CYCLIST EXPOSURE TO TRAFFIC POLLUTION: MICROSCALE VARIANCE, THE IMPACT OF ROUTE CHOICE AND COMPARISONS TO OTHER MODAL CHOICES IN TWO NEW ZEALAND CITIES

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Frontispiece

Sampling at the bottom of Queen Street, Auckland city, April 2009

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Abstract

This study aimed to investigate various aspects of cyclist exposure to common urban air pollutants, including CO, PM_{10}, $PM_{2.5}$, $PM_{1.0}$ and UFPs. The initial part of the study compared cyclist exposure to that of other transport modes, while the second part addressed the implications of route choice. The final part analysed the effect of proximity to traffic.

Data was collected in Christchurch and Auckland cities over a nine week period, with a total of 53 inter-modal and 7 separate cyclist sampling runs completed. Mobile sampling was conducted using a collection of instruments in four portable kits. Fixed-site meteorological data was used to find associations between pollutants and temperature and wind speed. Spatial patterns were also considered by means of time-series comparative graphs and colour-coded pollutant concentration GPS mapping.

The cyclist mode was up to 61% less exposed than the car for primary pollutants (CO and UFPs), but up to 26% more exposed for $PM_{1.0\text{-}10}$. The bus was generally the most exposed for all pollutants apart from CO. The effect of route choice was substantial, with the off-road cyclist route recording a reduction of 31% for CO and $PM_{1.0}$, and 53% for UFPs while PM_{10} was 6%. At a distance of 7 m from traffic, exposure dropped by 30% (UFPs), 22% (CO) and 14% ($PM_{2.5}$). At 19 m, concentrations decreased a further 17%, 13% and 8%, respectively. When moving much further away from traffic (~700 m), the effect was far less pronounced and no difference was observed for CO past 19 m.

Conclusions suggest that for most pollutants studied, the cyclist mode faces much lower exposure than other modes, especially when traveling through backstreets and cycle tracks. Significant exposure reductions can also be made when only a very small distance away from traffic emissions. This has positive implications for health, sustainable city planning and active-mode transport promotion.

Chapter One: Introduction

1.1 Introduction

Urban air pollution, primarily due to the burning of fuels for heating and industry, has been a problem ever since cities existed. Intermittent attempts at regulation occurred up until the mid twentieth century, when the Great London Smog of 1952 led to the introduction of the Clean Air Act of 1956 (Phalen 2002). Over the next 30 years, most industrialised nations increased efforts to control private and industrial emissions by means of policy and legislation. The exponential growth of the human population, subsequent demand for material goods and an increasing reliance on fossil fuels slowly worsened air quality to the point of global concern, at least within many developed nations. The most notable early report in recognition of environmental problems as a global issue, was the Brundtland Report of 1987 in which the term 'sustainable development' was coined (Bruntland 1987). This was the beginning of an era of discussing ways of attempting to tackle global environmental degradation collectively. Since the Brundtland report, various pivotal reports, summits and international agreements have followed. The Rio Earth Summit of 1992 led the development of the Kyoto protocol, in which 37 industrialised countries agreed to commit to reducing several key primary and secondary greenhouse gases. The Kyoto protocol has been heavily criticised around the globe, with notable economists labelling it as a political and technical failure (Prins & Rayner 2008).

Transport contributes to 14% of all global greenhouse emissions, with 72% of that portion made up of vehicular road transport (IEA 2006). In the US alone, 33% of carbon dioxide emissions originate from the transport sector, predicted to rise another 3% by 2020 (Greene & Schafer 2003). Curtailing emissions from vehicles through use of sustainable city transport and alternative modes is now a pressing issue for many local and state governments. The relative unwillingness of national governments to adopt radical policies and allocate funding to sustainable transport initiatives has left actual measurable change largely in the hands of local governmental bodies.

Reducing urban pollution is not only a paramount step as part of the ultimate goal of slowing environmental degradation; it is a vital necessity for protecting human health. Traffic generated emissions are responsible for more deaths than traffic accidents, in many major metropolises, including London, New York and Sydney. The New Zealand situation is not far behind the trend, with 399 pollutant associated premature mortalities per annum, compared with 502 by motor accident (Fisher et al. 2002). Reducing pollutant related mortalities can only be achieved through a reduction of emissions, cleaner fuel technologies (reduced toxicity) or using alternative completely alternative options; alternative industrial production methods, sustainable transport systems and active mode transport. Firstly, the New Zealand situation requires adequate personal exposure research to get up speed with international literature.

1.2 Personal pollution exposure: prior investigation and thesis rationale

Much of the earlier pollution research has relied on data from fixed site monitors which has been extrapolated across wider city areas. While sufficient for informing emissions guidelines and policies, fixed site methods often result in the underestimation of concentrations for some areas (Gulliver & Briggs 2004). It has also been shown that background and kerbside monitoring stations provide poor indications of personal exposure (Gulliver & Briggs 2004; Kaur et al. 2005a). Although the spatial distribution of some pollutants (especially small non-reactive particles) can be relatively uniform, concentrations fluctuate substantially, with levels generally highest closest to their source. Therefore, only direct personal exposure assessment can provide accurate measures of exposure while traveling.

There is now a wealth of published international research which has focused on personal journey time exposure. Results vary substantially, with the highest levels found in large cities in underdeveloped nations (Lindén et al. 2008; Saksena et al. 2008; Wöhrnschimmel et al. 2008). Significant differences also occur between transport modes. While the relative ratio between modes provides an idea of the general picture across studies, there are many exceptions to the more common conclusions. Different geographical settings (ambient sources, traffic density), instrumentation, methodologies and sampling conditions inevitably result in conflicting findings. It is for these reasons that overseas data cannot be relied on for informing health promotion and policy at the local level.

While fixed site monitoring has long been in place in New Zealand cities, currently, no study investigating personal pollution exposure while traveling exists. This provides a major research gap which needs to be addressed to see how the situation compares to overseas cities. Decisions can then be made on the usefulness for informing policy and promoting healthy transport decisions.

1.3 Research context

New Zealand has a relatively small population for its area, with approximately 4.3 million inhabitants and a population density of 16 persons per km^2. Public transport patronage is low, with the number of people traveling to work by private vehicle increasing from 64.8% in 1976 to 83% in 2006. This change is reflected in pubic transport use, dropping from 12.8% to 5% over the same period (Tin Tin et al. 2009). In Australia, national patronage is at 12% and is as high as 18% in New South Wales, where electric rail is the most popular means of public travel (Australian Bureau of Statistics 2007). Larger cities are afforded the luxury of having the population to support extensive public transport networks and are often well inter-connected to other cities by rail e.g. Europe, Australia and the UK. Due to a combination of a lack of demand and a lack of public spending, only two New Zealand cities have a commuter rail service (Auckland and Wellington) and inter-city services are limited as well as very expensive. All other cities are serviced by bus.

The situation for commuter cyclists could be greatly improved. Cycle lanes typically exist in the form of painted broken lines on the roadside but there are still many major streets and roads where no lanes are present. While no safety barrier exists between cyclists and traffic, visibility has been improved in recent years by fully painting lanes and improving signage to create greater drive awareness. Even so, the general perception that cycling is too dangerous remains the key barrier to preventing greater uptake for commuting purposes (Taylor et al. 2009).

New Zealand cities and towns are characteristic of typical colonial planning, with large areas dedicated to parkland and grid-based street configuration. The wide nature of colonial streets and abundant parkland provides ample opportunity for safe and efficient active mode networks, especially in areas of flat topography. In most cases, the possible options for greatly improved active-mode routes are well under-utilised. Many European cities, especially in The Netherlands and Germany, have world-class cycle infrastructure where cycling is safer, more direct and often quicker than driving. Segregated cycle ways, shared cyclist/pedestrian paths, cyclist give-way priority, large parking facilities and cyclist/pedestrian only city centres are some of the key features necessary for effective networks. While some progress is being made in New Zealand, more funding and faster implementation would likely see a greater uptake at a much faster rate.

The general public's perception is not only that cycling is unsafe, but one is also exposed to more pollution while cycling. This may well be a misconception, at least for certain pollutants. Hence there is a need to clarify the situation so that the public can be better informed. In addition to finding out how cyclists' exposure measures up to that of other modes, there is scope for addressing more detailed questions about exposure while cycling. Segregated cycle ways provide the opportunity to measure exposure at different distances from the traffic. The effect of proximity to traffic at the microscale level has never been investigated for cyclists anywhere in the world. While some studies have considered the effect of taking backstreets and parkland tracks, it is uncertain if any difference occurs between on-road exposure and levels experienced just several metres from traffic. As New Zealand also lacks research on the effect of route choice, two major cycling research gaps exist.

Improving alternative mode infrastructure will not only help improve air quality but also improve population health and fitness. Similar to the UK, USA and Australia, New Zealand, has serious problems with obesity, with prevalence now at 25% for the population aged 15 years and above (Ministry of Social Development 2009). Not surprisingly, research shows there is an inverse correlation between cycling, walking or using alternative transport to commute, and being overweight (Ming Wen & Rissel 2008). Health problems arising from limited physical activity are not confined to people who are overweight. Those who work in sedentary jobs and are relatively inactive have an increased risk of premature death from cancers and other serious ailments (Manson et al. 2004). Furthermore, the number of road accidents are increasing, with the health and social costs to New Zealand exceeding $200 million in 2005 (Billante 2008). Improving alternative transport infrastructure has long-lasting benefits that can positively affect the environment, individuals and society as a whole. Cyclist and pedestrian infrastructure is relatively cheap compared with other alternative transport solutions, making it seemingly a more logical area of investment. Research that supports and builds-on active mode transport should be encouraged and greater efforts made to shift results into policy.

1.4 Objectives and thesis structure

In consideration of the research context and identified research gaps, this thesis has four key objectives:

1. To ascertain how cyclists' personal pollutant exposure compares to levels experienced by other transport commuters, for key traffic pollutants including carbon monoxide and several sizes of particulate matter.

2. To investigate the effect of cyclist route choice – the difference between on-road cyclist exposure, backstreet exposure and off-road cycle path exposure along routes within the same vicinity.

3. To determine the degree to which, if any, cyclist pollutant exposure variance exists between on-road exposure and that experienced approximately 5 and 15 metres away from traffic flows.

4. To compare results to previous literature and discuss any implications of the results for the New Zealand situation.

This thesis continues by first defining pollutants associated with traffic emissions, looking at some of the associated health problems and then presenting an in-depth synthesis of findings from previous personal exposure studies (Chapter Two). Chapter Three outlines the methods employed in the study. Chapters Four and Five are the results chapters in which discussion elements are also included. Chapter Four solely focuses on the results from inter-modal comparisons while Chapter Five covers all the cycling research; including the effect of route choice and the effect of proximity to traffic. The conclusions chapter (Chapter Six) presents separate conclusions for both parts of the study, along with limitations and suggestions for further research.

1.5 Summary

This chapter has drawn attention to some of the main issues surrounding personal pollution exposure, why such research is important and how it fits into other pressing issues such as human health, sustainability and climate change. Most importantly, this chapter has outlined the New Zealand context and provided clear and logical research gaps that legitimately warrant enquiry. The main aim of this thesis is to provide a key piece of research that can be drawn on for policy-related reports and future academic enquiry.

Chapter Two: Literature Review

2.1 Introduction

Approximately half of the world's population currently resides in urban centres and the percentage living in rural areas is projected to decline as cities swell into mega-metropolises (Cohen 2003). Cities are home to a raft of social and environmental problems and air pollution is a key issue due to the detrimental effects on human health. Although urban pollution originates from a variety of sources, in most urban areas the majority comes from transport emissions. High air pollution exposure has been linked to increased allergies, respiratory illnesses, birth defects and numerous forms of cancer (Brunekreef & Holgate 2002).

Major pollutants associated with vehicle emissions include: particulate matter (expressed as PM_{10}, $PM_{2.5}$, $PM_{1.0}$ and UFPs [ultrafine particles $<=0.1$ μm]), black carbon, volatile organic compounds (common VOCs found in petrol include benzene, toluene, ethylbenzene and xylenes [BTEX]), polycyclic aromatic hydrocarbons (PAHs), ozone (O_3), carbon monoxide (CO), carbon dioxide (CO_2), sulfur dioxide (SO_2) and mono-nitrogen oxides (NO_x), which consist of nitric oxide (NO) and nitrogen dioxide (NO_2). PAHs are chemicals present in particulate matter and these compounds are primarily responsible for PM toxicity and accompanying adverse health effects (de Kok et al. 2006). Vehicular traffic also produces non-tail pipe pollutants such as material from clutch, brake and tyre abrasion, which include many heavy metals (Wahlin et al. 2006).

Given the growing environmental and health concerns, a substantial body of literature has been produced on both ambient pollutant recordings and personal exposure data. Much of the work on personal exposure in the past decade has tended to focus on journey-time exposure while traveling. Research from the United States shows that 60% of a persons total daily pollution exposure is attributable to their daily commute (Hill & Gooch 2007). Therefore it is important commuters be made aware of ways they can reduce personal exposure by changing transport modes or using different routes. Such information is highly useful for public health campaigns and in city planning.

This review synthesises findings from most of the transport exposure literature of the past fifteen years. Some of these studies have compared exposure for different transport modes. The main objective of this review is to investigate which modes are subject to the highest concentrations of key pollutants. Different variables affecting exposure are then summarised, followed by a brief overview of known health implications.

2.2 Effect of transport mode on personal exposure levels

This section first provides summarised results tables of most previous studies for the five main pollutants of interest. Table 1 outlines CO results. Tables 2, 3, 4 and 5 present the four main PM fractions: PM_{10}, $PM_{2.5}$, $PM_{1.0}$ and UFPs, respectively. Findings are then discussed in order of the most commonly used modes transport modes.

Table 1 Modal Studies - CO Summarised Results

Author	Location	Instrument/s	Mode	Vehicle type/make	Window and ventilation settings	Sampling duration	Sampling setting	Number of samples	Mean values (ppm unless specified)
Bevan et al. (1991)	Southampton, UK	Neotronics sampling pump	Bicycle			~35 min	Suburban commute Urban commercial Parkland	16 16 16	10.5 4.5 0.8
Chan et al. (1991)	Raleigh, NC, USA	Interscan 4146	Car	1983/87 Mercury four door sedan	Windows & vents closed, A/C on Windows closed, vent fan on, A/C off Front windows half opened, vent fan on, A/C off	1 hr	Urban Interstate beltway Rural	30 34 6	13 11 4
Koushki et al. (1992)	Riyadh, Saudia Arabia	Ecolyser series 2000	Car	Unknown	Unknown	25 min (avg)	Main arterial roadways	634	31.3
Limasset et al. (1993)	Paris, France Bordeaux, France	Ecolyser	Bus Bus	Unspecified	Unspecified	1.5-2 hrs 35-60 min	City centre City centre	18 24	10, 11.3 5.8, 8.1, 12.5
Liu et al. (1994)	Taipei, Taiwan	Horiba APMA-350E	Car Bus Motorcycle	Unspecified	Not controlled for Actual settings unspecified	30-50 min	Major city transportation mains	30 94 295	11.6 11.0 17.5
Ott et al. (1994)	Menlo Park Palo Alto Los Altos, CA, USA	GE Model 15ECS3CO3	Car	Unspecified	Driver's window fully open, front passenger open 3 inches, all others closed	31-61 min	Major urban arterial	88	9.8
Dor et al. (1995)	Paris, France	Draeger PAC II	Car Bus Walk Subway	Renault Express (petrol) Peugeot (electric) Renault (diesel)	Unknown	~82 min ~105 min ~82 min - - -	Urban centre Suburban Urban centre	22 30 2 - - -	12, 9,10 10,9 9,10 5 4 2
Fernandez-Bremauntz & Ashmore (1995)	Mexico City, Mexico	GE COED-1 PEM	Car Bus Trolleybus Minibus Minivan Metro system	Several small 1972-88 VWs Unspecified Electric Unspecified Unspecified Electric	Not specified	~44 min	Key city routes consisting of different traffic densities Rail/subway	43 205 53 185 44 111	55.2-57.0 20.5-41.1 21.6-31.9 31.6-64.4 39.5-66.1 16.8-26.5
Van Wijnen et al. (1995)	Amsterdam, Netherlands	Dupont pump	Car Bicycle Walk	Peugeot 205 (petrol)	Windows closed No specific ventilation settings	30 min- 1 hr 30 min-1 hr 30 min-1 hr	Inner-city, tunnel and rural	41 66 10	6730 $\mu g/m^3$ 2670 $\mu g/m^3$ 2460 $\mu g/m^3$
Clifford et al. (1997)	Nottingham, UK	Crowcon CO sensors	Van (morning) Van (morning)	Rover Maestro	Adjusted daily to suit prevailing conditions – generally, fan on at lowest setting	35 min (avg)	Key commuting routes	33 35	4.8 4.9

Table 1 Modal Studies - CO Summarised Results cont.

Rodes et al. (1998)	Sacramento, CA, USA	Draeger model 190	Car	1991 Chevrolet Caprice (sedan)	Vent on high (windows closed)	2 hrs	Freeway commute	6	2.0, 3.5
					Vent on low (windows closed)	2 hrs		4	2.2, 2.7
	Los Angeles, CA, USA	Draeger model 190	Car	1997 Ford Taurus (sedan)	Vent on high (windows closed)	2 hrs	Freeway commute	8	4.3, 4.5
					Vent on low (windows closed)	2 hrs		4	4.9, 5.3
Vellopoulou & Ashmore (1998)	Athens, Greece	Draeger model 190	Car	Unspecified	Not specified	45.5 min (avg)	Urban commute to city commercial triangle	20	16.9
			Bus					19	10.3
			Motorcycle					4	12.3
			Walk					1	6.8
Alm et al. (1999)	Kuopia, Finland	Langan T15	Car (morning)	1999 VW Jetta GL with three-way catalytic converter	Vent set to 2 (windows up)	18-24 min	Small urban town	11	5.7
			Car (afternoon)		Vents set to 2 (windows up)	18-24 min		12	3.1
Chan et al. (1999)	Hong Kong	SKC pump, Gas Filter Correlation CO Analyser	Car	Unspecified	Windows open/Windows closed	11-90 min	Various districts throughout the metropolis	25	10.1
			Bus	Unspecified		20-41 min		64	1.9
			Light bus	Unspecified	Windows open/Windows closed	14-38 min		32	2.4
			Tram	Electric		45-50 min		16	2.0
			Train	Electric	Windows open/Windows closed	42 min (avg)		16	1.0
			Subway	Electric		-		48	1.5
			Walk			-		96	1.8
Zagury et al. (2000)	Paris, France	Draeger PAC II	Taxi	Various diesel taxis	Unknown but states A/C not used due to weather conditions	8 hrs	>75% within city limits	28	3.8
Chan & Liu (2001)	Hong Kong	Interscan 4148	Taxi	LPG/diesel	A/C on	2 hrs	Urban-Urban	4	3.1
							Urban-Suburban	4	4.0
							Urban-Rural	4	2.7
			Bus	Diesel	Non-A/C	40-80 min	Urban-Urban	16	1.5
							Urban-Suburban	10	2.1
							Urban-Rural	8	1.2
			Minibus	Diesel	A/C on	18-40 min	Urban-Urban	10	2.8
							Urban-Suburban	8	3.3
							Urban-Rural	8	2.3
Chan et al. (2002b)	Guangzhou, China	Interscan 4148	Taxi	Petrol	A/C on	30 min (avg)	Urban commericial and residential zones	10	28.7
					Non-A/C	30 min (avg)		10	18.7
			Bus	Diesel	Windows closed, A/C on	49 min (avg)		20	8.9
					Non-A/C	53 min (avg)		20	8.3
			Subway	Electric	A/C on	30 min (avg)	Subway	20	3.1

Table 1 Modal Studies - CO Summarised Results cont.

Duci et al. (2003)	Athens, Greece	Solomat MPM4100 Environmental Monitoring System	Car Bus Trolley Train Walk	Unspecified Unspecified Electric Electric	Unspecified Unspecified Unspecified Unspecified	1-2 hrs 1-2 hrs 1-2 hrs 1-2 hrs 1-2 hrs	Urban *(winter)* Urban *(winter)* Urban *(summer)* Urban *(winter)* Urban *(summer)* Urban *(winter)* Urban *(summer)* Urban *(winter)* Urban *(summer)*	34 40 144 87 24 18 8 15 30	21.4 9.4 10.4 9.6 8.2 4.0 3.4 10.1 11.4
Bruinen de Bruin et al. (2004)	Milan, Italy	Langan Model T15	Car/taxi Bus/tram Train/metro Motorcycle Walking	Unspecified Unspecified Unspecified Unspecified	Unspecified Unspecified Unspecified	- - - - -	Urban commutes	207 158 57 14 241	5.7 3.8 3.0 4.5 3.0
Gómez-Perales et al. (2004)	Mexico City, Mexico	Langan T15	Bus Minibus Subway	Unspecified Unspecified Electric	Unspecified Unspecified Unspecified	2-3 hrs 2-3 hrs 2-3 hrs	Urban commute Urban commute Metro system	15 23 16	12 15 7
Mackay (2004)	Leeds, UK	Langan T15v	Car Bus Walk	Unknown Unknown	Unknown Unknown	2 hrs 3 hrs 2 hrs 3 hrs 2 hrs 3 hrs	Urban *(peak)* Urban *(off-peak)* Urban *(peak)* Urban *(off-peak)* Urban *(peak)* Urban *(off-peak)*	- - - - - -	0.85 0.95 0.59 0.51 0.37 0.47
Han et al. (2005)	Trujillo, Peru	Draeger colour stain diffusion tube Draeger Pac III	Taxi Bus Taxi Bus	Petrol Diesel Petrol Diesel	Unspecified Unspecified Unspecified Unspecified	~8 hrs ~8 hrs ~8 hrs ~8 hrs	Urban Urban Urban Urban	10 8 5 8	3.1 2.36 0.87 0.24
Kaur et al. (2005a)	London, UK	Langan T15	Car Taxi Bus Bicycle Walk	Unspecified Unspecified Unspecified	Unspecified Unspecified Unspecified	~20 min ~20 min ~20 min ~20 min ~20 min	Urban centre Urban centre Urban centre Urban centre Urban centre	13 16 27 29 26	1.3 1.1 0.8 1.1 0.9
Kaur et al. (2005b)	London, UK	Langan T15v	Walk			~20 min	Urban centre	173	1.3
Saksena et al. (2008)	Hanoi, Vietnam	Langan T15n	Car Bus Motorcycle Walk	Unspecified Diesel Unspecified	Both A/C on & A/C off (no diff) A/C on	18 min (avg) 25 min (avg) 19 min (avg) 18 min (avg)	Major arterial roads	32 16 32 16	18.5 11.5 18.6 8.5

Table 1 Modal Studies - CO Summarised Results cont.

Lindén (2008)	Ouagadougou, Burkina Faso	Langan T15v	Car	Unspecified	One window open	~1.5 hrs	Urban *(In-traffic)*	32	16
		Tpi A701	Car	Unspecified	One window open	~1.5 hrs	Urban *(Roadside)*	32	6.5
Wöhrnschimmel et al. (2008)	Mexico City, Mexico	Langan T15	Car	1999 Nissan Pickup (petrol)	Unspecified	1 hr 30 min	Urban	23	15.3-16.3
			Bus	Diesel	Unspecified	1 hr 30 min	Urban	34	11.5
			Minibus	LPG/CNG/diesel	Unspecified	1 hr 30 min	Urban	34	7.8
			Metrobus (BRT)*	Diesel	Unspecified	1 hr 30 min	Urban	57	20.3
Kaur & Nieuwenhuijsen (2009)	London, UK	Langan T15/T15v	Car	Toyota Starlet (petrol w/ three way catalyst)	Unspecified	18 min+	Urban centre	10	1.3
			Taxi	Diesel	Unspecified	18 min+	Urban centre	13	1.2
			Bus	Diesel	Unspecified	18 min+	Urban centre	19	0.8
			Bicycle			18 min+	Urban centre	14	0.9
			Walk			18 min+	Urban centre	16	0.7

*Bus Rapid Transit system (dedicated lanes)

Table 2 Modal Studies - PM10 Summary Results

Author	Location	Instrument/s	Mode	Vehicle type/make	Window and ventilation settings	Sampling duration	Sampling setting	Number of samples	Mean values ($\mu g/m^3$)
Fromme et al. (1998)	Berlin, Germany	Gravikon PM4/ Strohlein sampler	Car	1996 VW Golf with 3 way catalytic convertor	Unspecified	11 hrs	Urban *(winter)*	1	42.8
							Urban *(summer)*	1	43.5
			Subway	Electric (series F76)	Non-A/C	11 hrs	Urban *(winter)*	1	141
							Urban *(summer)*	1	153
Chan et al. (2002b)	Guangzhou, China	TSI DustTrak 8520	Taxi	Petrol	A/C on	30 min (avg)	Urban commericial and residential zones	8	88
					Non-A/C	30 min (avg)		8	140
			Bus	Diesel	Windows closed, A/C on	49 min (avg)		11	125
					Non-A/C	53 min (avg)		15	184
			Subway	Electric	A/C on	30 min (avg)	Subway	14	55
Lewné et al. (2006)	Stockholm, Sweden	Data-RAM	Taxi	Petrol/diesel	Participants free to adjust settings as desired	9 hrs 50 min	Urban/rural	39	26
			Bus	Diesel/ethanol		8 hrs 20 min	Urban/rural	42	44
			Truck	Diesel		8 hrs 35 min	Urban/rural	40	57
Branis (2006)	Prague, Czech Republic	TSI DustTrak 8520	Subway	Electric	A/C on	7 min (avg)	Urban *(winter)*	77	125.5
							Urban *(summer)*	31	82.3
			Walk			13 min (avg)	Urban *(winter)*	77	84.9
							Urban *(summer)*	31	51.4
Li et al. (2007)	Beijing, China	Dustmate sampler	Rail	Electric	A/C on	Unspecified	Rail	83	108
			Subway	Electric	Vents on (mechanical fans)		Subway	156	324.8
Gulliver & Briggs (2007)	Leicester, UK	OSIRIS/Dustmate sampler	Car	2002 Fiat Doblo (petrol)	Windows closed, vents set to 2	1 hr	Urban arterials and residential zones	33	18.2
			Walk			1 hr		33	19.1
Saksena et al. (2008)	Hanoi, Vietnam	PDR-1000	Car	Unspecified	Windows closed, A/C on	18 min (avg)	Major arterial roads	32	222
					Windows open, A/C off				595
					Overall mean (closed, A/C on)				408
			Bus	Diesel	A/C on	25 min (avg)		16	262
			Motorcycle	Unspecified		19 min (avg)		32	580
			Walk			18 min (avg)		16	495
Tsai et al. (2008)	Taipei, Taiwan	Grimm 1.108 dust monitor	Car	Petrol	Windows closed, A/C on	30 min (avg)	Urban	16	41.9
			Bus	Diesel	Windows closed, A/C on	43 min (avg)	Urban	16	70.0
			Motorcycle	Petrol		28 min (avg)	Urban	16	112.8
			Metro System	Electric	A/C on	34 min (avg)	Rail/subway	16	64.9
Gulliver & Briggs (2004)	Northampton, UK	OSIRIS sampler	Car	1995 Ford Fiesta	Windows closed, A/C off, vents off	1 hr	Main commuting routes	36	43.16
			Walk			1 hr		36	38.18
Briggs et al. (2008)	London, UK	OSIRIS sampler	Car	2001 Ford Focus (diesel)	Windows closed, vents set to 2	4 min (avg)	Urban	46	5.87
			Walk			13 min (avg)	Urban	46	27.56
Cheng et al. (2008)	Taipei, Taiwan	TSI DustTrak 8520	Metro System	Electric	A/C on	16-24 min	Rail/subway	294	41

Table 2 Modal Studies PM10 - Summary Results cont.

Wöhrnschimmel et al. (2008)	Mexico City, Mexico	SKC sampling pump	Bus Minibus Metrobus (BRT)*	Diesel LPG/CNG/diesel Diesel	Unspecified Unspecified Unspecified	1 hr 30 min 1 hr 30 min 1 hr 30 min	Urban Urban Urban	34 34 57	212 201 188
Thai et al. (2008)	Vancouver, BC, Canada	Grimm 1.108 dust monitor	Bicycle			2 hrs	Urban commercial, residential, industrial and parkland	14	21.6-74.8
Berghmans et al. (2009)	Mol, Flanders, Belgium	Grimm 1.108 dust monitor	Bicycle (range) Bicycle (all data) Bicycle (track)			~1 hr ~1 hr 20 min	Small urban centre/residential Cycle track	6 7 1	34.8-102 62.4 54.3
Nasir & Colbeck (2009)	Colchester, UK Various UK regions	Grimm 1.101/1.108 dust monitor	Car (morning) Car (evening) Train (peak) Train (off-peak) 	Unspecified Unspecified Electric Electric Electric Electric	Unspecified Unspecified A/C on Non-A/C A/C on Non-A/C	1 hr 1 hr 1 hr 1 hr 1 hr 1 hr	Suburban/rural Suburban/rural Urban/rural Urban/rural Urban/rural Urban/rural	80 80 10 5 10 5	22 21 44 94 21 90

*Bus Rapid Transit system (dedicated lanes)

Table 3 Modal Studies – $PM_{2.5}$ Summarised Results

Author	Location	Instrument/s	Mode	Vehicle type/make	Window and ventilation settings	Sampling duration	Sampling setting	Number of samples	Mean values (μg/m^3) unless specified
Rodes et al. (1998)	Sacramento, CA, USA	Modified BGI AFC123	Car	1991 Chevrolet Caprice (sedan)	Vent on high (windows closed) Vent on low (windows closed)	2 hrs 2 hrs	Freeway commute	6 4	13.3, 7.6 11.3, 11.0
	Los Angeles, CA, USA	Modified BGI AFC123	Car	1997 Ford Taurus (sedan)	Vent on high (windows closed) Vent on low (windows closed)	2 hrs 2 hrs	Freeway commute	8 4	49.8,40.6 47.2,37.4
Pfeifer et al. (1999)	London, UK	SKC 224PCXR3	Taxi	Unspecified	Unspecified	~8 hrs (taxi shift)	Urban	14	33.36
Adams et al. (2001)	London, UK	Casella vortex ultraflow	Car	Petrol/diesel combination	Not controlled for	<1hr	Urban *(winter)* Urban *(summer)*	12 42	33.7 37.7
			Bus	Diesel		<1hr	Urban *(winter)* Urban *(summer)*	32 36	38.9 39
			Bicycle			20-50 min	Urban *(winter)* Urban *(summer)*	56 40	23.5 34.5
Chan et al. (2002a)	Hong Kong	TSI DustTrak 8520	Taxi	LPG/diesel	A/C on	2 hrs	Urban	30	58
			Bus	Diesel	A/C on Non-A/C	2-2.5 hrs 2-2.5 hrs	Urban Urban	24 12	74 112
			Light bus	Diesel	A/C on Non-A/C	2-2.5 hrs 2-2.5 hrs	Urban Urban	7 7	63 137
			Train	Electric	-	2-2.5 hrs	Urban	70	50
			Tram	Electric	Non-A/C	2-2.5 hrs	Urban	17	175
Chan et al. (2002b)	Guangzhou, China	TSI DustTrak 8520	Taxi	Petrol	A/C on Non-A/C	30 min (avg) 30 min (avg)	Urban commericial and residential zones	8 8	73 106
			Bus	Diesel	Windows closed, A/C on Non-A/C	49 min (avg) 53 min (avg)		11 15	101 145
			Subway	Electric	A/C on	30 min (avg)	Subway	14	44
Dennekamp et al. (2002)	Abderdeen, UK	TSI DustTrak 8520	Car Landrover Bus Walk	Unspecified	Unspecified	Unspecified	Urban	13 7 14 10	~11 (median) ~65 (median) ~38 (median) ~22 (median)
Levy et al. (2002)	Buston, USA	TSI DustTrak 8520	Car Bus Subway	Unspecified Diesel Electric	Windows open Windows open	~2 hrs ~2 hrs ~2 hrs	Urban	- - -	~105 (median) ~110 (median) ~60 (median)
Chertok et al. (2004)	Sydney, Australia	Micro-Vol sampler	Car	Petrol (1997 sedans)	Unspecified	~40 min ~40 min	Urban	8 8	20.75 29.61 (geometric)

Table 3 Modal Studies – $PM_{2.5}$ Summarised Results cont.

Gómez-Perales et al. (2004)	Mexico City, Mexico	Casella vortex ultraflow	Bus	Unspecified	Unspecified	2-3 hrs	Urban commute	16	71
			Minibus	Unspecified	Unspecified	2-3 hrs	Urban commute	28	68
			Subway	Electric	Unspecified	2-3 hrs	Subway	18	61
Gulliver & Briggs (2004)	Northampton, UK	OSIRIS sampler	Car	1995 Ford Fiesta	Windows closed, A/C off, vents off	1 hr	Main commuting routes	36	15.54
			Walk			1 hr		36	15.06
Krausse (2004)	Leicester, UK	OSIRIS sampler	Van	Electric	Unspecified	~1 hr	Urban	133	9.7-25.6
Han et al. (2005)	Trujillo, Peru	SKC pump, BGI KTL Cyclone	Van	Unspecified	Unspecified	~8 hrs	Urban	5	114
			Bus	Diesel	Unspecified	~8 hrs	Urban	3	161
Kaur et al. (2005a)	London, UK	Casella vortex ultraflow	Car	Unspecified	Unspecified	~20 min	Urban centre	29	38
			Taxi	Unspecified	Unspecified	~20 min	Urban centre	22	41.5
			Bus	Unspecified	Unspecified	~20 min	Urban centre	42	34.5
			Bicycle			~20 min	Urban centre	48	33.5
			Walk			~20 min	Urban centre	56	27.5
Kaur et al. (2005b)	London, UK	Casella vortex ultraflow	Walk			~20 min	Urban centre	155	37.7
den Breejen (2006)	Utrecht, Netherlands	TSI DustTrak 8520	Car	Unspecified	Windows closed, vents set to 2	5-25 min	Urban	6	14
					Windows closed, vents set to 4	5-25 min	Urban	6	13
					Windows open	5-25 min	Urban	6	16
Hill & Gooch (2007)	Austin, TX, USA	TSI DustTrak 8520	Car	2006 Dodge Minivans (petrol)	Windows open	~80 hrs (figures given for this study are total sample time by mode)	Freeway/urban commute	15	55
					Windows closed, A/C on			13	66
					Windows open		Special lane (no trucks permitted)	7	30
	Columbus, OH		Car	2006 Dodge Minivans	Windows open		Freeway/urban commute	34	48
					Windows closed, A/C on			8	58
			Bus	B90 Biodiesel				15	36
			Walk			~5 hrs	Urban centre	3	21
	Boston, MA		Car	Unspecified	Windows open		Freeway/urban commute	16	35
					Windows closed, A/C off, vent recirc			12	34
								2	65
			Bus	Diesel	Windows closed, vent fresh	~14 hrs	Urban	5	26
				DPF	Diesel Particulate Filter		Urban	9	55
				CNG			Urban	10	24
			Train	Diesel	Locomotive in front (pull)	~12 hrs	Urban commute	6	70
				Diesel	Locomotive in rear (push)		Urban commute	6	56
			Subway	Electric		~3 hrs	Subway	3	47
			Walk				Urban centre	6	14
	New York, NY		Train	Diesel	Locomotive in front (pull)		Urban commute	3	13
				Diesel	Locomotive in rear (push)		Urban commute	2	5
			Subway	Electric			Subway	3	55

Li et al. (2007)	Beijing, China	Dustmate sampler	Rail Subway	Electric Electric	A/C on Vents on (mechanical fans)	Unspecified	Rail Subway	83 156	39.9 112.6
Gulliver & Briggs (2007)	Leicester, UK	OSIRIS/Dustmate sampler	Car Walk	2002 Fiat Doblo (petrol)	Windows closed, vents set to 2	1 hr 1 hr	Urban arterials and residential zones	33 33	8.3 10.9
Gómez-Perales et al. (2007)	Mexico City, Mexico	Casella vortex ultraflow	Bus Minibus Subway	Unspecified Unspecified Electric	Unspecified Unspecified Unspecified	2-3 hrs 2-3 hrs 2-3 hrs	Urban *(morning)* Urban *(evening)* Urban *(morning)* Urban *(evening)* Urban *(morning)* Urban *(evening)*	15,14 15,14 15,14 15,13 15,13 15,14	~48, ~39 ~39,~16 ~61,~35 ~30,~22 ~38,~32 ~22,~20 (medians)
Adar et al. (2008)	Seatttle/Tahoma, WA, USA	DataRAM pDR-1000AN	Car Bus	Toyota Prius petrol/electricity hyrbrid Diesel	Windows open Various	22 min (avg) 22 min (avg)	Urban commute Urban commute	57 85	12.4 20.9
Tsai et al. (2008)	Taipei, Taiwan	Grimm 1.108 dust monitor	Car Bus Motorcycle Metro System	Petrol Diesel Petrol Electric	Windows closed, A/C on Windows closed, A/C on A/C on	30 min (avg) 43 min (avg) 28 min (avg) 34 min (avg)	Urban Urban Urban Rail/subway	16 16 16 16	22.1 38.5 67.5 35.0
Briggs et al. (2008)	London, UK	OSIRIS sampler	Car Walk	2001 Ford Focus (diesel)	Windows closed, vents set to 2	4 min (avg) 13 min (avg)	Urban Urban	46 46	3.01 6.59
Cheng et al. (2008)	Taipei, Taiwan	TSI DustTrak 8520	Metro System	Electric	A/C on	16-24 min	Rail/subway	294	32
Wöhrnschimmel et al. (2008)	Mexico City, Mexico	SKC sampling pump	Bus Minibus Metrobus (BRT)*	Diesel LPG/CNG/diesel Diesel	Unspecified Unspecified Unspecified	1 hr 30 min 1 hr 30 min 1 hr 30 min	Urban Urban Urban	37 33 51	146 155 112
Thai et al. (2008)	Vancouver, BC, Canada	Grimm 1.108 dust monitor	Bicycle			2 hrs	Urban commercial, residential, industrial and parkland	14	7.3-33.6
McNabola (2008a)	Dublin, Ireland	Casella vortex ultraflow	Walk			20 min	Urban pavement Urban boardwalk (<3 m away)	10 10	2.83:1 (ratio of pavement to boardwalk)
McNabola et al. (2008b)	Dublin, Ireland	High flow personal sampler (HFPS)	Car Bus Bicycle Walk	1994 Landrover/ 1994 Nissan Vanette (diesel) Diesel	Windows closed, A/C off, vents closed Non-A/C, ventilation random	20-45 min 25-30 min 20 min 25-30 min	Urban Urban Urban Urban	46 44 56 48	82.73 128.16 88.14 63.45
Rim et al. (2008)	Austin, TX, USA	TSI SidePak	Bus	Diesel	Windows closed, A/C on	102-110 min	Suburban-urban school commute	9325	7-20

*Bus Rapid Transit system (dedicated lanes)

Table 3 Modal Studies – $PM_{2.5}$ Summarised Results cont.

