the industrial and residential end at Manor Road respectively. Variable having equal magnitude on either segment of a road are denoted without any of these two subscript. Subscript 'm' stands for variables either measured directly or estimated from the measurement and variables without this subscript or with any other subscripts are the ones predicted by the model. All particulate matter concentration in air refers to PM_{10} size.

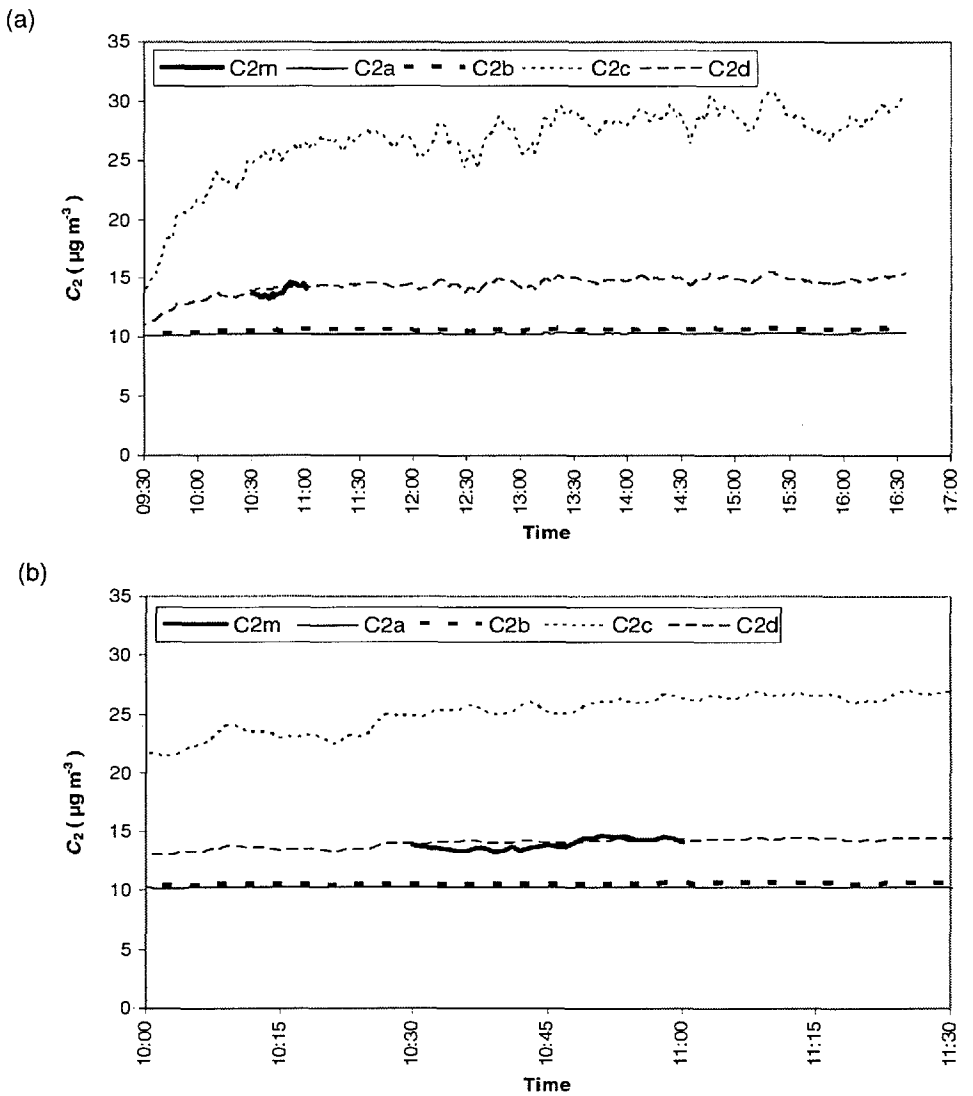


Fig. 6.1: Comparison of PM_{10} model prediction with measurement in absence of gritting: (a) Model prediction for entire duration of modelling and measurement available for short duration; and (b) Enlarged view of model prediction and measurement for a short duration.

The next step involved increasing parameters e_e, e_{br}, ω_t and v_d to their maximum reported values keeping τ_a zero. The model prediction (C_{2a}) still fell short of the measurement (C_{2m}) (Fig. 6.1). This indicated that, in the absence of any major local source, abrasion is the main source of material on a paved road surface. This agrees with the observation reported by earlier researchers (Gámez et al., 2001b). However inclusion of the minimum reported value for road abrasion rate ($\tau_a = 1.8 \times 10^{-11}$ m v^{-1}) did not cause any improvement to the model prediction (C_{2b}) from the previous one with zero road abrasion. This indicated that the minimum reported value of τ_a is too

low for an urban paved road. On the other hand, the maximum reported value for road abrasion ($\tau_a = 1.0 \times 10^{-9}$ m v^{-1}) resulted in model prediction (C_{2c}) much higher than the measurement. The model calibration indicated that the τ_a value of 2.5×10^{-10} m v^{-1} (corresponding model prediction is C_{2d}) is about the right approximation for the rate of the road abrasion.

6.2.1.2 Road surface loading

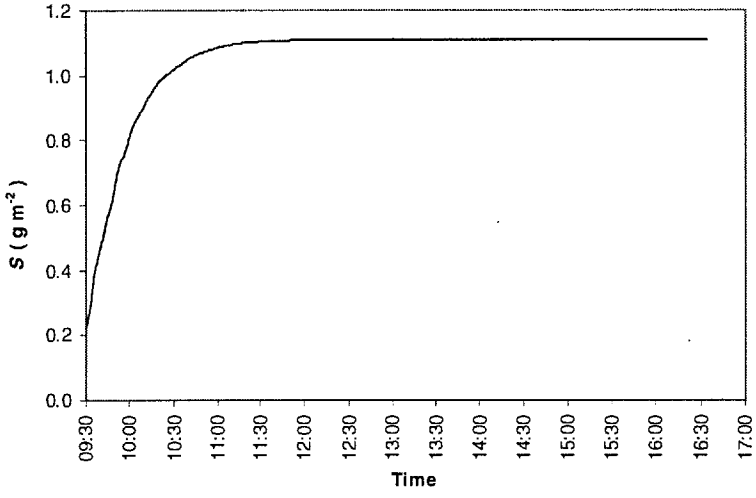


Fig. 6.2: Road surface loading.

Fig. 6.2 shows that, with present model parameterisation, the surface loading (S) at Gloucester Place is 1.1 g m^{-2}. For the US urban paved roads with significantly less traffic flow rate, the EPA has reported silt fraction between 3.5% to 21.7% (USEPA, 1997). Gloucester Place has a very high traffic and therefore it is expected that the silt fraction at Gloucester Place will be higher than the values reported for US condition. However, applying this range to the Gloucester Place results in the silt loading estimate between 0.04 g m^{-2} and 0.24 g m^{-2}. This estimate is within the silt loading range of 0.01 g m^{-2} to 1.02 g m^{-2}, reported by the EPA for high-ADT[2] roads in the USA (Section 2.12.1).

[2]High-ADT roads refer to roads with at least 5000 vehicles per day (USEPA, 1997).

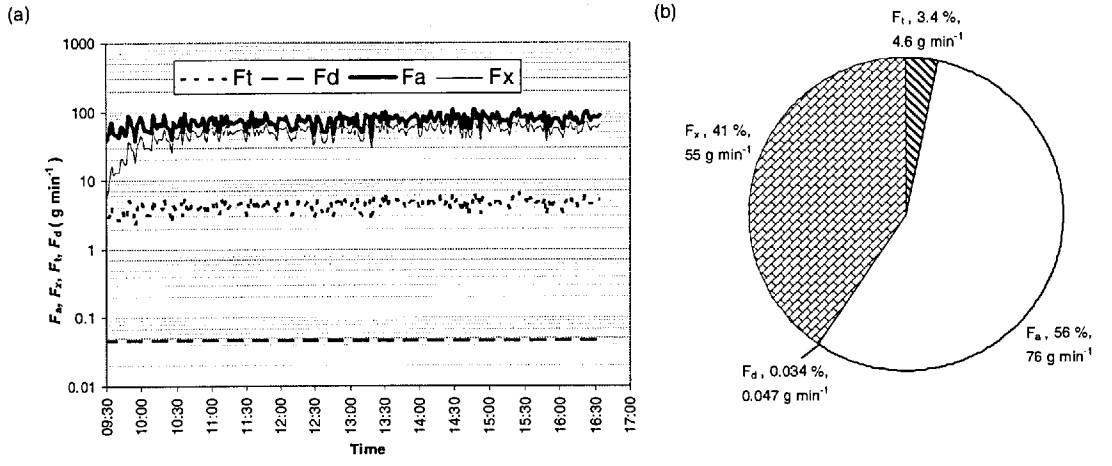


Fig. 6.3: Source apportionment of road surface loading: (a) Time series of road abrasion (F_a), material input from the upstream road segment (F_x), tyre wear (F_t) and dry deposition (F_d); and (b) Average contribution of these sources.

In addition to the general assumption that the sources for surface loading are the universal sources road abrasion, tyre wear deposition and dry deposition, the model suggests that the along-street flow of material from the upstream section is one more source that contributes to surface loading at any road segment. Fig. 6.3a shows that dry deposition (F_d), with a deposition rate of about 0.05 g min^{-1}, contributes the least among all the sources that replenish the road surface loading. Corresponding figures for road abrasion (F_a), material input from the upstream segment (F_x) and tyre-wear erosion (F_t) are 50 g min^{-1} to 100 g min^{-1}, 30 g min^{-1} to 70 g min^{-1} and 2 g min^{-1} to 6 g min^{-1} respectively. Road abrasion is the largest single source of surface loading. Material moved in from the upstream road segment is also a significant source of surface loading. On average F_a, F_x, F_t and F_d contribute 56%, 41%, 3.4% and 0.034% respectively to the surface loading (Fig. 6.3b).

6.2.1.3 Material removal fluxes

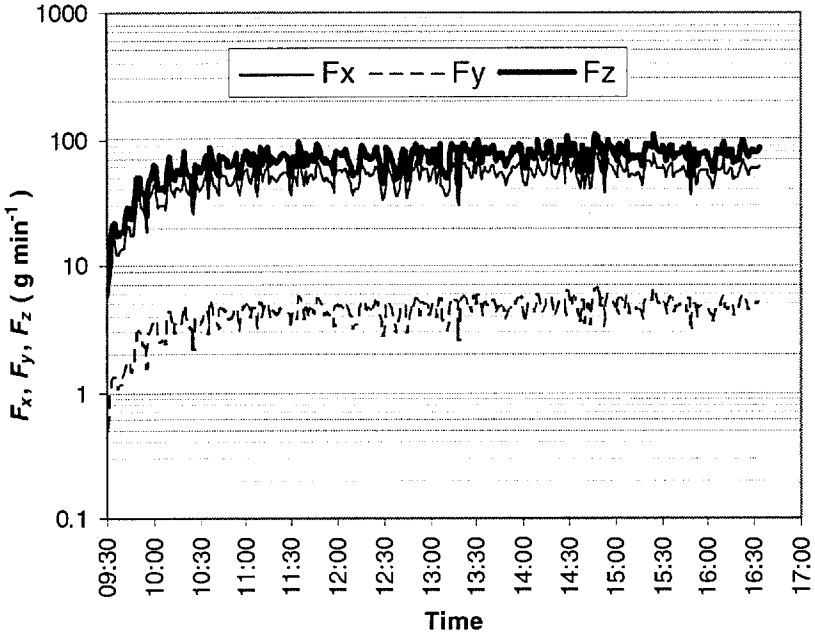


Fig. 6.4: Time series of material removal fluxes.

The depleting fluxes F_x, F_y and F_z at both road segments are proportional to f_x, f_y and f_z, the parameters that control these fluxes; i.e., F_x and F_z are about a factor of 10 higher than F_y (Fig. 6.4). The variations of these fluxes are entirely due to the varying traffic flow rate on Gloucester Place.

6.2.1.4 PM_{10}

The contribution of local background (C_b) to PM_{10} at Gloucester Place, which is about 1000 times that contributed by brake wear (C_{br}) and 200 times that contributed by exhaust emission (C_e), is the dominant source (Fig. 6.5a). Contribution from resuspension (C_z) is 100 times higher than contribution from the exhaust emissions. This shows the importance of having a credible emission inventory for resuspension in an urban area with an extensive road network. On average, roadside PM_{10} at Gloucester Place comprises of 68% of background ground level (C_b), 32% of resuspension (C_z), 0.36% of exhaust emission (C_e) and 0.06% of brake wear emissions (C_{br}) (Fig. 6.5b). Source apportionment of PM_{10} from resuspension is done with the assumption that emissions from each source is proportional to its contribution to the surface loading.

This gives the estimate of abrasion (C_a), material input from upstream segment (C_x), tyre wear (C_t) and dry deposition (C_d) contribution of 18%, 13%, 1.1% and 0.01% respectively to roadside PM_{10} at Gloucester Place (Fig. 6.5c).

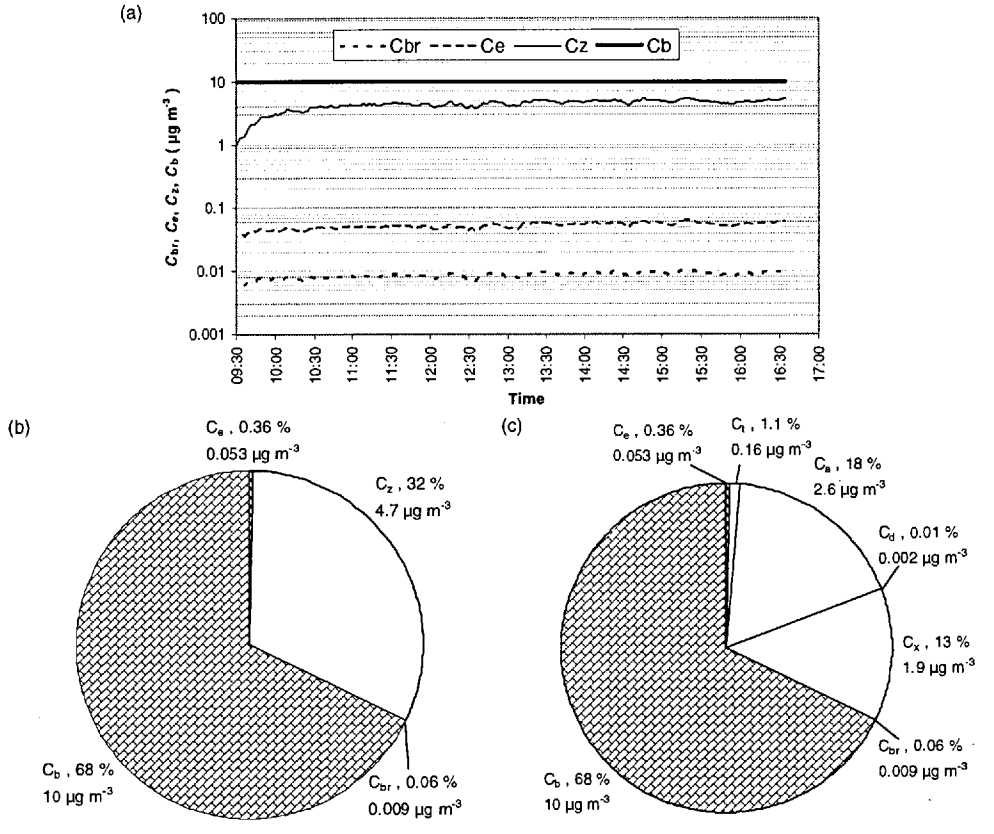


Fig. 6.5: (a) Time series of PM_{10} from background (C_b), exhaust (C_e), brake wear (C_{br}) and resuspension (C_z); (b) Average contribution of background, exhaust, brake wear and resuspension; and (c) Average contribution of background, exhaust, brake wear, tyre wear (C_t), abrasion (C_a), dry deposition (C_d) and material input from the upstream road segment (C_x).

6.2.2 Gritting event and subsequent material movement

6.2.2.1 Comparison of model prediction with measurement

Fig. 6.6 shows the comparison of maximum (C_{1max} & C_{2max}[3]) and minimum (C_{1min} & C_{2min}) model predictions with the measurement (C_{1m} & C_{2m}). The effect of gritting is marked by the sharp rise of roadside PM_{10} shortly after gritting at both sites. During the time of material transport from the gritted road segment to the ungritted road

[3]Subscripts '1' and '2' represent variables at the gritted and ungritted road segment at Gloucester Place (Fig. 3.4).

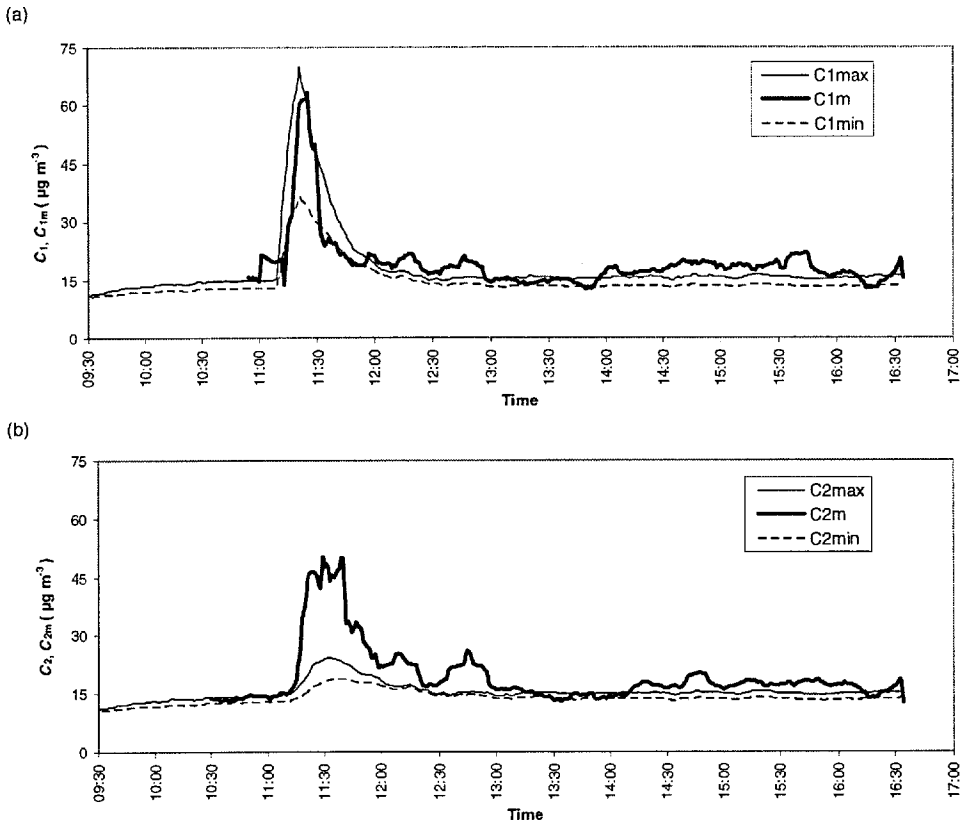


Fig. 6.6: Range of model prediction: (a) PM_{10} at the gritted road segment; and (b) PM_{10} at the ungritted road segment.

segment, some of it is lost to air and to the kerbs and therefore the material available at any time on the ungritted segment is lower than that available on the gritted segment. This results in lower peaks of PM_{10} level at the ungritted segment than the peaks at the gritted segment immediately after gritting. However, with time, more and more material from the gritted segment moves to the ungritted segment through the along-street material movement process and, after about an hour and half, the PM_{10} level at both sites reaches the pregritting level indicating removal of the majority of grit from the road surface. As discussed earlier (Section 4.1.5), the rise in PM_{10} level between 13:54 and 16:14 was due to a rise in urban background level, which is not reflected in model output because the model uses the the lowest pre-gritting PM_{10} level as the background level for Gloucester Place (Section 4.1.5.1). Occasional spikes above the pregritting level observed from the measurement, but not predicted by the model, could be due to occasional lift-up by the heavy vehicles of material locked in

road interstices that is otherwise not available for resuspension. It could also be due to change of dispersion conditions from quick variation of wind speed and direction that the model could not detect due to its application of a single 12-minute averaged dispersion relation obtained from the on-site tracer study. The difference between the maximum and minimum model prediction on both road segments is prominent during the first hour when rapid material transfer from the gritted segment to the ungritted segment took place. Within the ranges of parameter values obtained from the experiment, the maximum model prediction is obtained when f_x and f_y are minimum and f_z is maximum. The minimum model prediction is obtained when f_x and f_z are minimum and f_y is maximum. Variation of f_y causes smaller change to model output than the variation of f_x and f_z does. This indicates the model is less sensitive to parameterisation of the material removal process across the road.

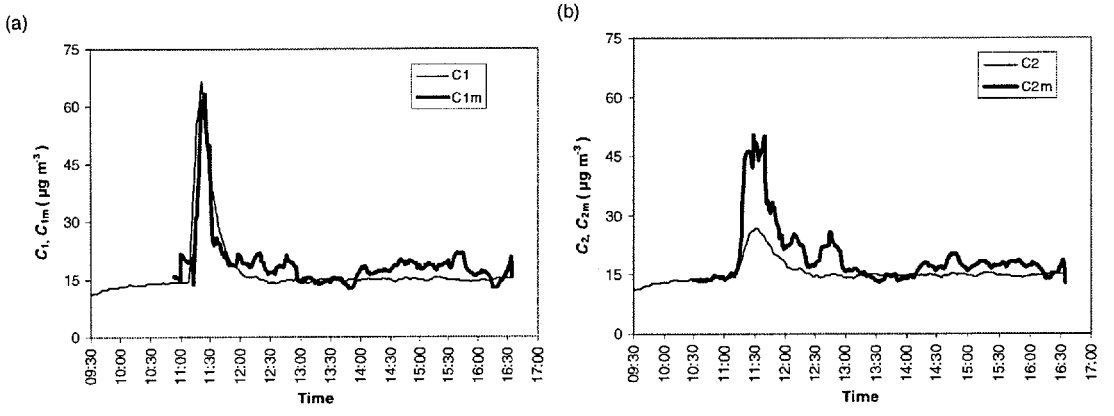


Fig. 6.7: Optimum model prediction: (a) PM_{10} at the gritted road segment; and (b) PM_{10} at the ungritted road segment.

The optimum model prediction is obtained when f_x is average, f_y is minimum and f_z is maximum, indicating removal of material to air as the dominant process (Fig. 6.7). While the model closely predicts the PM_{10} measurement at the gritted segment, it underpredicts the corresponding measurement by a factor of 2 at the ungritted segment during the first hour of rapid material movement. Possible reasons for this are discussed in Section 7.2.6.

6.2.2.2 Road surface loading

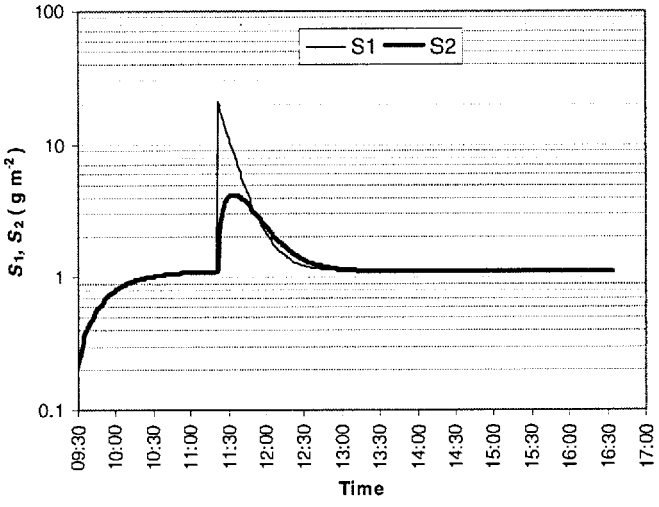


Fig. 6.8: Road surface loading.

About two hours after the gritting event, the surface loadings on both road segments reached their pre-gritting level of 1.1 g m^{-2} (Fig. 6.8). The surface loading at the gritted segment (S_1), immediately after gritting, reached a maximum value of 21.1 g m^{-2}, the sum of the surface loading before gritting and additional surface loading due to gritting. However the highest surface loading at the ungritted segment (S_2) was 4.2 g m^{-2}. This reduced peak surface loading at the ungritted segment is the result of loss of material across the road and to air during the time material moved from the gritted segment to the ungritted segment.

6.2.2.3 Material removal fluxes

The depleting fluxes at both road segments are proportional to f_x, f_y and f_z, the parameters that control these fluxes; i.e., fluxes along the road and to air are about 10 times higher than flux across the road (Fig. 6.9). During the first hour after the gritting event, depleting fluxes at the ungritted segment (F_{x2}, F_{y2} and F_{z2}) were lower than the corresponding fluxes of the gritted segment (F_{x1}, F_{y1} and F_{z1}) because of lower surface loading at the ungritted segment (S_2) than the surface loading at the gritted segment (S_1), discussed in Section 6.2.2.2.

