Determination of personal exposure to traffic pollution while travelling by different modes

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Simon Kingham
Woodroe Pattinson
Kreepa Shrestha
Department of Geography, University of Canterbury, Christchurch

Ian Longley
National Institute of Water and Atmospheric Research, Auckland

Jenny Salmond
School of the Environment, University of Auckland, Auckland

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NZ Transport Agency
Private Bag 6995, Wellington 6141, New Zealand
Telephone 64 4 894 5400; facsimile 64 4 894 6100
research@nzta.govt.nz
www.nzta.govt.nz

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1 Department of Geography, University of Canterbury, Christchurch
2 National Institute of Water and Atmospheric Research, Auckland
3 School of the Environment, University of Auckland, Auckland

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An important note for the reader

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Executive summary

Introduction

Most research suggests that car commuters are exposed to higher concentrations of air pollutants than those who walk, cycle or use public transport, although several more recent studies consider active modes of travel to be the most affected. This purpose of this project is to assess the comparative risk associated with exposure to traffic pollution when travelling via different transport modes in New Zealand cities. The research objectives were to:

- provide an accurate measure of personal pollution exposure by mode
- provide information for transport decision-making at personal and societal levels
- provide a stronger base for advocating a change in consumer behaviour.

Method

Concentrations of the key traffic-related pollutants (particulate matter (PM): PM_{10}, $PM_{2.5}$, PM_1; ultrafine particles (UFPs) and carbon monoxide (CO)) were simultaneously monitored on pre-defined routes in Auckland and Christchurch during the morning and evening commutes for people travelling by car, bus, on-road bike, train (Auckland only) and off-road bike (Christchurch only) from February to May 2009. Additional sampling took place on cycle routes on and parallel to roads at three consistent distances from traffic to assess the importance of proximity to traffic. The impact of transport mode on pollution exposure was assessed by calculating relative ratios between modes and comparing them to ambient levels at the time of day/day of week when the modal sampling took place. Changes in the levels of pollution were compared to global positioning system data to examine the impact of geography and other factors on pollution levels. Pollution levels were related to changes in weather conditions.

Results

The key results of this research are as follows:

- Car drivers are consistently exposed to the highest average levels of CO: 60% higher than cyclists, 40–100% higher than bus passengers and over 100% higher than train passengers.
- On-road cyclists are exposed to higher levels of CO (10%), PM_1 (25%) and UFPs (over 100%) than off-road cyclists. This could have significant policy implications for the location of cycle routes.
- Car drivers and bus passengers are exposed to higher average levels of UFP than cyclists. However, for very short acute exposures (a few seconds), on-road cyclists can be exposed to higher peaks.
- At some parts of their journeys, travellers are exposed to very high levels of pollution, often for short periods of time. This has potential health implications.
- Locating cycle paths just a short distance from roads can reduce pollution exposure significantly: for example, locating a cyclist 5–7m away can reduce exposure by 20–40%.
- One hour of commuting (ie 4% of the day) could contribute up to 20% of the total daily dose of CO and UFP.
- PM_{10} and $PM_{2.5}$ are inappropriate indicators of exposure to vehicle emissions.

Conclusion

Significant differences in pollution exposure are apparent for people travelling on different transport modes. For journeys deemed typical for urban New Zealand, the exposure of cyclists to traffic-related air pollution was significantly lower than that of car or bus users. This research also makes it clear that the exposure of cyclists can be substantially reduced further by providing cyclists with spatial separation from vehicle exhausts, and that substantial benefits can be achieved by relatively small degrees of separation (5–10m).

Abstract

This purpose of this project is to assess the comparative risk associated with exposure to traffic pollution when travelling via different transport modes in New Zealand cities. Concentrations of the key traffic-related pollutants (particulate matter: (PM): PM_{10}, $PM_{2.5}$, PM_1; ultrafine particles (UFPs) and carbon monoxide (CO)) were simultaneously monitored on pre-defined routes in Auckland and Christchurch during the morning and evening commute on people travelling by car, bus, on-road bike, train (Auckland only) and off-road bike (Christchurch only) from February to May 2009. The key results of this research are:

- Car drivers are consistently exposed to the highest average levels of CO.
- On-road cyclists are exposed to higher levels of CO, PM_1 and UFPs than off-road cyclists.
- Car drivers and bus passengers are exposed to higher average levels of UFP than cyclists.
- At some parts of their journeys, travellers are exposed to very high levels of pollution, often for short periods of time.
- Locating cycle paths just a short distance from roads can reduce pollution exposure significantly.
- One hour of commuting could contribute up to 20% of total daily CO and UFP.
- PM_{10} and $PM_{2.5}$ are inappropriate indicators of exposure to vehicle emissions.

1 Introduction

1.1 Urban pollution and the transport sector

Globally, transport contributes to 14% of all global greenhouse emissions, with 72% of that portion made up by vehicular road transport (International Energy Agency (IEA) 2006); in New Zealand, vehicular road transport is estimated to contribute 45% of carbon dioxide emissions and 12% of all greenhouse gas emissions (Ministry for the Environment 2010). Curtailing emissions from vehicles through use of sustainable city transport and alternative modes is now a pressing issue for many local and state governments. This will have implications for air quality. Reducing urban pollution is not only important in reducing 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 places, including London, New York and Sydney. The New Zealand situation is not far behind the trend, with over 500 pollutant-associated premature mortalities per annum for those aged over 30 years, a number similar to those killed by motor vehicle accidents (Fisher et al 2007; Kingham et al 2008; Kuschel and Mahon 2010). Reducing pollutant-related mortalities can only be achieved through a reduction of emissions and cleaner fuel technologies (reduced toxicity), or by using completely alternative options, alternative industrial production methods and sustainable transport systems, and encouraging active modes of transport.

1.2 Personal pollution exposure

Much of the earlier pollution research has relied on data from fixed site monitors which has been extrapolated across wider city areas. While they are sufficient for informing emissions guidelines and policies, fixed site methods often result in the underestimation of concentrations for some areas (Gulliver 2004). It has also been shown that background and kerbside monitoring stations provide poor indications of personal exposure (Gulliver and 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 being generally highest closest to their source. Therefore, only direct personal exposure assessment can provide accurate measures of exposure while travelling.

A wealth of published international research 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, many exceptions can be found 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 travelling 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 of fixed site monitoring for informing policy and promoting healthy transport decisions.

1.3 Research aims

This purpose of this project, undertaken 2009, was to assess the comparative risk associated with exposure to traffic pollution[1] when travelling via different transport modes.

The research objectives are to:

- provide an accurate measure of personal pollution exposure by mode
- provide information for transport decision-making at personal and societal levels
- provide a stronger base for advocating change in consumer behaviour.

1 It should be noted that in some places in New Zealand, especially parts of South Island, traffic is not the main source of pollution in wintertime. In these situations, woodsmoke from domestic heating can dominate. However, this research specifically focuses on traffic pollution, and so exposure to other sources is intentionally not considered and sampling is carried out to minimise the impact of other sources.

2 Literature review

2.1 Overview

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 grow into mega-metropolises (O'Neill et al 2003). Cities are home to a raft of social and environmental problems, and air pollution is a key issue because of its adverse effects on human health. Although urban pollution originates from a variety of sources, in most urban areas, the majority comes from transport emissions, although in some places in New Zealand, especially in South Island, woodsmoke is the major source in wintertime. High air pollution exposure has been linked to increased allergies, respiratory illnesses, birth defects and numerous forms of cancer (Brunekreef and Holgate 2002).

Pollutants associated with vehicle emissions include: particulate matter (PM; expressed as PM_{10}, $PM_{2.5}$, and PM_1) and UFPs (ultrafine particles ≤0.1µm)), black carbon, volatile organic compounds (VOCs; common VOCs found in petrol include benzene, toluene, ethylbenzene and xylenes, collectively known as BTEX), polycyclic aromatic hydrocarbons (PAHs), ozone (O_3), carbon monoxide (CO), carbon dioxide (CO_2), sulphur dioxide (SO_2) and mononitrogen 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 1996). Vehicular traffic also produces non-tailpipe pollutants, such as materials from clutch, brake and tyre abrasion, which include many heavy metals and resuspended road dust (Wahlin et al 2006).

Given these 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 travelling. Research from the United States (US) shows that 60% of a person's total daily pollution exposure is attributable to their daily commute (Hill and Gooch 2007). Therefore, it is important that commuters are 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 much of the transport exposure literature of the past 15 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 to traffic pollution

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 travelling by almost all alternative modes, including walking, bicycle, bus, subway and train (Adams et al 2002; Batterman et al 2002; Boogaard et al 2009; Cahill 2007; Chan et al 1991; Chan et al 1999; Chertok et al 2004; Dor et al 1995; Duci et al 2003; Duffy 1997; Georgoulis et al 2002; Gulliver and Briggs 2004; Kaur et al 2007; Kingham et al 1998; Lam 1999; Löfgren et al 1991; McNabola et al 2008b; Rank et al 2001; Shiohara et al 2005; Taylor and Ferguson 1998; Torre et al 2000; van Wijnen 1995; Vellopoulou and 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 because of 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; Cahill 2007; Dennekamp et al 2002; Levy et al 2002; McNabola et al 2008b; Peretz et al 2008).

Although these findings show that car exposure levels are generally among the highest, variation between transport modes at different study sites can be considerable. Their results are affected by variables such as vehicle makeup and configuration, ambient pollutant levels and local environmental factors, meaning that car drivers could 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 those of 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 that car drivers and passengers 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 (air conditioning on), exposure for $PM_{2.5}$ was higher at this setting than with the windows open (Cahill 2007).

Overall, the literature consistently presents comparatively high levels of CO and VOCs for the car mode (Chan et al 1991; Chertok et al 2004; Dor et al 1995; Duci et al 2003; Georgoulis et al 2002; McNabola et al 2008b; van Wijnen et al 1995; Vellopoulou and Ashmore 1998). Coupled with research which also ranks cars as receiving the highest levels of PM pollution, these findings suggest that the car commuter does not fare well compared to those who have other modal choices. However, when total accumulative intake is measured, car travel might not be the most detrimental mode when travel times and breathing rates are taken into account, especially compared to 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 1999; Piechocki-Minguy et al 2006; Saksena et al 2006; Saksena et al 2008; Tsai et al 2008). This is likely to be because motorcyclists are 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 to be approximately three times higher in motorcycles than in cars for PM_{10} (112.8 v 41.9 μg/m³) and $PM_{2.5}$ (67.5 v 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 a high density of traffic lights 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

Commuters using electric rail 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 1996; Chertok et al 2004; Lau and Chan 2003; Shiohara et al 2005). Currently, no PM data comparing aboveground electrified rail and other roadway modes are available. Exposure rates for VOCs, CO and NO_2 are lower because train tracks are generally situated away from traffic flows, cabins provide protection and the train itself is not a strong source of pollutants. However, the 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 as shown in table 2.1. Such findings are in agreement with a study by Lau and Chan (2003) in Hong Kong, where mean concentrations for BTEX were considerably lower for electric rail than those recorded in a taxi.

Table 2.1 Adjusted geometric means of BTEX and other pollutant levels (parts per billion) found in cars and trains (Chertok et al 2004)

Pollutant	Cars	Trains
Benzene	12.29	3.77
Toluene	28.76	12.44
Ethyl benzene	4.38	1.73
Xylenes	19.91	7.26
NO_2	29.70	14.85

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 (pulling), UFPs, black carbon and PAHs are much higher than for any other mode. When the locomotive is in the rear (pushing), levels are comparable to that of subway electric rail. However, fine particles ($PM_{2.5}$) for Boston were at around the same concentration as those of subway and car (windows up, vents open), regardless of where the locomotive was located (Cahill 2007).

Subway studies seem to differ, with some reporting the lowest exposure levels of all modes for PM, PAH, CO and benzene (Cahill 2007; Chan et al 2002; Gómez-Perales et al 2004), and others finding PM_{10} and $PM_{2.5}$ to be 3–10 times higher than for road surface transport modes (Aarnio et al 2005; Adams et al 2001; Johansson and Johansson 2003). Fromme et al (1998) found substantially higher PAH concentrations in the Berlin subway compared to inside cars. The explanations suggested included ambient seasonal variation and the influence of tar preservatives in the wooden railway ties. Gómez-Perales et al (2004) put such variance across subway studies down to differences in brake systems, ventilation systems and tunnel depth, while Kim et al (2008) suggest it could be caused by 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 that buses have the highest concentrations because of self-pollution from diesel engines (Chertok et al 2004; Farrar et al 2001). Tsai et al (2008) found PM 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; Cahill 2007; Dennekamp et al 2002; Fondelli et al 2008; Levy et al 2002; McNabola et

al 2008b; Peretz et al 2008). Hill and Gooch's results (2007) for $PM_{2.5}$ in a conventional diesel bus were around half that of cars (windows up), but UFPs (particles per 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. Substantial evidence also shows that exposure to CO in buses is much lower than in cars (Dor et al 1995; Duci et al 2003; Georgoulis et al 2002; Han and Naeher 2006; Kaur et al 2005a; Saksena et al 2008; Scotto di Marco et al 2005; van Wijnen 1995; Vellopoulou and Ashmore 1998).

Experiments with diesel particulate filters resulted in a reduction of UFP concentrations by around three-quarters to match ambient air levels – and the same was found in buses powered by compressed natural gas – but $PM_{2.5}$ concentrations were doubled and PAH concentrations were elevated. Biodiesel buses emit the lowest levels of UFPs and PAH, but slightly higher levels of $PM_{2.5}$ than traditional engines (Cahill 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 and 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 doors are open compared to when they are 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 that 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 (McNabola et al 2008b), Milan (Cattaneo et al 2009) and London (Kaur and Nieuwenhuijsen 2009), reported pedestrians were the least exposed to $PM_{2.5}$ and UFPs compared to car and bus users. However, a similar study completed in London produced opposite findings for all PM, including UFPs (Briggs et al 2008). Such results are supported by other research for PM_{10} and $PM_{2.5}$ (Dennekamp et al 2002; Gulliver and 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). Nazelle et al not only measured exposure concentrations, but also factored in inhalation rates. Preliminary findings suggest pedestrians could actually inhale greater amounts of UFPs than users of any other mode.

The literature does not disagree on CO exposure, with pedestrians being the least exposed (Saksena et al 2008; Zhao et al 2004). This is likely to be because vehicles are the only source of CO, whereas PM can be resuspended, 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 caused by the wind dispersion that is not experienced in the closed microenvironment setting of the vehicles used in most studies.

In Hill and Gooch's study (2007), 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 compressed natural gas or diesel particulate filter-equipped buses. Therefore, they were relatively low compared to most modes, but not lower than biodiesel buses or cars with the windows closed (air conditioning 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 for cyclists to be substantially lower than for cars and lower than for pedestrians during most sampling instances (van Wijnen 1995). Later research confirmed the contrary, with cyclists receiving higher levels than walking, car and bus (Mackay et al 1992). In London during 2005, a study found levels to be about the same as in cars (Kaur et al 2005a).

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

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

Apparently, no studies report higher VOC concentrations for cyclists than for cars and buses. Alongside 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 the 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 in cyclist breathing zones. Bernmark et al (2006) produced a different view:

> *Relationships between heart rate… and oxygen uptake, and between [heart rate] and pulmonary ventilation… for each participant were established in laboratory tests. The [pulmonary ventilation] 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*

The following year, O'Donoghue et al (2007) compared cyclist VOC inhalation to that of 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 could result in a 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, various potential factors influence cyclist exposure (Kaur et al 2007). These include:

- position on the road
- traffic light timings
- ability to pass between congested traffic
- height of cyclist from ground
- chosen route
- traffic density
- use of bus or cyclist lanes.

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 the position on the 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 particles per cm^3) and building side (73,329 particles per cm^3) is indicative of a rapid decrease in particle concentrations when moving just a very small distance away from emission 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 increasing 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 while 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 and 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 concentrations 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 three-lane roadway (on the pavement) and on a boardwalk only two metres away. The footpath and boardwalk are separated only by a small 'low-boundary' wall, meaning the boardwalk is a mere 1-2 metres further 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 modelling showed that because of 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, they potentially have large implications for future walkway planning and design.

2.3.2 Cyclists

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

An early investigation by Bevan et al (1991) compared CO, respirable suspended particle and VOC concentrations along a busy roadway to those in a common parkland area. This study found CO and respirable suspended particle 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 a 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 modelling – explored the differences between cycling along the shortest possible route, cycling along a low-exposure route (back streets) and taking the shortest direct route by bus. The study found that total exposure for the shortest cyclist route was 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; for bus patrons, this figure was between 79% and 115% lower. 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 some 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 dependent on the presence of mechanical or manual construction work. The lack of difference in PM_{10} levels for back streets and main road areas are consistent with the findings of Hertel et al (2008). As with the 2005a study by Kaur et al, the findings of Berghmans et al (2009) were only presented as overall mean concentrations and were 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 back streets 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, demonstrating how concentrations varied geospatially and compared to background PM_3 modelling. Unlike the heterogeneous distribution of UFPs, $PM_{2.5}$ was found to be more spatially uniform across the study route because of the 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. Positioning cycleways as far away from the road as possible has obvious positive outcomes, but it is uncertain at which distance it becomes worthwhile. A separation as little as only two or three metres could 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 the cyclist rode 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 could only be applicable under particular conditions.

Because of the dependence on associated sources and the behaviour of different pollutants, it is apparent that coarse particle measurement is less important for cyclist exposure studies; instead, UFPs, CO and perhaps PM_1 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 could 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 to traffic pollution

2.4.1 Environmental factors

A multitude of variables 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
- vehicle makeup and configuration.

2.4.2 Physical environment

Building configuration, road layout, trees and roadside structures have an effect on the accumulation and dispersion of pollutants (Baldauf et al 2008; Briggs et al 2008). Comparisons between an open area of terrain 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 were well mixed vertically (Baldauf et al 2008).

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 three-dimensional computation fluid dynamics, McNabola et al (2009) discovered that low boundary walls can reduce pedestrian exposure by 40% for perpendicular wind directions and up to 75% for parallel wind directions.

2.4.3 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, the weather can have an indirect influence on less obvious factors. For example, in countries with very cold climates, particulates from studded tyre abrasion are reported to significantly elevate levels of high particle mass concentrations (Gustafsson et al 2008).

2.4.4 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).

2.4.5 Travel behaviour

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

2.4.6 Vehicle makeup and configuration

Older and poorly maintained vehicles are more likely to emit higher amounts of exhaust fumes. New vehicles often have very high in-cabin concentrations of VOCs because of their 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 and Gooch 2007) experience shows that 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 and El-Fadel (2008) found that in-vehicle CO ingression varied between 250 and 1250mg/h depending on the vehicle ventilation settings. Again, having the windows closed and the air conditioning on resulted in the lowest recordings, while having the windows half-opened and the vents closed resulted in the highest recordings, as it 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 traffic 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; Fischer et al 1998; Gauderman et al 2005). PM exposure can cause various cancers, chronic respiratory diseases and cardiovascular diseases (Miller et al 2007; Pandya et al 2002; Smith 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. Because of their incredibly small size, UFPs are able to enter the body easily, 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 leukaemia, even after only short-term, low-level exposure (Feng et al 2010; 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 skin cancers (Boffetta et al 1997). High PAH exposure is also thought to cause premature birth and limit neurodevelopment during the first three years of life (Jedrychowski 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 and 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 by those 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$\mu g/m^3$ 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

Initially, car commuters were overwhelmingly seen to be exposed to higher concentrations of total air pollutants than those who walk, cycle or use public transport. While the majority of recent studies continue to support this position, several consider active modes of travel to be the most affected. Additionally, research has begun to try to take higher respiration rates into account. 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 and lastly train/subway, although this ranking is somewhat uncertain, as it is largely dependent on local environmental conditions and a range of other variables. It is for this reason that the current study explores pollutant exposure in the active travel modes of walking and cycling in more depth and in the New Zealand context.

3. Method

3.1 Overview

This section describes the methodology used in our study to achieve the study objectives:

- to provide an accurate measure of personal pollution exposure by mode
- to provide information to inform transport decision-making at personal and societal levels
- to provide a stronger base for advocating change in consumer behaviour.

It describes the monitoring regime, the instruments and the choice of study location, and outlines the methods used. Detailed results are presented in chapter 4.

3.2 Monitoring regime

A programme of monitoring took place under a number of different scenarios:

- Daily journeys to work by car, bus, bicycle and (in Auckland) train were compared simultaneously. These were repeated on a number of weekdays when conditions allowed (anticyclonic conditions, when wind speeds were light and pollution concentrations were expected to be at a maximum). Sampling took place at the same time of day (during the morning and evening rush hours) to minimise confounding factors.
- For bicycle exposure, the monitoring was done on major routes and on paths away from traffic.
- Personal sampling of individuals was undertaken during complete 24-hour periods so we could calculate the contribution of the daily commute to total daily pollution exposure.

3.3 Physical setting and selection of modes

3.3.1 Location

The sampling took place in Christchurch and Auckland. Christchurch has more cyclists and the topography is more straightforward (thereby increasing the chances of being able to establish clear relationships between exposure and modal choice). In Auckland, traffic pollution is much higher and the city has the largest population in the country. Sampling took place from 26 February to 1 April (Christchurch) and 27 April to 21 May 2009 (Auckland). Autumn was chosen as the ideal sampling period because of 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 cooler climate during winter.

3.3.2 Pilot study and sampling configuration

Two pilot runs were conducted along the Christchurch routes to test the equipment and to confirm that the sampling timing was realistic in relation to bus timetables and cycling times. The timing of the runs was successful from the first trial but a few minor changes were made in regard to the equipment. The plastic inlet tubes were replaced with stainless steel tubes to ensure concentration measurements were not affected by particles sticking to the inside of plastic tubing. It was also found that the 3007 monitors (see section 3.5.2 for a description of the equipment) were prone to 'tilt errors' when shaken around on the bikes. Tilt errors occur when the instrument optics are contaminated with alcohol, and have been

problematic in other recent bicycle research (Boogaard et al 2009). The presence of front suspension on the bike appeared to almost completely alleviate tilt events, so one of the bikes was changed.

3.3.3 Selection of time of day and modes

All modes were sampled on all occasions. 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 (journey 1) and then another to the University of Canterbury (journey 2).

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 (in Auckland, problems with some equipment meant PM and UFP sampling was focused on the road modes and no data is available for the train).

For Christchurch, the modes consisted of bus, car, cycling off the road and cycling on the road. One cyclist rode an off-road route via dedicated cycleways, through parks and back streets, 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 on-road route.

In Auckland, the train was also sampled at the expense of the off-road cycle mode, partly because few suitable comparative off-road cycle routes exist in Auckland. The cyclist, car and bus again travelled the same route, which ran as closely as possible to the train line.

As only three 3007 meters were available for the four modes, one was switched between modes to ensure data was collected across all modes.

3.3.4 Effect of proximity to traffic

To investigate the impact of proximity to traffic, 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 4.5–7m away and the third on an off-road path approximately 17.5–19m away on average. Cyclists rode along a specified road/path section and turning around before going 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.

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 modelling might provide clearer answers than monitoring by means of numerous fixed sites, it may not be entirely representative of exposure while moving.

3.4 Routes

3.4.1 Christchurch inter-modal routes

The Christchurch run was split into two separate journeys to replicate two normal commutes within the rush hour. The first of these journeys started at 7:40am and ran 8.2km from 340 Main North Road to the city bus exchange (figure 3.1). 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, the

second part of the journey ran 7.5km to the University of Canterbury Geography department, arriving at 9:00am (figure 3.2). In the afternoon, the journey left the University at 4:45pm, arriving at Redwood at 6:05pm.

Figure 3.1 Journey 1: Redwood to Christchurch city bus exchange

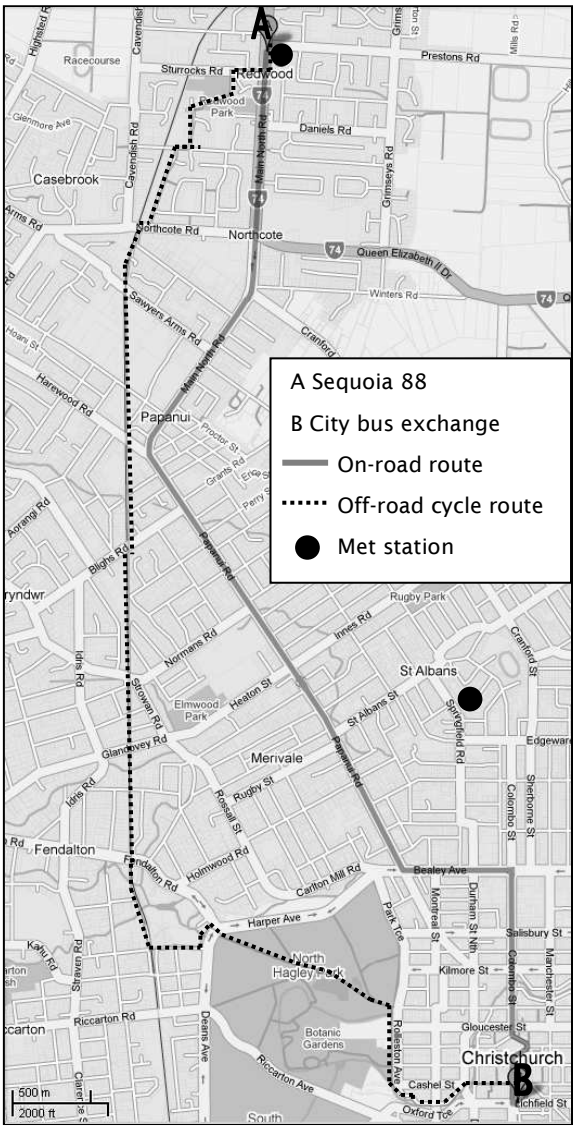
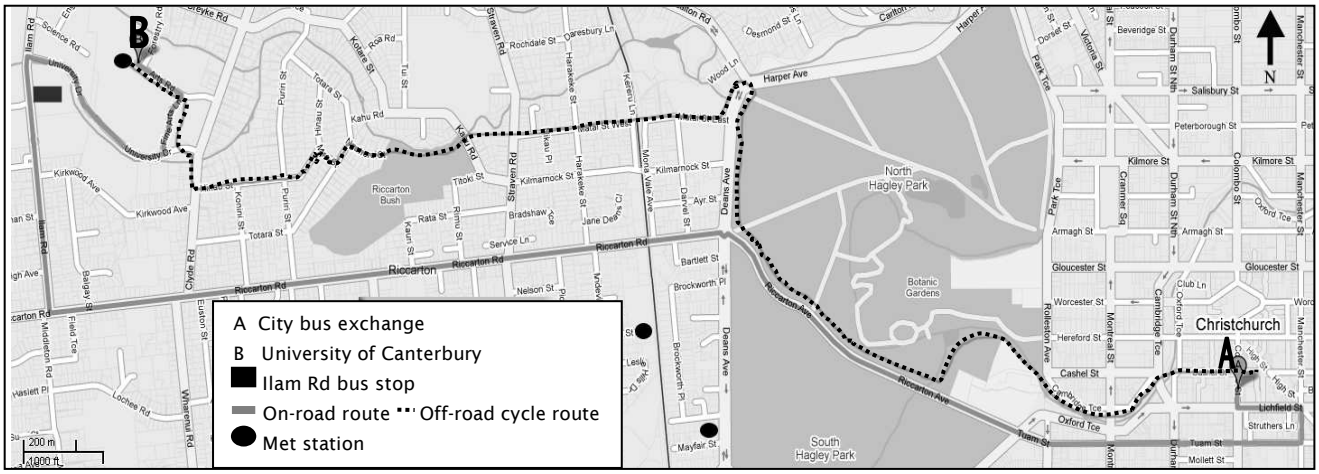


Figure 3.2 Journey 2: Christchurch city bus exchange to University of Canterbury



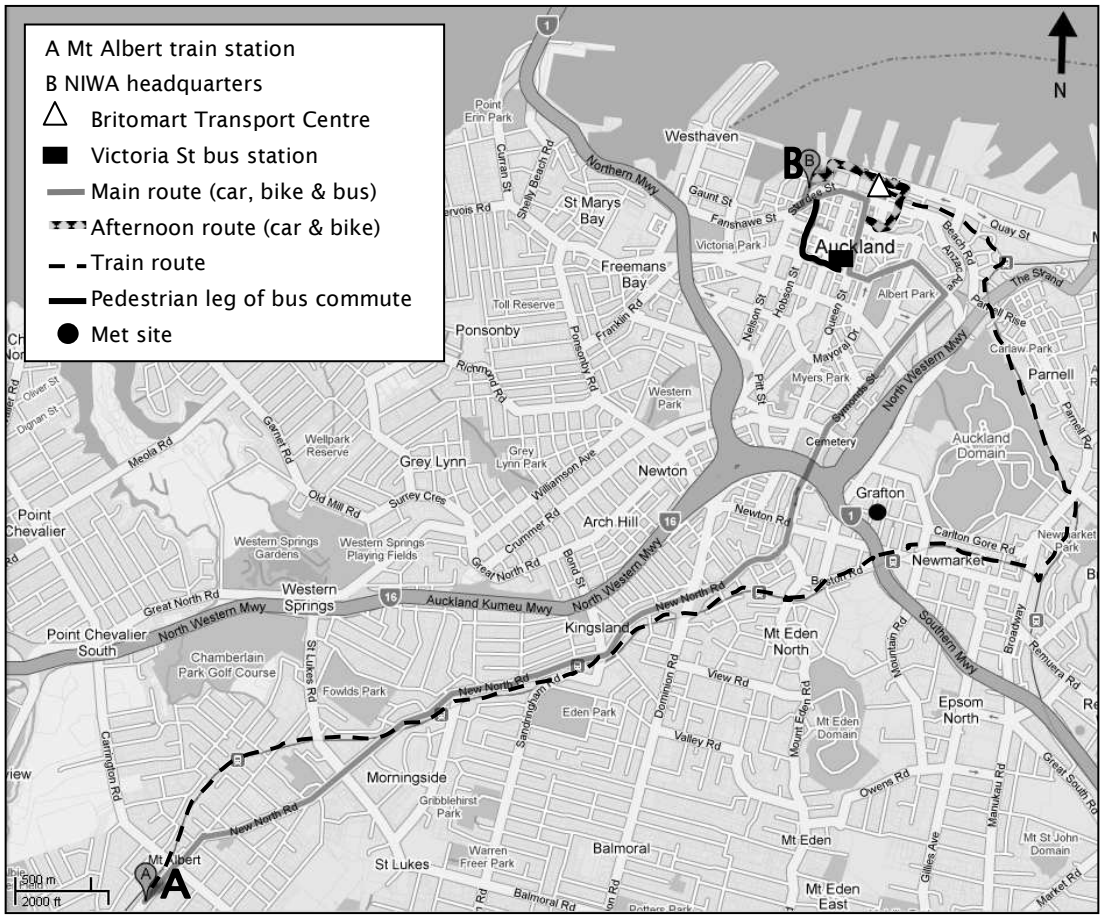
3.4.2 Auckland inter-modal route

The Auckland route ran from 947 New North Road at Mt Albert to the National Institute for Water and Atmospheric Research (NIWA) headquarters at Market Lane in the city centre (figure 3.3). This route was chosen because of:

- its proximity to the train track
- its proximity to volunteers' residences
- its use as a key commuting route to the city centre
- its use as a key bus route featuring dedicated rush hour bus lanes.