Morabia et al. (2009)	New York, NY, USA	TSI SidePak AM510	Car	Various	Participants free to choose Windows closed, vents on Windows closed, vents off	1 min avg	Urban centre	7941	13.1 (geo) 18 (per min) 44 (per min)
			Subway	Electric		1 min avg	Subway	6299	19.6 (geo)
			Walk			1 min avg	Urban centre	5929	23.9 (geo)
Berghmans et al. (2009)	Mol, Flanders, Belgium	Grimm 1.108 dust monitor	Bicycle (range)			~1 hr	Small urban centre/residential	6	12.3-75.8
			Bicycle (all data)			~1 hr		7	38.8
			Bicycle (track)			20 min	Cycle track	1	31.7
Nasir & Colbeck (2009)	Colchester, UK	Grimm 1.101/1.108 dust monitor	Car (morning)	Unspecified	Unspecified	1 hr	Suburban/rural	80	9
			Car (evening)	Unspecified	Unspecified	1 hr	Suburban/rural	80	8
	Various UK regions		Train (peak)	Electric	A/C on	1 hr	Urban/rural	10	14
					Non-A/C	1 hr	Urban/rural	5	30
			Train (off-peak)	Electric	A/C on	1 hr	Urban/rural	10	6
					Non-A/C	1 hr	Urban/rural	5	14
Kaur & Nieuwenhuijsen (2009)	London, UK	High flow personal sampler (HFPS)	Car	Toyota Starlet (petrol w/ three way catalyst)	Unspecified	18 min+	Urban centre	22	33.4
			Taxi	Diesel	Unspecified	18 min+		18	43.4
			Bus	Diesel	Unspecified	18 min+		33	33.1
			Bicycle			18 min+		29	33.8
			Walk			18 min+		39	27.1
Boogaard et al. (2009)	Apeldoorn, Netherlands	TSI DustTrak 8520	Car	Unspecified	Unspecified	1 min avg	Urban centre	163	14
			Bicycle					168	11
	Delft		Car					117	33
			Bicycle					155	26
	Den Bosch		Car					170	95
			Bicycle					149	99
	The Hague		Car					184	15
			Bicycle					154	6
	Eindhoven		Car					102	34
			Bicycle					145	39
	Groningen		Car					170	20
			Bicycle					138	13
	Haarlem		Car					167	36
			Bicycle					176	29
	Maastricht		Car					202	31
			Bicycle					148	20
	Nijmegen		Car					131	93
			Bicycle					122	95
	Utrecht		Car					186	122
			Bicycle					174	112
	Zwolle		Car					102	45
			Bicycle					103	44
	Combined total		Car					1694	49.4
			Bicycle					1632	44.5

Table 4 Modal Studies – $PM_{1.0}$ Summarised Results

Author	Location	Instrument/s	Mode	Vehicle type/make	Window and ventilation settings	Sampling duration	Sampling setting	Number of samples	Mean values (µg/m³)
Gulliver & Briggs (2004)	Northampton, UK	OSIRIS sampler	Car Walk	1995 Ford Fiesta	Windows closed, A/C off, vents off	1 hr 1 hr	Main commuting routes	36 36	7.03 7.14
Li et al. (2007)	Beijing, China	Dustmate sampler	Rail Subway	Electric Electric	A/C on Vents on (mechanical fans)	Unspecified	Rail Subway	83 156	14.7 38.2
Gulliver & Briggs (2007)	Leicester, UK	OSIRIS/Dustmate sampler	Car Walk	2002 Fiat Doblo (petrol)	Windows closed, vents set to 2	1 hr 1 hr	Urban arterials and residential zones	33 33	2.9 4.8
Tsai et al. (2008)	Taipei, Taiwan	Grimm 1.108 dust monitor	Car Bus Motorcycle Metro System	Petrol Diesel Petrol Electric	Windows closed, A/C on Windows closed, A/C on A/C on	30 min (avg) 43 min (avg) 28 min (avg) 34 min (avg)	Urban Urban Urban Rail/subway	16 16 16 16	16.2 31.3 48.4 26.5
Briggs et al. (2008)	London, UK	OSIRIS sampler	Car Walk	2001 Ford Focus (diesel)	Windows closed, vents set to 2	4 min (avg) 13 min (avg)	Urban Urban	46 46	1.82 3.37
Berghmans et al. (2009)	Mol, Flanders, Belgium	Grimm 1.108 dust monitor	Bicycle (range) Bicycle (all data) Bicycle (track)			~1 hr ~1 hr 20 min	Small urban centre/residential Cycle track	6 7 1	7.32-70.9 37.4 29.8
Nasir & Colbeck (2009)	Colchester, UK Various UK regions	Grimm 1.101/1.108 dust monitor	Car (morning) Car (evening) Train (peak) Train (off-peak) 	Unspecified Unspecified Electric Electric 	Unspecified Unspecified A/C on Non-A/C A/C on Non-A/C	1 hr 1 hr 1 hr 1 hr 1 hr 1 hr	Suburban/rural Suburban/rural Urban/rural Urban/rural Urban/rural Urban/rural	80 80 10 5 10 5	6 5 12 19 4 6

Table 5 Modal Studies –UFP Summarised Results

Author	Location	Instrument/s	Mode	Vehicle type/make	Window and ventilation settings	Sampling duration	Sampling setting	Number of samples	Mean values (pt/cm^3) unless specified
Dennekamp et al. (2002)	Abderdeen, UK	TSI 3934 Scanning Mobility Particle Sizer (SMPS)	Car Landrover Bus Walk	Unspecified	Unspecified	Unspecified	Urban	22 24 11 10	~25000 ~55000 ~60000 ~40000 (medians)
Levy et al. (2002)	Buston, USA	TSI DustTrak 8520	Car Bus Subway	Unspecified Diesel Electric	Windows open Windows open	~2 hrs ~2 hrs ~2 hrs	Urban	- - -	~39000 ~33000 ~22000 (medians)
Kaur et al. (2005a)	London, UK	TSI P-Trak 8525 UPC	Car Taxi Bus Bicycle Walk	Unspecified Unspecified Unspecified	Unspecified Unspecified Unspecified	~20 min ~20 min ~20 min ~20 min ~20 min	Urban centre Urban centre Urban centre Urban centre Urban centre	13 9 18 21 25	99736 87545 101364 93968 67773
Kaur et al. (2005b)	London, UK	TSI P-Trak 8525 UPC	Walk			(Kaur et al. 2005b) ~20 min	Urban centre	120	80009
Vinzents et al. (2005)	Copenhagen, Denmark	TSI 3007 CPC	Bicycle			93 min (avg)	Urban centre	5	32400 (geometric)
Kaur et al. (2006)	London, UK	TSI P-Trak 8525 UPC	Car Taxi Bus Bicycle Walk	Unspecified Unspecified Unspecified	Unspecified Unspecified Unspecified	~20 min ~20 min ~20 min ~20 min ~20 min	Urban centre Urban centre Urban centre Urban centre Urban centre	8 5 6 10 2	36821 108063 95023 84005 46072
den Breejen (2006)	Utrecht, Netherlands	TSI 3007 CPC	Car Bicycle	Unspecified	Unspecified	5-25 min 5-25 min	Urban Urban	52 51	22125 22823

Table 5 Modal Studies – UFP Summarised Results cont.

Hill & Gooch (2007)	Austin, TX, USA	TSI DustTrak 8520	Car	2006 Dodge Minivans (petrol)	Windows open Windows closed, A/C on Windows open	~80 hrs (figures given for this study are total sample time by mode)	Freeway/urban commute Special lane (no trucks permitted)	15 13 7	25928 21248 8671
	Columbus, OH		Car	2006 Dodge Minivans	Windows open Windows closed, A/C on		Freeway/urban commute	34 8	43337 14328
			Bus	B90 Biodiesel				15	17196
			Walk			~5 hrs	Urban centre	3	22502
	Boston, MA		Car	Unspecified	Windows open Windows closed, A/C off, vent recirc		Freeway/urban commute	16 12	29401 17429
					Windows closed, vent fresh	~14 hrs	Urban	2	28981
			Bus	Diesel			Urban	5	83227
				DPF	Diesel Particulate Filter		Urban	9	29788
				CNG	Compressed Natural Gas		Urban	10	23452
			Train	Diesel	Locomotive in front (pull)	~12 hrs	Urban commute	6	118218
				Diesel	Locomotive in rear (push)		Urban commute	6	13607
			Subway	Electric		~3 hrs	Subway	3	11909
			Walk				Urban centre	6	30273
	New York, NY		Train	Diesel	Locomotive in front (pull)		Urban commute	3	137366
				Diesel	Locomotive in rear (push)		Urban commute	2	51591
			Subway	Electric			Subway	3	49045
Briggs et al. (2008)	London, UK	OSIRIS sampler	Car Walk	2001 Ford Focus (diesel)	Windows closed, vents set to 2	4 min (avg) 13 min (avg)	Urban Urban	46 46	21639 30334
Rim et al. (2008)	Austin, TX, USA	TSI 3007 CPC	Bus	Diesel	Windows closed, A/C on	102-110 min	Suburban-urban school commute	9325	6040-34500
Thai et al. (2008)	Vancouver, BC, Canada	TSI P-Trak 8525 UPC	Bicycle			2 hrs	Urban commercial, residential, industrial and parkland	7	18830-57692
Weichenthal et al. (2008)	Montréal, Canada	TSI P-Trak 8525 UPC	Car (morning) Car (evening) Bus (morning) Bus (evening) Walk (morning) Walk (evening)	Unspecified Unspecified	Participants free to choose	20.5-55 min 21-50 min 8-15.5 min 11-33 min 3-16 min 3-10 min	Urban highway and busy roadway Two-lane roadway	22 (total) 42 (total) -	38348 31489 28029 22626 25161 15778
Zhu et al. (2008)	Los Angeles, LA, USA	TSI 3934 Scanning Mobility Particle Sizer (SMPS)	Van	2002 Chevrolet Express	Windows closed, A/C on	2 hrs 2 hrs	Freeway (mostly diesel traffic) Freeway (mostly petrol traffic)	2 2	134000 83800

Table 5 Modal Studies – UFP Summarised Results cont.

Berghmans et al. (2009)	Mol, Flanders, Belgium	TSI P-Trak 8525 UPC	Bicycle (range)			~1 hr	Small urban	6	10851-30576
			Bicycle (all data)			~1 hr	centre/residential	7	21226
			Bicycle (track)			20 min	Cycle track	1	21626
Kaur & Nieuwenhuijsen (2009)	London, UK	TSI P-Trak 8525 UPC	Car	Toyota Starlet (petrol w/ three way catalyst)	Unspecified	18 min+	Urban centre	9	101770
			Taxi	Diesel	Unspecified	18 min+		8	91947
			Bus	Diesel	Unspecified	18 min+		14	100018
			Bicycle			18 min+		8	77621
			Walk			18 min+		16	63065
Cattaneo et al. (2009)	Milan, Italy	TSI P-Trak 8525 UPC/TSI 3007 CPC	Car	Petrol w/ three way catalyst & anti-particulate filter	Windows closed, vents on	15 min+	Urban centre	21	107000
			Bus	Diesel	Unspecified	15 min+		21	117600
			Walk			15 min+		21	100200
Boogaard et al. (2009)	Apeldoorn, Netherlands	TSI 3007 CPC	Car	Unspecified	Unspecified	1 min avg	Urban centre	163	20796
			Bicycle					167	17070
	Delft		Car					112	24460
			Bicycle					153	27998
	Den Bosch		Car					170	23012
			Bicycle					147	21191
	The Hague		Car					184	15430
			Bicycle					131	15697
	Eindhoven		Car					102	23461
			Bicycle					143	28141
	Groningen		Car					170	22234
			Bicycle					138	21326
	Haarlem		Car					167	34739
			Bicycle					175	30363
	Maastricht		Car					202	35538
			Bicycle					87	28220
	Nijmegen		Car					131	24064
			Bicycle					121	20244
	Utrecht		Car					186	29722
			Bicycle					173	27246
	Zwolle		Car					89	23583
			Bicycle					101	31354
	Combined total		Car					1676	25545
			Bicycle					1536	24329

2.2.1 Car

Much of the available literature suggests that car or light vehicle commuters are generally exposed to higher levels of pollution than those traveling by almost all alternative modes; including walk, bicycle, bus, subway and train (Adams et al. 2002; Batterman et al. 2002; Boogaard et al. 2009; Bruinen de Bruin et al. 2004; Chan et al. 1991; Chan et al. 1999; Chertok et al. 2004; Dor et al. 1995; Duci et al. 2003; Duffy & Nelson 1997; Gulliver & Briggs 2004; Hill & Gooch 2007; Kaur et al. 2007; Kingham et al. 1998; Leung & Harrison 1999; Löfgren et al. 1991; McNabola et al. 2008b; Rank et al. 2001; Shiohara et al. 2005; Taylor & Fergusson 1998; Torre et al. 2000; van Wijnen et al. 1995; Vellopoulou & Ashmore 1998). While this may be true for most transport pollutants including PM, UFP, VOCs, CO, PAHs and black carbon, different results occur for NO_2 for example, where exposure in buses is usually higher than that found in cars due to in-vehicle sources (Chertok et al. 2004; Farrar et al. 2001). Some studies have also recorded slightly higher levels of $PM_{2.5}$ and UFPs for buses than for cars (Adams et al. 2001; Adar et al. 2008; Dennekamp et al. 2002; Hill & Gooch 2007; Levy et al. 2002; McNabola et al. 2008b).

Although these findings show that car exposure levels are generally among the highest, there can be considerable variation between transport modes at different study sites. Results are affected by variables such as vehicle makeup and configuration, ambient pollutant levels and local environmental factors, meaning car drivers may actually be the least exposed to PM in certain conditions. Recent research by Briggs et al. (2008) found that walking exposure rates for PM were greater than vehicle exposure by a factor of 4.7 (PM_{10}), 2.2 ($PM_{2.5}$), 1.9 ($PM_{1.0}$) and 1.4 (UFPs). These ratios for $PM_{2.5}$ and UFPs are very close to those reported by Dennekamp et al. (2002), who gave factors of 2.0 and 1.65 respectively. Furthermore, comparisons between motorcycles, cars, buses and the train/subway system in Taipei, Taiwan, showed car commuters received the lowest PM concentration exposure of all vehicular modes (Tsai et al. 2008). While sampling was conducted with windows closed, it is interesting to note that these three studies used different ventilation settings. The influence of vent settings appears to vary greatly between studies. An investigation in three US cities, Boston, Austin and Columbus, found that while UFP exposure was lowest with windows closed (A/C on), exposure for $PM_{2.5}$ was higher at this setting than with the windows open (Hill & Gooch 2007).

Overall, the literature consistently presents comparatively high levels of CO and VOCs for the car mode (Bruinen de Bruin et al. 2004; Chan et al. 1991; Chertok et al. 2004; Dor et al. 1995; Duci et al. 2003; McNabola et al. 2008b; van Wijnen et al. 1995; Vellopoulou & Ashmore 1998). Coupled with research which also ranks cars as receiving the highest levels of PM pollution, the car commuter does not fare well against other modal choices. However, when measuring total accumulative intake, car travel may not be the most detrimental mode when travel times and breathing rates are taken into account, especially for active modes.

2.2.2 Motorcycle

The exception to cars possibly being the most affected mode is commuting by motorcycle, where exposure is substantially higher than all other modes of transport. Studies have so far reported this for PM, CO, NO_2 and VOCs (Bugajny et al. 1999; Chan et al. 1993; Kuo et al. 2000; Piechocki-Minguy et al. 2006; Saksena et al. 2006; Saksena et al. 2008; Tsai et al. 2008). This is likely to be due to motorcyclists being situated directly in the 'stream of pollutants' without any shielding, along with their relatively close proximity to the exhaust tailpipes of traffic ahead. One study found mean exposure concentrations in Taipei city to be approximately three times higher than cars for PM_{10} (112.8 vs 41.9 µg/m³) and $PM_{2.5}$ (67.5 vs 22.1 µg/m³), while $PM_{1.0}$ recordings were 48.4 and 16.2 µg/m³ respectively (Tsai et al. 2008).

An important factor affecting motorcycle exposure is time spent idling at traffic lights, which increases PM levels by 5-7% compared to when moving (Tsai et al. 2008). Hence trips through areas with high traffic light density are likely to render far higher overall exposure rates. To date, exposure differences between motorcycles and bicycles on the same route have not been explored and this is an area requiring further research.

2.2.3 Train and subway

Electrified rail commuters are thought to receive the lowest amount of pollutants compared with all other modes. This has been found to be the case for NO_2 (Chertok et al. 2004; Piechocki-Minguy et al. 2006), CO (Duci et al. 2003) and VOCs (Barrefors & Petersson 1996; Chertok et al. 2004; Lau & Chan 2003; Shiohara et al. 2005). Currently, no PM data comparing above-ground electrified rail and other roadway modes are available. Exposure rates for VOCs, CO and NO_2 are lower because tracking is generally situated away from traffic flows, cabins provide protection, and the train itself is not a strong source of pollutants. However, results are influenced by background levels and frequency of passenger movements, with far higher levels found in some cities compared to others (Li et al. 2007).

Research conducted on Sydney's CityRail electrified rail network found VOCs and NO_2 to be under half the levels found in private cars, which had the highest recordings of all modes (Chertok et al. 2004). Adjusted geometric means for cars and trains were (expressed as parts per billion): benzene (12.29, 3.77), toluene (28.76, 12.44), ethyl benzene (4.38, 1.73), xylenes (19.91, 7.26) and NO_2 (29.70, 14.85). Such findings are in agreement with a study by Lau & Chan (2003) in Hong Kong, where mean concentrations for benzene, toluene, ethyl benzene, *m*/p-xylene and *o*-xylene were considerably lower for electric rail than those recorded in a taxi.

Results for diesel-powered locomotives differ greatly depending on locomotive position. Recordings from the Boston and New York rail networks show that when the locomotive is in front of the carriages (pull), UFPs, black carbon and PAH are much higher than any other mode. When the locomotive is in the rear (push), levels are comparable to that of subway electric rail. However, fine particles ($PM_{2.5}$) for Boston were at around the same concentration as subway and car (windows up, vents open), regardless of where the locomotive was located (Hill & Gooch 2007).

Subway studies seem to differ in agreement, with some reporting the lowest exposure levels of all modes for PM, PAH, CO and benzene (Chan et al. 2002b; Gómez-Perales et al. 2004; Hill & Gooch 2007), and others finding PM_{10} and $PM_{2.5}$ to be between 3 to 10 times higher than for road surface transport modes (Aarnio et al. 2005; Adams et al. 2001; Johansson & Johansson 2003). Fromme et al. (1998) found substantially higher PAH concentrations in the Berlin subway than for cars. The explanations suggested included ambient seasonal variation and the influence of tar preservatives in the wooden railway ties. Gómez-Perales et al. (2004) puts such variance across subway studies down to differences in brake systems, ventilations systems and tunnel depth, while Kim (2008) suggests it could be due to different monitoring conditions such as equipment, outdoor climate and season. The most recent subway study on the Taipei system found lower levels of PM_{10} and $PM_{2.5}$ than those reported in all previous studies (Cheng et al. 2008).

2.2.4 Bus

Investigation into NO_2 levels has shown buses have the highest concentrations due to self-pollution from diesel engines (Chertok et al. 2004; Farrar et al. 2001). Tsai et al. (2008) found particulate matter to be highest in buses (excluding motorcycles) for all PM fractions. This is supported by various studies for $PM_{2.5}$ (Adams et al. 2001; Adar et al. 2008; Dennekamp et al. 2002; Fondelli et al. 2008; Hill & Gooch 2007; Levy et al. 2002; McNabola et al. 2008b). Hill & Gooch's (2007) results for $PM_{2.5}$ in a conventional diesel bus were around half that of cars (windows up), but UFPs (pt/cm^3) were around four times higher. PAH levels on buses were substantially lower than in cars, regardless of the in-vehicle setting. VOC concentrations have also been found to be highest in buses for butadiene, ethylene and acetylene (McNabola et al. 2008b) and BTEX, apart from toluene (Chertok et al. 2004). Conversely, Shiohara et al. (2005) observed higher VOC concentrations in cars. There is also substantial evidence showing that exposure to CO in buses is much lower than in cars (Bruinen de Bruin et al. 2004; Dor et al. 1995; Duci et al. 2003; Han & Naeher 2006; Kaur et al. 2005a; Saksena et al. 2008; Scotto di Marco et al. 2005; van Wijnen et al. 1995; Vellopoulou & Ashmore 1998).

Experiments with Diesel Particulate Filters (DPF) resulted in a reduction of UFP concentrations by around three-quarters to match ambient air levels – and the same found in compressed natural gas (CNG) powered buses - but $PM_{2.5}$ concentrations double and PAH concentrations are elevated. Biodiesel buses emit the lowest levels of UFPs and PAH, but slightly higher levels of $PM_{2.5}$ than traditional engines (Hill & Gooch 2007).

As with cars, self-pollution intake can vary depending on whether windows are open or closed, along with the age of the vehicle (Marshall & Behrentz 2005). Bus commuters are also affected by doors opening and closing, with concentrations for $PM_{2.5}$ and PM_{10} increasing by 2% and 5% when opened compared to when closed (Tsai et al. 2008).

2.2.5 Pedestrian

Pedestrian exposure is an uncertain area with results varying between studies. Research finding lower exposure has often cited the relative separation from the traffic emission stream as the primary explanation. Evidence supporting this idea has been provided by Kaur et al. (2005b), who found pedestrian exposure varied greatly with distance from traffic, and was highest at the kerbside. However, differences inevitably occur between studies in the form of sampling settings (geographic location, buildings, vegetation) and methodologies. Three of the most recent studies, conducted in Dublin, Milan and London, reported pedestrians were the least exposed to $PM_{2.5}$ and UFPs compared to car and bus (Cattaneo et al. 2009; Kaur & Nieuwenhuijsen 2009; McNabola et al. 2008b). Yet a similar study completed in London produced opposite findings for all PM, including ultrafines (Briggs et al. 2008). Such results are supported by other research for PM_{10}, $PM_{2.5}$ (Dennekamp et al. 2002; Gulliver & Briggs 2004; Morabia et al. 2009; Saksena et al. 2008; Zhao et al. 2004), and UFPs in the pilot-study phase of research underway in Barcelona (de Nazelle et al. 2008). This study not only measures exposure concentrations, but also factors in inhalation rates. Preliminary findings suggest pedestrians may actually inhale greater amounts of UFPs than any other mode.

There is no disagreement between the literature on carbon monoxide exposure, with pedestrians being the least exposed (Saksena et al. 2008; Zhao et al. 2004). This is likely to be due to vehicles being the only source of CO, whereas PM can be re-suspended having originated from other sources. For VOC exposure, pedestrians are also ranked lowest for all BTEX pollutants combined (Chertok et al. 2004; McNabola et al. 2008b). It is thought that this is due to wind dispersion not experienced in the closed microenvironment setting of vehicles used in most studies.

For Hill & Gooch's (2007) study, pedestrian commuters were exposed to the lowest levels of $PM_{2.5}$ and black carbon, but UFPs and PAH levels were comparable to those found on CNG or DPF equipped buses. Therefore, they were relatively low compared to most, but not lower than biodiesel buses or cars with windows closed (A/C on).

2.2.6 Bicycle

As with walking, cyclist exposure is also quite a contentious issue, with research providing conflicting results. One of the earlier exposure studies (completed in Amsterdam) found CO levels to be substantially lower than cars, and lower than pedestrians during most sampling instances (van Wijnen et al. 1995). Later research confirmed the contrary, with cyclists receiving higher levels than walk, car and bus (Mackay 2004). In London during 2005, a study found levels to be about the same as in cars (Kaur et al. 2005a).

For NO_2, van Wijen et al. (1995) found levels to be higher for cycles than for cars. Australian research in Perth was in agreement, reporting 22 ppb compared with 15 in taxis and 14 in couriers (Farrar et al. 2001). Yet for Sydney, cars and buses measured 29.70 and 44.30 ppb, but cycles only 24.58 (Chertok et al. 2004). In this study, exposure was even lower than for pedestrians (26.08 ppb). The variance between vehicle exposures in Perth and Sydney may have been due to differences in in-vehicle settings (windows, A/C), sampling time-of-day (peak versus off-peak traffic), or differences between the types of measurement equipment used.

$PM_{2.5}$ has been found to be quite a bit lower for bicycles than for cars (Adams et al. 2001; Gee & Raper 1999; Kaur et al. 2005b; McNabola et al. 2008b; Rank et al. 2001). Seasons appear to have a marked affect, with wintertime recordings in London showing a mean exposure difference of 10.2 µg/m³ higher compared with summer (Adams et al. 2001). No data is currently available comparing course particle concentrations and few peer-reviewed published studies have addressed cyclist ultrafine exposure (Kaur et al. 2005a; Thai et al. 2008; Vinzents et al. 2005). Only one of these compared results with other modes of travel. Geometric means were 64,861, 88,055, 92,824 and 99,266 UFPs/cm³ for walk, cycle, car and bus, respectively (Kaur et al. 2005a). In 2006, a Dutch report found overall mean UFP comparisons (*N*=52) for cycle (22,823 UFPs/cm³) and car (22,125 UFPs/cm³) to be virtually the same (den Breejen 2006). For the Barcelona pilot study, de Nazelle et al. (2008) found the mean concentration to be roughly 40,000 UFPs/cm³.

Although this was slightly lower than for bus and walk, after inhalation rates had been considered, walking and cycling climbed well above subway and bus.

There do not appear to be any studies reporting higher VOC concentrations for cyclists than cars and buses. Besides electric train commuters and pedestrians, cyclists are exposed to the lowest amounts of VOCs including BTEX (Chertok et al. 2004), butadiene, ethane, ethylene and acetylene (McNabola et al. 2008b; O'Donoghue et al. 2007). These findings are supported by previous VOC measurements, including initial BTEX investigation by van Wijnen et al. (1995); further BTEX work by Rank et al. (2001) and a study that just measured benzene (Kingham et al. 1998). Moreover, following consideration of increased respiration rates experienced by cyclists, Rank et al. (2001) concluded that car drivers were still more exposed than cyclists as cabin concentrations were 2-4 times greater than cyclist breathing zones. In 2006, a study produced a different view, stating “Relationships between heart rate (HR) and oxygen uptake, and between HR and pulmonary ventilation (VE) for each participant were established in laboratory tests. The VE during cycling was four times higher than resting value. The level of air pollution exposure when cycling seemed to be comparable with the levels of exposure when sitting inside a vehicle” (Bernmark et al. 2006, p. 1486). The following year, O’Donoghue et al. (2007) compared cyclist VOC inhalation to bus passengers. Although exposure was lower, after respiration rates and travel times were accounted for, cyclists received slightly higher VOC intake than bus patrons.

The evidence seems to suggest that although cyclists have the benefit of greater wind dispersion and do not typically have to wait behind queued traffic, faster respiration rates may result in higher overall intake of VOCs. As the majority of pollutant studies have not considered respiration rates, it is possible that actual pollutant intake for cyclists and pedestrians has been greatly underestimated.

Nevertheless, there are various potential factors influencing cyclist exposure. These include: position on the road; traffic light timings; ability to pass between congested traffic; height of cyclist from ground; chosen route; traffic density and use of bus or cyclist lanes (Kaur et al. 2007).

2.3 Effect of proximity to traffic: pedestrians and cyclists

2.3.1 Pedestrians

Pedestrian exposure relative to traffic proximity has been investigated in many key research papers. Much of the initial investigation focused on position on pavement and time spent crossing at busy intersections. Kaur et al. (2005b) measured CO, $PM_{2.5}$ and UFP variation along a heavily trafficked London road, finding significant UFP reductions for the building side of the pavement as well as for the south side of the road. The reduction between kerbside (89,469 pt/cm^3) and building side (73,329 pt/cm^3) is indicative of a rapid decrease in particle concentrations when moving just a very small distance away from emissions sources. Higher concentrations on the north side can be explained by meteorology and street topography. Although little or no difference was recorded for $PM_{2.5}$ and CO for side of street and pavement position in this study, a reduction in CO concentrations with distance from the kerb was observed by Wright et al. in 1975 (cited in Kaur et al. 2005b). Kaur et al. (2006) later concluded that walking on the building side of the pavement whilst avoiding smokers and industrial work sites, can reduce mean UFP pedestrian exposure by 10-30%.

Walking along routes in busy areas with lengthy traffic signal delays can also increase exposure. A study using a micro-simulation model to track pedestrian and vehicle movements found that longer pedestrian crossing signal lengths result in greater exposure to CO and PM (Ishaque & Noland 2008). The study also noted that giving signal priority to pedestrians could greatly reduce overall exposure, despite an increase in traffic emissions. Such simulated results are supported by time-activity exposure profiles showing immense spikes (to maximum recorded UFP levels) when pedestrians wait at crossings (Kaur et al. 2006). Built-up city streets with tall structures are prone to urban street canyon effects where microscale wind flow characteristics cause the formation of high pollutant zones, exacerbating the higher levels experienced when taking heavily trafficked routes.

Clearly it is optimal for pedestrians to choose backstreet routes, avoid dusty/smoky areas and generally keep as far away from roadside high pollutant zones as possible. General background concentration and exposure variability have been confirmed to be much lower for pedestrians using a quieter backstreet route compared to a busier option (Kaur et al. 2006).

A recent noteworthy study investigated differences between $PM_{2.5}$ and benzene exposure right next to a 3 lane roadway (on the pavement) and on a boardwalk, only 2 metres away. The footpath and boardwalk are separated only by a small 'low-boundary' wall, meaning the boardwalk is a mere further 1-2 metres away from traffic than the footpath. Simultaneous recordings of pedestrians walking along each side of the wall found $PM_{2.5}$ and benzene levels to be higher by a factor of 2.83 and 2.0 on the pavement side. Computation Fluid Dynamics (CFD) modeling showed that due to the dispersive effect of the wall, levels would always be lower on the boardwalk, regardless of different wind characteristics (McNabola et al. 2008a).

These results, along with the aforementioned studies, highlight the degree to which substantial differences can occur at the microscale level. If significant differences can be observed at only a few metres from traffic sources, it potentially has large implications for future walkway planning and design.

2.3.2 Cyclists

As for pedestrians, the effect of chosen route also has important implications and can significantly reduce exposure for cyclists, especially when using backstreets, and away-from-road cycle tracks.

An early investigation by Bevan et al. (1991) compared CO, Respirable Suspended Particles (RSPs) and VOC concentrations along a busy roadway to a common parkland area. This study found CO and RSP levels to be higher along the roadway by a factor of 13 and 6, respectively. A range of 18 different VOCs were also sampled with all but four being substantially higher on the road. A similar study completed in 1998 also recorded consistently lower levels of benzene and particulates (measured by absorbance) for the cyclist riding on an exclusive cycle path (Kingham et al. 1998).

Similarly, taking backstreet routes provides cyclists with a relatively low-exposure option. Kaur et al. (2005a) looked at backstreet versus main road exposure in Central London, finding significantly lower concentrations of CO and UFPs across five different modes, indicating the positive effect of travelling on less heavily trafficked routes. Unfortunately the study did not break the findings down into exact comparative figures for each mode on each study route. However, research by Hertel et al. (2008) – based on street pollution modeling - explored the differences between cycling along the shortest possible route, cycling along a low-exposure route (backstreets), and taking the shortest direct route by bus. The study found that total exposure for the shortest cyclist route was between 10-30% lower for primary pollutants (NO_x and CO), but differences were insignificant for secondary pollutants (NO_2 and PM_{10}/$PM_{2.5}$). When traffic-generated concentrations were excluded, accumulated exposure was up to 67% lower for the low-exposure route, while for bus patrons, this figure was between 79% and 115%. The study also observed that travelling during off-peak times reduces exposure between 10% and 30% for primary pollutants, and 5% and 20% for secondary.

Berghmans et al. (2009) conducted interesting research in a small town in Flanders, Belgium, where a cyclist rode around various parts of the town and PM_{10} and UFP exposure was mapped according to concentration. They found that while UFP exposure was considerably higher in the city centre and along busy roads, PM_{10} variance was almost entirely dependant on the presence of mechanical or manual construction work. The lack of difference in PM_{10} levels for backstreet and main road areas are consistent with the findings of Hertel et al. (2008). As with the 2005 study by Kaur et al., the findings of Berghmans et al. (2009) were only presented as overall mean concentrations and not split into main road and backstreet areas. Although concentration variability was presented by means of time-exposure profiles and concentration 'dust maps', these methods do not allow for a clear distinction between overall mean exposure levels and mean levels experienced within different land use zones.

Somewhat similar methods were employed in a study conducted by Thai et al. (2008), where $PM_{10,}$ $PM_{2.5}$ and UFP concentrations were measured by cycling across a variety of land use zones. Comparable observations were made, with PM_{10} levels peaking in construction zones and UFPs near heavy traffic. Exposure-distance profiles were presented, outlining clear transitions between a main transit corridor, an off-road seaside cycle route, construction sites and the central business district. Sudden drops in UFP concentrations were evident when transferring from key commuting roads to smaller backstreets or off-road cycleways. Recorded $PM_{2.5}$ data was also mapped by colour-coding concentrations and overlaying them onto a land-use regression model (LUR), demonstrating how concentrations varied geospatially and compared to background PM_3 modeling. Unlike the heterogeneous distribution of UFPs, $PM_{2.5}$ was found to be more spatially uniform across the study route due to ability of $PM_{2.5}$ to stay airborne for long periods. This lack of variance was also noted by Hertel et al. (2008).

One area not previously explored in detail, is microscale variance at different distances from the roadway. In many cities, most notably in the Netherlands and Germany, cycle lanes are often situated in between parked cars and the road rather than directly on the roadside. This provides an interesting situation for exposure measurement. There are obvious positive implications when positioning cycle ways as far away from the road as possible, but it is uncertain at which distance it becomes worthwhile. A separation as little as only two or three metres may even be beneficial and it is possible that parked cars provide some degree of protection, as found with the small dividing wall in the study by McNabola et al. (2008a). O'Donoghue et al. (2007) noted considerable differences in VOC levels between travelling on the congested side of the road as opposed to going against the main flow of traffic, suggesting a 5-7 metre gap is highly beneficial, even without the presence of dispersive barriers. However, local wind conditions undoubtedly influence the degree to which distance from sources is significant. Berghmans et al. (2009) noted dust concentrations from construction work rose substantially when riding on the windward side of the road whereas when riding on the other side, almost no increase occurred. Traffic pollutants behave in a similar fashion and are also influenced by temperature and precipitation. Therefore, any positive results presented can only be viewed with consideration of associated factors and may only be applicable under particular conditions.

Due to dependence on associated sources and the behaviour of different pollutants, it is apparent that coarse particle measurement is less important for cyclist exposure studies and that UFPs, CO and perhaps $PM_{1.0}$ should be of key concern. While time-exposure profiles and particle mapping techniques are useful for displaying the variance across routes, a clear research gap exists where comparative mean exposure for different route types could be ascertained.

Cyclists are generally not able to commute exclusively on dedicated off-road cycleways, but a combination of parkland, trail and backstreet routes are realistic options in many towns and cities. Exploring total mean exposure between such an option and taking a more direct busier roadway is an area worthy of further investigation. The possibly negative consequence of a longer commuting duration may greatly outweigh the associated health cost of higher pollution intake. Additionally, the degree to which pollutant levels drop off at different distances parallel to the road has only been explored for pedestrians, leaving another key aspect open to investigation.

2.4 Other variables affecting personal exposure levels

There are a multitude of variables that affect exposure levels. These can be grouped into five main categories: physical environment (geographic location, topography and urban built environment), meteorological conditions, traffic conditions, travel behaviour; and vehicle makeup and configuration.

2.4.1 Physical environment

Building configuration, road layout, trees and roadside structures have an effect on the accumulation and dispersion of pollutants (Bowker et al. 2007; Briggs et al. 2008). Comparisons between an open terrain area and an area with vegetation and noise barriers found higher concentrations of UFPs for the open area. Concentrations in the vegetated area were more uniform and vertically well-mixed (Bowker et al. 2007).

Street canyon environments – streets amongst dense blocks of structures such as skyscrapers – can increase concentrations at the pedestrian level by up to 30% (Bogo et al. 2001). Using 3D Computational Fluid Dynamics, McNabola et al. (2009) have discovered that low boundary walls can reduce pedestrian exposure by 40% for perpendicular wind directions and up to 75% for parallel wind directions .

2.4.2 Meteorological conditions

Wind speed/direction, seasonal variation, precipitation, temperature, humidity and sea spray can all influence pollutant levels (Briggs et al. 2008; Jamriska et al. 2008; Minguillón et al. 2008). While some conditions have more obvious effects on chemical behaviour and pollutant concentrations, there are also less obvious factors where weather can have an indirect influence. For example, in countries with very cold climates, particulate from studded tyre abrasion is reported to significantly elevate levels of high particle mass concentrations (Gustafsson et al. 2008).

2.4.3 Traffic conditions

Clearly, the more congested the traffic conditions, the higher the levels of traffic-related pollutants. Other influences which increase certain pollutant concentrations are time spent idling at traffic lights and heavy traffic density. Heavy traffic density especially increases NO_2, and high truck density has been shown to elevate $PM_{2.5}$ above levels in traffic without trucks present (Janssen et al. 2003).

2.4.4 Travel behaviour

Various elements of an everyday typical commute can affect total daily exposure. Some of these may include frequency of stops - e.g. opening doors, gasoline refueling, time spent in parking lots, which side of the footpath you walk on, and so on (Kaur et al. 2005b). Cyclists can take shortcuts and dodge through traffic, resulting in less time spent in congestion if they choose to.

2.4.5 Vehicle makeup and configuration

Older vehicles and vehicles running poorly are more likely to emit higher amounts of exhaust fumes. New vehicles often have very high in-cabin concentrations of VOCs due to the construction materials (Yoshida et al. 2006). In-vehicle settings; including windows, ventilation settings and air conditioning are other key factors influencing in-cabin levels. The Clean Air Task Force (Hill & Gooch, 2007) experience shows having the windows up and the air conditioning on is the most protective setting. Having the windows open is the next best option, while setting the vents to fresh (windows closed) is the worst as pollutants infiltrate but cannot disperse. Esber & El-Fadel (2008) found that in-vehicle CO ingression varied between 250 to 1250 mg/h depending on the vehicle ventilation settings. Again, windows closed (A/C on) resulted in the lowest recordings, while ‘windows half-opened, vents closed’ resulted in the highest. Having the windows half-opened and the vents closed provides a similar environment to having the windows closed and the vents open, reducing dispersion while allowing significant infiltration.

2.5 Health implications of personal exposure to transport pollution

Transport related pollutants are widely known to be associated with various cancers and other medical ailments. NO_2 has been linked to wheezing in infants (Ryan et al. 2005), childhood asthma and increased rates of respiratory illnesses such as bronchitis (Duhme et al. 1996; Gauderman et al. 2005; Pikhart et al. 2000). PM exposure can cause various cancers, chronic respiratory diseases and cardiovascular diseases (Miller et al. 2007; Pandya et al. 2002; Samet et al. 2000; Sørensen et al. 2003). The smaller PM fractions are known to have the highest toxicity as they penetrate deeper into the lungs and contain higher concentrations of organic matter. Due to their incredibly small size, UFPs are able to easily enter the body, transfer between blood cells and access bone marrow, the heart, spleen and lymph nodes (Oberdorster et al. 2005). Certain VOCs are extremely carcinogenic and can cause damage to the central nervous system (Bolla 1991). Benzene and 1,3-butadiene are considered the most toxic and are known to cause leukemia, even after only short-term, low-level exposure (Glass et al. 2003; Murray 2000). As for VOCs, some PAH compounds are also highly carcinogenic. PAHs have been linked to multiple organ cancers, including lung, bladder, kidney, larynx and cancer of the skin (Boffetta et al. 1997). High PAH exposure is also thought to cause premature birth and limit neurodevelopment during the first 3 years of life (Perera et al. 2006).

Some research has specifically linked proximity to traffic with adverse health effects, such as low birth weight and premature births among women living near busy roads (Wilhelm & Ritz 2003), and increased allergies and respiratory illness among street vendors (Kongtip et al. 2006).

More recently, long-term research has also concluded that excessive exposure to air pollution (experienced living in highly polluted cities) can cause neuroinflammation and an altered brain immune response, which increases the likelihood of developing Alzheimer's and Parkinson's disease (Calderon-Garciduenas et al. 2008). A large epidemiological study based on 23 European cities estimated 16,926 premature deaths could be prevented annually if long-term exposure to $PM_{2.5}$ levels were reduced to 15 lg/m³ in each city (Boldo et al. 2006). This highlights the sheer scale of damage vehicle pollutants contribute to, and this is for only one particle fraction.

2.6 Summary

Formerly, car commuters were overwhelmingly thought to be exposed to higher concentrations of total air pollutants than for walking, cycling and taking the bus. While recent studies continue to support this position, there are also several which consider active mode travel to be the most affected. Additionally, such research has been expanded to take higher respiration rates into account, which suggests pollutant intake disparities are increased even further. However, actual individual pollutant inhalation can vary considerably depending on physical characteristics, fitness level and overall health. For this reason, results that factor in breathing rates should be viewed with caution.

For total pollutant exposure, motorcycle commuters are clearly the most exposed. According to the bulk of the literature, motorcycle is then followed by bus, car, pedestrian, cycle, train/subway. While this order is equivocal - largely dependant on local environmental conditions and a range of other variables - it remains the status quo for now.

In order to more accurately advise the public on transport exposure, investigation must be carried out at the local level under realistic settings. The numerous conflicting results discussed in this review emphasise the effects just one or two conditions can have e.g. it appears the best way to reduce exposure in a vehicle is to re-circulate the air with air conditioning on, but this may not be suitable advice in countries that remain cool over summer. Planning the position of cycle paths and pedestrian walkways relative to traffic is one way of mitigating the effects of air pollution. The extent to which microscale differences impact exposure levels has not been fully explored and further research is required. While a small distance could make a big difference in some environments; built structures (e.g. canyon-effects), trees, topography, and ambient levels may prevent sizable improvements in others.

As years of air pollutant intake is potentially very harmful, there is an increasing need to find ways to avoid it as much as possible. Given the climbing global population, cheaper vehicles and rising incomes in developing nations, fuel pollutant reductions are not likely to be achievable without significant advances in fuel and emissions technology. While fuel modifications are relatively easy to implement, it will take decades to replace or upgrade the world's current vehicle fleet when viable petroleum alternatives are developed. Therefore it is vital to inform citizens so that they can make better behavioural choices for themselves. Even after factoring in increased respiration rates, the health benefits of choosing an active mode may far outweigh the negatives. This is an area which is likely to receive greater attention in future exposure research.

Chapter Three: Physical Setting, Tools, and Methodology

3.1 Introduction

This chapter outlines the exact process that took place to prepare for and to conduct the pollutant sampling, starting with the physical setting. Section 3.3 details all instruments and equipment needed for the study, 3.5 describes the preparation phase and 3.6 defines the sampling strategy. Sections 3.8 and 3.9 provide an overview of the required data corrections and analyses used.

3.2 Physical setting

3.2.1 Location

Traffic pollutant sampling was conducted in New Zealand's two largest urban areas, Christchurch and Auckland.

Christchurch city, situated at 43° 32'S, 172°, 37'E, sits on very flat Canterbury floodplains at the southern end of Pegasus Bay on the east coast of the South Island (Figure 1). Christchurch city covers 1610 km^2 and has a population of approximately 350,000, giving a population density of around 217 persons per km^2.

Auckland is situated at one of the narrowest parts of the country (36° 51'S, 174°, 46'E), a volcanic isthmus between the Waitemata Harbour on the Pacific Ocean and the Manukau Harbour on the Tasman Sea (Figure 1). The greater Auckland region, split into several territories, is the largest (6,059 km^2) and most populous New Zealand metropolis with 1.4 million inhabitants (234 persons per km^2).