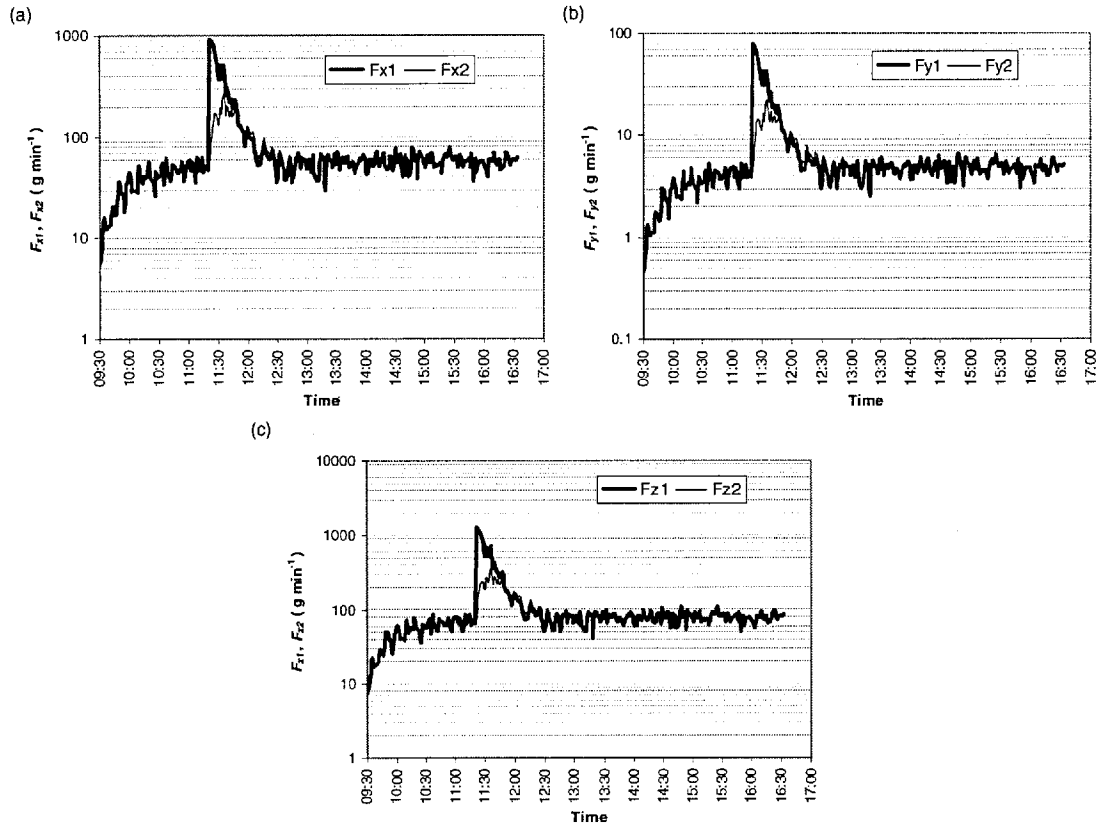


Fig. 6.9: (a) Along-street flux; (b) Across-street flux; and (c) Flux to air.

6.2.2.4 PM_{10}

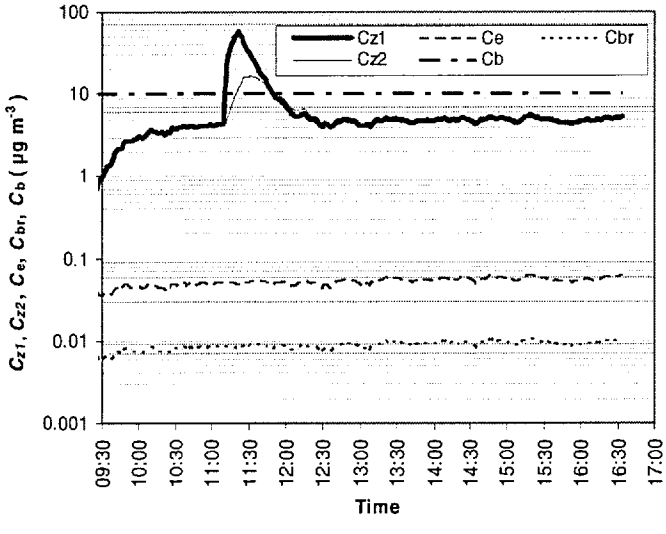


Fig. 6.10: Time series of PM_{10} from background (C_b), exhaust (C_e), brake wear (C_{br}) and resuspension (C_{z1} & C_{z2}).

The contribution of background, exhaust and brake wear to roadside PM_{10} remained the same as that before gritting. The important difference is that, for about 30 minutes after the gritting event, PM_{10} from resuspension was higher than the background PM_{10}. The highest modelled contribution of resuspension to roadside PM_{10} was 53 μg m^{-3} at the gritted segment and 16 μg m^{-3} at the ungritted segment, approximately 13 times and 4 times of the pregritting resuspension contribution (Fig. 6.10).

6.3 Site 2: Manor Road

Erith Group Transfer Station (EGTS), a waste recycling site at Manor Road, is suspected as the major source of surface loading at Manor Road. In order to appreciate the contribution of EGTS as a source at Manor Road, at first it is not included in the model as a source so that the model predicts a likely particulate matter level at Manor Road in absence of EGTS (Section 6.3.1). Subsequently this source is added to the model and output is compared with measurement (Section 6.3.2).

6.3.1 Model output with no material input from EGTS

6.3.1.1 Comparison of model prediction with measurement

Fig. 6.11 shows that, with no material input from EGTS, the model underpredicts the PM_{10} and surface loading measurements even when all the parameters for universal sources are at their optimum values (Section 6.2.1.1). The 90th percentile of the hourly PM_{10} model prediction is twice the 90th percentile of the hourly measured background PM_{10}. However, it is 14 μg m^{-3} less than the 90th percentile of the hourly measured PM_{10} at Manor Road. (Fig. 6.11a).

The difference between measurement and the model prediction indicates that the model is missing one or more sources at Manor Road. EGTS has been suspected to be the important missing source (Sections 3.3.1 & 5.6.2.5). The performance of the model with inclusion of material input from EGTS is presented in Section 6.3.2.

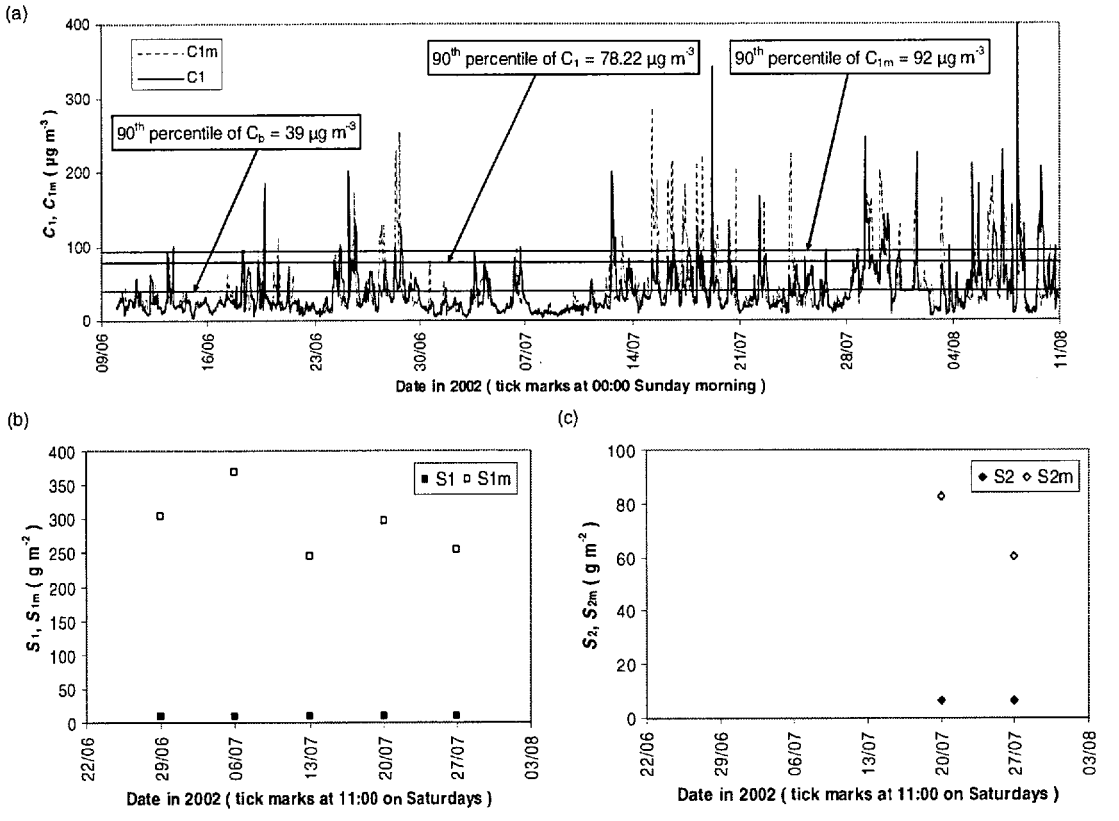


Fig. 6.11: Comparison of model prediction with measurement: (a) PM_{10} at the industrial end; (b) Surface loading at the industrial end; and (c) Surface loading at the residential end.

6.3.1.2 Road surface loading

Fig. 6.12 shows that, in absence of any local source and without the effect of rain, the model estimation for surface loading on road segments at either end of Manor Road is 3.7 g m^{-2}. Applying the average relationship between silt loading and surface loading at the industrial end, presented in Table 4.2, to the above estimate, the silt loading at Manor Road becomes 0.85 g m^{-2}. This estimate is within the silt loading range of 0.01 g m^{-2} to 1.02 g m^{-2}, reported by the EPA for high-ADT[4] roads in the USA (Section 2.12.1). The modelled influence of rain on lowering the silt loading is clearly seen for rain events with intensity of rain (I_r) beyond 1 mm h^{-1}. A rainfall event with very high intensity of rain above 10 mm h^{-1} has reduced the surface loading by a factor of 3.

[4]High-ADT roads refer to roads with at least 5000 vehicles per day (USEPA, 1997).

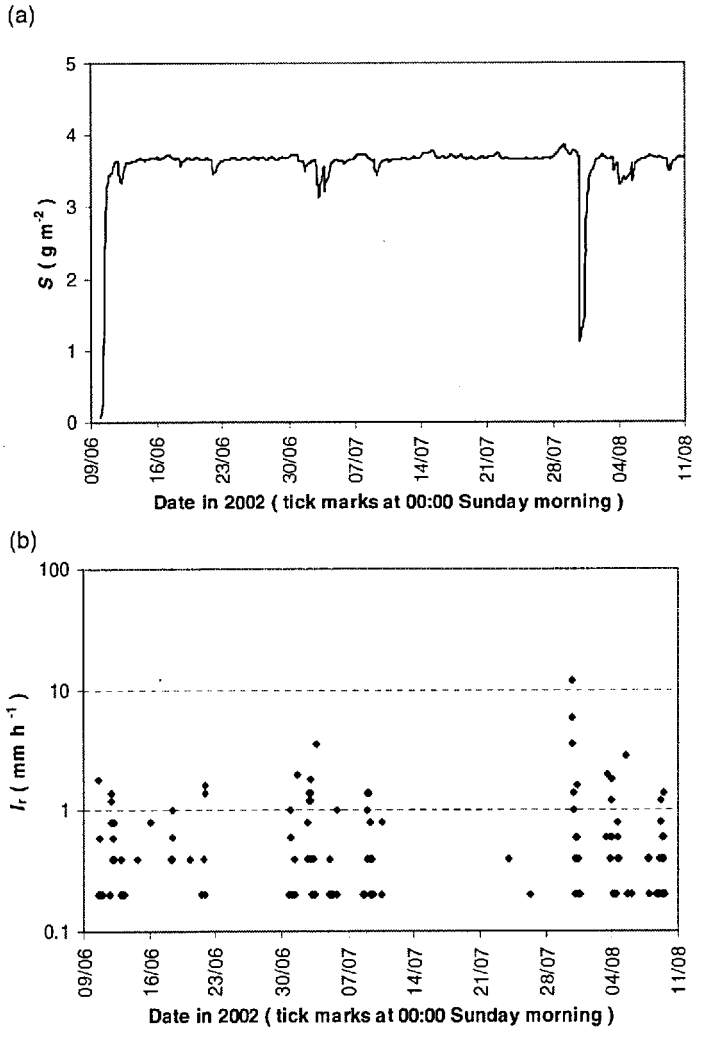


Fig. 6.12: (a) Road surface loading; and (b) Rainfall.

Figs. 6.13a & b show that, similar to the source contributions at Gloucester Place, road abrasion and material moved in from the upstream of the road segment are the two dominant sources of the surface material loading at Manor Road. Dry deposition contributes the least among all the sources to the road surface loading (Fig. 6.13c). On average F_a, F_x, F_t and F_d contribute 53%, 42%, 4.9% and 0.65% respectively to the surface loading (Fig. 6.13d).

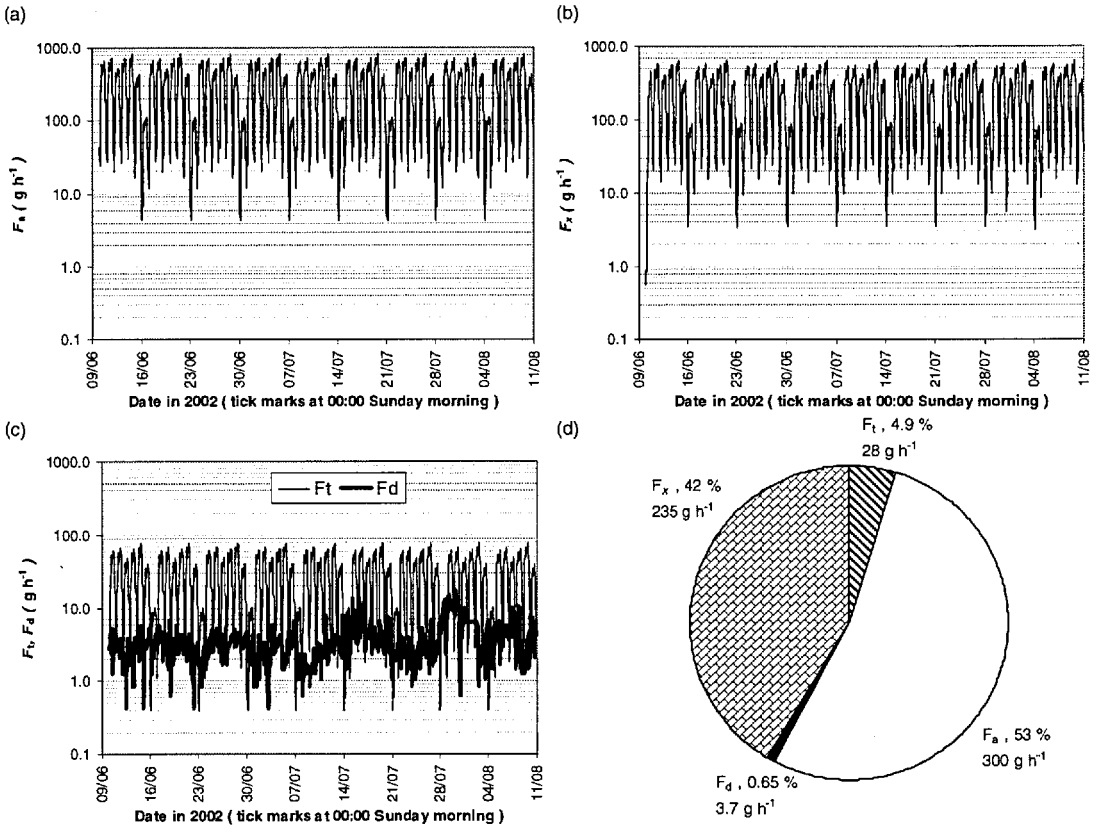


Fig. 6.13: Source apportionment of road surface loading: (a) Time series of road abrasion (F_a); (b) Time series of material input from the upstream road segment (F_x); (c) Time series of tyre wear (F_t) and dry deposition (F_d); and (d) Average contribution of these sources.

6.3.1.3 Material removal fluxes

On both road segments the depleting fluxes to air and along the road are higher than the flux across the road by a factor of 5 and 6 respectively (Fig. 6.14). The relative magnitude of the fluxes are determined by f_x, f_y and f_z. The peaks of the fluxes refer to daily peak traffic flow during the daytime and lowest points of fluxes refer to the night time reduced traffic flow. On early mornings of Sundays the traffic flow is the least on Manor Road and therefore the least values for fluxes are seen at 00:00 hours on Sundays.

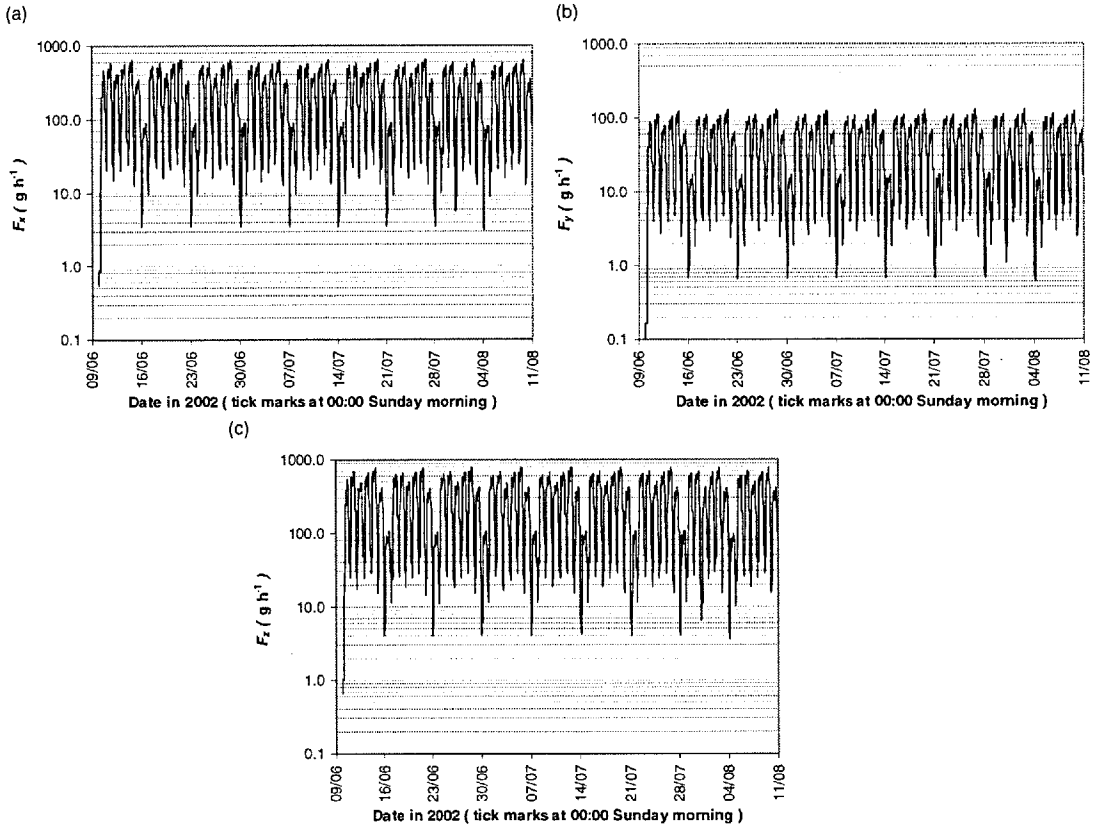


Fig. 6.14: (a) Along-street flux; (b) Across-street flux; and (c) Flux to air.

6.3.1.4 PM_{10}

Variability in traffic flow rate and atmospheric dispersion cause the modelled contribution of resuspension (C_z), exhaust emissions (C_e) and brake wear emissions (C_{br}) to PM_{10} in roadside air to vary approximately by a factor of 10^4, 10^3 and 10^2 respectively. Background PM_{10} (C_b) also varies by a factor of 10. On a number of occasions, the PM_{10} contribution from resuspension is higher than background PM_{10} (Fig. 6.15).

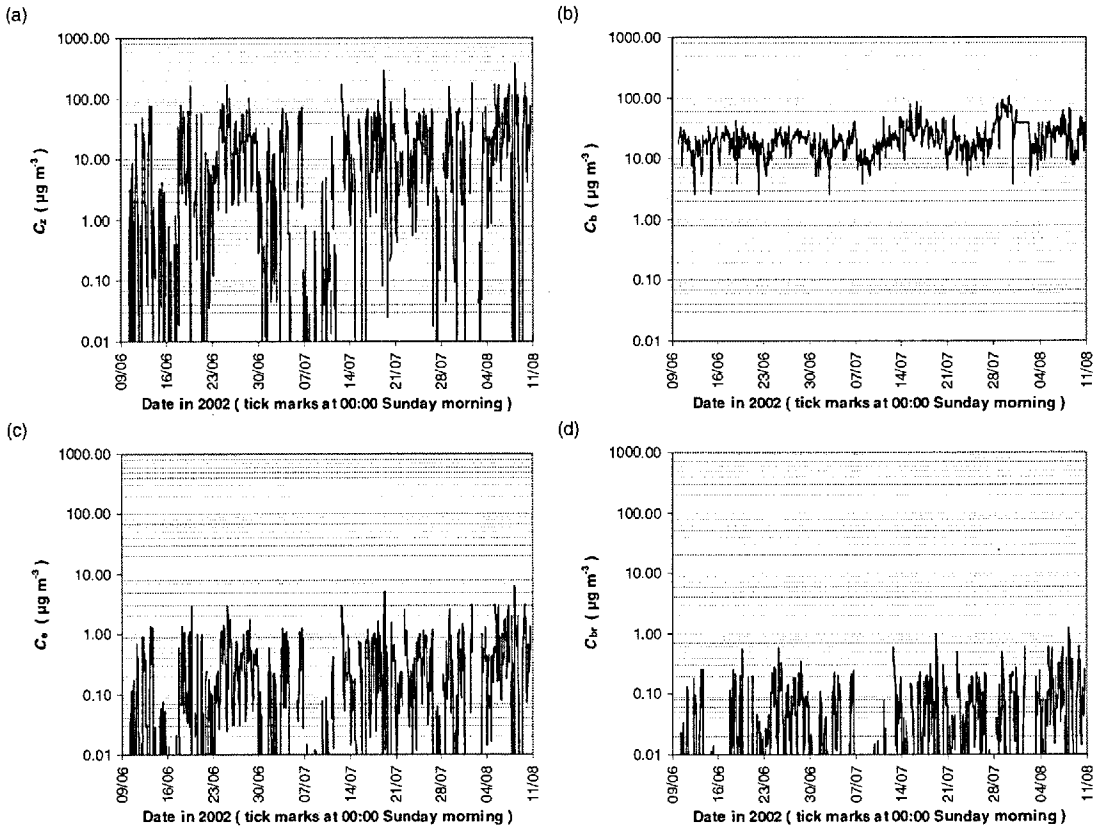


Fig. 6.15: Time series of PM_{10} contribution from different sources: (a) Resuspension; (b) Background; (c) Exhaust emissions; and (d) Brake wear emissions.

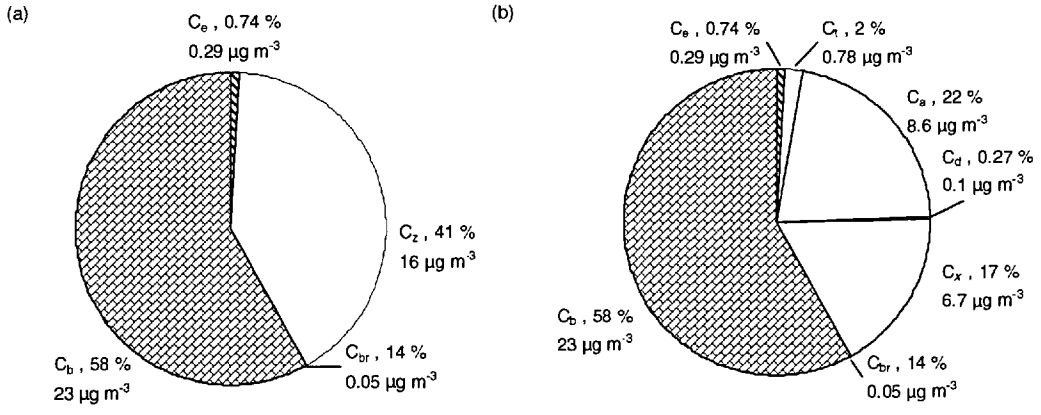


Fig. 6.16: Average PM_{10} contribution from different sources: (a) Average contribution of background (C_b), exhaust (C_e), brake wear (C_{br}) and resuspension (C_z); and (b) Average contribution of background, exhaust, brake wear, tyre wear (C_t), abrasion (C_a), dry deposition (C_d) and material input from the upstream road segment (C_x).

On average, in the absence of material input from EGTS, airborne PM_{10} at Manor

Road comprise 58% of background (C_b), 41% of resuspension (C_z), 0.74% of exhaust emission (C_e) and 0.14% of brake wear emissions (C_{br}) (Fig. 6.16a). Resuspension alone raises the average Manor Road airborne PM_{10} level by $16\,\mu g\ m^{-3}$. This is approximately 50 times higher than the the contribution from exhaust emission and brake wear emission taken together. The source apportionment of resuspension is presented in Fig. 6.16b.

6.3.2 Model output with material input from EGTS

6.3.2.1 Comparison of model prediction with measurement

Because no measurement of material input rate from EGTS (F_w) is available, it is appropriate first to find the optimum F_w value with which the model prediction closely agrees with the measurement. Calibration of the model indicated that, with the parameter values f_x, f_y and f_z obtained from Gloucester Place experiment, it is impossible to find a F_w value that would result in the model prediction agreeing to both the airborne PM_{10} and surface loading measurements at Manor Road.

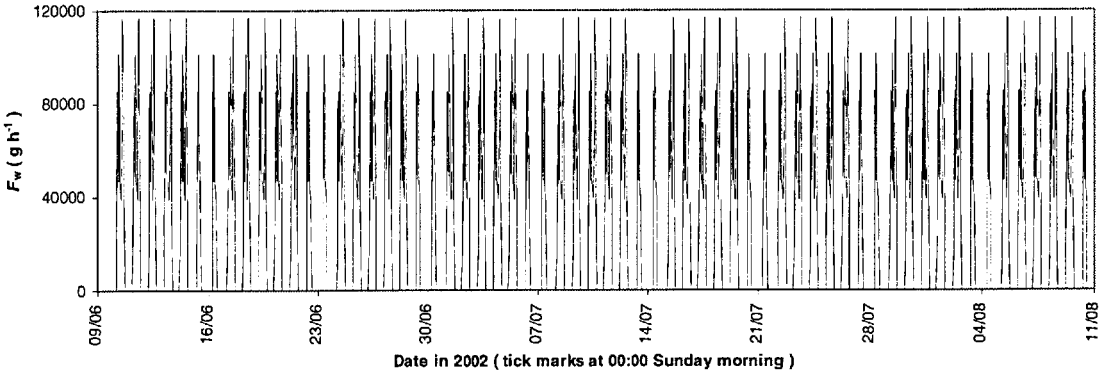


Fig. 6.17: Calibrated time-series of material input rate from EGTS.