The car, bus and cyclist travelled 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 the cyclist. The car and cycle route also varied slightly during the afternoon because of 'Bus Only' turning restrictions but this was not considered to alter the results significantly. The total distance of the morning journey was 9km and the afternoon journey was slightly longer at 9.4km. In the morning, the journey started at 7:40am and finished at 8:40am; in the evening, it ran from 4:25 to 5:30pm.

Figure 3.3 Auckland inter-modal sampling route



3.4.3 Proximity to traffic routes

In both Christchurch and Auckland, routes were selected that enabled people to cycle parallel to the road at three consistent distances from traffic. One was situated on the road, a second 6–8m off the road and the third 17–19m from the traffic (see the Christchurch example in figure 3.4). In Christchurch, the route for the proximity sampling was along the Riccarton Avenue stretch of Hagley Park; in Auckland, Tamaki Drive in St Heliers Bay was selected.

Figure 3.4 Satellite image of Christchurch cycle sampling area showing one section and the position of travel paths on both sides of the road

3.5 Pollutants, instruments and equipment

3.5.1 General notes

This project monitored concentrations of the key traffic-related pollutants:

- particulates (those smaller than 10µm, PM_{10}; those smaller than 2.5µm, $PM_{2.5}$; and those smaller than 1µm, PM_1)
- UFPs (represented by particle number concentrations)
- CO.

All of the pollutants were measured simultaneously (wherever possible). At the start and end of sampling in each city, and at other appropriate times, all sampling units were co-located indoors to ensure consistency. Each of these samplers records real-time pollution levels with a temporal resolution of seconds.

3.5.2 Instruments and equipment

A variety of pollution monitoring units were used. All instruments and equipment were portable and could be carried by an individual for the purposes of personal pollution exposure sampling. A summary of the equipment used for all modes can be seen in table 3.1. Instrument choice was primarily based on practical suitability and successful use in previously 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. 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

UFPs (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). Kestrel 4500s were used to collect meteorological data at the start and end of the mobile sampling route and Nokia N82 Global positioning system (GPS) cellular phones were used to collect GPS coordinates as well as sound and photographs. The latter were able to be used to identify possible causes of peaks in the data.

Table 3.1 Instruments used for data collection

Instrument	Measures	Sampling range	Sampling resolution	Manufacturer	Number employed
Langan T15n	CO	0–200ppm	0.05 ppm 1 second intervals	Langan Instruments, San Francisco, CA, USA	4
GRIMM Environmental Dust Monitor (models 1.101, 1.107 and 1.108)	PM_{10}, $PM_{2.5}$ and $PM_{1.0}$	1–6500 µg/m^3	120nm to 30µm 6 second intervals	GRIMM Aerosol Technik GmbH & Co. KG, Ainring Dorfstrasse, Germany	4
TSI 3007 Condensation Particle Counter	UFP	0–500,000 pt/cm^3	0.01 to >1.0µm 1 second intervals	TSI Incorporated, Knoxville, TN, USA	3
Kestrel 4500	Meteorological data	N/A	1 minute intervals	Nielsen-Kellerman Inc., Boothwyn, PA, USA	6
Nokia N82 GPS cellular phone	GPS coordinates, sound and photographs	N/A	3 second intervals	Nokia Inc., Keilaniemi, Espoo, Finland	4

3.5.3 Sampling kit development

Four Kincrome heavy-duty tool kit bags were used to house the sampling instruments. The kit bag's 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 3.5).

Because of 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 200,000pt/cm^3, but 3007s can only reliably record concentrations up to 100,000pt/cm^3, so diluters were put together to dilute values by ~0.1. Knibbs et al (2009) have observed coincidence-related undercounting at concentrations greater than 100,000pt/cm^3. This occurs from multiple particles simultaneously passing through the single-particle counting optics. Diluters were made following the design reported by Knibbs et al (2009) 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 high-efficiency particulate air filter made by TSI Incorporated. The filter is a zero-check filter with a Y-type flow splitter to draw in 'dead air'.

Solid steel mounting racks were made to hold the kits securely 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 3.5). In the car, the kit was placed on the passenger seat, which meant the

sample inlets were at approximately chest height. In the bus, the bag was on the lap of the passenger, resulting in the sample inlets being around head height.

Figure 3.5 Example of a sampling kit attached to a cycle

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

GPS coordinate maps of pollutant concentration 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.

3.6 Analysis

Measured personal exposures consist of two main components: the contribution from near-field emission sources (motor vehicles on or very near the subject's route) and the contribution from other more diffuse sources (the 'urban background'). Urban background ambient pollution levels were determined from fixed monitoring sites, but localised and short-term variations were also extracted from the high-resolution personal monitoring data for periods when the near-field contribution was zero or close to zero.

The impact of transport mode on pollution exposure was assessed by calculating relative ratios between modes and comparing them to ambient levels at the time of day/day of week when the modal sampling took place.

Changes in pollution levels were compared to GPS data to examine the impact of geography and other factors on pollution levels.

Pollution levels were related to changes in weather conditions.

3.7 Study vehicles

The car used for all sampling in both cities was a standard 1992 Toyota Corolla four-door sedan imported from Japan. It was three years older than the average New Zealand car at the time of sampling (NZTA 2010). The car had been serviced regularly and was thought to be running cleanly and efficiently. To prevent biased results, it was important that the vehicle was not overly susceptible to the exchange of indoor/outdoor air. 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 and compared to a control vehicle of similar age and design. These tests confirmed that the vehicle used in this study was no more or less leaky than expected. In other words, we are confident that the in-car levels measured in the study vehicle will be reasonably representative of the New Zealand vehicle fleet and, in the context of this study, which looks at pollution exposure and not specifically at vehicle emissions, this is perfectly acceptable. Internal ventilation was set to reflect the most typical setting in New Zealand: all sampling runs were completed with the windows closed and the vents set to 'fresh', with the fan set to position 2 (of four 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). A related NZTA-funded project has investigated the relative importance of different characteristics of the in-vehicle environment including air conditioning, ventilation and windows being open or closed (Longley et al 2011).

The bus fleets in both Christchurch and Auckland cities predominantly consist of diesel engine buses. Red Bus Ltd in Christchurch mainly operates German-made MAN 17.223 model diesel buses, and Stagecoach Auckland mainly runs the Swedish Scania L94 model. In both cities, some older buses are running as well. 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. Nine ADK/ADB class DMUs, 10 ADL/ADC class DMUs, 2 DBR class locomotives and 14 DC class locomotives are in operation. 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 not consistent.

The cyclists rode at a speed of 10–12km/h. This speed reduced the likelihood of equipment being disturbed, especially as wind tunnel tests indicated that faster speeds reduced the ability of the instruments to sample effectively.

3.8 Statistical analysis

Data was analysed and statistics produced using SPSS. As the data were skewed (the values were not evenly distributed on both sides of the mean), non-parametric tests were used to test for differences between modal exposures.

4 Results and discussion

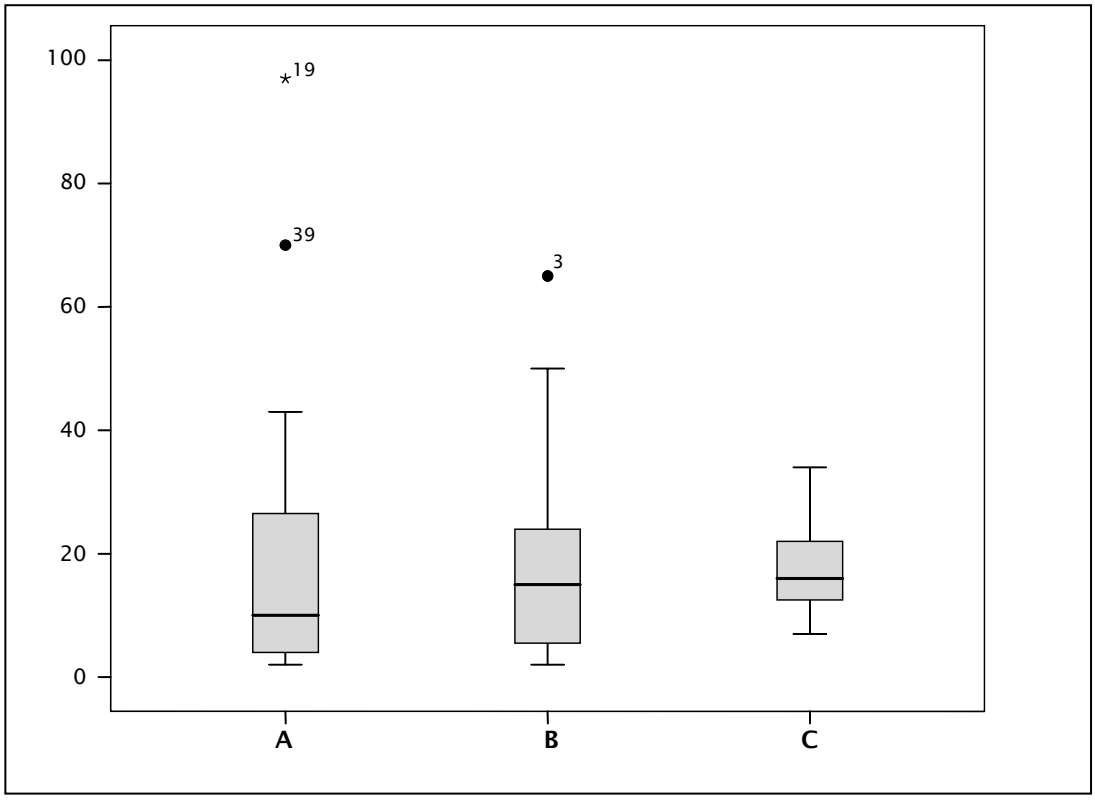
4.1 Modal variability

4.1.1 Interpreting the data

Pollution exposure levels for each mode are presented as box plots in this section (the descriptive statistics are in appendix A).

In these plots, the lines represent median values and the boxes represent the interquartile range (all values between the 25th and 75th percentile). The T-bars are 'whiskers' and extend to 1.5 times the height of the box (or maximum and minimum values); outside of these are outliers, where the circles are between 1.5 and three times the height of the bars, and the stars are extreme outliers if values are more than three times the height of the boxes.

Figure 4.1 Example of a box plot



Some care needs to be taken interpreting the median values for individual modes, as not all modes were sampled on each trip. This was the result of a range of factors, including scarcity of equipment (eg only three UFP counters were available) and equipment failure. Consequently, it is possible that high pollution days when some modes were not sampled could spuriously raise the mean value relative to the modes that may not have been sampled on those days. The data for individual modes is useful for getting an idea of pollution levels and for comparison to other studies, but is not ideal for comparing between modes. A more accurate way is to compare ratios of pollution exposure by mode, and these are also presented and discussed (descriptive statistics are presented in appendix A). In these, a ratio over 1 indicates that the first named mode of transport has a greater level than the second. Again, box and whisker plots are used to show the median, interquartile range and outliers. Unless otherwise indicated, reported differences between modes are statistically significant ($P < 0.05$).

4.1.2 Carbon monoxide

Despite the marked differences in background CO concentrations and local traffic patterns between the two cities, the results showed that in both Auckland and Christchurch, car drivers were consistently exposed to higher levels of CO than either bus passengers or cyclists (figures 4.2–4.5). The car driver was also exposed to a much wider range of mean trip concentrations (as shown by the larger interquartile range). These trends have been reported in a number of previous studies (see, for example, Chan et al 1999 and 2002; Han and Naeher 2005; Kaur et al 2005a; Mackay 2004 and Van Wijnen et al 1995). In most cities, the primary source of CO is from vehicle exhaust emissions. Therefore, thus the proximity of the air intake for cars to the exhaust plume, combined with low cabin volume for dilution and minimal exchange rates is likely to account for this trend. CO leakage into the cabin from the cars evaluated could also contribute to higher exposures; however, the consistency of the results with previous studies suggests this is unlikely.

It is interesting to note that in both cities, the car drivers experienced proportionately and statistically significantly higher exposures to CO compared to all the other modes. In Christchurch, the car driver was, on average, exposed to levels of CO 50% greater than the on-road cyclist, 150% greater than the off-road cyclist and 80% greater than the bus passenger (figure 4.3). In Auckland, the car driver was, on average, exposed to levels of CO 140% greater than the cyclist and 180% greater than the bus passenger and 480% greater than the train passenger (figure 4.5). Furthermore, in Auckland, the lowest individual trip exposure level for the car driver (2.6ppm) was higher than the highest mean exposure recorded on any other mode (bike, 2.2ppm).

More subtle differences between the cities were revealed between cyclist and public transport passenger exposures. In Auckland, the bus passenger and the cyclist were not statistically different ($P = 0.065$) but the cyclist was exposed to nearly twice the pollution of the train passenger. In Christchurch and Auckland, the bus passengers were exposed to similar levels to the on-road cyclists ($P = 0.43$ and $P = 0.65$) but to 40% more than the off-road cyclist in Christchurch.

Figure 4.2 Box plot of CO levels for trips in Christchurch

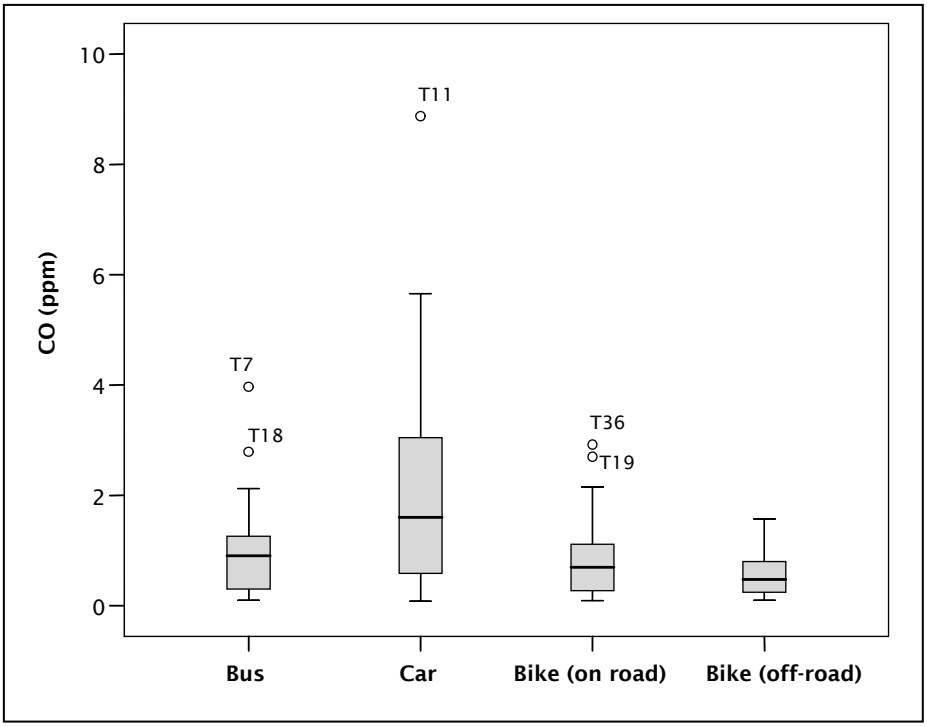
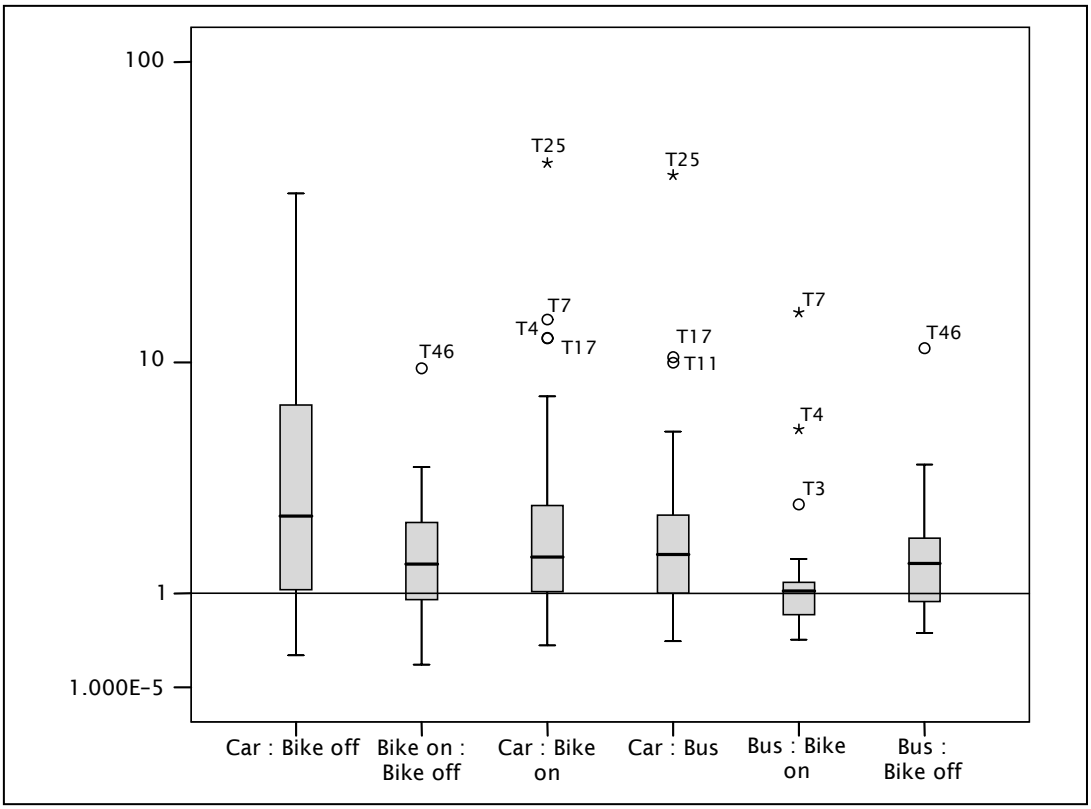


Figure 4.3 Box plot of ratios of CO levels for trips in Christchurch



Note: Because of some extreme outliers, the scale in this graph is logarithmic to make it easier to interpret.

Figure 4.4 Box plot of CO levels for trips in Auckland

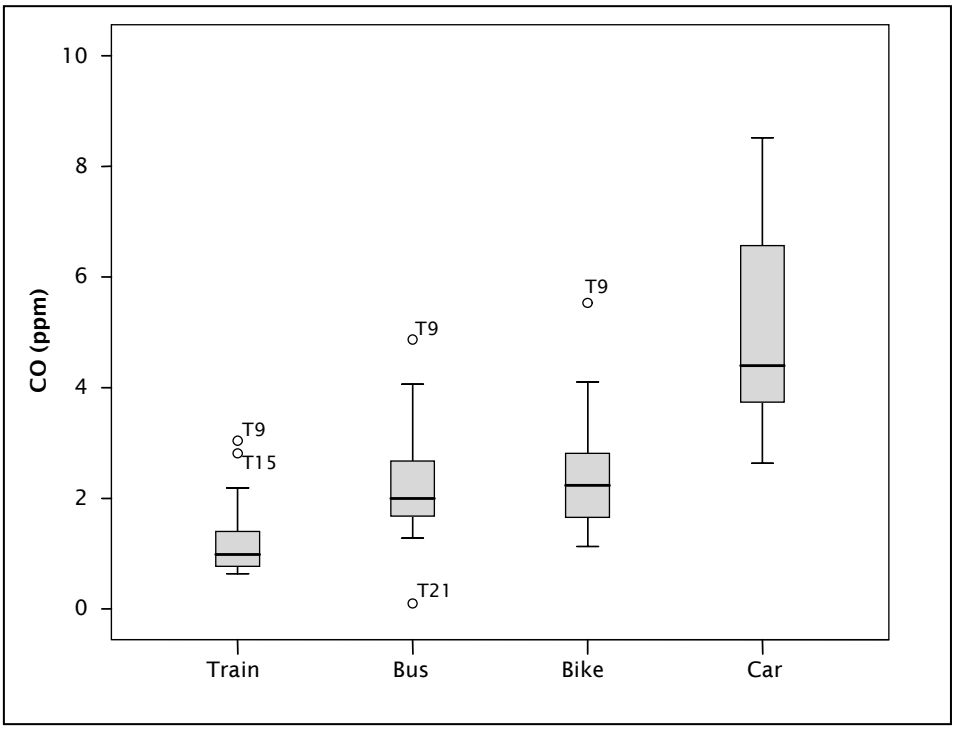
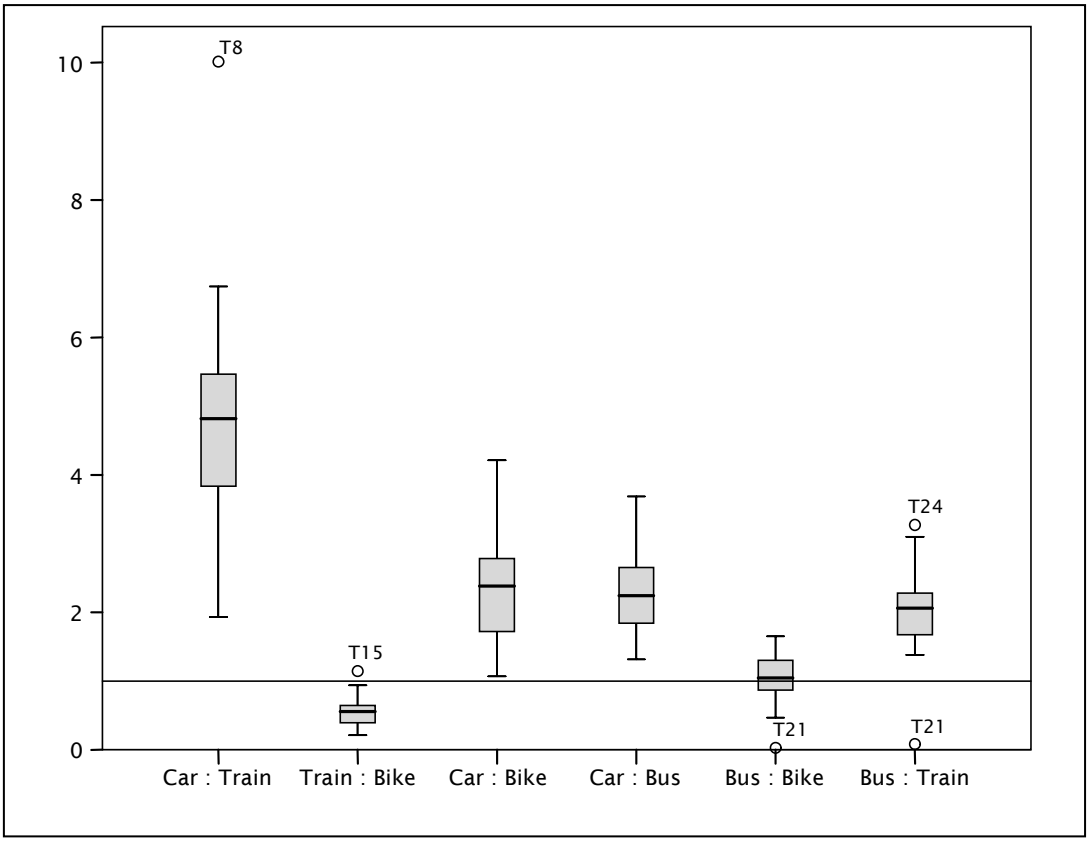


Figure 4.5 Box plot of ratios of CO levels for trips in Auckland



In Christchurch, the on-road cyclist was exposed to 60% more CO than the off-road cyclist. A visual example of the difference between the on- and off-road cyclists can be seen by mapping the pollution exposure of the two cyclists. Figure 4.6 shows the CO levels for the two cyclists on 12 March in the morning as they travelled from Redwood to the city centre. The mean exposure for the on-road cyclist for this trip was 2.15ppm and was 0.97ppm for the off-road cyclist, with an on:off ratio of 2.2. By looking at the map, the difference between the two cyclists can be easily seen, with higher exposures evident on certain parts of the on-road route.

Figure 4.7 shows the second part of that journey from the city centre to the University. On this trip, the on-road exposure level was 2.92ppm while that of the off-road cylist was 1.12ppm, a ratio of 2.6. Again, the difference between exposures is clear.

In Christchurch, the car driver exhibited the greatest number of outliers and the bus passenger the least, with the maximum recorded value of 52ppm being recorded in the car (figure 4.8). In Auckland, the trends were very different. The lowest mean exposure and the smallest variation in concentration about the mean were experienced by the train passenger. This is likely to be a product of both the absence of CO emissions from the train itself (diesel-powered) and the route taken, which was generally away from any local traffic sources. Furthermore, in Auckland, the cyclists experienced the greatest number of outliers, with the maximum reported being well over 100ppm (figure 4.9). The latter is quite different to the situation in Christchurch. A suggested reason for this is that the Auckland route had limited on-road cycle lanes, so that the cyclists were stuck in traffic, thus potentially exposing themselves to occasional high peaks of pollution. Christchurch had consistent cycle lanes so, although cyclists were still just as close to traffic, they rarely had to stop between motor vehicles. This is an interesting finding, and has potentially significant implications for cycle planning and health policy.