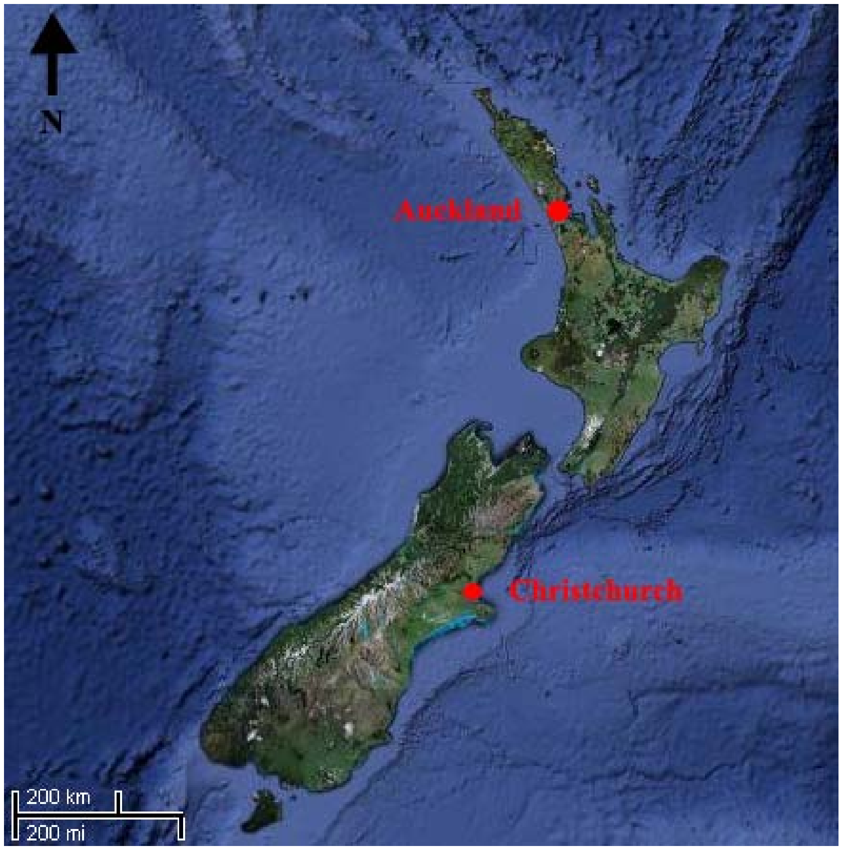

Figure 1 Physical setting of the two study cities

3.2.2 Local climatology and the influence of the physical setting

Christchurch has a relatively dry, temperate climate. During the warm summer months the city often receives strong nor'west winds. Winters are notably calmer with overnight temperatures often dropping below 0 °C. These cool, calm winter conditions often form a stable inversion layer above the city. Emissions from domestic heating combine with industry and traffic pollutants, forming a layer of smog under the inversion layer. As a result, far greater ambient coarse particle concentrations are experienced during winter, potentially affecting exposure samples (Town 2001). Ultrafine particles are negatively correlated with temperature, as higher temperatures are thought to increase coagulation which results in a rapid loss of concentrations as particles grow into larger size fractions (Kaur et al. 2006; Vinzents et al. 2005). Increased windspeed also negatively affects ultrafine particles whereas it can have the opposite effect with coarse particles as it aids resuspension (Zhu et al. 2002). While sampling under a range of conditions was important, calm conditions with moderate temperatures would provide the most comparable data. Hence it was decided that Autumn would be the optimum period.

Conversely, Auckland has a very wet, humid climate with the highest rainfall occurring in June and July. As the sampling instruments were not waterproof, data collection could not take place during wet weather, so it was ideal to have the sampling finished before the winter months.

With all of these variables considered, sampling was scheduled for March (Christchurch) and May (Auckland).

3.3 Instruments, equipment & tools

3.3.1 Sampling instruments

Instrument choice was primarily based on practical suitability and successful use in previous published research. Although resource availability was a factor in determining which would be used, the study ended up securing a collection of mid to top-range instruments that have been commonly used in past research and are still considered to be the industry standard (Table 6). GRIMM aerosol instruments have been widely used in previous fine to coarse particle studies ($PM_{1.0}$, $PM_{2.5,}$ PM_{10}), and TSI 3007s are considered the current leading portable instrument for measuring ultrafine particles (Thai et al. 2008; Tsai et al. 2008). Langan T15n CO measurement devices have also been successfully used in previous transport pollutant exposure research (Gómez-Perales et al. 2004; Kaur et al. 2005a; Lindén et al. 2008).

Table 6 Instruments used for data collection

Instrument	Measures	Sampling Range	Sampling Resolution	Other Capabilities	Technical Notes	Manufacturer	Number Employed
Langan T15n	Carbon Monoxide	0-200 ppm	0.05 ppm 1 second intervals	Temp, rH	Switching on display causes spikes in logged data. Has button which can be used as an 'event marker'	Langan Instruments, San Francisco, CA, USA	4
GRIMM Environmental Dust Monitor – Models 1.101, 1.107 & 1.108	PM_{10}, $PM_{2.5}$ and $PM_{1.0}$	1-6,500 $\mu g/m^3$	120 nm to 30 microns 6 second intervals	Temp, rH Moisture Compensation System	Counts particles across 32 size channels and use that information to estimate $PM_{1.0}$, $PM_{2.5}$ and PM_{10} mass	GRIMM Aerosol Technik GMBH & CO. KG, Ainring Dorfstraße, Germany	4
TSI 3007 Condensation Particle Counter (CPC)	Ultrafine particles	0-500,000 pt/cm^3	0.01 to >1.0 µm 1 second intervals	Although capable of counting particles as small as 10 nm (and even smaller), there is only a 50% chance (or less) at this size due to different particle makeup. The shape, surface area and solubility determine whether the particle takes on enough alcohol to be recognised.		TSI Incorporated, Knoxville, TN, USA	3
Kestrel 4500	Meteorological Data	N/A	1 minute intervals	Wind dir, wind speed, temp, wind chill, rH, barometric pressure and more		Nielsen-Kellerman Inc., Boothwyn, PA, USA	6
Nokia N82 GPS Cellular Phone	GPS co-ordinates, sound, photographs	N/A	3 second intervals		Ran custom software to log GPS co-ordinates, sound and low-res images	Nokia Inc., Keilaniemi, Espoo, Finland	4

3.3.2 Sampling kit development

Four Kincrome heavy duty tool kit bags were purchased to house the sampling instruments and Nokia N82 phones. The kit bag main compartment provided a perfect fit for a 3007 and a GRIMM dust monitor sitting side by side. Instrument inlet tubes were positioned horizontally using an adjustable plastic stalk. The Langans and Kestrels sat in the front pockets of the kit and the Nokia phones were attached to an adjustable clip-in mobile phone holder (Figure 2).

Due to high concentration recording limitations for the 3007s, a filter had to be developed to dilute incoming values. Concentrations behind buses and other smoky vehicles often exceed 200,000 pt/cm³ - 3007s can only reliably record concentrations up to 100,000 pt/cm³ - hence diluters were put together to dilute values to approximately $1/10^{th}$. Knibbs et al. (2007) have observed coincidence-related undercounting at concentrations greater than 100,000 pt/cm³. This occurs due to multiple particles simultaneously passing through the single-particle counting optics. Diluters were made by crimping the end of a bicycle valve to create a very small orifice. It was then attached to a plastic tube connected to a HEPA TSI zero-check filter with a Y-type flow splitter to draw in 'dead air' (Figure 2).

Solid steel mounting racks were made to securely hold the kits in place while used on bicycles. They clipped into brackets attached to the handlebars and were also secured with hose clamps and cable ties for extra strengthening. The kits themselves were attached to the racks with tie down cables, bungee cords and G-clamps (Figure 2).

Figure 2 Example of a sampling kit attached to a cycle

3.3.3 Logging software and analysis tools

All logged data was downloaded using the instruments' proprietary software - Hoboware Pro (Langan), DustMonitor (GRIMM), Aerosol Instrument Manager (3007) and Kestrel Weather Tracker (Kestrel). Data was then exported into Microsoft Excel formats, manually collated into master spreadsheets and averaged up to a uniform logging interval (6 seconds) for analysis, using NI LabVIEW.

A customized logging application (GEOGDataLogger) was written to run on the Nokia N82 phones. The software recorded GPS coordinates, sound and took photographs every 3 seconds. Data could then be mapped using another custom application written for the project, GRC Media Mapper (see Bartie & Kingham 2009). This software displayed pollutant concentrations to the left of the screen, along with mapped GPS points and still images to the right (Figure 3).

Pollutant concentration GPS co-ordinate maps were produced by plotting XY data and colour coding corresponding concentration values using ArcGIS 9.3. All other maps were created using a combination of Google Earth Pro 5.1 and Adobe Photoshop 7.0. For the cyclist study, the average distance between cyclist positions was calculated by digitising a series of 19 points at equal intervals along the section of road and ultilising ArcINFO's 'PointDistance_analysis' function.

Microsoft Excel 2003/2007 was used for primary summary statistics and descriptive statistics were produced using StatSoft Statistica 8.

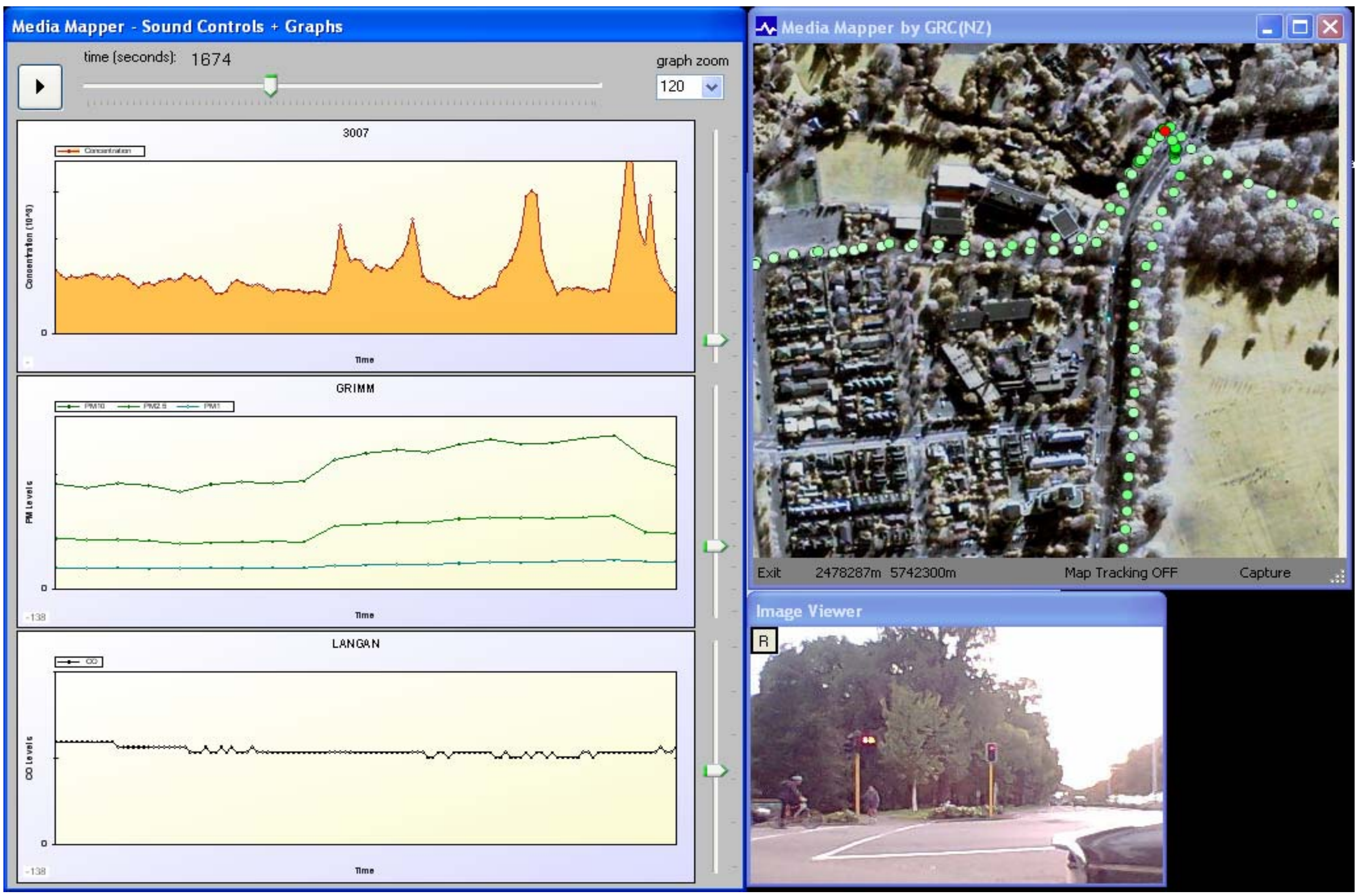

Figure 3 Screenshot of GRC Media Mapper

3.4 Study vehicles

The car used for all sampling in both cities was a stock-standard 1992 Toyota Corolla four door sedan. The car had had regular servicing and was thought to be running cleanly and efficiently. To prevent biased results, it was important the vehicle was not overly susceptible to the exchange of indoor/outdoor air. Refer to section 3.5.2 for a more detailed explanation.

The bus fleets in both Christchurch and Auckland cities predominantly consist of diesel engine buses. Red Bus Ltd in Christchurch operates German made MAN 17.223 model diesel buses and Auckland's MAXX runs the Swedish Scania L94 model. While both cities operate gas-turbine hybrid electric buses within the city centres, only the diesel models were ridden during the study.

The trains operating on the Auckland rail network consist of a combination of diesel multiple units (DMUs) and diesel locomotives. There are 9 ADK/ADB class DMUs, 10 ADL/ADC class DMUs, 2 DBR class locomotives and 14 DC class locomotives. The DMU engines are situated either at the front or rear of the units and the locomotives operate in push-pull mode. Trains are intermittently switched between different services so the position of the engine and the train type is somewhat random.

3.5 Pre-fieldwork tests and setup

3.5.1 Wind tunnel tests

Initial tests were conducted to determine how best to configure the instrument inlet tubing and to discover how different travelling speeds affected instrument readings. While less important in the closed microenvironments of cars and buses, it was necessary to investigate for the cycling and pedestrian elements. This was done by setting up a

GRIMM dust monitor at the rear of a wind tunnel and burning an incense stick to artificially create smoke. Particulate measurements of one minute duration were taken with the inlet tubing set at both vertical and horizontal positions at a range of tunnel speed settings. The results confirmed that the horizontal position rendered far higher concentration recordings (Figure 4). A significant drop-off in concentrations was also noted for both settings at 20 km/hr. A speed of approximately 15 km/hr was thought to be most appropriate for cyclists and inlet tubes were configured horizontally for all four kits.

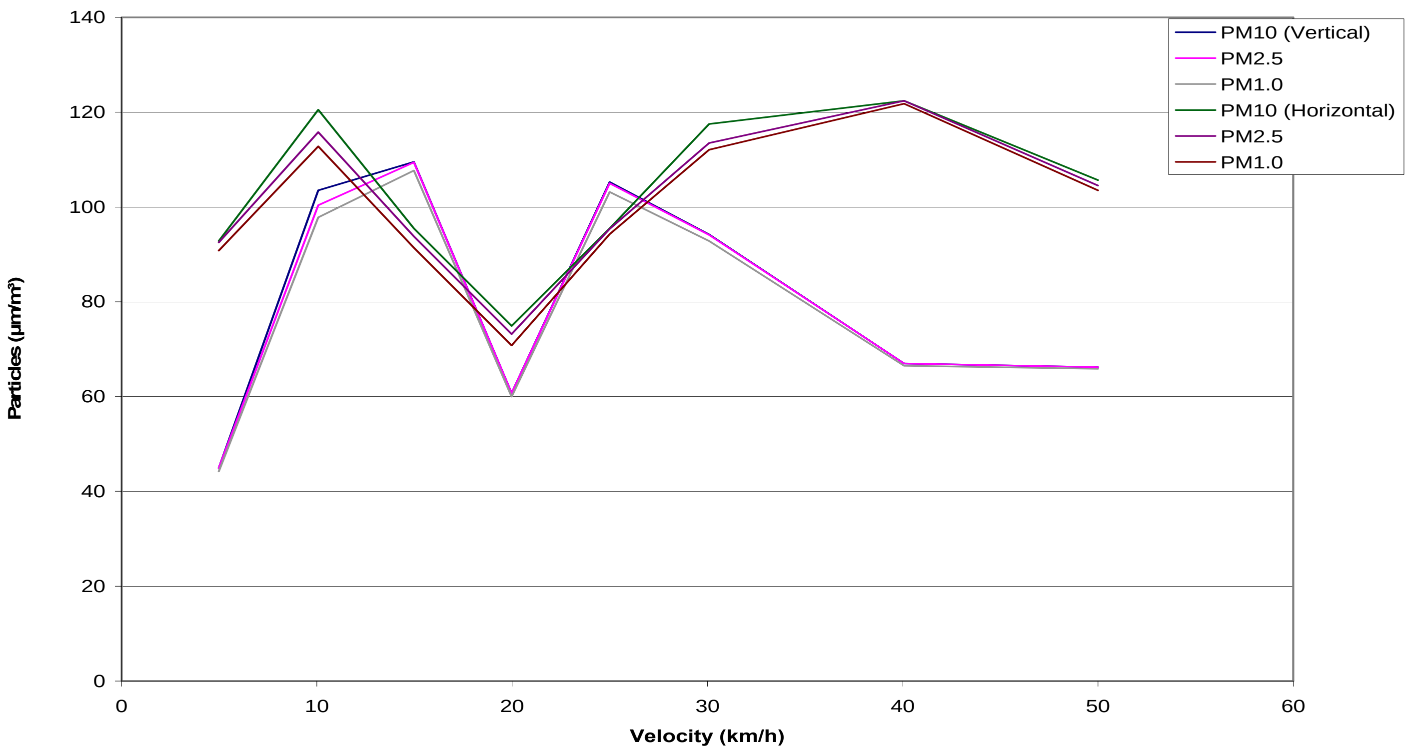

Figure 4 One minute average particulate recordings at different velocities and inlet tube positions

3.5.2 Car air exchange characterisation

Some vehicles, particularly older models, have higher rates of cabin pollutant decay and are also prone to greater self-pollution and outdoor infiltration. To prevent the results being affected by the use of a 'leaky vehicle', the air exchange rate of the study vehicle was measured. This was done by igniting an incense stick in the vehicle while traveling along a straight road at a constant speed of 50 km/hr and measuring the concentration decay with a GRIMM dust monitor (the incense was extinguished once the cabin filled with smoke). The results were then normalised by dividing each data point by the first and the data was then trimmed to the start point of exponential decay (Figure 5). An exponential curve of the form $y = Ae^{Bt}$ was applied (B = Air Exchange Rate), giving a raw AER value of 0.0117 units/s for PM_{10}. Multiplied to per hour, results were 42, 31 and 29 for PM_{10}, $PM_{2.5}$ and $PM_{1.0}$ respectively. Based on similar trials conducted by the National Institute of Water & Atmospheric Research (NIWA), these values fell within the normal range of 10 – 100 for a typical car with vents open, windows closed. This setting was decided to reflect the most typical setting in New Zealand and for this study, all sampling runs were completed with the windows closed, vents set to 'fresh' with the fan set to position 2 (of 4 possible settings). This configuration has been used in previous research in the UK, where it is also considered to represent typical urban driving behaviour (Briggs et al. 2008).

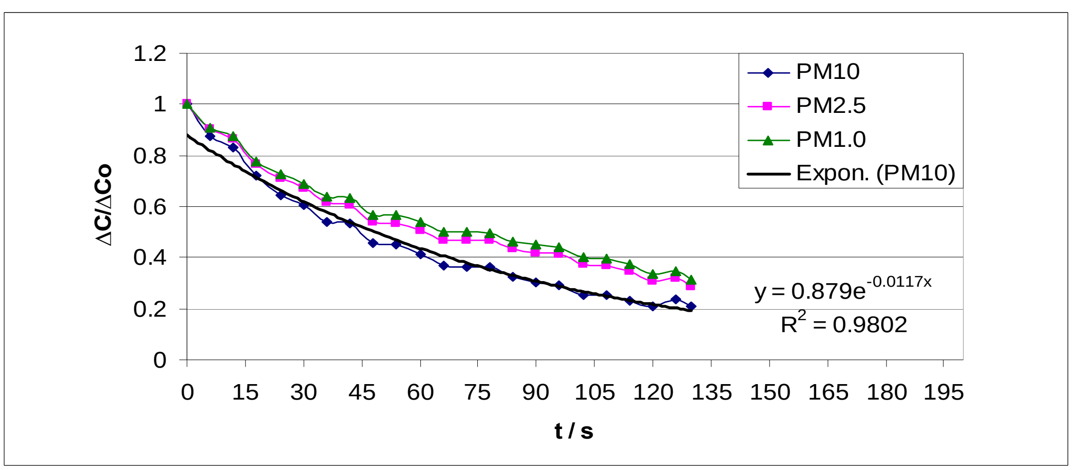

Figure 5 Air exchange characteristics of the study car

To fully confirm the vehicle was not prone to biased sampling results, a test was conducted using a control vehicle of similar age and design. The study car and a 1992 petrol Nissan Primera sedan were driven together (one behind the other) up and down a busy urban road for 40 minutes, followed by a 20 minute countryside drive along a two-lane regional highway. All windows were closed and vents set to fresh, with the fan on level 3 (of 4). Mean results for PM_{10}, $PM_{2.5}$ and $PM_{1.0}$ were in close agreement: 11, 5.9 and 4.2 µg/m³ (study car) and 9.9, 5.5 and 3.9 µg/m³ (control car).

3.5.3 Meteorological data

Background weather data required for the study included wind speed, wind direction and temperature. Precipitation was not considered as the sampling kits were not capable of operating in rainy weather. In order to obtain local scale 10 minute wind data for Christchurch, two Kestrel 4500s were mounted within the study area, one situated at 340 Main North Road and another at 69 Deans Avenue. This data was then checked against two larger meteorological sites located at Coles Place and the Department of Geography at Canterbury University. Data from the two northern sites was combined and averaged to provide overall results for the north-south route (Journey 1). The same was done for the east-west (Journey 2) route. The journey routes and the locations of the met sites are depicted in Figure 6 and Figure 7.

Mean wind data were calculated by averaging the wind vector components:

$$V_e = \Sigma\, [U_i \sin(A_i)]/N$$
$$V_n = \Sigma\, [U_i \cos(A_i)]/N$$

And calculating average wind speed and direction from the vector averages:

$$UV = (V_e^2 + V_n^2)^{1/2}$$
$$AV = ArcTan(V_e/V_n)$$

Wind speed was also included in the analysis as a fixed category of high (>2 m/s) or low (<2 m/s) readings. At <2 m/s, wind direction becomes very variable, consisting of numerous slow-moving eddies whereby the dispersive influence on pollutants becomes significantly reduced. The threshold of 2 m/s has been applied in previous air pollutant research for Christchurch (Marsh & Wilkins 2004).

Temperature recordings were taken from Kestrels attached to the sampling kits, accounting for variability throughout each journey.

For Auckland, background data was retrieved from the Auckland Regional Council Khyber Pass met station, situated in the middle of the sampling route (see Figure 10). A lack of resources and suitable sites made the use of static Kestrel sites too difficult.

3.5.4 Pilot study and sampling configuration

Two pilot runs were conducted along the Christchurch routes to test the equipment and to confirm sampling logistics were realistic in relation to bus timetables and cycling times. The timing of the runs was successful from the first trial but changes were made in regard to the equipment. The plastic inlet tubes were replaced with stainless steel tubes to ensure concentration measurements weren't affected by particles sticking to the inside of plastic tubing. It was also found that the 3007s were prone to 'tilt errors' when shaken around on the cycles. Tilt errors occur when the instrument optics are contaminated with alcohol and has been problematic in other recent cycle research (Boogaard et al. 2009). The presence of front suspension on the cycle appeared to almost completely alleviate tilt events, hence one of the cycles was changed.

3.6 Sampling strategy

The sampling strategy involved two distinct lots of data collection. Inter-modal data was collected along set commuting routes while comparative cyclist data was recorded separately by three cyclists riding at different distances from the traffic flow, along a short section of road.

Sampling took place during February 26 – April 1 (Christchurch) and April 27 – May 21 (Auckland). Autumn was chosen as the ideal sampling period due to moderate rainfall and mild temperatures. Warmer temperatures also resulted in a reduced risk of domestic heating emissions augmenting traffic pollutants, especially for Christchurch which has a far cooler climate during winter.

3.6.4 Inter-modal sampling

Four commuters set out on specified routes that were designed (as closely as possible) to replicate typical commutes to and from sites of work or study. Journeys did not fully reflect the most logical commuting route for the car and main cyclist as it was important they took the same path as the bus commuter. Sampling trips were made during rush hour traffic to reflect when most people travel and to yield higher (more comparable) concentration recordings. The Christchurch study allowed for the replication of two separate journeys per sampling run – one from the northern fringe of the city to the city centre and then another to the University of Canterbury.

A total of 27 Journey 1 and 26 Journey 2 legs were completed in Christchurch with another 26 journeys completed in Auckland. Data was lost for multiple journeys and not all of the collected data was useful.

For Christchurch, the modes consisted of bus (Kit 1), car (Kit 2), cycle off-road (Kit 3) and cycle on-road (Kit 4). One cyclist rode an off-road route via dedicated cycle-ways, through parks and backstreets, while another took exactly the same route as the bus and car. This was to explore the exposure implications of taking a longer off-road route versus a more direct route on-road.

In Auckland, bus became Kit 3, Kit 1 became train and there was no off-road cycle mode due to equipment restrictions and lack of suitable comparative routes. Kits 2 and 4 remained the same as for Christchurch. The cyclist, car and bus again travelled the same route which ran as closely as possible to the train line.

As there were only three 3007s available among four kits, one was switched between kits near the end of each sampling campaign to ensure data was collected across all modes. In Christchurch, a 3007 was placed in Kit 3 for Runs 1-17 and then moved to Kit 1 for the remaining ten runs. In Auckland, a 3007 was switched from the bus to the train for the final journey only. NIWA had already collected substantial UFP data for the train mode and data loss and time constraints meant greater priority was given to the other three modes.

3.6.5 Cyclist sampling: Effect of proximity to traffic

To investigate differences in microscale exposure levels, a number of sampling runs were made using three cyclists riding simultaneously at different distances from the flow of traffic. One cyclist was situated on the road right next to traffic, another on the footpath 4.5 - 7 metres away and the third was on an off-road path approximately 17.5 – 19 metres away on average. Cylists rode along a specified road/path section and then turned around and went back the other way, repeating the process until at least 20 lengths were completed. This was done three times in each city to account for different weather conditions.

An additional scenario was tested in Christchurch where the off-road cyclist rode on a path in the middle of a park, ~700 metres north of the other two cyclists. The purpose was to ascertain how much exposure decreased at this distance compared to concentrations just 5 – 20 metres away. The cyclist in the middle of the park was always at least 30 metres away from traffic sources, with the radius of distance to traffic in all directions as large as 420 metres. The extent to which pollutant levels decrease at very small distances from traffic has important implications for the positioning of cyclist and pedestrian pathways. While microscale computer modeling may provide clearer answers than monitoring by means of numerous fixed sites, it may not be entirely representative of exposure whilst moving.

3.7 Study areas

3.7.4 Christchurch inter-modal routes

The Christchurch run was split into two separate journeys to replicate two normal commutes within the rush hour timeframe. The first of these journeys, referred to as 'Journey 1' or J1, ran 8.2 km from 340 Main North Road to the city bus exchange at 36-54 Lichfield Street (Figure 6). On arrival, the car driver parked in a parking lot above the bus terminal and met the bus commuter and the cyclists at Cashel Mall (a street closed off to traffic). After a short wait, 'Journey 2' (J2) to the University of Canterbury Geography department commenced (Figure 7). Journey 2 was 7.5 km long. As the bus was, in terms of timing, the least flexible, the sampling schedule was designed to fit around the bus timetable (Table 7).

Table 7 Christchurch inter-modal run timetable

Morning	**Bus**	**Car**	**Cycle – Off-Road**	**Cycle – On-Road**
Meet Redwood	**7:40**	**7:40**	**7:40**	**7:40**
Depart Redwood	7:51 (#12)	7:50	7:45	7:45
Arrive BusX	8:15	8:15	8:25	8:15
Meet Cashel Mall	**8:25**	**8:25**	**8:25**	**8:25**
Depart BusX	8:42 (#21 or 3)	8:35	8:30	8:42
Arrive UC	8:57	8:50	9:00	9:00
Meet GEOG	**9:00**	**9:00**	**9:00**	**9:00**
Afternoon				
Meet GEOG	**4:45**	**4:45**	**4:45**	**4:45**
Depart UC	4:55 (#21)	4:55	4:50	4:50
Arrive BusX	5:13	5:10	5:20	5:20
Meet Cashel Mall	**5:20**	**5:20**	**5:20**	**5:20**
Depart BusX	5:25 (#12)	5:30	**5:25**	5:25
Arrive Redwood	6:01	6:00	6:05	6:00
Depart Redwood	**6:10**	**6:10**	**6:10**	**6:10**

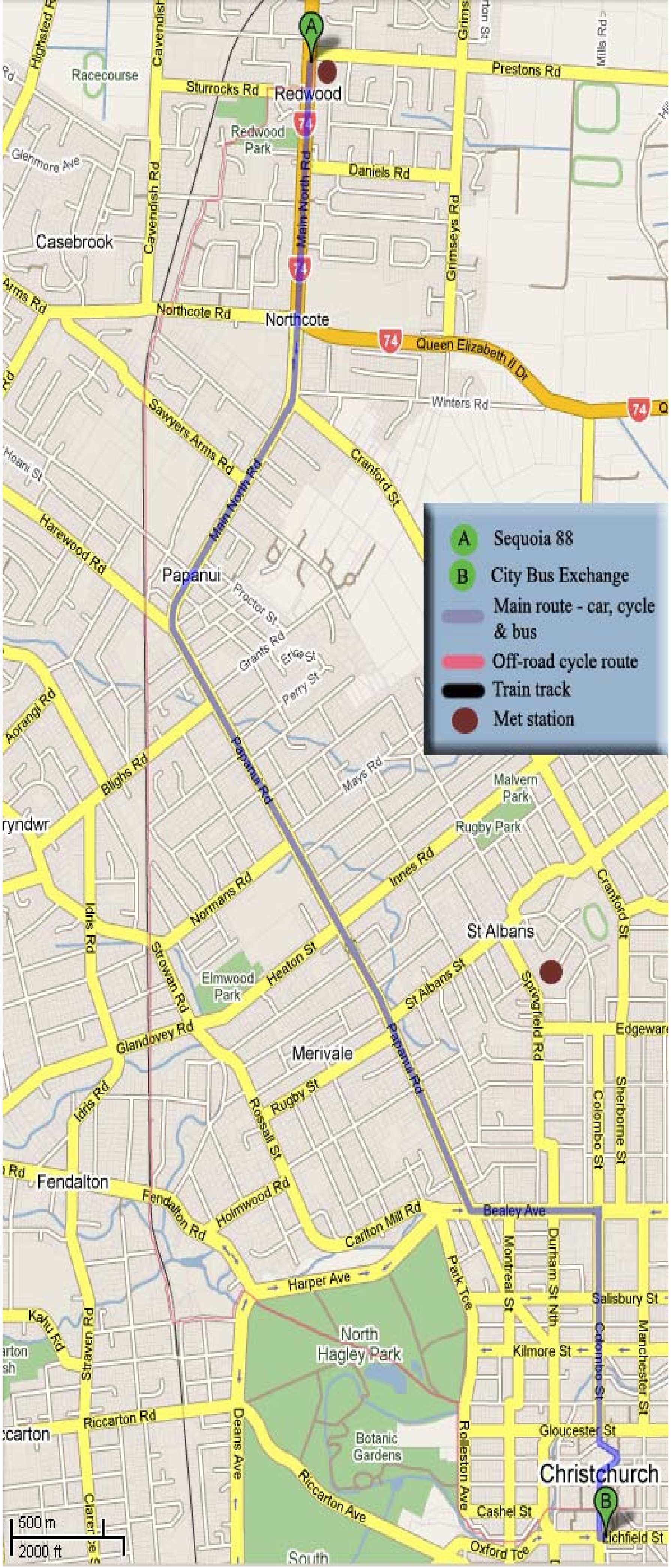

Figure 6 Journey 1 – Redwood to Christchurch City Bus Exchange

Figure 7 Journey 2 - Christchurch City Bus Exchange to University of Canterbury

3.7.5 Christchurch cyclist study area

The cycling aspect of the Christchurch study was conducted along the Riccarton Avenue stretch of Hagley Park (Figure 8). This area was chosen as it has both a footpath and an off-road track only 19 metres from the road. Riccarton Avenue is also very busy throughout the day and the parking spaces are usually full (Figure 9). Parked cars were thought to possibly provide a protective barrier, resulting in lower concentrations on inside paths. The area is also well vegetated with large trees and gardens. Figure 10 provides a clearer idea of the sampling setting and path layout. Figure 9 shows the travel direction of the three paths. The path marked in red to the north of the map shows the location of the third cyclist during the additional test scenario. One cyclist was situated on-road, the other off-road and the third, right in the centre of the park.

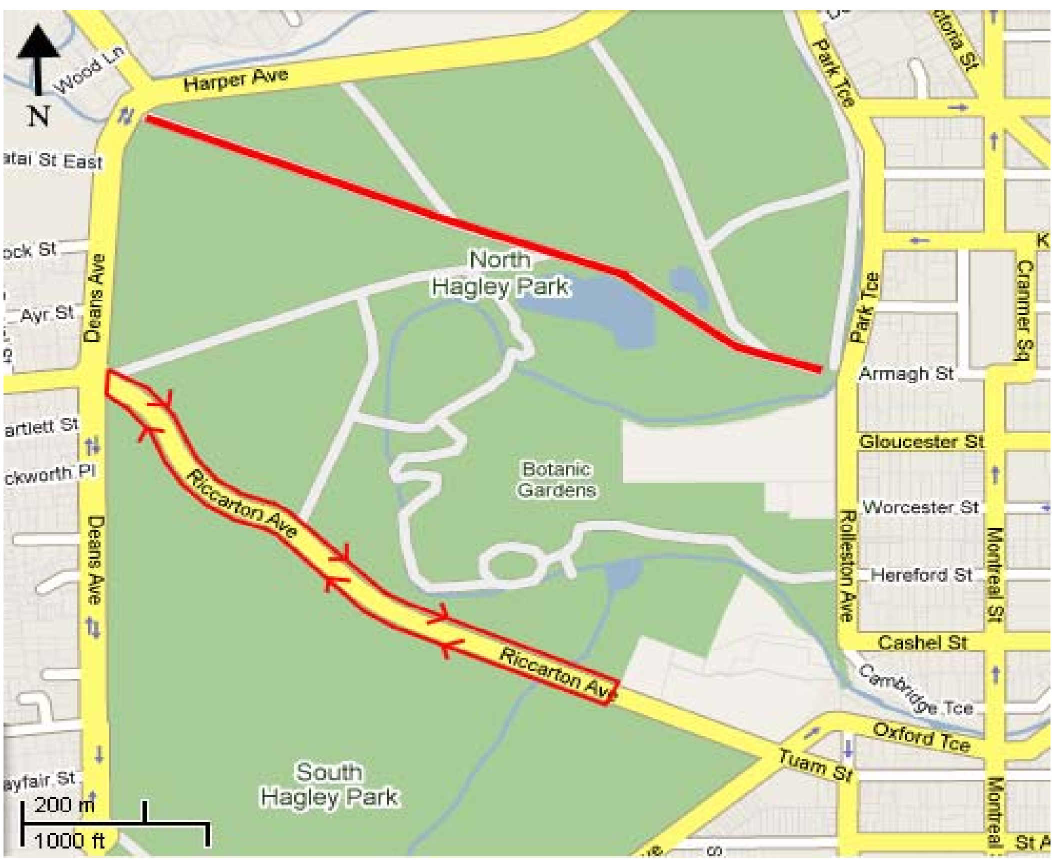

Figure 8 Christchurch cycle sampling area showing location and direction of travel

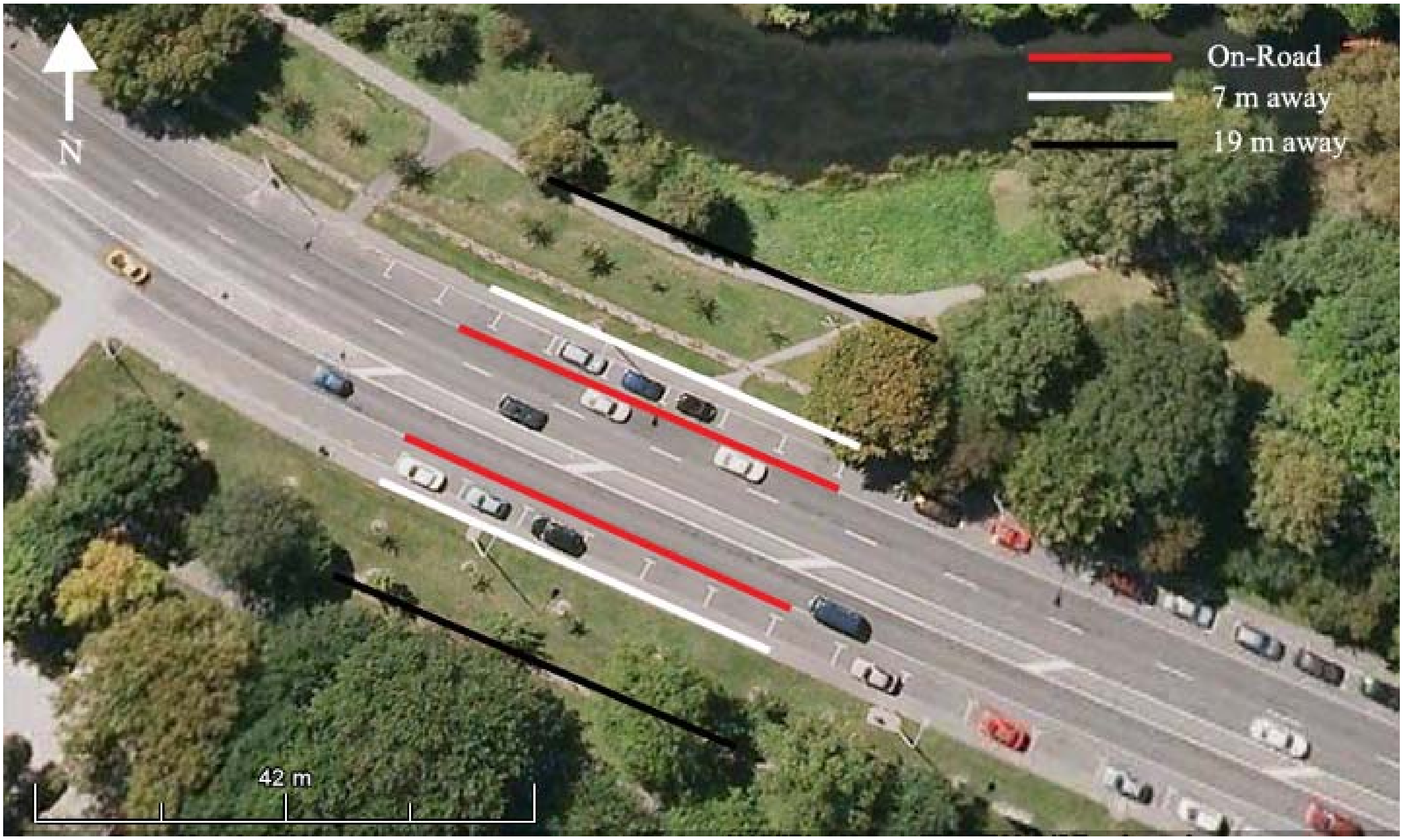

Figure 9 Satellite image of Christchurch cycle sampling area showing one section and the position of travel paths on both sides of the road

3.7.6 Auckland inter-modal route

The Auckland route ran from 947 New North Road at Mt Albert to NIWA headquarters at Market Lane in the city centre (Figure 10). This route was chosen due to its: proximity to the train track; proximity to volunteers' residences; use as a key commuting route to the city centre; use as a key bus route featuring dedicated rush hour bus lanes. The car, bus and cycle traveled along exactly the same route but the bus commuter walked part of the journey; to and from the Victoria Street bus station. Similarly, the train commuter walked part of the leg, to and from the Britomart Transport Centre along the same route as the car and cycle. The car and cycle route also varied slightly during the afternoon due to ‘Bus Only’ turning restrictions but this was not considered to significantly alter the results (see Figure 10). The total distance of the morning journey was 9 km and the afternoon journey was slightly longer at 9.4 km. The timing of the runs, outlined in Table 8, was designed to fit as closely as possible to bus and train travel times.

Table 8 Auckland inter-modal run timetable

Morning	**Bus**	**Car**	**Train**	**Cycle**
Meet Mt Albert	**7:40**	**7:40**	**7:40**	**7:40**
Depart Mt Albert	7:51 (#210)	7:50	7:53	7:50
Arrive city station	8:27	-	8:18	-
Depart city station	8:27	**-**	8:23	**-**
Arrive at NIWA	**8:37**	**8:35**	**8:39**	**8:31**
Afternoon				
Meet NIWA	**4:25**	**4:25**	**4:25**	**4:25**
Depart NIWA	4:25	4:45	4:40	4:45
Arrive city station	4:35	-	4:52	-
Depart city station	4:48 (#211)	-	4:58	-
Arrive Mt Albert	**5:24**	**5:30**	**5:21**	**5:30**

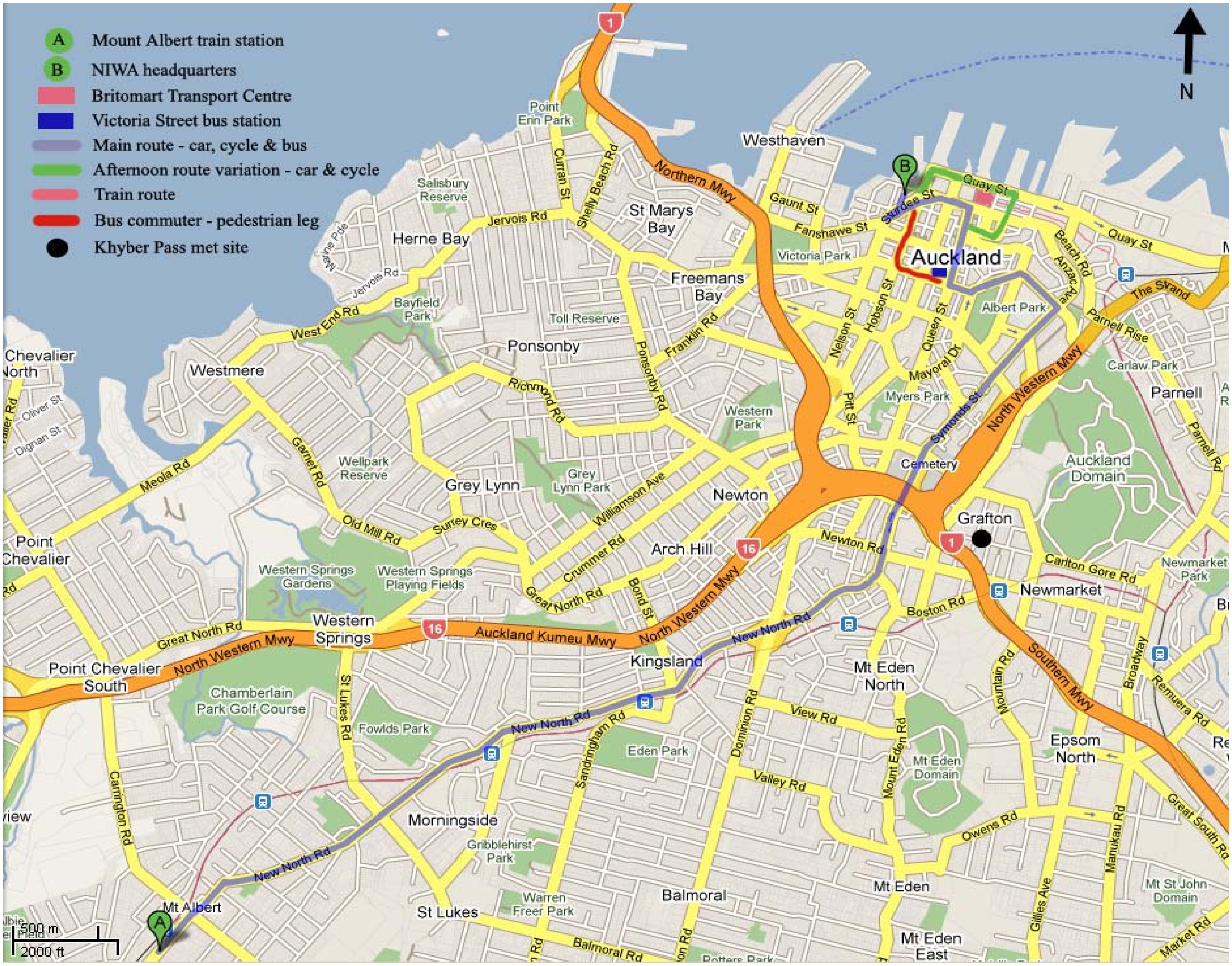

Figure 10 Auckland inter-modal sampling route

3.7.7 Auckland cyclist study area

Tamaki Drive, St Heliers Bay was chosen for the cycle sampling in Auckland. St Heliers Bay is an affluent area lined with boutique stores and expensive restaurants along the landward side of Tamaki Drive. It is also popular location for water sports and family recreation. The area provided a 500 metre section consisting of a shared cycle and pedestrian walkway as well as a boardwalk running alongside the beach. This made for a suitable replication of the study area in Christchurch except the footpath and on-road cyclists were slightly closer together and only one side of the road could be used (Figure 11 and Figure 12).