F_w is first calibrated to arrive at a surface loading value that agrees with the surface loading measurements obtained at Manor Road. With a F_w range shown in Fig. 6.17, the modelled surface loading replicates the measurement within a factor of 2 (Figs. 6.18b & c). However the modelled PM_{10} is a order of magnitude higher than the measurement (Fig. 6.18a).

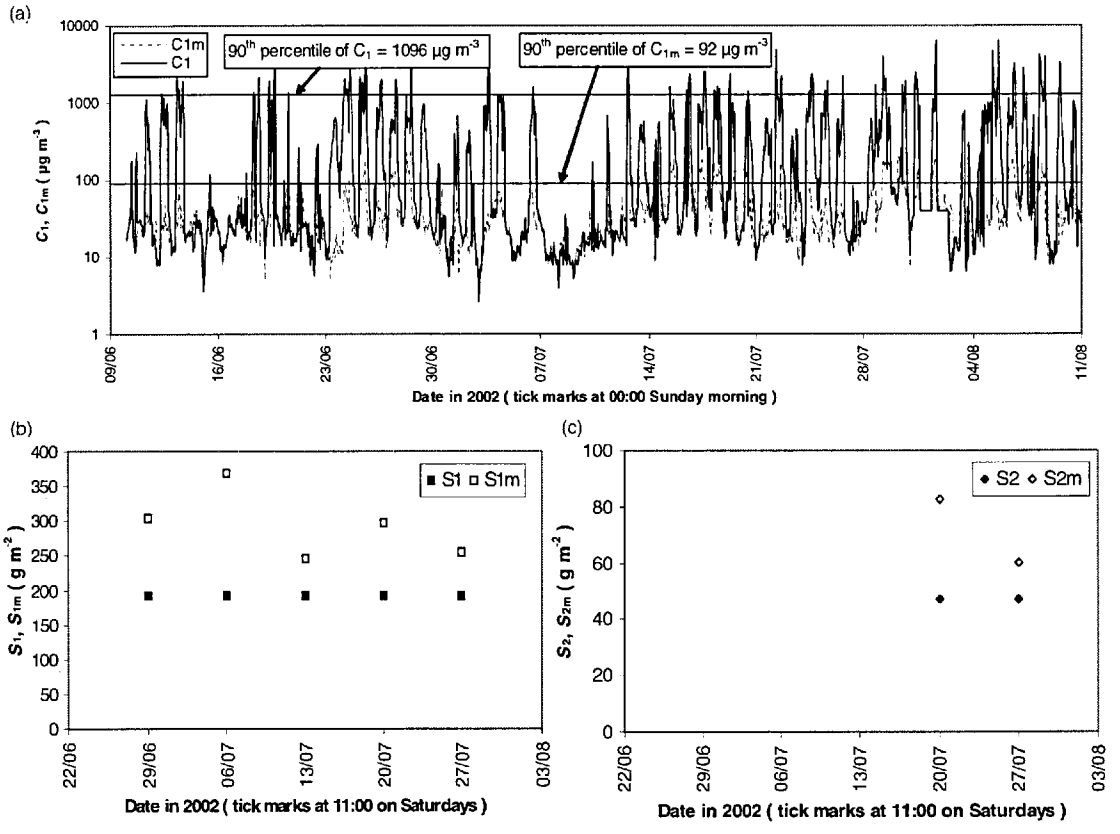


Fig. 6.18: Performance of the model with material input from EGTS: (a) PM_{10} at the industrial end; (b) Surface loading at the industrial end; and (c) Surface loading at the residential end.

Therefore the objective of the calibration procedure was narrowed down to estimate F_w at which the model output agrees with the measured PM_{10} at Manor Road regardless of surface loading measurement. With this evaluation criterion, the calibration process indicated that for the time series of F_w presented in Fig. 6.19, the model prediction agrees with the PM_{10} measurement (Fig. 6.20a).

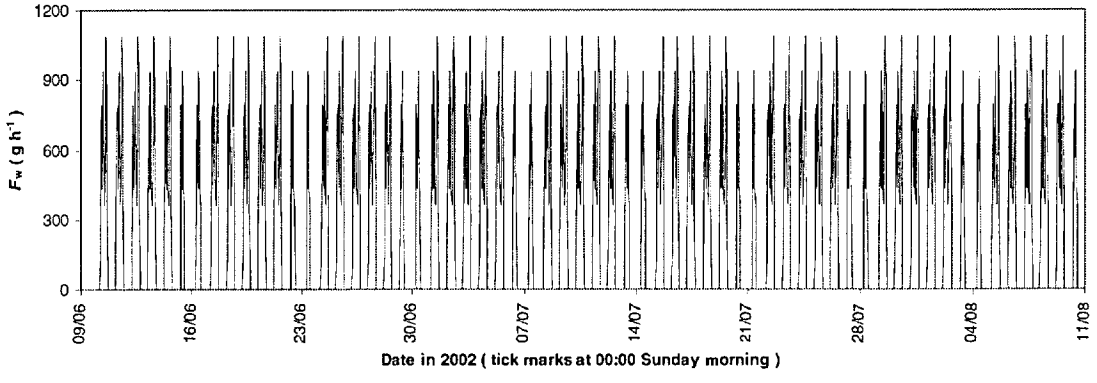


Fig. 6.19: Calibrated time-series of material input rate from EGTS.

The model estimate shows that highest rate at which material is carried over from EGTS to Manor Road is over 1 kg h^{-1}. During early morning and evening when the EGTS is closed, there is no material input from the EGTS (Fig. 6.19). The material inflow rate is proportional to the traffic flow rate from EGTS to Manor Road because of the way it is parameterised in the model (Section 5.6.2.5).

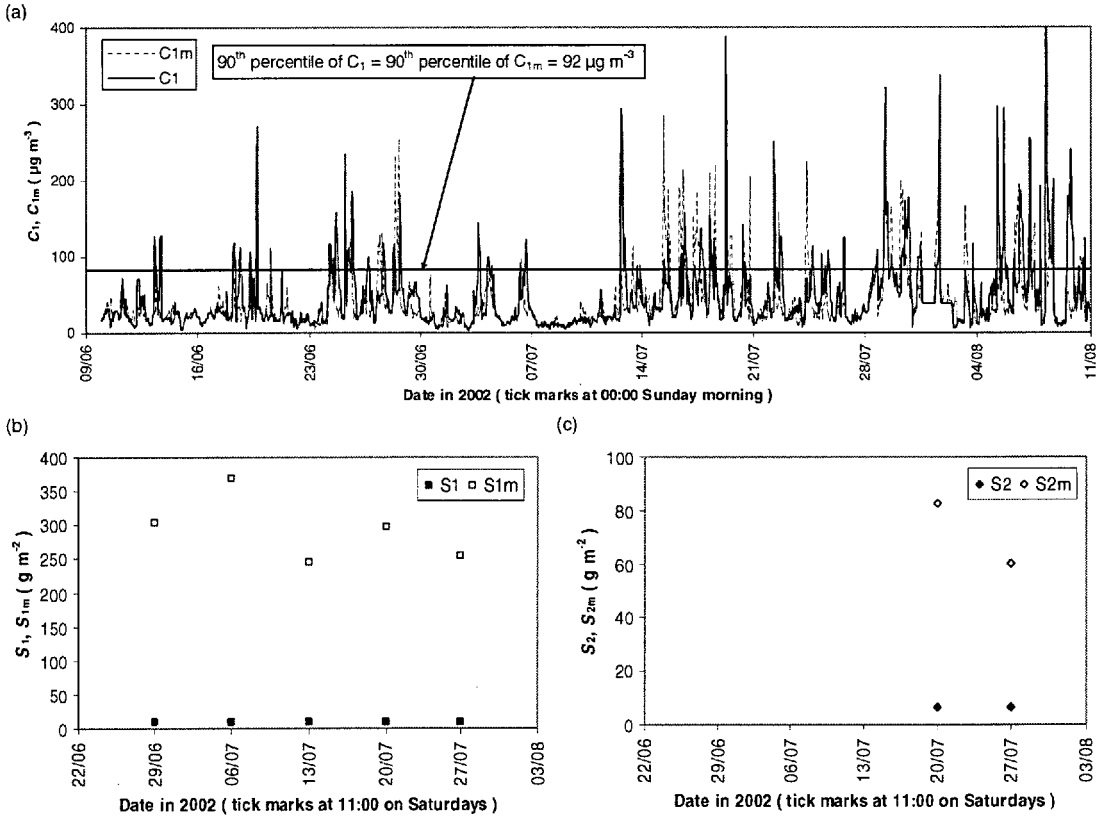


Fig. 6.20: Performance of the model with material input from EGTS: (a) PM_{10} at the industrial end; (b) Surface loading at the industrial end; and (c) Surface loading at the residential end.

However, as mentioned earlier, within the parameter values obtained from Gloucester Place experiment, the model could not reproduce the surface loading measurements (Figs. 6.20b & c). Possible reasons for this are discussed in Section 7.2.7

6.3.2.2 Road surface loading

The average surface loadings on industrial and residential end of Manor Road are 8 g m^{-2} and 6 g m^{-2} respectively (Fig. 6.21), twice and one and half times of the respective surface loadings when there was no material input from EGTS (Section 6.3.1.2).

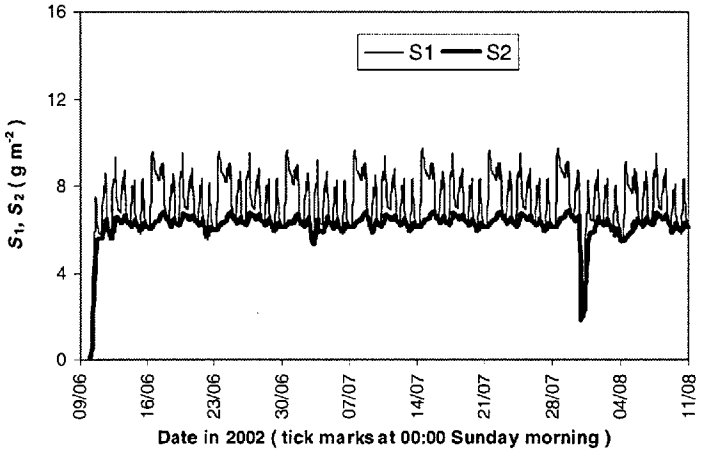


Fig. 6.21: Road surface loading.

Applying the average relationship between silt loading and surface loading presented in Table 4.2 to the above estimate, the silt loading at the industrial end and residential end become 2.04 g m^{-2} and 0.96 g m^{-2} respectively. While the silt loading estimate for the residential end is within the EPA reported silt loading range[5] for high-ADT[6] US roads, the estimate for the industrial end is twice the upper limit of this range. The reduced surface loading at the residential end is the result of loss of material across the road and to air during the time material moved from the industrial end to the residential end. The fluctuations of surface loading due to traffic flow variability were able to obscure the impact of low-intensity rain on surface loading.

Figs. 6.22a & c show the time series of the material input rates from the upstream road segments at the industrial end and residential end of Manor Road respectively. Other fluxes that contribute to the surface loading are presented earlier (Figs. 6.13a & c & 6.19). All the fluxes are proportional to the traffic flow rate. At the industrial end, input to road surface loading comprises of 214 g h^{-1} from the upstream road segment (F_{x0}) and 270 g h^{-1} from EGTS (F_w), making a total of 59% of material that is not generated on the road segment itself (Fig. 6.22b). This reduces to 342 g h^{-1} at the residential end (F_{x1}), making 51% of the surface loading because of loss of material to air and across the road (Fig. 6.22d).

[5]The EPA reports the silt loading range of 0.01 g m^{-2} to 1.02 g m^{-2} for high-ADT roads (Section 2.12.1).

[6]High-ADT roads refer to roads with at least 5000 vehicles per day (USEPA, 1997).

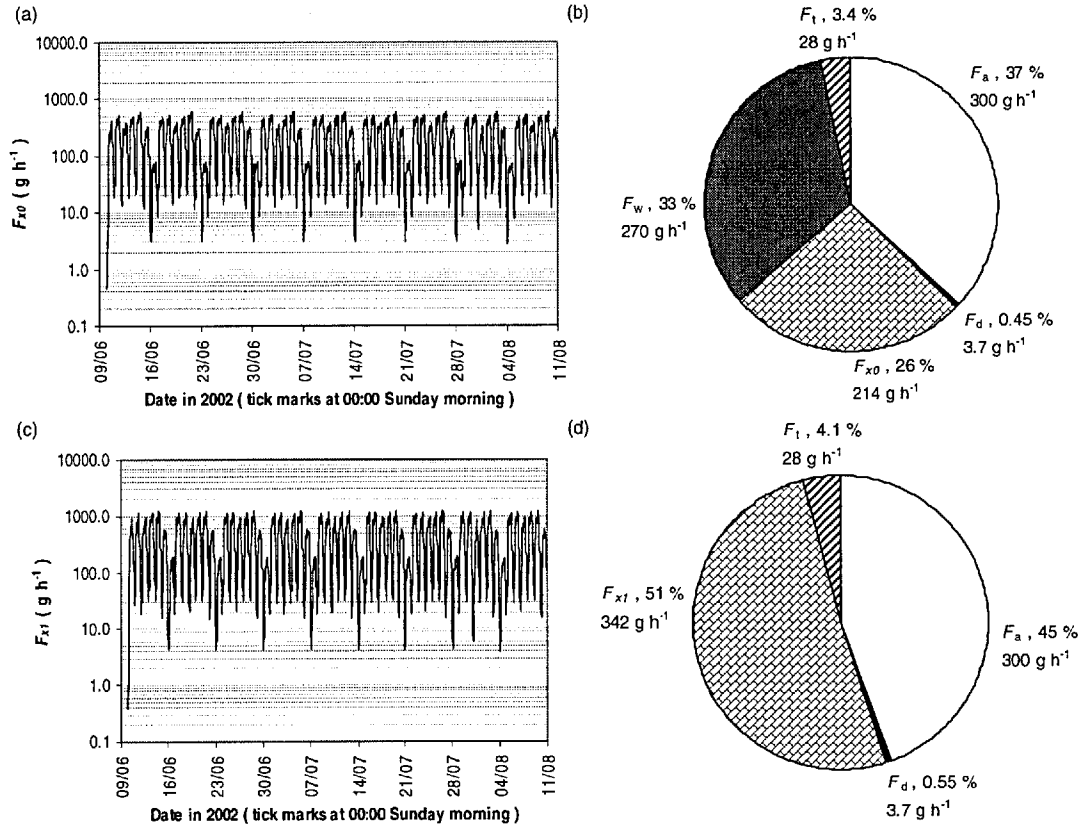


Fig. 6.22: Source apportionment of road surface loading: (a) Rate of material input from the upstream road segment at the industrial end; (b) Average contribution of abrasion (F_a), tyre wear (F_t), dry deposition (F_d), material input from the upstream road segment (F_{x0}) and material input from EGTS at the industrial end (F_w); (c) Rate of material input from the upstream road segment at the residential end; and (d) Average contribution of abrasion, tyre wear, dry deposition and material input from the upstream road segment (F_{x1}) at the residential end.

6.3.2.3 Material removal fluxes

On both road segments the depleting fluxes to air and along the road are marginally higher than the flux across the road (Fig. 6.23). This is in contrast to the factor of 5 to 6 difference between these fluxes when there was no material input from EGTS. This is because of the use of the lower f_x, than the earlier one, that is used to achieve the best model prediction.

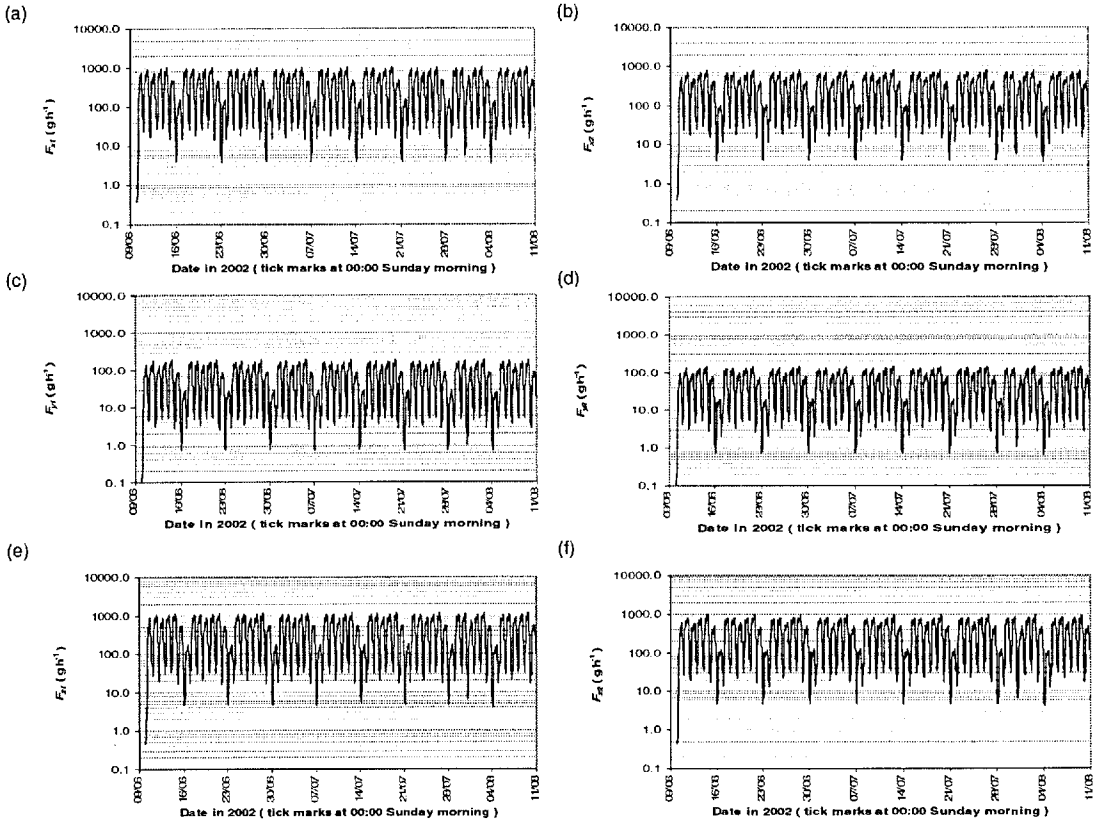


Fig. 6.23: (a) Along-street flux at the industrial end; (b) Along-street flux at the residential end; (c) Across-street flux at the industrial end; (d) Across-street flux at the residential end; (e) Flux to air at the industrial end; and (f) Flux to air at the residential end.

6.3.2.4 PM_{10}

Fig. 6.24 shows that resuspension (C_{z1} and C_{z2}), exhaust emissions (C_e) and brake wear emissions (C_{br}) vary by a factor of 10^4, 10^3 and 10^2 respectively because of variations in traffic and dispersion condition. Some of the hourly values of C_{z1} and C_{z2} are larger than background level (C_b) by a factor of 10 to 100.

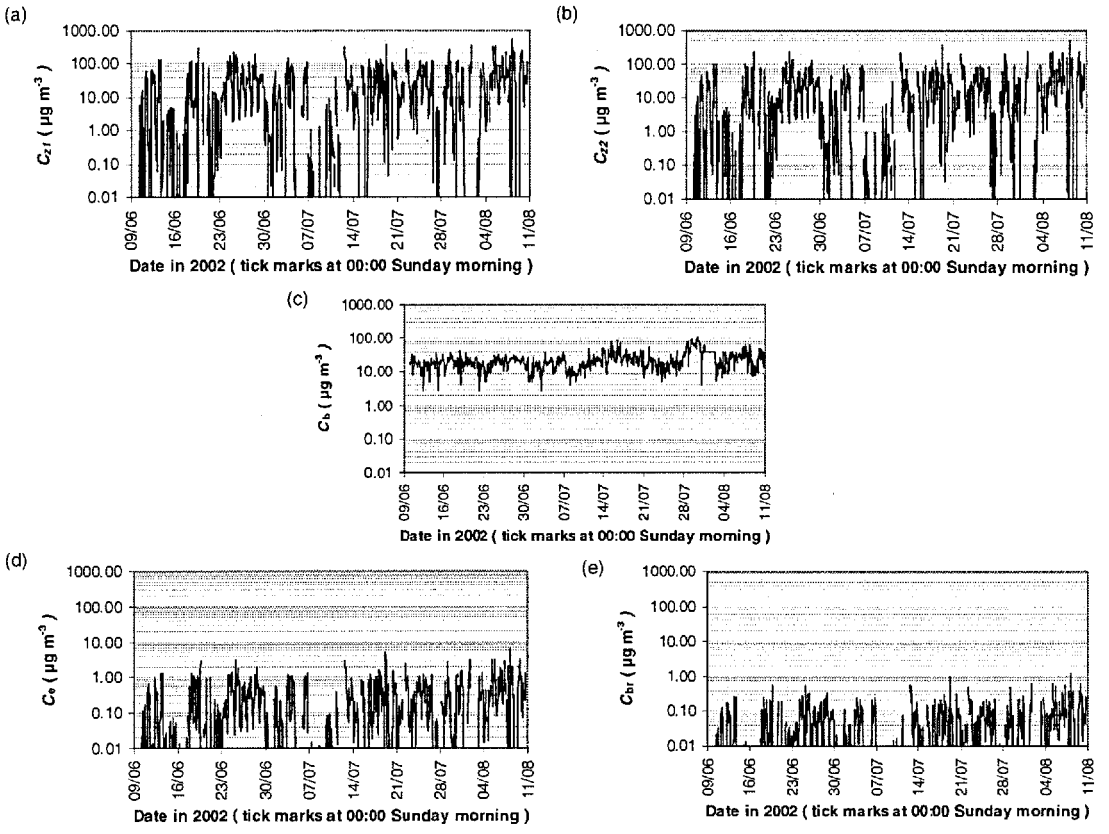


Fig. 6.24: Source apportionment of PM_{10}: (a) Resuspension at the industrial end; (b) Resuspension at the residential end; (c) Background; (d) Exhaust emissions; and (e) Brake wear emission.

Removal of material to kerbs and to air during the time the material moved from the industrial end to the residential end because of along-street movement has caused the lower contribution of resuspension at the residential end ($19\,\mu$g m^{-3}) than that at the industrial end ($22\,\mu$g m^{-3}) (Figs. 6.25a & b). Resuspension and background are the dominant sources at the industrial end (Fig. 6.25a). The amount and proportion of resuspension due to material input at the residential end (C_{x1}) is higher than the that at the industrial end (C_{x0}) because it now includes resuspension from the additional material added from the EGTS (Figs. 6.25c & d).

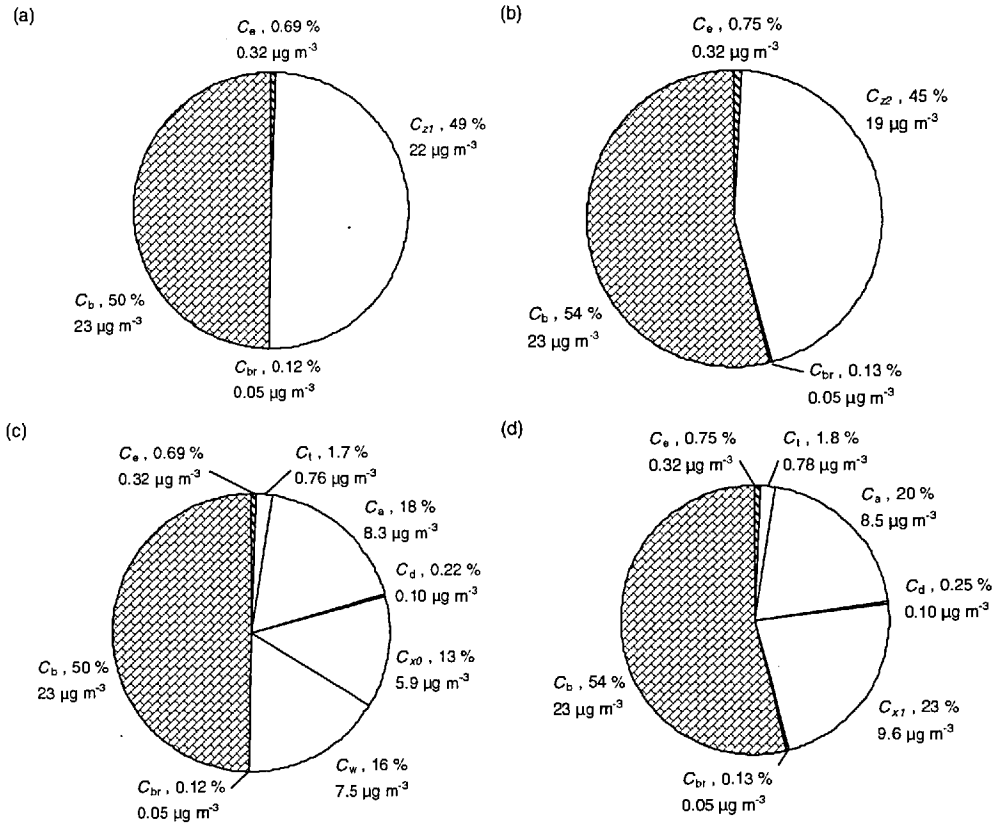


Fig. 6.25: Average PM_{10} contribution from different sources: (a) Average contribution of background (C_b), exhaust (C_e), brake wear (C_{br}) and resuspension (C_{z1}) at the industrial end; (b) Average contribution of background, exhaust, brake wear and resuspension (C_{z2}) at the residential end; (c) Average contribution of background, exhaust, brake wear, tyre wear, abrasion (C_a), dry deposition (C_d), material input from EGTS (C_w) and material input from the upstream road segment (C_{x0}) at the industrial end; and (d) Average contribution of background, exhaust, brake wear, tyre wear, abrasion, dry deposition and material input from the upstream road segment (C_{x1})at the residential end.