Figure 4.6 Map of on-road and off-road cyclists' comparative CO exposure: Redwood to Christchurch city centre, 7:40–8:20am, 12 March 2009

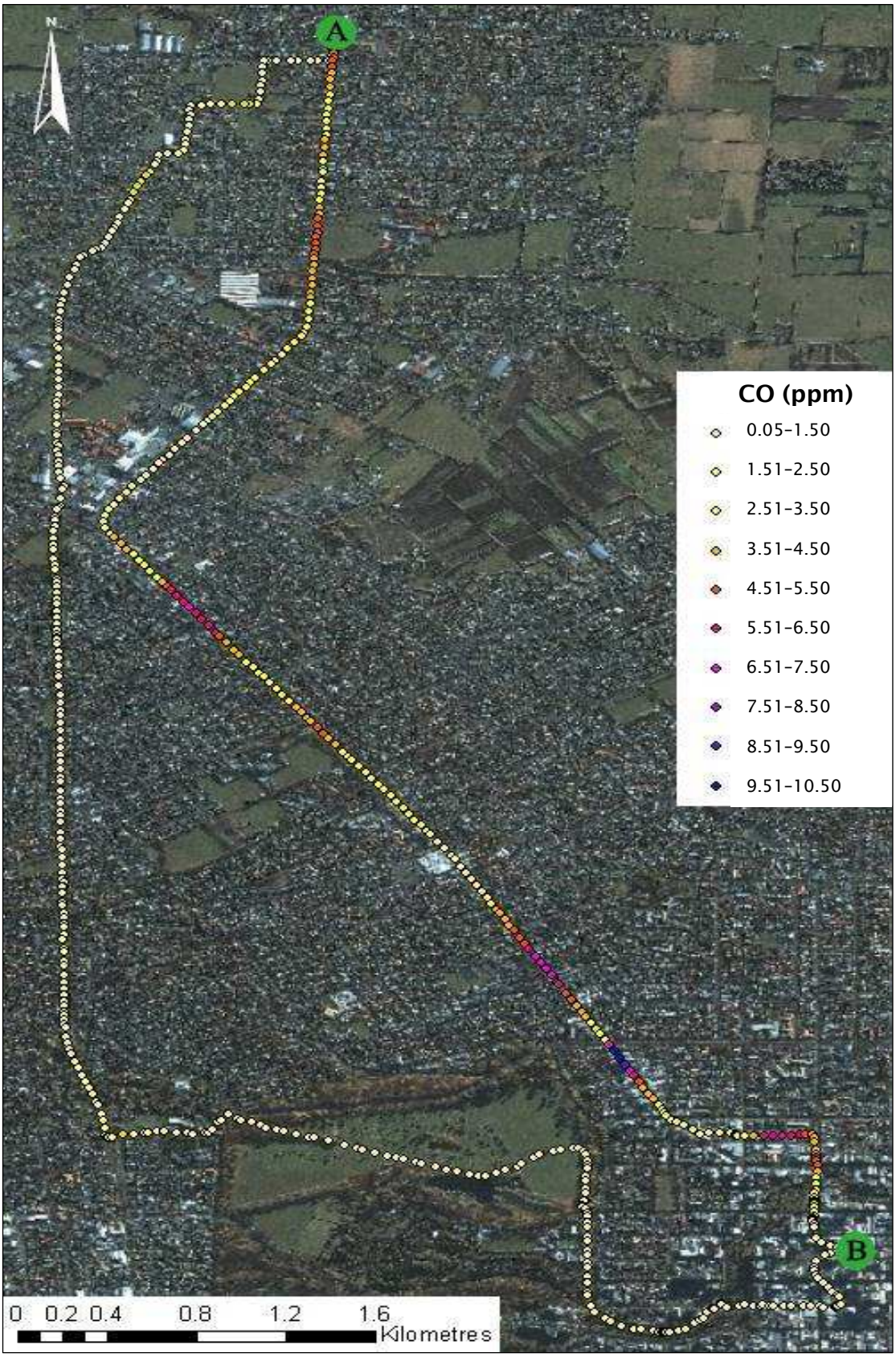


Figure 4.7 Map of on-road and off-road cyclists' comparative CO exposure: Christchurch city centre to Canterbury University, 8:30–9:00am, 12 March 2009

Figure 4.8 Box plot of CO levels for all data in Christchurch

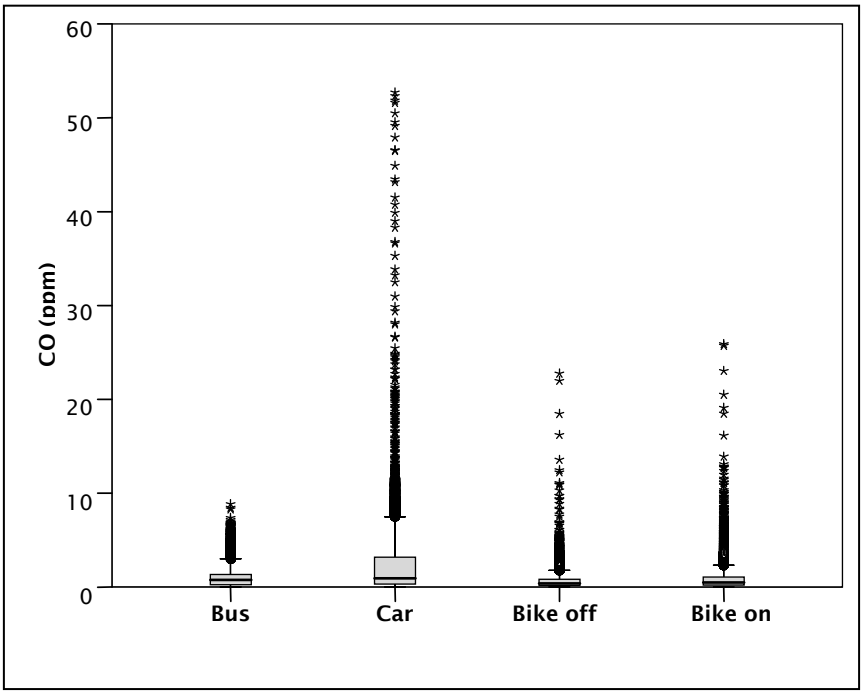
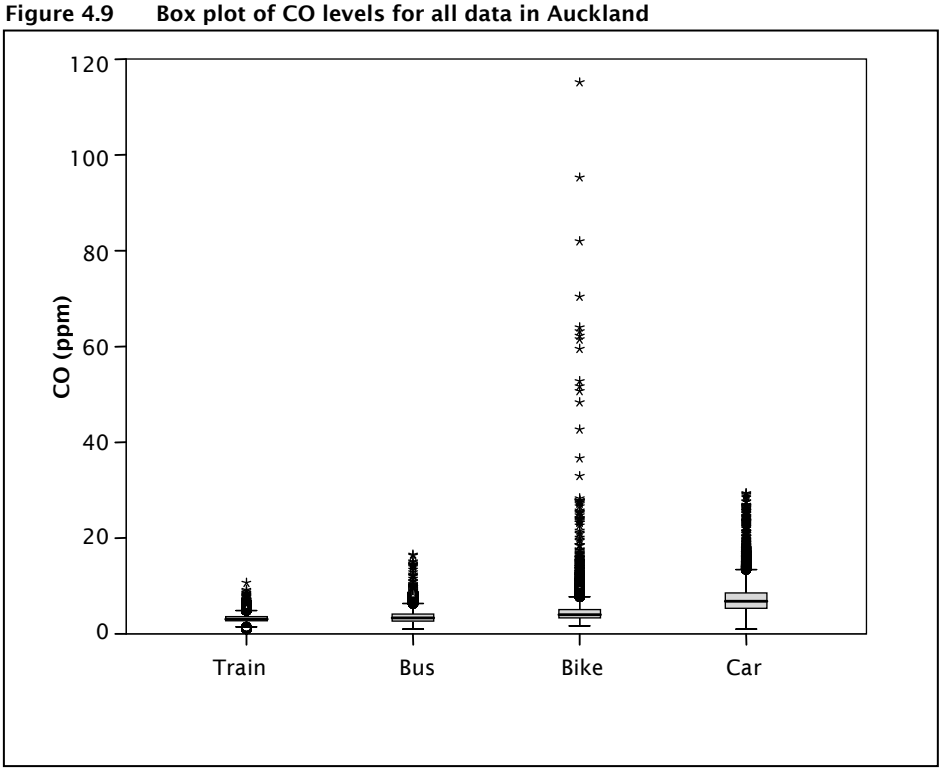


Figure 4.9 Box plot of CO levels for all data in Auckland



4.1.3 PM_{10}

Again, exposure results for particulate pollution in Auckland and Christchurch showed some consistency between the two. In both cities, PM_{10} concentrations showed less variability between transport modes and more within-mode variation compared to those observed for CO. In Christchurch, the bus passenger was, on average, exposed to 20% greater levels of PM_{10} than the car driver and 10% greater than the on-road cyclist, but was not statistically different from the off-road cyclist (P = 0.19). Somewhat surprisingly, the off-road cyclist was exposed to 20% more than the car driver but showed no statistically significant difference to the on-road cyclist (P = 0.31) (figures 4.10 and 4.11). These findings are somewhat surprising and contrary to much of the international literature. We suggest two possible interpretations. Firstly, this may suggest that the car had an effective filter system for removing this larger fraction of the particles (although the age of the car suggests that this is unlikely). Secondly, PM_{10} is not directly related to traffic exhaust emissions. Almost all particles emitted from diesel exhausts are below 0.05µm in diameter (Kittleson 1998) and thus contribute very little to mean particle mass relative to aged particles from background sources and coarse particles (mineral dust, sea spray, some wear products). Additionally, the Grimm instruments used in this study do not detect particles much smaller than the wavelength of light (below ~0.25µm in diameter in practice) because of its optical technology. The modal size of vehicle exhaust is well below 0.1µm, and thus measures of PM_{10} are insensitive to newly emitted traffic exhaust. Certainly, no traffic-related reason could explain why the off-road cyclist was exposed to higher levels that those experienced on the road. Furthermore, the off-road cyclist could have been exposed to intermittent sources of resuspended dust (from vegetation debris and soil) on unsealed bike paths. The resulting expected intermittent exposure is supported by the larger interquarter range in concentrations experienced by the off-road cyclist. Auckland showed no statistically significant differences between any combination of modes (figures 4.12 and 4.13).

Figure 4.10 Box plot of PM_{10} levels for trips in Christchurch

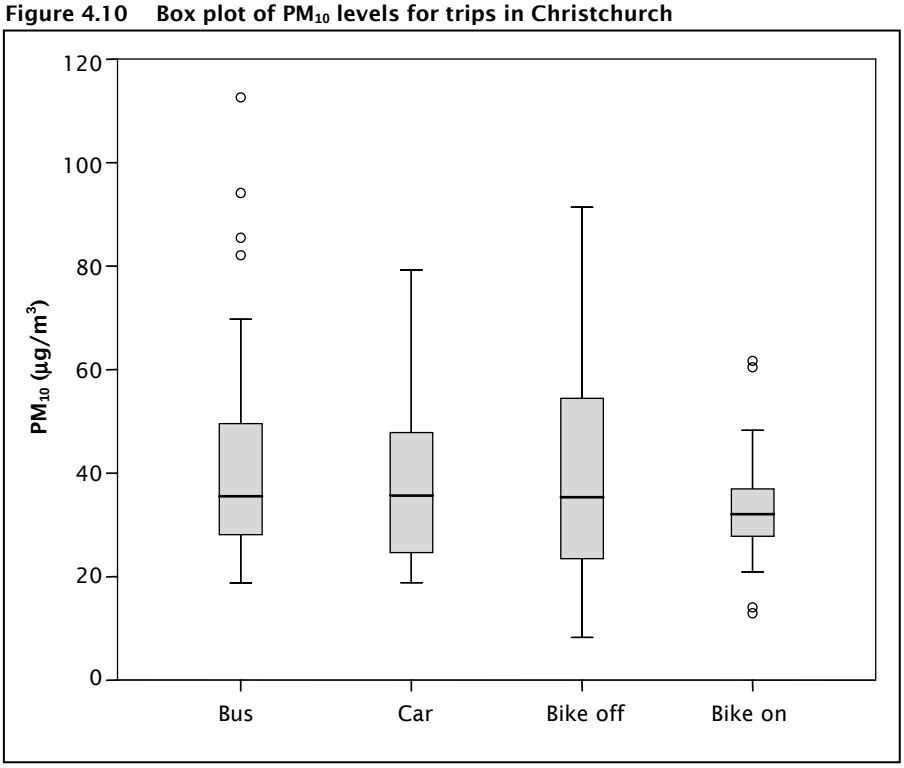
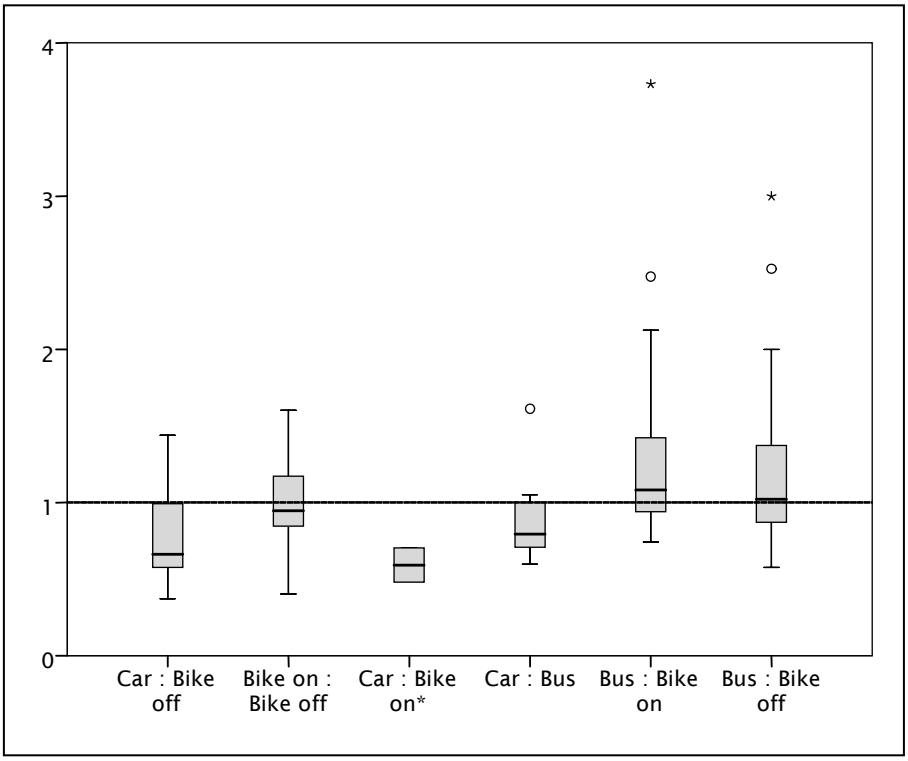


Figure 4.11 Box plot of ratios of PM_{10} levels for trips in Christchurch



*The car and on-road bike were only measured simultaneously on two occasions.

Figure 4.12 Box plot of PM_{10} levels for trips in Auckland

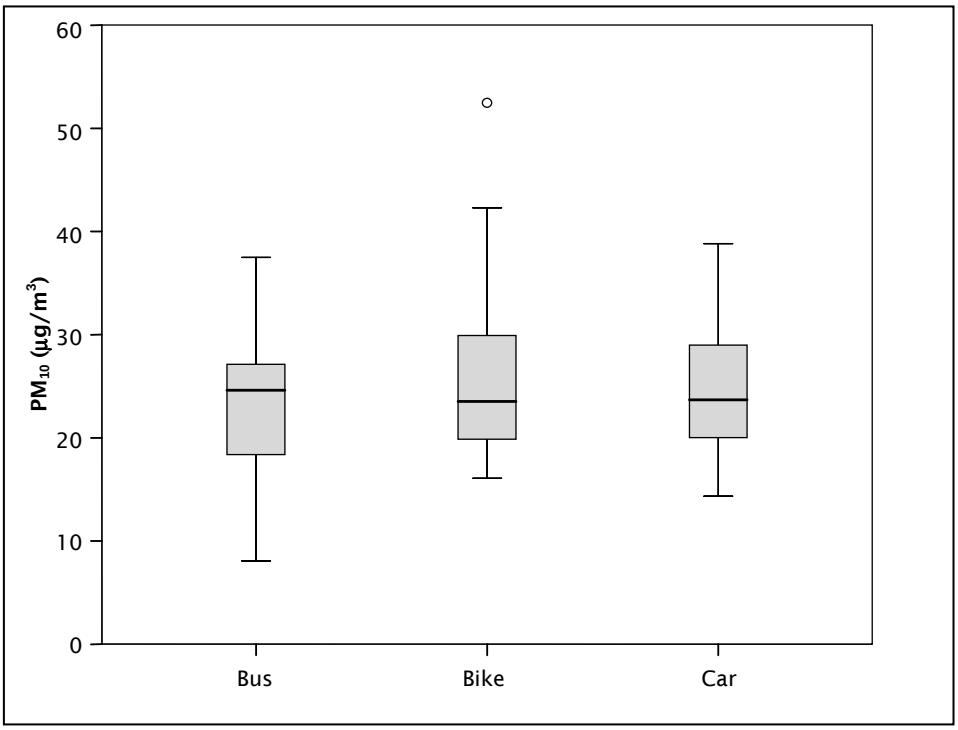
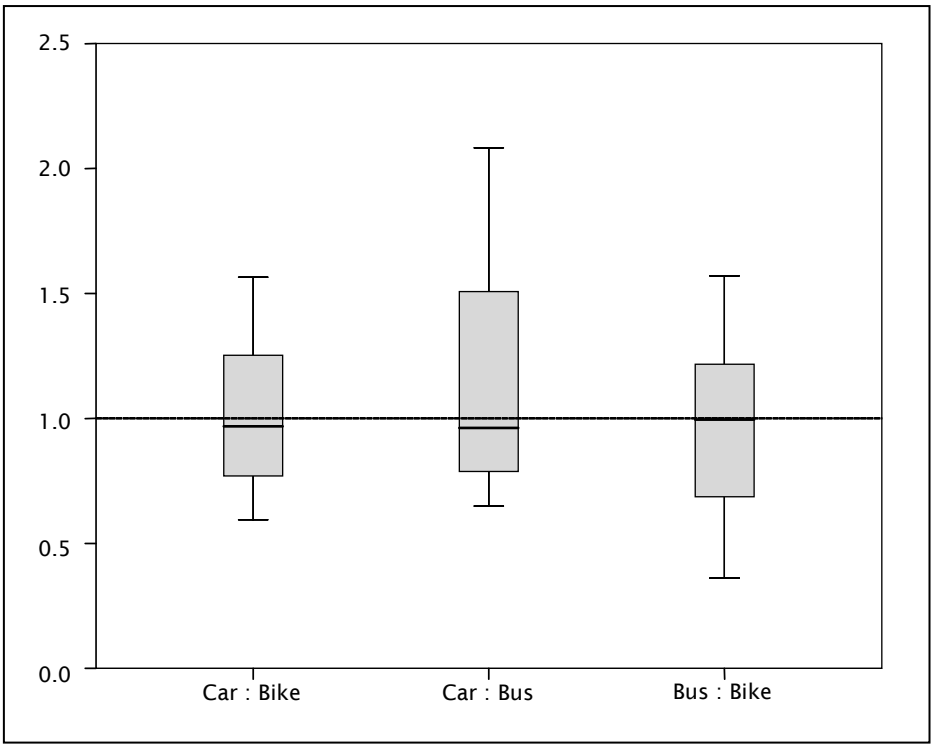


Figure 4.13 Box plot of ratios of PM_{10} levels for trips in Auckland



In Christchurch, the bus passenger exhibited the greatest number of outliers and the highest peak; the on-road cyclist the fewest outliers and the lowest peak (figure 4.14). In comparison, the bus passenger in

Auckland observed the lowest modal peak and lower outliers (figure 4.15). The reason for this is not obvious but it could be related to the nature of the bus fleet, and merits some further investigation.

Figure 4.14 Box plot of PM_{10} levels for all data in Christchurch

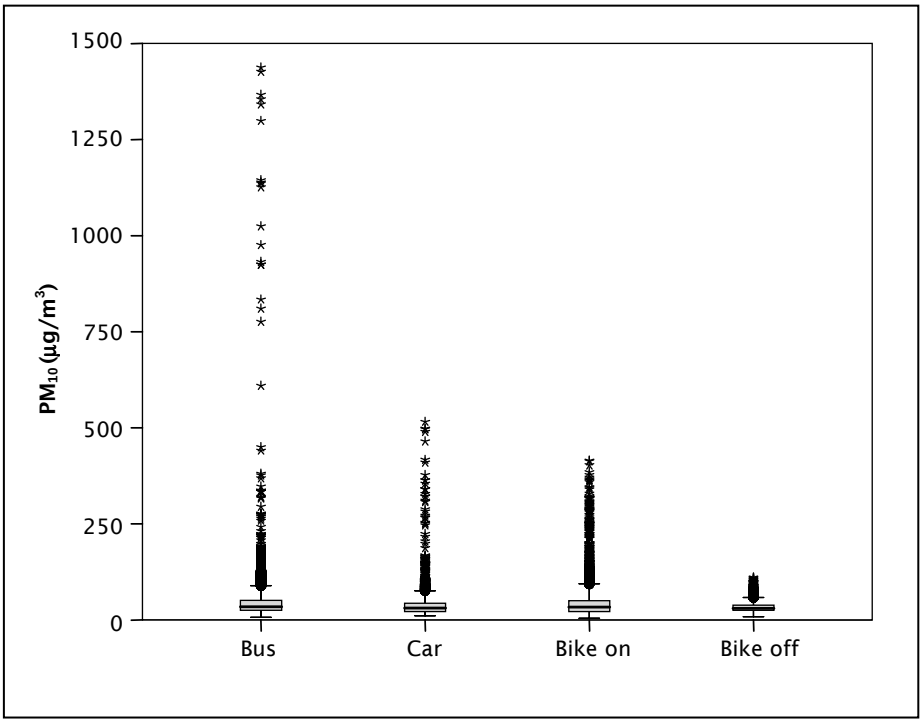
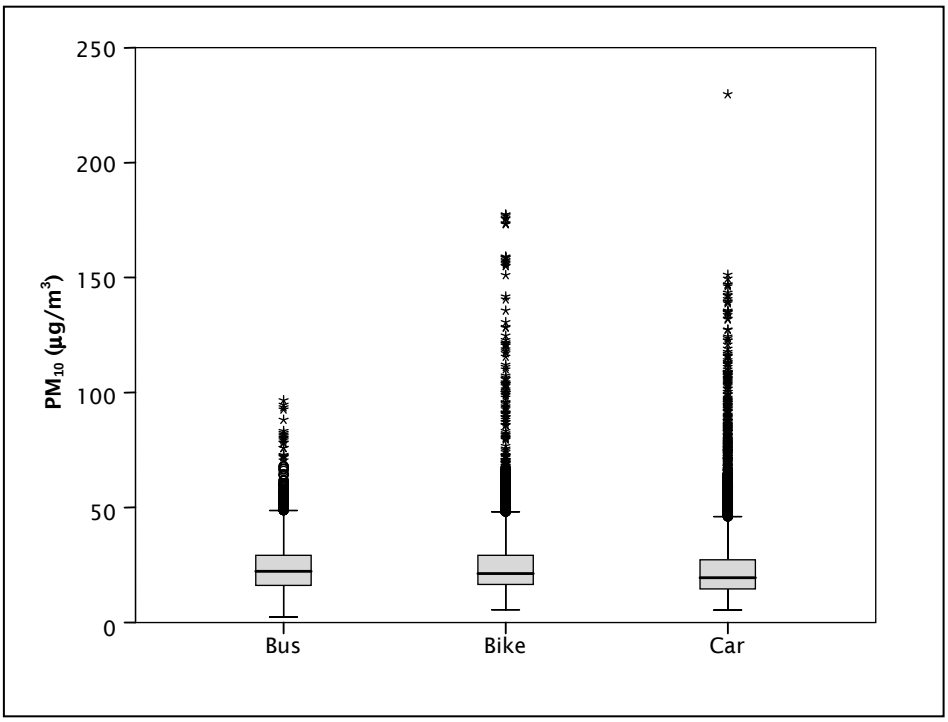


Figure 4.15 Box plot of PM_{10} levels for all data in Auckland



Examination of the correlation observed between pollutants during different commutes shows that in both cities, PM_{10} was poorly correlated with CO regardless of transport mode. Berghmans et al (2009) also

reported poor correlations between UFP and PM_{10} exposure observed by cyclists in Flanders, Belgium. This was attributed to different sources of the two pollutants and the strong influence of construction work in the city. Thus this leads us to conclude that PM_{10} is an inappropriate measure of exposure to traffic emissions, a fact supported by a recent report produced for Auckland Regional Council that found that less than 20% of PM_{10} emissions are from traffic sources (Davy et al 2009).

4.1.4 $PM_{2.5}$

In both Auckland and Christchurch, exposure to $PM_{2.5}$ was more significantly influenced by choice of transport mode. The highest mean exposures to $PM_{2.5}$ were reported by the bus passengers in both cities (figures 4.16–4.19), although differences between modes were not always statistically significant. In Christchurch, the bus passenger was, on average, exposed to 20% greater levels of $PM_{2.5}$ than the car driver and 10% greater than the on-road cyclist, but no other statistically significant differences were apparent between modes. In Auckland, the bus passenger's exposure was 50% higher than that of the cyclist but was not different from the car driver ($P = 0.067$). Exposure reported by the car driver was 10% greater than by the cyclist ($P < 0.05$). In Christchurch, the bus passenger was exposed to the greatest number of outliers and the highest peak in concentration (figure 4.20). In Auckland, this is very different: the bus passenger was exposed to lower peak concentrations but had a greater interquartile range, indicating a wider range of mean trip concentrations (figure 4.21). This is similar to the finding for PM_{10} and the same interpretation applies.

Figure 4.16 Box plot of $PM_{2.5}$ levels for trips in Christchurch

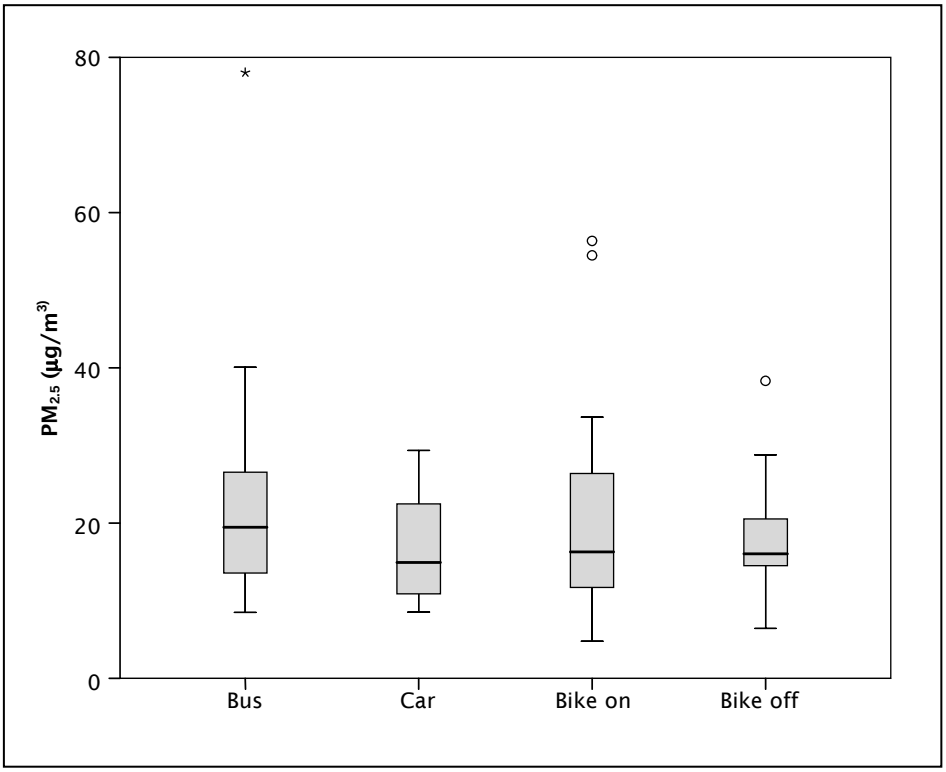
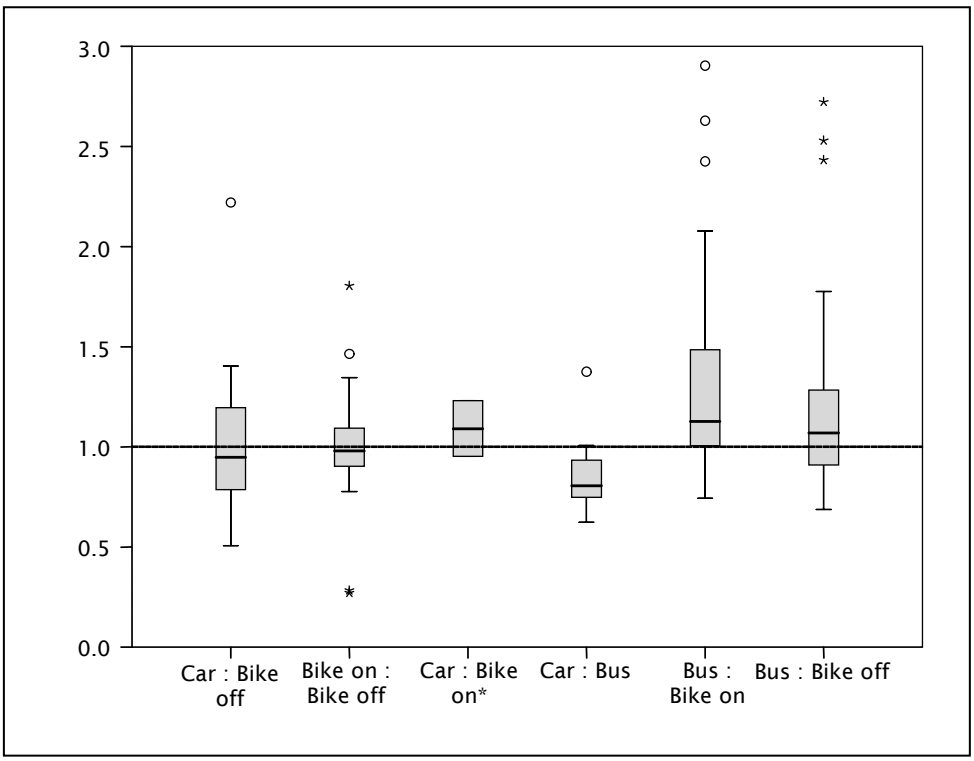


Figure 4.17 Box plot of ratios of $PM_{2.5}$ levels for trips in Christchurch



* The car and on-road bike were only measured simultaneously on two occasions.