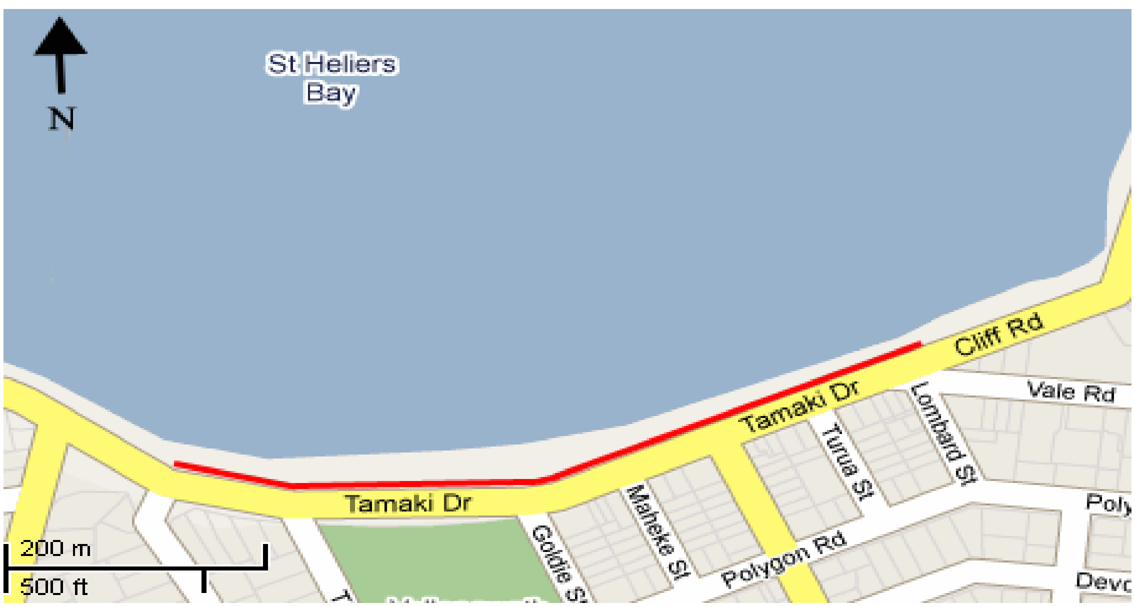

Figure 11 St Heliers Bay cycle sampling area

Figure 12 Satellite image of St Heliers Bay cycle sampling area

3.8 Sample data correction

3.8.1 Langan data

The Langan CO data required a considerable amount of correction, including manual correction and adjustment for temperature error. After collating the data, it was scanned by eye for extreme peaks and unusual troughs where readings dropped below zero. Many spikes of only 1-2 seconds duration were present, indicating an erroneous reading as the slow response time of these instruments means they cannot record such rapid spike events. All run data was plotted on a time series and suspicious sudden peaks above 5 ppm were removed manually. The same was done for recordings below zero parts per million. Langans also seem to 'black out' on occasion, reporting numerous instances of 0.031 ppm. Following these events, the instrument usually takes up to 30 seconds to recover. Such events and the subsequent 30 seconds of data were removed. Further adjustment was then applied to correct temperature error. An electrical current is generated by the result of the chemical interaction between CO and CO_2, which is detected by the electrochemical sensor. This causes temperature-dependant variation of output from the sensor, consequently requiring correction to recorded data. At temperatures above 20º C, the instrument over-reads, while at lower temperatures, under-reporting occurs (Langan 2006). After applying appropriate temperature corrections, CO errors were corrected to within +/- 0.2 ppm.

3.8.2 3007 data

A minor issue with the TSI 3007s was the occurrence of tilt errors, most commonly experienced while cycling and walking with the kits. Tilt errors cause the instrument to log false values of 1.68E+07 for around 8-10 seconds and then values gradually increase from less than 10 pt/cm^3 as the pump recovers. It takes around 20 seconds for the instrument to fully recover and all 'tilt error blocks' of around 30 seconds in duration were manually removed. Occasionally, automatic recovery did not occur and the instrument had to be reset by removing the battery pack, resulting in substantial data loss when left unnoticed.

A dilution system developed by Knibbs et al. (2007) was used during UFP measurements with the TSI 3007 instruments, meaning all recorded data first needed to be multiplied by ten. While the goal was to dilute concentrations to 1/10th of incoming values, it was discovered that the behaviour of the diluter was likely to change over time as the HEPA filter absorbed small particles. Two of three filters retained the original set dilution ratio of 10:1 throughout the Christchurch sampling campaign, while the ratio for the third filter grew substantially over time. The change in performance was able to be observed from the approximate 20 minute period when all three instruments were recording together in between Journey 1 and Journey 2, in the city centre. Data from the affected diluter was plotted against the reliable data, clear outliers were removed, and a linear regression with a forced zero intercept applied. This was done for every run and the resulting slope applied to the original data. Unfortunately, while the instruments were co-located in an office environment following each run in Auckland, the sampling campaign lacked a similar period where the instruments were co-located in a high concentration environment. Due to an undocumented change in performance among two of the filters, it was decided the data was not correctable and all Auckland UFP data was removed from the analysis.

3.8.3 GRIMM data

The major concern surrounding the recorded PM concentration data was that none of the Grimms were characterised against a reliable reference instrument. The preferred method is to concurrently run the Grimm dust monitors alongside a Tapered Element Oscillating Microbalance (TEOM), measure association across size fractions and adjust if required (Chan et al. 2004, Tsai et al. 2008). TEOMs are Environmental Protection Agency (EPA) approved Federal Reference Method (FRM) samplers. Secondly, Grimm Environmental Dust Monitors or Portable Aerosol Spectrometers (PAS) are typically calibrated under controlled conditions using polydisperse aerosol composed of Arizona test dust. Grimms have been observed to size and count particles differently when sampling other dust types such as monodisperse aerosols (Peters et al. 2006). Although monodisperse concentrations are extremely rare in normal environments, the instruments may not always function in the same fashion when encountering dusts of other composition. For these reasons, absolute PM values presented within this study should be viewed with due caution. While absolute values are presented, where possible, results are discussed in terms of ratios and relative percentage differences.

One instrument, the Grimm 1.108, under-read throughout the entire campaign by a factor of two. This was determined using the same correction method used for the 3007 diluters; using the most reliable instrument as a reference. A second instrument, the Grimm 1.101, produced strange output with values ranging from the tens to thousands. It was hoped at least some of the data could be recovered, but with no clear pattern observed, all data was rendered unusable and excluded from analysis. This meant all PM data for the train mode was lost.

3.9 Statistical analysis

Overall summary statistics were produced using Microsoft Excel. Raw data was then transferred into Statsoft Statistica for advanced analysis. As there was such a large amount of skewed data to analyse, advanced linear non-parametric tests were required. Analysis of variance (ANOVA) was chosen as the most suitable method, applied in the form of a linear mixed effects model. Analysis was blocked by Run (sampling trip) or Leg (for cyclist sampling) as the random factor. Other variables included Mode, Journey (1 or 2, for Christchurch only), Direction, Time of Day, Wind Strength (average wind speed categorised as either high [>2 m/s] or low [<2 m/s]), Wind Influence (upwind or downwind) as fixed effects, with Average Temperature and Average Wind Speed as covariates. All variance components analyses (VCA) were computed using Type I sequential sums of squares (Type I SS) and Error d.f. were calculated using Satterthwaite's method of denominator synthesis (Satterthwaite 1946). A star (*) next to any factor in the output tables specify that tests assume entangled fixed effects are zero. The type of relationship between any significant independent factors and the dependant variable (pollutant sampled) was determined using correlation matrices.

3.10 Summary

This chapter first introduced the physical setting of the two study cities and briefly discussed the influence of local topography and climatology. It gave a detailed account of the sampling methodology, including the pilot study, timetabling and an overview of the study design. Various problems were encountered with data correction and analysis, which were outlined as clearly as possible. The final section explained the reasoning behind the chosen statistical models, which set the scene for the next two chapters; the results chapters.

Chapter Four: Inter-modal Results and Discussion

4.1 Introduction

This chapter provides all summary statistics and ANOVA tables based on the raw results. All logged data were trimmed to include only samples taken while traveling. Results for the two Christchurch routes were analysed together for the summary statistic tables, but 'Journey' was included as a fixed factor to ascertain whether any significant differences occurred between routes. Overall mean results for each city are then discussed in relation to previous studies consisting of similar methodologies.

4.2 Christchurch

4.2.1 Carbon monoxide

Mean results for Christchurch ranked the car mode as the most exposed, at around 2.6 times higher than bus and the on-road cyclist. The off-road cyclist was the least exposed and 4.4 times lower than car. The max concentration was 52.33 ppm, recorded by the car mode (see Table 9). All modes recorded the lowest possible resolution of 0.05 ppm as the minimum value. There was a statistically significant difference across modes ($F_{3,172.6}$ =33.45, p<0.001). Both wind speed (p=0.003) and temperature (p=0.005) were negatively correlated with CO (Table 10). Time of day was non-signficant, as was wind speed, which was grouped into fixed categories. There was a significant interaction between Journey and Mode (p=0.008), with CO levels being considerably higher for Journey 1, but only for the car and on-road cyclist.

Table 9 Summary statistics for CO modal exposure in Christchurch

Mode of transport	*N* Journeys (samples)	Arithmetic Mean (+/-0.2 ppm)	Std. Dev.	Min	Max	Median	-95% Cnf.Lmt	+95% Cnf.Lmt
All samples	*184 (42153)*	*1.35*	*2.00*	*0.05*	*52.33*	*0.66*	*1.33*	*1.37*
Bus	49 (11172)	1.10	1.12	0.05	8.85	0.86	1.08	1.12
Car	48 (8810)	2.87	3.22	0.05	52.33	2.37	2.81	2.94
Cycle on-road	39 (9161)	1.12	1.58	0.05	25.90	0.49	1.08	1.15
Cycle off-road	48 (12110)	0.65	0.88	0.05	22.75	0.37	0.64	0.67

Table 10 ANOVA results for CO modal exposure in Christchurch

Variable	Effect (F/R)	df Effect	MS Effect	df Error	MS Error	F	p
Avg Wind Speed	***Fixed**	**1**	**27.59**	**34.19**	**2.66**	**10.37**	**0.003**
Avg Temp	***Fixed**	**1**	**32.90**	**25.66**	**3.43**	**9.60**	**0.005**
{1}Journey	***Fixed**	**1**	**13.12**	**75.39**	**1.76**	**7.47**	**0.008**
{2}Time of Day	*Fixed	1	1.30	23.75	3.74	0.35	0.561
{3}Mode	***Fixed**	**3**	**39.47**	**172.65**	**1.18**	**33.45**	**<0.001**
{4}Wind Strength	Fixed	1	4.50	146.19	1.38	3.26	0.073
{5}Run	**Random**	**25**	**3.52**	**150.00**	**1.10**	**3.21**	**<0.001**

4.2.2 PM_{10}, $PM_{2.5}$ and $PM_{1.0}$

For average PM_{10} exposure, the bus mode ranked highest at 45.79 µg/m³ followed by the cyclists and then the car, which was the least exposed at 27 µg/m³ (Table 11). Results were significant across modes ($F_{3,103.6}=4.21$, $p=0.007$). Similar results were found for $PM_{2.5}$, except the off-road cyclist had slightly lower exposure than the on-road cyclist, which also had a far lower PM_{10} result ($F_{3,84.09}=3.88$, $p=0.012$). The off-road cyclist received the lowest recordings for $PM_{1.0}$; 7%, 31% and 43% lower than car, on-road cyclist and bus, respectively ($F_{3,90.52}=5.08$, $p=0.002$). The influence of wind strength was significant for PM_{10} and $PM_{2.5}$, with lower median scores for the high wind speed category of >2 m/s. Average wind speed was negatively correlated with all fine-coarse PM fractions, but was only significant for $PM_{1.0}$ ($p=0.004$). Full ANOVA results are presented in Table 12.

Table 11 Summary statistics for PM modal exposure in Christchurch

Mode of transport		***N* Journeys (samples)**	**Arithmetic Mean** (μg/m³)	**Std. Dev.**	**Min**	**Max**	**Median**	**-95% Cnf.Lmt**	**+95% Cnf.Lmt**
All samples	*PM_{10}*	*143 (34221)*	*38.12*	*30.40*	*4.00*	*1138.80*	*32.20*	*37.80*	*38.44*
	$PM_{2.5}$		*19.29*	*15.84*	*1.30*	*505.40*	*16.60*	*19.13*	*19.47*
	$PM_{1.0}$		*10.44*	*10.77*	*0.30*	*312.30*	*7.80*	*10.33*	*10.55*
Bus	PM_{10}	43 (9922)	45.79	40.09	4.80	1138.80	36.20	45.00	46.58
	$PM_{2.5}$		23.59	20.54	2.20	268.00	18.80	23.17	24.01
	$PM_{1.0}$		13.74	13.70	0.80	118.60	10.40	13.47	14.01
Car	PM_{10}	19 (3950)	27.00	12.96	4.00	94.84	24.36	26.60	27.41
	$PM_{2.5}$		14.03	6.97	1.60	62.80	11.79	13.82	14.25
	$PM_{1.0}$		8.40	5.48	0.80	55.90	6.67	8.23	8.58
Cycle Off-Road	PM_{10}	47 (12351)	37.45	31.41	4.30	573.50	31.20	36.90	38.01
	$PM_{2.5}$		17.94	16.63	1.30	505.40	15.40	17.65	18.24
	$PM_{1.0}$		7.83	10.04	0.30	312.30	5.70	7.66	8.01
Cycle On-Road	PM_{10}	34 (7984)	35.12	14.47	9.01	117.52	31.92	34.81	35.44
	$PM_{2.5}$		18.99	8.61	4.98	77.55	16.43	18.80	19.17
	$PM_{1.0}$		11.38	8.14	1.86	73.13	8.45	11.20	11.56

Table 12 ANOVA results for PM modal exposure in Christchurch

PM_{10}							
Variable	**Effect (F/R)**	**df Effect**	**MS Effect**	**df Error**	**MS Error**	**F**	**p**
Avg Wind Speed	*Fixed	1	213.13	31.32	674.63	0.32	0.578
Avg Temp	*Fixed	1	79.54	26.35	809.36	0.10	0.756
{1}Journey	*Fixed	1	2.28	61.79	427.65	0.01	0.942
{2}Time of Day	*Fixed	1	2479.86	24.80	873.70	2.84	0.105
{3}Mode	***Fixed**	**3**	**1417.46**	**103.61**	**336.56**	**4.21**	**0.007**
{4}Wind Strength	**Fixed**	**1**	**2937.85**	**99.04**	**343.68**	**8.55**	**0.004**
{5}Run	**Random**	**25**	**864.32**	**109.00**	**239.89**	**3.60**	**<0.001**
$PM_{2.5}$							
Avg Wind Speed	*Fixed	1	522.98	29.27	316.93	1.65	0.209
Avg Temp	*Fixed	1	53.76	25.89	385.27	0.14	0.712
{1}Journey	*Fixed	1	188.53	51.62	182.71	1.03	0.314
{2}Time of Day	*Fixed	1	427.87	24.80	418.08	1.02	0.321
{3}Mode	***Fixed**	**3**	**531.77**	**84.09**	**136.98**	**3.88**	**0.012**
{4}Wind Strength	**Fixed**	**1**	**752.26**	**81.42**	**139.33**	**5.40**	**0.023**
{5}Run	**Random**	**25**	**411.46**	**107.00**	**84.11**	**4.89**	**<0.001**
$PM_{1.0}$							
Avg Wind Speed	***Fixed**	**1**	**1182.75**	**29.92**	**120.22**	**9.84**	**0.004**
Avg Temp	*Fixed	1	331.70	26.06	146.83	2.26	0.145
{1}Journey	*Fixed	1	193.28	53.62	71.45	2.71	0.106
{2}Time of Day	*Fixed	1	276.79	24.84	159.53	1.74	0.199
{3}Mode	***Fixed**	**3**	**271.39**	**90.52**	**53.46**	**5.08**	**0.003**
{4}Wind Strength	Fixed	1	209.14	85.99	54.87	3.81	0.054
{5}Run	**Random**	**25**	**157.68**	**109.00**	**34.38**	**4.59**	**<0.001**

4.2.3 Ultrafine particles

Ultrafine exposure was highly significant across modes ($F_{3,105.77}$ =13.12, p<0.001) and significantly negatively correlated with average wind speed (p<0.001) and average temperature (p=0.007). Time of day was also significant (p=0.009), with far higher levels generally experienced during mornings than in the afternoons – overall means by morning and afternoon for all samples were 81691 and 62095 pt/cm^3 respectively. The off-road cyclist was by far the least exposed; 53%, 69% and 70% lower than the on-road cyclist, bus and car, respectively (Table 13). ANOVA results are given in Table 14.

Table 13 Summary statistics for UFP modal exposure in Christchurch

Mode of transport	***N* Journeys (samples)**	**Arithmetic Mean** (pt/cm³)	**Std. Dev.**	**Min**	**Max**	**Median**	**-95% Cnf.Lmt**	**+95% Cnf.Lmt**
All samples	*184 (28909)*	*52895.28*	*70818.37*	*23.33*	*1304048*	*26993.33*	*52078.93*	*53711.63*
Bus	19 (3809)	76481.74	68833.83	544.83	506990	57056.67	74295.07	78668.40
Car	42 (7802)	77654.53	85145.92	3202.98	970369.02	46594.22	75764.91	79544.16
Cycle on-road	38 (8545)	49842.85	71568.47	85	1304048	25516.67	48325.19	51360.51
Cycle off-road	33 (8753)	23541.97	37176.55	23.33	741751.67	11115	22763.04	24320.90

Table 14 ANOVA results for UFP modal exposure in Christchurch

Variable	**Effect (F/R)**	**df Effect**	**MS Effect**	**df Error**	**MS Error**	**F**	**p**
Avg Wind Speed	***Fixed**	**1**	**5.93E+10**	**31.68**	**2.69E+09**	**22.05**	**<0.001**
Avg Temp	***Fixed**	**1**	**2.48E+10**	**28.03**	**2.98E+09**	**8.32**	**0.007**
{1}Journey	*Fixed	1	2.47E+09	71.60	1.70E+09	1.45	0.233
{2}Time of Day	***Fixed**	**1**	**2.85E+10**	**23.90**	**3.49E+09**	**8.16**	**0.009**
{3}Mode	***Fixed**	**3**	**1.88E+10**	**105.77**	**1.43E+09**	**13.12**	**<0.001**
{4}Wind Strength	Fixed	1	4.76E+08	106.49	1.42E+09	0.33	0.565
{5}Run	**Random**	**25**	**3.32E+09**	**98.00**	**1.08E+09**	**3.09**	**<0.001**

4.3 Auckland

4.3.1 Carbon monoxide

Mean results for Auckland ranked the car mode as the most exposed, at 2.3 times higher than bus and the on-road cyclist, which faced the same level of exposure. Train was the least exposed; lower than car by a factor of 4.3 and 1.9 times lower than bus/cyclist. The cyclist recorded the highest concentration, at 112.91 ppm (Table 15). There was a statistically significant difference across modes ($F_{3,69.06}=134.3$, $p<0.001$). Both wind speed ($p=0.029$) and temperature ($p=0.001$) were negatively correlated with CO (Table 16). Time of day was non-significant, as was wind speed grouped into fixed categories.

Table 15 Summary statistics for CO modal exposure in Auckland

Mode of transport	***N* Journeys (samples)**	**Mean** (+/-0.2 ppm)	**Std. Dev.**	**Min**	**Max**	**Median**	**-95% Cnf.Lmt**	**+95% Cnf.Lmt**
All samples	*95 (36707)*	*3.13*	*2.81*	*0.05*	*112.91*	*2.41*	*3.11*	*3.17*
Bus	23 (9942)	2.51	1.45	0.05	13.78	2.27	2.48	2.54
Car	24 (9530)	5.74	2.85	0.35	27.11	5.19	5.67	5.79
Train	24 (6579)	1.34	0.83	0.05	8.62	1.08	1.32	1.36
Cycle on-road	24 (10656)	2.51	2.99	0.05	112.91	1.96	2.46	2.57

Table 16 ANOVA results for CO modal exposure in Auckland

Variable	**Effect (F/R)**	**df Effect**	**MS Effect**	**df Error**	**MS Error**	**F**	**p**
Avg Wind Speed	***Fixed**	**1**	**11.48**	**18.85**	**2.04**	**5.6**	**0.029**
Avg Temp	***Fixed**	**1**	**30.35**	**19.02**	**2.02**	**15.0**	**0.001**
{1}Time of Day	*Fixed	1	1.94	18.85	2.04	1.0	0.342
{2}Mode	***Fixed**	**3**	**88.75**	**69.06**	**0.66**	**134.3**	**<0.001**
{3}Wind Strength	Fixed	1	2.98	18.85	2.04	1.5	0.241
{4}Run	**Random**	**19**	**2.02**	**68.00**	**0.66**	**3.1**	**<0.001**

4.3.2 PM_{10}, $PM_{2.5}$ and $PM_{1.0}$

Mean results for $PM_{2.5}$ ($F_{2,41.81}=6.85$, $p=0.003$) and $PM_{1.0}$ ($F_{2,40.66}=9.13$, $p=0.001$) were significant across modes but PM_{10} was non-significant ($F_{2,43.85}=0.39$, $p<0.982$). This was possibly due to an uncorrectable instrument error causing under-reading for the PM_{10} channel in the bus Grimm. Time of day significantly affected $PM_{2.5}$ ($p=0.005$) with overall morning concentrations (26.3 µg/m³) being substantially higher than those of the afternoons (17.6 µg/m³). Average temperature significantly affected $PM_{1.0}$ ($p=0.025$), for which there was a negative correlation. Refer to Table 18 for full ANOVA results. The on-road cyclist was the least exposed for both $PM_{2.5}$ and $PM_{1.0}$, at 33% and 44% lower than bus, and 11% and 28% lower than car (Table 17).

Table 17 Summary statistics for PM modal exposure in Auckland

Mode of transport		*N* Journeys (samples)	Arithmetic Mean (µg/m³)	Std. Dev.	Min	Max	Median	-95% Cnf.Lmt	+95% Cnf.Lmt
All samples	*PM_{10}*	*65 (26200)*	*24.61*	*19.95*	*5.40*	*803.85*	*21.37*	*24.37*	*24.85*
	$PM_{2.5}$		*19.66*	*18.64*	*3.23*	*696.70*	*15.60*	*19.43*	*19.88*
	$PM_{1.0}$		*14.40*	*12.29*	*1.86*	*141.80*	*10.54*	*14.25*	*14.55*
Bus	PM_{10}	21 (7988)	24.56	10.22	8.34	89.72	23.91	24.33	24.78
	$PM_{2.5}$		24.63	14.08	3.23	149.89	22.41	24.31	24.93
	$PM_{1.0}$		18.87	11.74	2.72	122.99	15.55	18.62	19.13
Car	PM_{10}	22 (8052)	23.55	16.07	5.4	151.23	19	23.20	23.90
	$PM_{2.5}$		18.66	15.11	4.73	144.40	14	18.33	19.00
	$PM_{1.0}$		14.72	15.03	2.90	141.83	9.8	14.40	15.06
Cycle On-Road	PM_{10}	22 (10161)	25.49	27.17	7.67	803.86	21.84	24.96	26.02
	$PM_{2.5}$		16.54	23.01	4.68	696.65	13.64	16.09	16.99
	$PM_{1.0}$		10.63	8.47	1.86	98.62	8.24	10.47	10.80

Table 18 ANOVA results for PM modal exposure in Auckland

PM$_{10}$							
Variable	**Effect (F/R)**	**df Effect**	**MS Effect**	**df Error**	**MS Error**	**F**	**p**
Avg Wind Speed	*Fixed	1	76.94	16.93	104.12	0.74	0.402
Avg Temp	*Fixed	1	12.76	17.21	103.02	0.12	0.729
{1}Time of Day	***Fixed**	**1**	**899.51**	**17.56**	**101.71**	**8.84**	**0.008**
{2}Mode	*Fixed	2	17.89	43.85	46.31	0.39	0.682
{3}Wind Strength	Fixed	1	143.35	17.14	103.28	1.39	0.255
{4}Run	**Random**	**19**	**96.91**	**38.00**	**44.42**	**2.18**	**0.020**
PM$_{2.5}$							
Avg Wind Speed	*Fixed	1	187.53	15.78	121.06	1.55	0.231
Avg Temp	*Fixed	1	23.94	16.21	120.38	0.20	0.662
{1}Time of Day	***Fixed**	**1**	**1248.33**	**16.75**	**119.56**	**10.44**	**0.005**
{2}Mode	***Fixed**	**2**	**583.63**	**41.81**	**85.20**	**6.85**	**0.003**
{3}Wind Strength	Fixed	1	192.34	16.10	120.54	1.60	0.224
{4}Run	Random	19	116.59	38.00	84.03	1.39	0.191
PM$_{1.0}$							
Avg Wind Speed	*Fixed	1	86.57	14.41	47.00	1.84	0.196
Avg Temp	***Fixed**	**1**	**293.42**	**15.01**	**47.04**	**6.24**	**0.025**
{1}Time of Day	*Fixed	1	143.62	15.77	47.09	3.05	0.100
{2}Mode	***Fixed**	**2**	**449.03**	**40.66**	**49.18**	**9.13**	**0.001**
{3}Wind Strength	Fixed	1	3.51	14.87	47.03	0.07	0.788
{4}Run	Random	19	47.27	38.00	49.25	0.96	0.523

4.4 Combined Results

4.4.1 Carbon monoxide

For both cities combined, the car mode was higher than the on-road cyclist by a factor of 2.3, and by 2.5 and 3.3 for bus and train (Table 19). However, a lack of an off-road cyclist and train mode for Christchurch renders these modes less inter-comparable. Overall mean CO levels for Auckland (3.13 ppm) were greater than Christchurch by a factor of 2.3 (1.35 ppm). The maximum level sampled was 112.91 ppm, recorded by the on-road cyclist mode in Auckland. Across four modes in each city, results were significant ($F_{4,256.63}$ =56.52, p<0.001). Table 20 provides full ANOVA results.

Table 19 Summary statistics for combined CO modal exposure

Mode of transport	*N* Journeys (samples)	Mean (+/-0.2 ppm)	Std. Dev.	Min	Max	Median	-95% Cnf.Lmt	+95% Cnf.Lmt
All samples	*279 (77960)*	*2.19*	*2.58*	*0.05*	*112.91*	*1.45*	*2.17*	*2.21*
Bus	72 (21114)	1.76	1.46	0.05	13.78	1.55	1.74	1.78
Car	72 (18340)	4.36	3.35	0.05	52.32	4.04	4.31	4.41
Train	24 (6579)	1.34	0.83	0.05	8.62	1.08	1.32	1.36
Cycle off-road	48 (12110)	0.65	0.88	0.05	22.75	0.37	0.64	0.67
Cycle on-road	63 (19817)	1.87	2.54	0.05	112.91	1.33	1.83	1.90

Table 20 ANOVA results for combined CO modal exposure

Variable	Effect (F/R)	df Effect	MS Effect	df Error	MS Error	F	p
Avg Windspeed	*Fixed	1	9.73	74.81	6.20	1.56	0.214
Avg Temp	*Fixed	1	20.04	75.31	6.10	3.28	0.074
{2}Time of Day	*Fixed	1	0.08	74.60	6.24	0.01	0.906
{3}Mode	***Fixed**	**4**	**86.38**	**256.63**	**1.52**	**56.52**	**<0.001**
{4}Wind Strength	Fixed	1	0.24	95.39	3.93	0.06	0.804
{5}Run	**Random**	**76**	**5.97**	**194.00**	**1.04**	**5.72**	**<0.001**

4.4.2 PM_{10}, $PM_{2.5}$ and $PM_{1.0}$

Results for combined PM exposure show $PM_{2.5}$ ($F_{3,196.32}$ =5.10, p=0.002) and $PM_{1.0}$ ($F_{3,198.86}$ =3.01, p=0.031) differed significantly across modes (Table 22). $PM_{1.0}$ was significantly affected by wind strength (p=0.039) for which there was a negative correlation. Median values for run means in wind conditions >2 m/s were 8.38 and 13.36 when average wind speed was <2 m/s. Overall $PM_{1.0}$ exposure for Auckland was 28% higher than Christchurch, while $PM_{2.5}$ was virtually the same with a difference of only 1.8%. The Christchurch bus commuter recorded the highest PM_{10} value of 1138.8 µg/m³, followed by the Auckland cyclist (803.8 µg/m³) which also had the highest $PM_{2.5}$ peak of 695.65 µg/m³ (Table 21). The Christchurch off-road cyclist was exposed to the highest $PM_{1.0}$ recording of 312.3 µg/m³.

Table 21 Summary statistics for combined PM modal exposure

Mode of transport		***N* Journeys (samples)**	**Arithmetic Mean** (µg/m³)	**Std. Dev.**	**Min**	**Max**	**Median**	**-95% Cnf.Lmt**	**+95% Cnf.Lmt**
All samples	*PM_{10}*	*208 (60407)*	*32.26*	*27.22*	*4.00*	*1138.80*	*27.40*	*32.04*	*32.48*
	$PM_{2.5}$		*19.46*	*17.13*	*1.30*	*696.65*	*16.20*	*19.32*	*19.59*
	$PM_{1.0}$		*12.16*	*11.62*	*0.30*	*312.30*	*8.80*	*12.07*	*12.25*
Bus	PM_{10}	64 (15416)	38.85	34.05	4.80	1138.80	32.00	38.31	39.39
	$PM_{2.5}$		24.75	18.48	2.20	268.00	20.15	24.46	25.05
	$PM_{1.0}$		16.02	13.50	0.80	122.99	12.00	15.81	16.23
Car	PM_{10}	41 (12002)	24.69	15.20	4.00	151.20	20.30	24.42	24.96
	$PM_{2.5}$		17.14	13.19	1.60	144.40	13.30	16.91	17.38
	$PM_{1.0}$		12.65	13.05	0.80	141.80	8.74	12.41	12.88
Cycle Off-Road	PM_{10}	47 (12351)	37.45	31.41	4.30	573.50	31.20	36.90	38.01
	$PM_{2.5}$		17.94	16.63	1.30	505.40	15.40	17.65	18.24
	$PM_{1.0}$		7.83	10.04	0.30	312.30	5.70	7.66	8.01
Cycle On-Road	PM_{10}	56 (20638)	28.64	21.96	7.67	803.86	26.48	28.34	28.94
	$PM_{2.5}$		17.90	17.64	3.23	696.65	15.29	17.66	18.14
	$PM_{1.0}$		11.58	8.77	1.86	98.62	8.55	11.46	11.70

Table 22 ANOVA results for combined PM modal exposure

			PM_{10}				
Variable	**Effect (F/R)**	**df Effect**	**MS Effect**	**df Error**	**MS Error**	**F**	**p**
Avg Wind Speed	*Fixed	1	1175.49	71.96	577.85	2.03	0.158
Avg Temp	*Fixed	1	78.38	71.64	581.11	0.13	0.715
{1}Time of Day	***Fixed**	**1**	**3587.95**	**69.20**	**607.92**	**5.90**	**0.018**
{2}Mode	*Fixed	3	571.15	198.60	216.95	2.63	0.051
{3}Wind Strength	Fixed	1	1155.23	103.17	400.72	2.88	0.093
{4}Run	**Random**	**74**	**558.22**	**125.00**	**163.17**	**3.42**	**<0.001**
			$PM_{2.5}$				
Avg Wind Speed	*Fixed	1	433.62	71.36	252.52	1.72	0.194
Avg Temp	*Fixed	1	132.71	71.25	253.00	0.52	0.471
{1}Time of Day	***Fixed**	**1**	**1458.89**	**68.84**	**263.74**	**5.53**	**0.022**
{2}Mode	***Fixed**	**3**	**501.75**	**196.32**	**98.42**	**5.10**	**0.002**
{3}Wind Strength	Fixed	1	436.43	103.82	176.75	2.47	0.119
{4}Run	**Random**	**74**	**242.23**	**123.00**	**74.82**	**3.24**	**<0.001**
			$PM_{1.0}$				
Avg Wind Speed	*Fixed	1	292.76	72.04	126.11	2.32	0.132
Avg Temp	***Fixed**	**1**	**860.78**	**71.73**	**126.84**	**6.79**	**0.011**
{1}Time of Day	*Fixed	1	481.94	69.38	132.77	3.63	0.061
{2}Mode	***Fixed**	**3**	**138.90**	**198.86**	**46.15**	**3.01**	**0.031**
{3}Wind Strength	**Fixed**	**1**	**381.73**	**102.06**	**86.87**	**4.39**	**0.039**
{4}Run	**Random**	**74**	**121.76**	**125.00**	**34.23**	**3.56**	**<0.001**

4.5 Comparisons to previous studies

This section presents simplified versions of the comparative tables presented in the literature review chapter. Results from the current study are presented with results of the most comparative previous studies, primarily based on similar ventilation and sampling settings where possible.

4.5.1 Carbon monoxide

Table 23 Inter-modal CO results compared to previous studies

Author	Location	Mode	Sampling setting	Mean (ppm)	Ratio (car:alternative mode)
Bevan et al. (1991)	Southampton, UK	Bicycle	Suburban commute Urban commercial Parkland	10.5 4.5 0.8	
Chan et al. (1991)	Raleigh, NC, USA	Car	Interstate beltway Rural	11 4	
Clifford et al. (1997)	Nottingham, UK	Van (morning) Van (afternoon)	Key commuting routes	4.8 4.9	
Rodes et al. (1998)	Sacramento, CA, USA Los Angeles, CA, USA	Car Car	Freeway commute Freeway commute	2.0, 3.5 2.2, 2.7 4.3, 4.5 4.9, 5.3	
Alm et al. (1999)	Kuopia, Finland	Car (morning) Car (afternoon)	Small urban town	5.7 3.1	
Kaur et al. (2005a)	London, UK	Car Taxi Bus Bicycle Walk	Urban centre	1.3 1.1 0.8 1.1 0.9	- 1.18 1.63 1.18 1.44
Wöhrnschimmel et al. (2008)	Mexico City, Mexico	Car Bus Minibus Metrobus (BRT)	Urban	16.3 11.5 7.8 20.3	- 1.42 2.09 0.80
Kaur & Nieuwenhuijsen (2009)	London, UK	Car Taxi Bus Bicycle Walk	Urban centre	1.3 1.2 0.8 0.9 0.7	- 1.08 1.63 1.45 1.86
Current study	Christchurch, NZ Auckland, NZ	Car Bus Bicycle (on-road) Bicycle (off-road) Car Bus Train Bicycle	Urban commute	2.87 1.10 1.12 0.65 5.74 2.51 1.34 2.51	- 2.61 2.57 4.42 - 2.29 4.28 2.29

For the car mode, absolute results for Christchurch were most comparable to those found by Rodes et al. (1998) in Sacramento along a freeway (Table 23). Similarly, their results for Los Angeles are very close to those found in Auckland. The mean for Auckland (5.7 ppm) was also exactly the same as the mean observed for a small urban Finnish town during morning sampling (Alm et al. 1999). Both of these studies sampled with windows closed, vents set to 2.

Much of the research including a bus mode has been conducted in very large urban centres such as Mexico and Taipei, making results less comparable. However, results from London are very similar to those for Christchurch for both the bus and on-road cyclist (Kaur et al. 2005). The off-road cyclist result agreed with that found for parkland in Southampton, UK (Bevan et al. 1991). Cyclist results are discussed in greater detail in Chapter 5.

No comparable study exists for the train mode as most large urban centres around the world have electrified rail systems, where exposure is generally lower than all other modes.

When comparing ratios, the ratios for car:bus and car:bicycle for the current study were almost double those found in other cities. The closest found was car:minibus in Mexico city (Wöhrnschimmel et al. (2008)

4.5.2 PM_{10}, $PM_{2.5}$ and $PM_{1.0}$

Comparative results for PM_{10} are presented in Table 24. Auckland results were omitted as they were non-significant and most likely erroneous.

Table 24 Inter-modal PM_{10} results compared to previous studies

Author	Location	Mode	Sampling setting	Mean (µg/m³)	Ratio (car:alternative mode)
Lewné et al. (2006)	Stockholm, Sweden	Taxi Bus Truck	Urban/rural	26 44 57	- 0.59 0.46
Gulliver & Briggs (2007)	Leicester, UK	Car Walk	Urban arterials and residential zones	18.2 19.1	- 0.95
Briggs et al. (2008)	Northampton, UK	Car Walk	Urban	5.87 27.56	- 0.21
Thai et al. (2008)	Vancouver, BC, Canada	Bicycle	Urban commercial, residential, industrial and parkland	21.6-74.8	
Berghmans et al. (2009)	Mol, Flanders, Belgium	Bicycle	Small urban centre/residential Cycle track	62.4 54.3	
Nasir & Colbeck (2009)	Colchester, UK	Car (morning) Car (evening)	Suburban/rural Suburban/rural	22 21	
Current study	Christchurch, NZ	Car Bus Bicycle (on-road) Bicycle (off-road)	Urban commute	27 45.8 35.1 37.5	- 0.59 0.77 0.72

Results from the current study are in agreement with previous research in that the car mode is ranked the least exposed. This is due to the origins of PM_{10} being mostly background sources and heavy diesel vehicles, which are particularly prone to self-pollution. Active modes are more influenced by background concentrations, whereas cars are primarily affected by CO and other emissions generated in front of them.

Mean results for Christchurch were very similar to those for Stockholm, giving the same ratio of 0.59 for car:bus (Lewné et al. 2006). For active-mode transport, mean cyclist results fell within the range reported by Thai et al. (2008) across a variety of urban settings in Vancouver, while those for small Belgian town ~11 times smaller than Christchurch (population), were far higher (Berghmans et al. 2009). The on-road cyclist result was greater than car by a factor of 1.3, whereas Briggs et al. (2008) found pedestrian exposure to be greater by a factor of 4.7. The reason given for such a difference is that the fanned ventilation system filters out particles, creating an independent microenvironment inside the vehicle cabin. Additionally, greater travel time for the pedestrian further increased exposure.

For Christchurch car $PM_{2.5}$ exposure (14 μg/m^3), results were again comparable to those of Rodes et al. (1998) in Sacramento (11.3, 11 μg/m^3) and exactly the same as found in Utrecht, Netherlands (den Breejen 2006). Auckland results were slightly higher and comparable to those for New York (Morabia et al. 2009). As for ratios, car:cyclist were very close to those for car:walk in Leceister, Northampton and New York (Table 25). The car:cyclist ratios were comparable to those found in London and across numerous cities in the Netherlands. Overall, it could be stated that cyclists are slightly less exposed. There were no comparable findings for the bus mode.

Table 25 Inter-modal $PM_{2.5}$ results compared to previous studies

Author	Location	Mode	Sampling setting	Mean (µg/m³)	Ratio (car:alternative mode)
Rodes et al (1998)	Sacramento, CA	Car	Freeway commute	11.3, 11.0	
	Los Angeles, CA, USA	Car	Freeway commute	47.2, 37.4	
den Breejen (2006)	Utrecht, Netherlands	Car	Urban	14	
Gulliver & Briggs (2007)	Leicester, UK	Car	Urban arterials and residential zones	8.3	-
		Walk		10.9	0.76
Briggs et al. (2008)	Northampton, UK	Car	Urban	3.01	-
		Walk		6.59	0.46
Thai et al. (2008)	Vancouver, BC, Canada	Bicycle	Urban commercial, residential, industrial and parkland	7.3-33.6	
Morabia et al. (2009)	New York, NY, USA	Car	Urban centre	18	-
		Walk		23.9	0.75
Berghmans et al. (2009)	Mol, Flanders, Belgium	Bicycle	Small urban centre/residential	38.8	
			Cycle track	31.7	
Nasir & Colbeck (2009)	Colchester, UK	Car (morning)	Suburban/rural	9	
		Car (evening)	Suburban/rural	8	
Kaur & Nieuwenhuijsen (2009)	London, UK	Car	Urban centre	33.4	-
		Taxi		43.4	0.77
		Bus		33.1	1.01
		Bicycle		33.8	0.99
		Walk		27.1	1.23
Boogaard et al. (2009)	Apeldoorn, Netherlands	Car	Unspecified	14	-
		Bicycle		11	1.27
	Delft	Car		33	-
		Bicycle		26	1.27
	Den Bosch	Car		95	-
		Bicycle		99	0.96
	The Hague	Car		15	-
		Bicycle		6	2.5
	Eindhoven	Car		34	-
		Bicycle		39	0.87
	Groningen	Car		20	-
		Bicycle		13	1.54
	Haarlem	Car		36	-
		Bicycle		29	1.24
	Maastricht	Car		31	-
		Bicycle		20	1.55
	Nijmegen	Car		93	-
		Bicycle		95	0.98
	Utrecht	Car		122	-
		Bicycle		112	1.09
	Zwolle	Car		45	-
		Bicycle		44	1.02
	Combined total	Car		49.4	-
		Bicycle		44.5	1.11
Current study	Christchurch, NZ	Car	Urban commute	14	-
		Bus		23.6	0.59
		Bicycle (on-road)		19	0.74
		Bicycle (off-road)		17.9	0.78
	Auckland, NZ	Car		18.7	-
		Bus		24.6	0.76
		Bicycle		16.5	1.13

Mean results for car $PM_{1.0}$ in Christchurch were closest to those in Colchester, UK, while mean results for Auckland were almost as high as in Taipei, Taiwan (Table 26). Ratios for car:cycle (on-road) in Christchurch were comparable to car:walk in Leicester and New York, yet car exposure was 28% higher than cyclist in Auckland. A relative lack of $PM_{1.0}$ literature on inter-modal journey exposure makes comparison with other cities difficult.