6.4 Summary

The model results show that the model formulation is able to explain the measurement except for surface loading at Manor Road. Possible reasons for this are discussed in Section 7.2.7. Application of the model to Gloucester Place resulted in an evaluation of the wide range of abrasion rate reported in literature (Section 6.2.1.1). Material carried over from the upstream road segment is found to be a major source of the surface loading (Sections 6.2.1.2, 6.3.1.2 & 6.3.2.2). Comparison of PM_{10} model prediction

with measurement indicated the non-inclusion of one or more processes in the model. Grinding of material, discussed in Section 7.2.6, appears to be this process that controls the reservoir of fine material on the road surface. In absence of any significant local source, resuspension is found to be a major source of airborne PM_{10}. In an urban paved road abrasion contributes the bulk of material to the road surface loading and therefore to resuspension.

Chapter 7

Discussion and conclusions

7.1 Recapitulation of thesis content

Adverse impacts of PM_{10} on human health and therefore the need to control it, were discussed in Section 2.1. The increasing importance of paved road surfaces as the source of airborne PM_{10}, especially in urban areas, was recognised and the evidence of traffic activity being the most important agent of PM_{10} emission was presented in Sections 2.2 & 2.3. The growing importance of traffic-induced resuspension of road dust was discussed in Sections 2.4 & 2.5. For proper understanding of the emissions from paved roads and for a reliable estimate of the magnitude of such emissions, the need to study processes behind resuspension was highlighted in Section 2.6. Lack of existing modelling methods to consider such processes was discussed in Sections 2.11 & 2.12.

To improve our understanding of the processes leading to entrainment of particles from the road surface into roadside air, a mass-balance model that explains the material movement on urban paved roads has been developed (Chapter 5). The model includes the important processes behind paved road emissions and therefore places emissions to the atmosphere in the context of a budget of sources and sinks. The sources universal to a paved road were identified from literature review. The estimates for parameter values representing these source terms were also obtained from literature review (Sections 5.6.1 & 5.6.2). From the wide range of parameter values,

an appropriate parameter value for road abrasion, the universal source having significant contribution to road surface loading, has been obtained through model calibration (Section 6.2.1.1). Estimates for the local sources have been obtained either from measurement (Section 5.6.2.4) or through model calibration (Section 5.6.2.5). Removal of material by rain has been parameterised on the basis of earlier studies (Section 5.6.3.4). The traffic-induced material removal processes have been conceived on the basis of our observation of the material movement on road surfaces. Experiment at Gloucester Place provided parameter values for these processes (Sections 3.2 & 4.3). These parameter estimates were then used to assess the scientific correctness and to evaluate the model (Chapter 6).

7.2 Enhanced understanding of particle movement on paved roads

The main achievement of this research is the enhanced understanding of particle movement on urban paved roads. It consists of quantification of the effect of the important particle movement processes on urban paved roads. All these processes are then mathematically formulated to be the components of a model. Some additional processes are identified during the research and the possible implications of these processes on particle movement on road surfaces are discussed.

7.2.1 Resuspension of material

The rate of suspension of particulate matter into the roadside air was estimated from the roadside concentrations in air, measured during the Gloucester Place experiment. Point source tracer release measurements and emissions factor ratio methods produce broadly consistent emissions estimates, giving grounds for confidence that the measurement, although approximate, is robust. When a vehicle passes over the gritted surface, the particulate matter level recorded by the Grimm monitor increases by a factor between 5 and 30 for different particle sizes, indicating the process of resus-

pension. Immediately after gritting, very fast movement of material along the road is observed. The rate of movement slows down with time as the material available on the road surface decreases. Grit thrown to the edge of the road indicates a process of across-street movement of material. Coarse material becomes resuspended more easily than finer material. This agrees with observation from earlier studies (Abu-Allaban et al., 2003a,b; AQEG, 2004; Barrowcliffe et al., 2002; Corn and Stein, 1965; Garland, 1983; Harrison et al., 1997a; Lin et al., 1999; Manoli et al., 2002; Nicholson and Branson, 1990; Nicholson, 1993). Although strong dependence of resuspension on particle size has largely been reported from studies involving resuspension due to wind (Corn and Stein, 1965; Garland, 1983; Harrison et al., 1997a), Nicholson and Branson (1990) have observed from field investigation that they are equally applicable to traffic-induced resuspension. Hinds (1982) attributes this to drag force increasing faster than adhesive force with increasing particle diameter and Corn and Stein (1965) suggest protrusion of large particles further into the turbulent air stream as the reason. The exponential decay rates of coarse fractions with time are in contrast to the inverse relationship between resuspension and time proposed from theoretical (Reeks et al., 1988) as well as experimental works (Garland, 1979; Nicholson, 1993) on resuspension by wind. Traffic-induced turbulence is generally stronger than wind-induced turbulence in an urban street, except at times when strong prevailing and in-street air flows are present. Therefore it is not surprising that the exponential decay rates of material, consistent with observations reported from some earlier works on wind-induced resuspension (Anspaugh et al., 1975; Linsey, 1978), are seen in this case. Fine fractions up to 2 μm remained on the road, and therefore available for resuspension, for a longer time, as indicated by Grimm data that showed their level above the local background until the end of the experiment. Similar features can be observed with ordinary winter gritting, but the importance of this controlled experiment is the short length of road to which gritting was applied, which allows the time-constants of decay of grit concentration to be used to parameterise the movement of the particulate matter.

7.2.1.1 Resuspension rate and resuspension factor

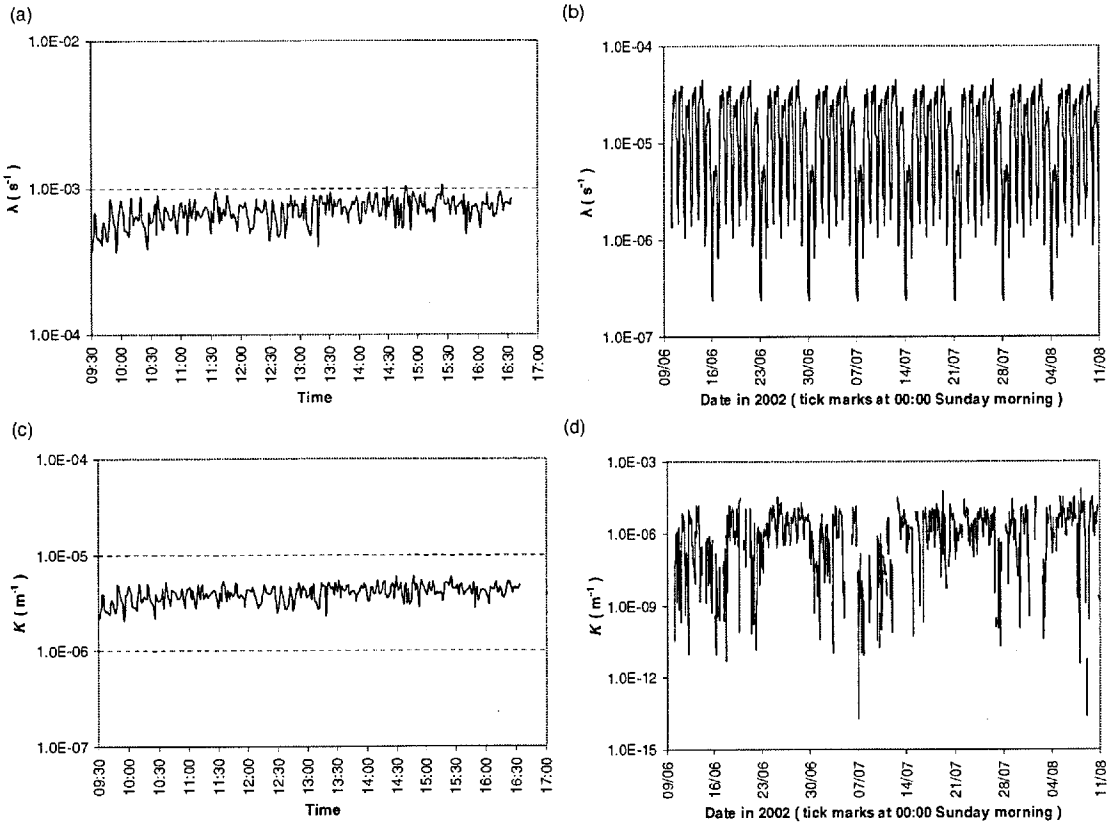


Fig. 7.1: (a) Resuspension rate at Gloucester Place; (b) Resuspension rate at Manor Road ; (c) Resuspension factor at Gloucester Place ; and (d) Resuspension factor at Manor Road.

The resuspension rates[1] (λ) vary within the range of $10^{-4}\,s^{-1}$ to $10^{-3}\,s^{-1}$ at Gloucester Place and $10^{-7}\,s^{-1}$ to $10^{-5}\,s^{-1}$ at Manor Road (Figs. 7.1a & b). For a particular resuspension fraction, traffic causes change in surface loading (S) and resuspension flux (R) and therefore λ follows a pattern similar to that of traffic. For the first few hours on each Sunday, traffic on Manor Road is very low, i.e., about a factor of 10 less than the average daily traffic. This has resulted in low λ values on Sunday mornings at Manor Road. At Gloucester Place traffic flow is fairly uniform during the duration of the experiment and therefore no large variation of λ is observed. The resuspension factors (K) lie in the range of $10^{-6}\,m^{-1}$ at Gloucester Place and within

[1]Resuspension rate, Λ, is defined as the fraction of surface deposit removed per unit time and Resuspension factor, K, is defined as the ratio of airborne concentration of the deposited material and surface loading of the material (Section 2.8).

the range of $10^{-14}\,m^{-1}$ to $10^{-5}\,m^{-1}$ at Manor Road (Figs. 7.1c & d). Use of a single 12-minute averaged dispersion coefficient from the tracer study to calculate roadside concentration of PM_{10} (C) from road surface emission at Gloucester Place has resulted in less variation in K. The wide range for K at Manor Road is obtained because of use of hourly dispersion coefficient for a period of 9 weeks, which varies by a factor of 10^9. No earlier study that includes estimates for λ and K for traffic-induced resuspension was found from the literature. However, excluding a few very low values at Manor Road, the above estimates for K are comparable with the range of $10^{-10}\,m^{-1}$ to $10^{-2}\,m^{-1}$, reported earlier for wind-induced resuspension with mechanical turbulence (Sehmel, 1980b, 1984).

7.2.2 Along-street and Across-street movement of material

Based on the data obtained from the Gloucester Place experiment, a quantitative description of the emission processes was carried out to calculate the speed with which particulate matter on a road arrives at some distance downstream of the initially gritted patch, to estimate across-street speed that causes movement of particulate matter to kerbs and thus the significance of loss of particulate matter along and across the road with respect to resuspension to air (Section 4.3). The rate of movement of material along the road in the direction of traffic flow was estimated by observing the difference in arrival time of elevated concentrations of micrometer-sized particles in roadside air adjacent to and a short distance downstream of a section of three-lane, one-way building-lined street onto which rock salt was applied. Initial movement, dominated by particles several micrometres in size and above, was $0.11\,m\,s^{-1}$. After several minutes, slower movement of finer particulate matter was observed, at a speed of $0.06\,m\,s^{-1}$. Vehicle speeds were fairly typical for Central London, moving at 15 miles per hour to 30 miles per hour during the green phase of the traffic lights, with queues forming when the lights were red. Assuming $5.6\,m\,s^{-1}$ (equivalent to $20\,km\,h^{-1}$) as the average speed of vehicles along the Gloucester Place, the average speed of the material movement along the road is about 1.1 % of the vehicle speed and 3.8 % of the average street-level

wind speed. These speeds were observed at an average traffic flow of 4800 vehicles per hour (1600 per lane), and it is reasonable to assume the speed would be proportional to traffic flow on a similar road with fewer or a larger number of vehicles per lane.

Assuming the rates of movement of material along and across the road are proportional to the fraction of material removed in these directions, the maximum and minimum across-street particle removal speeds are estimated to be $0.014\,\mathrm{m\ s^{-1}}$ and $0.007\,\mathrm{m\ s^{-1}}$ respectively. The direction of the across-street movement is at an angle of approximately 8° to the direction of vehicle movement. This rate of lateral movement is consistent with the fraction of the initially deposited rock salt found collected against the kerb after several hours when most of the material had been removed from the main part of the carriageway. This rate also is likely to be proportional to the number of vehicles using the road.

7.2.3 Quantification of material removal processes

The resuspension fraction, the removal fraction along the road and the removal fraction across the road were in the range of 2.1×10^{-4} per vehicle to 5.4×10^{-4} per vehicle, 2.6×10^{-4} per vehicle to 7.1×10^{-4} per vehicle and 3.3×10^{-5} per vehicle to 9.1×10^{-5} per vehicle respectively, making the total removal fraction between 5.0×10^{-4} per vehicle and 1.3×10^{-3} per vehicle (Section 4.3.4). These estimates agree with the findings of the earlier studies that had followed almost identical technique as ours and reported resuspension fractions of the order of 10^{-5} to 10^{-2} per vehicle on the same day of tracer application (Sehmel, 1973, 1976b) (Section 2.10). The most important new information provided by this experiment however, is that while Sehmel (1973, 1976b) attributed the entire loss to resuspension, this study identifies the total loss as a sum of losses along and across the road and emission to air, and suggests that the flux of material to air is approximately 40% of the total outgoing fluxes in all directions, 70% of the sum of the fluxes along and across the road and is a factor of 10 higher than the amount lost across the road to the kerbs. Material lost along the road is more than the combined loss due to resuspension and material removed across

the road. Therefore this study gives an estimate for the first time of the contributions of different processes that cause road dust redistribution due to traffic on an urban street. The fluxes to air in Section 4.3.3 refer to PM_{10}. Particles larger than PM_{10} may also become suspended into the air, but these then redeposit to the road surface. These have therefore been included in along-street and across-street movement, but not in PM_{10} emissions to air. Simple formulae summarising the relative contribution of different material removal processes are given in Equations 4.14, 4.15, 4.16, 4.17, 4.18 and 4.19. The quantification of material removal processes is used to develop a formula that describes the likely spatial variation of emissions from a point source of particulate matter on a paved road surface (Equation 4.20). Such a formula for material dispersion on paved roads is the first of its kind.

7.2.4 Development of a model

A dynamic model that encompasses all the important particle movement processes is successfully developed and evaluated, the requirement of which is emphasised by several researchers (Countess et al., 2001; Kuhns et al., 2001a; Nicholson, 2001; Venkatram, 2001). For the first time all the important sources of road surface loading on a paved road (road abrasion, tyre wear and dry deposition) are combined together into this model and their widely varying estimates from different studies are evaluated. In the absence of any major local source, abrasion is identified to be the major process that replenishes the surface loading. Similar estimation is presented by earlier researchers (Gámez et al., 2001b). The significant contribution of material from an upstream road segment to the surface loading suggests that silt loading at the point of emission is not only dependent on the processes happening at the road segment itself, but also it depends on sources and processes that contribute to the surface loading at other sections. This is particularly significant in urban areas where major roadside activities such as pavement repair, construction/demolition of roadside buildings, etc., at one location could be the cause of excess airborne particulate matter concentration at a new location. The information on this type of cause-effect relationship of the particulate

pollution in urban areas cannot be obtained from the empirical models. Combined with sources that directly emit to air (brake wear emission and exhaust emission), this gives a detailed synthesis of sources of particulate matter emission. Mathematical formulation of each source term gives the underlying processes and the methods to calculate each source contribution. Therefore the need to calculate separately the exhaust and non-exhaust emissions, expressed by many researchers (Düring et al., 2002a; Schulze, 2001), is fulfilled.

Mathematical formulation of particle removal processes, quantification of these processes and testing their validity are entirely new contribution of this research. The formulation itself is independent of any measurement. The removal processes are quantified from a field experiment and therefore it adds some empiricism to the model. However inclusion of the processes imparts the scientific and mechanistic basis to the model, which was completely lacking in case of more empirical models, including the EPA AP-42 model. Testing the model for its scientific correctness and its evaluation showed the interdependency of different material movement processes and their dependence on traffic to give a consistent picture against independent measurement, suggesting interplay of these processes are able to explain the paved road emission.

7.2.5 Comparison with AP-42 emission factor model

Figs. 7.2 & 7.3 show that the emission factors predicted by the model are proportional to the surface loadings, somewhat similar to that of AP-42 algorithm where emission factor is proportional to the silt loading. However, the surface loadings and therefore the emission factors here are the result of interplay of different traffic-induced material movement processes described in Chapter 5, unlike that of AP-42 where no information is available regarding the processes that contribute to the silt loading. Surface loading changes with time depending on the interaction of processes that control the source and sink unlike a single point measurement of silt loading that is often used for derivation of AP-42 emission factor.

Silt loading at Gloucester Place is not measured. Therefore to estimate the emis-

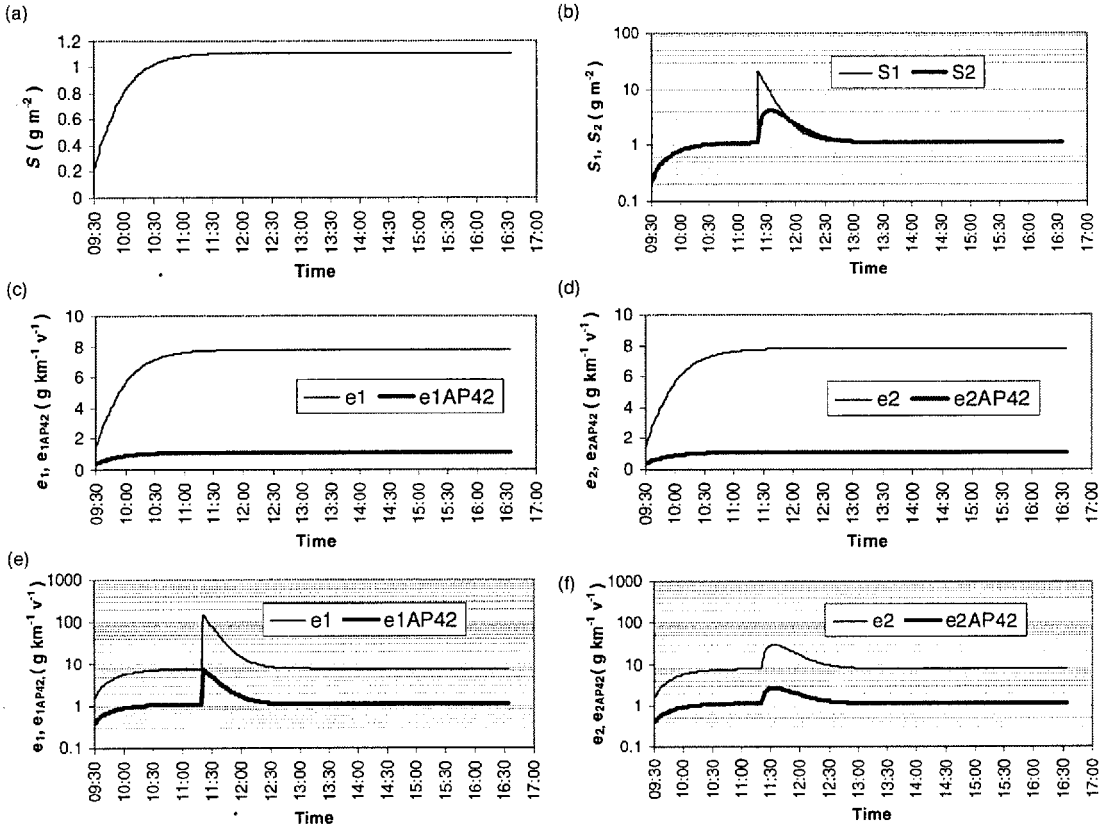


Fig. 7.2: Surface loading and emission factor at Gloucester Place: (a) Surface loadings before gritting; (b) Surface loadings after gritting (c) Emission factors at gritted segment before gritting; (d) Emission factors at ungritted segment before gritting; (e) Emission factors at gritted segment after gritting; and (f) Emission factors at ungritted segment after gritting.

sion factor at Gloucester Place by applying AP-42 algorithm, a silt fraction of 21.7%, the maximum measured value for US paved roads with significantly lower traffic flow (USEPA, 1997), is used. With this assumption, the model estimates the emission factors at Gloucester Place a factor of 7 greater than the AP-42 estimate (Fig. 7.2). This could be due to the assumed silt fraction of 21.7% from the US database being too low for the Gloucester Place that has significantly higher traffic flow rate than the roads for which the database provides the silt fractions (Section 6.2.1.2). In case of Manor Road, the silt loading measurement is available and the emission factor estimated by the model broadly agrees with the AP-42 estimation (Fig. 7.3).

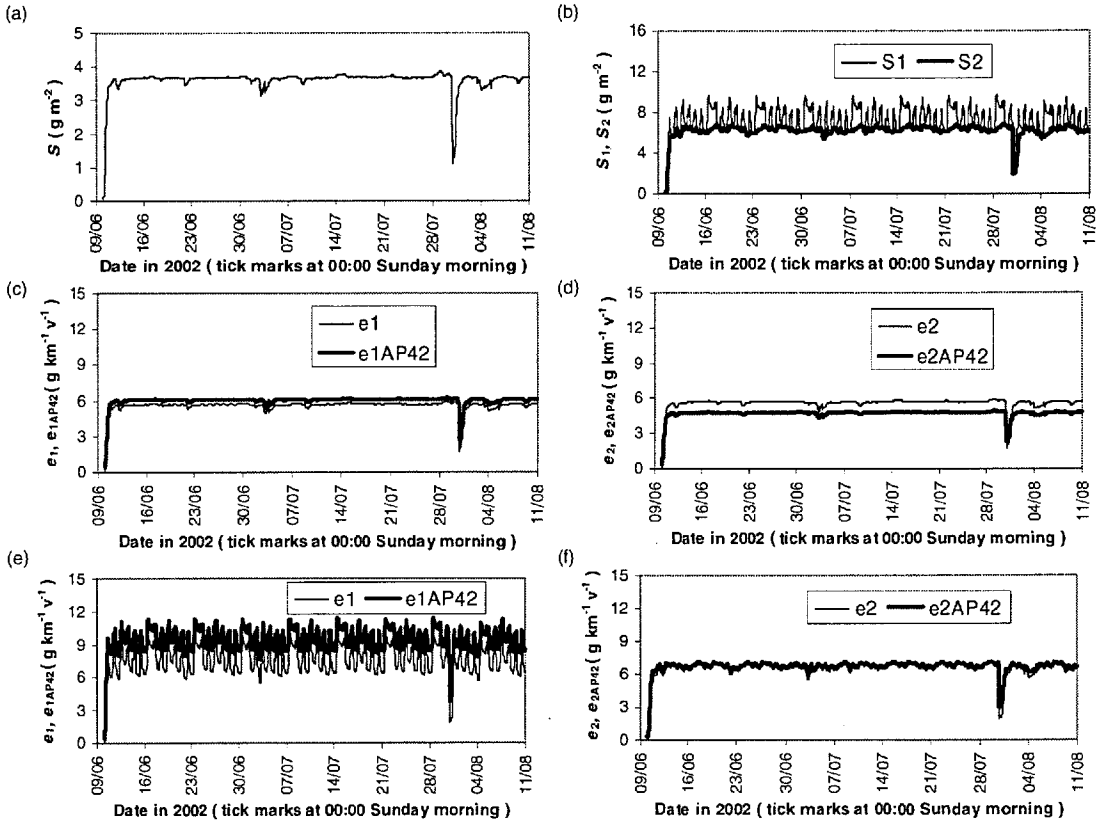


Fig. 7.3: Surface loading and emission factor at Manor Road: (a) Surface loadings without input of material from EGTS; (b) Surface loadings with input of material from EGTS (c) Emission factors at industrial end without input of material from EGTS; (d) Emission factors at residential end without input of material from EGTS; (e) Emission factors at industrial end with input of material from EGTS; and (f) Emission factors at residential end with input of material from EGTS.

As far as data requirement is concerned, AP-42 requires surface loading and silt content data to calculate silt loading, whereas the new approach only requires surface loading. This reduces the additional task of laboratory analysis to find the silt fraction, needed for estimation of AP-42 emission factors. AP-42 requires vehicle composition and weight of each vehicle class; this model requires only vehicle composition. If values of f_x, f_y and f_z are established through a series of detailed validation studies, this model will not only quantify the emissions from the paved road, but also will give an understanding of the process and therefore the mechanistic basis of the paved road emission which is absent in case of empirical models including the AP-42 emission factor model.

7.2.6 Underprediction of downstream PM_{10} at Gloucester Place

The model underprediction of PM_{10} measurement at Gloucester Place (Section 6.2.2) could be attributed to grinding of material during the along-street movement that causes increase of fine particles with time at the ungritted segment. While the grit is carried along the road to the ungritted segment, some of the bigger fractions of it are ground to fine size by the wheel pressure and by colliding with each other as they travel on-surface and above-surface under the influence of vehicle wake. With time as more and more material is removed from the road surface, the material available for grinding decreases. Further large particles are removed to kerbs through saltation and creep (Section 5.6.3). This leaves less material as well as less coarser fractions of it, both being unfavourable conditions for grinding. Thus grinding becomes less effective as time since gritting increases resulting in no appreciable difference in PM_{10} level between the gritted and ungritted segment after 2 hours of the gritting event. This process of grinding of coarse material to fine size is not included in the present model formulation because of lack of information and understanding, resulting in model underprediction immediately after gritting when the rate of material transfer along the road was high.

7.2.7 Underprediction of surface loading at Manor Road

Reasons for model underprediction of surface loading measurements at Manor Road (Section 6.3.2.1) are discussed below:

1. Road surface dust at Manor Road was collected by both dry and wet vacuuming. It is not known whether the additional material collected by wet vacuuming could contribute to PM_{10} emission because if the material is difficult to be removed by dry vacuuming, it could be equally difficult to become airborne due to traffic-induced turbulence (Section 2.12.1.1).