Figure 4.18 Box plot of $PM_{2.5}$ levels for trips in Auckland

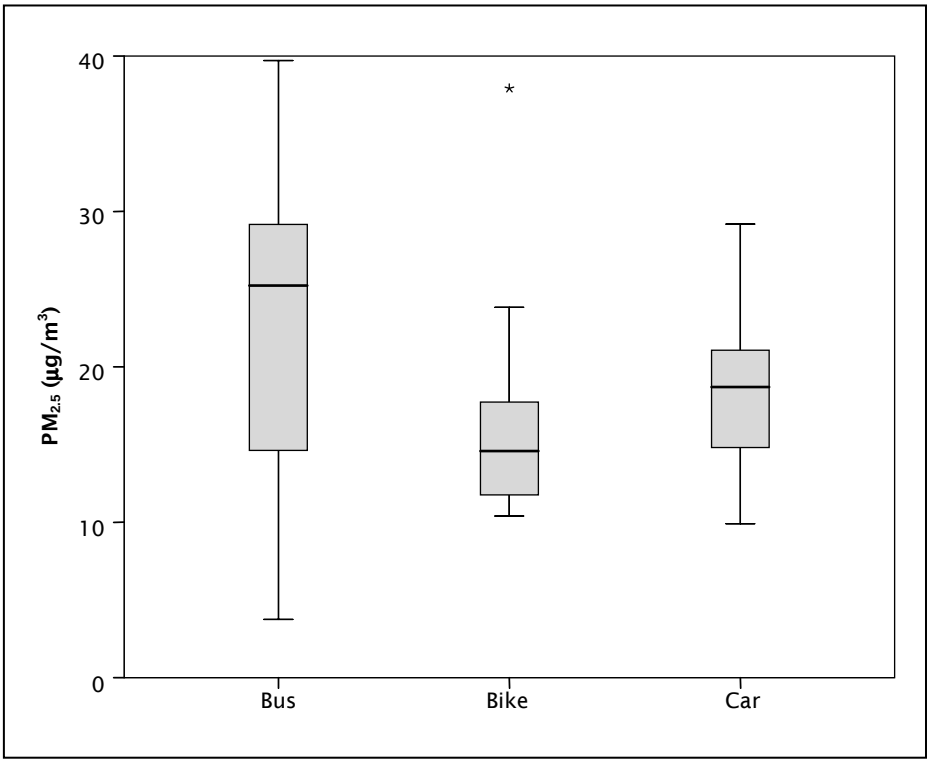


Figure 4.19 Box plot of ratios of $PM_{2.5}$ levels for trips in Auckland

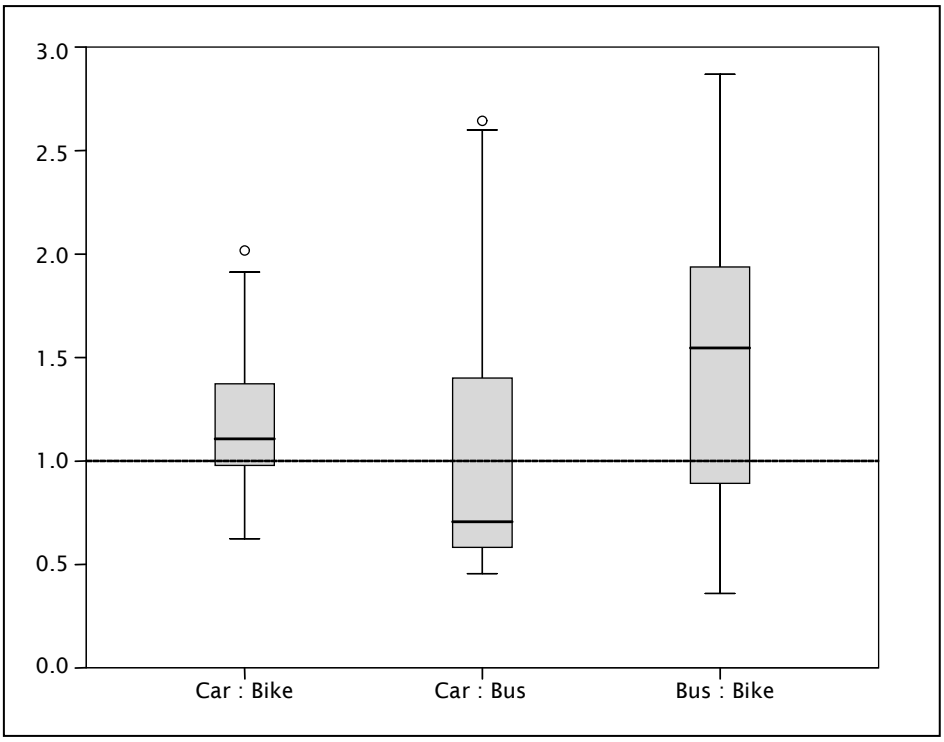


Figure 4.20 Box plot of $PM_{2.5}$ levels for all data in Christchurch

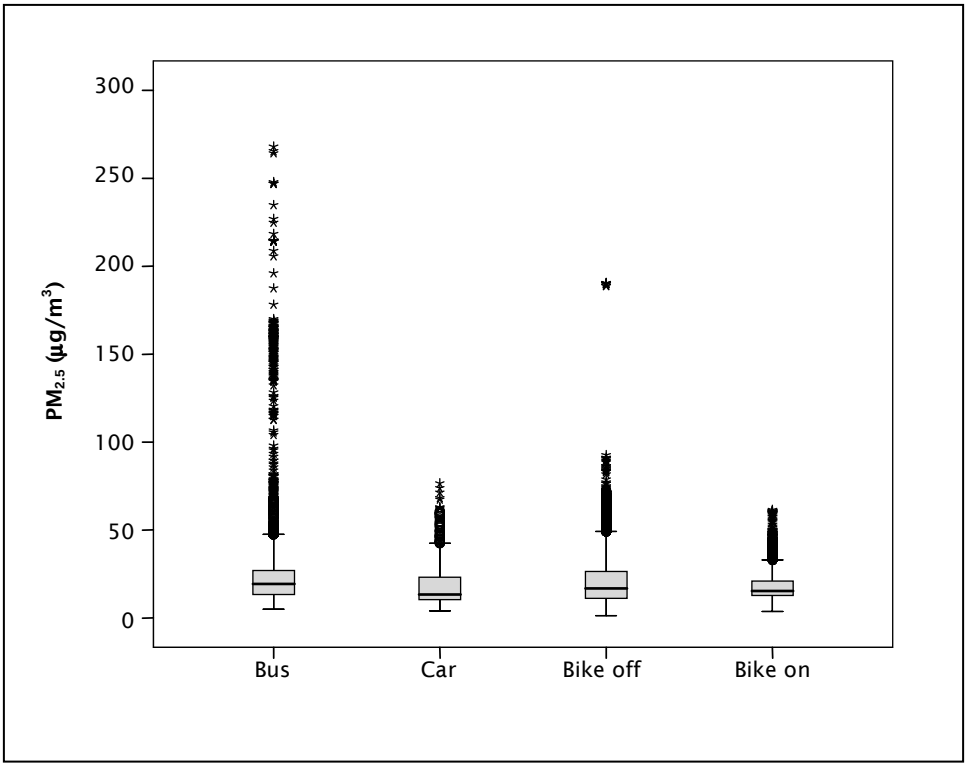
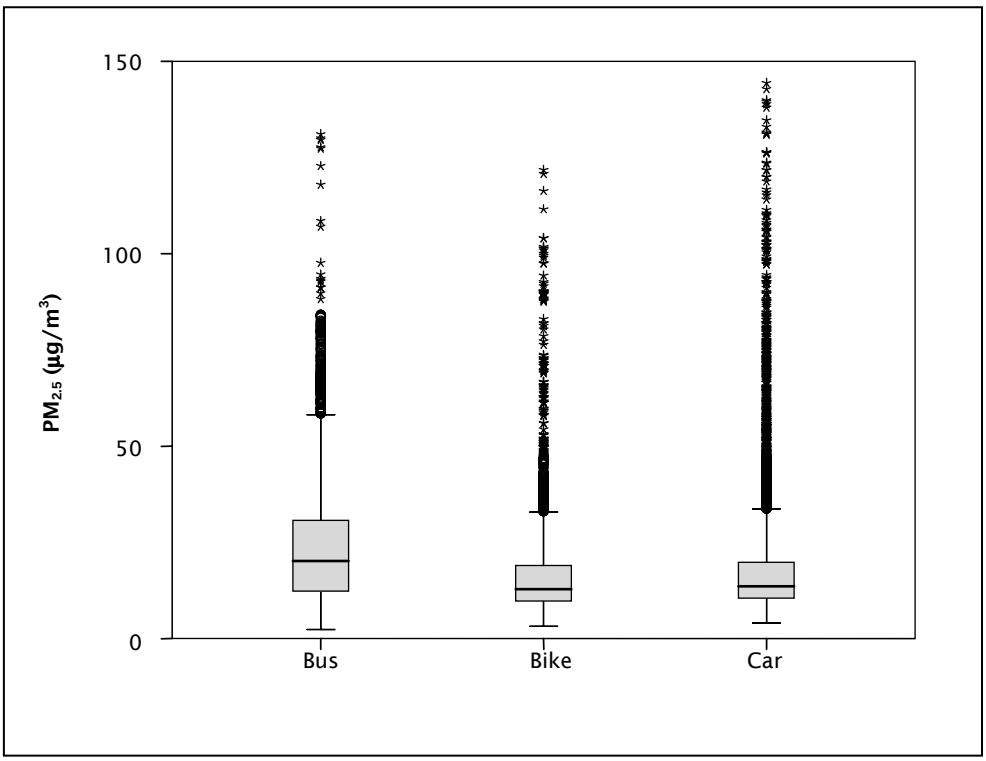


Figure 4.21 Box plot of $PM_{2.5}$ levels for all data in Auckland



In both cities, bus passenger exposure to $PM_{2.5}$ was often higher than in other transport modes. This result is often reported in the literature (see, for example, Adams et al 2001; Chan et al 2002; Dennekamp et al 2002; Kaur et al 2005b). Bus travel is typified by increased stop–start activity and consequent opening and closing of doors, enabling traffic-related pollutants to enter the bus cabin. However, the poor correlation between $PM_{2.5}$ and CO suggests that local external sources are not the only explanation for increased exposure to $PM_{2.5}$. Previous studies have suggested that diesel fuel emissions from the bus itself could contribute to increased exposure on board, as noted by Behrentz et al (2004), Marshall and Behrentz, (2005) and Rim et al (2008). This is more likely to occur as a result of the slower speeds of travel associated with stop–start activity around bus stops (Chan et al 2002b). In addition, passenger movement within the cabin could contribute to resuspension of material.

It is interesting to note that Kaur and Nieuwenhuijsen (2009) report a very low correlation between traffic density counts and $PM_{2.5}$ concentrations, suggesting that even this size of particle may not be a good indicator of local traffic-related emissions, and attributing spatial and temporal trends in concentration to long-range transport processes. This study was conducted in London, where not only are the diesel vehicle fleets much more modern and subject to much tighter emission regulations, but transboundary pollutant transport is likely to be a significant source of pollution. In Auckland and Christchurch, despite the impact of modal choice on exposure to $PM_{2.5}$, a poor correlation between CO and $PM_{2.5}$ was observed regardless of transport mode. This also suggests that $PM_{2.5}$ concentrations are not strongly related to local vehicle emissions but could be influenced by the bus itself. Further analysis of the bus fleet is required.

4.1.5 PM_1

In Auckland and Christchurch, the mode of transport significantly affected exposure to PM_1. The results show that in both cities, the bus passenger was consistently exposed to the highest mean concentrations of PM_1, followed by the car driver and the cyclist (figure 4.22 and 4.23). In Christchurch, the car driver was exposed to 40% more than the off-road cyclist but was not significantly different from the bus passenger

(P = 0.22) (figures 4.24 and 4.25). Only two simultaneous measurements of the car driver and on-road cyclist were taken, so no statistical comparison can be done, but in both cases, the car passenger experienced a higher level of exposure. The bus passenger was exposed to statistically significantly higher levels of PM_1 than both cyclists. However, in Auckland, the difference in exposure experienced by the car driver and bus passenger was not statistically significant (P = 0.17), but both the car driver and bus passenger were statistically significantly higher than the cyclist (30% for car and 60% for bus). In both cities, the bus passenger experienced a large interquartile range, indicating a wider range of mean trip concentrations; interestingly, in Christchurch, this range in exposure was also observed by the off-road cyclist. The on-road cyclist was exposed to levels 20% higher than the off-road cyclist.

In Christchurch, the bus passenger was exposed to the greatest number of outliers and the highest peak in concentration (figure 4.26). In Auckland, this was very different: the bus passenger was exposed to lower peak concentrations but a greater interquartile range than other modes, indicating a wider range of mean trip concentrations (figure 4.27). This is similar to the finding for PM_{10} and $PM_{2.5}$, and the same interpretation applies.

Figure 4.22 Box plot of PM_1 levels for trips in Christchurch

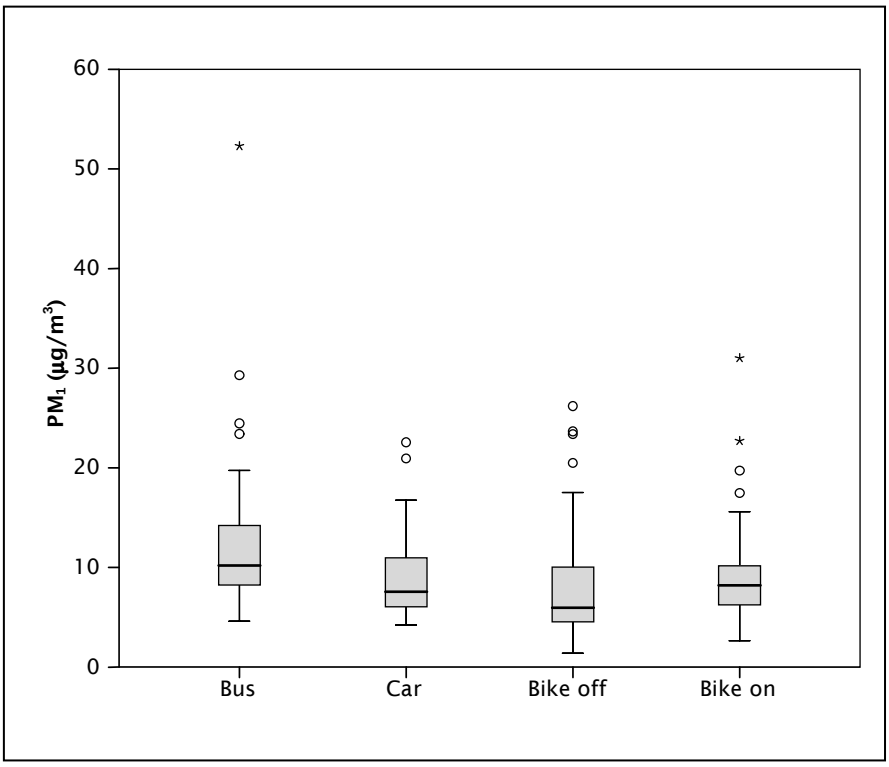


Figure 4.23 Box plot of PM_1 levels for trips in Auckland

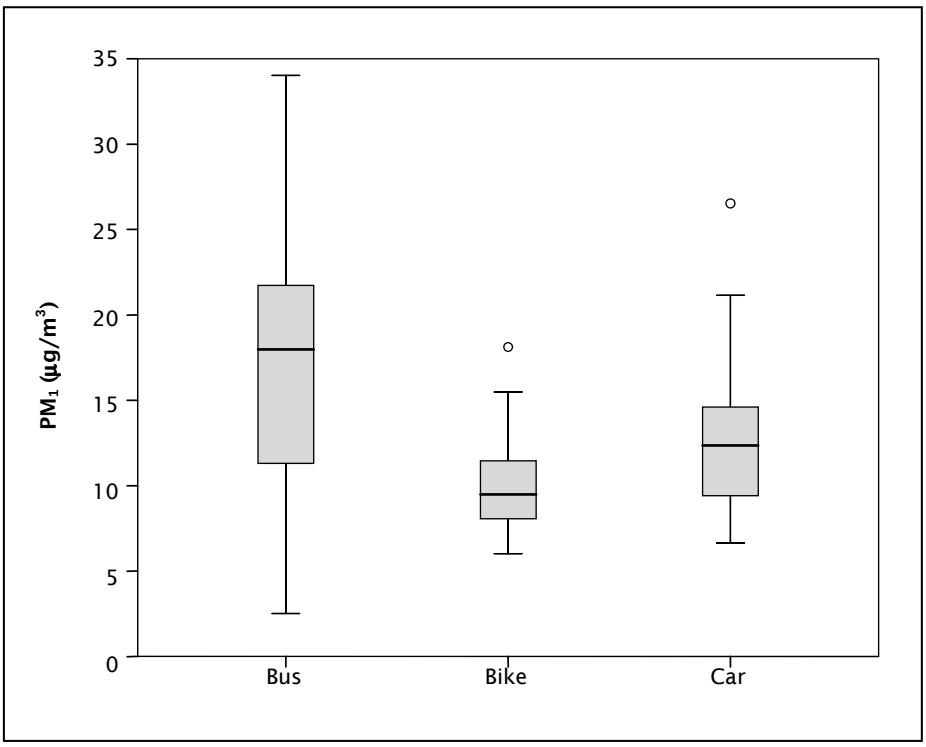
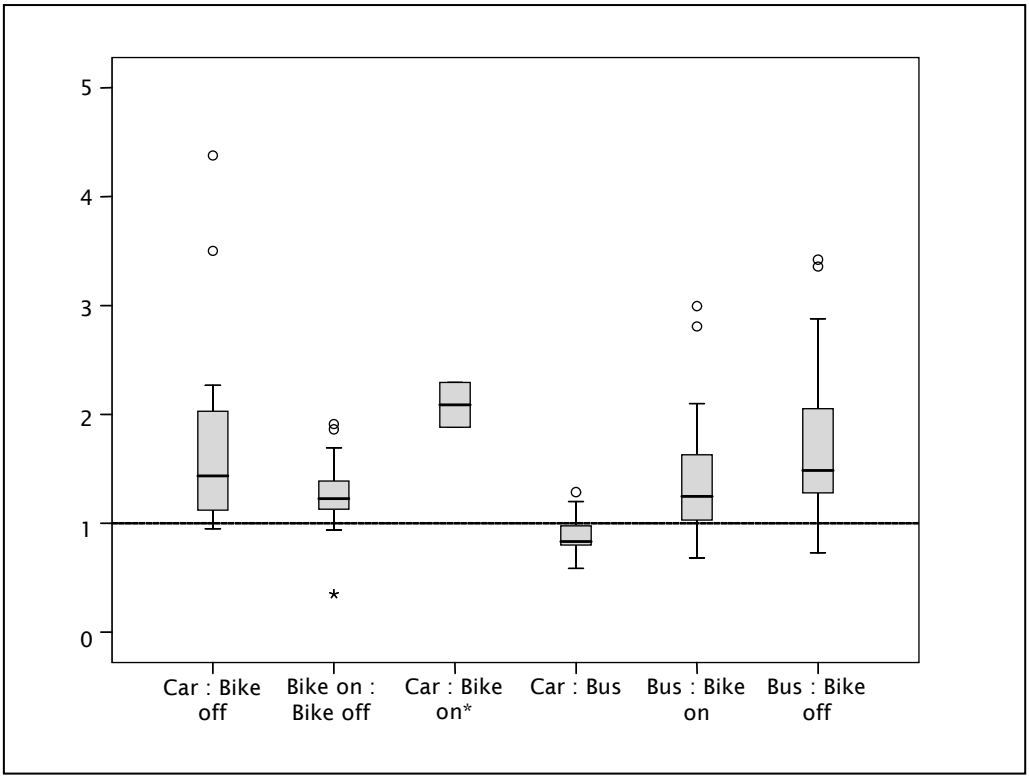


Figure 4.24 Box plot of ratios of PM_1 levels for trips in Christchurch



* The car and on-road bike were only simultaneously measured on two occasions.

Figure 4.25 Box plot of ratios of PM_1 levels for trips in Auckland

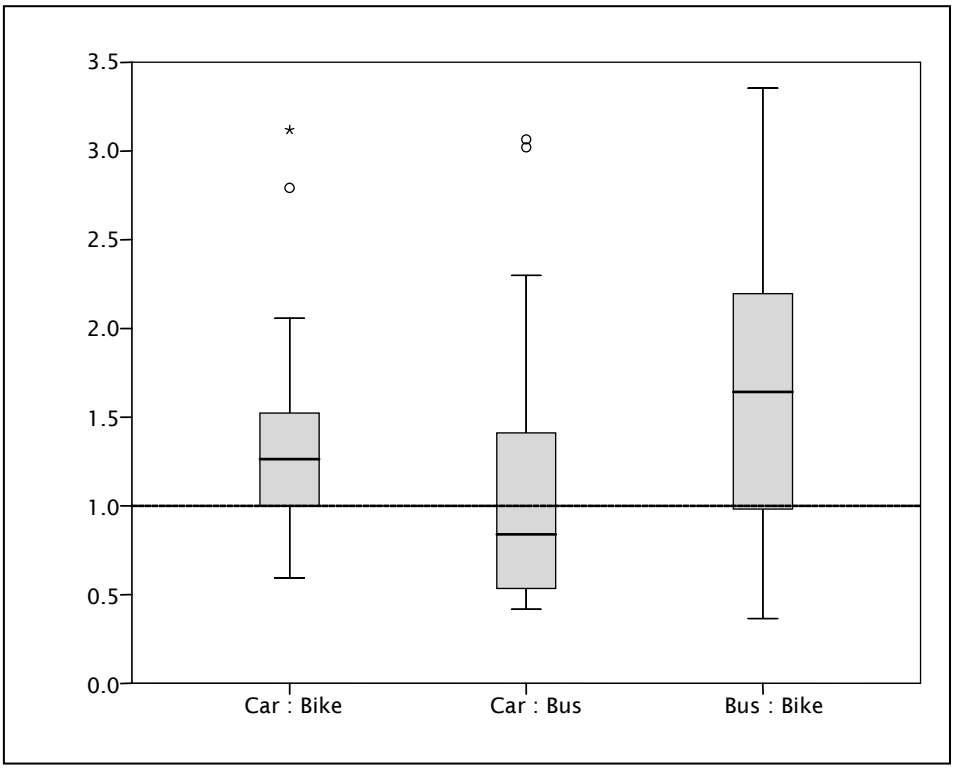


Figure 4.26 Box plot of PM_1 levels for all data in Christchurch

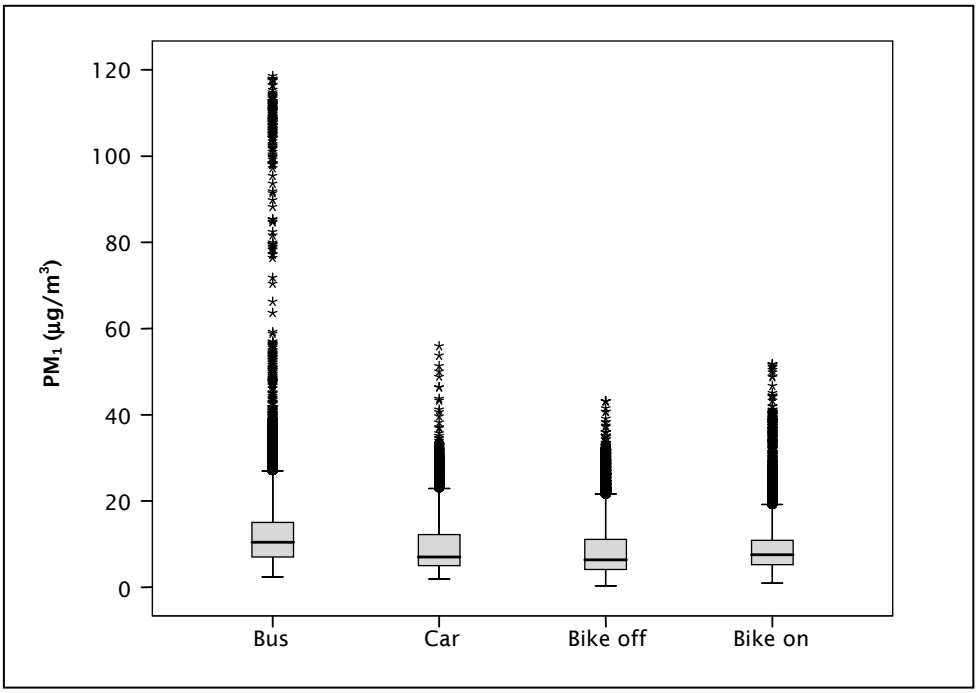
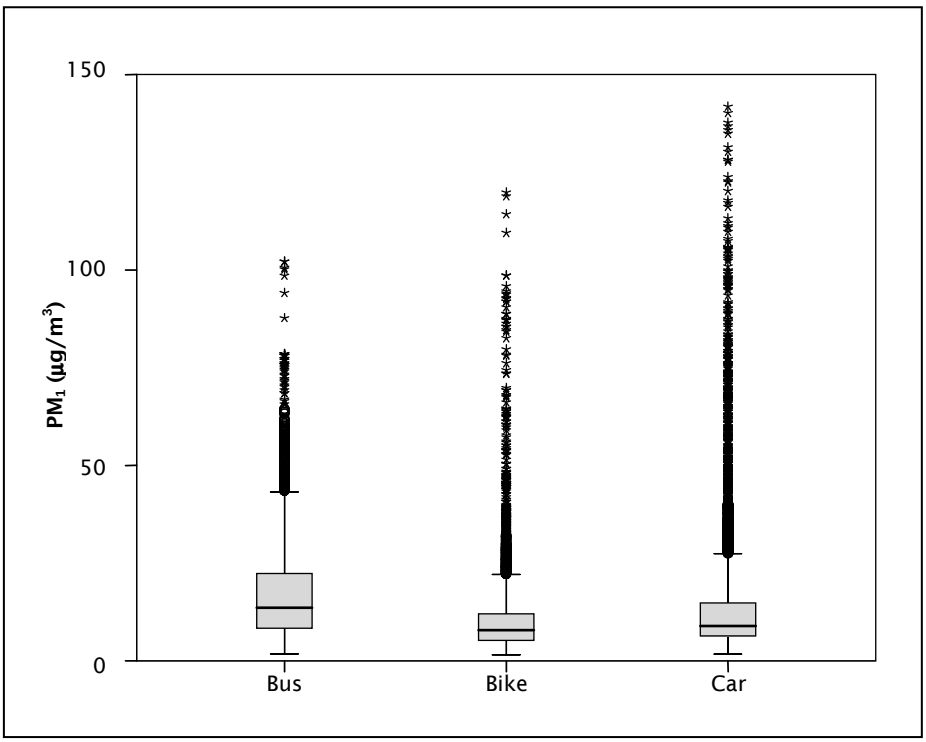


Figure 4.27 Box plot of PM_1 levels for all data in Auckland



In Auckland, a significant correlation was observed between CO and PM_1 during cycle commutes (0.83) but not between CO and $PM_{2.5}$ or PM_{10}. However, it is interesting to note that a significant correlation was not observed between these pollutants during either the bus or car journeys. Similarly, in Christchurch, weak or insignificant correlations between PM_1 and CO were reported. It is also interesting that the ratio of PM_1 to $PM_{2.5}$ was different during different transport modes. In both cities, the mean ratios during car and bike journeys were higher than during bike journeys. This suggests that while PM_1 may be a more appropriate measure of exposure to traffic emissions than the larger fractions, other sources of PM_1 are also important.

4.1.6 UFPs

Unfortunately, technical problems resulted in no UFP data being collected in Auckland. Furthermore, as only three UFP counters were available, simultaneous measurements on all four modes were not possible, and thus the number of samples available for comparative purposes was reduced.

Mean results for Christchurch show that the bus passenger had the highest median levels, followed by the car driver and then the two cyclists (figures 4.28 and 4.29). The car driver was exposed to levels 110% greater than the off-road cyclist; the on-road cyclist to 220% greater than the off-road cyclist. The bus passenger was exposed to levels 20% more than the on-road cyclist, while the bus passenger and car driver had similar exposure levels ($P = 0.58$). Again, having only two simultaneous samples meant we could not test the statistical significance between the car driver and on-road cyclist. The off-road cyclist had a significantly smaller interquartile range of exposures, which makes sense, as UFPs are closely linked to fresh emissions, and an off-road environment should experience a lower exposure to these. The peak short exposures and number of outliers (figure 4.30) were similar to those of the bus passenger, while the car driver and on-road cyclist had higher outliers and more of them, higher for the cyclist than the car

driver. This possibly suggests that buses and cars afford some protection from very high acute exposures but little against high long-term average exposures.

Figure 4.28 Box plot of UFP levels for trips in Christchurch

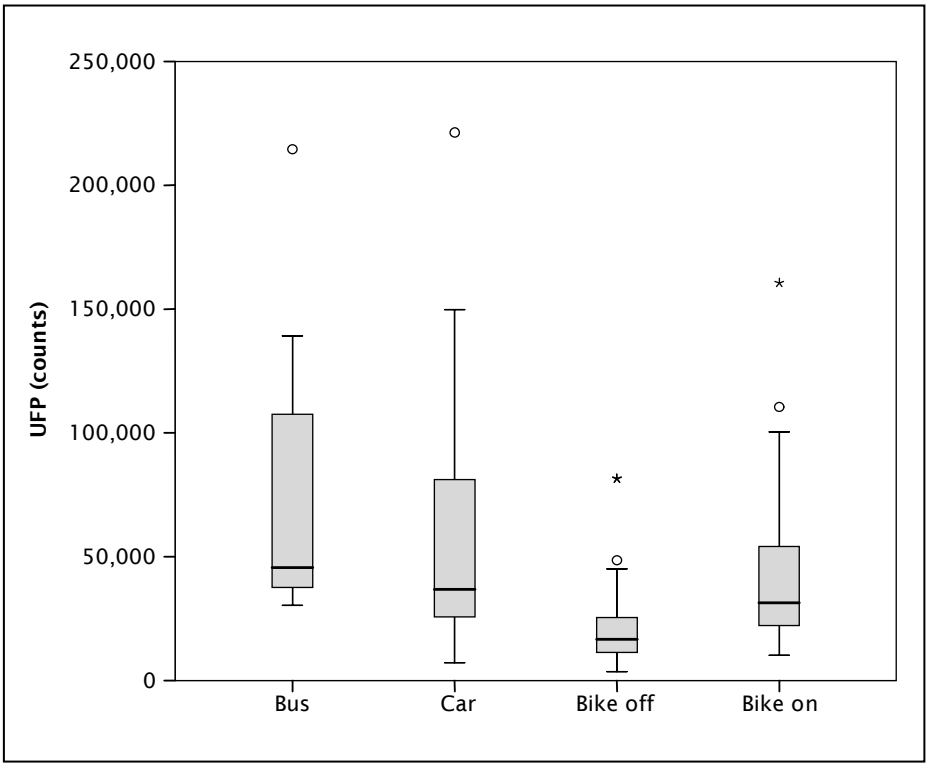
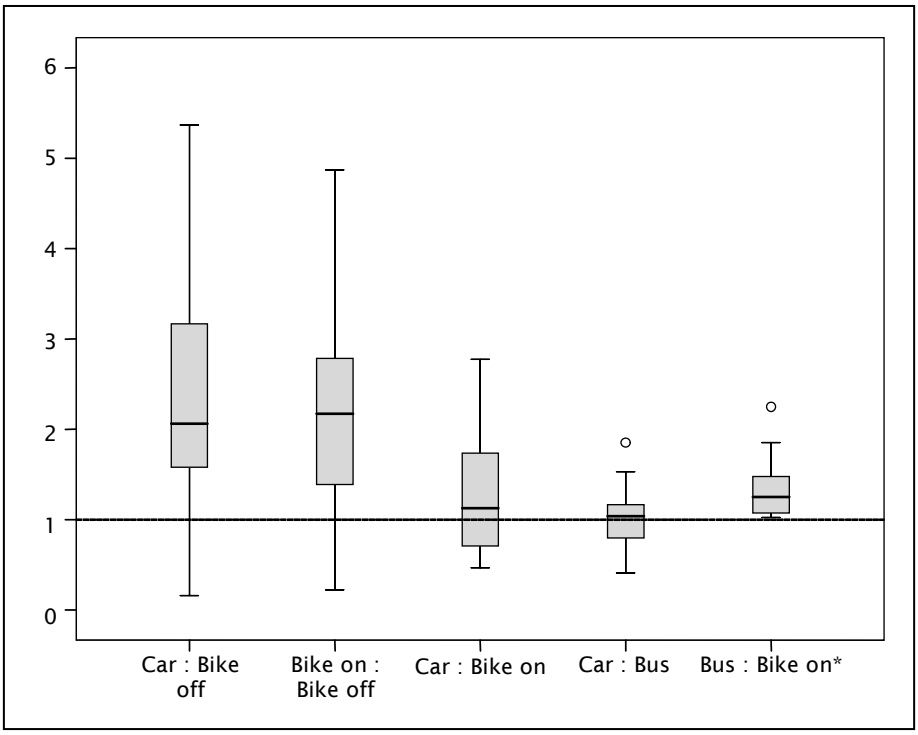
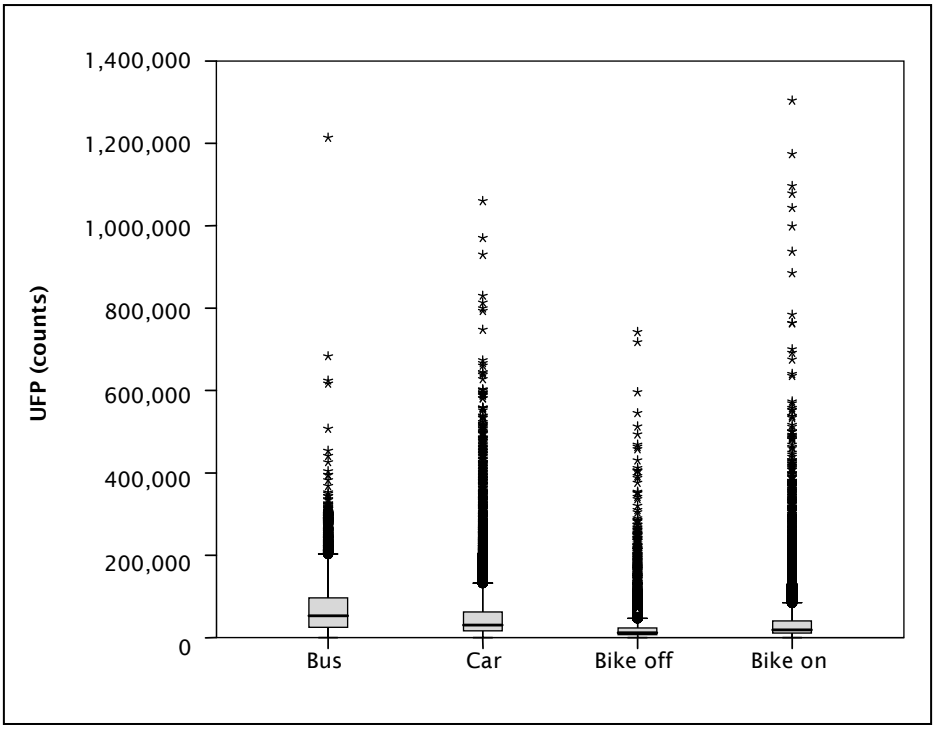


Figure 4.29 Box plot of ratios of UFP levels for trips in Christchurch



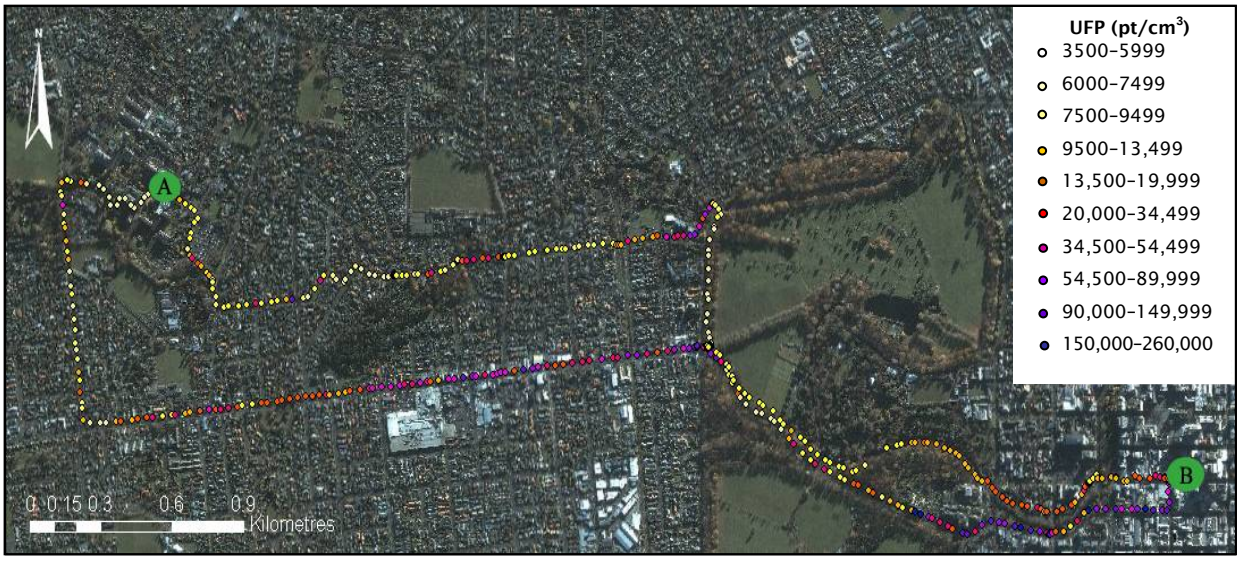
* Bus and off-road bike were never simultaneously measured.