Table 26 Inter-modal $PM_{1.0}$ results compared to previous studies

Author	Location	Mode	Sampling setting	Mean (µg/m³)	Ratio (car:alternative mode)
Gulliver & Briggs (2007)	Leicester, UK	Car Walk	Urban arterials and residential zones	2.9 4.8	- 0.60
Briggs et al. (2008)	Northampton, UK	Car Walk	Urban	1.82 3.37	- 0.54
Tsai et al. (2008)	Taipei, Taiwan	Car Bus	Urban	16.2 31.3	- 0.52
Morabia et al. (2009)	New York, NY, USA	Car Walk	Urban centre	18 23.9	- 0.75
Berghmans et al. (2009)	Mol, Flanders, Belgium	Bicycle	Small urban centre/residential Cycle track	37.4 29.8	
Nasir & Colbeck (2009)	Colchester, UK	Car (morning) Car (evening)	Suburban/rural Suburban/rural	6 5	
Current study	Christchurch, NZ Auckland, NZ	Car Bus Bicycle (on-road) Bicycle (off-road) Car Bus Bicycle	Urban commute	8.4 13.7 11.4 7.8 14.7 18.9 10..6	- 0.61 0.74 1.08 - 0.77 1.39

4.5.3 Ultrafine particles

For the car mode, mean UFP results were greatly in excess of those found in Boston, Northampton and Montréal, yet lower than in London (Table 27). Findings are generally far higher for car than for bus and active modes (Boogaard et al. 2009; Kaur & Nieuwenhuijsen 2009; Weichenthal et al. 2008). Ratios for car:bus and car:bicycle (on-road) are in agreement with Kaur & Nieuwenhuijsens' (2009) findings for London.

Table 27 Inter-modal UFP results compared to previous studies

Author	Location	Mode	Sampling setting	Mean (μg/m³)	Ratio (car:alternative mode)
Hill & Gooch (2007)	Boston, MA, USA	Car	Urban	28981	-
		Bus		83227	0.35
		Walk		30273	0.96
Briggs et al. (2008)	Northampton, UK	Car	Urban	21639	-
		Walk		30334	0.71
Thai et al. (2008)	Vancouver, BC, Canada	Bicycle	Urban commercial, residential, industrial and parkland	18830-57692	
Weichenthal et al. (2008)	Montréal, Canada	Car (morning)	Urban highway and busy roadway	38348	-
		Car (evening)		31489	-
		Bus (morning)		28029	1.37
		Bus (evening)		22626	1.39
		Walk (morning)	Two-lane roadway	25161	1.52
		Walk (evening)		15778	1.99
Berghmans et al. (2009)	Mol, Flanders, Belgium	Bicycle	Small urban centre/residential	21226	
			Cycle track	21626	
Kaur & Nieuwenhuijsen (2009)	London, UK	Car	Urban centre	101770	-
		Taxi		91947	1.11
		Bus		100018	1.02
		Bicycle		77621	1.31
		Walk		63065	1.61
Boogaard et al. (2009)	Apeldoorn, Netherlands	Car	Unspecified	20796	-
		Bicycle		17070	1.22
	Delft	Car		24460	-
		Bicycle		27998	0.87
	Den Bosch	Car		23012	-
		Bicycle		21191	1.09
	The Hague	Car		15430	-
		Bicycle		15697	0.98
	Eindhoven	Car		23461	-
		Bicycle		28141	0.83
	Groningen	Car		22234	-
		Bicycle		21326	1.04
	Haarlem	Car		34739	-
		Bicycle		30363	1.14
	Maastricht	Car		35538	-
		Bicycle		28220	1.25
	Nijmegen	Car		24064	-
		Bicycle		20244	1.19
	Utrecht	Car		29722	-
		Bicycle		27246	1.09
	Zwolle	Car		23583	-
		Bicycle		31354	0.75
	Combined total	Car		25545	-
		Bicycle		24329	1.05
Current study	Christchurch, NZ	Car	Urban commute	77654	-
		Bus		76481	1.02
		Bicycle (on-road)		49842	1.56
		Bicycle (off-road)		23541	3.30

4.6 Summary

The results of the current study place the car mode as the most exposed for CO and UFPs but the least exposed for PM, with the exception of $PM_{2.5}$ and $PM_{1.0}$ in Auckland, where the on-road cyclist was least exposed. CO levels were comparatively quite high, with car:bus and car:cyclist ratios in Christchurch and Auckland around double those observed in London (Kaur et al. 2005a; Kaur & Nieuwenhuijsen 2009). Particulate findings were generally consistent with previous research. Briggs et al. (2008) suggest that while car filtration systems effectively prevent the ingress of course particles, finer particles are able to penetrate and accumulate. Secondly, the number of intense exposure peaks (heavy vehicles, construction activity, passing smokers) has been known to significantly influence overall exposure for pedestrians and cyclists (Briggs et al. 2008; Kaur et al. 2006). Vehicles, acting as independent 'indoor' microenvironments, are far less affected by these events; due to the less penetrable environment and/or the ability to move more quickly through high exposure situations. This may explain the heightened car:cyclist PM2.5 and PM1.0 exposure in Auckland. The car was constantly queued behind buses and other traffic while the cyclist was able to quickly move to the front of queues and always finish the study route first. In Christchurch, congestion was much less of an issue and the on-road cyclist and car generally completed routes simultaneously. UFP results also support previous findings, with car being higher than bus and active modes (Boogaard et al. 2009; Kaur & Nieuwenhuijsen 2009; Weichenthal et al. 2008). However, some research comparing car to pedestrians, found opposing results (Hill & Gooch 2007, Briggs et al. 2008). The differences in findings between Kaur & Nieuwenhuijsen (2009) and Briggs et al. (2008) are difficult to explain given the same vehicle configuration was used and sampling methodology was similar, taking place in and around major urban centres.

Chapter Five: Cyclist results

5.1 Introduction

This chapter presents and discusses all results recorded by the cyclists in Christchurch and Auckland. The first section explores the implications of taking either the more direct route on-road or a longer, off-road route consisting of cycle ways and parkland. Results are discussed in terms of spatial variability/uniformity observed and the effect of meteorological variables. Inferences about spatial variation were made by adding comparative data to transparent area charts for at least eight runs. Two charts for each pollutant are included to illustrate observed trends. Note that due to instrument failure and data loss, a mixture of examples are given from different days and are not always inter-comparable. The second part looks at any changes in pollutant concentrations at the microscale level. The influence of traffic proximity is considered using mean exposure data and frequency distributions where significant differences are found. Further consideration is given to wind influence, where relative position in relation to wind direction is addressed. The final sections consider the results in light of any comparable literature.

5.2 *Christchurch: Effect of route choice*

Table 28 compares cyclist summary results. Overall, the off-road cyclist was exposed to lower concentrations of all pollutants apart from PM_{10}. The largest decreases found were for UFPs, CO and $PM_{1.0}$, where there was a 53, 42 and 31% reduction, respectively. $PM_{2.5}$ dropped by 6% while PM_{10} increased by 7%.

Table 28 Summary cyclist results

Mode	Pollutant	*N* Journeys (samples)	Mean	Std. Dev.	Min	Max	Median	-95% Cnf.Lmt	+95% Cnf.Lmt
Cycle Off-Road	CO	48 (12110)	0.65	0.88	0.05	22.75	0.37	0.64	0.67
	PM_{10}	47 (12351)	37.45	31.41	4.30	573.50	31.20	36.90	38.01
	$PM_{2.5}$		17.94	16.63	1.30	505.40	15.40	17.65	18.24
	$PM_{1.0}$		7.83	10.04	0.30	312.30	5.70	7.66	8.01
	UFP	33 (8753)	23541.97	37176.55	23.33	741751.67	11115	22763.04	24320.90
Cycle On-Road	CO	39 (9161)	1.12	1.58	0.05	25.90	0.49	1.08	1.15
	PM_{10}	34 (7984)	35.12	14.47	9.01	117.52	31.92	34.81	35.44
	$PM_{2.5}$		18.99	8.61	4.98	77.55	16.43	18.80	19.17
	$PM_{1.0}$		11.38	8.14	1.86	73.13	8.45	11.20	11.56
	UFP	38 (8545)	49842.85	71568.47	85	1304048	25516.67	48325.19	51360.51

The off-road cyclist recorded the lowest minimum values for all pollutants, but also had the highest particulate values in excess of 18 times the standard deviation. This can be put down to a single event on the morning of March 10 during which the cyclist passed an old idling diesel school bus while traveling down a suburban backstreet. The peak event lasted for approximately one minute, indicating that the vehicle had been idling for some time, emitting a plume of pollutants that the rider slowly passed through. Wind conditions were weak at only 1.3 m/s and no other vehicles were present. Although the on-road cyclist would have also passed idling diesel vehicles at traffic lights, it is likely that this bus was running particularly poorly, with concentrations elevated by greater idling time. A higher mean PM_{10} value can be explained by a greater susceptibility to background influences from home heating and industrial sources, whereas the on-road cyclist is primarily affected by traffic emissions. The same can be said for $PM_{2.5}$ which mainly originates from home heating. Conversely, the majority of $PM_{1.0}$ emerges from vehicular fuel combustion and gas-to-particle conversion, accounting for the substantial difference in exposure (Phalen 2002). The off-road cyclist rode past several factories in a small industrial area near the start of the route and with the exception of the parkland section, the remainder was residential zoning. Results from the 2006 Christchurch inventory of emissions to air show that motor vehicles and industry are comparatively small contributors to PM_{10} (11%, 13%) and $PM_{2.5}$ (9%, 8%), with the balance made up of domestic heating (Smithson 2009). Although levels drop over summer, relatively cool temperatures and poorly insulated housing necessitate heating during most months of the year. Throughout the sampling period, average morning temperatures ranged from 8.1 - 18.6ºC and smoke from residential chimneys was occasionally noted by volunteers.

5.2.1 Carbon monoxide

Figure 13 displays mean CO concentrations for all comparable runs. Exposure varies greatly, highlighting the dependence on traffic fumes and the influence of meteorological conditions. For example, on the low exposure days where mean exposure for both modes was only a fraction of 1 ppm, average wind speed was 4.16 m/s and higher.

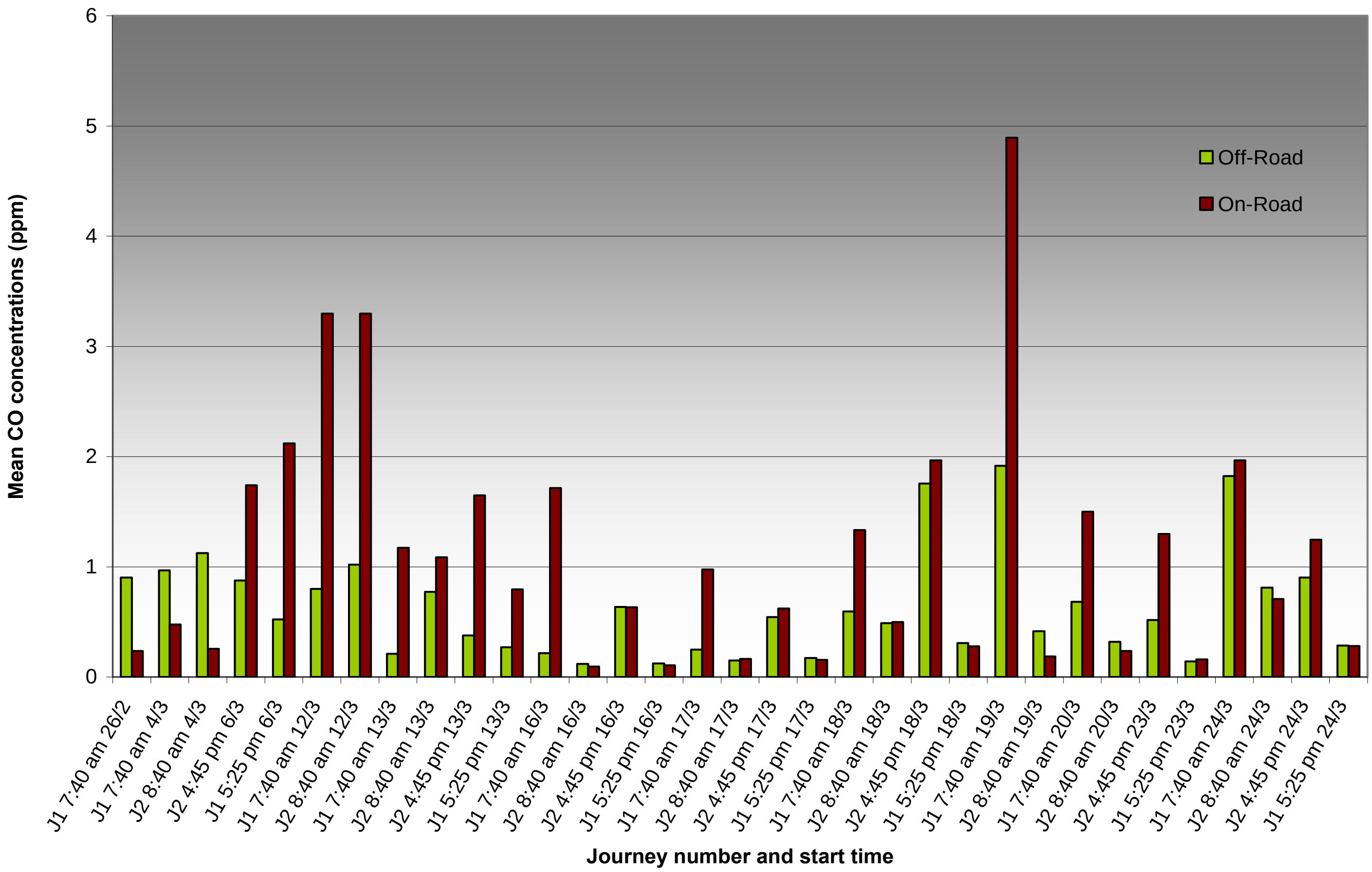

Figure 13 Cyclists' comparative mean CO exposure by journey

For the first three runs displayed in Figure 13, mean exposure is more than double for the off-road cyclist, indicating the occurrence of extreme peak episodes. These events also caused higher mean exposure for ultrafine particles during the same runs, shown in Figure 33. CO and UFP concentrations are generally found to be highly correlated e.g. $r = 0.7$ (Kaur et al. 2005). Locating peak events for the off-road cyclist in GRC Media Mapper showed that whenever the cyclist emerged from a heavily vegetated area in Hagley Park onto a roadway crossing, both CO and UFP levels skyrocketed. Average temperatures were lower during these two days (15.8, 14.3ºC) were lower than all other sampling days, making temperature a key predictor of concentration, along with traffic density. Average wind speed was also relatively high, at 3.2 and 3.8 m/s for 26 February and 4 March, respectively. It is likely that the low temperatures made for higher concentrations which remained more stagnant at the fringe of highly vegetated sheltered areas. While most vegetated areas act as pollutant sinks, they have also been shown to reduce airflow and cause stagnancy, significantly elevating concentrations; "the special structure of vegetation reduces near-surface air exchange, leading to an increase in atmospheric particle concentration and thus deterioration in the pollution situation near to emission sources" (Litschke & Kuttler 2008, p. 232). The abovementioned conditions, coupled with crossing wait times of around 3-4 minutes, led to higher mean exposure for the off-road cyclist.

Figure 14 and Figure 16 illustrate comparative CO concentrations by geographic location, during a morning commute from Redwood to Christchurch city centre and then to Canterbury University, on 12th March 2009. Overall, exposure for the off-road cyclist was extremely low, rising only when coming into contact with traffic at intersections and crossings. On-road was highest when cycling alongside queued traffic and waiting at traffic lights. Exposure peaked at queues situated within street canyons, most evident in the southwest corner of Figure 16, where the cyclist faces a very long queue and concentrations reach 25.9 ppm. These figures represent typical colour-coded concentration comparisons between the two cyclists, with the exception of the first three runs. Note that due to volunteer error, the on-road cyclist route differs slightly for this particular run but was not considered to make much of a difference to the mean, if any.

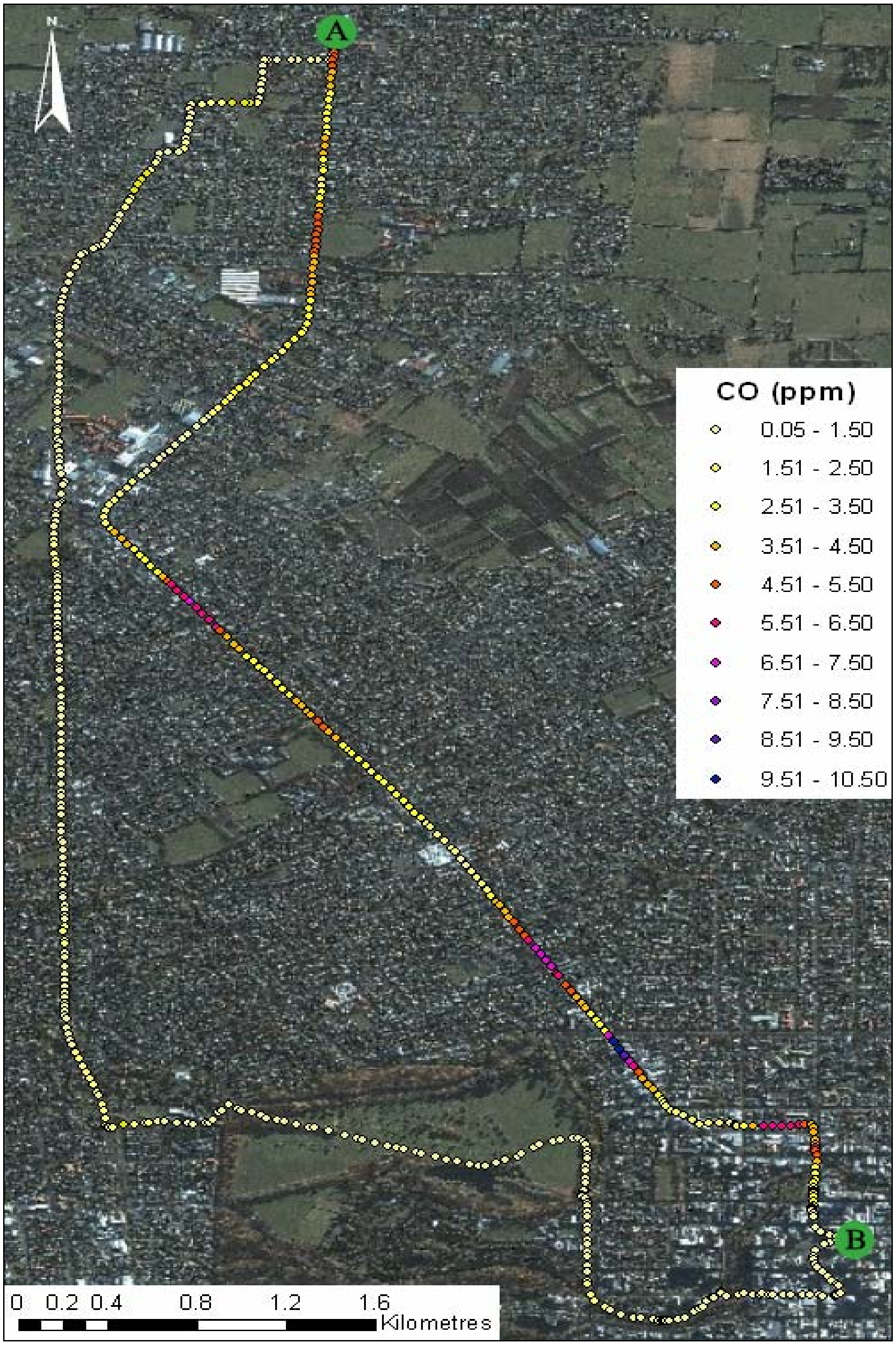

Figure 14 Cyclists' real-time comparative CO exposure by GPS co-ordinates: Redwood to Christchurch city centre, 7:40 – 8:20 am,12 March 2009

Figure 15 and Figure 17 illustrate the general pattern of exposure for the same two journeys. CO spatial distribution between the two cyclists was found to be heterogeneous, which is consistent with literature outlining the limited spatial extent of CO (Zhou & Levy 2007). Exposure was very low for the cyclist traveling away from traffic, often only just above zero, with mean concentrations only exceeding 1 ppm three times.

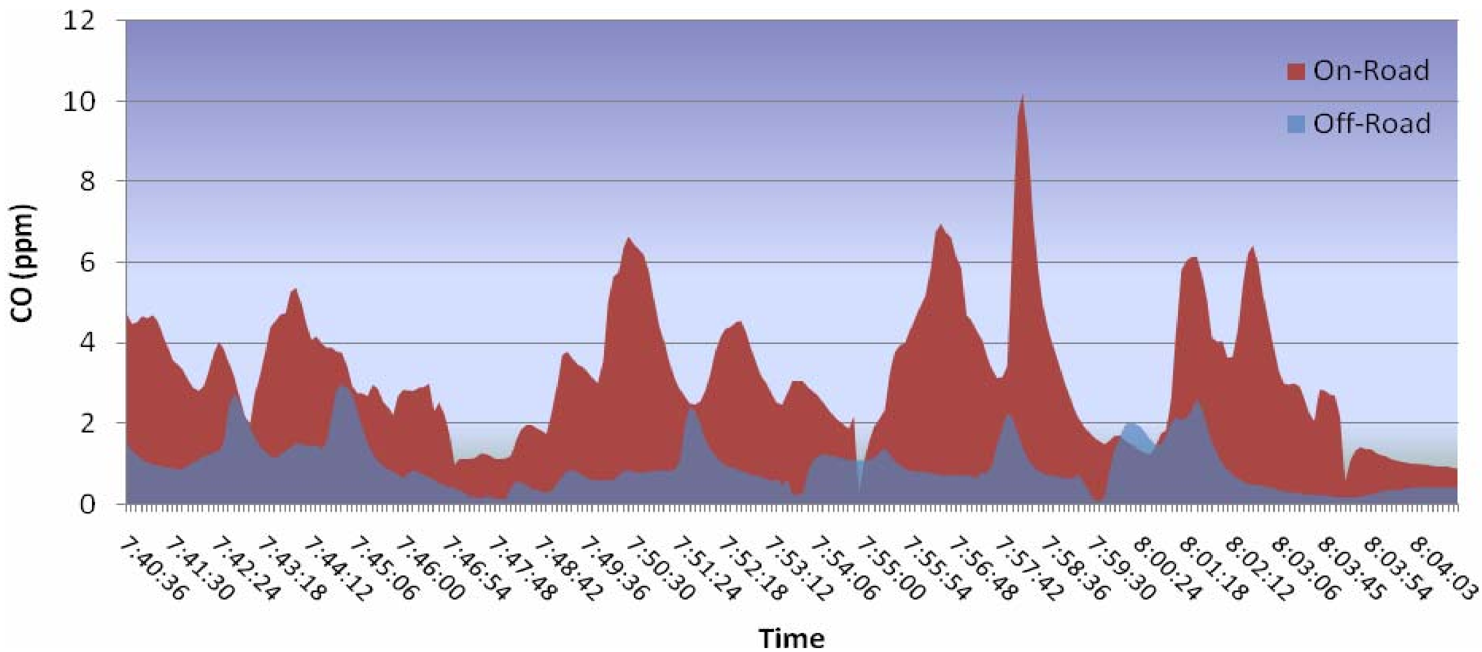

Figure 15 Cyclists' real-time comparative CO exposure showing a heterogeneous spatial distribution: Redwood to Christchurch city centre, 7:40 – 8:20 am, 12 March 2009

Figure 16 Cyclists' real-time comparative CO exposure by GPS co-ordinates: Christchurch city centre to Canterbury University, 8:30 – 9:00 am, 12 March 2009

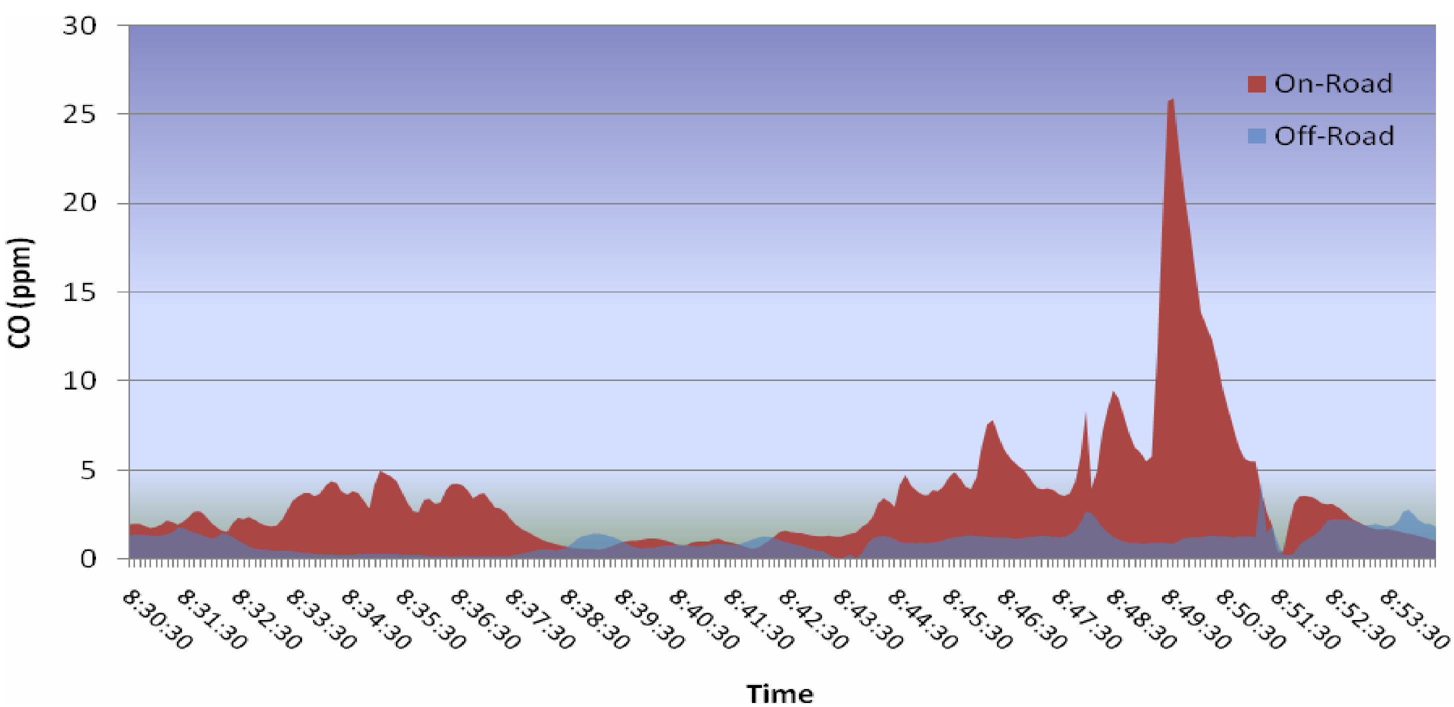

Figure 17 Cyclists' real-time comparative CO exposure showing a heterogeneous spatial distribution: Christchurch city centre to Canterbury University, 8:30 – 9:00 am, 12 March 2009

5.2.2 PM_{10}

Mean exposure for PM_{10} shows a clear pattern of spatial uniformity across the distance between the two cyclists, which, at certain points, was as wide as 3.3 km for Journey 1 and 0.6 km for Journey 2. While on-road exposure was occasionally higher, for the most part, exposure was relatively even or slightly higher for the off-road cyclist, indicating increased susceptibility to non-traffic sources (Figure 18). This resulted in an overall mean disparity of 6%, with the off-road cyclist being more exposed.

Conclusions on spatial homogeneity for coarse particles across large metropolitan centres are mixed, with many earlier studies finding high correlation between fixed sites (Burton et al. 1996; Martuzevicius et al. 2004). More recent research has shown that high correlation is not always indicative of homogeneity and that coefficients of divergence provide a more accurate measure. A study in Iowa City utilising 33 sites, found distribution for $PM_{10–2.5}$ to be heterogeneous, with maximum coefficients of divergence ranging from 0.21 – 0.36 (Ott et al. 2008). The average distance between sites was only 4.4 km and results closely matched those of mobile sampling. Concentration mapping corresponded well with known sources, including quarries, large constructions sites and industrial areas. The differences in findings for spatial variability between studies highlight the importance of considering local sources and utilising appropriate statistical methods. Traditionally, the greatest source affecting Christchurch concentrations has been home heating. While emissions from home heating are declining, recent research across 11 Christchurch sites found PM_{10} to be substantially spatially variable (Wilson et al. 2006).

While no analysis was done between the two mobile datasets and fixed site data, basic comparisons between the two cyclist routes found PM_{10} and $PM_{2.5}$ to be extremely spatially uniform (Figures 20, 22, 25 & 27). Although such evidence is by no means reflective of city-wide uniformity, it does pose interesting questions for small-scale variability. Large particles (greater than 5 µg m^3) from ambient sources are said to settle within a few kilometres (Phalen 2002), while traffic-generated PM settles within 400 m

(Zhou & Levy 2007). The slightly elevated exposure experienced by the off-road cyclist is likely to reflect a complex mixture of ambient and mobile-source emissions, whereas on-road cyclist exposure is more dependant on fluctuations within immediate traffic sources.

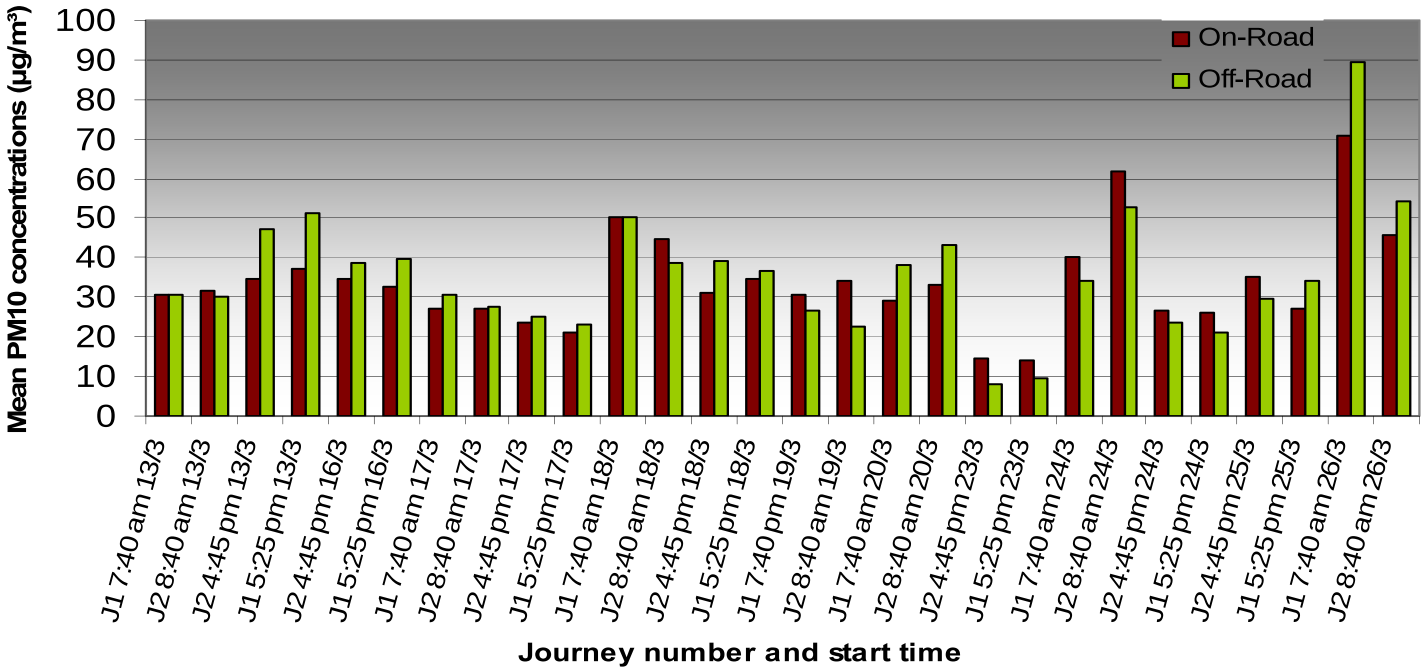

Figure 18 Cyclists' comparative mean PM_{10} exposure by journey

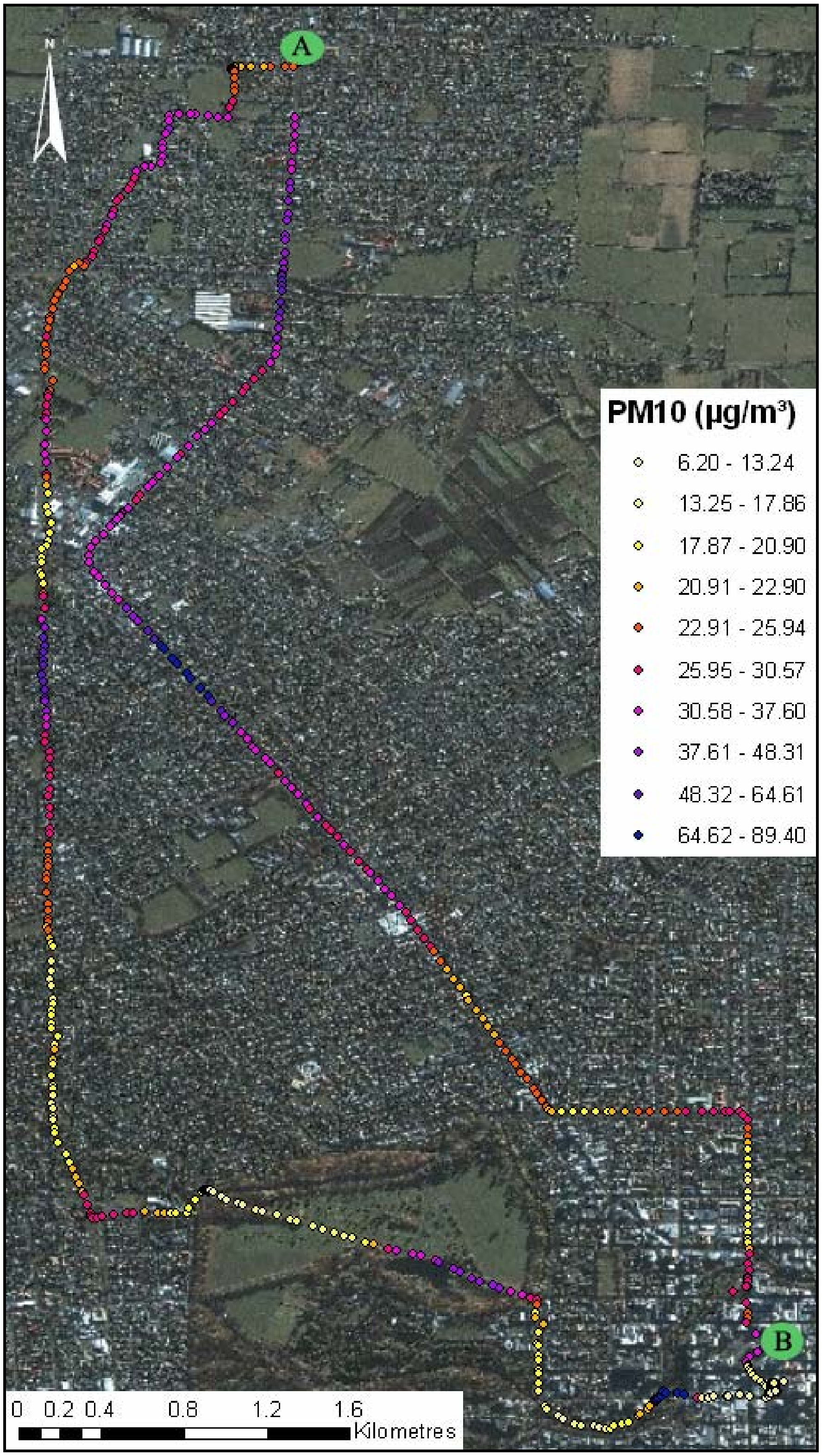

Figure 19 Cyclists' real-time comparative PM_{10} exposure by GPS co-ordinates: Redwood to Christchurch city centre, 7:40 – 8:20 am, 19 March 2009

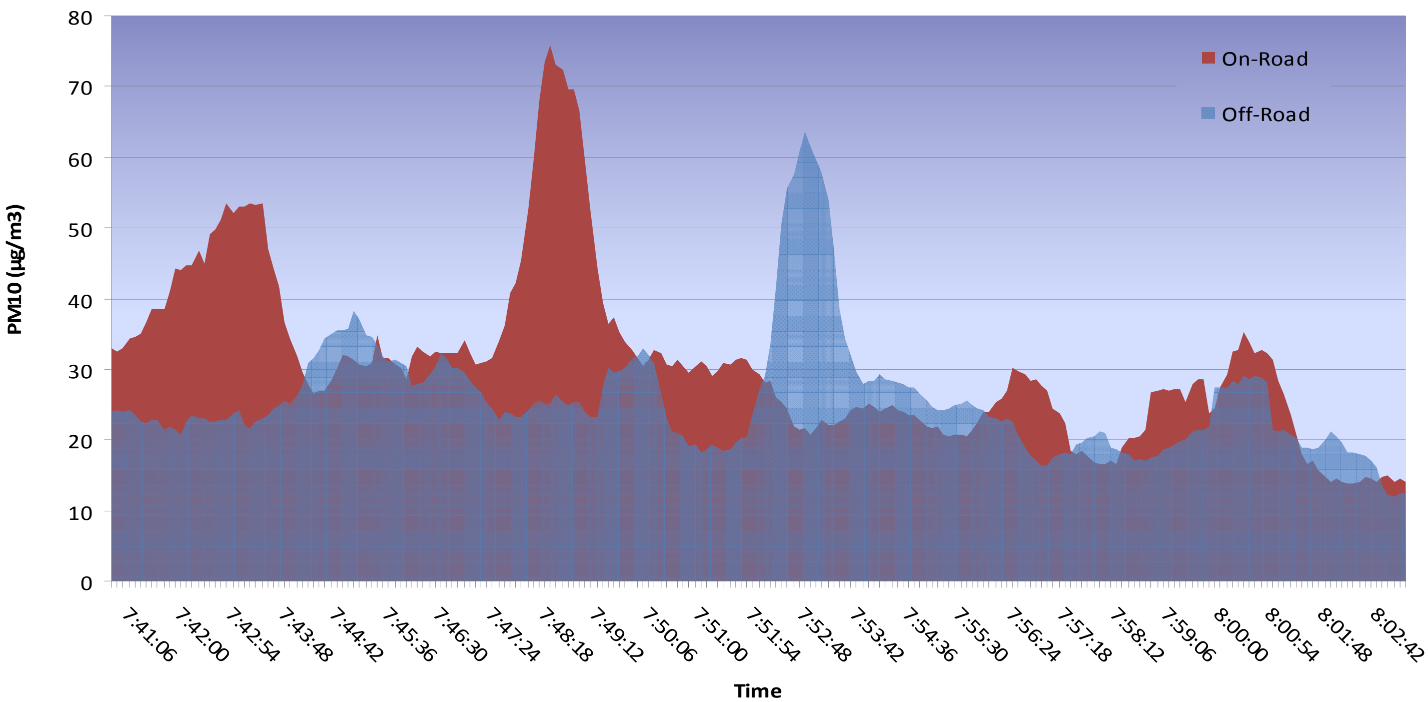

Figure 20 Cyclists' real-time comparative PM_{10} exposure showing relative spatial uniformity: Redwood to Christchurch city centre, 7:40 – 8:20 am, 19 March 2009

Figure 20 and Figure 22 show an almost perfect relationship, keeping in mind that the on-road cyclist was usually ahead by 2-5 minutes due to the shorter route. Figure 19 and Figure 21 show an obvious influence of heavy diesel vehicles on-road where peak exposure occurs, while the off-road tracks also represent very high concentrations in areas with no vehicles present, such as the middle of Hagley Park.