2. During the experimental period, rainfall at Manor Road was a frequent occurrence (Fig. 4.12d). Although the model includes the effect of rain, a simple

parameterisation is used that may fail to reflect some aspects of the wash-off. A number of uncertainties associated with the impact of rain on surface loading are yet not properly understood. For example, a German study indicated that particulate matter concentration reduction occurs with rainfall intensity as low as 0.1 mm h^{-1} and the effect of rain in lowering the emission can continue until 7 hours after the rain has stopped (Düring et al., 2002a). Myers (2001) suggested that light rain increases the availability of silt for suspension as a result of the washing of the material out of crevices of the road and therefore increases the particulate matter concentration in air. From a four-week monitoring campaign in Schildhornstraße in Berlin, Düring and Lohmeyer (2001) observed no reduction of PM_{10} emissions on days with rain (Section 5.6.3.4).

3.

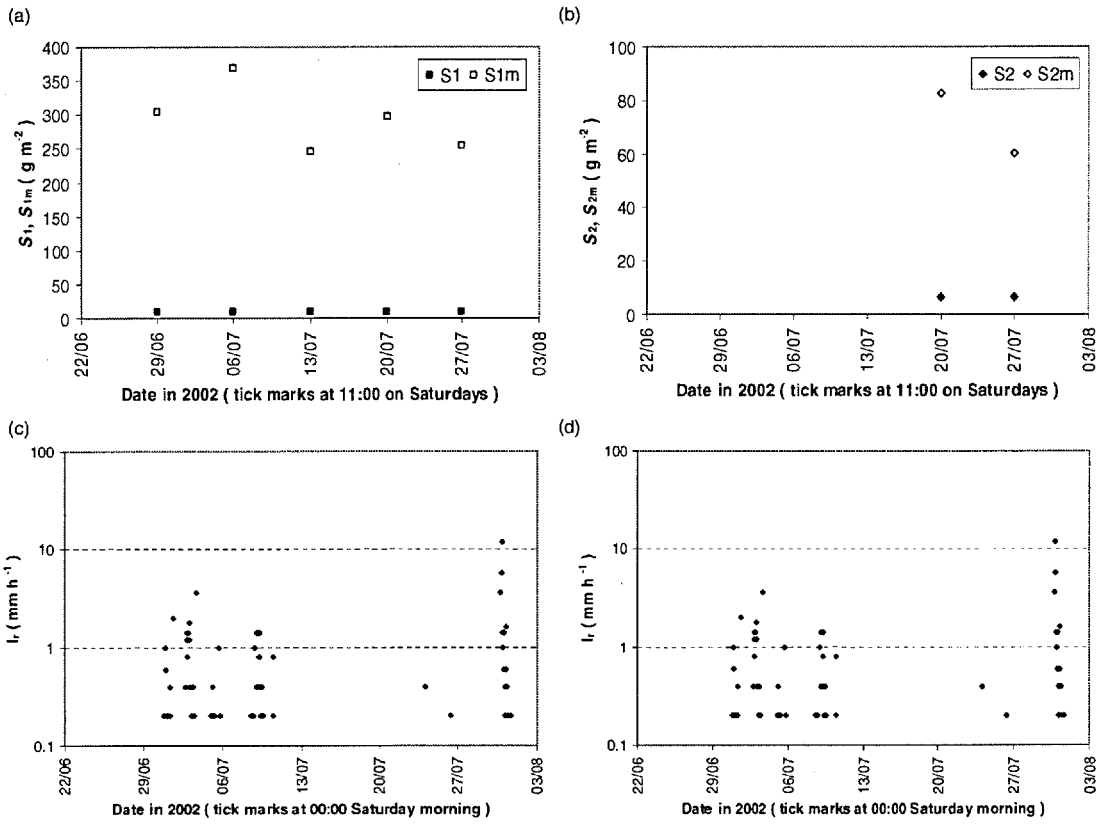


Fig. 7.4: Surface loading and rainfall: (a) Surface loading at the industrial end; (b) Surface loading at the residential end; (c) Rainfall; and (d) Rainfall.

There appears to be some inconsistency in measurements of the surface loading. Between 29 June 2002 and 6 July 2002, the rain event is quite wide-spread.

Therefore one would expect measurement of the surface loading at the industrial end on 6 July 2002 to be somewhat less than the surface loading of 29 June 2002. However the measurement shows the opposite to it (Fig. 7.4). The measurement could be correct if either the surface loading recovers very quickly to reach the normal level after the rain has stopped or the rain intensity of as high as 3.6 mm h^{-1} has no impact on surface loading. No detailed study is available to verify this. This again emphasises the previous argument.

4. Rain increases the moisture content of the surface loading material and this would have affected its removal rates in different directions. However, the removal fractions applied in the model are obtained from the Gloucester Place experiment with no rain during the experiment (Section 4.1.1). Therefore, removal fractions obtained in a dry condition may not be entirely applicable for a material removal processes during periods with intermittent rain.

5. The traffic density and fraction of HGVs are higher at Manor Road than at Gloucester Place and it may have affected the rate of replenishment and depletion of material on road surface. This is not included in this model because of lack of understanding.

6. The road condition itself may be the reason. Visible pot-holes on Manor Road could have arrested material from being removed in any of the removal processes thereby increasing the surface loading to very high level which the model is not able to predict. Although the model includes a road condition factor to take account of the road condition, this essentially determines the amount of abrasion and assumes that all of these are available for removal due to traffic movement.

7.3 Uncertainty

In broad terms, the uncertainties associated with this model can be discussed under three categories as suggested by Thiessen et al. (1999):

7.3.1 Conceptual uncertainty

Conceptual uncertainty relates to the structure of the model, i.e., whether all of the relevant processes have been incorporated and whether they have been appropriately related to each other.

7.3.1.1 Non-incorporation of a process due to lack of knowledge

The conceptual uncertainty in the present model could be due to the ignorance regarding the processes of pollutant transport and dispersion that are not incorporated into the model. For example, in case of Manor Road, with the ranges of the model parameters obtained from Gloucester Place, the model could not reproduce the surface loading measurement. This suggests that either the processes included in the present model are not enough to represent the processes happening on the road-vehicle environment and/or the parameter ranges obtained from the Gloucester Place experiment are too narrow to be applicable in roads with varying traffic and road conditions and with different road geometry. It may well be due to some material movement process that is active at Manor Road, but not included in the model. This can only be identified with experience and repetition of field studies in roads with varying traffic and road surface characteristics and with varying source.

7.3.1.2 Non-incorporation of a possible process: grinding

The largest source of uncertainty in the whole study is probably the assumption that particles in road sediment remain discrete single particles until the time it is removed from the road surface. During the process of particle movement on the paved road surface, some grinding almost certainly occurs, by which coarser particles are lost and an additional source of finer particles appears. However lack of proper understanding and non-availability of data are the constraints for this process to be included in the model. Therefore the absence of a source term in the present model that increases the fine particle reservoir on road surface could be overestimating the material removal rate because of easy removal of coarser material either to kerb or to air. Therefore how the

total emission and movement of all particle sizes relate to the emission of PM_{10} that is specifically of current interest is not precisely known. Plausible evidence of grinding and effect of its non-inclusion on model performance are discussed in Section 7.2.6. More research is required to understand and estimate this dynamic nature of particle size variation so that it can be parameterised for inclusion in the model.

7.3.2 Representational uncertainty

Representational uncertainty arises because some processes can be represented mathematically in different ways and it includes matters relating both to the choice of equations to be used and to the selection of the analytical or numerical techniques for solving those equations. Conceptual and representational uncertainty are sometimes referred to together as model uncertainty (Thiessen et al., 1999).

7.3.2.1 Effect of rain

Although removal by rain is parameterised in the model, its precise effect on changing the surface loading on a road surface is not known. Not many studies are available on this. Available studies provide, to some extent, contradicting views (Sections 5.6.3.4). Some of the uncertainty associated with particle removal by rain is discussed in Section 6.3.2.1. Improved parameterisation of rain event by improving the calculation method presented in this model will reduce this uncertainty.

7.3.3 Data uncertainty

Also known as parameter uncertainty, this uncertainty is associated with input values. This relates to the reliability with which the parameters describing the transport and dispersion process can be measured or quantified.

7.3.3.1 Variability associated with atmospheric processes

Like any other environmental model, in the present model this type of uncertainty is associate with wind strengths that are highly variable with height, atmospheric

turbulence that varies depending on the time of the day and the amount of insolation, emissions that are diluted to varying degrees based on meteorological conditions, and deposition that is highly dependent on particle size.

7.3.3.2 Wide range of values for input parameters

As shown in Table 6.1, rates of road abrasion and tyre wear vary by a factor of 10^2. Variability associated with abrasion rate is critical because, in absence of any major local source, it is the most important source of surface loading. Therefore it is a source of uncertainty for this type of model.

Ideally, road wear emissions estimates should take into account the fact that total emissions are likely to be dominated by small sections of road where the wear is especially rapid, due to road surface being in poor condition or due to driving pattern (i.e., higher values are reported at Freeway exits where vehicle must slow rapidly, maximising the occurrence of brake and road wear (Abu-Allaban et al., 2003b)), and that heavy goods vehicles are likely to be responsible for a disproportionate fraction of the emissions in such circumstances, consistent with inferences drawn from other studies (AQEG, 2004; Sehmel, 1973, 1976b). Further, at present the rate of road abrasion is available from the limited studies based on modelling. Therefore to verify this, the average rate of wear of paved road surfaces should be estimated from highways engineering data and from field measurement. This should include comparison of new and worn surfaces, especially since very new surfaces have anecdotally been observed to emit small or negligible quantities of particulate matter. Development of the connection of this stream of data into atmospheric emissions inventory calculations should be the subject of an additional research that could be integrated with the automated surveying of road surface condition.

7.3.3.3 Effect of wind stress

Earlier field experiments suggest resuspension by wind forces is minimal compared to the resuspension caused by traffic (Sehmel, 1973, 1976b). However, Countess et al. (2001) suggest to quantify the emission due to wind for a better precision of traffic-

induced emission. Harrison et al. (2001) have reported effect of wind speed on particle resuspension by fitting the curve to the data obtained from measurement at five locations in the UK, including two locations in London. They suggested the following formula, similar to that described in Section 2.9, for wind-induced resuspension.

$$C_r = b\,u^a \tag{7.1}$$

Where,

C_r = wind-induced resuspended PM_{coarse} concentration ($\mu g\ m^{-3}$)

u = wind speed ($m\ s^{-1}$)

a & b = coefficients derived from curve fit

For the summer of 1997, Harrison et al. (2001) reported average values for b of 0.43 for London Marylebone Road and 0.02 for London Bloomsbury. Average values of a for the same period were 1.50 for London Marylebone Road and 2.26 for London Bloomsbury. Using the 90^{th} percentile street-level wind speed of 2.6 m s^{-1} at Gloucester Place and 3.4 m s^{-1} at Manor Road and values of a and b mentioned above in Equation 7.1, the estimates of resuspension by wind-stress lie between 0.17 $\mu g\ m^{-3}$ and 1.8 $\mu g\ m^{-3}$ at Gloucester Place and 0.31 $\mu g\ m^{-3}$ and 2.7 $\mu g\ m^{-3}$ at Manor Road. These estimates are equivalent to between 3.7% and 40% of the total resuspension at Gloucester Place and between 1.6% and 13.5% of the total resuspension at Manor Road.

Our assumption is that no grit was resuspended by wind stresses between field test times. However, if significant amounts (the estimates above show that it could have been as high as 13.5% at Gloucester Place and 40% at Manor Road) of grit were resuspended by wind stresses, then the true traffic-induced resuspension rates would be smaller than those estimated. Consequently, these data are the higher limits of traffic-induced resuspension rates expected for the test conditions. Sehmel (1973) has expressed similar views for this type of study. Therefore, the uncertainty of the value of critical parameters, such as the material removal rates, mainly limits the precision of model prediction.

7.3.3.4 Nature of material used for the study

Rock salt is hygroscopic in nature and therefore its movement on urban roads could be a little different from the crustal material and material used for road construction. No account has been taken of the chemical properties of the material during the experiment.

7.4 Application

7.4.1 Development of non-exhaust emission inventory

As discussed in Section 2.4, with better control of exhaust emission, non-exhaust emission from paved roads are becoming increasingly important and it is expected that for meeting stricter future air quality standards, control of non-exhaust emissions will be the focus. It is proposed that the quantification of particle movement processes presented in this thesis should inform the improvement of atmospheric emissions inventories. Combined with information on rainfall (Deletic et al., 1997; Deletic and Orr, 2005) and the presence of kerb at side of road, the resulting estimate of emissions to air will for the first time provide a non-exhaust particle emissions inventory that can potentially reflect policy options such as road surface renewal and construction dust control, allowing models to be validated and then used to assess air quality management scenarios.

7.4.2 Estimating source contribution from site-specific sources

The model structure gives an idea as to how different sources of particulate matter on road surfaces can be included as long as appropriate parameterisation can be obtained, whether it is a point source such as that of gritting at Gloucester Place or a continuous source such as that of input of material from EGTS at Manor Road. Examples include soiling of the road surface following flooding, the use of amphibious vehicles (http://www.londonducktours.co.uk/gallery.htm), armed forces vehicles causing excessive road surface wear, material falling onto road surfaces from trees including seeds,

flower fragments, and leaves, and road cleaning methods that result in increased road wear.

7.4.3 Emissions from accidental spills

The gritting event at Gloucester Place is analogous to the short-time high-concentration release of particulate pollutants and therefore the quantification of material removal processes at Gloucester Place can be used to estimate the spatial and temporal scale of contamination from accidental spillage of radioactive material, pesticides, etc.

7.4.4 Quantification of traffic-induced pollution migration

The formula proposed for road surface material dispersion gives an initial estimate of the on-road movement of the particles which was not addressed by earlier studies. While the amount going to air is the primary concern, this formula includes the nature of material translocation which shows how the pollutants that escaped from resuspension at one location can get resuspended at other places and therefore it emphasises the importance of on-road pollutant migration process. The formula for spatial variation of emissions can be used with a Geographical Information Systems (GIS) road network database, to produce a map of atmospheric emissions from a map of road surface particulate matter sources, analogous to the way a map of atmospheric concentrations can be produced in GIS from a map of atmospheric emissions.

7.4.5 Quantification of emissions from winter gritting

Finally, the emissions to atmosphere arising from winter gritting of urban paved roads, can be estimated directly by using the parameter values obtained from the Gloucester Place experiment.

7.5 Summary and recommendations

An experiment of traffic-induced dust dispersion was successfully conducted to quantify the rate of movement of particulate matter on paved road surfaces and its emissions to the atmosphere in the context of particulate matter pollution removal mechanisms. The experimental design was sufficiently robust to allow the loss of some measurement data due to instrument malfunction, to be overcome. It was also able to accommodate the movement of material along the road being much more rapid than expected.

The experiment and modelling exercise confirmed that a model for paved road particle emission needs to be two-dimensional and dynamic, including material build-up and removal processes on the road surface and their characteristic times. It is proposed that this will not only increase the physical realism of the model but also reduce the gap between observation and model prediction, two main criticisms faced by the models that parameterise the emission only as instantaneous silt loading and traffic parameters at the point of emission (Countess et al., 2001; Fitz and Bufalino, 2002; Kantamaneni et al., 1996; Nicholson, 2001; Venkatram and Fitz, 1998; Venkatram et al., 1999; Venkatram, 2000, 2001; Zimmer et al., 1992). The development and evaluation of a simple mass-balance model to predict non-exhaust paved road emissions based on this approach are presented.

The experiment was a pilot experiment of short duration which was not repeated due to the public nuisance it created. The interpretation of results is subject to the uncertainties involved in this type of study, discussed in Section 7.3. A longer experiment should be conducted with detailed modelling to estimate and verify these parameters. Extensive field measurements should be carried out to estimate the range of the parameters under different street and meteorological conditions, particularly at an air-flow perpendicular to the road or against the flow of traffic. In this case it was almost a channelled flow in the same direction as the traffic and therefore it is anticipated that the along-street movement of material may have been overestimated which can be checked from an experiment with predominant wind direction perpendicular to road. The across-street parameter could be different if the experiment was conducted

on a road without the effect of street canyon. The initial surface loading in this experiment was significantly higher than what one would find on an urban paved road. This could have affected the estimation of the material removal rates. A study on a paved road without gritting is likely to show a slower material removal rate. There is also uncertainty on how much has precisely gone to the kerb and in what time, because the deposits on the pavement may have partially come from the across-street component. Material of different mineralogical composition could allow collection of on-road samples to improve the accuracy of these initial estimates. Further studies of this type are needed for comparison of these results to pollutant removal rates on streets with different topography on either side.

Any activity that provides a source of material onto the road surface, for example waste transfer stations and construction or demolition sites can be a source of particulate matter on a paved road surface. This can be observed visually where soiling of the road surface occurs due to vehicles with muddy wheels accessing the paved road, including the rate of decline of soiling with distance downstream of the point of entry. But similar movement of material will apply to dust deposited to the road surface from the air, for example from the cutting of paving slabs, mixing of cement and activities associated with routine highway maintenance and other civil engineering works. The UK Quality of Urban Air Review Group (QUARG) estimates the total UK emissions of PM_{10} from these sources have a range from 500 tonnes per annum to 18000 tonnes per annum (Buckingham et al., 1997). Speciation studies by Querol et al. (2004) attribute the mineral city background contribution to demolition, construction and road dust as $5\,\mu g\ m^{-3}$ in Spain and Sweden and $2\,\mu g\ m^{-3}$ - $4\,\mu g\ m^{-3}$ in PM_{10} in other European studies. Rates of production of dust from these activities are likely to be highly variable. Many of the larger construction works, which are also the larger sources of PM_{10}, by their vary nature move site as each project is completed. The future study should aim at parameterising these sources to be included in a model to estimate their likely contribution to urban particulate matter emission. The significance of these sources of emissions, especially as exhaust emissions are controlled, is such that combining GIS datasets of construction and roadworks with atmospheric emissions calculations, is at

least worth exploring. It is likely that the general spatial and temporal variability, and long-term spatial average total emissions could be captured by such an undertaking, even if the daily emissions at a specific location would remain subject to very large uncertainty. Most importantly, the response of atmospheric emissions to changes in the amount of construction and road works, and possibly to the wider application of more effective dust reduction measures, would be able to be modelled. Dry and wet deposition and tyre wear deposition contribute significantly less to the paved road surface loading. The future study should therefore be aimed at obtaining more accurate estimate for road abrasion which is the single largest source of road surface loading. This will increase the accuracy of non-exhaust emission inventory and therefore source terms of the model when significant local sources are absent.

The model presented here has not been subjected to thorough sensitivity and uncertainty analysis. This should certainly be part of the future study to increase the accuracy and reliability of the model. Incorporation of sensitivity and uncertainty analysis to the model will help in identifying critical parameters that control the model performance. Like any other model, this model should not be used as a substitute for measured data where these are available. With inclusion of site-specific source terms following estimation of their likely contribution through measurement, the model can be used to identify and estimate the spatial and temporal distribution of material in a road network. After adequate field studies aimed at estimating the model parameters and evaluating the model, application of the new model has the potential to give an improved assessment and better understanding of the paved road particulate emission than the existing approaches.

Bibliography

Abbey, D. E., Burchette, R. J., Knutsen, S. F., McDonnell, W. F., Lebowitz, M. D., and Enright, P. L. (1998). Long-term particulate and other air pollutants and lung function in nonsmokers. *American Journal of Respiratory and Critical Care Medicine*, 158:289–298.

Abbey, D. E., Nishino, N., McDonnell, W. F., Lebowitz, M. D., and Enright, P. L. (1999). Long-term inhalable particles and other air pollutants related to mortality in nonsmokers. *American Journal of Respiratory and Critical Care Medicine*, 159:373–382.

Abu-Allaban, M., Gillies, J. A., and Gertler, A. W. (2003a). Application of a multi-lag regression approach to determine on-road PM_{10} and $PM_{2.5}$ emission rates. *Atmospheric Environment*, 37:5157–5164.

Abu-Allaban, M., Gillies, J. A., Gertler, A. W., Clayton, R., and Proffitt, D. (2003b). Tailpipe, resuspended road dust, and brake-wear emission factors from on-road vehicles. *Atmospheric Environment*, 37:5283–5293.

AckermannLiebrich, U., Leuenberger, P., Schwartz, J., Schindler, C., Monn, C., Bolognini, C., Bongard, J. P., Brandli, O., Domenighetti, G., Elsasser, S., Grize, L., Karrer, W., Keller, R., KellerWossidlo, H., Kunzli, N., Martin, B. W., Medici, T. C., Perruchoud, A. P., Schoni, m. H., Tschopp, J. M., Villiger, B., Wuthrich, B., Zellweger, J. P., and Zemp, E. (1997). Lung function and long term exposure to air pollutants in Switzerland. *American Journal of Respiratory and Critical Care Medicine*, 155(1):122–129.

Adamson, I. Y. R., Prieditis, H., and Vincent, R. (1999). Pulmonary toxicity of an atmospheric particulate sample is due to soluble fraction. *Toxicology and Applied Pharmacology*, 157:43–50.

ADB (1992). Vehicular emission control planning in Metro Manila. Final report, Asian Development Bank.

ADMS (2000). *ADMS-Urban:An Urban Air Quality Management System User Guide, Version 1.6.* CERC Limited, 3 King's Parade, Cambridge CB2 1SJ. http://www.cerc.co.uk (last accessed: 15 June 2005).

Ahagon, A. and Kaidou, H. (1990). Aging of tire parts during service. I. *Rubber Chemistry and Technology*, 63:683–697.

Alastuey, A., Querol, X., Rodriguez, S., Viana, M. M., Plana, F., Artiñano, B., Salvador, P., Garćia do Santos, S., Fernandez Patier, R., Rosa, J. D. L., Sanchez de la Campa, A., Menéndez, M., and Gil, J. J. (2004). Levels of PM at rural, urban and industrial sites in Spain. In *Proceedings of the International Conference on Particles in the size of 2.5 to 10 microns in urban areas, 4-6 November 2002.* Institute for Meteorology, Free University and the Brandenburg Technical University Cottbus, Berlin, Geramny. http://secus.met.fu-berlin.de/veranstaltungen/Abstracts%20PM10/Alastuey.htm (last accessed: 6 April 2005).

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

American Lung Association (2005). Facts about fine particle air pollution. http://www.cleanairstandards.org/article/articlereview/232/1/15 (last accessed: 3 November 2005).

Anderson, H. R., Bermner, S. A., Atkinson, R. W., Harrison, R. M., and Walters, S. (2001). Particulate matter and daily mortality and hospital admissions in the west

midlands conurbation of the United Kingdom: associations with fine and coarse particles, black smoke and sulphate. *Occupational Environmental Medicine*, 58:504–510.

Anderson, K. R., Avol, E. L., Edvards, S. A., Shamoo, D. A., Peng, R. C., Linn, W. S., and Hacknery, J. D. (1992). Controlled exposures of volunteers to respirable carbon and sulfuric acid aerosols. *Journal of the Air and Waste Management Association*, 42:771.

Angstorm, A. (1964). The parameters of atmospheric turbidity. *Tellus*, 16:64–75.

Anspaugh, L. R., Shinn, J. H., Phelps, P. L., and Kennedy, N. C. (1975). Resuspension and redistribution of plutonium in soils. *Health Physics*, 29(4):571–582.

APEG (1999). Source apportionment of airborne particulate matter in the United Kingdom. Technical Report ISBN 0-7058-1771-7, Airborne Particles Expert Group (APEG), Air and Environment Quality Division, Department of the Environment, Transport and the Regions, London (UK).

ApSimon, H. M., Gonzalez del Campo, M. T., and Adams, H. S. (2001). Modelling long-range transport of primary particulate material over Europe. *Atmospheric Environment*, 35:343–352.

AQEG (2004). Particulate matter in the United Kingdom. Draft report for Comment, Air Quality Expert Group (AQEG), AQEG Secretariate, DEFRA, 4/E14, Ashdown House, 123 Victoria Street, London SW1E 6DE. http://www.defra.gov.uk/environment/airquality/aqeg/index.htm,http://www.defra.gov.uk/corporate/consult/particulate-matter/index.htm (last accessed: 29 September 2005).

Arnold, S. J., ApSimon, H., Barlow, J., Belcher, S., Bell, M., Boddy, J. W., Britter, R., Cheng, H., Clark, R., Colvile, R. N., Dimitroulopoulou, S., Dobre, A., Greally, B., Kaur, S., Knights, A., Lawton, T., Makepeace, A., Martin, D., Neophytou, M., Neville, S., Nieuwenhuijsen, M., Nickless, G., Price, C., Robins, A., Shallcross, D.,

Simmonds, P., Smalley, R. J., Tate, J., Tomlin, A. S., Wang, H., and Walsh, P. (2004). Introduction to the DAPPLE air pollution project. *Science of the Total Environment*, 332:139–153.

Artíñano, B., Salvador, P., Alonso, D. G., Querol, X., and Alastuey, A. (2004). Influence of traffic on the PM_{10} and $PM_{2.5}$ urban aerosol fractions in Madrid (Spain). *Science of the Total Environment*, 334-335:111–123.

BADC (2003). The British Atmospheric Data Centre, Space Science and Technology Department, R25 - Room 1.106, CCLRC Rutherford Appleton Laboratory, Fermi Avenue, Chilton, nr Didcot, Oxfordshire OX11 0QX, England, UK. http://badc.nerc.ac.uk/ (last accessed: 13 April 2005).

Bagley, S. T., Baumgard, K. J., Gratz, L. G., H, J. J., and Leddy, D. G. (1996). Characterization of fuel and after-treatment device effects on diesel emissions. Research Report No. 76, Health Effects Institute, Cambridge, Massachusetts, USA.

Bagnold, R. A. (1941). *The physics of blown sand and desert dunes.* Methuen & Co, London.

Baker, V. R. (1998). Paleohydrology and the hydrological sciences. In Benito, G., Baker, V. R., and Gregory, K. J., editors, *Palaeohydrology and Environmental Change*, pages 1–10. Jon Wiley & Sons, Chichester.

Baldauf, R. W., Bailey, R. C., and Somers, H. J. (2002). Comments on HEI tunnel study. Annual Meeting, Health Effects Institute, Seattle, WA.

Ball, D. J. and Caswell, R. (1983). Smoke from diesel engined road vehicles: An investigation into the basis for British and European emission standards. *Atmospheric Environment*, 17(1):169–181.