Figure 4.30 Box plot of UFP levels for all data in Christchurch



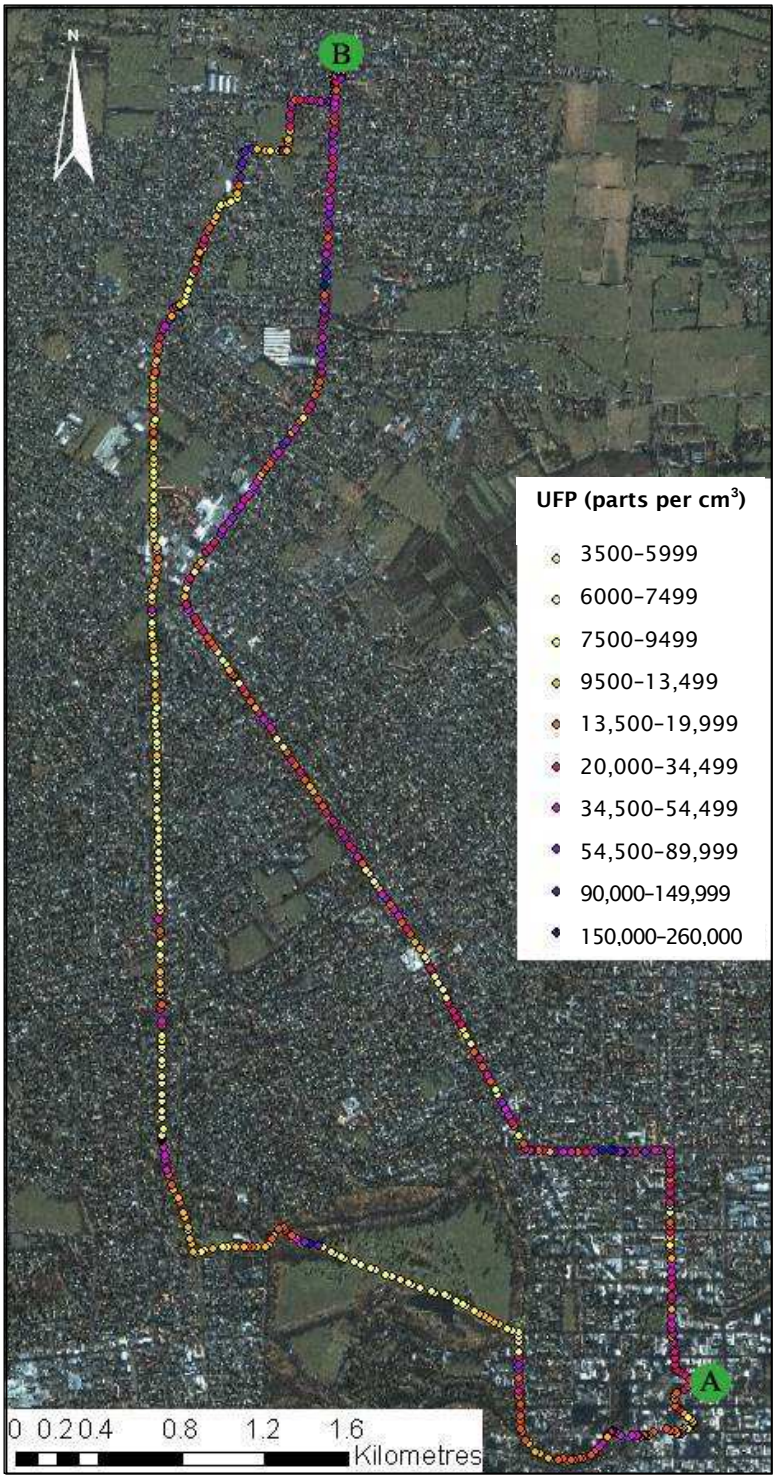
A visual example of the difference between the on- and off-road cyclists can again be seen by mapping the pollution exposure of the two cyclists. Figure 4.31 shows the UFP levels for the two cyclists on 17 March in the evening as they travelled from the University to the city centre. The mean exposure for the on-road cyclist for this trip was a count of 24,256 particles, and off-road a count of 11,1106 particles with an on : off ratio of 2.1. By looking at the map, the difference between the two cyclists can be easily seen. The distinct spatial variation in exposures is very clear, reflecting the effect of proximity to source on levels of UFP. Figure 4.32 shows the second part of that journey from the city centre to Redwood. On this trip, the exposure level was a count of 24,467 particles on-road and a count of 16,230 particles off-road, a ratio of 1.6. Again, the difference between exposures is very clear.

Figure 4.31 Map of on- and off-road cyclists' real-time comparative UFP exposure: University of Canterbury to Christchurch city centre, 4:45–5:05pm, 17 March 2009



These results are supported by the literature, which shows that UFPs are a much better indicator of vehicle-related pollutants than large particles sizes (Kaur 2009). Interestingly, however, although the correlation between UFPs and CO was high during the off-road cycle commutes (0.93), it was low for the on-road cyclist (0.52) and not significant for the car passenger.

Figure 4.32 Map of on- and off-road cyclists' comparative UFP exposure: Christchurch city centre to Redwood, 5:25–6:00pm, 17 March 2009



4.1.7 Summary

A number of main conclusions can be drawn from analysing the modal difference in exposure.

- Car drivers are consistently exposed to the highest average levels of CO: 60% higher than cyclists, 40–100% higher than bus passengers and over 100% higher than train passengers.

- On-road cyclists are exposed to higher levels of CO (10%), PM_1 (25%) and UFPs (over 100%) than off-road cyclists. This could have significant policy implications for the location of cycle routes.
- Car drivers and bus passengers are exposed to higher average levels of UFP than cyclists. However, for very short acute exposures (a few seconds), on-road cyclists were exposed to higher peaks.
- Cycle lanes could reduce pollution exposure for cyclists. The reduction in on-road cyclist peak exposures relative to other on-road modes (bus and car) was less in Auckland, which had few on-road cycle tracks, than in Christchurch, where on-road lanes were more prevalent.
- PM_{10} is a poor indicator of exposure to vehicle emissions. While the exposure levels experienced by the various on-road modes of transport are debateable, it is clear from this detailed dataset that exposure to traffic emissions cannot lead to the consistently higher levels of PM_{10} observed by the off-road cyclist compared to the on-road cyclist. It seems much more plausible that the off-road cyclist is differentially exposed to PM_{10} from other sources, and thus we have to conclude that PM_{10} is not a good indicator of exposure to vehicle emissions. For particulate matter, the finer fractions ($PM_{2.5}$ and PM_1) are a marginally better indicator, while particle number concentrations seem to be the more responsive indicator to fresh exhaust emissions.

4.2 Determinants of exposure

4.2.1 Urban background levels

4.2.1.1 Overview

One factor which will play an important role in determining exposure to pollution during daily commutes is the local urban background concentration. This refers to the level of pollutants observed in urban areas away from (ie in the order of 100m or more) any immediate sources of pollution such as traffic or industrial emissions. It should be distinguished from the regional background, which refers to concentrations upwind of cities arising from long-range and natural emission sources. In urban background locations, we would expect pollutant concentrations to be well mixed and relatively homogeneous in space and lower than in areas closer to the sources of pollutants. In urban areas that have higher total emissions, we would expect the urban background concentrations to be higher. Thus we might expect that urban background concentrations of pollutants would be higher in Auckland compared to Christchurch because of Auckland's greater size. (This pattern could be reversed in winter with the more prevalent use of woodburning for domestic heating in Christchurch; however, this study project was designed to avoid the woodburning season in both cities.) The urban background concentrations are relevant to this study, as they provide the baseline on top of which local emissions from transport are evaluated.

Fixed air quality monitoring stations (FAQMSs) provide a good estimate of background concentrations, as long as they provide good coverage of the spatial variability of each pollutant within the city. By selecting the data from the FAQMSs for the periods when personal exposure data was collected, it is possible to improve our understanding of the spatial variability of pollutant concentrations and to ascertain the importance of local emissions in determining exposure on each transport mode. Different pollutants are likely to display different trends in time and space.

It is also possible to estimate background concentrations from the mobile transect data collected during the field campaign. For example, during parts of the journey, the cyclist is likely to be travelling through areas which could be considered representative of background concentrations. One method of isolating these periods is to look at the parts of the journey where the cyclist is exposed to the lowest concentrations. These can be identified visually for each time series; alternatively, the lowest 5% or 10% of

the dataset can be considered to be representative of background concentrations. These methods were compared and no significant difference was observed between the techniques.

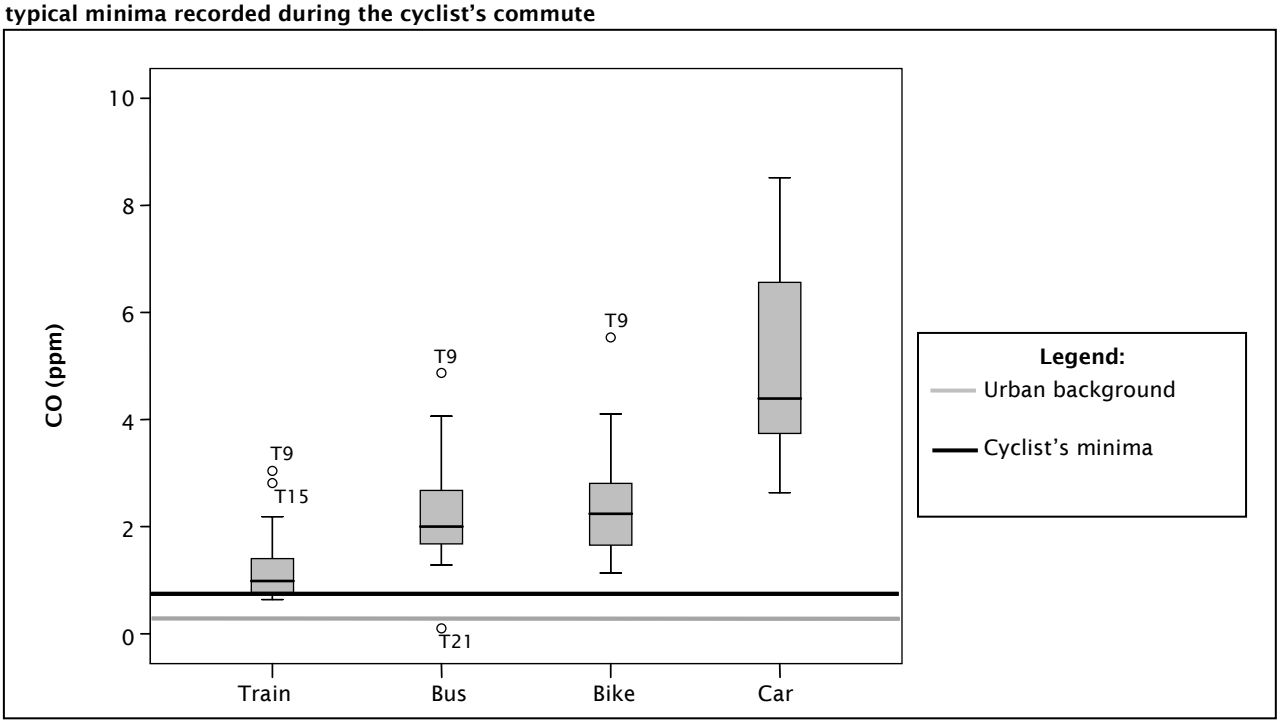
4.2.1.2 Carbon monoxide

In Auckland, three urban background sites were chosen: Pakuranga, Glen Eden and Takapuna. The closest urban background site to the study was Glen Eden. The average concentration of CO recorded by FAQMSs located at urban background sites during the exposure study ranged from 0.26ppm to 0.46ppm. Clearly, a large degree of spatial variability can be found within the Auckland study area. The overall background value reported from the cyclist data was up to 85% higher (0.75ppm), while the mean exposure of the cyclist was significantly higher at 2.28ppm.

These different levels are illustrated in figure 4.33. The grey horizontal line (urban background) represents the contribution of diffuse CO sources that are distributed across Auckland. Traffic could still be a major contributor to this level, but the emissions cannot be apportioned to any particular road. The black line represents the cyclist's exposure en route during periods without close encounters with individual vehicles, eg in gaps in the traffic. This probably represents recent (but not immediate) emissions from traffic sources on that road at their maximum level of dilution. The actual exposure of our commuters represents an elevation above the black line arising from close encounters with individual vehicles or streams of vehicles.

Thus it can be seen that the train commuter was exposed almost entirely to background and diffuse CO sources (more detailed analysis reported in Shrestha et al (2010) has shown that most of the peaks in the 'train' exposure occur before and after boarding the train). The contribution to the mean exposure of the bus and cycle commuters consists of approximately 15% background source, 20% diluted on-road sources and 65% near-field vehicle sources. The car commuter's exposure has a much greater contribution from near-field vehicles. This is consistent with the hypothesis discussed elsewhere that the enclosed cabin of the car accumulates emissions from the vehicles it follows. The cyclist does not experience this 'trapping' but the bus commuter occupies a larger cabin volume and spends part of their journey outside of the cabin.

Figure 4.33 Box plot of CO levels for trips in Auckland, showing typical urban background CO levels and the typical minima recorded during the cyclist's commute



Although the magnitude of the exposure was poorly represented by the background FAQSMs, a strong correlation could be found between both calculated background concentrations observed during the cycle commute and total trip exposure (R = 0.73–0.83). Interestingly, although a poor correlation could be seen between CO concentrations observed at the kerbside FAQMS in Queen Street and exposure during the commute (R = –0.13), the magnitude of the mean exposure was predicted more accurately. This is consistent with a number of exposure studies which report that urban background FAQMSs generally underpredict exposure in the transport microenvironment (Adams 2001; Duci et al 2003; Kaur et al 2005a and 2005b; Zagury et al 2000). Previous studies report an improved correlation between transport exposure and FAQMSs located at kerbside sites (Kaur et al 2005a). However, in this example, the Queen Street FAQMS is located in an area which has quite different urban morphology (taller buildings which flank both sides with few gaps) and traffic flow patterns compared to those experienced on the commuting route.

In Christchurch, data were available from two urban background FAQMSs: St Albans and Burnside. Both are located near the study area, although St Albans is closer to the city centre. The average concentration of CO recorded by the FAQMSs ranged from 0.21ppm to 0.38ppm during the exposure studies. Again, the background concentrations observed by the (on-road) cyclist were higher than those of the FAQMSs by 0.32ppm and the mean exposure by 1.13ppm. Interestingly, although a poor correlation between the urban background FAQMSs and both background and mean exposures observed by the on-road cyclist (R = 0.23–0.44), the concentrations observed by the off-road cyclist showed a much better correlation (R = 0.711–0.822). This suggests that proximity to traffic plays an important role in determining exposure, as discussed in later sections. A kerbside FAQMS was not available in Christchurch.

Unfortunately, it was not possible to sample the exposure to pollutants during different transport modes in both Auckland and Christchurch simultaneously. This makes it very difficult to compare the mean exposure between cites, as different weather conditions affect the local background concentrations through their influence on the stability of the atmosphere (pollutant dispersion) and pollutant emissions (cold starts generate higher traffic emissions and could result in increased emissions from other sources such as space heating). However, by using the urban background concentrations to normalise the data, it is possible to determine the differences in exposure between the two cities. This process is complicated by the unexpected spatial variability in CO concentrations in Auckland. Nevertheless, it can be seen that although the urban background concentrations are only slightly raised in Auckland compared to Christchurch, the exposure during the commute is more than doubled. This suggests that local traffic emissions play a much more significant role in determining on-road cyclist exposure in Auckland than they do in Christchurch. This is likely to be a product of the density of the traffic flows.

4.2.1.3 Particulate matter

In Auckland, FAQMS data was available for PM_{10} from Glen Eden and Takapuna. As noted earlier, concentrations of PM_{10} were much more spatially homogeneous in Auckland compared to CO and no statistical difference between the background concentrations reported by the cyclist (17.04µg/m³) and the FAQMSs were reported. However, the FAQMSs underpredicted the mean exposure (26.46µg/m³). The correlation between the background transport exposure and the Glen Eden and Takapuna FAQMSs were R= 0.508 and 0.713, respectively. The FAQMS data was not correlated with mean exposure during the commute. This is consistent with the argument that PM_{10} is a poor indicator for exposure to traffic-related pollution.

FAQMS measurements of $PM_{2.5}$ were not available at Glen Eden. Again, the mean concentrations observed at the FAQMSs and the background concentrations observed during the bike commute were not significantly different and a poor correlation was observed between the datasets.

In Christchurch, mean background exposure during the bike commute (25.61µg/m³) was very similar to that observed at the Burnside FAQMS but no significant correlation was reported between the datasets. Again, the FAQMS underpredicted the mean exposure (36.54µg/m³). It is interesting to note that, unlike CO concentrations, the magnitude of the increased exposure in the transport microenvironment observed in Christchurch can be explained by increased background concentrations that are not attributable to traffic.

4.2.1.4 Summary of FAQMS results

In summary, care must be taken when using FAQMSs to predict personal pollutant exposure in the transport microenvironment. In Auckland, although background and mean transport exposure to CO is well correlated with the FAQMS data, the magnitude of the exposure is significantly underpredicted. In Christchurch, the magnitude of the background exposure is correctly predicted by the FAQMS data, but mean exposure is underpredicted. Further temporal variations in CO show a poor correlation between the transport microenvironment and FAQMSs. For PM_{10} and $PM_{2.5}$, FAQMSs correctly predict the magnitude of background exposures in both Auckland and Christchurch, although they underpredicted mean exposures. Although a weak correlation between transport exposures and FAQMS data was reported for PM_{10} in Auckland, all other correlations were poor. These results indicate that it is important to take the pollutant type into consideration, and also the spatial variation within a city and local conditions, before trying to predict pollutant exposure in the transport microenvironment from urban background FAQMSs. It also highlights the differences between the two cities. In Auckland, the influence of traffic on CO concentrations could be detected over and above increased background concentrations, whereas in Christchurch, traffic signals could not account for increased background, mean or transportation exposure.

4.2.2 Exposure and proximity to traffic

This section reports the results where three cyclists simultaneously cycled the same route, one on the road, one on the pavement and the third on a separated cycle path parallel to but away from the road (see section 3.3.4 for more details).

Changes in levels of CO, PM_1 and UFPs with increasing distance from traffic are presented for Christchurch (table 4.1) and Auckland (table 4.2). Average values (whether expressed as means or medians) decrease significantly as the cyclist is located further from the traffic. For example, in Christchurch (table 4.1), mean exposure for the cyclist cycling on the pavement was 78% that of the on-road cyclist (74% using median values) and the CO exposure for cyclist completely off the road was under half the level of the on-road cyclists (46% using mean and 42% using median values). Similar but less dramatic reductions are shown for PM_1 and UFPs. In Auckland, the reductions are as consistent: the cyclists away from the road see reductions in exposure of 20–40%; for CO, however, the footpath seems to afford as much pollution protection as riding further from the road.

Overall, it can be concluded that even short distances from traffic sources seem to reduce pollution concentrations and ultimately pollution exposure quite significantly. The nature of the method adopted here means these results apply for non-enclosed modes of transport, predominantly cyclists, as demonstrated here, but are probably also applicable for pedestrians. The results indicate significant potential health benefits from locating such active modes even just a few metres away from motor vehicles.

Table 4.1 Cycle runs in proximity to the road in Christchurch: descriptive statistics

Pollutant	Route	*N* legs (samples)	Mean	Standard deviation	Min	Max	Median	Mean % of on-road cyclist	Median % of on-road cyclist
CO	On-road	153 (3106)	1.19	0.75	0.05	12.8	1.00	100	100
	Footpath	153 (3101)	0.93	0.79	0.05	7.14	0.74	78	74
	Off-road	153 (3114)	0.55	0.45	0.05	3.66	0.42	46	42
PM_1	On-road	169 (4023)	5.61	3.75	1.4	37	4.40	100	100
	Footpath	145 (2913)	5.13	2.92	1.5	30.5	4.68	91	78
	Off-road	157 (3711)	3.80	2.28	0.4	18	3.97	68	61
UFP	On-road	146 (3597)	43,450	61,034	1,091	1,588,280	29,495	100	100
	Footpath	52 (1044)	30,235	38,518	4,320	490,908	19,440	70	63
	Off-road	157 (3711)	25,014	32,191	185	1,149,812	22,155	58	53

Table 4.2 Cycle runs in proximity to road in Auckland: descriptive statistics

Pollutant	Route	*N* legs (samples)	Mean	Standard deviation	Min	Max	Median	Mean % of on-road cyclist	Median % of on-road cyclist
CO	On-road	78 (2295)	0.75	0.30	0.39	3.58	0.68	100	100
	Footpath	78 (2297)	0.55	0.17	0.23	3.14	0.55	73	57
	Off-road	78 (2297)	0.48	0.19	0.17	1.21	0.52	64	63
PM_1	On-road	78 (2293)	5.36	2.49	2.23	26.30	5.30	100	100
	Footpath	78 (2292)	4.35	2.04	1.56	12.77	4.73	81	82
	Off-road	78 (2291)	4.04	1.84	1.40	6.98	4.79	75	74

4.2.3 Peak exposures

In both Auckland and Christchurch, intermittent spikes in CO and UFP concentration were observed in the time series recorded during journeys by bike, bus and car. These were highly variable in both maximum exposure concentration and duration, lasting from a few seconds to over five minutes. (Other pollutants were not measured at a high enough temporal resolution for such short-scale events to be detected.)

Peaks in CO and UFP (figures 4.34–4.39 and table 4.3) were often experienced simultaneously by the cyclist. Detailed analysis of the trip photos showed that the majority of these events occurred at intersections. Cycling (or waiting) behind a diesel vehicle was also identified as a common cause of pollutant spikes, while general congestion accounted for a much smaller percentage. As this study took samples in the morning and evening rush hours, buses were more prevalent than trucks; consequently,

proximity to diesel-emitting buses was of greater significance than diesel-emitting trucks. This does not mean that diesel trucks are not significant; it merely reflects the timing of this exposure-based study.

Figure 4.34 Identification of localised spikes on personal exposure to CO using time–activity exposure profiles in the morning of 13 March as observed by the on-road cyclist in Christchurch

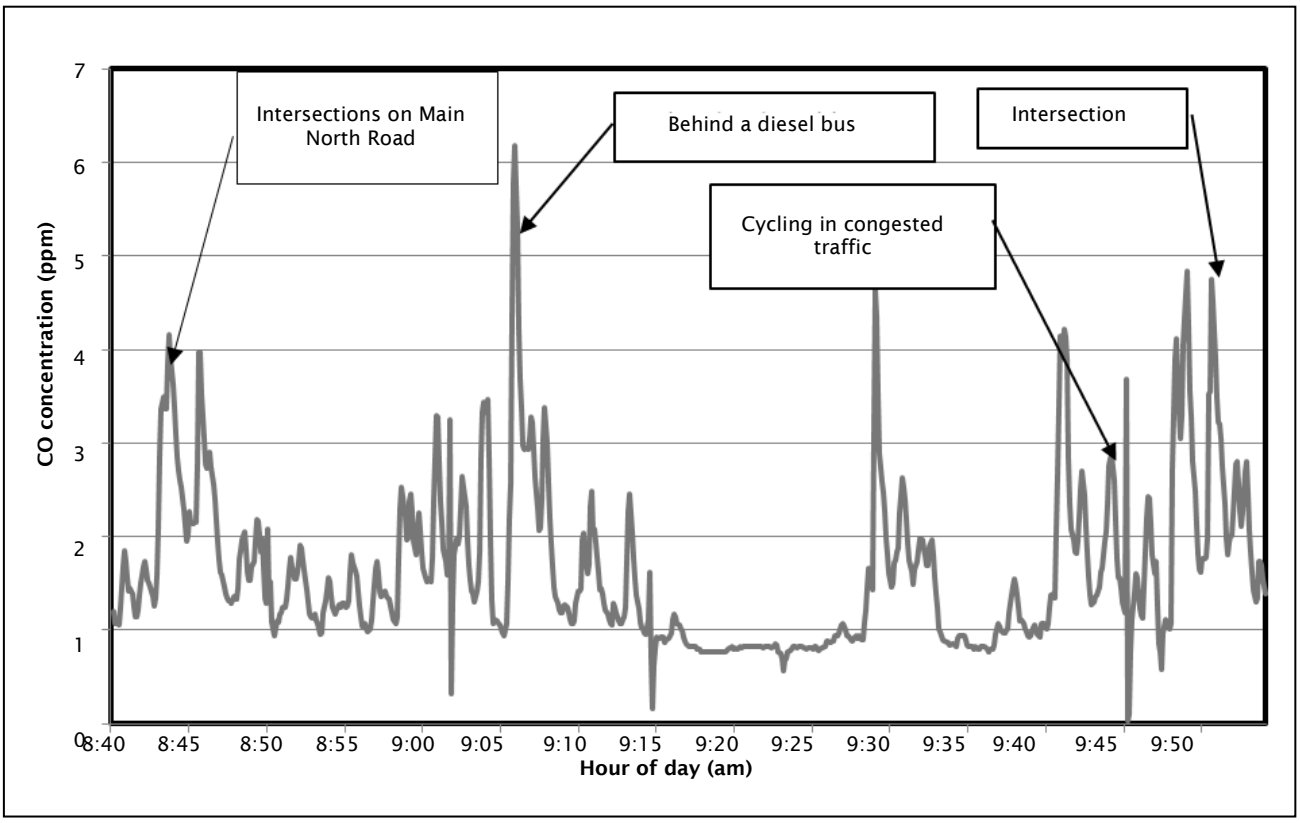


Figure 4.35 Identification of localised spikes in personal exposure to UFP counts using time-activity exposure profiles in the morning of 13 March as observed by the on-road cyclist in Christchurch.