Figure 21 Cyclists' real-time comparative PM_{10} exposure by GPS co-ordinates: Christchurch city centre to Canterbury University, 8:30 – 9:00 am, 19 March 2009

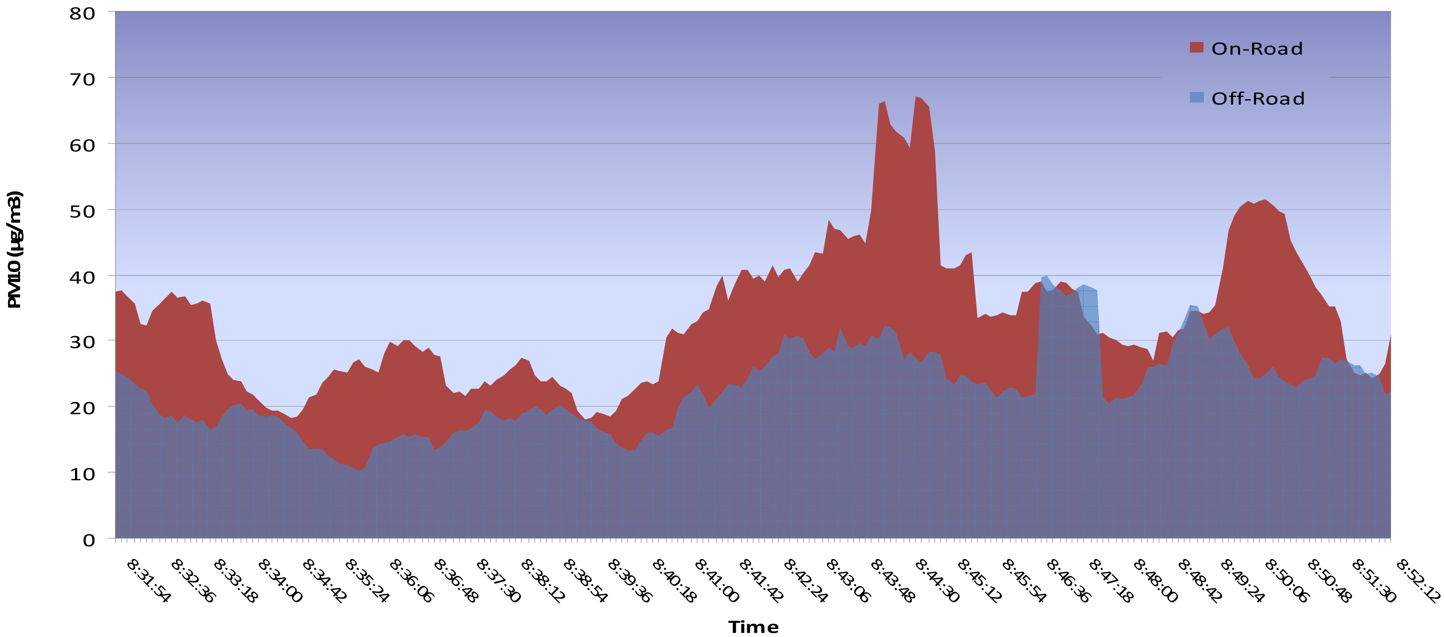

Figure 22 Cyclists' real-time comparative PM_{10} exposure showing relative spatial uniformity: Christchurch city centre to Canterbury University, 8:30 – 9:00 am, 12 March 2009

5.2.3 $PM_{2.5}$

As with PM_{10}, $PM_{2.5}$ also shows very close uniformity, with notably higher means for the off-road cyclist during some runs (Figure 23). As the dominant sources of $PM_{2.5}$ in Christchurch are home heating and industry (to a far lesser extent), the 6% difference in overall mean exposure is presumptively due to the same reasons given for PM_{10}. Again, spatial patterns are profoundly uniform (Figure 25 and Figure 27) and high concentrations are observed off-road, with no vehicles present (Figure 24 and Figure 26). These findings are in strong agreement with spatial homogeneity observed across a similar small-scale mobile study for cyclists in Vancouver (Thai et al. 2008).

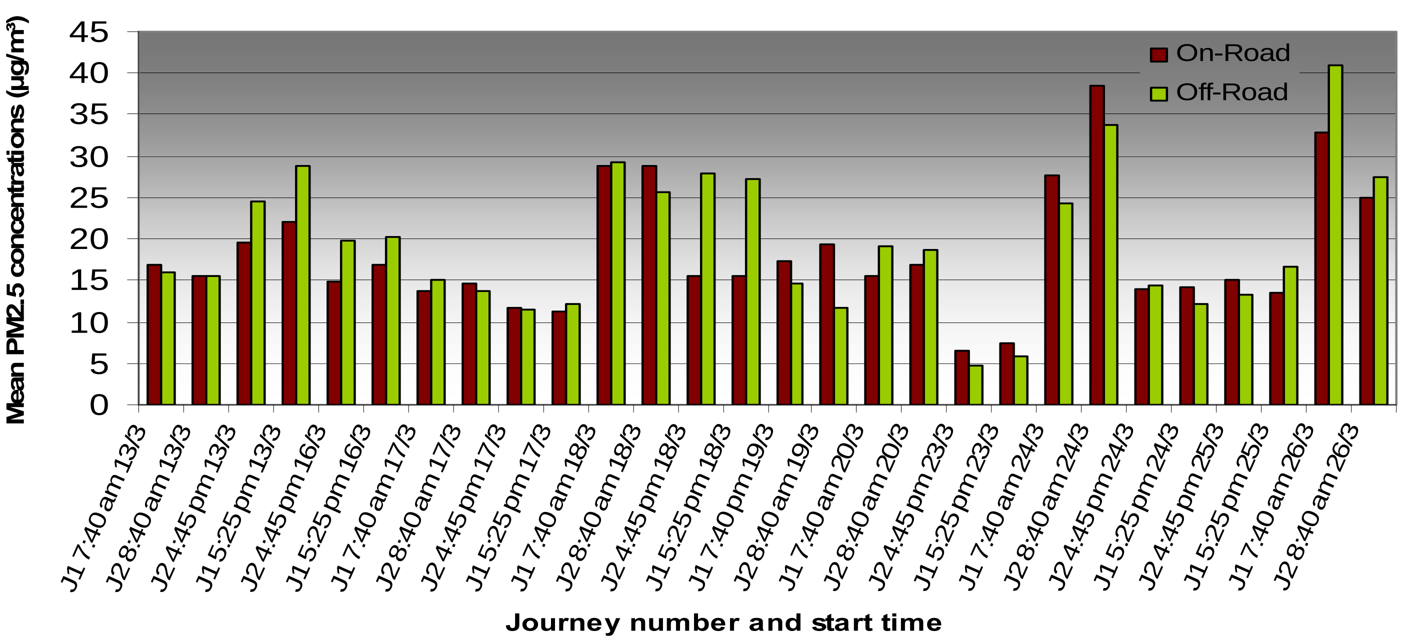

Figure 23 Cyclists' comparative mean $PM_{2.5}$ exposure by journey

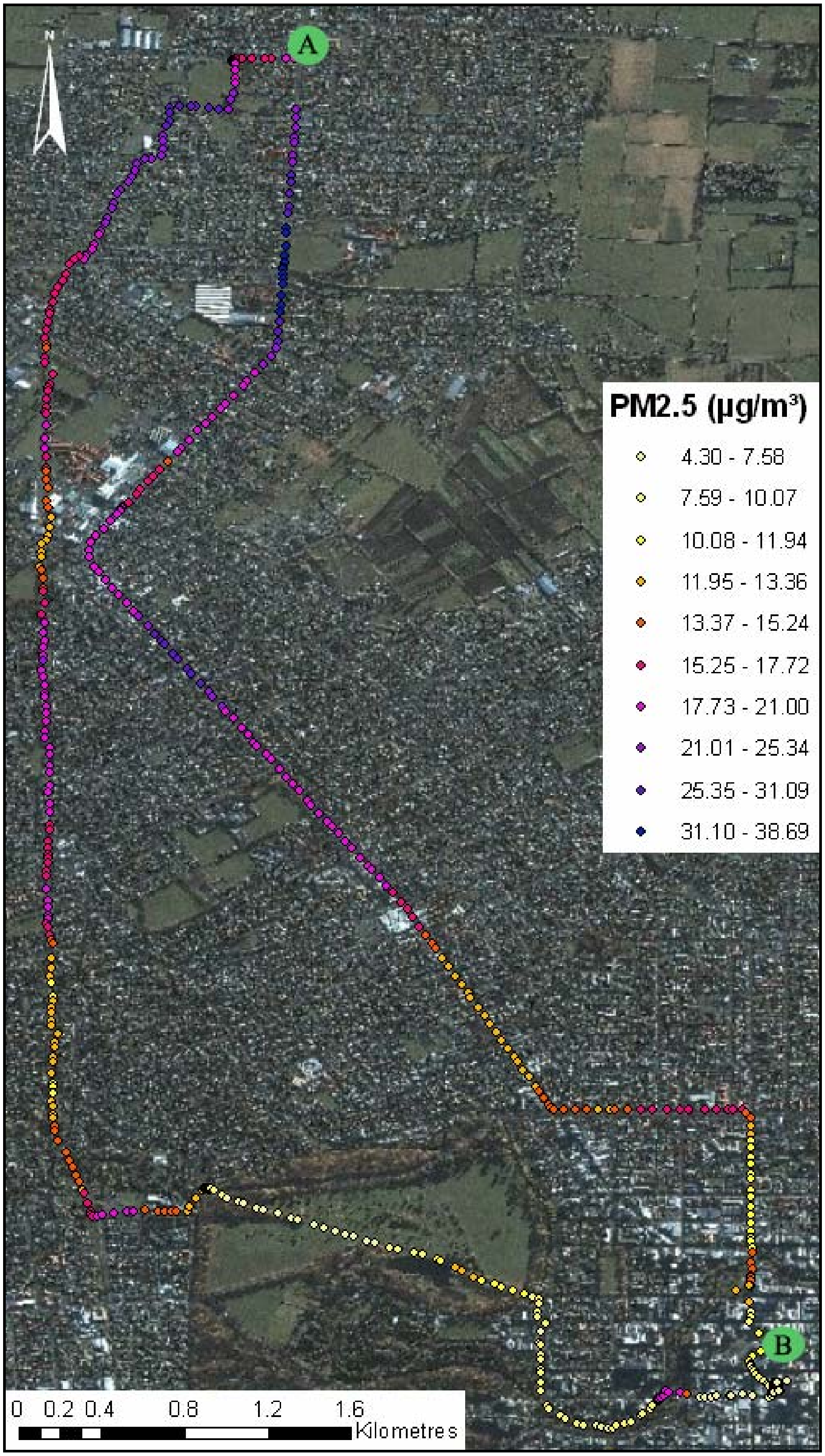

Figure 24 Cyclists' real-time comparative $PM_{2.5}$ exposure by GPS co-ordinates: Redwood to Christchurch city centre, 7:40 – 8:20 am, 19 March 2009

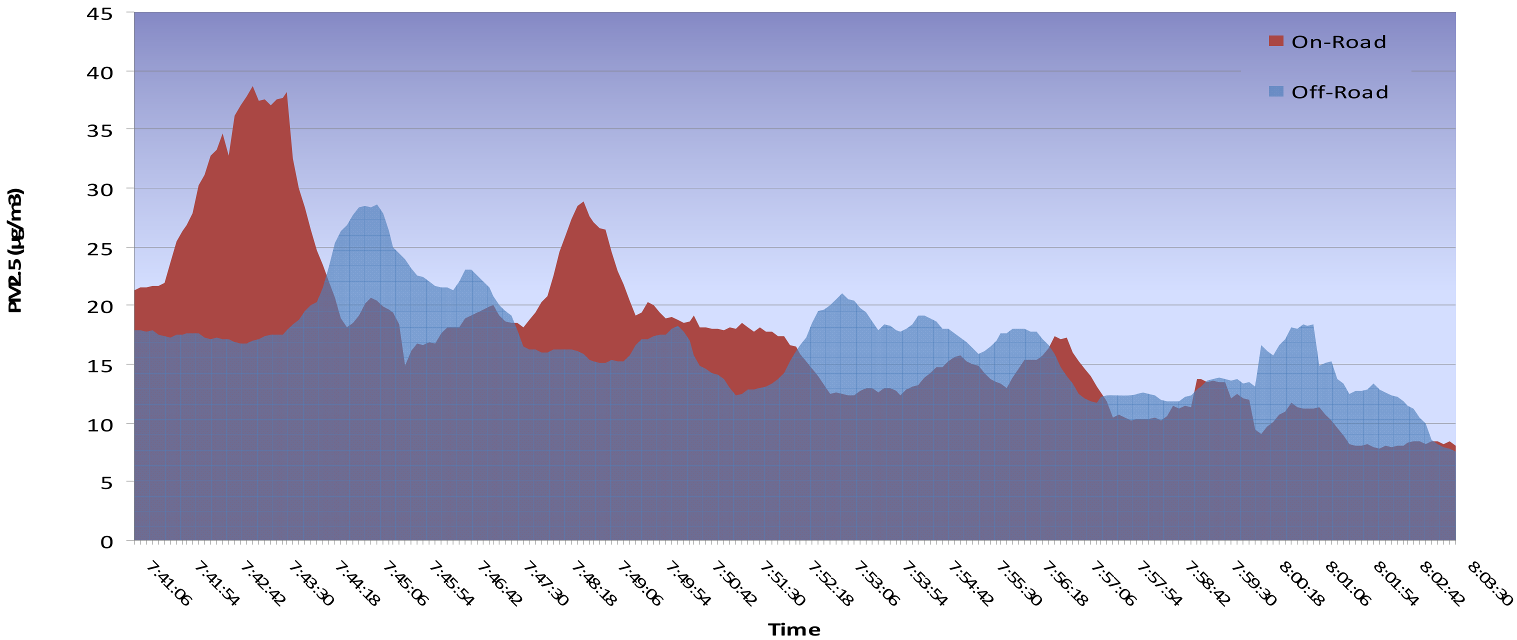

Figure 25 Cyclists' real-time comparative $PM_{2.5}$ exposure showing relative spatial uniformity: Redwood to Christchurch city centre, 7:40 – 8:20 am, 19 March 2009

Figure 26 Cyclists' real-time comparative PM_{25} exposure by GPS co-ordinates: Christchurch city centre to Canterbury University, 8:30 – 9:00 am, 19 March 2009

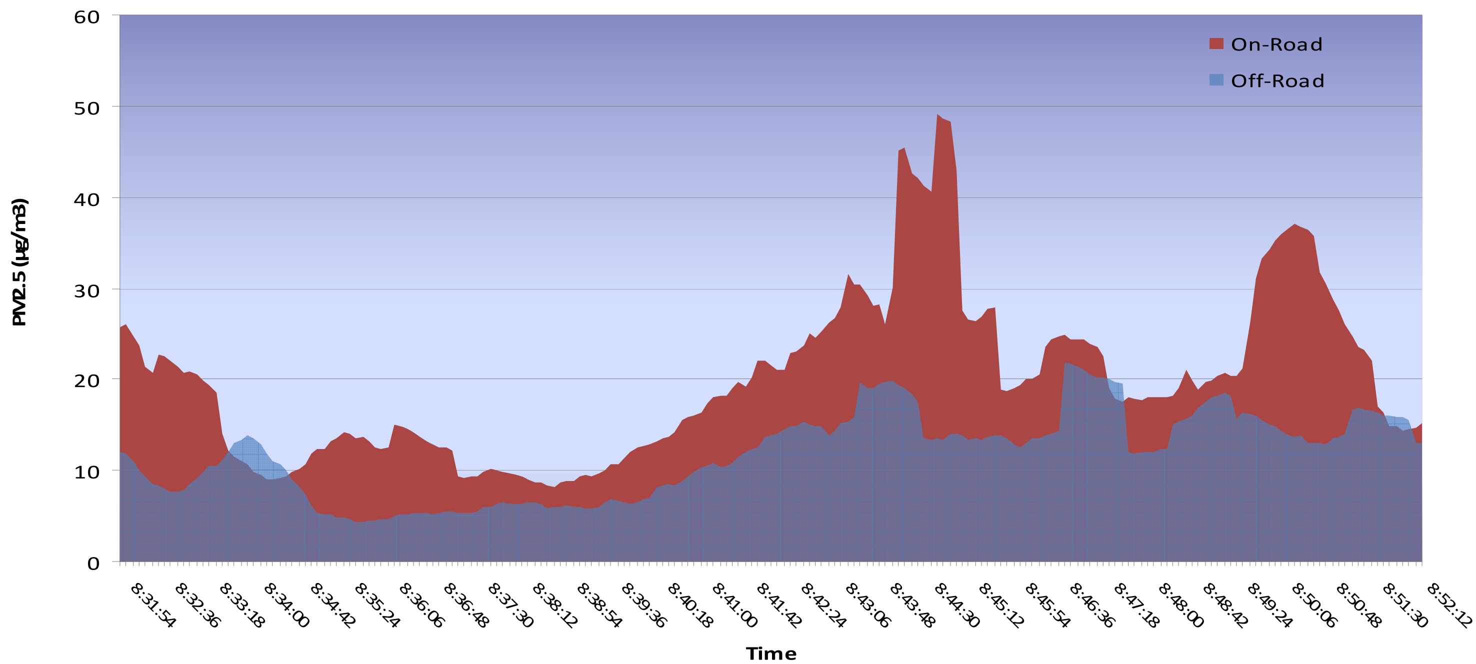

Figure 27 Cyclists' real-time comparative $PM_{2.5}$ exposure showing relative spatial uniformity: Christchurch city centre to Canterbury University, 8:30 – 9:00 am, 12 March 2009

5.2.4 $PM_{1.0}$

As for CO, a significant reduction in overall mean exposure (31%) was observed for the off-road cyclist due to the reliance on traffic emissions as the primary source. However, in contrast to CO, spatial distribution appeared to be highly uniform. This is reflected in pollutant traces for numerous days (see Figure 30 and Figure 32 for an example), as well as in comparative mean exposure by journey (Figure 28). Uniformity is also visible in pollutant concentration mapping, with high off-road exposure observed in residential backstreet areas, similar to on-road levels (Figure 29 and Figure 31). The only differences causing such a large overall disparity was the occurrence of peak exposure points for the on-road cyclist (queued traffic, street canyons) and minimum exposure areas experienced by the off-road cyclist (parkland sections). On-road exposure was higher for all days with the exception of the afternoon runs for the 18th of March.

This observed lack of spatial variation was also found in Mol, Belgium, where bicycle measurements along a 17 kilometre route traversing a variety of land uses rendered little variation (Berghmans et al. 2009). Exposure peaks only occurred when coming into contact with heavy traffic, reflecting the impact of traffic proximity and the long atmospheric residency of fine particles. Such uniformity is attributable to the lightweight, non-reactive characteristics of $PM_{1.0}$ and associated slow settling time (Phalen 2002).

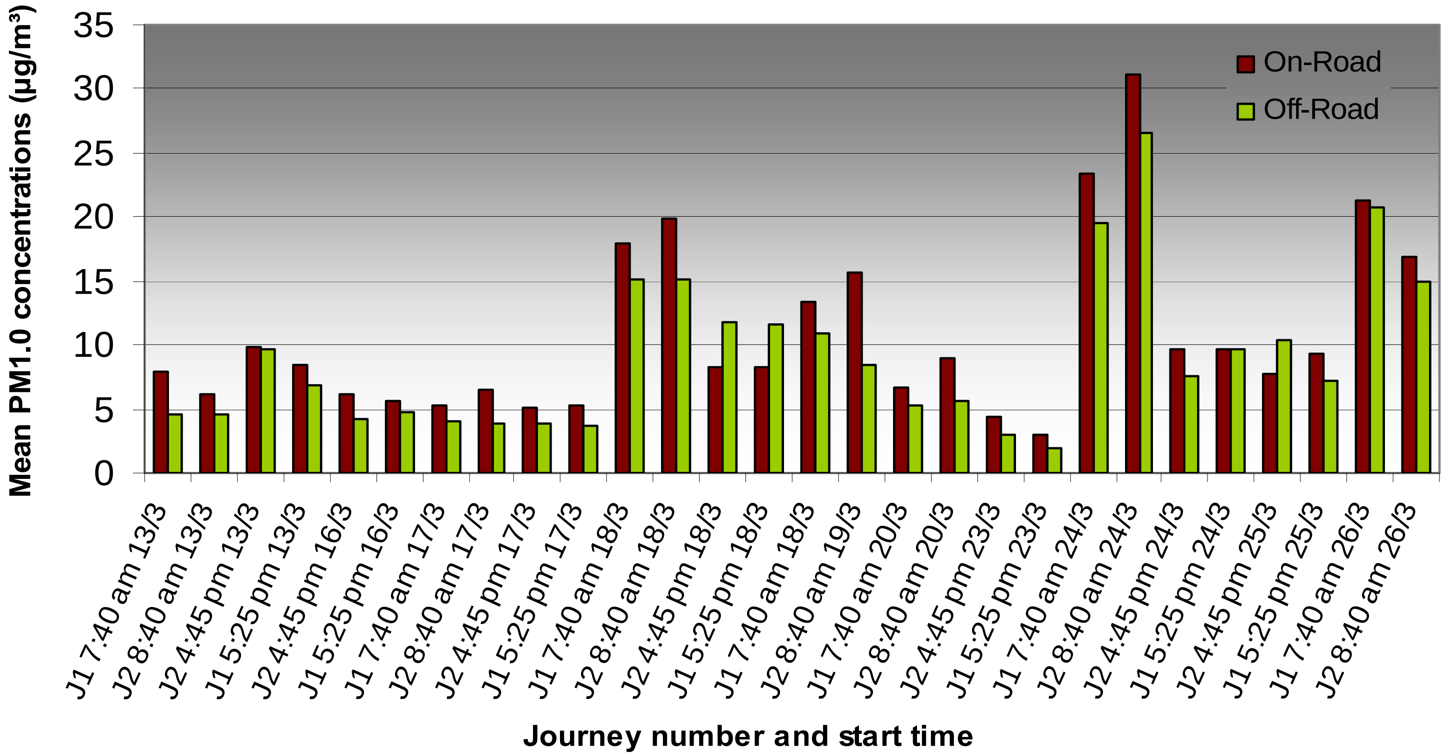

Figure 28 Cyclists' comparative mean $PM_{1.0}$ exposure by journey

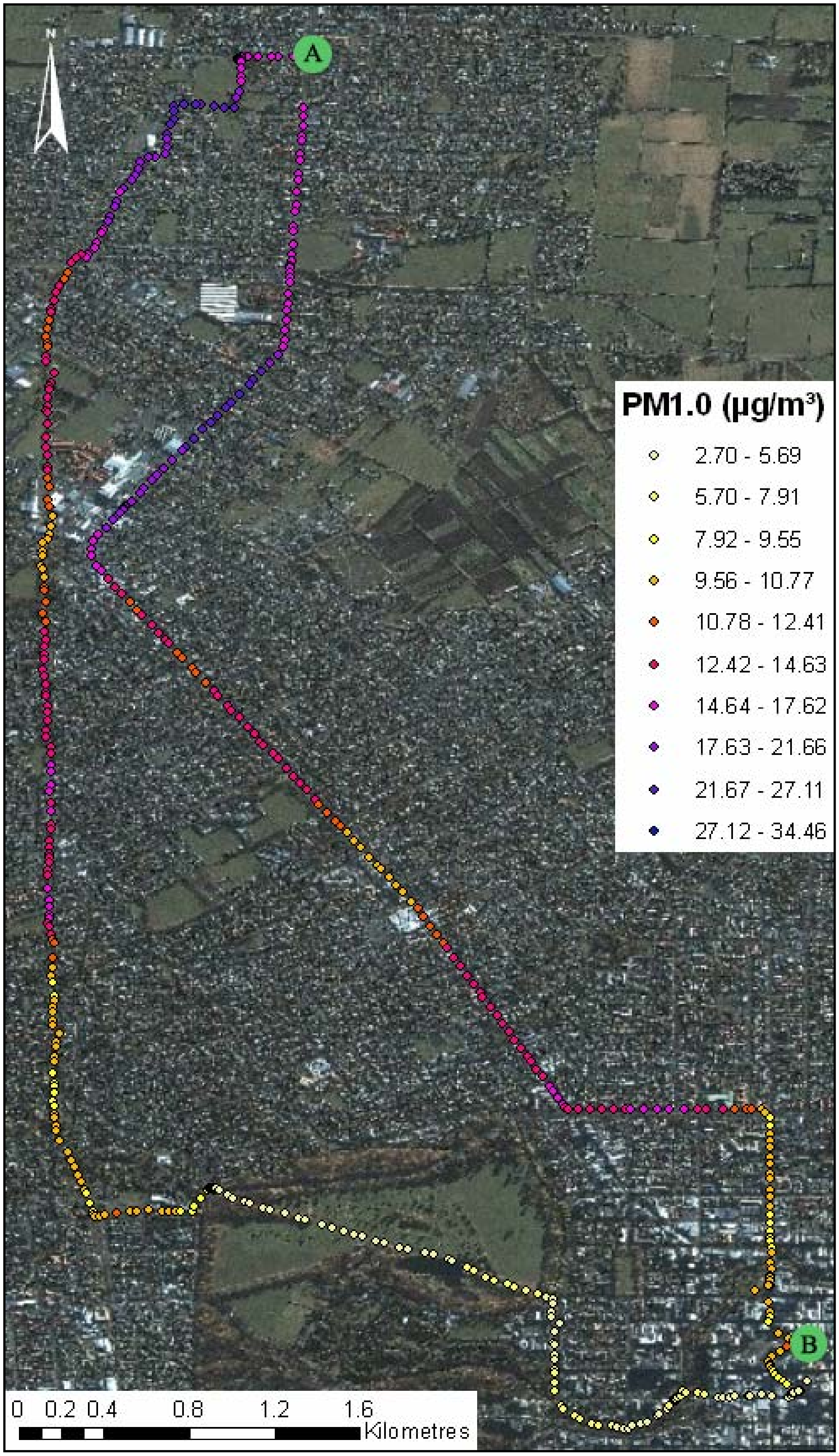

Figure 29 Cyclists' real-time comparative $PM_{1.0}$ exposure by GPS co-ordinates: Redwood to Christchurch city centre, 7:40 – 8:20 am, 19 March 2009

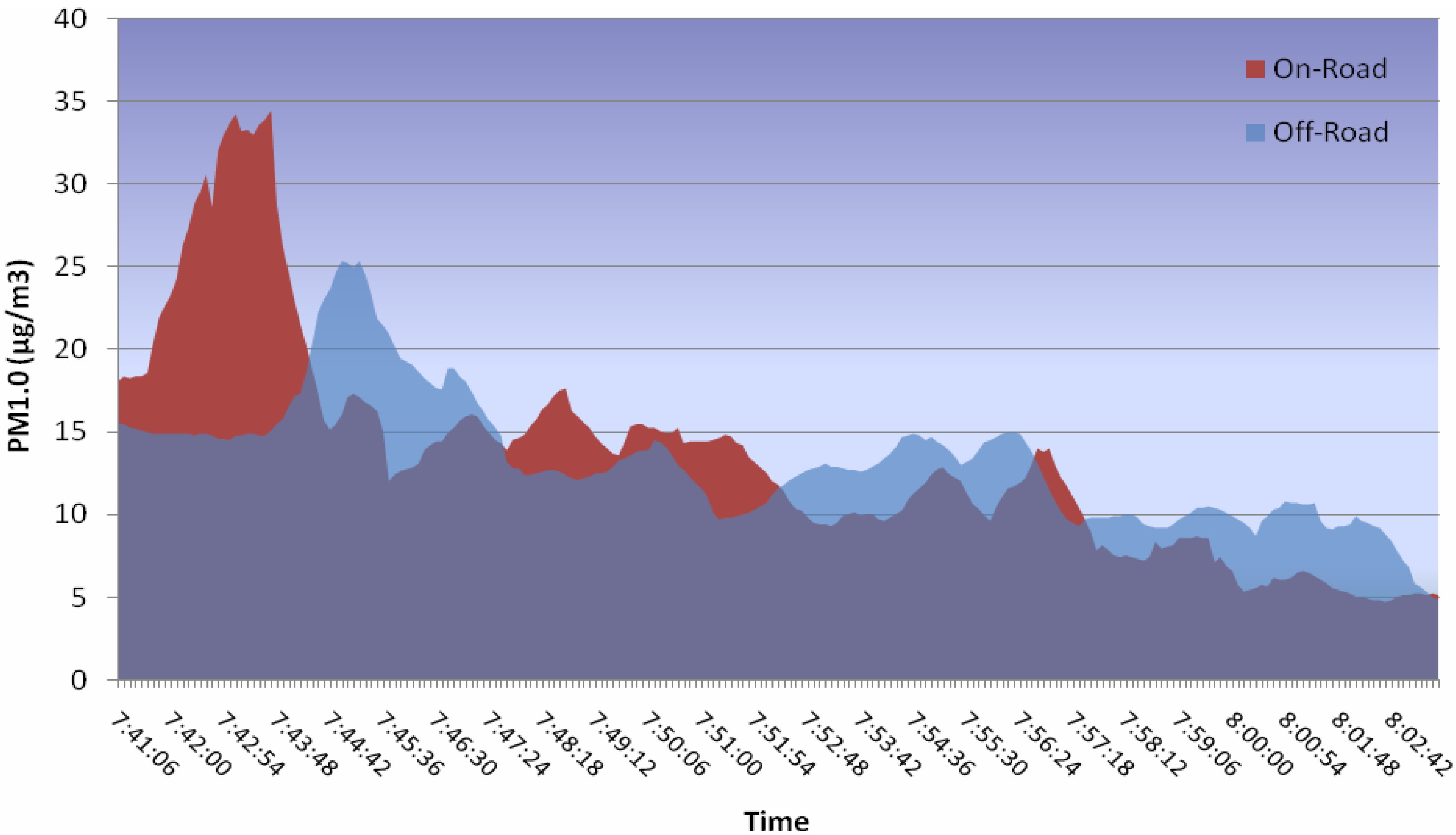

Figure 30 Cyclists' real-time comparative $PM_{1.0}$ exposure showing relative spatial uniformity: Redwood to Christchurch city centre, 7:40 – 8:20 am, 19 March 2009

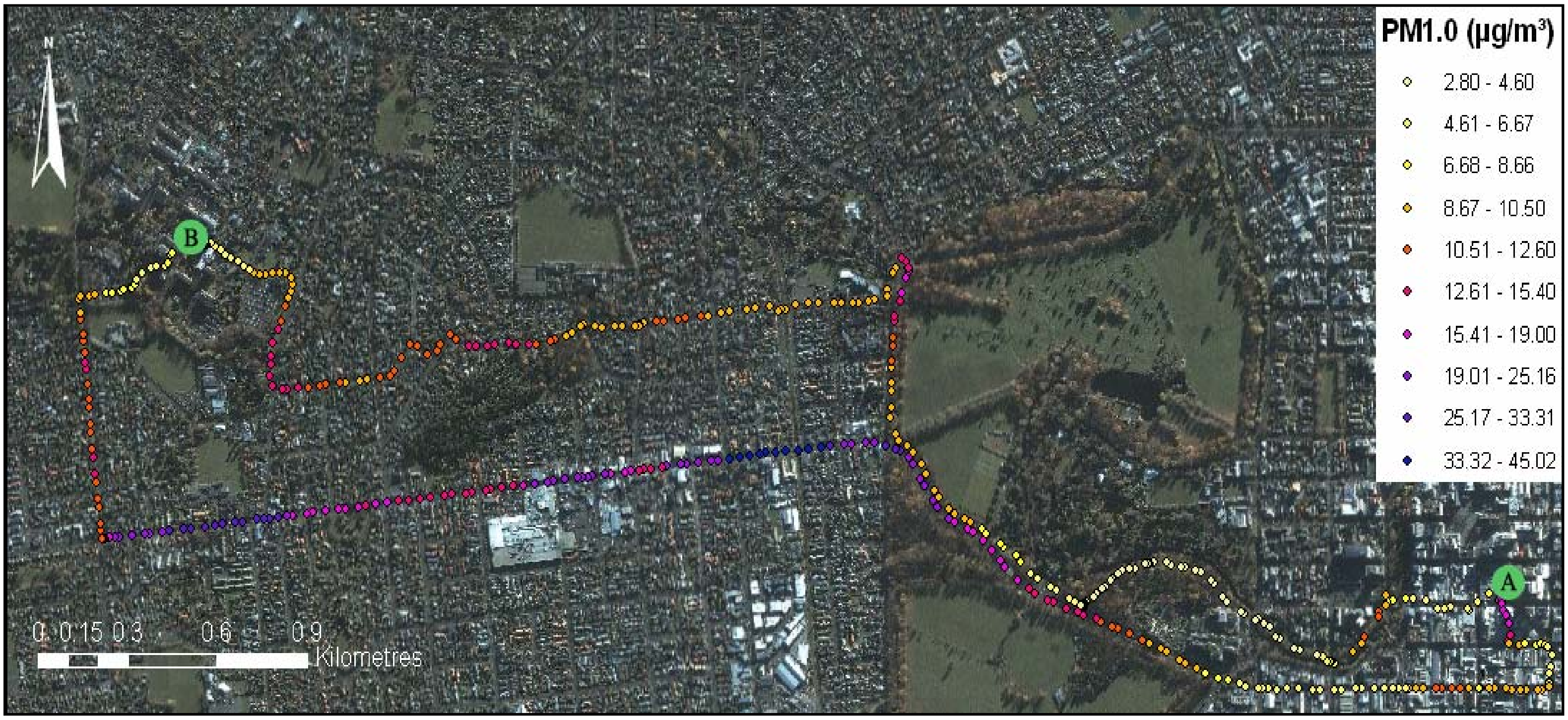

Figure 31 Cyclists' real-time comparative $PM_{1.0}$ exposure by GPS co-ordinates: Christchurch city centre to Canterbury University, 8:30 – 9:00 am, 19 March 2009

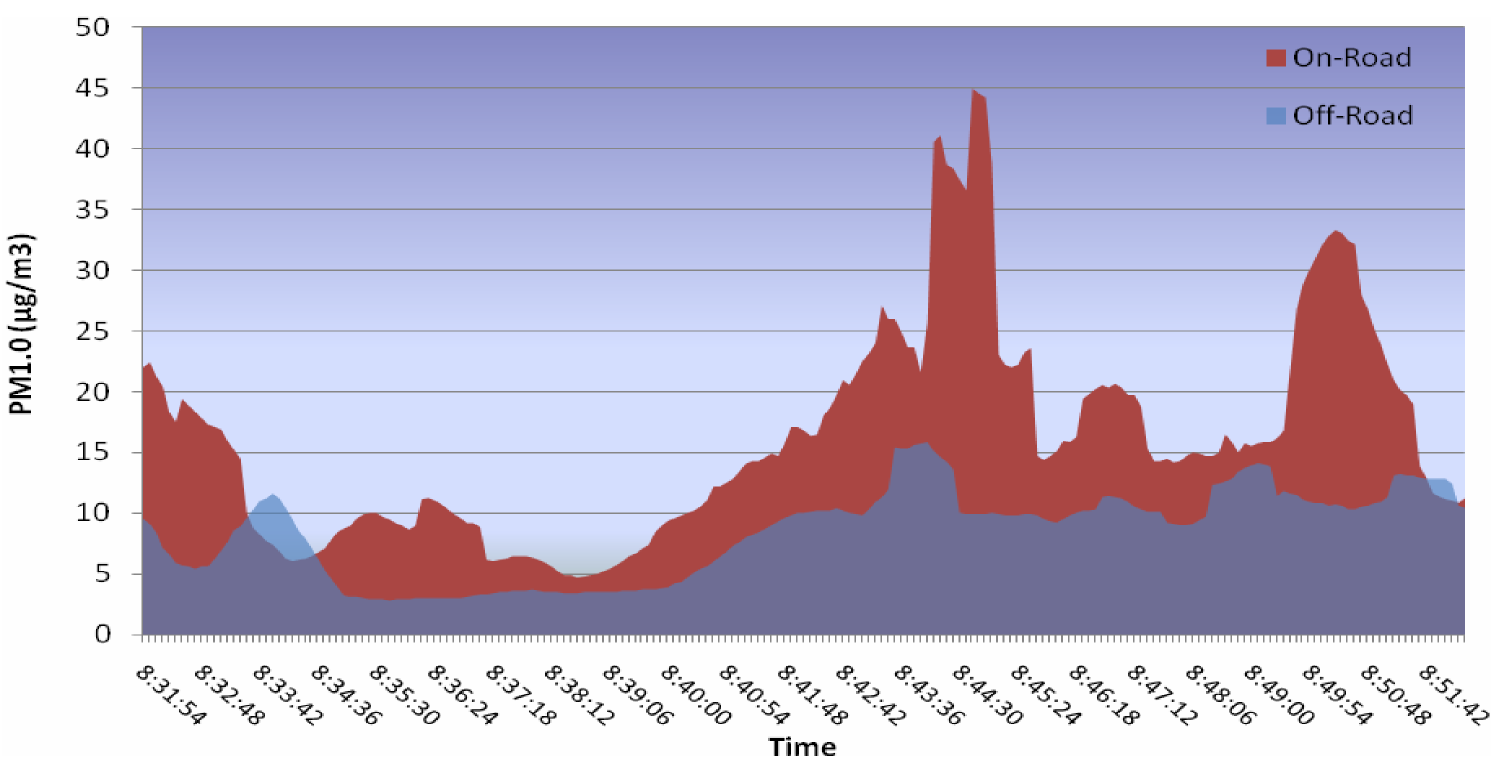

Figure 32 Cyclists' real-time comparative $PM_{1.0}$ exposure showing relative spatial uniformity: Christchurch city centre to Canterbury University, 8:30 – 9:00 am, 12 March 2009

5.2.5 Ultrafine particles

Journey mean exposure for the on-road cyclist was often in excess of off-road exposure by a factor of 3 or more, giving a disparity of 53% between means (Figure 33). Exposure was always lowest off-road, with the exception of the first three sampling runs discussed in section 5.2.1. A significant negative relationship was found for UFPs and temperature, explained by accelerated coagulation and condensation at higher temperatures (Vinzents et al. 2005). Comparisons between ultrafine particle exposure showed no spatial agreement (Figure 35 and Figure 37). Significant temporal variation is also evident from these figures when looking at high exposure events. While all extreme peak events are very rapid, usually lasting only several seconds or more, the fluctuations are generally more gradual for the on-road cyclist. Levels can stay elevated for several minutes, sustained by the constant presence of fresh emissions when in high traffic situations. The influence of traffic is clearly visible in Figure 34 and Figure 36. When not in the vicinity of running vehicles, UFP levels are virtually zero. This is reflected in the minimum values for both the on-road (23 pt/cm^3) and off-road cyclists (85 pt/cm^3). Concentration mapping only shows elevated levels when in very close proximity to traffic, with the exception of riding along a gravel path section in Hagley Park and along the riverside (the orange and red sector just before arriving at point B, southeast corner, Figure 34).

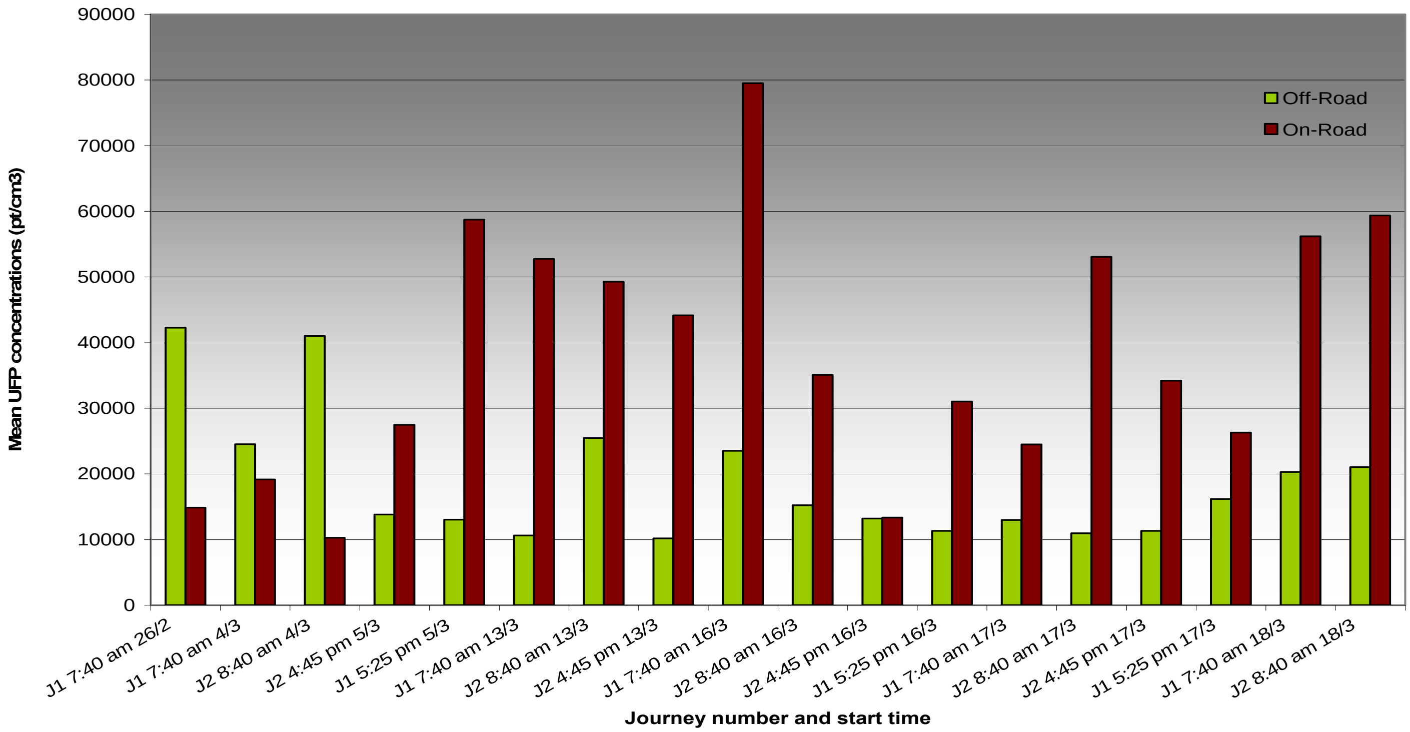

Figure 33 Cyclists' comparative mean UFP exposure by journey

The observed spatial heterogeneity of UFPs is consistent with findings of previous recent cyclist sampling campaigns (Berghmans et al. 2009; Kaur & Nieuwenhuijsen 2009; Thai et al. 2008). A meta-analysis of mobile source dispersion modeling studies concluded that the spatial extent of UFPs is limited to 100-300 metres (Zhou & Levy 2007). However, a large portion of UFPs are lost within seconds of being emitted due to coagulation into larger particles. This process is known as the transition between the *transient nuclei mode* (formed by condensation and nucleation) and the *accumulation mode*, when vapours condense onto existing particles (Phalen 2002). Secondly, a complex interplay between influential variables effects actual distance traveled. Under many circumstances it is likely most particles not lost to coagulation travel far less than 100 metres. The atmospheric behaviour of ultrafine particles is further discussed in section 5.3.3.