Barrie, L. A. (1988). Aspects of atmospheric pollutant origin and deposition revealed by multielemental observations at a rural location in eastern Canada. *Journal of Geophysical Research*, 93:3773–3788.

Barrowcliffe, R., Newton, A., Harrison, R. M., and Jones, A. (2002). Sources of particulate matter in urban areas: TRAMAQ project UG 250. Final report, Department for Transport, 8 Cavendish Square, London W1G 0ER. http://www.dft.gov.uk/stellent/groups/dft_roads/documents/page/dft_roads_507455.pdf (last accessed: 13 April 2005).

Bascom, R., Bromberg, P. A., Costa, D. L., Devlin, R., Dockery, D. W., Frampton, M. W., Lambery, W., Samet, J., Speizer, F. E., and Utell, M. (1996). Health effects of outdoor air pollution. *Respiratory and Critical Care Medicine*, 153(2):477–498.

Benitez, J. V. O., Vergel, K. N., and Eng, D. (2001). Relationship of traffic flow characteristics and roadside suspended particulate matter (SPM) emissions in Metro Manila. In *Proceedings of the 3rd Conference of Eastern Asia Society for Transportation Studies (EASTS)*, volume 3, pages 45–56, Hanoi, Vietnam. http://www.easts.info/p3-3.htm (last accessed: 11 November 2004).

Berdowski, J., Visschedijk, A., Creemers, E., Pulles, T., Pacyna, J., Fudala, J., and Querreveld, D. (2001). Emission estimates for particulate matter under Coordinated European programme on Particulate Matter Emission Inventories, Projections and guidance (CEPMEIP). Technical report, TNO-MEP. http://www.air.sk/tno/cepmeip/em_factors_results.php (last accessed: 15 November 2005).

Berdowski, J. J. M., Mulder, W., Veldt, C., Visschedijk, A. J. H., and Zandveld, P. Y. J. (1996). Particulate matter emissions (PM_{10}, $PM_{2.5}$, $PM_{0.1}$) in Europe in 1990 and 1993. Technical Report TNO_report TNO_MEP_R 96/472, TNO. pp. 90.

Berico, M., Luciani, A., and Formignani, M. (1997). Atmospheric aerosol in urban area measurements of TSP and PM_{10} standards and pulmonary deposition assessments. *Atmospheric Environment*, 31:3659–3665.

Beverland, I. J., Heal, M. R., Agius, R. M., Hibbs, K. R., Elton, R. A., and Fowler, D. (2002). The metal contents of airborne particles in Edinburgh: application to epidemiology. Technical report, Department of Health, London.

Bexley (2001). Stage 3 review and assessment of local air quality in London Borough of Bexley. Technical report, Bexley Council, London, Environmental Health (Home and Environment), 2A Hadlow Road, Sidcup, Kent DA14 4AF. http://www.bexley.gov.uk (last accessed: 13 April 2005).

Bexley (2003). Review and assessment of local air quality in Manor Road, Erith, Consultation draft-Stage 4. Technical report, Bexley Council, London, Environmental Health (Home and Environment), 2A Hadlow Road, Sidcup, Kent DA14 4AF. http://www.bexley.gov.uk (last accessed: 13 April 2005).

Braaten, D. A., Paw, U., and Shaw, R. H. (1990). Particle resuspension in turbulent boundary layer - observed and modelled. *Journal of Aerosol Science*, 21(5):613–628.

Brain, D. V. and Valdberg, P. A. (1979). Deposition of aerosol in the respiratory tract. *American Review of Respiratory Disease*, 120:1325–1373.

Braun-Fahrländer, C., Vuille, J. C., Sennhauser, F. H., Neu, U., Künzle, T., Grize, L., Gassner, M., Minder, C., Schindler, C., Varonier, H. S., Wüthrich, B., and the SCARPOL Team (1997). Respiratory health and long-term exposure to air pollutants in Swiss schoolchildren. *American Journal of Respiratory and Critical Care Medicine*, 155(3):1042–1049.

Brown, D. M., Wilson, M. R., MacNee, W., Stone, V., and Donaldson, K. (2002). Size-dependent proinflammatory effects of ultrafine polystyrene particles: A role for surface area and oxidative stress in the enhanced activity of ultrafines. *Toxicology and Applied Pharmacology*, 175:191–199.

Buckingham, C., Clewley, L., Hutchinson, D., Sadler, L., and Shah, S. (1997). London Atmospheric Emissions Inventory. Technical report, London Research Centre, Environment and Transport Studies, 81 Black Prince Road, London SE1 7SZ.

Bultnyck, H. and Malet, L. M. (1972). Evaluation of atmospheric dilution factors for effluents diffused from an elevated continuous point source. *Tellus*, XXIV:455–472.

Cadle, S. H. and Chock, D. P. (1977). General Motors' sulphate dispersion experiments: experimental procedures and results. *Journal of Air Pollution Control Assessment*, 27:33–34.

CAFE (2004). Summary of the discussion groups of the workshop. In *Proceedings of the International Conference on Particles in the size of 2.5 to 10 microns in urban areas, 4-6 November 2002*, Berlin (Germany). Institute for Meteorology, Free University and the Brandenburg Technical University Cottbus. http://secus.met.fu-berlin.de/veranstaltungen/Abstracts%20PM10/summary.htm (last accessed: 28 April 2005).

CalTOX (1993). CalTOX, A multimedia total exposure model for hazardous-waste sites: Part I, II & III. Technical report, The office of Scientific Affairs, Department of Toxic Substances Control, California Environmental Protection Agency. http://www.dtsc.ca.gov/sciencetechnology/ctox_dwn.html (last accessed: 16th April 2004).

CARB (1998). Emission Inventory 1996. Technical report, Technical Support Division, Emissions Inventory Branch, California Air Resources Board (CARB), Sacramento, California, USA.

Carlton, A. g., Turpin, J. B., Johnson, W., Buckley, B. T., Simcik, M., Eisenreich, S. J., and Porcja, R. J. (1999). Methods for characterisation of personal aerosol exposures. *Aerosol Science Technology*, 31:66–80.

CASAC (2005). Casac endorses stricter fine particle standards. http://www.cleanairstandards.org/article/articleprint/394/-1/41 (last accessed: 03 November 2005).

Chamberlain, A. C. and Chadwick, R. C. (1953). Deposition of airborne radioiodine vapor. *Nucleonics*, 8:22–25.

Chan, W. H., Tang, A. J. S., and Chung, H. S. (1986). Concentration and deposition of trace metals in Ontario -1982. *Water, Air, & Soil Pollution*, 29:373–389.

Chan, Y. C., Simpson, R. W., McTainsh, G. H., and Vowles, P. D. (1997). Charac-

terisation of chemical species in $PM_{2.5}$ and PM_{10} aerosols in Brisbane, Australia. *Atmospheric Environment*, 31:3773–3785.

Chang, M. (2001). Email communication from Michelle Chang, letter dated November 28, 2001 to Mr. Bill Kuykendal, US EPA, Research Triangle Park, NC 27711. http://www.epa.gov/ttn/chief/ap42/ch13/index.html (last accessed: 13 April 2005).

Charlson, R. J. and Rodhe, H. (1982). Factors controlling the acidity of natural rain water. *Nature*, 295:683–685.

Chepil, W. S. (1945). Dynamics of wind erosion-I. Nature of movement of soil by wind. *Soil Science*, 60:305–320.

Chepil, W. S. (1959). Equilibrium of soil grains at the threshold of movement by wind. *Soil Science Society of America Proceedings*, 23:422–428.

Chow, J. C., Watson, J. G., Egami, R. T., Frazier, C. A., Zhiqiang, L., Goodrich, A., and Bird, A. (1990). Evaluation of regenerative-air street sweeping on geographical contributions to PM_{10}. *Journal of the Air and Waste Management Association*, 40:1134–1142.

Chow, J. C., Watson, J. G., Frjita, E. M., Lu, Z., and Lawson, D. R. (1994). Temporal and spatial variations of $PM_{2.5}$ and PM_{10} aerosol in Southern California air quality study. *Atmospheric Environment*, 28:2061–2080.

Chow, J. C., Watson, J. G., Lu, Z., Lowenthal, D. H., Frazier, C. A., Solomon, P. A., Thuillier, R. H., and Magliano, K. (1996a). Descriptive analysis of $PM_{2.5}$ and PM_{10} at regionally representative locations during SJVAQS/AUSPEX. *Atmospheric Environment*, 30:2079–2112.

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

Claiborn, C., Mitra, A., Adams, G., Bamesberger, L., Allwine, E., Kantamaneni, R., Lamb, B., and Westberg, H. (1995). Evaluation of PM_{10} emission rates from paved

and unpaved roads using tracer techniques. *Atmospheric Environment*, 29(10):1075–1089.

CLEA (2002). Contaminated Land Exposure Assessment (CLEA) Model, Report CLR 10: Technical basis and Algorithm, Version 1.3.0.2. Technical report, Department for Environment, Food & Rural Affairs (DEFRA), Nobel House, 17 Smith square, London SW1P 3JR. http://www.defra.gov.uk/environment/landability/pubs.htm (last accessed: 12 March 2004).

Colvile, R. N., Hutchinson, E. J., Mindell, J. S., and Warren, R. F. (2001). The transport sector as a source of air pollution. *Atmospheric Environment*, 35:1537–1565.

COMEAP (1995). Health effects of non-biological particles. Technical report, Committee on the Medical Effects of Air Pollutants, Department of Health, United Kingdom, HMSO, London.

COMEAP (1998). Quantification of the effects of air pollution on health in the United Kingdom. Technical report, Committee on the Medical Effects of Air Pollutants, Department of Health, London, United Kingdom.

COMEAP (2001). Statement on long-term effects of particles on mortality. Technical report, Committee on the Medical Effects of Air Pollutants, Department of Health, London, United Kingdom. http://www.advisorybodies.doh.gov.uk/comeap/ (last accessed: 13 April 2005).

Commission of the European Communities (2005). Proposal for a directive of the european parliament and of the council on ambient air quality and cleaner air for europe. Provisional Version COM(2005) 447, European Commission, Brussels. http://europa.eu.int/comm/environment/air/cafe/pdf/cafe_dir_en.pdf (last accessed: 3 November 2005).

Corn, M. and Stein, F. (1965). Re-entrainment of particles from a plane surface. *American Industrial Hygiene Association Journal*, 26:325–336.

Countess, R. (2001). Communication from Richard J. Countess, PhD, COUNTESS ENVIROMENTAL,November 21, 2001 to Mr. Bill Kuykendal, US EPA, Research Triangle Park, NC 27711. http://www.epa.gov/ttn/chief/ap42/ch13/index.html (last accessed: 13 April 2005).

Countess, R., Barnard, W., Claiborn, C., Gillette, D., Latimer, D., Pace, T., and Watson, J. (2001). Methodology for estimating fugitive windblown and mechanically resuspended road dust emissions applicable for regional air quality modelling. In *10^{th} International Emission Inventory Conference: "One Atmosphere, One Inventory, Many Challenges", Denver, Colorado, May 01-03.* Emission Factor and Inventory Group, US Environmental Protection Agency. http://www.epa.gov/ttn/chief/conference/ei10/fugdust/countess.pdf (last accessed: 13 April 2005).

Cowherd, C. J., Cuscino, T. A., and Small, M. (1982). Cost effectiveness of road dust control. Technical Report EPA-600/9-82-021, US Environmental Protection Agency, Research Triangle Park, NC.

Cuscino, T. A., Muleski, G. E., and Cowherd, C. J. (1983). Determination of the decay in control efficiency of chemical dust suppressants. In *Symposium on iron and steel pollution abatement technology for 1982*, Research Triangle Park, NC. US Environmental Protection Agency.

D'Alessandro, A., Lucarelli, F., Mandò, P. A., Marcazzan, G., Nava, S., Prati, P., Valli, G., Vecchi, R., and Zucchiatti, A. (2003). Hourly elemental composition and sources identification of fine and coarse PM_{10} particulate matter in four Italian towns. *Journal of Aerosol Science*, 34:243–259.

Dannis, M. L. (1974). Rubber dust from normal wear of tyres. *Rubber Chemistry and Technology*, 47:1011–1037.

DEFRA (2000). The Air Quality Strategy for England, Scotland, Wales and Northern Ireland: Working Together for Clean Air (updated 17 September 2001). Technical report, Department for Environment, Transport & Regions (DETR), Nobel House, 17

Smith Square, London SW1P 3JR. http://www.defra.gov.uk/environment/airquality/strategy/pdf/foreward.pdf (last accessed: 20 March 2004).

DEFRA (2003). The Air Quality Strategy for England, Scotland, Wales and Northern Ireland: Addendum. Technical report, Department for Environment, Food & Rural Affairs (DEFRA), Nobel House, 17 Smith Square, London SW1P 3JR. http://www.defra.gov.uk/environment/airquality/strategy/addendum/pdf/aqs_addendum.pdf (last accessed: 20 March 2004).

Deletic, A., Ashley, R., and Rest, D. (2000). Modelling input of fine granular sediment into drainage systems via gully-pots. *Water Research*, 34(15):3836–3844.

Deletic, A. and Makismovic, C. (1998). Evaluation of water quality factors in storm run-off from paved roads. *Journal of Environmental Engineering*, 124(9):869–879.

Deletic, A., Makismovic, C., and Ivetic, M. (1997). Modelling storm wash-off of suspended solids from impervious surfaces. *Journal of Hydraulic Research*, 35(1):99–119.

Deletic, A. and Orr, D. W. (2005). Pollution buildup on road surfaces. *Journal of Environmental Engineering*, 131(1):49–59.

Dockery, D. W., Cunningham, J., Damokosh, A. I., Neas, L. M., Spengler, J. D., Koutrakis, P., Ware, J. H., Raizenne, M., and Speizer, F. E. (1996). Health effects of acid aerosols on North American children: respiratory symptoms. *Environmental Health Perspectives*, 104:500–505.

Dockery, D. W., Pope, C. A., Xu, X., Spengler, J. D., Ware, J. H., Fay, M. E., Ferris, B. G., and Speizer, F. E. (1993). An association between air pollution and mortality in six U.S. cities. *The New England Journal of Medicine*, 329(24):1753–1759.

Dockery, D. W. and Pope III, C. A. (1994). Acute respiratory effects of particulate air pollution. *Annual Reviews of Public Health*, 15:107–132.

Dockery, D. W., Schwartz, J., and Spengler, J. D. (1992). Air pollution and daily mortality: association with particulates and acid aerosols. *Environmental Research*, 59:362–373.

Donaldson, K., Li, X. Y., and MacNee, W. (1998). Ultrafine (nanometer) particle mediated lung injury. *Journal of Aerosol Science*, 29:553–560.

Dorsey, J. R., Nemitz, E., Gallagher, M. W., Fowler, D., Williams, P. I., Bower, K. N., and Beswick, K. M. (2002). Direct measurements and parameterisation of aerosol flux, concentration and emission velocity above a city. *Atmospheric Environment*, 36(6):791–800.

Düring, I., Jacob, J., Lohmeyer, A., Lutz, M., and Reichenbächer, W. (2002a). Estimation of the "non-exhaust pipe" PM_{10} emissions of streets for practical traffic air pollution modelling. In *Proceedings of the 11th International Symposium on Transport and Air Pollution, June 19-21, 2002*, volume I, pages 309–316. Graz University of Technology, Institute for Internal Combustion Engines and Thermodyanmics, Graz, Austria.

Düring, I. and Lohmeyer, A. (2001). Validierung von pm_{10}-immissionsberechnungen im nahbereich von straßen und quantifizierung der feinstaubbildung von straßen. Technical report, Bericht für Senatsverwaltung für Stadtentwicklung, BrückenstraSSe 6, 10173 Berlin und Sächsisches Landesamt für Umwelt und Geologie, Zur Wetterwarte 11, 01109 Dresden. www.Lohmeyer.de, click there "Literatur" (last accessed: 10 June 2005).

Düring, I., Schulze, E., Jacob, J., Reichenbächer, W., and Lohmeyer, A. (2002b). Estimation of the "non-exhaust pipe" PM_{10} emissions of streets for practical traffic air pollution modelling. In *Proceedings of the 8th International Conference on Harmonisation, October 14-17, 2002*. Sofia, Bulgaria.

Edwards, J., Walters, S., and Griffith, R. K. (1994). Hospital admissions for asthma in preschool children: relationship to major roads in Birmingham, United Kingdom. *Archives of Environmental Health*, 49:223–227.

EEA (1999). EEA guidelines for defining and documenting data on costs of possible environmental protection measures. Technical report 27, European Environment

Agency. http://glossary.eea.eu.int/EEAGlossary/E/emission_factor (last accessed: 13 April 2005).

Englert, N. (2004). Fine particles and human health - a review of epidemiological studies. *Toxicological Letters*, 149(1-3):235–242.

EPAQS (1998). A recommendation for a United Kingdom air quality standard for particles. Technical report, Expert Panel on Air Quality Standards (EPAQS), Department of the Environment, Transport and the Regions (DETR) (now Department for Environment, Food & Rural Affairs (DEFRA)), Nobel House, 17 Smith Square, London SW1P 3JR. http://www.defra.gov.uk/environment/airquality/aqs/air_measure/index.htm (last accessed: 20 July 2004).

EPAQS (2001). Airborne particles: What is the appropriate measurement on which to base a standard? A discussion document. Technical report, Expert Panel on Air Quality Standards (EPAQS), Department for Environment, Food & Rural Affairs (DEFRA), Nobel House, 17 Smith Square, London SW1P 3JR. http://www.defra.gov.uk/environment/airquality/aqs/air_measure/index.htm (last accessed: 13 April 2005).

Ferin, J., Oberdörster, G., Soderholm, S. C., and Gebein, R. (1991). Pulmonary tissue access of ultrafine particles. *Journal of Aerosol Medicine*, 4:57–58.

Fitz, D. R. (1998). Evaluation of street sweeping as a PM_{10} control method. Final Report. Technical report, Centre for Environmental Research and Technology, College of Engineering, University of California, Riverside, USA. Prepared for the Mobile Source Air Pollution Review Committee (MSRC), South Coast Air Quality Management District.

Fitz, D. R. and Bufalino, C. (2002). Measurement of PM_{10} emission factors from paved roads using on-board particle sensors. In *Proceedings of the 11th Annual Emission Inventory Conference: "Emission Inventories - Partnering for the Future", Atlanta, Georgia, April 16-18.* Emission Factor and Inventory Group, United States Environmental Protection Agency.

Fuller, G. W. and Green, D. (2004). The impact of local fugitive PM_{10} from building works and road works on the assessment of the European Union Limit Value. *Atmospheric Environment*, 38:4993–5002.

Fwa, T. F. and Ang, B. W. (1991). Tire-wear characteristics of public buses. *Journal of Transportation Engineering*, 117(3):298–310.

Gaffney, P., Bode, R., and Murchison, L. (1995). PM_{10} emission inventory improvement program for California. Technical report, Air Resource Board, 2020 L Street, Sacramento, California 95814.

Gámez, A. J., Berkowicz, R., Ketzel, M., Lohmeyer, A., and Reichenbächer, W. (2001a). Determination of the "non-exhaust pipe" PM_{10} emissions of roads for practical traffic air pollution modelling. In *Proceedings of the 7th International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes, Belgirate, Italy, May 28-31.* JRC-EI. http://www.harmo.org/conferences/belgirate/7harmo.asp (last accessed: 13 April 2005),.

Gámez, A. J., Düring, I., Bösinger, R., Rabl, P., and Lohmeyer, A. (2001b). Determination of the 99.8-percentile of NO_2 concentrations and PM_{10} emissions for EIA Studies. In *Proceedings of the 3rd International Conference on Urban Air Quality, Loutraki, Greece, March 19-23.*

Garg, B. D., Cadle, S. H., Mulawa, P. A., Groblicki, P. J., Laroo, C., and Parr, G. A. (2000). Brake wear particulate matter emission. *Environmental Science and Technology*, 34:4463–4469.

Garland, J. A. (1979). Resuspension of particulate matter from grass and soil. Technical Report AERE-R9452, HMSO, London.

Garland, J. A. (1983). Some recent studies of the resuspension of deposited material from soil and grass. In Pruppacher, H. R., Semonin, R. G., and Slinn, W. G. N., editors, *Precipitation Scavenging, Dry Deposition and Resuspension*, volume 2, pages 1087–1097. Elsevier, Amsterdam.

Garland, J. A. and Playford, K. (1992). Resuspension of Cs-137 from the Chernobyl accident. In Schwartz, S. E. and Slinn, W. G. N., editors, *Precipitation Scavenging and Atmosphere-Surface Exchange*, volume 3, pages 1605–1614. Hemisphere, Washington, DC.

Gehrig, R. and Buchmann, B. (2003). Characterising seasonal variations and spatial distribution of ambient PM_{10} and $PM_{2.5}$ concentrations based on long-term Swiss monitoring data. *Atmospheric Environment*, 37:2571–2580.

Gertler, A. W., Gillies, J. A., and Pierson, W. R. (2000). An assessment of the mobile sources contributions to PM_{10} and $PM_{2.5}$ in the United States. *Water, Air & Soil Pollution*, 123:203–214.

Gillette, D. A. (1978). Tests with a portable wind tunnel for determining wind erosion threshold velocities. *Atmospheric Environment*, 12:2309–2313.

Gillette, D. A. (1983). Threshold velocities for wind erosion on natural terrestrial arid surfaces (a summary). In Pruppacher, H. R., Semonin, R. G., and Slinn, W. G. N., editors, *Precipitation Scavenging , Dry Deposition and Resuspension*, volume 2, pages 1047–1058. Elsevier, Amsterdam.

Gillette, D. A., Blifford, I. H., and Fryrear, D. W. (1974). The influence of wind velocity on the size distributions of aerosols generated by the wind erosion of soils. *Journal of Geophysical Research*, 79:4068–4075.

Goodwin, J. W. L., Salway, A. G., Murrells, T. P., Dore, C. J., and Eggleston, H. S. (1999). UK emission of air pollutants 1970-1997. Technical report, Department of the Environment, Transport and the Regions (DETR), Nobel House, 17 Smith Square, London SW1P 3JR. http://www.aect.co.uk/netcen/airqual/naei/annreport/annrep99/index.htm (last accessed: 14 December 2004).

Göttle, A. (1979). Ursachen und mechanismen der regenwasserverschmutzung-ein beitrag zur modellierung der abflußbeschaffenheit in städt. Technical report, Ge-

bieten, Berichte aus Wassergüitewirtschaft und Genundgeitsingenieurwesen, TU München H.

Gramotnev, G., Brown, R., Ristovski, Z., Hitchins, J., and L, M. (2003). Determination of average emission factors for vehicles on a busy road. *Atmospheric Environment*, 37:465–474.

Green, D., Fuller, G., and Barratt, B. (2001). Evaluation of TEOM™ 'correction factors' for assessing the EU Stage 1 limit values for PM_{10}. *Atmospheric Environment*, 35:2589–2593.

Grosch, K. A. (1992). Abrasion of rubber and its relation to tyre wear. *Rubber Chemistry and Technology*, 65:78–106.

Hall, D. (1989). The time dependence of particle resuspension. *Journal of Aerosol Science*, 20(8):907–910.

Hanna, S. R., Briggs, G. A., and Hosker Jr., R. P. (1982). Handbook on Atmospheric Diffusion. Technical Report DOE/TIC-11223, U.S. Department of Energy.

Harrison, R. M. (2004). Key pollutants - airborne particles. *Science of the Total Environment*, 334-335:3–8.

Harrison, R. M., Charron, A., Lawrence, R., and Jones, A. M. (2005). Mass closure and source apportionment studies of urban atmospheric aerosol. http://aerosol.web.psi.ch/Final_reports/05%20%20Harrison.pdf (last accessed: 27 August 2005).

Harrison, R. M., Deacon, A. R., Jones, M., and Appleby, R. S. (1997a). Sources and processes affecting concentrations of PM_{10} and $PM_{2.5}$ particulate matter in Birmingham (U.K.). *Atmospheric Environment*, 31(24):4103–4117.

Harrison, R. M., Jones, A. M., and Barrowcliffe, R. (2004a). Field study of the influence of meteorological factors and traffic volumes upon suspended particle mass at urban roadside sites of differing geometries. *Atmospheric Environment*, 38:6361–6369.

Harrison, R. M., Jones, A. M., and Lawrence, R. G. (2003). A pragmatic mass closure model for airborne particulate matter at urban background and roadside sites. *Atmospheric Environment*, 37:4927–4933.

Harrison, R. M., Jones, A. M., and Lawrence, R. G. (2004b). Major component composition of PM_{10} and $PM_{2.5}$ from roadside and urban background sites. *Atmospheric Environment*, 38:4531–4538.

Harrison, R. M., Jones, M., and Collins, G. (1999). Measurements of the physical properties of particles in the urban atmosphere. *Atmospheric Environment*, 33:309–321.

Harrison, R. M., Smith, D. J. T., Pio, C. A., and Castro, L. M. (1997b). Comparative receptor modelling study of airborne particulate pollutants in Birmingham (United Kingdom), Coimbra (Portugal) and Lahore (Pakistan). *Atmospheric Environment*, 31(20):3309–3321.

Harrison, R. M. and Yin, J. (2000). Particulate matter in the atmosphere: which particle properties are important for its effects on health? *The Science of the Total Environment*, 249:85–101.

Harrison, R. M., Yin, J., Mark, D., Stedman, J., Appleby, R. S., Booker, J., and Moorcroft, S. (2001). Studies of the coarse particle (2.5 -10 μm) component in UK urban atmosphere. *Atmospheric Environment*, 35:3667–3679.

Hatch, T. F. and Gross, P. (1964). *Pulmonary Deposition and Retention of Inhaled Aerosols*. Academic Press, Inc., New York.

Hauck, H., Berner, A., Frischer, T., Gomiscek, B., Kundi, M., Neuberger, M., Puxbaum, H., and Preining, O. (2004). AUPHEP - Austrian project on health effects of particulates - general overview. *Atmospheric Environment*, 38(24):3905–3915.