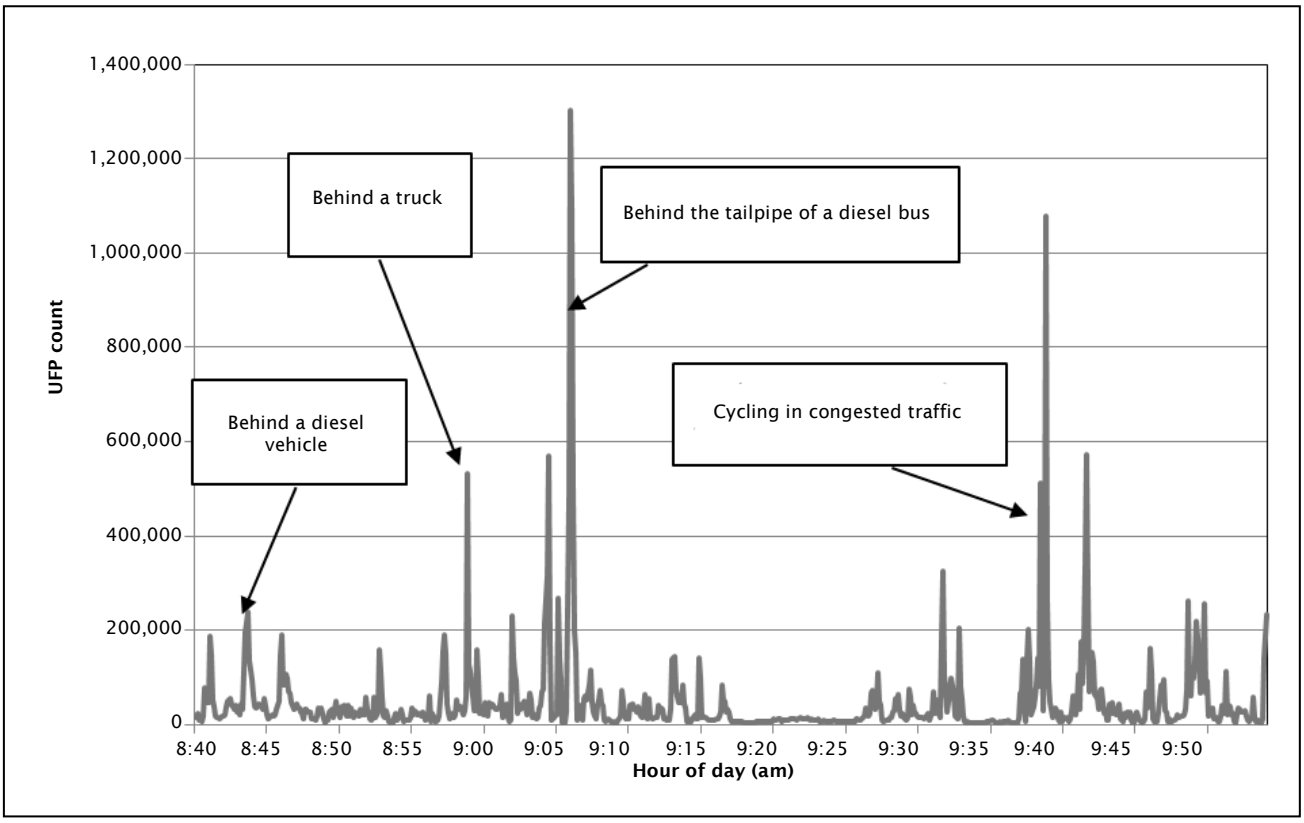


Table 4.3 Number of peaks during journeys in Christchurch

		Number of peaks					
		Behind a diesel vehicle		At intersection		Traffic congestion	
		CO	UFP	CO	UFP	CO	UFP
On-road cyclist*	am	16	16	20	21	3	3
	pm	7	11	21	19	1	1
Car**	am	1	3	9	5	-	-
	pm	3	3	7	6	-	-

*A total of eight journeys were analysed (four in the morning and four in the afternoon).

** A total of eight journeys were analysed (four in the morning and four in the afternoon).

Figure 4.36 Frequency of casual factors associated with peaks in CO during on-road cycle journeys in Christchurch

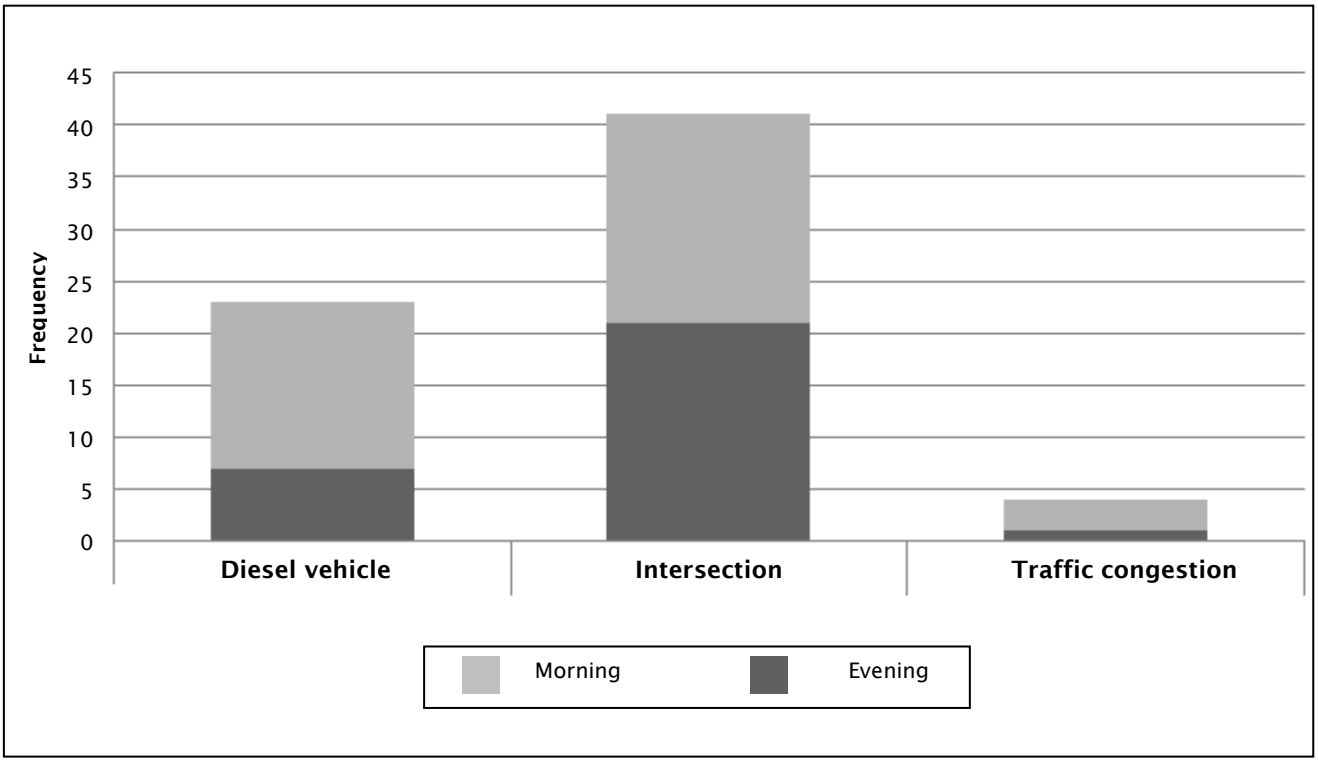


Figure 4.37 Frequency of causal factors associated with peaks in personal exposure to UFP counts during on-road cycle journeys in Christchurch

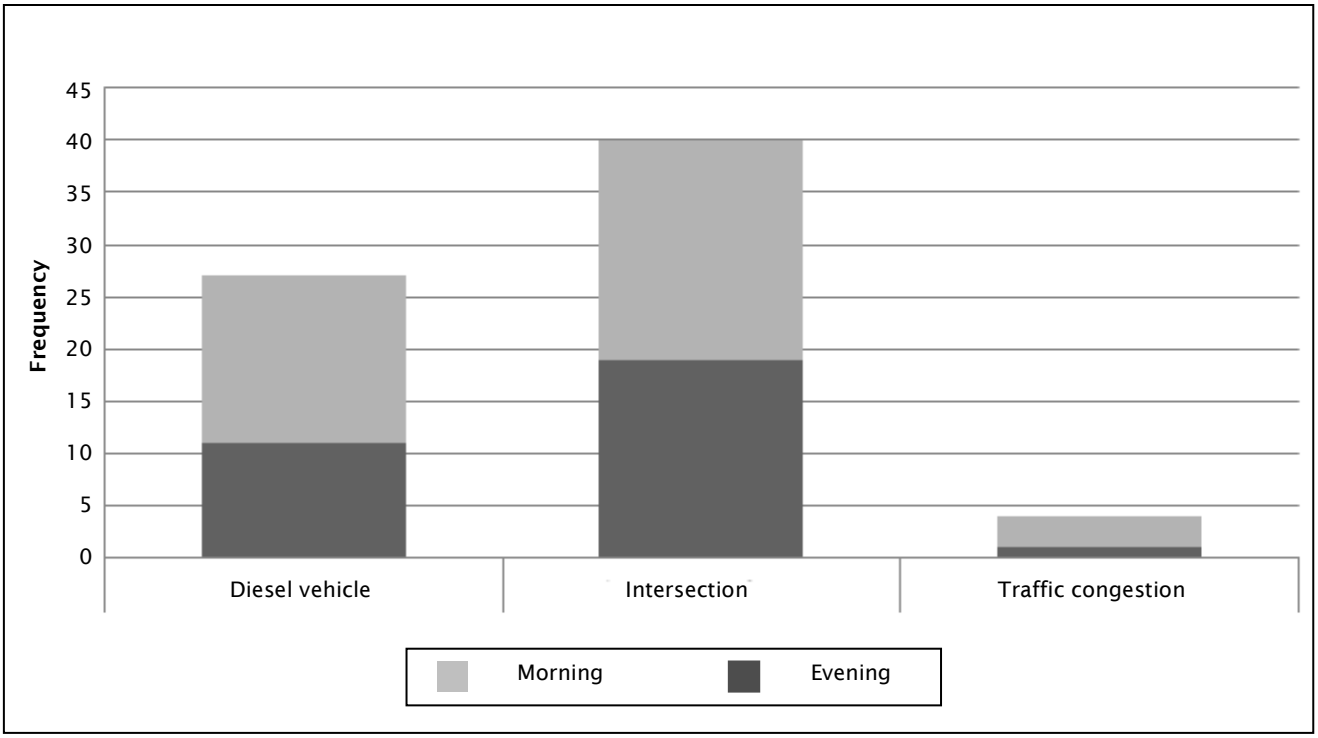


Figure 4.38 Frequency of casual factors associated with peaks in personal exposure to CO during car journeys in Christchurch

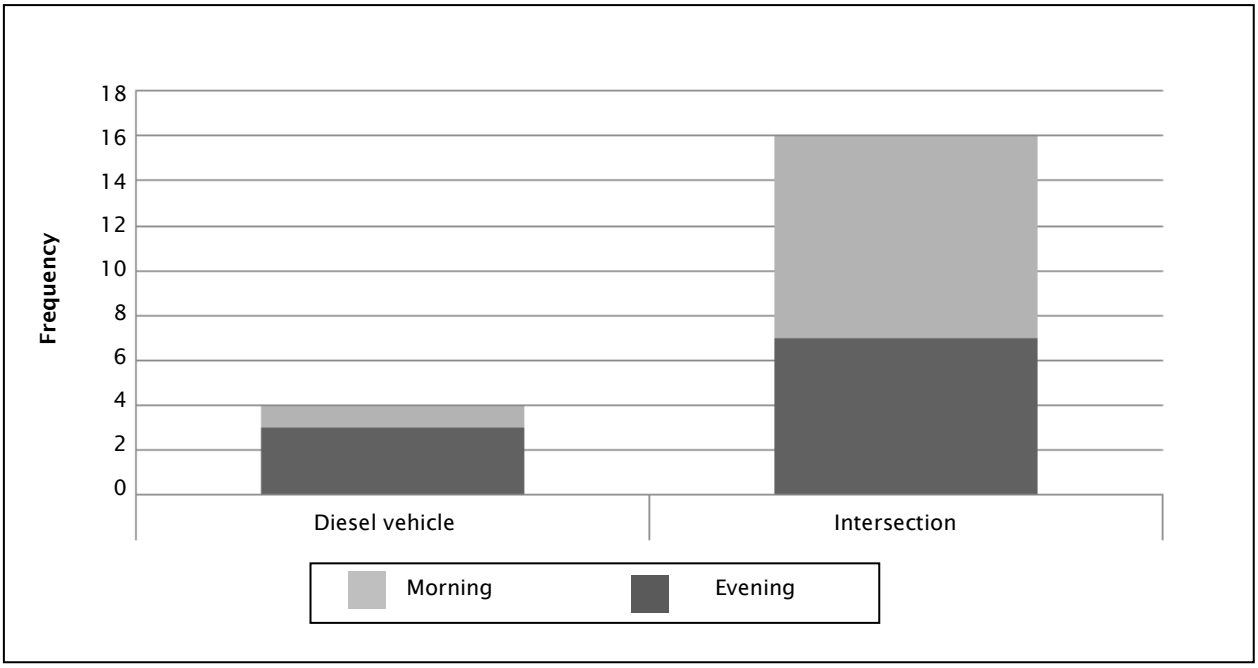
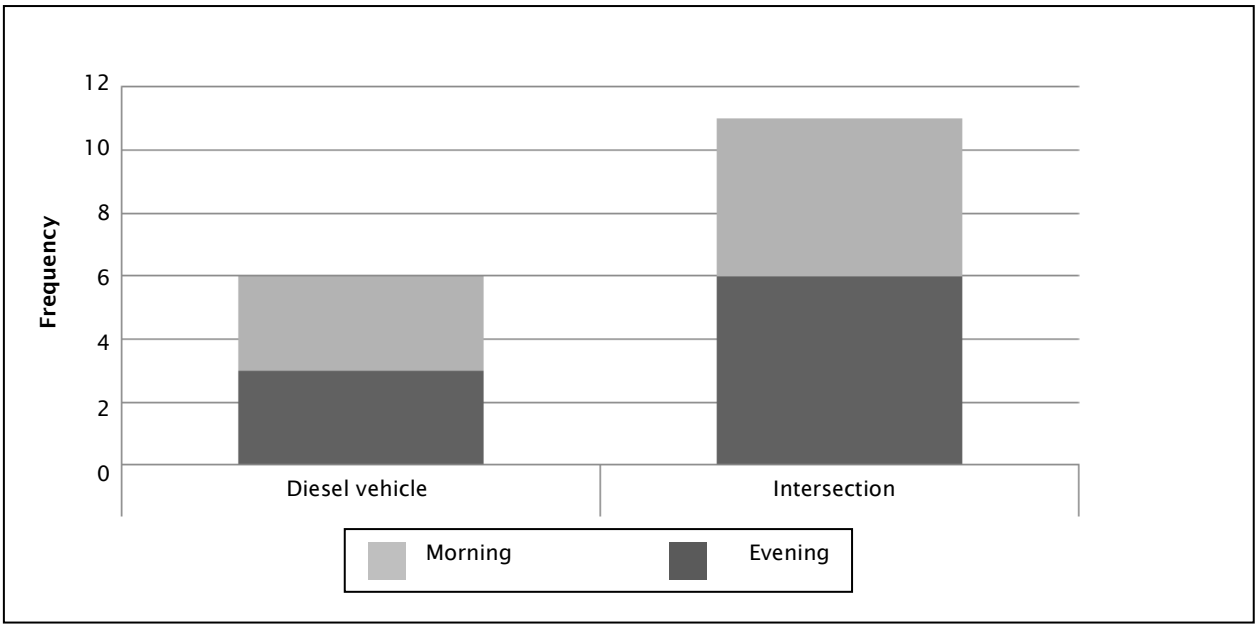


Figure 4.39 Frequency of causal factors associated with peaks in personal exposure to UFP during car journeys in Christchurch



Spikes in pollutant concentration observed during car and bus journeys were typically lower in concentration, with a longer duration than those measured by the cyclist. This is consistent with polluted air entering the cabin and then remaining because of slower dilution rates. Interestingly, peaks in CO and UFP were rarely coincident during bus and car journeys, suggesting either:

- different causal mechanisms
- a degree of filtering occurring as the air entered the cabin for one or both pollutants

- differential instrument response times.

The details of these mechanisms are being investigated in current research (partly funded by the NZTA) by the NIWA team. Again, travel through or waiting at intersections accounted for the highest proportion of spikes, while trailing diesel vehicles accounted for a significant proportion of the remainder. This is consistent with previous studies (eg Clifford et al 1997; Duci et al 2003; Kaur et al 2006; Behrentz et al 2004). Overall, car and bus passengers were less likely than the cyclist to be exposed to the spikes in pollutant concentration resulting from following a diesel vehicle. During bus journeys, significant spikes in pollution were also observed when the bus was stationary at a stop with the doors open. This has been observed in previous studies (eg Chan et al 2002; Kaur et al 2005b) and is accounted for by the increased penetration of vehicle emissions from stationary buses in front.

Detailed analysis of where the spikes in pollution concentration took place revealed the presence of a number of 'hotspots' in the Christchurch area (figure 4.40 and 4.41; table 4.4). These were remarkably consistent for both CO and UFP pollution, and provide an indication of where improved traffic management could result in lower pollutant exposure. Berghmans et al (2009) showed that the consistent identification of such hotspots could be achieved with relatively few data samples, which has important implications for future air quality management strategies.

Table 4.4 Frequency of 'hotspots' of personal exposure to CO and UFP during bike (on-road) journeys identified at different locations in Christchurch

Hotspot frequency*	A**	B	C	D
CO hotspots	5	4	5	4
UFP hotspots	5	4	4	3

*A total of eight journeys were analysed, four journeys in the morning and four in the afternoon.

** A, B, C and D refer to the locations of hotspots as shown in figures 4.40 and 4.41.

Figure 4.40 Map of hotspot locations (both on Papanui Road) along the Redwood to city centre route in Christchurch, indicating where commuters may experience significantly higher exposures to air pollution than elsewhere

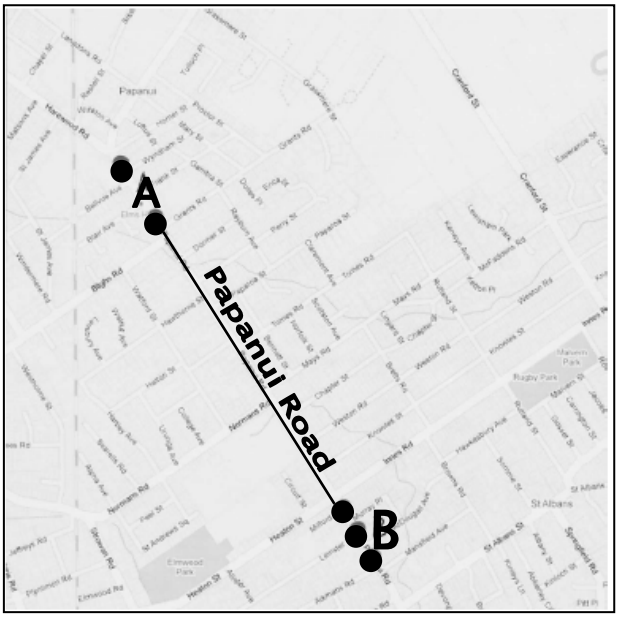
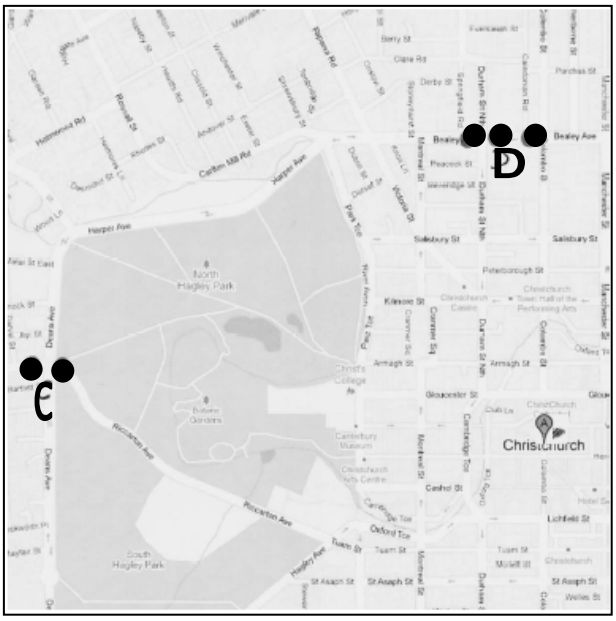


Figure 4.41 Map of hotspot locations along commuting route near the city centre in Christchurch, indicating where commuters may experience significantly higher exposures to air pollution than elsewhere



4.3 Journey segments

4.3.1 Subdividing the journeys

Each trip by each mode was subdivided into a series of journey legs to home in on the segments of the journey that created the most exposure to pollutants. These segments were:

- travelling
- waiting at start or end of the journey
- waiting at stops or stations (bus and train passenger only, differentiated between indoor and outdoor)
- walking through car-parks (car driver only).

Each trip started and ended with a short waiting period while all the travellers prepared to start, or finished the journey and waited for the other mode users to arrive. The main part of the journey consisted of a period of travelling. In addition, the motorised modes included a section of the journey waiting at or passing through a bus stop, train station or car-park. For the car driver, this was walking through an enclosed car-park; for the bus passenger in Christchurch, it included being at an outside bus stop and an enclosed bus station; for the Auckland bus passenger, it was waiting at an outside bus stop; and for the Auckland train passenger, it involved being at an outside train station and an inside train station (Britomart).

The natures of the different environments could potentially create different levels of exposure. Figures 4.42 to 4.52 show box plots of the various pollutants for Auckland and Christchurch for the motorised modes subdivided into the various legs of the journey, and descriptive statistics are presented in appendix A.

4.3.2 CO

For CO, the Christchurch car driver (figure 4.42), while exposed to an overall mean level of 1.6ppm, was exposed to some very high three-second peak levels (over 50ppm) during both the travel part of the trip and in the car-park, compared to the bus passenger (figure 4.43) (mean 1.9ppm), for whom the peak did not reach 10ppm. This suggests that the bus and the bus station afford greater protection from CO than the car or the car-park. In both cases, the median value for the travelling component of the journey is significantly higher than in the enclosed car-park or bus station (appendix A). A similar situation can be seen in Auckland, although the peaks are much lower (figures 4.44 and 4.45) even through the average levels are higher (appendix A). The lower peaks in the Auckland car-park could be possibly partly attributable to this car-park being a private work car-park, whereas the one in Christchurch is a public car-park that probably has a greater volume of cars. The levels for train are lower, with fewer peaks, especially in the new enclosed Britomart station (figure 4.46).

Figure 4.42 Box plot of car driver CO levels for journey legs in Christchurch

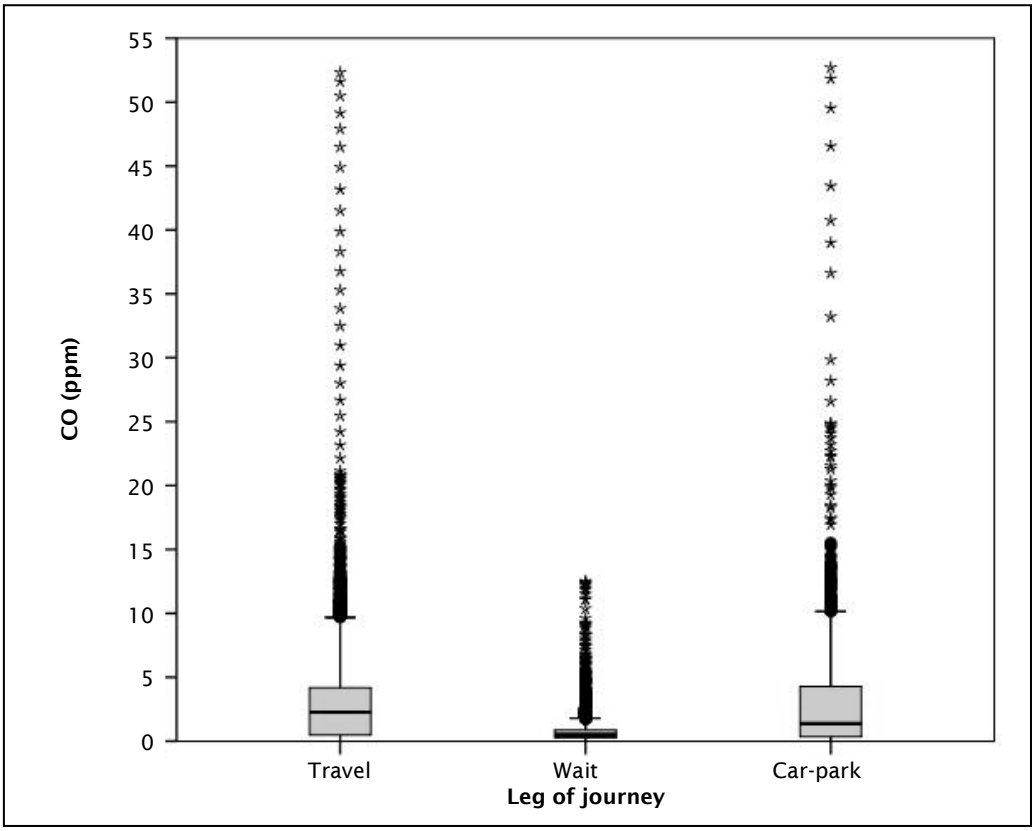


Figure 4.43 Box plot of bus passenger CO levels for journey legs in Christchurch

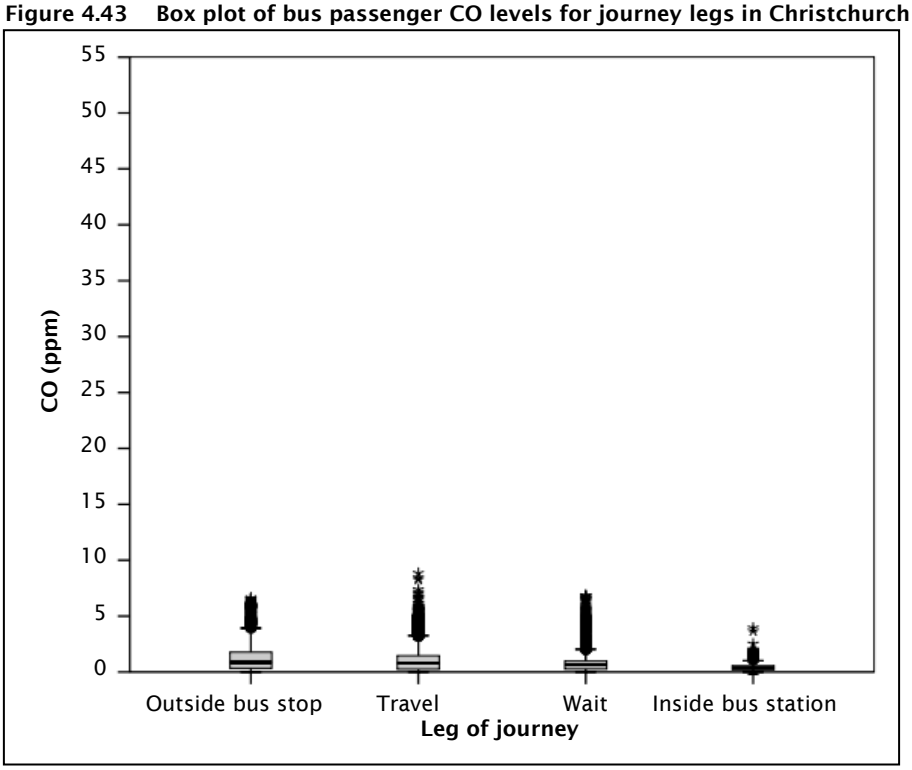


Figure 4.44 Box plot of car driver CO levels for journey legs in Auckland

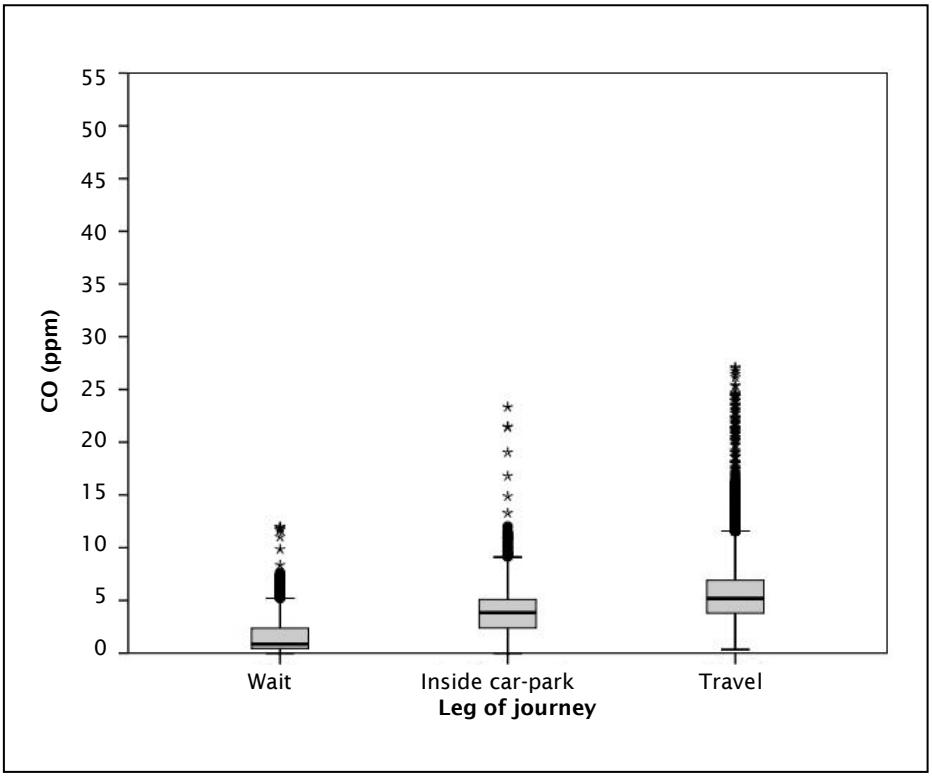


Figure 4.45 Box plot of bus passenger CO levels for journey legs in Auckland

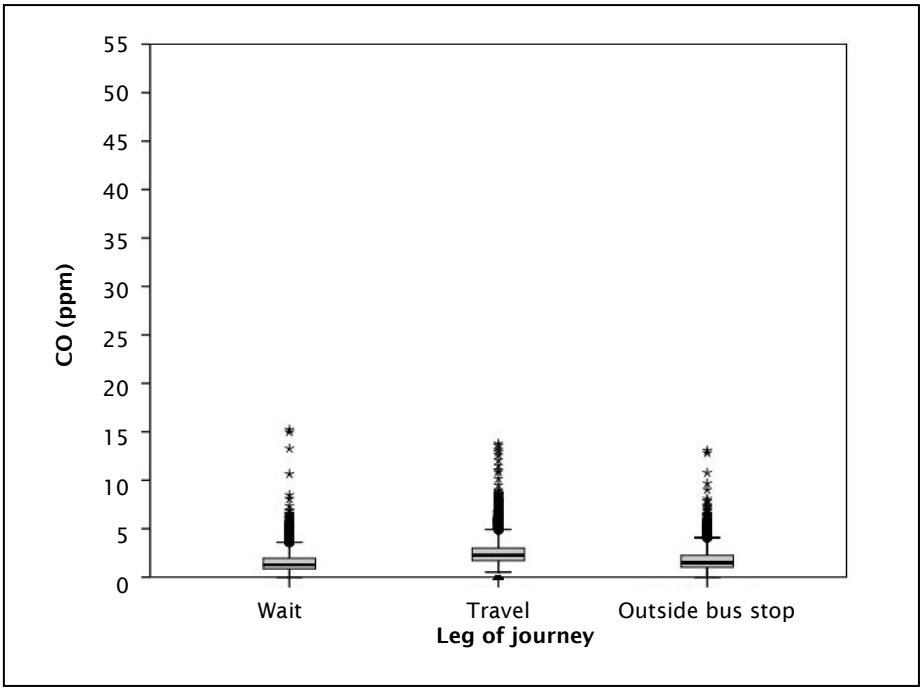
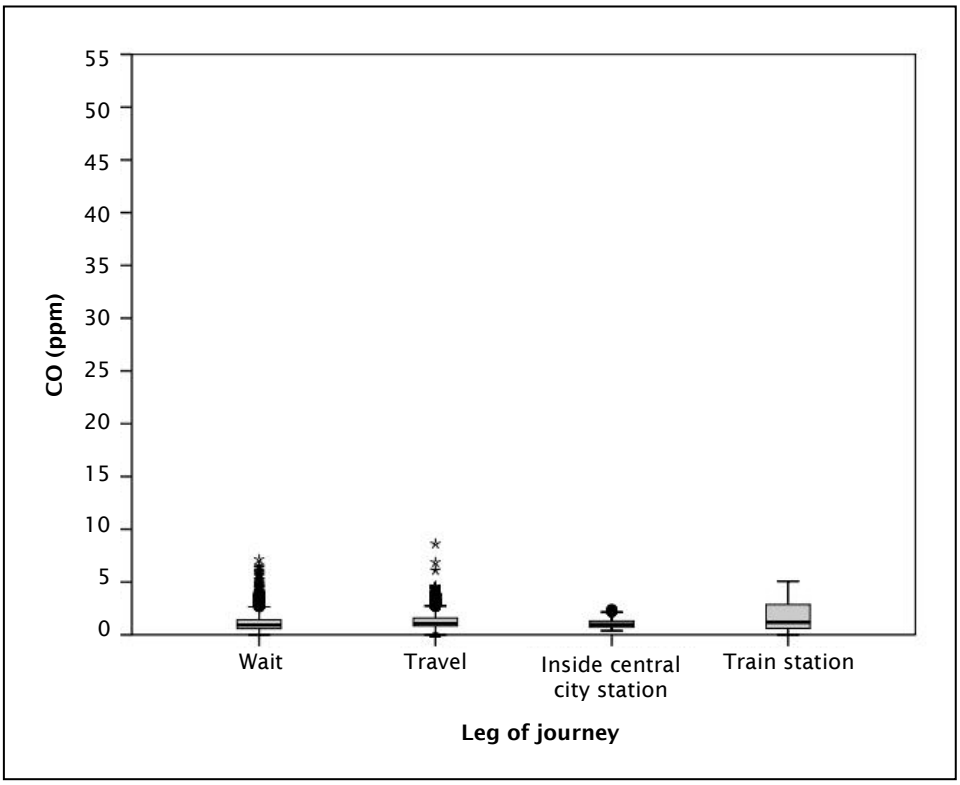


Figure 4.46 Box plot of train passenger CO levels for journey legs in Auckland



4.3.3 Particulate matter

For PM_1 in Christchurch, the travel part of the journey is, on average, a part of the journey with lower levels of this pollutant (appendix A), but has the most extreme peak values for both modes (figures 4.47 and 4.48), especially for the bus passenger. The high peaks are repeated for the Auckland car driver (figure 4.49), with the peak values exceeding 140µg/m^3, although the median levels are lower than for the bus passenger. The bus passenger is also exposed to high short peak values while travelling but also while waiting at the outside bus stop (figure 4.50).