Figure 34 Cyclists' real-time comparative UFP exposure by GPS co-ordinates: Canterbury University to Christchurch city centre, 4:45 – 5:05 pm, 17 March 2009

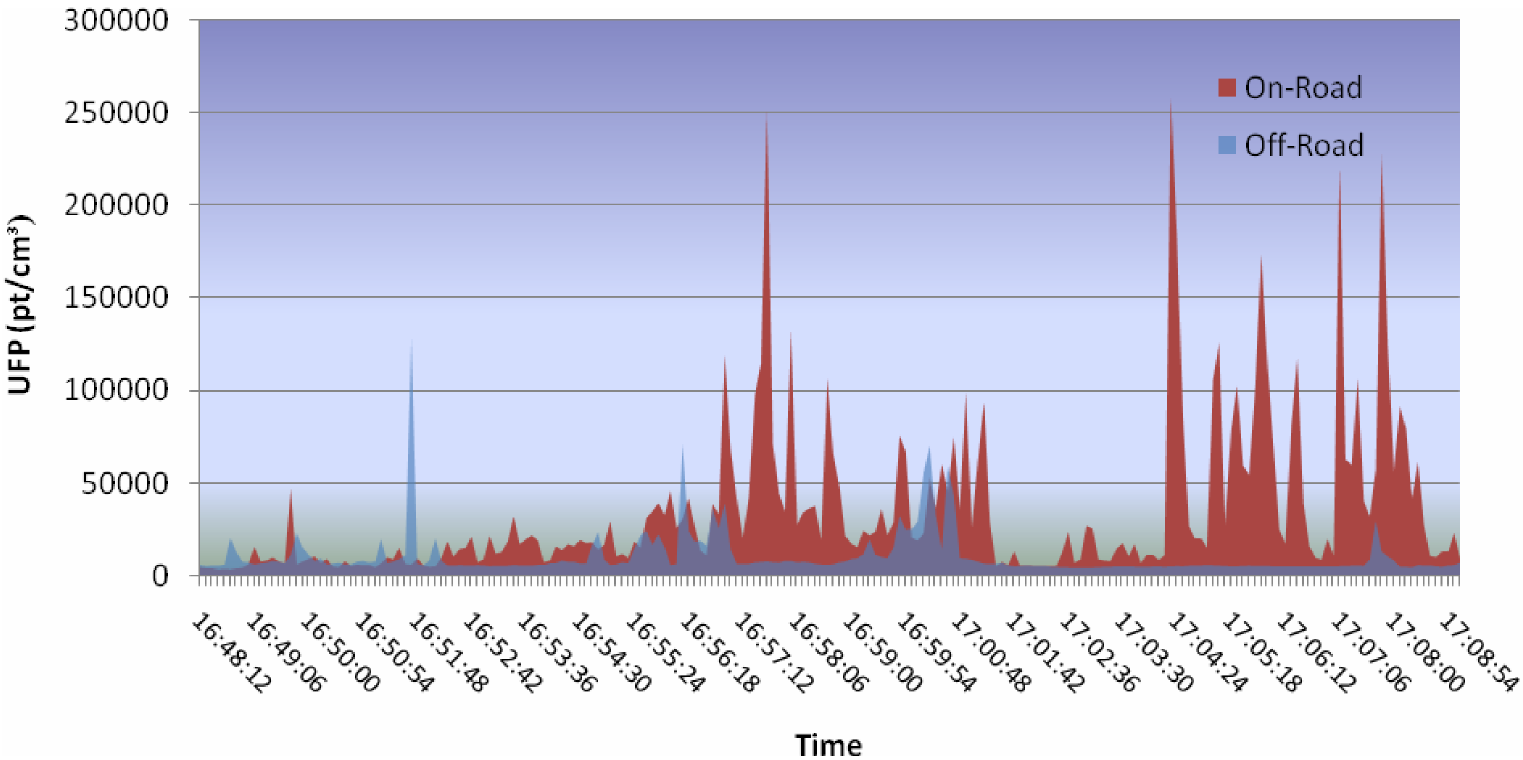

Figure 35 Cyclists' real-time comparative UFP exposure: Canterbury University to Christchurch city centre, 4:45 – 5:05 pm, 17 March 2009

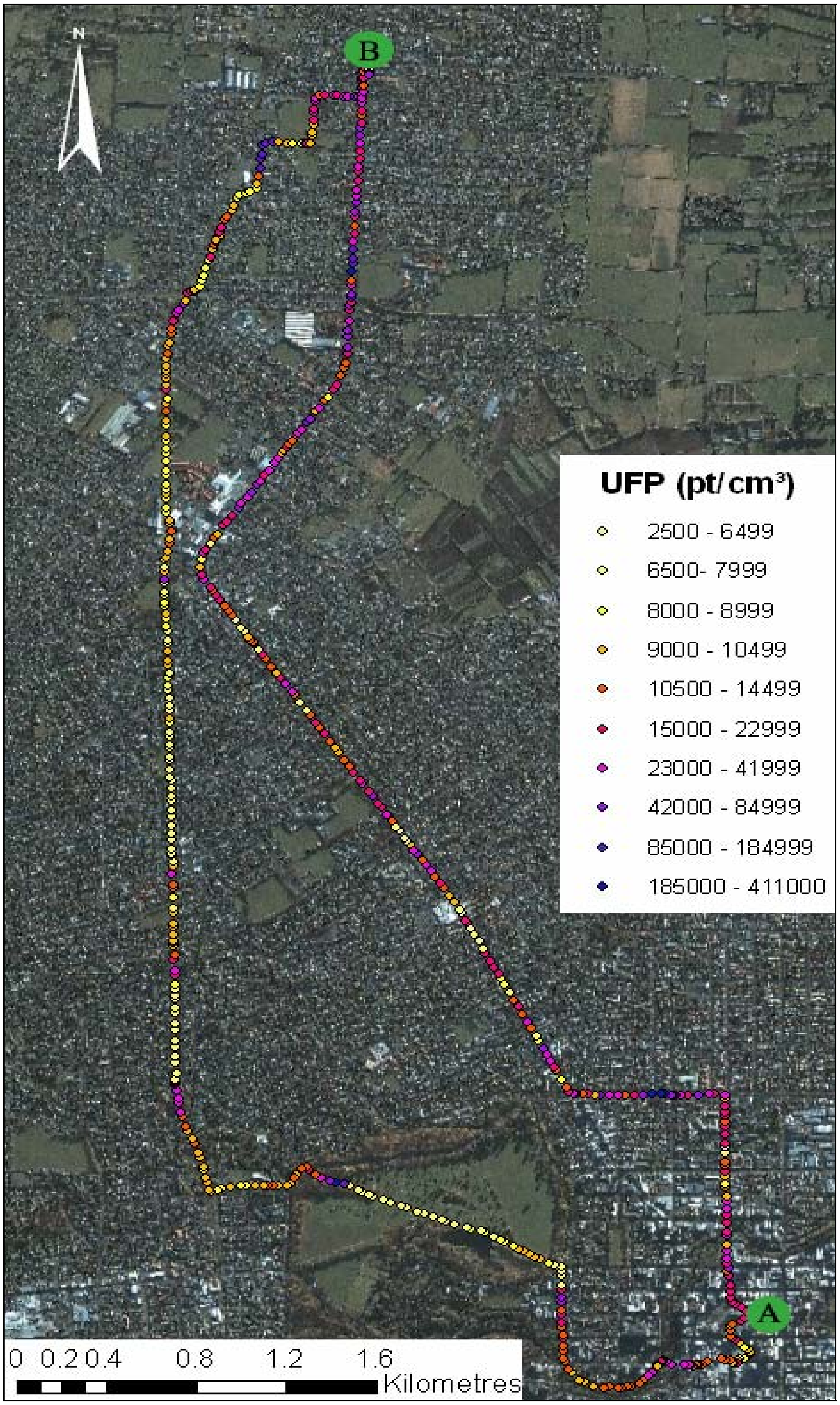

Figure 36 Cyclists' real-time comparative UFP exposure by GPS co-ordinates: Christchurch city centre to Redwood, 5:25 – 6:00 pm, 17 March 2009

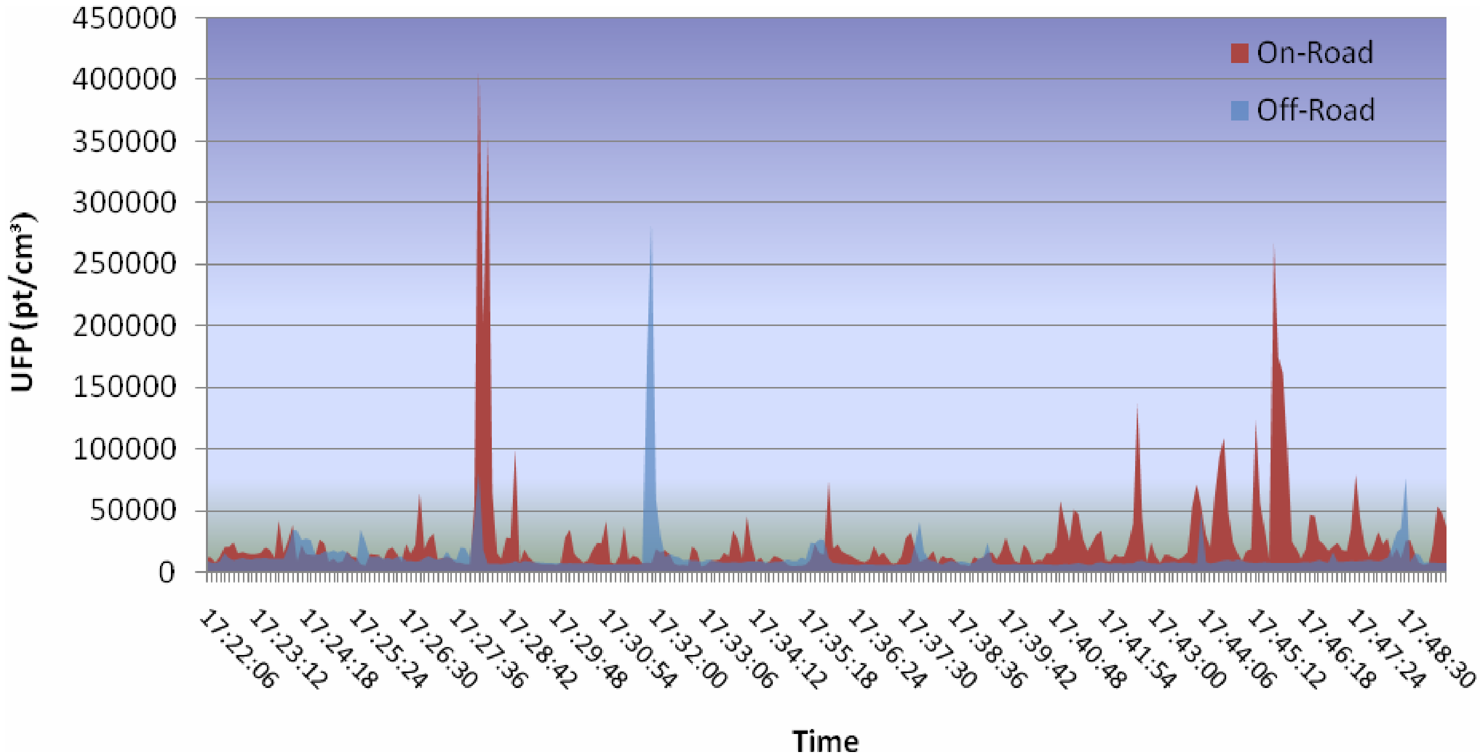

Figure 37 Cyclists' real-time comparative UFP exposure: Christchurch city centre to Redwood, 5:25 – 6:00 pm, 17 March 2009

5.3 Christchurch: Effect of proximity to traffic

5.3.1 Carbon monoxide

A substantial decrease with distance from traffic was observed for CO. The off-road cyclist had lower exposure than the one on the sidewalk and on the road, with a difference of 41% and 54%, respectively. Mean exposure for the cyclist riding along the pavement was 22% lower than the cyclist on the road (Table 29). Reduced exposure was significant across cyclist position in relation to traffic ($F_{2,449.12}$=82.27, p<0.001). Table 30 presents full ANOVA results.

Average temperature (p<0.001) and wind speed (p=0.001) were negatively correlated with CO. This mirrors findings in previous research (Kaur & Nieuwenhuijsen 2009). The Wind Strength fixed factor was also significant (p=0.002), with a higher median value for leg means and a greater number of extreme outliers under low wind speed conditions (Figure 38). The position of cyclist relative to wind direction and traffic did not render a significant relationship. This may be due to the majority of cyclist sampling being conducted during low wind speed periods (<2 m/s), when wind direction is quite variable.

Table 29 Summary statistics for cyclist CO exposure in Christchurch

Mode of transport	***N* Legs (samples)**	**Mean** (+/-0.2 ppm)	**Std. Dev.**	**Min**	**Max**	**Median**	**-95% Cnf.Lmt**	**+95% Cnf.Lmt**
All samples	459 (9325)	0.89	0.73	0.05	12.8	0.69	0.87	0.90
Cycle on-road	153 (3106)	1.19	0.75	0.05	12.8	1.00	1.16	1.22
Cycle footpath	153 (3101)	0.93	0.79	0.05	7.14	0.74	0.90	0.96
Cycle off-road	153 (3114)	0.55	0.45	0.05	3.66	.42	0.53	0.56

Table 30 ANOVA results for cyclist CO exposure in Christchurch

Variable	Effect (F/R)	df Effect	MS Effect	df Error	MS Error	F	p
Avg Wind Speed	***Fixed**	**1**	**3.61**	**129.16**	**0.34**	**10.69**	**0.001**
Avg Temp	***Fixed**	**1**	**4.36**	**129.26**	**0.34**	**12.89**	**<0.001**
{1}Direction	*Fixed	2	0.01	55.00	0.46	0.03	0.970
{2}Mode	***Fixed**	**2**	**19.93**	**449.12**	**0.24**	**82.27**	**<0.001**
{3}Wind Strength	***Fixed**	**1**	**4.72**	**57.88**	**0.45**	**10.46**	**0.002**
{4}Wind Influence	Fixed	1	0.40	170.31	0.32	1.28	0.259
{5}Leg	**Random**	**23**	**1.02**	**427.00**	**0.23**	**4.34**	**<0.001**

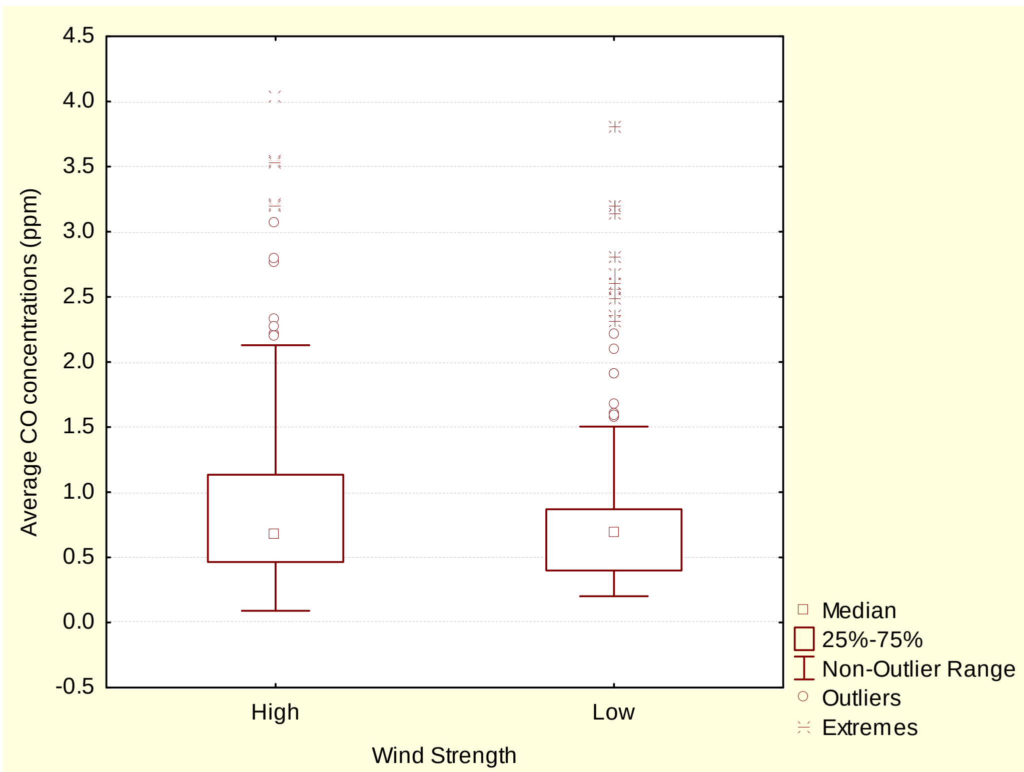

Figure 38 Mean CO exposure values grouped by Wind Strength

5.3.2 PM_{10}, $PM_{2.5}$ and $PM_{1.0}$

The off-road cyclist was exposed to 26% and 32% less $PM_{1.0}$ than the footpath and on-road cyclist (Table 31). While the on-road to off-road reduction for $PM_{2.5}$ and PM_{10} was relatively high (21%, 16%), the drop-off in concentrations was much lower from the footpath to the off-road path (8%, 3%). Differences between the on-road and footpath cyclists were also observed, with a reduction of 13% (PM_{10}), 14% ($PM_{2.5}$) and 9% ($PM_{1.0}$), only 7 metres from traffic. Results were significant across cyclist position in relation to traffic for all three fine-coarse particle fractions (Table 32): $F_{2,352.92}=6.52$, $p=0.002$ (PM_{10}), $F_{2,351.03}=5.55$, $p=0.004$ ($PM_{2.5}$) and $F_{2,352.42}=8.24$, $p<0.001$ ($PM_{1.0}$). All max concentrations occurred on the road, with the exception of PM_{10}, sampled at the footpath. For average temperature, there was a significant positive correlation for PM_{10} ($p=0.003$), no correlation for $PM_{2.5}$ ($p=0.557$) and a negative correlation for $PM_{1.0}$ ($p<0.001$). Average wind speed was significant and relationships were negative; $p<0.001$ (PM_{10}/$PM_{2.5}$) and $p=0.033$ ($PM_{1.0}$). The fixed wind strength effect (Figure 39) was significant for $PM_{1.0}$ ($p<0.001$), with many outliers towards the upper end of the range within the low speed category (<2 m/s), yet the median and overall range was much higher for higher wind speeds (>2 m/s).

Table 31 Summary statistics for cyclist PM exposure in Christchurch

Mode of transport		*N* Legs (samples)	Arithmetic Mean (µg/m³)	Std. Dev.	Min	Max	Median	-95% Cnf.Lmt	+95% Cnf.Lmt
All samples	*PM_{10}*	*414 (9558)*	*21.16*	*11.46*	*4.0*	*197*	*19.28*	*20.93*	*21.39*
	$PM_{2.5}$		*9.97*	*4.50*	*1.2*	*41*	*9.52*	*9.88*	*10.06*
	$PM_{1.0}$		*4.97*	*3.24*	*0.4*	*37*	*4.37*	*4.90*	*5.03*
Cycle on-road	PM_{10}	169 (4023)	23.14	10.51	6.9	105	21.68	22.82	23.47
	$PM_{2.5}$		11.08	4.83	3.23	41	11.27	10.93	11.23
	$PM_{1.0}$		5.61	3.75	1.4	37	4.40	5.49	5.72
Cycle footpath	PM_{10}	145 (2913)	20.05	13.09	7.24	197.2	17.20	19.57	20.52
	$PM_{2.5}$		9.50	3.10	3.04	33.8	9.10	9.39	9.62
	$PM_{1.0}$		5.13	2.92	1.5	30.5	4.68	5.03	5.24
Cycle off-road	PM_{10}	157 (3711)	19.36	10.39	4	99	17.00	18.96	19.76
	$PM_{2.5}$		8.78	4.87	1.2	27	7.40	8.59	8.97
	$PM_{1.0}$		3.80	2.28	0.4	18	3.97	3.71	3.89

Table 32 ANOVA results for cyclist PM exposure in Christchurch

PM_{10}							
Variable	**Effect (F/R)**	**df Effect**	**MS Effect**	**df Error**	**MS Error**	**F**	**p**
Avg Wind Speed	***Fixed**	**1**	**2957.09**	**310.76**	**70.13**	**42.17**	**<0.001**
Avg Temp	***Fixed**	**1**	**651.74**	**168.19**	**73.65**	**8.85**	**0.003**
{1}Direction	*Fixed	2	59.59	144.41	74.53	0.80	0.452
{2}Mode	***Fixed**	**2**	**435.70**	**352.92**	**66.79**	**6.52**	**0.002**
{3}Wind Strength	*Fixed	1	24.84	149.90	74.31	0.33	0.564
{4}Wind Influence	Fixed	1	54.79	348.70	69.20	0.79	0.374
{5}Leg	Random	23	93.73	350.00	66.70	1.41	0.104
$PM_{2.5}$							
Avg Wind Speed	***Fixed**	**1**	**542.93**	**372.89**	**12.64**	**42.95**	**<0.001**
Avg Temp	*Fixed	1	4.05	333.99	11.73	0.34	0.557
{1}Direction	*Fixed	2	6.63	314.55	11.51	0.58	0.563
{2}Mode	***Fixed**	**2**	**74.86**	**351.03**	**13.50**	**5.55**	**0.004**
{3}Wind Strength	*Fixed	1	26.82	319.66	11.56	2.32	0.129
{4}Wind Influence	Fixed	1	6.59	371.64	12.88	0.51	0.475
{5}Leg	Random	23	6.57	350.00	13.52	0.49	0.980
$PM_{1.0}$							
Avg Wind Speed	***Fixed**	**1**	**18.13**	**333.96**	**3.96**	**4.58**	**0.033**
Avg Temp	***Fixed**	**1**	**395.91**	**199.34**	**4.04**	**97.98**	**<0.001**
{1}Direction	*Fixed	2	4.96	172.54	4.06	1.22	0.297
{2}Mode	***Fixed**	**2**	**31.99**	**352.42**	**3.88**	**8.24**	**<0.001**
{3}Wind Strength	***Fixed**	**1**	**706.04**	**178.83**	**4.06**	**174.09**	**<0.001**
{4}Wind Influence	Fixed	1	1.31	360.99	3.94	0.33	0.565
{5}Leg	Random	23	4.50	350.00	3.88	1.16	0.280

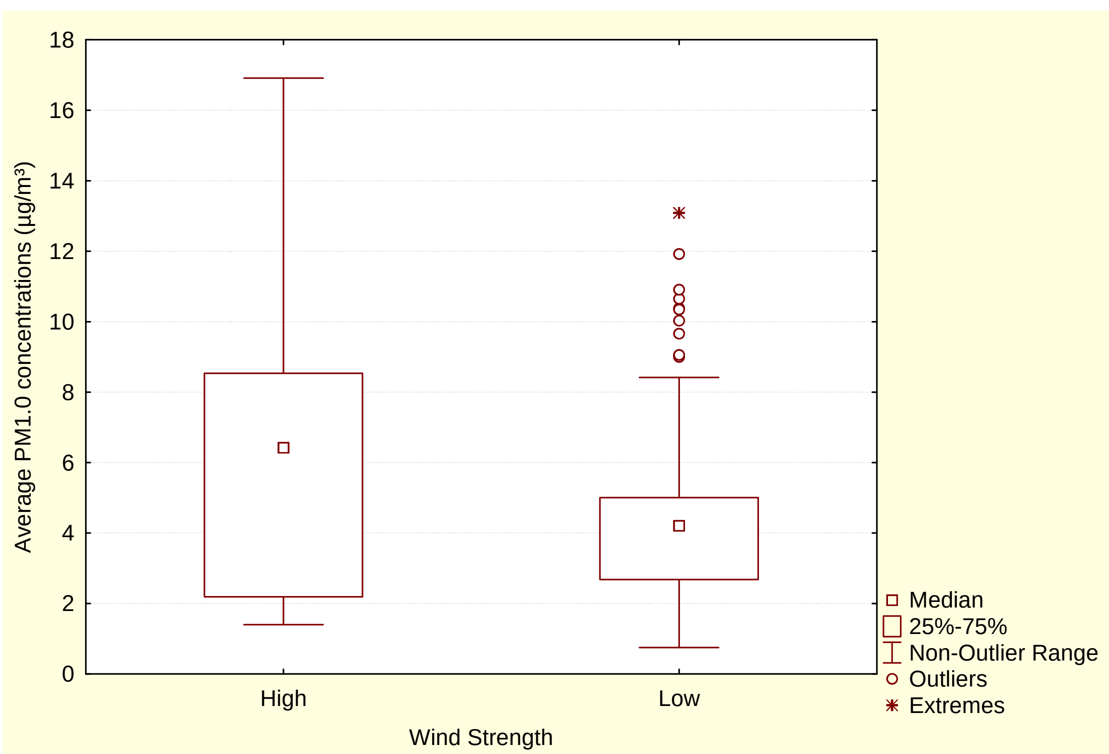

Figure 39 Mean $PM_{1.0}$ exposure values grouped by Wind Strength

5.3.3 Ultrafine particles

Overall mean UFP exposure decreased substantially with greater distance from traffic. At 7 metres away, the footpath cyclist was 30% less exposed than the on-road cyclist (Table 33). At 19 metres away, the off-road cyclist's exposure was lower by 17% (footpath) and 42% (on-road). The maximum concentration was observed by the on-road cyclist and the minimum by the off-road cyclist.

Exposure was highly significant across cyclist position in relation to traffic ($F_{2,314.94}$=24.25, p<0.001). UFP levels were positively correlated with wind speed (p=0.007) and no association was found for temperature (Table 34). Wind strength (p<0.001) and the influence of position in relation to wind direction were found to be significant (p=0.003). The median value for mean exposure at >2 m/s was almost double that of low wind speed exposure (Figure 40). Being positioned downwind from traffic made a substantial difference, resulting in a higher median and more extreme values (Figure 41). Figure 42 illustrates the influence of wind direction, showing two traverses of the sampling area – A to B on the north side and then C to D on the south side, with the three series of points representing the cyclists at different positions to the road. Note the general pattern showing the lowest concentrations when position is upwind relative to traffic and the highest when position is downwind.

Table 33 Summary statistics for cyclist UFP exposure in Christchurch

Mode of transport	***N* Legs (samples)**	**Arithmetic Mean** (pt/cm^3)	**Std. Dev.**	**Min**	**Max**	**Median**	**-95% Cnf.Lmt**	**+95% Cnf.Lmt**
All samples	*355 (8352)*	*33606.85*	*48226.19*	*185.0741*	*1588280*	*23938.84*	*32572.42*	*34641.27*
Cycle on-road	146 (3597)	43450.45	61034.96	1091.67	1588280.42	29495.70	41455.18	45445.73
Cycle footpath	52 (1044)	30235.72	38518.22	4320	490908.33	19440.83	27896.51	32574.93
Cycle off-road	157 (3711)	25014.02	32191.09	185.07	1149812	22155	23977.98	26050.07

Table 34 ANOVA results for cyclist UFP exposure in Christchurch

Variable	Effect (F/R)	df Effect	MS Effect	df Error	MS Error	F	p
Avg Wind Speed	***Fixed**	**1**	**4.42E+09**	**217.74**	**588963793**	**7.51**	**0.007**
Avg Temp	*Fixed	1	2.04E+09	104.47	684788363	2.98	0.087
{1}Direction	*Fixed	2	1.24E+09	81.27	728362756	1.70	0.190
{2}Mode	***Fixed**	**2**	**1.27E+10**	**314.94**	**525005877**	**24.25**	**<0.001**
{3}Wind Strength	***Fixed**	**1**	**1.23E+10**	**89.44**	**710723481**	**17.27**	**<0.001**
{4}Wind Influence	**Fixed**	**1**	**5.16E+09**	**242.39**	**575978592**	**8.95**	**0.003**
{5}Leg	**Random**	**23**	**1.28E+09**	**292.00**	**500494517**	**2.55**	**<0.001**

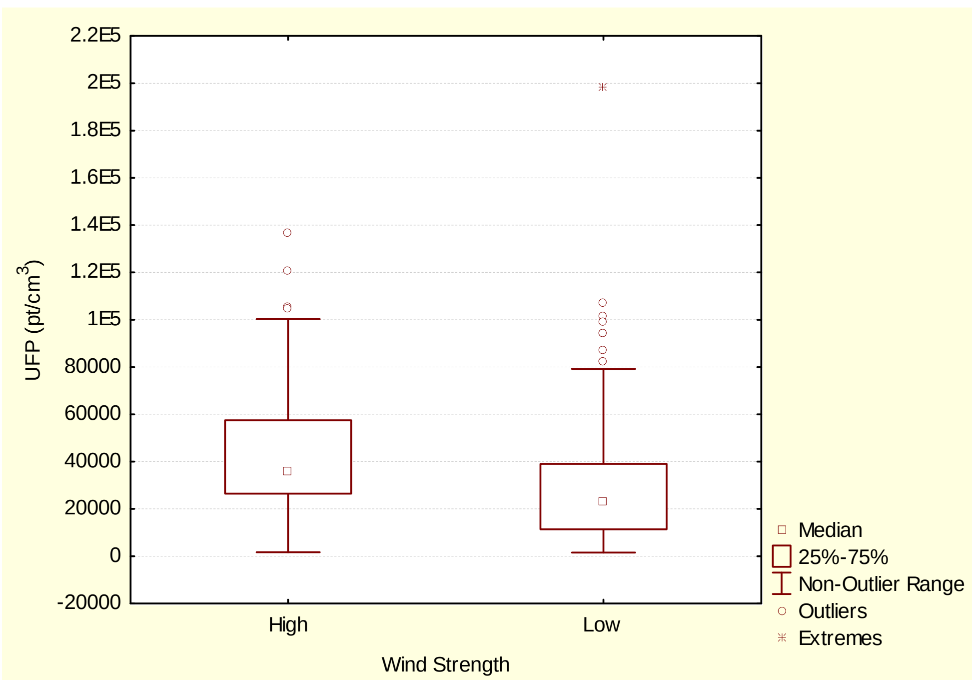

Figure 40 Mean UFP exposure values grouped by Wind Strength

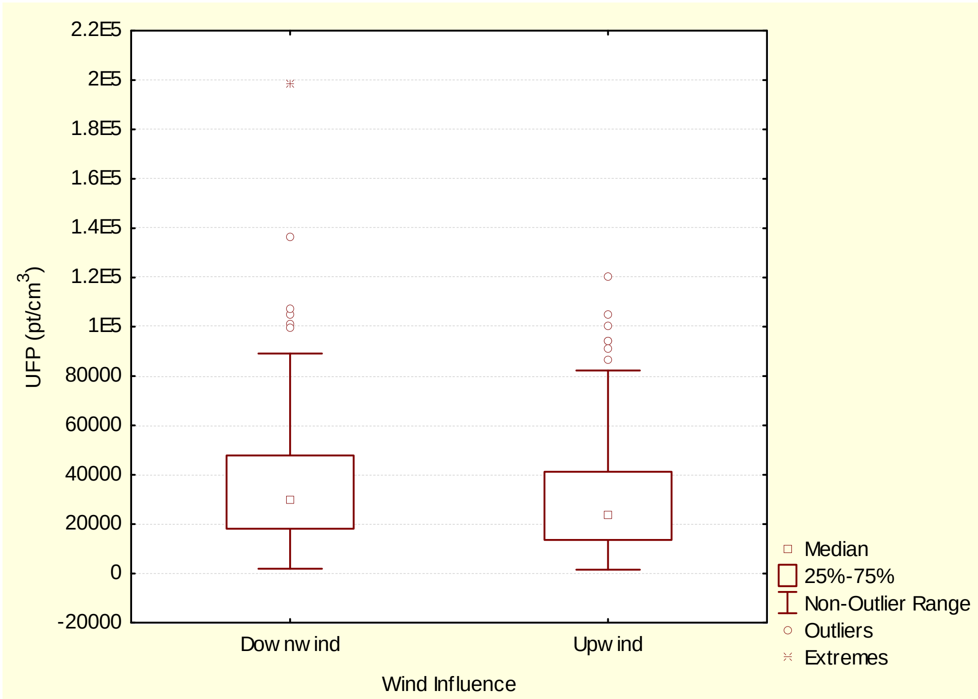

Figure 41 Mean UFP exposure values grouped by Wind Influence

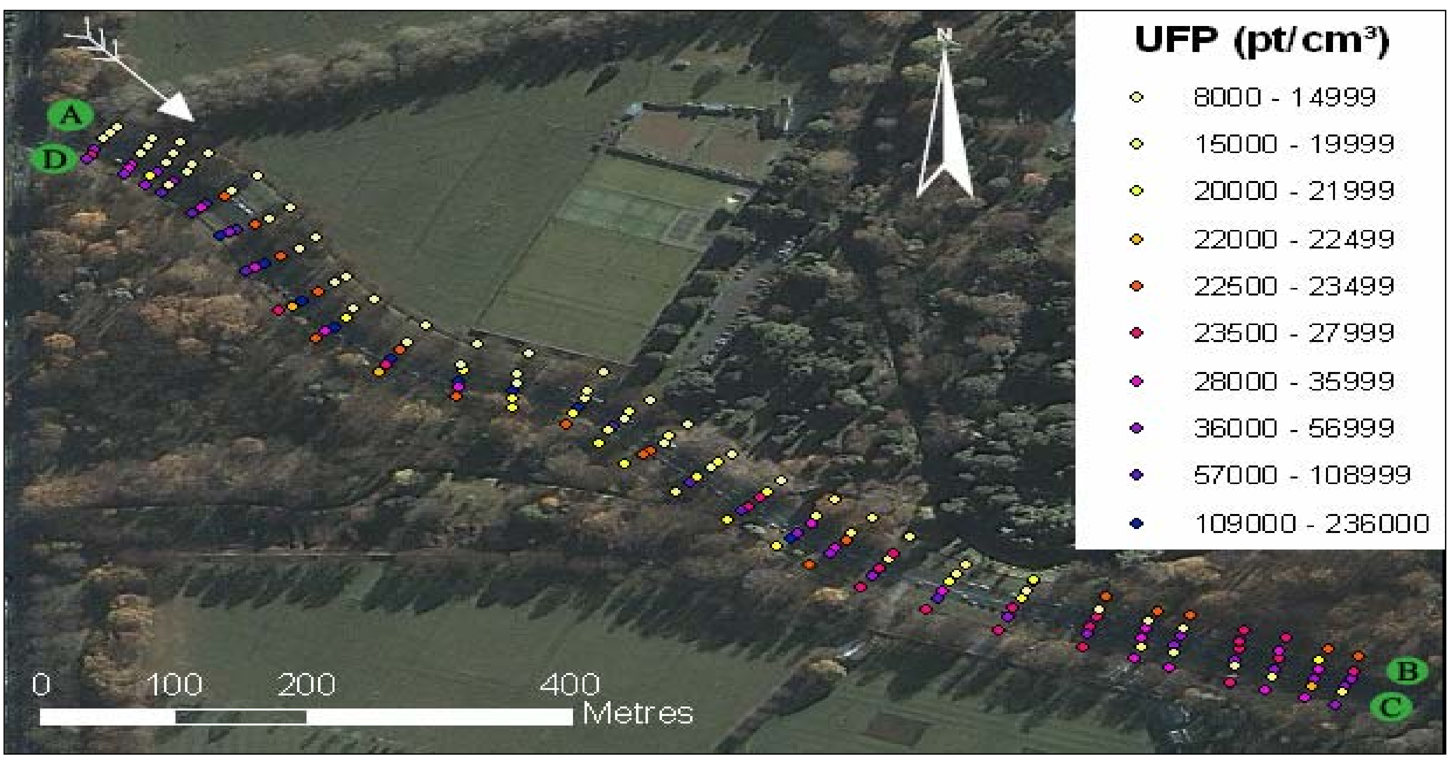

Figure 42 Cyclists' comparative UFP exposure at different distances from traffic: a sample section showing two legs (west-east and back). Wind speed = 1.6 m/s. Wind direction indicated on figure.

The lack of association between UFPs and temperature was due to the limited sampling period of ~2 hours across 3 days and the fact that sampling took place during warm afternoons. Temperature only fluctuated by ~2°C, compared to an approximate 16°C fluctuation throughout the inter-modal campaign, for which there was a significant inverse relationship. Such a relationship was also observed by Thai et al. (2008) and Kaur et al. (2009).

The positive association found for wind speed can possibly be explained by the decreased opportunity for particles to diffuse closer to their source. At such a close distance to emissions sources, the positive correlation is not surprising (Figure 40). When measuring concentrations across greater distances, a negative association can be expected due to increased time for coagulation and dispersion. This was confirmed for the inter-modal sampling (see section 4.2.3) and in prior research (Vinzents et al. 2005). The calculated time taken for a 0.1 μm particle to settle in still air at a distance of 2 metres from its source is 14 hours (Phalen 2002). At elevated wind speeds, while most particles not lost to larger size fractions settle within minutes, the remainder may travel several kilometres. This is evident from the current study when considering the sharp drop-off in concentrations when moving away from the road. Concentrations decreased by 30% at a 7 metre distance, 42% at a 19 metre distance and 62% in the middle of the park, approximately 700 metres away (see Table 35). Results are further supported by the inter-modal on-road-off-road comparisons, where exposure was 53% lower, despite being exposed to peak event situations behind vehicles and at crossings. Furthermore, unlike all other pollutants in the study, the influence of wind direction was highly significant for ultrafines (p=0.003). This is likely owing to their extremely small size and aerodynamic properties, while heavier particle concentrations are less affected by wind direction due to constant formation processes and the re-suspension of road dust. Additionally, average wind speeds were quite low at 2.14 m/s or less, with the exception of one day (4.16 m/s). If sampling took place across a greater number of days under higher wind speed conditions, the influence may have been non-significant.

Figure 43 shows the frequency distribution for all collected UFP data. The size of the on-road and footpath cyclist distributions is affected by a reduced number of observations, yet their shape is very similar. The bulk of observations occur at the mid-range level, tapering down towards the high end of the range. This pattern is especially pronounced for the on-road cyclist. This shows fairly even exposure to roadside levels and the high susceptibility to peak events such as queued traffic and heavy diesel vehicles passing, although exposure is clearly lessened at the footpath. The off-road distribution is likely to represent two distinct source contributions. The greatest number of observations either sit at extremely low levels or within the mid-range zone. It is probable that the lower end of the spectrum is representative of background levels attributable to city traffic from all surrounding roads (as observed in the middle of the park), while the mid-upper range reflects the influence of traffic sources within the direct vicinity (Riccarton Avenue).

(a)

(b)

(c)

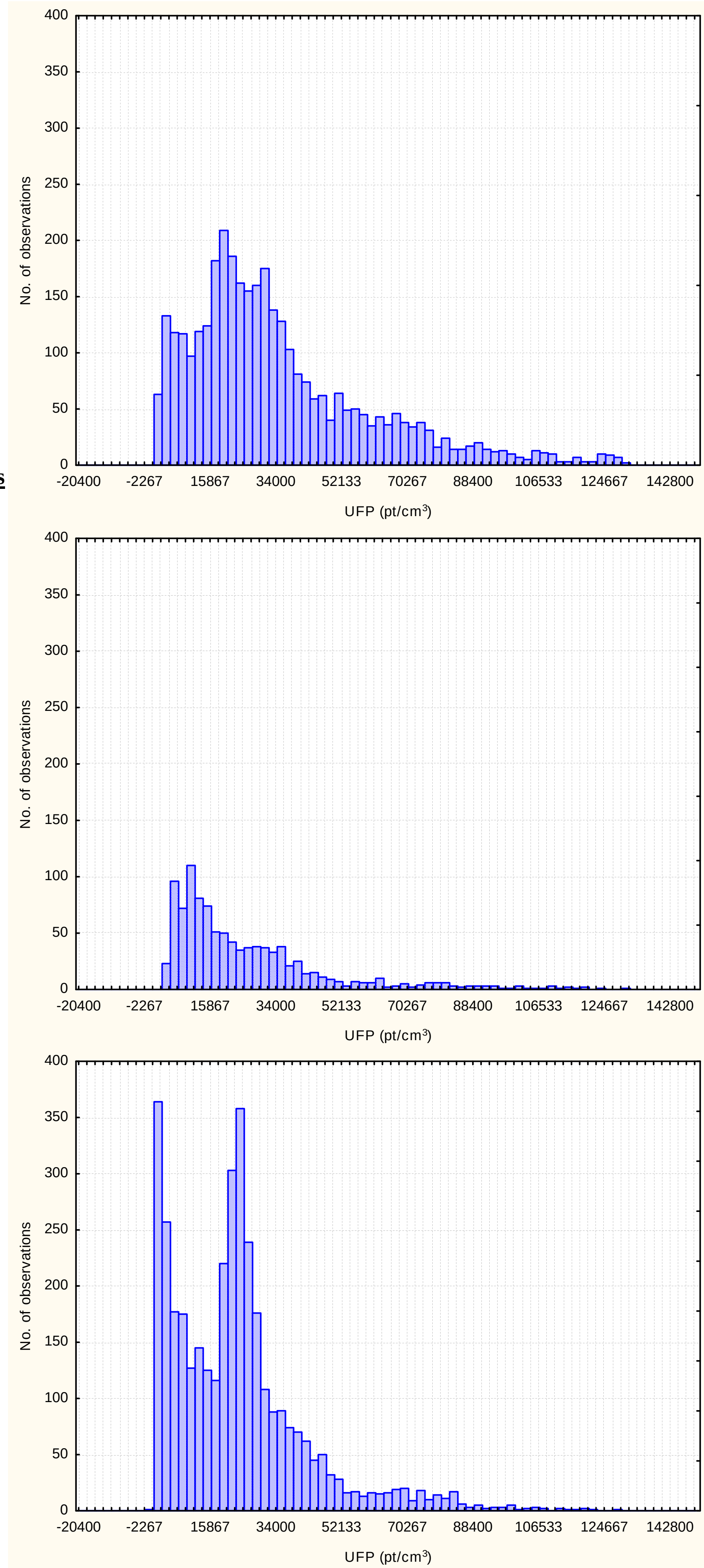

Figure 43 Freq. distribution for UFPs
(a) On-road cylist
(b) Footpath cyclist
(c) Off-road cyclist

5.3.4 Additional park scenario

This extra scenario was completed for one sampling afternoon (total of 16 legs) in order to gain an idea of how concentrations in the middle of the park compared to those closer to the road. Data was collected independently of other sampling days, to supplement the main microscale variance results for further comparison. While only the descriptive results are presented for discussion (Table 35), ANOVA results were significant across modes for all pollutants ($p<0.001$). Due to the very small sample size, results cannot be treated with absolute confidence and this part of the study is intended for indicative purposes only.