Haywood, J. and Boucher, O. (2000). Estimates of the direct and indirect radiative forcing due to tropospheric aerosols: a review. *Reviews of Geophysics*, 38:513–543.

Heldstab, J., de Hann, P., Künzle, T., and Filliger, P. (1999). PM_{10} map and population exposure for Switzerland. In *Proceedings of the 6th Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes, 11 -14 October*, Rouen, France. CORIA-UMR 6614, CNRS-Université & INSA de Rouen. http://www2.dmu.dk/atmosphericenvironment/6harmo.htm (last accessed: 23 November 2005).

Henshaw, S. (2003). CO_2 measurement in the urban environment. Technical report, University of Bristol, UK. 4th year MSci report.

Hewitt, C. N. (1990). An integrated budget for selected pollutants from a major rural highways. *The Science of the Total Environment*, 93:375–384.

Hildemann, L. H., Markowshi, G. R., and Cass, G. R. (1991a). Chemical composition of emissions from urban sources of fine organic aerosol. *Environmental Science and Technology*, 25:744–759.

Hildemann, L. H., Markowshi, G. R., Jones, M. C., and Cass, G. R. (1991b). Submicrometer aerosol mass distributions of emissions from boilers, fireplaces, automobiles, diesel trucks, and meat cooking operations. *Aerosol Science Technology*, 14:138–152.

Hinds, W. C. (1982). *Aerosol Technology*. Jon Wiley & Sons, New York.

Hitchins, J., Morawaska, L., Wolff, R., and Gilbert, D. (2000). Concentrations of submicrometre particles from vehicle emissions near a major road. *Atmospheric Environment*, 34:51–59.

Hoek, G. and Brunekreef, B. (1993). Acute effects of a winter air pollution episode on pulmonary function and respiratory disease of children. *Archives of Environmental Health*, 48:328–335.

Hoek, G., Brunekreef, B., Goldbohm, S., Fischer, P., and van den Brandt, P. A. (2002). Association between mortality and indicators of traffic-related air pollution in the Netherlands. *The Lancet*, 360:1203–1209.

Hofken, K. D., Meixner, F. X., and Ehhalt, D. H. (1983). Deposition of atmospheric trace constituents onto different natural surfaces. In Pruppacher, H. R., Semonin, R. G., and Slinn, W. G. N., editors, *Dry Deposition and Resuspension*, volume 2, pages 825–836. Elsevier, New York.

Holgate, S., Samet, J. M., Koren, H. S., and Maynard, R. L. (1999). *Air pollution and health*. Academic Press, San Diego/London.

Horst, T. W. (1976). The estimation of air concentrations due to the suspension of surface contamination by the wind. Technical Report BNWL-2047, Pacific Northwest Laboratory, Richland, Washington.

HPS (2003). *STELLA Research, Version 8.0 for Windows*. High Performance Systems Inc., 46, Centerra Parkway, Suite 200, Lebanon, NH 03766. http://www.hps-inc.com (last accessed: 24 March 2004).

Hunt, P. B., Robertson, D. I., Bretherton, R. D., and Winton, R. I. (1982). SCOOT - a traffic responsive method of co-ordinating signals. TRRL Laboratory Report vol. 1014, Transport and Road Research Laboratory (TRRL), UK, Crowthorne, Berkshire, UK.

Ibald-Mulli, A., Wichmann, H. E., Kreyling, W., and Peters, A. (2002). Epidemiological evidence on health effects of ultrafine particles. *Journal of Aerosol Medicine*, 15(2):189–201.

Infante, R., Carrasquillo, A., and Pérez, V. (1990). Size distribution measurements of suspended particulate matter in Ponce, Puerto Rico. *Atmospheric Environment*, 24B(2):275–281.

Jacobson, M. Z. (1999). *Fundamentals of Atmospheric Modelling*. Cambridge University Press, Cambridge.

Jaecker-Voirol, A. and Pelt, P. (2000). PM_{10} emission inventory in Ile de France for transport and industrial sources: PM_{10} resuspension, a key factor for air quality. *Environmental Modelling Software*, 15:575–581.

Janssen, N., Vanmansom, D., Vanderjagt, K., and Harssema, H. (1997). Mass concentration and elemental composition of airborne particulate matte at street and background locations. *Atmospheric Environment*, 31(8):1185–1193.

Jansz, A. (2002). Characterisation of road dust and road side air borne particulate matter to help identify the sources of particulate pollution. Masters thesis, Imperial College of Science, Technology and Medicine, London.

Jansz, A., Fox, J., Williams, T., Colvile, R., Deletic, A., Patra, A., and ApSimon, H. (2004). Mass balance and characteristics of particles of road surface origin in roadside air in Bexley, London. In *Proceedings of the International Conference on Particles in the size of 2.5 to 10 microns in urban areas, 4-6 November 2002*, Berlin (Germany). Institute for Meteorology, Free University and the Brandenburg Technical University Cottbus. http://secus.met.fu-berlin.de/veranstaltungen/Abstracts\%20PM10/Colville.htm (last accessed: 6 April 2005).

Johansson, C., Hansson, H. C., and Areskoug, H. (2004). Concentrations and sources of PM in Sweden. In *Proceedings of the International Conference on Particles in the size of 2.5 to 10 microns in urban areas, 4-6 November 2002*, Berlin, Germany. Institute for Meteorology, Free University and the Brandenburg Technical University Cottbus. http://secus.met.fu-berlin.de/veranstaltungen/Abstracts%20PM10/Johansson.htm (last accessed: 6 April 2005).

Johnson, J. P., Kittelson, D. B., and Watts, W. F. (2005). Source apportionment of diesel and spark ignition exhaust aerosol using on-road data from the Minneapolis metropolitan area. *Atmospheric Environment*, 39:2111–2121.

Kagawa, J. (1984). Health effects of air pollutants and their management. *Atmospheric Environment*, 29:3465–3475.

Kaidou, H. and Ahagon, A. (1990). Aging of tire parts during service. *Rubber Chemistry and Technology*, 63:698–712.

Kalognomou, E. A. and Moussiopoulos, N. (2005). Atmospheric linkage between climate change and air pollution. In Moussiopoulos, N. and Kalognomou, E. A., editors, *Proceedings of the Atmospheric Transport and Transformation at Urban and Local Scales - The Report of the Valencia Workshop, 2005; hosted by Transport and Transformation of Pollutants (T & TP)*, ACCENT Secretariat, Urbino, Italy. http://www.accent-network.org/portal/joint-research-programme/transport-and-transformation-(tandtp) (last accessed: 25 August 2005).

Kantamaneni, R., Adams, G., Bamesberger, L., Allwine, E., Westberg, H., Lamb, B., and Claiborn, C. (1996). The measurement of roadway PM_{10} emission rates using atmospheric tracer ratio techniques. *Atmospheric Environment*, 30(24):4209–4223.

Karppinen, A., Kukkonen, J., Pohjola, M., Partanen, L., and Härkönen, J. (2004). Modelling of the urban concentrations of $PM_{2.5}$ and PM_{10}. In *Proceedings of the International Conference on Particles in the size of 2.5 to 10 microns in urban areas, 4-6 November 2002*, Berlin (Germany). Institute for Meteorology, Free University and the Brandenburg Technical University Cottbus. http://secus.met.fu-berlin.de/veranstaltungen/Abstracts%20PM10/patra.htm (last accessed: 6 April 2005).

Kathuria, V. (2005). Vehicular pollution control in Delhi: Impact of Compressed Natural Gas. *Economic and Political Weekly*, pages 1907–1915.

Katsouyanni, K., Touloumi, G, D., and Spix, C. (1997). Short-term effects of ambient sulphur dioxide and particulate matter on mortality in 12 European cities: results from time-series data from the APHEA project. *British Medical Journal*, 314:1658–1663.

Katsouyanni, K., Zmirou, D., Spix, C., Sunyer, J., Schouten, J. P., Pönkä, A., Anderson, H. R., Le Moullec, Y., Wojtyniak, B., Vigotti, M. A., and Bacharova, L. (1995). Short-term effects of air pollution on health: a European approach using epidemiological time-series data. *European Respiratory Journal*, 8:1030–1038.

Kim, M. G., Yagawa, K., Inoue, H., Lee, Y. K., and Shirai, T. (1990). Measurement of tyre tread in urban air by pyrolysis-gas chromatography with flame photometric detection. *Atmospheric Environment*, 24A(6):1417–1422.

King, A. M., Pless-Mulloli, T., Merefield, J., and Stone, I. (2000). New directions: TEOMs and the volatility of UK non-urban PM_{10}: a regulatory dilemma? *Atmospheric Environment*, 34:3211–3212.

Kirchner, T. B. (1991). Establishing model credibility involves more than validation. In *BIOMOVS: Proceedings of a Symposium on the Validity of Environmental Transfer Models*, pages 371–378. Swedish Radiation Protection Institute, Stockholm.

Kirchner, T. B. and Whicker, F. W. (1984). Validation of PATHWAY, a simulation model of the transport of radionuclides through agroecosystems. *Ecological Modelling*, 22:21–44.

Kittelson, D. B. (1998). Engines and nanoparticles: a review. *Journal of Aerosol Science*, 29:575–588.

Kittelson, D. B. (2001). Recent measurements of nanoparticle emissions from engines. Presented at the meeting on "Current research on diesel exhaust particles" at Japan Association of Aerosol Science and Technology, Tokyo. http://www.me.umn.edu/centere/mel/reports/JASSTpaper.pdf (last accessed: 17 July 2005).

Kittleson, D. B., Watts, W. F., and Johnson, J. P. (2004). Nanoparticle emissions on Minnesota highways. *Atmospheric Environment*, 38:9–19.

Kleeman, M. J. and Cass, G. R. (1988). Source contributions to the size and composition distribution of urban particulate air pollution. *Atmospheric Environment*, 32:2803–2816.

Kristensson, A., Johansson, C., Westerholm, R., Swietlicki, E., Gidhagen, L., Wideqvist, U., and Vesely, V. (2004). Real-world traffic emission factors of gases and particles measured in a road tunnel in Stockholm, Sweden. *Atmospheric Environment*, 38:657–673.

Kuhns, H., Etyemezian, V., Landwehr, D., MacDougall, C., Pitchford, M., and Green, M. (2001a). Testing re-entrained aerosol kinetic emissions from roads (TRAKER): a new approach to infer silt loading on roadways. *Atmospheric Environment*, 35(16):2815–2825.

Kuhns, H., Etyemezian, V., and Shinbein, P. (2001b). Relating road dust emissions surrogates to average daily traffic and vehicle speed in Las Vegas, Nevada. In *Proceedings of the 10th International Emission Inventory Conference: "One Atmosphere, One Inventory, Many Challenges", May 1-3*, Denver, Colorado. Emission Factor and Inventory Group, United States Environmental Protection Agency.

Kulmala, M., Riihiluoma, V., and Raunemaa, T. (1986). Particle emission from gasoline powered vehicles: Emission deposition and re-emission under different traffic density situation. *Journal of Aerosol Science*, 17(6):973–983.

Künzli, N., Kaiser, R., Medina, S., Studnicka, M., Chanel, O., Filliger, P., Herry, M., Horak Jr, F., Puybonnieux-Texier, V., Quénel, P., Schneider, J., Seethaler, R., Vergnaud, J.-C., and Sommer, H. (2000). Public-health impact of outdoor and traffic-related air pollution: a European assessment. *The Lancet*, 356(2):795–801.

Kuykendal, W. B. (2002). Technical memorandum on decisions on final ap-42 section 13.2.1 "paved roads". http://www.epa.gov/ttn/chief/ap42/ch13/index.html (last accessed: 13 April 2005).

Laden, F., Neas, L. M., Dockery, D. W., and Schwartz, J. (2000). Association of fine particulate matter from different sources with daily mortality in six U.S. cities. *Environmental Health Perspectives*, 108(10):941–947.

Le Bihan, O., Gámez, A., Lohmeyer, A., and Berkowicz, R. (2004). Particle emission and dispersion in street canyon. In Berkowicz, R., Britter, R., and Di Sabatino, S., editors, *Optimisation of Modelling Methods for Traffic Pollution in Streets (TRAPOS)*, CD 1, pages 60–66. TRAPOS.

Li, X. Y., Brown, D., Smith, S., MacNee, W., and Donaldson, K. (1999). Short-term inflammatory responses following intratracheal instillation of fine and ultrafine carbon clack in rats. *Inhalation Toxicology*, 11(8):709–731.

Lin, L. H., Harrison, R. M., and Harrad, S. (1999). The contribution of traffic to atmospheric concentrations of polycyclic aromatic hydrocarbons. *Environmental Science and Technology*, 33:3538–3542.

Lindgren, Å. (1996). Asphalt wear and pollution transport. *The Science of the Total Environment*, 189/190:281–286.

Linsey, G. S. (1978). Resuspensin of transuranium elements - a review of existiong data. Technical report, NRPB-R75, HMSO, London.

Lippmann, M. (1998). The 1997 US EPA standards for particulate matter and ozone. In RE, H. and RM, H., editors, *Issues in Environmental Science and Technology*, pages 75–99. Royal Society of Chemistry.

Lippmann, M., Bachmann, J. D., Bates, D. V., Cassec, F. R., van Bree, L., Driscoll, K. E., Phalen, R. F., Pope, C. A., Soderholm, S. C., and Wilson, W. E. (1998). Report of the particulate matter research strategies workshop, Park City, Utah, April 29-30, 1996. *Applied Occupational and Environmental Hygiene*, 13(6):485–493.

Lohmeyer, A., Lambrecht, U., Peranic, Z., and Boesinger, R. (2002). Validation of vehicle road PM_{10} emission models by the Karlsruhe PM validation data set and the results of the regular German state monitoring stations. In *Proceedings of the 8^{th} International Conference on Harmonisation, Sofia, October 14-17.*

Longley, I. D., Gallagher, M. W., Dorsey, J. R., Flynn, M., Bower, K. N., and Allan, J. D. (2004). Street canyon aerosol pollutant transport measurements. *Science of the Total Environment*, 334-335:327–336.

Lovelock, J. E. and Ferber, G. J. (1982). Exotic tracers for atmospheric studies. *Atmospheric Environment*, 16(6):1467–1471.

Luhana, L., Sokhi, R., Warner, L., Mao, H., Boulter, P. G., McCrae, I. S., Wright, J., and Osborn, D. (2004). Non-exhaust particulate measurements: results. Deliverable8 of the EU 5GP project PARTICULATES. Technical report, TRL Limited, Crowthorne, UK.

Lupker, H. A., Montanaro, F., Donadio, D., Gelosa, E., and Vis, M. A. (2002). Truck tyre wear assessment and prediction. In *Proceedings of the 7th International Symposium on Heavy vehicle weights & dimensions, Delft, The Netherlands, June 16-20.* http://www.automotive.tno.nl/VD/Docs/71SHVWD%20paper%20on%20TROWS%20project.pdf (last accessed: 9 June 2005).

Maheswaran, R., Brindley, P., Fryers, P., Wise, S., Payne, N., Campbell, M., L, R., and Haining, R. (2002). Chronic effects of outdoor air pollution on cardiovascular and respiratory symptoms, admissions and mortality in Sheffield. In Shuker, L. and Forster, P., editors, *Proceedings of the 6th annual UK review meeting on outdoor and indoor air pollution research, 15th-16th April 2002*, pages 68–69, University of Leicester, 94 Regent Road, Leicester LE1 7DD, UK. MRC Institute for Environment and Health,. http://www.le.ac.uk/ieh/ (last accessed: 16 May 2005).

Manoli, E., Voutsa, C., and Samara, C. (2002). Chemical characterisation and source identification/apportionment of fine and coarse air particles in Thessaloniki, Greece. *Atmospheric Environment*, 36:949–961.

Martin, D., Price, C., Shallcross, D., Nickless, G., and Makepeace, T. (2004). Report on sulphur hexafluoride dispersion experiments carried out on the 26th May 2004. Technical report, University of Bristol, Bristol BS8 1TS, UK.

MathWorks (2004). *MATLAB®7 & SIMULINK®6 Student Version, Version 7.0.0.27 (R14).* The Mathworks, Inc., USA. http://www.mathworks.com (last accessed: 24 March 2004).

Matsumoto, H. (1999). System dynamics model for life cycle assessment (LCA) of residential buildings. In *System Dynamics Review.* http://www.ibpsa.org/proceedings/bs99/papers/PB-07.pdf (last accessed: 20 July 2004).

Maynard, A. D. and Maynard, R. L. (2002). A derived association between ambient aerosol surface area and excess mortality using historic time series data. *Atmospheric Environment*, 36:5561–5567.

McKone, T. E., Hall, D., and Kastenberg, W. E. (1997). CalTOX Version 2.3 , Descriptions of modifications and revisions. Technical report, Human and Ecological Risk Division, Department of Toxic Substances Control, California Environmental Protection Agency. http://www.dtsc.ca.gov/sciencetechnology/ctox_dwn.html (last accessed: 29th April 2004).

Miass, M. and Brenninkmeijer, K. A. M. (1998). Atmospheric SF_6: Trends, sources and prospects. *Environmental Science Technology*, 32(20):3077–3086.

Moosmüller, H., Arnott, W. P., Rogers, C. F., Bowen, J. L., Gillies, J. A., Pierson, W. R., Collins, J. F., Durbin, T. D., and Norbeck, J. M. (2001). Time resolved characterisation of diesel particulate emissions. 1. Instrument for particle mass measurements. *Environmental Science Technology*, 35:781–787.

Morawska, L., Bofinger, N. D., Kocis, L., and Nwankwoala, A. (1998). Submicron and supermicron particles from diesel vehicle emissions. *Environmental Science Technology*, 32(14):2033–2042.

Mosello, R., Marchetto, A., and Tartari, G. A. (1988). Bulk and wet atmospheric deposition chemistry at Pallanza (N. Italy). *Water, Air & Soil Pollution*, 42:137–151.

Mukerjee, D. (1998). Critical review: assessment of risk from multimedia exposures of children to environmental chemicals. *Journal of the Air and Waste Management Association*, 48(6):483–501.

Muleski, G. E., Cowherd, C. J., and Brooks, G. (2001). Reevaluation of the unpaved road emission factor model. In *Proceedings of the 10th International Emission Inventory Conference: "One Atmosphere, One Inventory, Many Challenges", May 1-3,*

Denver, Colorado. Emission Factor and Inventory Group, United States Environmental Protection Agency.

Murphy, S. A., BeruBe, K. A., Pooley, F. D., and Richards, R. J. (1998). Response of lung epithelium to well characterised fine particles. *Life Sciences*, 62(19):1789–1799.

Muschack, W. (1990). Pollution of street run-off by traffic and local condition. *The Science of the Total Environment*, 93:419–431.

Myers, R. E. (2001). Communication from Ronald E. Myers, letter dated November 30, 2001 to Mr. Bill Kuykendal, US EPA, Research Triangle Park, NC 27711. http://www.epa.gov/ttn/chief/ap42/ch13/index.html (last accessed: 13 April 2005).

NAAQS (2002). Appeals Court upholds ozone and PM NAAQS. http://cleanairstandards.org/article/articlereview/175/1/12 (last accessed: 03 November 2005).

NAEI (2003). The UK Emission Factor Database, version 02.8. www.naei.org.uk/emissions/index.php (last accessed: 13 April 2005), www.defra.gov.uk/environemnt/airquality/laqm/guidance/pdf/laqm-tg03.pdf (last accessed: 13 April 2005).

Nemmar, A., Hoet, P. H. M., Vanquickenborne, B., Dinsdale, D., Thomeer, M., Hoylaerts, M. F., Vanbilloen, H., Mortelmans, L., and Nemery, B. (2002). Passage of inhaled particles into the blood circulation in humans. *Circulation*, 105:411–414.

NEPC (2003). Variation to the national environment protection (ambient air quality) measure. Technical report, National Environment Protection Council Service Corporation. http://www.ephc.gov.au/pdf/Air_Variation_PM25/PM2_5_Variation.pdf (last accessed: 3 November 2005).

NETCEN (2005). The UK National Air Quality Information Archive managed by NETCEN. http://www.airquality.co.uk/archieve/index.php (last accessed: 6 August 2005).

Newman, J. E., Abel, M. D., Harrison, P. R., and Yost, K. J. (1974). Wind as related to critical flushing speed vs reflotation speed by high volume sampler particulate loading. In *Proceedings of the Atmosphere-Surface Exchange of Particulate and Gaseous Pollutants, CONF 740921, 4-6 September*, pages 466–496, Energy Res. and Dev. Admin. Symp. Ser., Richland, Washington. Nat. Tech. Inf. Serv., Springfield, Virginia.

Nicholson, K. W. (1988). A review of particle resuspension. *Atmospheric Environment*, 22(12):2639–2651.

Nicholson, K. W. (1993). Wind tunnel experiments on the resuspension of particulate material. *Atmospheric Environment*, 27A(2):181–188.

Nicholson, K. W. (2001). A critique of empirical emission factor models: A case study of the AP-42 model for estimating PM_{10} emission from paved roads (Venkatram, A., Atmospheric Environment 34, 1-11). *Atmospheric Environment*, 35:185–186.

Nicholson, K. W. and Branson, J. R. (1990). Factors affecting resuspension by road traffic. *The Science of the Total Environment*, 93:349–358.

Nicholson, K. W., Branson, J. R., and Giess, P. (1991). Field measurements of the below-cloud scavenging of particulate material. *Atmospheric Environment*, 25A(3/4):771–777.

Nicholson, K. W., Branson, J. R., Giess, P., and Cannell, R. J. (1989). The effect of vehicle activity on particle resuspension. *Journal of Aerosol Science*, 20(8):1425–1428.

Nielsen, S. P., Kohler, H., and Peterson, S. (1991). Testing of models for the grass-cow-milk pathway. In *BIOMOVS: Proceedings of a Symposium on the Validity of Environmental Transfer Models*, pages 15–28. Swedish Radiation Protection Institute, Stockholm.

Ntziachristos, L. and Samaras, Z. (2003). New directions: Emerging demands for vehicle particle emission characterisation. *Atmospheric Environment*, 37:441–442.

Nyberg, F., Gustavsson, P., and Järup, K. (2000). Urban air pollution and lung cancer in Stockholm. *Epidemiology*, 11:487–495.

NYSERDA (2002). A survey of monitoring instruments for measurement of airborne pollutants. Final Report 03-05, New York State Energy Research and Development Authority, 17 Coulmbia Circle, Albany, New York 12203-6399. http://www.nyserda.org (last accessed: 08 October 2004).

Oberdörster, G., Ferin, J., Gelein, R., Soderholm, S. C., and Finkelstein, J. (1992). Role of the alveolar macrophage in lung injury: study with ultrafine particles. *Environment and Health Perspectives*, 97:193–199.

Oberdörster, G., Gelein, R., Ferin, J., and Weiss, B. (1995). Association of particulate air pollution and acute mortality: involvement of ultrafine particles. *Inhalation Toxicology*, 7:111–124.

OEG (2005). Onyx Environmental Group plc, Onyx House, 154A Pentonville Road, London N1 9PE. http://www.onyxgroup.co.uk/ (last accessed: 10 August 2005).

Okamoto, S., Kobayashi, K., Ono, N., Kitabayashi, K., and Katatani, N. (1990). Comparative study on estimation methods for NO_x emissions from a roadway. *Atmospheric Environment*, 24A:1535–1544.

Oreskes, N., Shrader-Frechette, K., and Belitz, K. (1994). Verification, validation, and confirmation of numerical models in the earth sciences. *Science*, 263:641–646.

Ostro, B. D., Hurley, S., and Lipsett, M. J. (1999). Air pollution and daily mortality in the Coachella Valley, California: a study of PM_{10} dominated by coarse particles. *Environmental Research*, 81:231–238.

Ostro, B. D., Lipsett, M. J., and Das, R. (1998). Particulate matter and asthma: a quantitative assessment of the current evidence. *Applied Occupational and Environmental hygiene*, 13(6):453–460.

Painter, D. E. (1974). *Air Pollution Technology*. Reston Publishing Company, Inc. (A Prentice Hall Company), Box 547, Reston, Virgnia 22090, USA.

Pasquill, F. (1961). The estimation of the dispersion of windborne material. *Meteorological Management*, 90:33–49.

PEDCo Environmental Inc. (1981a). Denver demonstration study. Technical report, Colorado division of Air Pollution Control, Denver, CO.

PEDCo Environmental Inc. (1981b). Study of street cleaning impact on particulate levels in Kansas City, Kansas. Technical report, US Environmental Protection Agency, Region VIII, Kansas City, MO.

Peters, A., Döring, A., Wichmann, H. E., and Godden, D. (1995). Particulate air pollution and acute health effects. *The Lancet*, 345:176–178.

Peters, A., Döring, A., Wichmann, H. E., and Koenig, W. (1997a). Increased plasma viscosity during the 1985 air pollution episode: a link to mortality. *The Lancet*, 349:1582–1587.

Peters, A., Wichmann, H. E., Tuch, T., Heinrich, J., and Heyder, J. (1997b). Respiratory effects are associated with the number of ultrafine particles. *American Journal of Respiratory and Critical Care Medicine*, 155(4):1376–1383.

Phillips, M. (1980). A force balance model for particle entrainment into a fluid stream. *Journals of Physics D - Applied Physics*, 13:221–233.

Phupinyokul, M. B. and Harrison, R. M. (1999). Mechanisms of health afflictions from traffic-related particles. Technical Report 2(7), Bureau of Technical Advisory Group, Department of Health, Ministry of Public Health, Thiland. http://advisor.anamai.moph.go.th/factsheet/index_en.htm#envi (last accessed: 13 January 2006).

Pierson, W. R. and Brachaczek, W. W. (1974). Airborne particulate debris from rubber tyres. *Rubber Chemistry and Technology*, 47(150):1275–1299.

Pillai, P. S., Suresh Babu, S., and Krishna Moorthy, K. (2002). A study of PM, PM_{10} and $PM_{2.5}$ concentration at a tropical coastal region. *Atmospheric Research*, 61:149–167.

Pope III, C. A. (2000). Epidemiology of fine particulate air pollution and human health: Biologic mechanisms and who's at risk? *Environmental Health Perspectives*, 108(S4):713–723.