Figure 4.47 Box plot of car driver PM_1 levels for journey legs in Christchurch

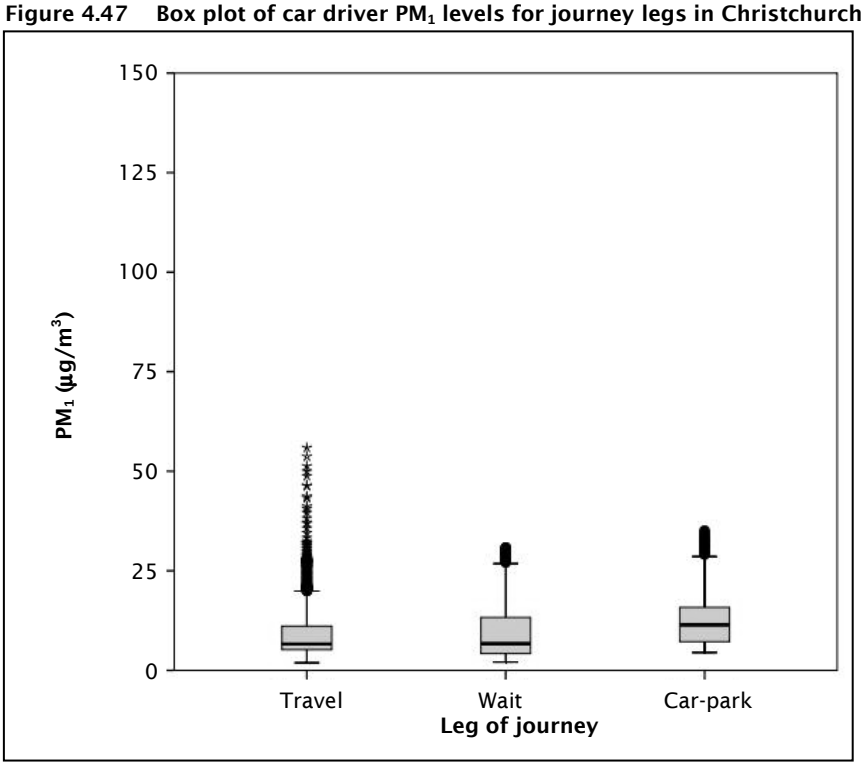


Figure 4.48 Box plot of bus passenger PM_1 levels for journey legs in Christchurch

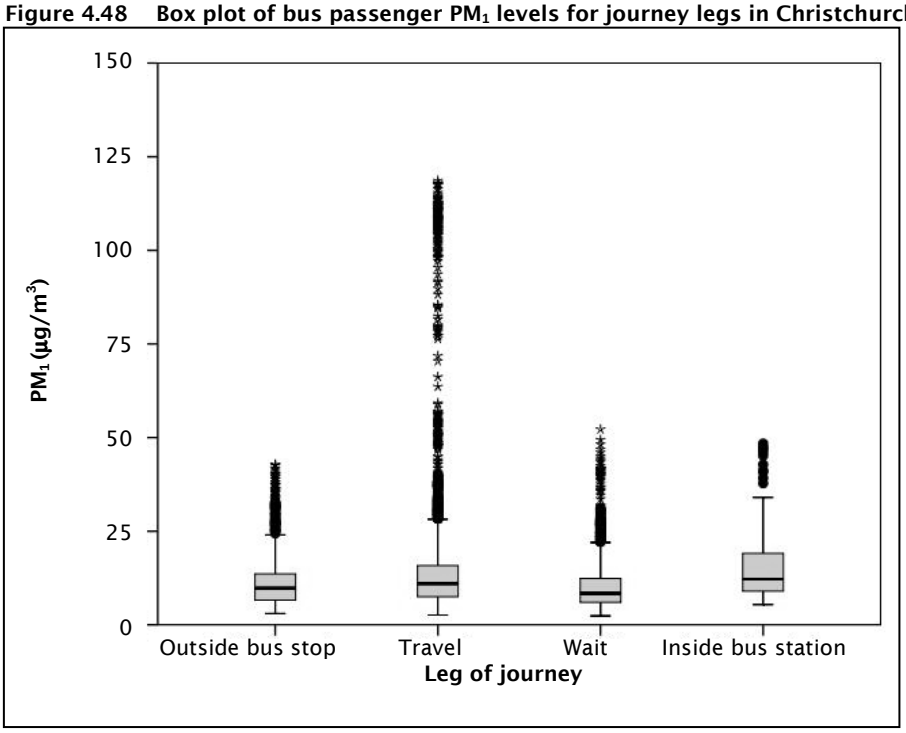


Figure 4.49 Box plot of car driver PM_1 levels for journey legs in Auckland

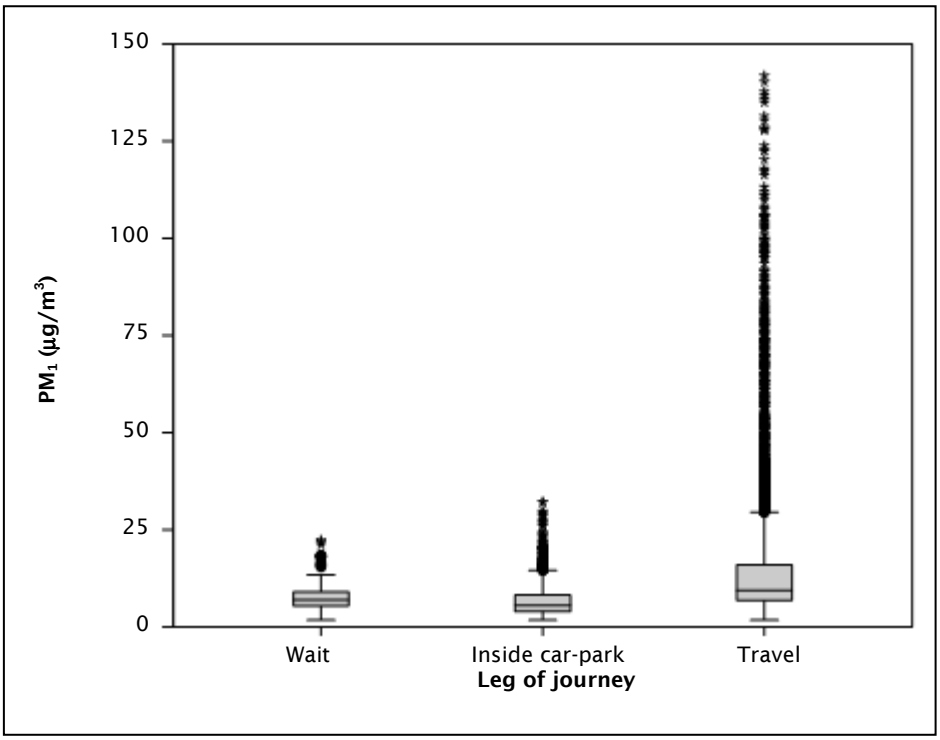
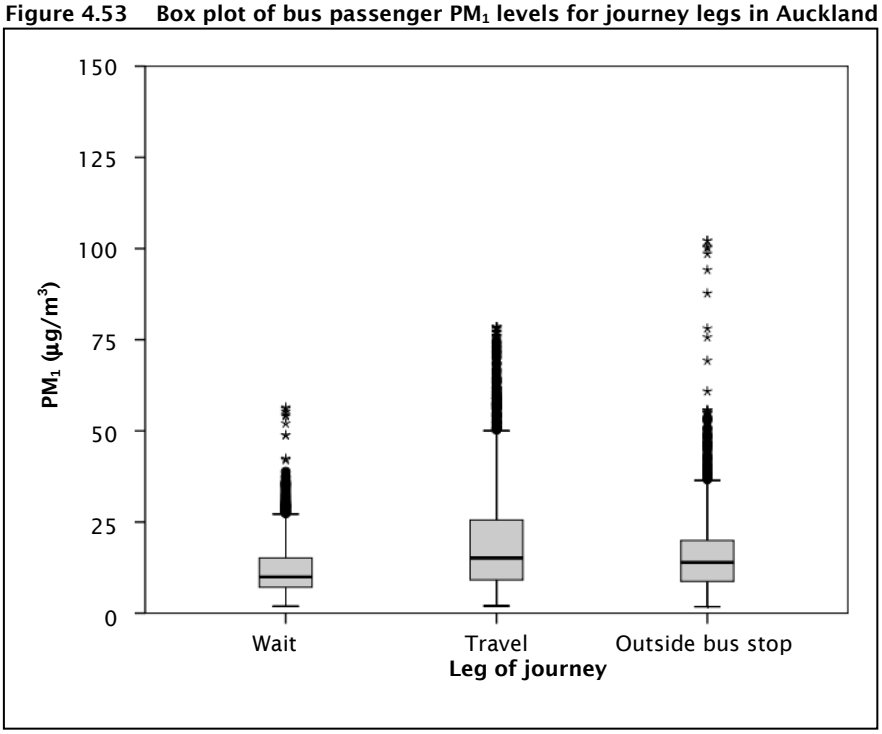


Figure 4.53 Box plot of bus passenger PM_1 levels for journey legs in Auckland



4.3.4 UFPs

For UFP (data only available in Christchurch), we seem some interesting values (figures 4.51 and 4.52). For the car driver, the travel component has the highest median value (appendix A) and a number of high extreme exposures, as does the car-park. The bus passenger is exposed to a median higher particle count than the car passenger (45,000 v 36,000; appendix A), but sees higher levels while in the bus station (median 91,000) than while travelling (64,000) or at the bus stop (22,000). This high average at the bus station is interesting, as it includes few peak extreme values (figure 4.52).

Figure 4.51 Box plot of car driver UFP levels for journey legs in Christchurch

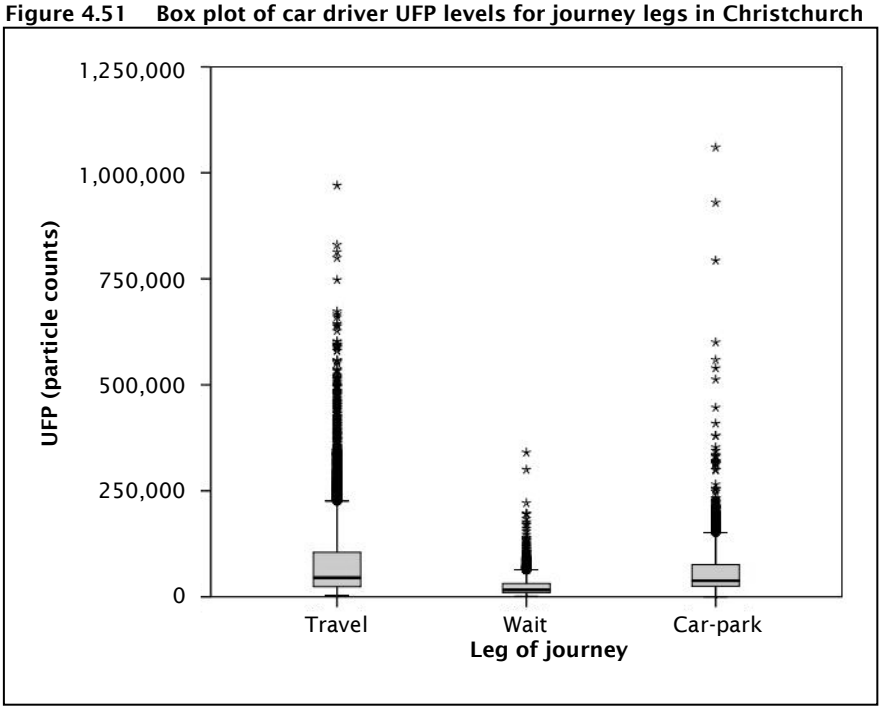
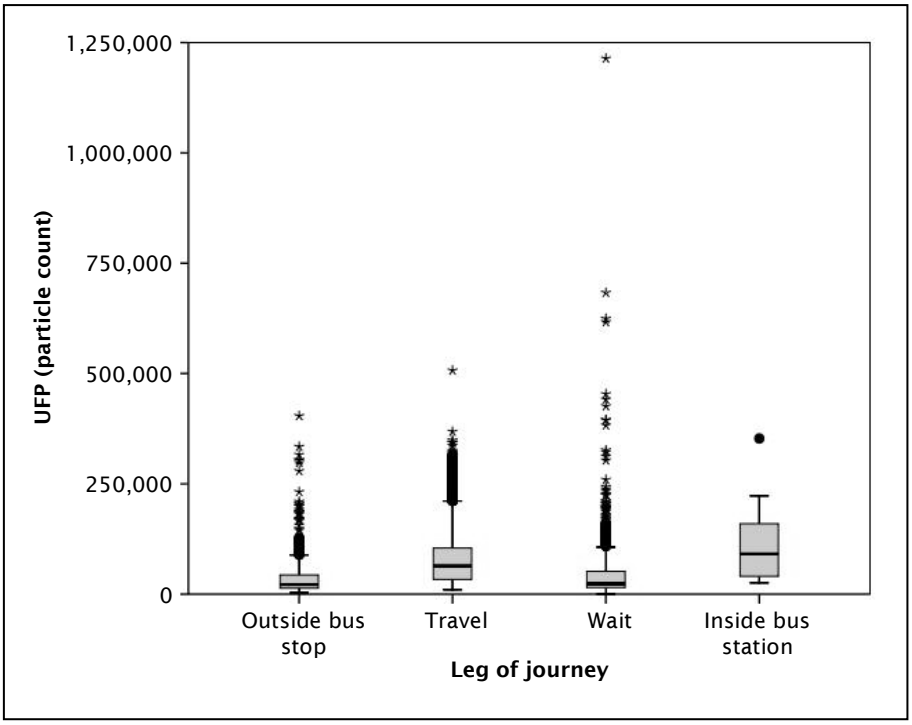


Figure 4.52 Box plot of bus passenger UFP levels for journey legs in Christchurch



4.3.5 Individual journeys

Individual journeys can also be examined. The first of these (figure 4.53) shows significantly raised CO levels for the car driver in Christchurch on 10 March in the afternoon while the driver was walking through the enclosed car-park in the centre of Christchurch to get to their car at the start of the journey, and the same location also sees elevated levels of PM_1 exposure for the car driver on the morning of 4 March (figure 4.54). On the other hand, figure 4.55 shows very high levels of CO for the bus passenger at the bus stop on the morning of 17 March.

This leads us to conclude that during some parts on the journeys, travellers are exposed to very high levels of pollution, often for short periods of time, which has potential health implications.

Figure 4.53 Box plot of car driver CO levels for journey legs in Christchurch on 10 March 2009 during the evening peak

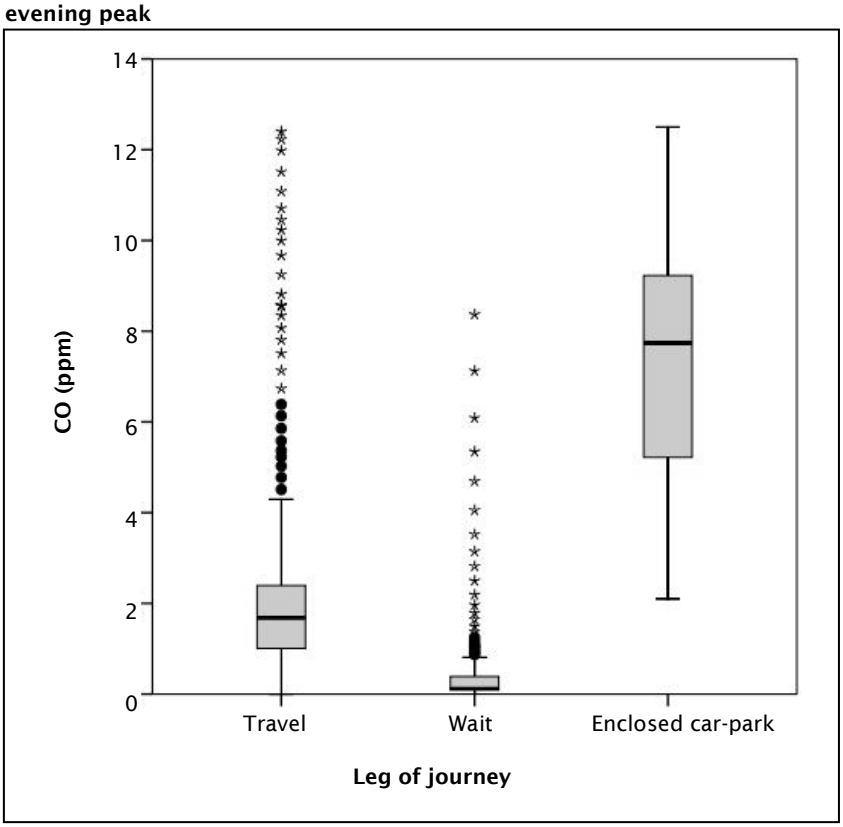


Figure 4.54 Box plot of car driver PM_1 levels for journey legs in Christchurch on 4 March 2009 during the morning peak

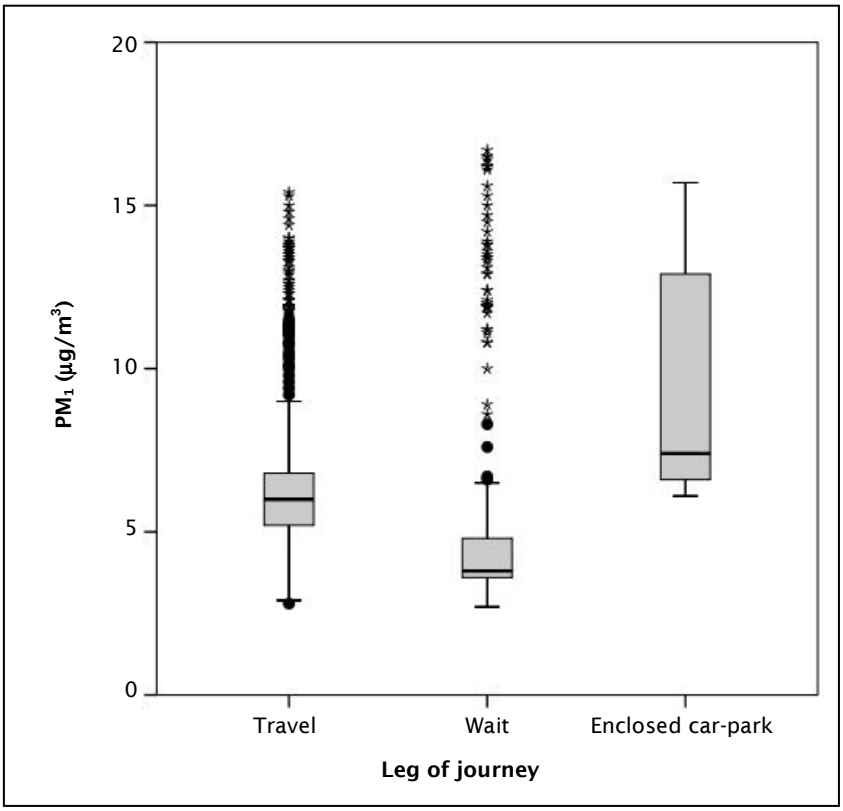
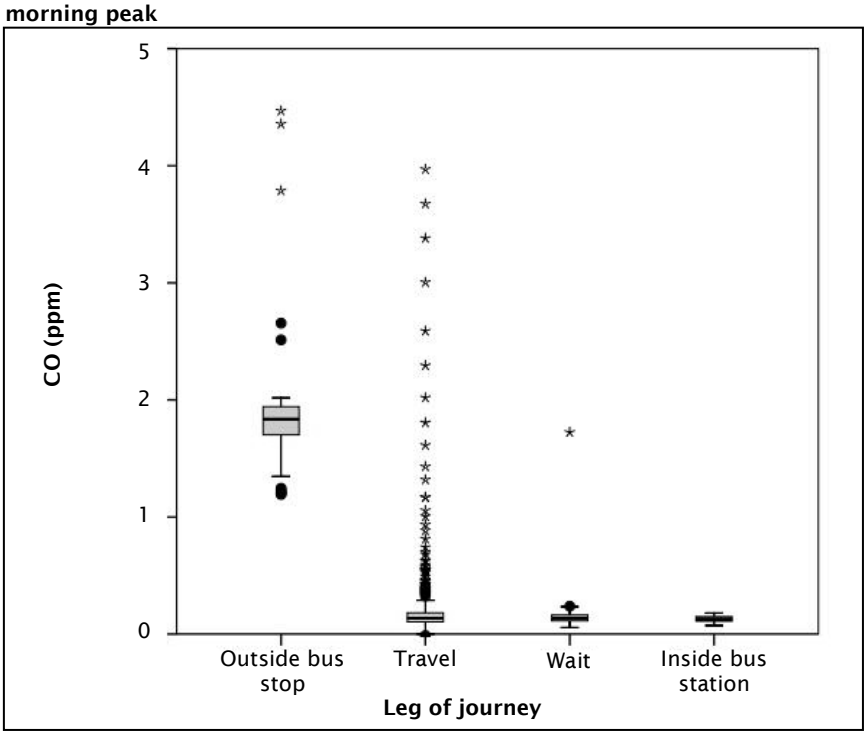


Figure 4.55 Box plot of bus passenger CO levels for journey legs in Christchurch on 17 March 2009 during the morning peak



4.4 The role of weather conditions

4.4.1 Introduction

Weather conditions such as wind speed, temperature and relative humidity can have a marked impact on pollutant emission rates and dispersion. For example, during windy conditions, we might expect to see lower concentrations of CO, UFP and potentially $PM_{2.5}$ but higher concentrations of PM_{10} because of resuspension processes. Cooler temperatures not only indicate reduced dispersion, but also potentially result in higher emissions of CO from motor vehicles because of the reduced efficiency of catalytic converters at lower temperatures. Thus meteorological conditions potentially affect exposure in the transport microenvironment.

Despite the different sampling periods, the weather conditions observed during the field campaigns were remarkably consistent between the two cities (table 4.5). This reduces the complexity and number of confounding factors in the study when comparing the two cities and ensures, as far as possible, that emission characteristics in both cities were similar.

Table 4.5 Summary of meteorological variables during journey times in Auckland and Christchurch

Location	Factor	Mean	SD*	*N*	Min	Max
Auckland	Temperature (C)	13.52	3.12	23	8.40	18.83
Christchurch	Temperature (C)	14.92	3.06	24	8.55	22.74
Auckland	Wind speed (m/s)	2.48	1.25	23	0.48	5.10
Christchurch	Wind speed (m/s)	2.87	1.63	24	0.50	7.54
Auckland	Relative humidity (%)	74.17	9.86	23	52.25	91.75
Christchurch	Relative humidity (%)	77.31	9.90	24	57.38	98.85

* SD, standard deviation

4.4.2 Carbon monoxide

In Auckland, the correlation between mean weather condition variables and CO concentration at the FAQMS of Takapuna (which showed the highest correlation between the transport exposure and FAQMS measurements) revealed that increased wind speed (R = –0.74) and increased temperature (R = –0.66) resulted in reduced concentrations (table 4.6). By comparison, in Christchurch, only a weak correlation was observed between CO observed at the St Albans FAQMS (which showed the highest correlation between the transport exposure and FAQMS measurements) and wind speed (–0.56), and no correlation was observed between temperature and CO. This difference between the two cities was unexpected and cannot easily be explained by local differences in geography or land use characteristics at the two FAQMS sites.

Table 4.6 Summary of Pearson correlations between weather variables and fixed station monitoring of CO in Auckland and Christchurch

Location	Wind speed	Temperature	Relative humidity
Auckland: Takapuna station monitoring CO	–0.74**	–0.66**	0.44
Christchurch: St Albans station monitoring CO	–0.67**	–0.24	0.14

**Correlation is significant at the 0.01 level (two-tailed).

In Auckland, a strong negative correlation was observed between the cyclist's and the bus passenger's exposure to CO and wind speed (table 4.7). Regression modelling revealed that wind speed explained approximately 64% variability in mean CO exposure for both the cyclist and bus passenger. Furthermore, for every 1m/s increase in wind speed, decreases in mean CO exposure of 0.67ppm for the cyclist and 0.56ppm for the bus passenger were reported. The enhanced dilution effect for the cyclist was expected, given the absence of shelter provided by a cabin. Previous studies have also reported similar results for bus journeys. For example, Gómez-Perales et al (2004) found an 18% decrease in CO exposure in a minibus was associated with every 1m/s increase in wind speed. Alm et al (1999) reported distinctly lower in-vehicle CO exposures in windy weather (≥2m/s), which accounted for 16% of the variability of CO exposure. Given the absence of a strong correlation between the CO concentrations observed at the FAQMSs and wind speed, it is perhaps not surprising that the relationship between exposure and wind speed was reduced in Christchurch.

Table 4.7 Summary of Pearson correlations between weather-related variables and mean exposure to CO for each transport mode in Auckland and Christchurch

Location	Factor	On-road bike	Bus	Car	Train (Auckland)/ Off-road bike (Christchurch)
Auckland	Wind speed	-0.80**	-0.80**	-0.58**	-0.69**
Christchurch	Wind speed	-0.69*	-0.54*	-0.34	-0.46*
Auckland	Temperature	-0.69**	-0.71**	-0.41**	-0.62*
Christchurch	Temperature	-0.19	-0.15	-0.12	-0.06
Auckland	Relative humidity	0.45*	0.40	0.05	0.36

*Correlation is significant at the 0.05 level (two-tailed).

**Correlation is significant at the 0.01 level (two-tailed).

The relationship between temperature and CO exposure during the commute was more complex. In Auckland, a weak correlation was observed between the exposure observed by the car driver and the temperature, whereas the exposures observed by the cyclist and bus passenger were again similar. This suggests that the observed relations are not a consequence of increased exhaust emissions but are perhaps caused by a secondary source such as space heating emissions. It is interesting, however, that in Christchurch, no correlation was observed between exposure and mean temperature. Given that the mean temperatures in the two cities were similar, if space heating were to account for increased correlation in Auckland, it should also have been observed in Christchurch. This needs more investigation. Few studies have identified this inverse relationship between CO concentration and temperature. A few studies in the literature report strong correlations between CO exposure and temperature. For example, Kaur et al (2006) observed that temperature corresponded to approximately 20% variability in CO exposure, and Elminir (2005) notes that concentrations were only sensitive to temperature in the temperature range between 10 and 15˚C.