No difference for mean CO levels was found between the park and off-road cyclists, yet the findings presented in section 5.3.1 show a 54% reduction from on-road to off-road and a 41% reduction for on-road to footpath. This potentially highlights the degree to which separation becomes beneficial. It appears that at distance of 7 and 19 metres away, CO concentrations become highly dilute, reaching the lowest levels for some distance. The spatial extent of CO is thought to be somewhere in the order of 100-400 metres (Zhou & Levy 2007). The results agree well, considering the influence of traffic approximately 300 metres to the north [for the park cyclist].

PM_{10} and $PM_{2.5}$ show a substantial decrease with distance from traffic, with levels lowest in the park centre. $PM_{1.0}$ appears to show the opposite trend for this dataset, which disagrees with findings in section 5.3.2. Wind direction was southwest, with an average speed of 1.68 m/s. No immediately obvious explanation exists.

UFP concentrations were 39% lower off-road than on road, in very close agreement with the 42% reduction found in section 5.3.3. Between off-road and park, concentrations decreased a further 38%.

In addition to particle loss over distance, it is likely that the flat parkland topography played a role in reducing concentrations. A large golf course is situated at the north top of Hagley Park and there are far fewer trees in this area. Large open areas reduce wind friction allowing for greater air flow and heightened dispersion.

Table 35 Summary statistics for additional park scenario

Mode	Pollutant	*N* Legs (samples)	Mean	Std. Dev.	Min	Max	Median	-95% Cnf.Lmt	+95% Cnf.Lmt
Cycle On-Road	CO	Data lost							
	PM_{10}	16 (946)	24.69	13.47	9.41	105.3	20.73	23.83	25.55
	$PM_{2.5}$		10.70	5.58	3.23	37.08	10.75	10.34	11.06
	$PM_{1.0}$		2.51	2.01	0.30	6.30	1.50	2.38	2.64
	UFP	16 (933)	39633.22	71938.71	5210.06	1588280.42	27487.31	35011.17	44255.26
Cycle Off-Road	CO	16 (949)	0.44	0.42	0.07	3.13	0.29	0.41	0.46
	PM_{10}	16 (940)	17.04	13.02	4.00	98.80	14.20	16.20	17.87
	$PM_{2.5}$		5.18	3.18	1.20	17.8	3.70	4.98	5.38
	$PM_{1.0}$		3.06	2.29	0.40	10.3	2.20	2.91	3.20
	UFP	16 (773)	24108.58	10298.32	4840.00	97638.33	22311.67	23381.46	24835.70
Cycle Park	CO	16 (949)	0.44	0.34	0.07	2.53	0.35	0.42	0.46
	PM_{10}	16 (913)	8.37	4.69	1.90	28.7	7.50	8.06	8.67
	$PM_{2.5}$		3.82	2.28	1.00	8.60	2.80	3.67	3.97
	$PM_{1.0}$		6.50	4.49	1.52	26.0	6.33	6.21	6.79
	UFP	16 (936)	14726.68	5735.80	2898.33	67271.67	15114.17	14358.75	15094.62

5.4 Auckland: Effect of proximity to traffic

5.4.1 Carbon monoxide

Mean CO results ranked the off-road cyclist as the least exposed; 12% lower than the footpath and 36% lower than the road (Table 36). The footpath cyclist observed a 27% reduction from the on-road position. Maximum peaks were picked up on the road and minimum values on the boardwalk, which also had the lowest minimum value. Results were significant across modes ($F_{2,210.03}$=36.59, p<0.001). Both average wind speed (p=0.005) and temperature (p<0.001) were positively correlated with CO (Table 37). The positive association with temperature is most likely related to the traffic influence. As sampling had to take place during the weekend, it was highly dependant on traffic volume. During the first day of sampling, it had been raining and temperatures were cool. As the weather improved, there were noticeably more vehicles during the last two sampling runs. Wind strength was excluded from the Auckland ANOVA models as average wind speed was 2.52 m/s or higher for all sampling days. The influence of wind in relation to traffic was also excluded as only one side of the road could be used and wind direction was always west or southwest.

Table 36 Summary statistics for cyclist CO exposure in Auckland

Mode of transport	*N* Legs (samples)	Mean (+/-0.2 ppm)	Std. Dev.	Min	Max	Median	-95% Cnf.Lmt	+95% Cnf.Lmt
All samples	234 (6889)	0.59	0.25	0.17	3.58	0.58	0.59	0.60
Cycle on-road	78 (2295)	0.75	0.30	0.39	3.58	0.68	0.73	0.76
Cycle footpath	78 (2297)	0.55	0.17	0.23	3.14	0.55	0.54	0.55
Cycle off-road	78 (2297)	0.48	0.19	0.17	1.21	.52	0.48	0.49

Table 37 ANOVA results for cyclist CO exposure in Auckland

Variable	Effect (F/R)	df Effect	MS Effect	df Error	MS Error	F	p
Avg Wind Speed	***Fixed**	**1**	**0.25**	**227.88**	**0.03**	**8.17**	**0.005**
Avg Temp	***Fixed**	**1**	**1.29**	**86.58**	**0.02**	**53.46**	**<0.001**
{1}Direction	*Fixed	1	0.04	14.58	0.02	2.33	0.148
{2}Mode	**Fixed**	**2**	**1.17**	**210.03**	**0.03**	**36.59**	**<0.001**
{3}Leg	Random	36	0.02	192.00	0.03	0.63	0.950

5.4.2 PM_{10}, $PM_{2.5}$ and $PM_{1.0}$

Mean results for the off-road mode showed a significant decrease for $PM_{2.5}$ and $PM_{1.0}$; 18%, 7% (from footpath) and 25%, 28% (from road). PM_{10} showed an opposite trend; highest at the boardwalk and decreasing at the footpath (8%) and road (19%). $PM_{2.5}$ and $PM_{1.0}$ also decreased between the road and footpath by 12% and 19%, respectively (Table 38). Results were significant across modes for all fractions (Table 39): $F_{2,221.87}=43.83$, $p<0.001$ (PM_{10}), $F_{2,211.15}=169.62$, $p<0.001$ ($PM_{2.5}$) and $F_{2,217.86}=76.61$, $p<0.001$ ($PM_{1.0}$). All PM was negatively correlated with average wind speed ($p<0.001$) and positively correlated with average temperature: $p=0.020$ (PM_{10}), $p=0.007$ ($PM_{2.5}$) and $p=0.005$ ($PM_{1.0}$). Direction was also significant: $p=0.040$ (PM_{10}) and $p=0.008$ ($PM_{2.5}$), with higher mean values observed for the east-west transverse.

Table 38 Summary statistics for cyclist PM exposure in Auckland

Mode of transport		***N* Legs (samples)**	**Artithmetic Mean** (μg/m³)	**Std. Dev.**	**Min**	**Max**	**Median**	**-95% Cnf.Lmt**	**+95% Cnf.Lmt**
All samples	*PM_{10}*	*234 (6881)*	*15.89*	*3.87*	*7.49*	*27.14*	*14.89*	*15.80*	*15.99*
	$PM_{2.5}$		*9.33*	*4.24*	*2.77*	*27.17*	*8.24*	*9.23*	*9.43*
	$PM_{1.0}$		*4.58*	*2.21*	*1.40*	*26.30*	*4.79*	*4.53*	*4.64*
Cycle on-road	PM_{10}	78 (2293)	14.53	3.60	7.49	25.75	13.54	14.38	14.67
	$PM_{2.5}$		10.74	4.66	4.06	27.17	11.17	10.55	10.93
	$PM_{1.0}$		5.36	2.49	2.23	26.30	5.30	5.26	5.46
Cycle footpath	PM_{10}	78 (2292)	15.91	3.93	10.11	27.14	14.01	15.75	16.07
	$PM_{2.5}$		9.50	4.07	3.13	17.95	10.00	9.33	9.66
	$PM_{1.0}$		4.35	2.04	1.56	12.77	4.73	4.27	4.43
Cycle off-road	PM_{10}	78 (2291)	17.25	3.59	8.80	26.92	17.20	17.10	17.40
	$PM_{2.5}$		7.75	3.35	2.77	14.67	7.46	7.61	7.88
	$PM_{1.0}$		4.04	1.84	1.40	6.98	4.79	3.96	4.11

Table 39 ANOVA results for cyclist PM exposure in Auckland

PM_{10}							
Variable	**Effect (F/R)**	**df Effect**	**MS Effect**	**df Error**	**MS Error**	**F**	**p**
Avg Wind Speed	***Fixed**	**1**	**1705**	**211.51**	**3.61**	**471.90**	**<0.001**
Avg Temp	***Fixed**	**1**	**24**	**60.01**	**4.06**	**5.96**	**0.018**
{1}Direction	***Fixed**	**1**	**27**	**24.16**	**4.55**	**6.01**	**0.022**
{2}Mode	**Fixed**	**2**	**153**	**221.87**	**3.49**	**43.83**	**<0.001**
{4}Leg	Random	36	4	192.00	3.42	1.26	0.167
$PM_{2.5}$							
Avg Wind Speed	***Fixed**	**1**	**3223.94**	**228.00**	**0.98**	**3304.88**	**<0.001**
Avg Temp	***Fixed**	**1**	**6.46**	**82.80**	**0.80**	**8.04**	**0.006**
{1}Direction	***Fixed**	**1**	**6.81**	**15.78**	**0.61**	**11.12**	**0.004**
{2}Mode	**Fixed**	**2**	**173.49**	**211.15**	**1.02**	**169.62**	**<0.001**
{4}Leg	Random	36	0.71	192.00	1.05	0.68	0.918
$PM_{1.0}$							
Avg Wind Speed	***Fixed**	**1**	**720.44**	**221.81**	**0.45**	**1587.32**	**<0.001**
Avg Temp	***Fixed**	**1**	**3.87**	**66.61**	**0.45**	**8.52**	**0.005**
{1}Direction	*Fixed	1	0.90	21.51	0.46	1.99	0.173
{2}Mode	**Fixed**	**2**	**34.76**	**217.86**	**0.45**	**76.61**	**<0.001**
{4}Leg	Random	36	0.45	192.00	0.45	1.00	0.473

The trend observed for PM_{10}, being higher on the boardwalk and lower at the road, is the complete reverse of findings for Christchurch, although the disparities between modes were much lower. This is likely due to the influence of sea spray and the far lower traffic volume, reflected in the CO results, with the on-road result almost twice as high in Christchurch. Sea spray is a significant contributor of PM_{10} in both cities; 23% in Christchurch and 22% in Auckland (Senaratne et al. 2005). Hagley Park is located ~6 kilometres from the coast, where PM_{10} originating from different sources is well mixed. Secondly, the associated short settling time would render sea spray an unlikely contributor this far inland. Conversely, it is most probably the primary source in an area like St Heliers Bay during autumn.

The inverse relationship for wind speed and all PM is consistent with results for Christchurch, along with the positive correlation with temperature for PM_{10}. For Auckland, increased vehicle movements may have had an influence, as with the increase in CO. However, it may be better explained by coincidental changes in ocean swells and meteorological conditions, thereby increasing the influence of sea spray. An explanation for Christchurch is less apparent.

5.5 Combined results: Effect of proximity to traffic

5.5.1 Carbon monoxide

Combined CO results showed that the off-road cyclist was 32% and 48% less exposed than the footpath and on-road cyclists, with the footpath exposure 23% lower than on the road ($F_{2,648.12}=52.32$, $p<0.001$). The max value was 12.81 ppm, sampled in Christchurch (Table 40). There was an overall negative correlation between average wind speed and CO ($p<0.001$). Grouped into fixed wind speed categories, average concentrations were greater at lower wind speeds with higher extremes (Figure 44). ANOVA results are displayed in Table 41.

Table 40 Combined summary statistics for cyclist CO exposure

Mode of transport	*N* Legs (samples)	Mean (+/-0.2 ppm)	Std. Dev.	Min	Max	Median	-95% Cnf.Lmt	+95% Cnf.Lmt
All samples	693 (16214)	0.76	0.59	0.05	12.81	0.62	0.75	0.77
Cycle on-road	231 (5401)	1.00	0.64	0.05	12.8	0.80	0.98	1.02
Cycle footpath	231 (5398)	0.77	0.64	0.05	7.14	0.61	0.75	0.79
Cycle off-road	231 (5411)	0.52	0.36	0.05	3.66	0.47	0.51	0.53

Table 41 Combined ANOVA results for cyclist CO exposure

Variable	Effect (F/R)	df Effect	MS Effect	df Error	MS Error	F	p
Avg Wind Speed	***Fixed**	**1**	**24.16**	**646.12**	**0.31**	**76.80**	**<0.001**
Avg Temp	*Fixed	1	0.58	673.42	0.31	1.88	0.171
{1}Direction	*Fixed	2	0.01	682.11	0.31	0.03	0.971
{2}Mode	***Fixed**	**2**	**15.67**	**648.12**	**0.30**	**52.32**	**<0.001**
{3}Wind Strength	***Fixed**	**1**	**4.66**	**684.00**	**0.31**	**15.18**	**<0.001**
{4}Wind Influence	Fixed	1	0.11	451.39	0.33	0.32	0.573
{5}Leg	**Random**	**37**	**0.54**	**647.00**	**0.30**	**1.80**	**0.003**

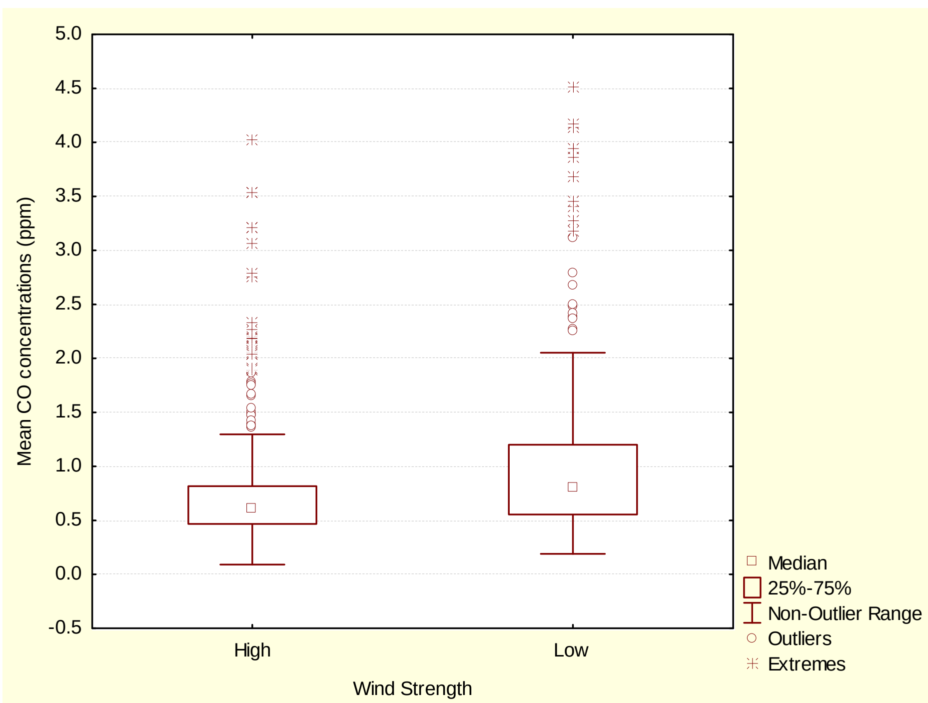

Figure 44 Combined CO exposure grouped by Wind Strength

5.5.2 PM_{10}, $PM_{2.5}$ and $PM_{1.0}$

Overall, the off-road cyclist was ranked lowest for all PM exposure apart from PM_{10} where the footpath mode was slightly lower. The greatest differences occurred between the off-road and on-road cyclists, with an 8% (PM_{10}), 24% ($PM_{2.5}$) and 29% ($PM_{1.0}$) reduction 18-19 metres from traffic (Table #). 5-7 metres from traffic, the footpath mode observed overall reductions of 9% (PM_{10}) and 13% ($PM_{2.5}$/$PM_{1.0}$). Results were significant across modes for $PM_{2.5}$ ($F_{2,570.86}$=7.72, p<0.001) and $PM_{1.0}$ ($F_{2,570.88}$=5.16, p<0.001) only. Wind strength and wind influence were also significant for $PM_{2.5}$ and $PM_{1.0}$ (Figure 45): p=0.005 and p<0.001. The median values for wind influence were heavily affected by the upwind position for all Auckland sampling, where the influence of sea spray was dominant.

Table 42 Combined summary statistics for cyclist PM exposure

Mode of transport		***N* Legs (samples)**	**Artithmetic Mean** (µg/m³)	**Std. Dev.**	**Min**	**Max**	**Median**	**-95% Cnf.Lmt**	**+95% Cnf.Lmt**
All samples	*PM_{10}*	*616 (16434)*	18.96	9.45	4.00	197.24	16.80	18.81	19.10
	$PM_{2.5}$		9.70	4.40	1.20	40.75	9.10	9.63	9.77
	$PM_{1.0}$		4.81	2.86	0.40	37.39	4.50	4.76	4.85
Cycle on-road	PM_{10}	231 (6316)	20.02	9.60	6.90	105.3	17.72	19.78	20.25
	$PM_{2.5}$		10.96	4.77	3.23	40.8	11.27	10.84	11.07
	$PM_{1.0}$		5.52	3.35	1.40	37.4	4.70	5.44	5.60
Cycle footpath	PM_{10}	223 (5205)	18.23	10.34	7.24	197.24	15.20	17.94	18.51
	$PM_{2.5}$		9.50	3.56	3.04	33.80	9.25	9.40	9.60
	$PM_{1.0}$		4.79	2.60	1.50	30.50	4.70	4.72	4.86
Cycle off-road	PM_{10}	162 (2291)	18.38	8.05	4.00	98.80	17.15	18.15	18.60
	$PM_{2.5}$		8.30	4.26	1.20	27.42	7.40	8.18	8.42
	$PM_{1.0}$		3.91	2.09	0.40	18.00	4.30	3.85	3.97

Table 43 Combined ANOVA results for cyclist PM exposure

Variable	Effect (F/R)	df Effect	MS Effect	df Error	MS Error	F	p
PM_{10}							
Avg Wind Speed	***Fixed**	**1**	**3665.95**	**606.93**	**47.37**	**77.39**	**<0.001**
Avg Temp	***Fixed**	**1**	**1618.89**	**603.67**	**47.41**	**34.15**	**<0.001**
{1}Direction	*Fixed	2	97.56	588.18	47.45	2.06	0.129
{2}Mode	*Fixed	2	27.68	572.29	47.48	0.58	0.559
{3}Wind Strength	***Fixed**	**1**	**2844.86**	**591.36**	**47.45**	**59.96**	**<0.001**
{4}Wind Influence	Fixed	1	96.16	573.27	47.27	2.03	0.154
{5}Leg	Random	37	45.69	570.00	47.49	0.96	0.536
$PM_{2.5}$							
Avg Wind Speed	***Fixed**	**1**	**649.81**	**593.24**	**10.94**	**59.37**	**<0.001**
Avg Temp	***Fixed**	**1**	**1776.61**	**586.68**	**11.10**	**160.06**	**<0.001**
{1}Direction	*Fixed	2	4.67	577.41	11.29	0.41	0.661
{2}Mode	***Fixed**	**2**	**88.06**	**570.86**	**11.41**	**7.72**	**<0.001**
{3}Wind Strength	***Fixed**	**1**	**88.61**	**578.93**	**11.26**	**7.87**	**0.005**
{4}Wind Influence	**Fixed**	**1**	**298.61**	**605.12**	**10.52**	**28.38**	**<0.001**
{5}Leg	Random	37	4.07	570.00	11.43	0.36	1.000
$PM_{1.0}$							
Avg Wind Speed	***Fixed**	**1**	**327.67**	**593.74**	**4.22**	**77.59**	**<0.001**
Avg Temp	***Fixed**	**1**	**48.99**	**587.08**	**4.28**	**11.44**	**0.001**
{1}Direction	*Fixed	2	10.23	577.61	4.35	2.35	0.096
{2}Mode	***Fixed**	**2**	**22.69**	**570.88**	**4.40**	**5.16**	**0.006**
{3}Wind Strength	***Fixed**	**1**	**478.17**	**579.17**	**4.34**	**110.10**	**<0.001**
{4}Wind Influence	**Fixed**	**1**	**84.65**	**605.46**	**4.06**	**20.83**	**<0.001**
{5}Leg	Random	37	1.61	570.00	4.41	0.37	1.000

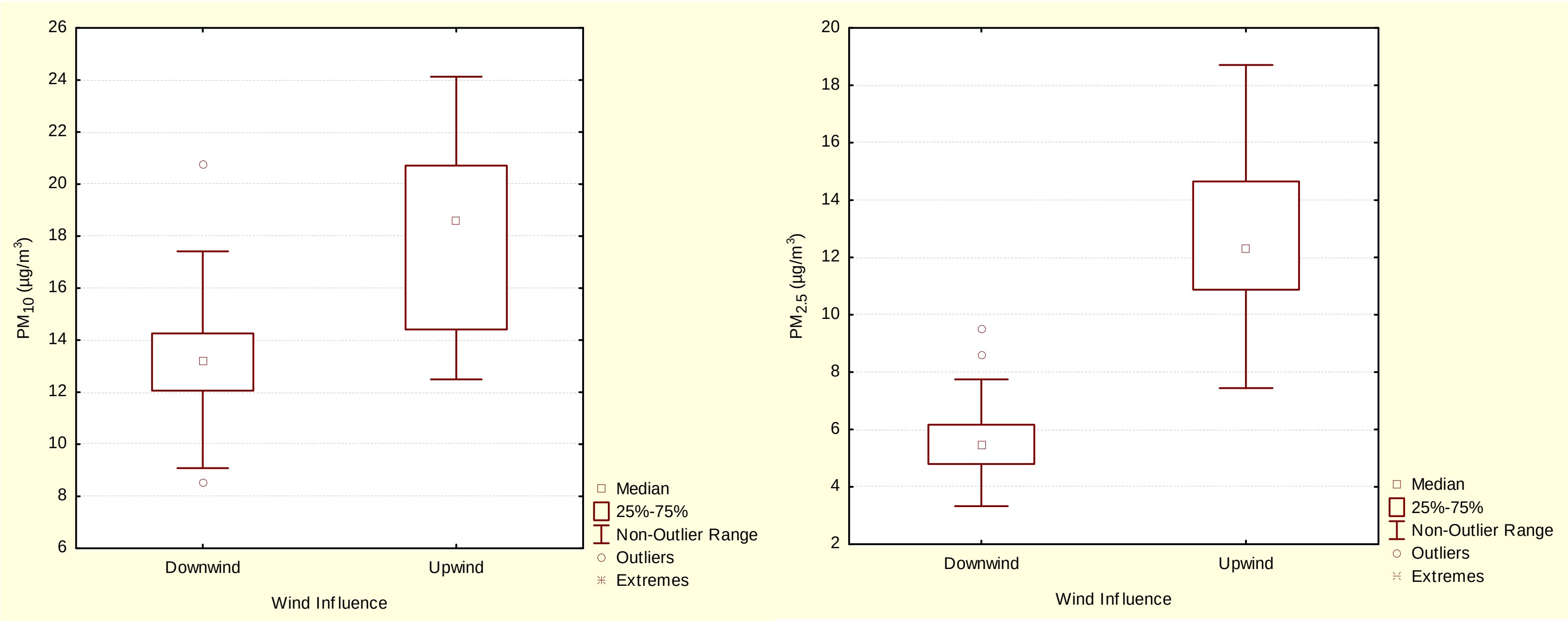

Figure 45 Mean $PM_{2.5}$ and $PM_{1.0}$ exposure grouped by Wind Influence

5.6 Cyclist results in the context of previous research

5.6.1 Effect of route choice

Research into the effect of cyclist route choice is relatively limited compared to inter-modal enquiry. Only three comparable studies exist for CO. The first, by Bevan et al. (1991) found CO levels to be higher on-road than within a parkland area by a factor of 13. These results are not completely comparable as sampling only took place within common parkland and did not replicate an actual commute, whereas the on-road element did. While the off-road element of the current study was largely situated within cycle ways and parkland areas, mean results were significantly elevated when coming into contact with traffic at backstreets or roadway crossings. Hence on-road exposure was only higher by a factor of 1.72. Even so, the 42% reduction found is very significant considering a cyclist commuter may be able to almost halve their exposure. Furthermore, the overall mean values of 10.5 ppm (Southampton; Bevan et al. 1991) and 1.12 in Christchurch highlight the relatively low-risk of commuting by bicycle in smaller cities where vehicle emissions are a lesser concern. Interestingly, commuting by bicycle in central London resulted in a mean exposure of only 1.1 ppm, exactly the same as in Christchurch (Kaur et al. 2005). The reduced exposure compared to Southampton may have been due to the use of bus lanes in the London study, providing more separation from the main stream of traffic. While this study did consider the use of backstreet travel, results were combined with other modes such as car and bus, although only pedestrians and cyclists could travel the full backstreet route. Combined mean exposure for the backstreet area was 0.6 ppm compared with 1.3 ppm on the main roads (Kaur et al. 2005). These findings are almost identical to those for Christchurch cyclists – 0.65 ppm and 1.12 ppm, respectively. A third study, utilising street pollution modeling based on the city of Copenhagen, found a 23 – 28% reduction for a backstreet route, compared to a shorter, more direct route (Hertel et al. 2008). This estimate is well below the findings of the current study.

For PM_{10}, the same modeling study simulated an 11 – 12% reduction for backstreet route concentrations, but found accumulative exposure to be 2 – 3% higher due to the increased travel time (Hertel et al. 2008). The current study reported a 6% increase in concentrations for the off-road cyclist. Results are expected to differ due to differences in traffic concentrations and background sources, in addition to the methodologies. Both Thai et al. (2008) and Berghmans et al. (2009) reported significant elevations in PM_{10} levels for cyclists near constructions sites. While neither study separated mean exposure by area type, the overall range of mean values largely agreed with those of the current study. For Vancouver, Thai et al. (2008) found the range to be within 21 – 74 $\mu g/m^3$. For Mol, Belgium, Berghmans et al. (2009) found the range to be within 34 – 102 $\mu g/m^3$. In Christchurch, the range for both on-road and off-road combined was within 8 – 93 $\mu g/m^3$. Berghmans et al. (2009) sampled along an additional cycle track area on one day, finding a mean exposure of 54 $\mu g/m^3$. This was only lower than all data combined by 13%.

$PM_{2.5}$ was 13 – 14% lower off-road in the simulation model but accumulative exposure was virtually the same (Hertel et al. 2008). The range of exposure for the current study (5 – 41 µg/m³) closely agreed with that of Vancouver (7 – 33 µg/m³), while Mol findings were much higher (12 – 75 µg/m³). Mean exposure for the cycle track in Mol was 32 µg/m³; 18% lower than the overall combined mean (Berghmans et al. 2009). The reduction found off-road for Christchurch was only 6%.

The study in Mol, Belgium is the only other piece of research to consider $PM_{1.0}$ exposure for cyclists. Mean exposure ranged from 7 – 77 µg/m³ compared to 1 – 27 µg/m³ in the current study. The cycle track results for Mol were 13% lower than for all data combined, while the off-road cyclist experienced a 31% reduction in Christchurch.

Ultrafine results for Mol ranged between 10851 and 30576, not far below those for Vancouver (18830 – 57692 pt/cm³). Mol cyclist track exposure was 2% higher than overall mean exposure for all data (21226 pt/cm³). Mean data for Christchurch ranged between 2387 and 160520 pt/cm³, while the on-road-off-road reduction was 53%. The reduced disparity between the results for Mol is likely to do with the limited sample size, with only one sample for the cycle track and seven main samples. The mean result for the on-road cyclist in Christchurch (49842 pt/cm³) was 47% lower than that recorded in London (93968 pt/cm³) by Kaur et al. (2005). The London main street (101142 pt/cm³), versus backstreet (71628 pt/cm³) sampling resulted in an overall difference of 41% (for all modes combined). This difference highlights the effect of taking backstreet routes, regardless of mode. The overall differences found for London are not far below those found for Christchurch cyclists.

5.6.2 Effect of proximity to traffic

The results for the current study show substantial reductions for CO, UFPs and PM with distance from traffic, with the exception of PM_{10} at the seaside setting in Auckland. Comparative research addressing exposure differences at such small distances from traffic is extremely limited, with studies (fixed site) tending to look at downwind levels much further from roads (Hitchins et al. 2000). However, reductions in pedestrian exposure when moving away from the kerb have been noted for CO (Wright et al. 1975) and UFPs (Kaur et al. 2005b, Kaur et al. 2006). In central London, Kaur et al. (2005b) observed a 15% reduction for average UFP exposure from kerb to buildingside. No difference was found for CO levels and $PM_{2.5}$ decreased by only 1%. The results for Christchurch indicate an average reduction of 30% (UFPs), 22% (CO) and 8% ($PM_{2.5}$) at 7 metres from traffic. The differences between the two studies can primarily be explained by the higher traffic volume and street canyon environment in central London. Secondly, no indication of the width of the pavement was provided by the London study. Furthermore, the parkland setting of the Christchurch study would have provided a greater dispersive effect. The London results for UFPs are encouraging for New Zealand, proving that microscale deterioration occurs even within street canyons. Due to lower traffic densities and building heights, such reductions may not be limited to UFPs in smaller cities like Christchurch. A subsequent study by Kaur et al. (2006) in the same area, suggested that with careful avoidance of cigarette smokers and construction sites, UFP exposure can be lessened by 10-30%. In certain situations, the same degree of reduction may be able to be achieved in busy street canyons as occurs alongside parkland.

Only one other study provides actual exposure data for microscale variance. In Dublin, McNabola et al. (2008) explored the effect of a low-boundary wall (less than 1 metre high) between a pavement and a boardwalk. The pavement is approximately 2 metres from the traffic flow and the boardwalk, 3-4 metres away from traffic. $PM_{2.5}$ was found to decrease by a factor of 2.83 on the boardwalk, while for Christchurch, this figure was only 1.17 at 7 metres from traffic. This highlights the powerful dispersive effect of the wall compared to an open area. It was thought that parked cars may have some dispersive benefit, but this did not appear to be the case when looking at the pollutant traces and photographs using the Media Mapper software. It is likely the airflow traveled under and over the car (as well as through gaps between parked cars), reaching the cyclist on the other side.

5.7 Summary

The results of this section have addressed spatial variation of concentrations over relatively short distances, as well as variation within very small distances. The results show that CO and UFP levels decrease substantially with distance from traffic, exhibiting a heterogeneous distribution. While the three PM fractions are more uniform in concentration patterns, $PM_{2.5}$ and $PM_{1.0}$ also rapidly decreases with distance from traffic. PM_{10} appears to be more reflective of background concentrations and is less dependant on traffic sources. These findings have significant implications for behavioural choices and city planning. Taking an off-road route can reduce exposure by 31-53% for traffic-dependant pollutants (UFPs, CO and $PM_{1.0}$), while differences in background exposure are minor, but can be slightly higher off-road. These primary traffic-generated pollutants appear to either dissipate or be lost to other processes such as coagulation, over extremely small distances. At an average distance of only 7 and 19 metres, CO dropped 41 & 54%, $PM_{1.0}$ 26 & 32% and UFPs 30 & 42%, respectively. This shows a very steep gradient and no difference was observed for CO at 19 metres or 700 metres north. However, UFPs, due to their interactive nature, did decrease a further 38% to levels likely representative of background. Correlations with temperature and wind speed largely agreed with findings of international studies, although some unusual results occurred for Auckland due to the seaside location and lack of traffic as a result of bad weather.

The Auckland data was less useful due to the loss of the UFP data, but did help confirm overall microscale reductions for CO, $PM_{2.5}$ and $PM_{1.0}$ across two distinct geographical settings.

Findings for the effect of route choice were in agreement with those for London, illustrating the benefits of taking backstreet routes in cities of various sizes (Kaur et al. 2005). Microscale variance somewhat agreed with previous work on pedestrian pavement exposure. Although the current study made use of open areas and previous work was situated within street canyons, UFP exposure reduction away from the kerb may sometimes be as high as within the Christchurch study. The fact a reduction of this magnitude (comparable to that in a parkland area) may be possible in large cities has important implications for pedestrian and cyclist planning, including the placement of temporary paths and detours that avoid construction activity.

Chapter Six: Conclusions

6.1 Inter-modal findings

The inter-modal section of this study successfully addressed the issue of which transport modes encounter the highest and lowest overall mean concentrations of key pollutants associated with transport emissions. While it failed to include motorcyclist and pedestrian exposure; or adequately address train commuter exposure, it did reasonably replicate the general study design of the majority of similar international research. Furthermore, significant results were produced from two distinct cities, providing the first mobile exposure literature for New Zealand.

Final comparative exposure results largely agreed with those of international studies. The cyclist (on-road) fared relatively well against other modes. Carbon monoxide exposure for the car was around 2.6 and 2.3 times higher than both the cyclist and bus, in Christchurch and Auckland, respectively. Train exposure was lower still - by a factor of 4.3, compared to car. The cyclist was also the least exposed for ultrafine particles – by a factor of 1.6, compared to car and bus concentrations which were roughly the same. There was not a lot of variation between modes for mean $PM_{1.0\text{-}10}$ exposure. PM_{10} was highest for the cyclist in Auckland, but only 4% greater than bus, whereas this difference was 23% in Christchurch, in favour of the cyclist. The bus mode also had the highest mean exposure for $PM_{2.5}$ and $PM_{1.0}$ in Auckland, while the cyclist was slightly lower than car. The bus was also highest in Christchurch, followed by the cyclist and then the car. The higher levels in the bus are likely due to diesel self-pollution and intake of outside air through open windows and the continual opening of doors. The car proved to be a more protective environment, filtering out larger particles through the ventilation system. Although the cyclist was exposed to higher PM than the other modes in Christchurch, modal differences across PM fractions only ranged from 6 – 26%. Comparatively, the cyclist was 36% and 61% less exposed than the car, for UFPs and CO, respectively.

The results of this study highlight a significant advantage for cyclists in regard to pollution exposure, as it is the ultrafine particles which are considered the most dangerous due to their penetrative ability and greater toxicity. Larger particles are less concerning, yet still pose serious health risks. This study has further shown cyclists may also be the least exposed to $PM_{2.5}$ and $PM_{1.0}$ in heavily trafficked large urban areas such as Auckland.

6.2 Cyclist findings

The results of the inter-modal study have demonstrated that a separation of just one or two metres from the traffic stream, coupled with the ability to move to the front of queued traffic, substantially reduces CO and UFP exposure compared to other modes.

The second part of this study investigated the effect of traveling as far away from traffic as possible, by means of backstreets and parkland. The results show off-road cyclists have reduced exposure in the order of 31% for CO and $PM_{1.0}$, and 53% for UFPs. Although exposure was 6% higher for both PM_{10} and $PM_{2.5}$, this is minor compared to the other reductions. Higher PM_{10} and $PM_{2.5}$ for the off-road cyclist represents the strong influence of background sources while high CO, $PM_{1.0}$ and UFPs levels on-road, highlight the impact of being in close proximity to traffic flows. Previous investigation into the effect of cyclist route choice is rather limited and has generally looked at activity-exposure profiles using one cyclist, rather than comparing real-time exposure along two distinct routes. This study has provided some of the first concrete evidence to suggest that taking a longer alternative route significantly lowers exposure, which is likely to be beneficial to one’s health over months or years of commuting. Furthermore, the study design allowed for a glimpse into patterns of spatial variation across distances of up to 3.3 km. All PM was found to be highly uniform, whereas CO and UFPs were rapidly dispersed or lost to other processes. Pollutant correlations with temperature and wind speed were also found to agree with the results of prior research.

The final section addressed a previously unexplored area of microscale exposure variance whilst traveling. The results showed a separation of only 5 – 7 m from the on-road position can greatly lower exposure of CO, UFP and $PM_{1.0}$; and to a lesser extent, PM_{10} and $PM_{2.5}$, in certain settings. At 15 – 19 m away, the reduction effect is more than doubled for all pollutants sampled. At approximately 700 m away, the effect wanes, where levels are more representative of background. This element of the study was also able to demonstrate the significance of cyclist position in relation to traffic and wind direction, with higher UFP concentrations experienced at the downwind side of the road. The influence was not significant for other pollutants due to their non-reactive nature and the fact larger particles are constantly being re-suspended by traffic movement. While a distance of 5 – 7 m may not render contrasts to the extent found in more open areas, past pedestrian enquiry within street canyons has shown reductions do occur, at least for CO and UFPs.

6.3 Implications for policy

The policy implications of the overall study results are very wide ranging. In the broader context, as many preceding inter-modal studies have concluded, traveling by private vehicle is not the optimal way to avoid high pollutant concentrations. Walking, taking a train, subway, tram or bicycle generally all prove much better than being amidst in-traffic sources. The implications here are obvious and policy-makers have been aware of the evidence for many years, especially overseas. Not only does investment in alternative and active transport infrastructure greatly improve population health, it prepares cities for uncertain futures in regard to energy availability, technology and international trade. Furthermore, it makes for a healthier and more active population, which makes economic sense in that improved psychological and physical health is positively correlated with increased productivity (Bloom et al. 2004).

The results of the cyclist components of this research provide an additional argument in campaigning for greater investment in cycle track/walkway infrastructure within local and national governmental bodies. While the safety advantages of having segregated paths are well researched and demonstrated overseas, the impact on pollution exposure is less understood. This study has shown that very small distances of separation from roadways can result in large exposure reductions. Secondly, the effect is greatly diminished between a separation of a few metres and a large distance of several hundred metres. Given the space limitations for long-distance segregations in built-up areas, often the only existing opportunity lies within small separations. Such an opportunity may seem futile, but is in fact extremely significant. The layout of Christchurch streets and roads is a prime example of this, easily allowing for segregated cycleways in wide streets and shared pathways within wide pavements. In countries such as The Netherlands and Germany, where this type of infrastructure is commonplace, active mode transport is extremely popular and on the rise (Pucher & Buehler 2008).

Various progressive studies outline the numerous benefits of implementing functional policies and investing in alternate mode infrastructure. A top example of this comes from Victoria, Canada, where Litman (2008) explains the logic behind various policy options that can render substantial benefits for resident populations, transport system efficiency and city attractiveness. The types of policies include: increased fuel taxes, smart growth, least-cost planning, pay-as-you-drive pricing, reduced employee parking, High Occupant Vehicle (HOV) priority, walking and cycling infrastructure improvements and more. Litman (2008) outlines how improved cyclist and pedestrian facilities fit into smart growth policy; active-mode oriented land use and building design, increased connectivity with shortcuts for non-motorised modes, improved paths and cycle lanes, and traffic calming, speed reductions and vehicle restrictions. Additionally, such improvements greatly support travel by public transport. If commuters are able to quickly and safely get to public transit stations (and securely store equipment), patronage will increase as people become less reliant on private vehicles. These 'Win-Win' strategies are examples of forward-planning that "provide multiple, economic, social and environmental benefits" and "are justified regardless of uncertainties about global warming or other environmental and social impacts" (Litman 2008, p. 1).

6.4 Limitations

One of the main limitations of this study is that it did not measure VOC or NOx exposure. Initially, NO_2 was to be included as one of the sampled pollutants but a lack of equipment prevented this from going ahead. However, as discussed in the literature review, the results of previous studies show that cyclist exposure to VOCs or NOx is comparatively low and is likely to be correlated with other primary pollutants such as CO. Secondly, the analysis did not include accumulative dose. The issue of accumulative dose based on breathing rates is rather complex and has received little attention up until recent years.

The use of the ultrafine diluter system proved rather problematic and requires continual co-location and data correction to maintain accurately recorded absolute concentrations. Unfortunately, the lack of attention to this necessity resulted in the loss of all UFP data for Auckland. Nonetheless, the inter-modal and cyclist data collected for Christchurch was sufficient to meet the objectives of the study.

The absolute PM data presented, obtained from two different Grimm spectrometer models, cannot be treated with absolute certainty due to the inability to characterise against instruments that meet Federal Reference Method requirements. However, meaningful ratios were able to be calculated from this data which were comparable to those found for other inter-modal studies. Furthermore, raw mean values seemed to be within the expected range for the associated mode for cities of this size (Christchurch and Auckland). It is probable that the error surrounding the lack of unequivocal characterisation is negligible.

The somewhat questionable nature of the PM data, coupled with the loss of Auckland UFP and PM train data, did not significantly reduce the quality of the study. An enormous amount of data was collected and carefully corrected and/or screened for erroneous occurrences. The final dataset was of a very high standard and able to provide a quality assessment of exposure for two different cities.

The final limitation was that this study did not compare mobile data to fixed site monitoring. This was mainly due to time restrictions and that many studies have already investigated relationships with fixed sites. Additionally, it was not a key objective of this enquiry, which principally focused on comparisons between modes and distance from traffic sources.

6.5 Future research

An interesting question apparent following a full literature review, is that of motorcyclist versus cyclist exposure, which has not yet been investigated. This would give a fully complete real-time representation of the effect of being just one or two metres away from the main traffic flow. Furthermore, it is possible that full-faced motorcycle helmets provide a protective area, limiting the ingress of particles into the helmet enclosure.

Future microscale cyclist research should include looking at the effect of proximity across a range of land uses, with a special focus on street canyons. This poses some difficulty, due to requiring the use of pedestrian paths. The pavement cyclist could be substituted by a pedestrian sampler, providing increased travel time is accounted for, yet this would not fully mirror the real-time method used in this study. This type of work could be supplemented by high resolution modeling techniques within various street settings. Such studies are limited, with researchers calling for high resolution modeling near emissions sources (Zhou & Levy 2007).

There is also a need for a standardised method of measuring accumulative dose and respiration rates. Further inter-modal and active mode research should aim to include a combination of exposure, accumulative dose, respiration and modeling techniques. Perhaps only then can fully accurate conclusions be made regarding different levels of intake between modes.

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