Pope III, C. A., Burnett, R. T., Thun, M. J., , Calle, E. E., Krewski, D., Ito, K., and Thurston, G. D. (2002). Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. *Journal of the American Medical Association (JAMA)*, 287(9):1132–1141.

Pope III, C. A. and Dockery, D. W. (1992). Acute health effects of PM_{10} pollution on symptomatic and asymptotic children. *American Review of Respiratory Disease*, 145:1123–1128.

Pope III, C. A. and Dockery, D. W. (1999). Epidemiology of particle effects. In Holgate, S. T., Samet, J. M., Koren, H. S., and Maynard, R. L., editors, *Air Pollution and Health*, pages 673–705. Academic Press, London.

Pope III, C. A., Dockery, D. W., and Schwartz, J. (1995a). Review of epidemiological evidence of health effects of particulate air pollution. *Inhalation Toxicology*, 7:1–18.

Pope III, C. A., Thun, M. J., Namboodiri, M. M., Dockery, D. W., Evans, J. S., Speizer, F. E., and Heath, C. W. (1995b). Particulate air pollution as a predictor of mortality in a prospective study of U.S. adults. *American Journal of Respiratory and Critical Care Medicine*, 151:669–674.

Powersim (2003). *Powersim Studio 2003*. Powersim Ltd., PO Box 3961 Dreggen, N-5835 Bergen, Norway. http://www.powersim.com (last accessed: 24 March 2004).

Punjrath, J. S. and Heldman, S. R. (1972). Mechanism of small particle re-entrainment from flat surfaces. *Aerosol Science*, 3:429–440.

QUARG (1996). Airborne particulate matter in the United Kingdom. The third report of the Quality of Urban Air Review Group (QUARG). Technical report, Department of the Environment, London, United Kingdom.

Querol, X., Alastuey, A., Ruiz, C. R., Artiñano, B., Hansson, H. C., Harrison, R. M., Buringh, E., ten Brink, H. M., Lutz, M., Bruckmann, P., Straehl, P., and Schneider, J. (2004). Speciation and origin of PM_{10} and $PM_{2.5}$ in selected European cities. *Atmospheric Environment*, 38:6547–6555.

Rao, P. S. P., Khemani, L. T., Momin, G. A., Safai, P. D., and Pillai, A. G. (1992). Measurement of wet and dry deposition at an urban location in India. *Atmospheric Environment*, 26B(1):73–78.

Rauterberg-Wulff, A. (2000). Untersuchung über die Bedeutung der Staubaufwirbelung für die PM_{10}-immission an einer Hauptverkehrsstraße. Technical report, Technical University Berlin, Fachgebiet Luftreinhaltung.

Reeks, M. W., Reed, J., and Hall, D. (1988). On the resuspension of small particles by turbulent flow. *Journals of Physics D - Applied Physics*, 21(4):574–589.

Ristovski, Z., Morawska, L., and Hitchins, J. (1998). Submicrometer and supermicrometer particulate emissions from spark ignition vehicles. *Environmental Science Technology*, 32(24):3485–3852.

Rogge, W. F., Hildemann, L. M., Mazurek, M. A., Cass, G. R., and Simonelt, B. R. T. (1993). Sources of fine organic aerosols. 3. road dust, tyre debris, and organometallic brake lining dust: Roads as sources and sinks. *Environmental Science and Technology*, 27(9):1892–1904.

Rosen, H. and Novakov, T. (1977). Raman scattering ad the characterisation of atmospheric aerosol particles. *Nature*, 266:708–710.

Ruellan, S. and Cachier, H. (2001). Characterisation of fresh particulate vehicular exhausts near a Paris high flow road. *Atmospheric Environment*, 35:453–468.

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

Samet, J. M., , Dominici, F., and Curriero, F. C. (2000a). Fine particulate air pollution and mortality in 20 US cities, 1987-1994. *New England Journal of Medicine*, 343:1742–1749.

Samet, J. M., Zeger, S. L., and Dominici, F. (2000b). The national morbidity, mortality, and air pollution study (NMMAPS). Part 2. Morbidity and mortality from air pollution in the United States. Research Report No. 94, Health Effects Institute, Cambridge, Massachusetts, USA.

Sanhueza, E. (2001). Hydrochloric acid from chlorocarbons: a significant global source of background rain acidity. *Tellus*, 53B:122–132.

Sapkota, B. C. (2002). Suspended matter in the urban air of Kathmandu Valley. In *Proceedings of the Regional Workshop on Better Air Quality in Asian and Pacific Rim Cities, 16-18 December 2002 (BAQ 2002)*, Hong Kong Convention and Exhibition Centre (HKCEC), Hong Kong. The Hong Kong Polytechnic University. http://www.cse.polyu.edu.hk/~activi/BAQ2002/presentations.htm (last accessed: 19 November 2005).

Sawicki, J., Johnson, H., Belanger, B., Lathlin, O., Newman, D., Anderson, D., Jardine, K., Baker, M., MacAdam, K., Langdon, O., Noel, W., Eftoda, D., Handley, J., and Kilabuk, P. (2000). Canada-wide standards for particulate matter (pm) and ozone. Technical report, Canadian Council of Ministers of the Environment, Quebec City. http://www.ccme.ca/assets/pdf/pmozone_standard_e.pdf (last accessed: 03 November 2005).

Saxena, V. K. (1997). Impact of stratospheric volcanic aerosols on climate: evidence for aerosol short-wave and long-wave forcing in the southeast US. *Atmospheric Environment*, 31:4211.

Schlesinger, R. B. and Cassee, F. (2003). Atmospheric secondary inorganic particulate matter: The toxicological perspective as a basis for health effects risk assessment. *Inhalation Toxicology*, 15:197–235.

Schneider, J., Spangl, W., Lorbeer, G., Trimbacher, C., and Nagl, C. (2004). Analysis of different PM fractions in Vienna and Illmitz in 2000/01. In *Proceedings of the International Conference on Particles in the size of 2.5 to 10 microns in urban areas, 4-6 November 2002*, Berlin, Germany. Institute for Meteorology, Free University and the Brandenburg Technical University Cottbus. http://secus.met.fu-berlin.de/veranstaltungen/Abstracts%20PM10/Nagl.htm (last accessed: 6 April 2005).

Schulze, E. (2001). Communication from Ronald Evelyn Schulze e-mail dated November 27, 2001 to Mr. Bill Kuykendal, US EPA, Research Triangle Park, NC 27711. http://www.epa.gov/ttn/chief/ap42/ch13/index.html (last accessed: 13 April 2005).

Schuring, D. J. and Clark, J. D. (1988). Load, speed, and pressure effects on passenger car tire rolling-loss distribution. *Rubber Chemistry and Technology*, 61:669–687.

Schwartz, J., Dockery, D. W., and Neas, L. M. (1996). Is daily mortality associated specifically with fine particles? *Journal of the Air and Waste Management Association*, 46:927–939.

Seaton, A., MacNee, W., Donaldson, K., and Godden, D. (1995). Particulate air pollution and acute health effects. *The Lancet*, 345:176–178.

Sehmel, G. A. (1976a). Airborne ^{238}Pu and ^{239}Pu associated with the larger than 'respirable' resuspended particles at Rocky Flats during July 1973. Technical Report BNWL-2119, Pacific Northwest Laboratory, Richland, Washington.

Sehmel, G. A. (1976b). Particle resuspension from an asphalt road caused by car and truck traffic. In *Proceedings of the Atmosphere-Surface Exchange of Particulate and Gaseous Pollutants, CONF 740921, 4-6 September*, pages 859–883, Energy Res. and Dev. Admin. Symp. Ser., Richland, Washington. Nat. Tech. Inf. Serv., Springfield, Virginia.

Sehmel, G. A. (1980a). Particle and gas dry deposition: a review. *Atmospheric Environment*, 14:983–1101.

Sehmel, G. A. (1980b). Particle resuspension: a review. *Environment International*, 4:107–127.

Sehmel, G. A. (1983). Resuspension rates from aged inert tracer sources. In Pruppacher, H. R., Semonin, R. G., and Slinn, W. G. N., editors, *Precipitation Scavenging , Dry Deposition and Resuspension*, volume 2, pages 1073–1086. Elsevier, Amsterdam.

Sehmel, G. A. (1984). Deposition and resuspension. In Randerson, D., editor, *Atmospheric Science and Power Production*, pages 533–583. US Department of Energy Tech. Inf. Centre.

Sehmel, G. H. (1973). Particle resuspension from an asphalt road caused by car and truck traffic. *Atmospheric Environment*, 7:291–309.

Seinfeld, J. H. and Pandis, S. N. (1998). *Atmospheric Chemistry and Physics: From Air Pollution to Climate Change.* John Wiley & Sons, Inc., New York.

Seton, Johnson & Odell, I. (1983a). Appendix to Final Report, Portland road dust demonstration project. Technical report, Department of Public Works, City of Portland, OR.

Seton, Johnson & Odell, I. (1983b). Final Report, Portland road dust demonstration project. Technical report, Department of Public Works, City of Portland, OR.

Shi, J. P., Evans, D. E., Khan, A. A., and Harrison, R. M. (2001). Sources and concentrations of nanoparticles (< 10 nm diameter) in the urban atmosphere. *Atmospheric Environment*, 35:1193–1202.

Siddiqi, A. A. and Woeley, Jr., F. L. (1977). Urban and industrial air pollution in Houston, Texas - I. hydrocarbons. *Atmospheric Environment*, 11:131–143.

Sieker, F. and Grottker, M. (1988). Beschaffenheit von straßenoberflächenwasser bei mittlerer verkehrsbelastung. Technical Report Heft 530, Rorschung Straßenbau und Straßenverkehrstechnik, Bundesminister für Verkehr, Abt. Straßenbau, Bonn-Bad Godesberg.

Simulistics (2004). *Simile.* the Simulistics Ltd., Edinburgh Technology Transfer Centre, King's Buildings, Edinburgh, Scotland EH9 3JL. http://simulistics.com (last accessed: 24 March 2004).

Slinn, W. G. N. (1978). Parameterisations for resuspension and for wet and dry deposition of particles and gases for use in radiation dose calculations. *Nuclear Safety*, 19:205–219.

Smith, S., Stribley, T., Barratt, B., and Perryman, C. (1997). Determination of partisol, TEOM, ACCU and cascade impactor instruments in the London Borough of Greenwich. *Clean Air*, 27:70–73.

Smith, W. H. (1970). Salt contamination of white pine planted adjacent to an interstate highway. *Plant Disease Reporter*, 54(12):1021–1025.

Spix, C., Anderson, H. R., Schwartz, J., Vigotti, M. A., LeTertre, A., Vonk, J. M., Touloumi, G., Balducci, F., Piekarski, T., Bacharova, L., Tobias, A., Ponka, A., and Katsouyanni, K. (1998). Short-term effects of air pollution on hospital admissions of respiratory disease in Europe: a quantitative summary of APHEA study results. *Archives of Environmental Health*, 53:54–64.

Stanger (2005). Site description in the Automatic Urban and Rural Monitoring Network: Site hosted and maintained by casella stanger on behalf of the department for the environment, food and rural affairs. http://www.stanger.co.uk/siteinfo/classification.htm (last accessed: 6 August 2005).

Stern, A. C. (1962). *Air Pollution*, volume VIII. Academic Press, Inc., New York.

Sternbeck, J., Sjödin, A., and Andréasson, K. (2002). Metal emissions from road traffic and the influence of resuspension - results from two tunnels studies. *Atmospheric Environment*, 36:4735–4744.

Stewart, K. (1967). The resuspension of particulate material from surfaces. In Fish, B. R., editor, *Surface Contamination - The proceedings of the symposium at Gatlinburg, Tennessee, June 1964*, pages 63–74. Pergamon Press, Oxford.

Sturm, P. J., Pucher, K., Sudy, C., and Almbauer, R. A. (1996). Determination of traffic emissions - intercomparison of different calculation methods. *The Science of the Total Environment*, 189/190:187–196.

Subramani, J. P. (1971). *Particulate air pollution from automobile tyre wear.* PhD thesis, University of Cincinnati.

Takano, H., Yoshikawa, T., Ichinose, T., Miyabara, Y., Imaoka, K., and Sagai, M. (1997). Diesel exhaust particles enhance antigen-induced airway inflammation and local cytokine expression in mice. *American Journal of Respiratory and Critical Care Medicine*, 156:36–42.

Tate, J. E. and Bell, M. C. (2002). Network monitoring, modelling and management to aid mitigate the impact of "event" traffic. In *Proceedings of the 11th International Conference on Road Traffic Monitoring and Control.* IEE Publication.

Thiessen, K. M., Thorne, M. C., Maul, P. R., Pröhl, G., and Wheater, H. S. (1999). Modelling radionuclide distribution and transport in the environment. *Environmental Pollution*, 100:151–177.

Thompson, R. N., Nau, C. A., and Lawrence, C. H. (1966). Identification of vehicle tyre rubber in roadway dust. *American Industrial Hygiene Association Journal*, 27:488–495.

Thurston, G. D. and Spengler, J. D. (1985). A quantitative assessment of source contributions to inhalable particulate matter pollution in metopolitan Boston. *Atmospheric Environment*, 19(1):9–25.

TNO (1997). Particulate matter emissions (PM_{10},$PM_{2.5}$ and $PM_{0.1}$) in Europe in 1990 and 1993. Technical report, TNO Institute of Environmental Sciences, Energy Research and Process Innovation, Apeldoorn, The Netherlands.

Triantafyllou, A. G., Kiros, E. S., and Evagelopoulos, V. G. (2002). Respirable particulate matter at an urban and nearby industrial location: concentrations and variability

and synoptic weather conditions during high pollution episodes. *Journal of the Air and Waste Management Association*, 52:287–296.

TSI (2004). *Model 8525 P-Trak Ultrafine Particle Counter: Operation and Service manual, P/N 1980380 Rev G.* TSI Incorporated, 500 Cardigan Road, Shoreview, MN 55126, USA. http://www.tsi.com (last accessed: 07 October 2004).

Turner, D. B. (1970). *Workbook of Atmospheric Dispersion Estimates.* Office of Air Programs, USEPA, Research triangle Park, NC.

USEPA (1993a). Appendix C.1: Procedures for sampling surface/bulk dust loading. Technical report, US Environmental Protection Agency, US EPA, Office of Air Quality Planning and Standards (OAQPS), Info CHIEF Help Desk, Mail Code D205-01, Research Triangle Park, NC 27711. http://www.epa.gov/ttn/chief/comments.html (last accessed: 06 October 2005). http://www.epa.gov/ttn/chief/ap42/appendix/app-c1.pdf (last accessed: 06 October 2005).

USEPA (1993b). Appendix C.2: Procedures for laboratory analysis of surface/bulk dust loading samples. Technical report, US Environmental Protection Agency, US EPA, Office of Air Quality Planning and Standards (OAQPS), Info CHIEF Help Desk, Mail Code D205-01, Research Triangle Park, NC 27711. http://www.epa.gov/ttn/chief/comments.html (last accessed: 06 October 2005). http://www.epa.gov/ttn/chief/ap42/appendix/app-c2.pdf (last accessed: 06 October 2005).

USEPA (1993c). Emission factor documentation for AP-42, Section 13.2.1. Paved Roads. EPA contact No. 68-D0-0123, Work assignment No. 44, MRI Project No. 9712-44. Technical report, US Environmental Protection Agency, US EPA, Office of Air Quality Planning and Standards (OAQPS), Info CHIEF Help Desk, Mail Code D205-01, Research Triangle Park, NC 27711. http://www.epa.gov/ttn/chief/comments.html (last accessed: 06 October 2005). http://www.epa.gov/ttn/chief/ap42/ch13/index.html (last accessed: 06 October 2005).

USEPA (1995). Introduction to AP-42, Volume I, Fifth Edition – January 1995. Technical Report Volume I, Fifth Edition, US Environmental Protection Agency, US

EPA, Office of Air Quality Planning and Standards (OAQPS), Info CHIEF Help Desk, Mail Code D205-01, Research Triangle Park, NC 27711. http://www.epa.gov/ttn/chief/comments.html (last accessed: 06 October 2005). http://www.epa.gov/ttn/chief/ap42/ch13/index.html (last accessed: 06 October 2005).

USEPA (1997). Addendum to Emission factor documentation for AP-42, Section 13.2.1. Paved Roads. Final Report. EPA contact No. 68-D2-0159, Work assignment No. 4-02, MRI Project No. 4604-02. Technical report, US Environmental Protection Agency, US EPA, Office of Air Quality Planning and Standards (OAQPS), Info CHIEF Help Desk, Mail Code D205-01, Research Triangle Park, NC 27711. http://www.epa.gov/ttn/chief/comments.html (last accessed: 06 October 2005). http://www.epa.gov/ttn/chief/ap42/ch13/index.html (last accessed: 06 October 2005).

USEPA (2002). AP-42, Section 13.2.1. Paved Roads. Final Report. Technical report, US Environmental Protection Agency, US EPA, Office of Air Quality Planning and Standards (OAQPS), Info CHIEF Help Desk, Mail Code D205-01, Research Triangle Park, NC 27711. http://www.epa.gov/ttn/chief/comments.html (last accessed: 06 October 2005). http://www.epa.gov/ttn/chief/ap42/ch13/final/c13s021.pdf (last accessed: 06 October 2005).

USEPA (2003). AP-42, Section 13.2.1. Paved Roads. Final Report. Technical report, US Environmental Protection Agency, US EPA, Office of Air Quality Planning and Standards (OAQPS), Info CHIEF Help Desk, Mail Code D205-01, Research Triangle Park, NC 27711. http://www.epa.gov/ttn/chief/comments.html (last accessed: 06 October 2005). http://www.epa.gov/ttn/chief/ap42/ch13/final/c13s0201.pdf (last accessed: 06 October 2005).

van der Gon, H. D. and Visschedijk, A. (2004). Source contributions to particulate matter in the size range of 2.5 - 10 microns in the Netherlands with special emphasis on sea salt and crustal material. In *Proceedings of the International Conference on Particles in the size of 2.5 to 10 microns in urban areas, 4-6 November 2002*, Berlin (Germany). Institute for Meteorology, Free University and the Brandenburg Tech-

nical University Cottbus. http://secus.met.fu-berlin.de/veranstaltungen/Abstracts%20PM10/Denier.pdf (last accessed: 6 April 2005).

van Wijnen, J. H. and van der Zee, S. C. (1998). Traffic-related air pollutants: exposure of road users and populations near busy roads. *Review of Environmental Health*, 13:1–25.

Vedal, S. (1995). Health effects of inhalable particles: Implications for British Coulmbia. Technical report, Water, Air and Climate Change Branch (Formerly known as Air Resources Branch), Ministry of Environment, Lands and Parks. http://www.env.gov.bc.ca/ske/skeair/pm10/pm10rx.html,http://wlapwww.gov.bc.ca/air/particulates/heoipifb.html (last accessed: 5 May 2005).

Vedal, S. (1997). Ambient particle and health: lines that divide. *Journal of the Air and Waste Management Association*, 47:551–581.

Veith, A. G. (1992). A review of important factors affecting treadwear. *Rubber Reviews*, 65:601–658.

Veith, A. G. (1995). Tyre tread wear - the joint influence of compound properties and environmental factors. *Tyre Science and Technology*, 23(4):212–237.

Venkatram, A. (2000). A critique of empirical emission factor models: A case study of the AP-42 model for estimating PM_{10} emission from paved roads. *Atmospheric Environment*, 34:1–11.

Venkatram, A. (2001). Response to comments by Nicholson. A critique of empirical emission factor models: A case study of the AP-42 model for estimating PM_{10} emission from paved roads (Atmospheric Environment 34, 1-11). *Atmospheric Environment*, 35:187.

Venkatram, A. and Fitz, D. (1998). Measurement and modelling of PM_{10} and $PM_{2.5}$ emissions from paved roads in California. Final report Contract 94-336, California Air Resources Board.

Venkatram, A., Fitz, D., Bumiller, K., Du, S., Boeck, M., and Ganguly, C. (1999). Using a dispersion model to estimate emission rates of particulate matter from paved roads. *Atmospheric Environment*, 33:1093–1102.

Vensim (2003). *Vensim, Version 5.2.* Ventana Systems, Inc., 60 Jacob Gates Road, Harvard, MA 01451, USA. http://www.vensim.com (last accessed: 24 March 2004).

Vermette, S. J., Williams, A. L., and Landsberger, S. (1992). PM_{10} source apportionment using local surface dust profiles: examples from Chicago. In Chow, J. C. and Ono, D. M., editors, *Proceedings of the PM_{10} Standards and Non-traditional Particulate Source Controls*, volume I, pages 262–278. Air and Waste Management Association, Pittsburgh, Pennsylvania.

Wainwright, J. and Mulligan, M. (2004). Modelling and model building. In Wainwright, J. and Mulligan, M., editors, *Environmental Modelling: Finding Simplicity in Complexity*, pages 7–73. John Wiley & Sons, Ltd.

Watson, J. G., Chow, J. C., Zhiqiang, L., Fujita, E. M., Lowenthal, D. H., and Lawson, D. R. (1994). Chemical mass balance source apportionment of PM_{10} during the Southern California air quality study. *Aerosol Science and Technology*, 21:1–36.

Wei, F., Teng, E., Wu, G., Hu, W., Wilson, W. E., Chapman, R. S., Pau, J. C., and Zhang, J. (1999). Ambient concentrations and elemental compositions of PM_{10} and $PM_{2.5}$ in four Chinese cities. *Environmental Science and Technology*, 33:4188–4193.

Weigand, G., Diegmann, V., and Pfäfflin, F. (2004). Statistical examination of the relationship between PM_{10} emissions and road traffic. In *Proceedings of the International Conference on Particles in the size of 2.5 to 10 microns in urban areas, 4-6 November 2002*, Berlin, Germany. Institute for Meteorology, Free University and the Brandenburg Technical University Cottbus. http://secus.met.fu-berlin.de/veranstaltungen/Abstracts%20PM10/Weigand.pdf;www.IVU-Umwelt.de (last accessed: 6 April 2005).

Weiland, S. K., Mundt, K. A., Rueckmann, A., and Keil, U. (1994). Self-reporting

wheezing and allergic rhinitis in children and traffic density on street of residence. *Annals of Epidemiology*, 4:243–247.

Whicker, F. W., Shaw, G., Voigt, G., and Holm, E. (1999). Radioactive contamination: state of the science and its application to predictive models. *Environmental Pollution*, 100:133–149.

WHO (2003). Health aspects of air pollution with particulate matter, ozone and nitrogen dioxide. Report on a WHO working group, Bonn, Germany, 13-15 January 2003. Technical report, World Health Organisation, Regional office for Europe, Copenhagen.

Wichmann, H. E. and Peters, A. (2000). Epidemiological evidence of the effects of ultrafine particle exposure. *Philoshopical Transaction of the Royal Society of London*, A358:2751–2768.

Wilson, R. and Spengler, J. (1996). *Particles in our air: concentrations and health effects*. Harvard University Press, Boston.

Wilson, W. E. and Suh, H. H. (1997). Fine particles and coarse particles: concentration relationships relevant to epidemiological studies. *Journal of the Air and Waste Management Association*, 47:1238–1249.

Wjst, M., Reitmeir, P., and Dold, S. (1993). Road traffic and adverse effects on respiratory health in children. *British Medical Journal*, 307:596–600.

Wyzga, E. R. (2002). Air pollution and health: Are particulates the answer? In $PM_{2.5}$ *and electric power generation: Recent findings and implications - Proceedings of the National Energy Technology Laboratory (NETL) conference, 9-10 April 2002*, Pittsburg, Pennsylvania, USA. http://www.netl.doe.gov/publications/proceedings/02/PM25/ (last accessed: 26 August 2005).

Xuan, J. (1999). Dust emission factors for environment of Northern China. *Atmospheric Environment*, 33:1767–1776.

Zimmer, R. A., Reeser, W. K., and Cummins, P. (1992). Evaluation of PM_{10} emission factors for paved roads. In Chow, J. C. and Ono, D. M., editors, *Proceedings of the PM_{10} Standards and Non-traditional Particulate Source Controls*, volume I, pages 311–323. Air and Waste Management Association, Pittsburgh, Pennsylvania.

Publication by the author during thesis research

Patra, A., Bowen, E., Colvile, R., Arnold, S., Martin, D., Price, C., Shallcross, D., Tate, J., Dobre, A., Robins, A., 2005a. Movement and dispersion of particulate matter due to traffic on road surface. In Sokhi, R. S., Millan, M. M. and Moussiopoulos, N., editors, *Proceedings of the 5th International Conference on Urban Air Quality*, 29-31 March 2005, Valencia (Spain): University of Hertfordshire, UK; FUNDACION CEAM, Spain and Aristotle University, Greece. ISBN No: 1-898543-92-5 `http://www.urbanairquality.org/UAQ2005`

Patra, A., Colvile, R., Arnold, S., Bowen, E., Tate, J., Robins, A., Martin, D., Price, C., and Shallcross, D. (2005b). Movement and dispersion of particulate matter due to traffic on road surface: Interim Report to the Greater London Authority. Technical report, Imperial College London, Environmental Processes and Systems Research Group, Department of Environmental Science and Technology, South Kensington, London SW7 2AZ.

Patra, A., Colvile, R., Arnold, S., Bowen, E., Tate, J., Robins, A., Martin, D., Price, C., and Shallcross, D. (2005c). Movement and dispersion of particulate matter due to traffic on road surface: Final Report to the Greater London Authority. Technical report, Imperial College London, Environmental Processes and Systems Research Group, Department of Environmental Science and Technology, South Kensington, London SW7 2AZ.

Patra, A., Colvile, R., Arnold, S., Bowen, E., Tate, J., Martin, D., Price, C., Shallcross, D., ApSimon, H., and Robins, A. (2005d). Particulate matter movement and dispersion due to traffic on an urban road I: on-street observations. *Atmospheric Environment.* Submitted.

Patra, A., Colvile, R., Martin, D., and Shallcross, D. (2005e). Particulate matter movement and dispersion due to traffic on an urban road II: quantitative parameterisation. *Atmospheric Environment.* Submitted.