4.4.3 Particulate matter

PM_{10} and $PM_{2.5}$ are both poorly correlated with wind speed, temperature and relative humidity at FAQMSs in Auckland and Christchurch. No data was available for PM_1 at the FAQMSs (table 4.8).

Table 4.8 Summary of Pearson correlations between meteorological variables and FAQMS monitoring of PM_{10} and $PM_{2.5}$ in Auckland and Christchurch

Site	Wind speed	Temperature	Relative humidity
Auckland: Takapuna station monitoring PM_{10}	0.08	0.27	0.23
Christchurch: St Albans station monitoring PM_{10}	0.22	0.20	-0.19
Auckland: Takapuna station monitoring $PM_{2.5}$	-0.47*	-0.36	0.40

**Correlation is significant at the 0.01 level (two-tailed).

The relationships between exposure to particulate matter during the commute and mode of transport used were strongly affected by particle size. No evidence of resuspension of coarse particles associated with increased wind speed was found in either city, as has been shown in previous studies (eg Alm et al 1999). However, a strong negative correlation was observed by the cyclist for PM_1 in both Auckland and Christchurch. In Christchurch, this was also observed by the bus passenger.

For example, in Auckland, wind speed was strongly negatively correlated with PM_1 for the cyclist (similar to CO) but not with any other size fraction; neither was the relation observed for any other mode of transport. Similarly, a positive correlation between temperature and exposure to PM_{10} and $PM_{2.5}$ was observed by the bus passenger but not for the other modes of transport.

In Auckland, a weak relationship between temperature and PM_{10} and $PM_{2.5}$ was observed during the bus commute. In Christchurch, a weak negative correlation was observed between $PM_{2.5}$ and temperature, and a weak positive correlation between PM_1 and temperature was observed for the cyclist only. In both cities, a significant positive correlation was observed between PM and the temperature by the cyclist only; again, this was size-dependent, with PM_{10} and $PM_{2.5}$ showing sensitivity in Christchurch and PM_{10} in Auckland. The complexity of these results and the lack of consistency between modes of transport and cities strongly suggest that many different sources of particulate matter are present in both cites which are not related to transportation sources. Furthermore, the difference in behaviour between the size fractions again suggests that $PM_{2.5}$ and larger fractions are poor indicators of exposure to traffic pollution.

Table 4.9 Summary of Pearson correlations between weather-related variables and mean exposure to PM_{10}, $PM_{2.5}$ and PM_1 in each transport modes in Auckland and Christchurch

Location	Factor	Particulate	On-road bike	Bus	Car	Train (Auckland)/ off-road bike (Christchurch)
Auckland	Wind speed	PM_{10}	-0.16	0.16	0.15	N/A
		$PM_{2.5}$	-0.21	0.11	0.05	
		PM_1	-0.76**	-0.18	-0.20	
Christchurch		PM_{10}	-0.50	-0.26	N/A	0.29
		$PM_{2.5}$	-0.64**	-0.35		-0.20
		PM_1	-0.67**	-0.62**		-0.56**
Auckland	Temperature	PM_{10}	0.22	0.59**	0.46*	N/A
		$PM_{2.5}$	-0.24	0.56**	0.30	
		PM_1	0.41	0.33	0.22	
Christchurch		PM_{10}	-0.50	0.03	N/A	0.37
		$PM_{2.5}$	-0.56*	0.02		0.21
		PM_1	0.52*	0.30		-0.30
Auckland	Relative humidity	PM_{10}	0.45*	0.28	0.22	N/A
		$PM_{2.5}$	0.49*	0.29	0.27	
		PM_1	0.48*	0.33	-0.04	
Christchurch		PM_{10}	0.64**	0.34	N/A	0.17
		$PM_{2.5}$	0.67**	0.41		0.39
		PM_1	0.50	0.36		0.57**

*Correlation is significant at the 0.05 level (two-tailed).

**Correlation is significant at the 0.01 level (two-tailed).

In summary, wind speed has a strong influence on exposure to CO (reducing exposure) and, to a lesser extent, on exposure to PM_1 in both cities. This was more likely to be detected by the cyclist than by other modes because of the absence of a cabin or shield, which acts to filter the atmosphere. Overall, the relation between wind speed and exposure to larger size fractions of PM is weak, and varies by particle size and mode of transport. The correlation between CO and temperature appears stronger in Auckland than in Christchurch and is independent of transport mode.

4.5 Contribution of commuting to total daily exposure

To determine the contribution that exposure during commuting makes to total exposure over a 24-hour period would require a substantially different experimental design that allowed for continuous 24-hour monitoring. This would require different instrumentation for UFPs (the 3007 instruments require hourly attention to maintain the correct level of working fluid). Such a design was beyond the capability of the resources available for this project. However, very limited observations were made outside of the commuting period to provide an initial comparative indication of the relative contribution of commuting to total exposure.

On the evening of Friday 20 March 2009, during the Christchurch study, each of the three sampling kits was taken home by the study participants and run continuously from approximately 6:00pm to midnight. No further monitoring was conducted until the afternoon commute of Monday 23 March. Following the morning observational run of Thursday 26 March, one set of instruments were run for approximately one hour around midday in an office of the University of Canterbury. The mean concentrations observed during these additional periods are shown in table 4.10.

Table 4.10 Mean concentrations observed for non-commuting exposures

Location	CO (ppm)	UFP (cm^3)	PM_{10} ($\mu g/m^3$)	$PM_{2.5}$ ($\mu g/m^3$)	PM_1 ($\mu g/m^3$)
Lyttelton	0.2	17.0	26.6	7.5	2.5
Deans Ave	0.3	13.0	29.1	16.1	7.6
Hansons Lane	0.2	8.0	20.0	6.6	2.3
University of Canterbury office	0.0	2.2	15.8	7.2	3.5

The range of these values is compared to the range of mean commuting exposures on the evening of the 23rd and the morning of the 24th (ie before and after the residential sampling), and for the whole Christchurch campaign in figures 4.56 to 4.58. A significant difference can be seen between commuting exposures on the evening of the 23rd and the morning of the 24th, especially exposure to UFP. Nevertheless, in general, it can be seen that for CO and UFP, the commuting exposures are between double and an order of magnitude greater than the evening exposures, and that the office exposure is much smaller again. For PM_{10}, the difference between commuting and non-commuting exposures is much smaller, consistent with the hypothesis that traffic emissions make a much smaller contribution to this metric.

Figure 4.56 Commuting exposures to CO on the runs before and after limited evening (residential) and daytime (indoor office) sampling in Christchurch

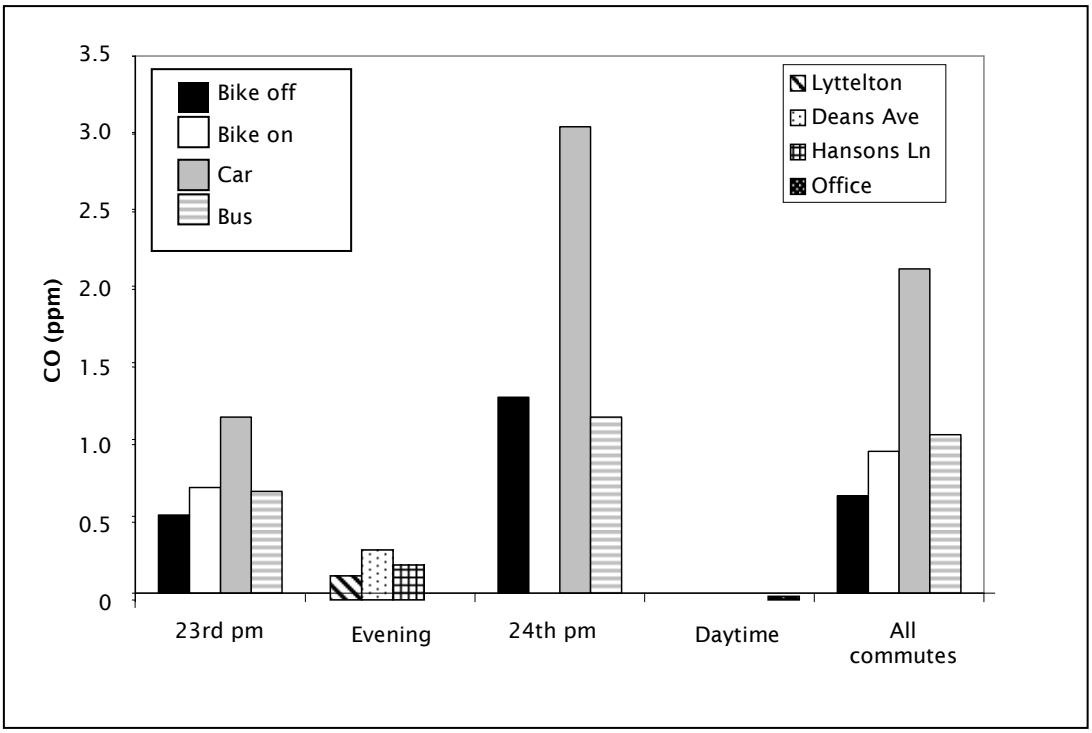
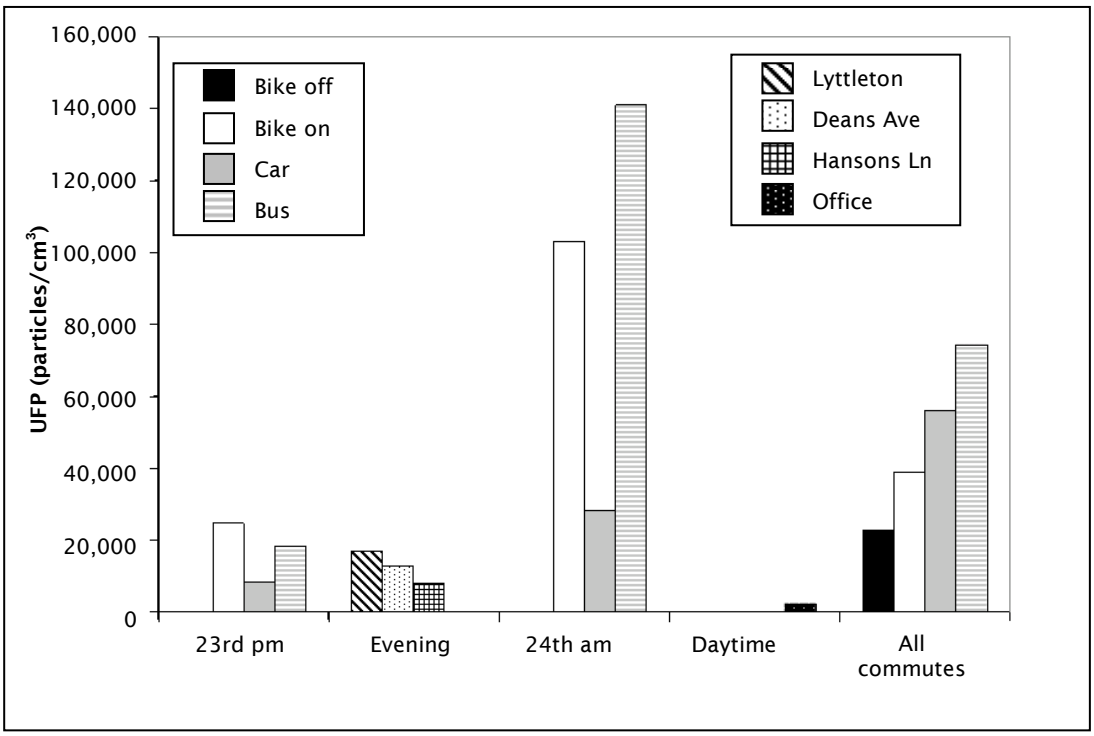


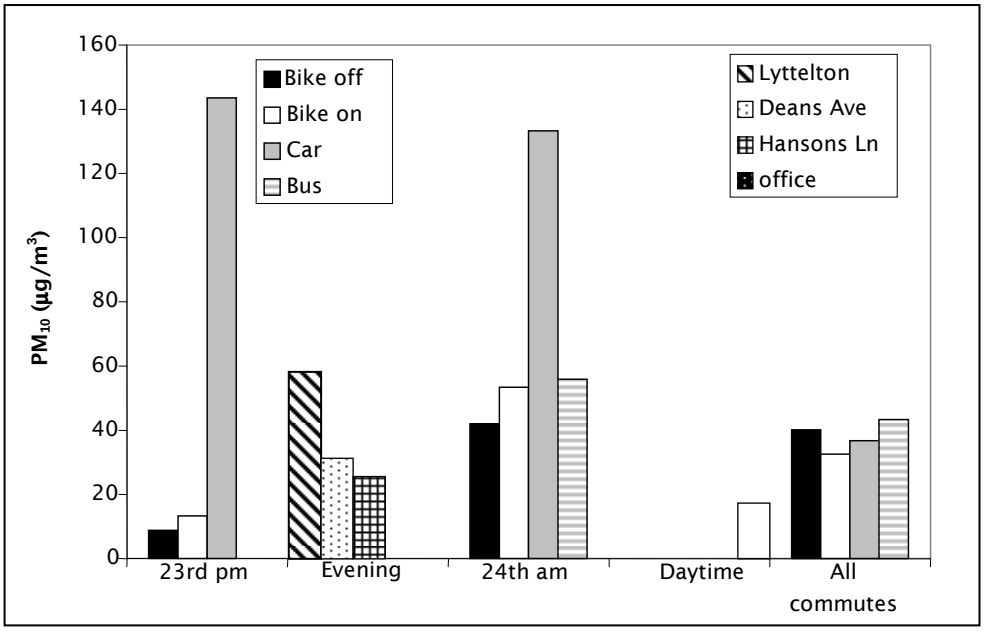
Figure 4.56 also shows the mean exposures for all commuting sampling in Christchurch. (Note that on-road cycling was not sampled on the morning of 24 March)

Figure 4.57 Commuting exposures to UFPs on the runs before and after limited evening (residential) and daytime (indoor office) sampling in Christchurch



The mean exposures for all commuter samples in Christchurch are also shown in figure 4.57. (Note that off-road cycling was not sampled on 23 or 24 March, and that bus was not sampled on 23 March.)

Figure 4.58 Commuting exposures to PM_{10} on the runs before and after limited evening (residential) and daytime (indoor office) sampling in Christchurch



It is not appropriate to be any more specific in presenting these data. Clearly, the commuting and non-commuting exposures are not directly comparable. The non-commuting data was collected over a much shorter period and cannot be presumed to be generally representative. However, the results for CO and UFP are striking, and a suggest that one hour of commuting (ie 4% of the day) could contribute between 15% and 25% of the total CO and UFP daily dose.

5 Conclusions

5.1 Scientific findings

- Car drivers are consistently exposed to the highest average levels of CO: 60% higher than cyclists, 40–100% higher than bus passengers and over 100% higher than train passengers.
- On-road cyclists are exposed to higher levels of CO (10%), PM_1 (25%) UFP (over 100%) than off-road cyclists. This could have significant policy implications for the location of cycle routes.
- Car drivers and bus passengers are exposed to higher average levels of UFP than cyclists. However, for very short acute exposures (a few seconds), on-road cyclists can be exposed to higher peaks.
- During some parts on the journeys, travellers are exposed to very high levels of pollution, often for short periods of time. This has potential health implications.
- Locating cycle paths just a short distance from roads can reduce pollution exposure significantly. For example, locating a cyclist 5–7m away from the roadway can reduce exposure by 20–40%.
- One hour of commuting (ie 4% of the day) could contribute between 15% and 25% of the total CO and UFP daily dose.
- PM_{10} and $PM_{2.5}$ are inappropriate indicators of exposure to vehicle emissions.

5.2 Discussion and further research

5.2.1 Personal exposure of transport users and its assessment

One of the five objectives of the New Zealand Transport Strategy (NZTS) (NZTA 2008) is 'protecting and promoting public health'. Associated with this objective are five targets, two relating to road traffic accidents, one relating to noise, one supporting increased active mode travel and the other to air quality. Specifically, the target relating to air quality aims to:

> *...reduce the number of people exposed to health-endangering concentrations of air pollution in locations where the impact of transport emissions is significant.*

Neither the NZTS nor the NZTA's environmental plan provide further explicit elaboration on the meaning of 'location', 'impact' or 'significant'. However, it is implicit in these documents, and in the NZTA's *Economic evaluation manual* (EEM; NZTA 2010b) and the NZTA's preferred approach to air quality assessment for road projects (NZTA 2010) that 'locations' refers to whole cities as well as to roadside corridors and locations of intense traffic activity. This aligns with the prevailing philosophy of air quality management in New Zealand and in much of the rest of the world, ie that air quality is to be managed spatially, so that the desired reduction expressed by the NZTS target would be achieved through a decrease in 'health-endangering concentrations' at such 'significant locations', once such locations had been identified.

Implicit in this approach is that 'exposure' refers to a population who share that exposure because they all inhabit the polluted location. Our study explores a somewhat different conception of exposure – that of individual or personal exposure. When one considers the exposure of an individual, it becomes necessary to consider that individual's mobility.

Our study has added to an existing body of data which demonstrates that a large proportion of an individual's exposure to air pollutants occurs not in a fixed location, but while on the move; exposures

which typically have durations of an hour or often less. The average concentrations we measured on the move were all higher than those measured in fixed locations by ambient air quality monitoring stations (section 4.2). This implies that when one assesses exposure at the personal scale, at least some of the 'locations' where road transport emissions were most significant were mobile, ie the interiors of vehicles and the few metres behind a vehicle where its exhaust plume can be found. Our study also shows that exposures can vary substantially depending on factors other than 'location', eg mode and route. This raises the possibility of how to achieve a reduction in exposure other than managing air quality in a given location. It also potentially forces a reappraisal of which locations are significant in terms of influencing exposure during travel.

Ultimately, transport funding decisions will determine the pace of progress towards the NZTS targets, and it is here that the EEM plays a pivotal role. The EEM states that 'Improvements to public health can occur through increased physical activity and fitness and through reducing exposure to pollutants' (section 2.16 of the EEM).

In the procedures provided by the EEM, 'exposure to pollutants' is evaluated in terms of reductions in ambient PM_{10} concentrations weighted by population. This represents a use of the term 'exposure' which may be appropriate in some instances, but less so in others, especially in local-scale projects and schemes. Whereas this is appropriate for large populations (eg whole towns and cities), it cannot be simply applied to projects such as off-road cycleways. This is because it is unclear what the change in PM_{10} is (is it a change in PM_{10} at a particular location or for particular travellers, and, if so, on which mode?) or what the exposed population is.

Volume 1 of the EEM implies a willingness to consider **personal** exposure of transport users when evaluating transport projects:

> *Walking and cycling can have significant health benefits through increased exercise levels. However, this could be offset by an increased exposure to pollutants if the activity involves sharing road space.*

Unfortunately, the EEM (volume 2, section 8.4) does not mention exposure and makes no provision for its inclusion or consideration in any evaluation. This is likely to be related to the lack of any current means of linking these sorts of exposures to quantifiable health costs (see below). Nevertheless, this research provides an indication of the relative 'increased exposure' which could be explored further.

We believe that this research represents a step towards a more rational and helpful means of capturing the potential public health benefits (and external costs saved) related to air pollution in cost–benefit analysis for transport projects. However, further research is required to develop a means of a quantitative and representative measure of exposure of transport users, and a means of assessing it and predicting it under alternative scenarios such that it could be a useful addition to the Transport Monitoring Indicator Framework, EEM or other relevant tool.

5.2.2 Health implications

Our study sought to compare exposures. It was not a study of health outcomes, although it is intended to provide data to support such studies in the future. Were our experimental subjects exposed to 'health-endangering concentrations of air pollution'?

In our research, concentrations of CO were lower than all recognised national and international health protection standards and guidelines (which are available for 15-minute, 30-minute and 60-minute exposures). In that sense, none of our study commuters were exposed to 'health-endangering concentrations' of CO.

Our journey durations were all less than one hour. The Air Quality National Environmental Standards (New Zealand Government 2004) and Ambient Air Quality Guidelines (Ministry for the Environment 2002) for particulate matter (PM_{10} and $PM_{2.5}$) refer to exposures of 24 hours or one year. They also represent the health risk to a large population exposed to roughly similar air quality, and are unsuitable for application to short-term (<24 hours) exposure or localised variations in exposure, or subtle variations within populations. It is not appropriate to compare our observations of PM_{10} (or $PM_{2.5}$) to these guidelines. The health implications of short-term (in the order of an hour) exposure to elevated levels of PM, including very brief exposures (minutes or seconds) to high concentrations, are a major scientific knowledge gap at present.

Many academics and specialists believe that the health risk arising from transport emissions is actually related to UFPs and is best expressed by particle number concentrations. However, insufficient scientific evidence has yet been gathered to define what a 'health-endangering concentration' might be and no standards or guidelines yet exist (Morawska et al 2008). Nevertheless, enough evidence of the importance of UFPs has been gathered for the World Health Organisation to have included it for its next review of its air quality guidelines. Providing data to inform that global research effort is one of the motivations of this study.

5.2.3 Causes of exposure – study limitations

The primary objective of this study was to provide an accurate measure of personal pollution exposure by mode. What was beyond the scope of this study was to explain the reasons for the variations observed between modes, routes and trips. Elucidating the determinants of exposure, and hence the causes of variation, could be informative for identifying means of reducing exposure for transport users, and quantifying or predicting that reduction. Some exploratory analysis is included in this report, which we briefly discuss here.

Studies such as this are difficult to design and execute because of the wide range of influencing variables (vehicle age, meteorology, routes chosen, time of day and design of cycleways being just a few) and our inability to control or influence most of these variables. We adopted a study design within the constraints of our available resources which allowed us to consider two cities (with differing meteorological and air quality climates, levels of traffic and transport systems), a range of meteorological conditions (but not all seasons), four transport modes and two journeys (origin–destination pairs) in the case of Christchurch. It is beyond the scope of this study to show that the results are generally applicable in other cities, in other climates, in other vehicles or on other routes. This is because to do so would require either vastly more observational data or a full understanding of the processes and variables determining exposure. Nevertheless, we chose heavily-used routes during peak travel times on working days which, combined with the large number of repeated observational trips, leads us to propose that our data does not represent particularly atypical conditions for New Zealand.

5.2.4 Personal exposure and the choice to cycle

Under the objective of 'protecting and promoting public health', the NZTS includes a target to 'increase walking, cycling and other active modes to 30 percent of total trips in urban areas by 2040.' However, the EEM recognises the conflict between health benefits and exposure risks:

> *Walking and cycling can have significant health benefits through increased exercise levels. However, this could be offset by an increased exposure to pollutants if the activity involves sharing road space.*

Our study strongly supports this statement, although this study does not permit health benefits and increased exposure to be quantified in a common currency, so the net effect cannot be quantified.

Other research has also shown that pollution exposure – or, more precisely, the perception and tolerance of cyclists and would-be cyclists to exposure to traffic exhaust – plays a major role in deterring cycling in urban areas. As such, it can act as an effective counterforce to the health benefits which could be gained through the exercise cycling provides.

However, our study suggests that, in general, a cyclist's exposure is actually lower than that of a bus or car user for the same journey, and that this exposure can be reduced further through careful consideration of the separation of cyclists from motorised vehicles. What is beyond the scope of this study is whether the potential health gains arising from reduced exposure for the cyclist are offset by their increased breathing rate.

5.2.5 Heavy-duty diesels and cycling infrastructure

It was not the objective of our study to investigate potential changes in exposure impacts associated with transport policies or projects. However, our exploratory analysis of the dataset has identified two issues which could be worthy of further exploration – proximity to heavy diesel exhausts and cycling infrastructure.

In our study, the highest levels of UFPs, PM_{10}, $PM_{2.5}$ and PM_1 were measured by the bus commuter. However, peaks in exposure for all modes were strongly associated with periods of following or being close to diesel vehicles, especially buses, including time spent at bus stops. (In this study, the timing of the sampling, ie the morning and evening rush hour, meant that buses were more prevalent than trucks.) This could indicate that such diesel vehicles have a disproportionately large impact on exposure. It also suggests that the way such vehicles use the road space or are scheduled (to enable more separation between point of emission and exposed travellers) could offer a means of exposure reduction by reducing the probability or frequency of close encounters between people and diesel vehicle exhausts. We recommend that the effect on exposure of (for example) bus convoys, bus or truck lanes, and bus prioritisation at signals is explored further.

Studies elsewhere have shown that bus new technologies have substantially lower particulate emissions (including UFPs) than older buses. This indicates that the emission standards of the buses employed could play a critical role in determining the exposure of commuters. We recommend that this causal relationship between vehicle emission standards and resulting exposure is investigated further to help understand the potential that such emission standards (across the national fleet or targeted, for example, at public transport fleets or low emission zones) have for reducing exposure of travellers specifically, as well as the population in general.

The NZTS makes repeated reference to 'cycling-friendly environments'. Our research indicates that design options exist for substantially reducing the exposure of cyclists to harmful pollution by providing cyclists with spatial separation from vehicle exhausts, and that substantial benefits can be achieved by relatively small degrees of separation. Our study showed clearly that the effect of providing cycleways 5–7m from traffic makes for a 20–40% reduction in exposure reduction. Although this was not explicitly studied, we speculate (based on our observational data) that further exposure reductions could be achieved by providing cycles with 'front of the queue' priority at signalised intersections, which would reduce the occurrence of cyclists finding themselves immediately behind accelerating diesel exhausts. We recommend that the potential for such gains be investigated further.

5.2.6 Summary of questions for further research

- What are the determinants of transport users' exposure, and the causes of variation between trips, routes and modes? Can a generalised predictive model be developed which permits scenario-based assessment, or which allows predictions to be made in other cities, on other routes, in different vehicles or at other times? What are the relationships between road design, vehicle design (and emission standards), traffic management and exposure?
- How can personal exposure during travel be quantified in a sufficiently simple and general manner so that it can be monitored and predicted, and form the basis of policy targets and assessment and evaluation procedures?
- What are the health implications of brief exposures (less than an hour) to transport-related air pollutants, as experienced every day by large populations?
- What role do variations in breathing and posture between different travel modes have on the resulting health effects?
- Is it possible to compare the health benefits of active travel against the health costs associated with air pollutant exposure?
- Is exposure a factor in modal choice and could exposure-reducing measures encourage behavioural change?

5.3 Recommendations

This research clearly found that for journeys we deem to be typical for urban New Zealand, the exposure of cyclists to traffic-related air pollution was lower than that of car or bus users. This research also makes it clear that the exposure of cyclists can be substantially reduced further by providing cyclists with spatial separation from vehicle exhausts, and that substantial benefits can be achieved by relatively small degrees of separation (5–10m).

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Appendix A Descriptive statistics

A1 Pollution values

Table A1 Descriptive statistics of pollution values for CO, PM_{10}, $PM_{2.5}$, PM_1 and UFPs of trips in Christchurch

Mode	Pollutant	Min	Percentile 05	Percentile 25	Median	Percentile 75	Percentile 95	Max
Bus	CO (ppm)	0.1	0.1	0.3	0.9	1.2	2.1	4.0
Car	CO (ppm)	0.0	0.1	0.7	1.6	3.0	5.4	8.9
On-road bike	CO (ppm)	0.09	0.15	0.28	0.7	1.13	2.55	2.92
Off-road bike	CO (ppm)	0.1	0.1	0.2	0.5	0.8	1.5	2.1
Bus	PM_{10} (µg/m^3)	18.7	22.2	28.1	35.5	49.5	85.5	112.6
Car	PM_{10} (µg/m^3)	18.8	18.8	24.6	35.7	47.8	79.2	79.2
On-road bike	PM_{10} (µg/m^3)	12.9	14.1	27.8	32.0	36.9	60.4	61.7
Off-road bike	PM_{10} (µg/m^3)	8.2	8.8	23.4	35.3	54.5	82.5	91.4
Bus	$PM_{2.5}$ (µg/m^3)	8.5	11.6	13.6	19.5	26.6	38.8	78.0
Car	$PM_{2.5}$ (µg/m^3)	8.5	8.5	10.9	14.9	22.5	29.4	29.4
On-road bike	$PM_{2.5}$ (µg/m^3)	6.4	7.5	14.5	16.0	20.5	28.8	38.3
Off-road bike	$PM_{2.5}$ (µg/m^3)	4.8	5.6	11.7	16.3	26.4	33.7	56.4
Bus	PM_1 (µg/m^3)	4.6	5.8	8.2	10.2	14.2	24.5	52.3
Car	PM_1 (µg/m^3)	4.2	4.2	6.0	7.6	11.0	22.6	22.6
On-road bike	PM_1 (µg/m^3)	2.6	3.1	6.2	8.2	10.2	22.7	31.0
Off-road bike	PM_1 (µg/m^3)	1.4	2.7	4.5	5.9	10.0	23.4	26.2
Bus	UFP (count)	30,308	30,308	37,574	45,561	107,455	214,492	214,492
Car	UFP (count)	7216	16,901	25,738	36,807	81,176	143,771	221,267
On-road bike	UFP (count)	10,121	12,681	22,107	31,414	54,157	100,363	160,520
Off-road bike	UFP (count)	3601	5903	11,314	16,641	25,466	81,441	81,626

Table A2 Descriptive statistics of pollution values for CO, PM_{10}, $PM_{2.5}$ and PM_1 of trips in Auckland

Mode	Pollutant	Min	Percentile 05	Percentile 25	Median	Percentile 75	Percentile 95	Max
Train	CO (ppm)	0.6	0.6	0.8	1.0	1.4	2.8	3.0
Bus	CO (ppm)	0.1	1.3	1.7	2.0	2.7	4.1	4.9
Bike	CO (ppm)	1.1	1.4	1.7	2.2	2.8	4.1	5.5
Car	CO (ppm)	2.6	3.1	3.7	4.4	6.5	7.2	8.5
Bus	PM_{10} (µg/m^3)	8.1	11.5	18.4	24.6	27.1	34.7	37.5
Bike	PM_{10} (µg/m^3)	16.1	17.3	19.7	23.5	32.0	42.3	52.5
Car	PM_{10} (µg/m^3)	14.3	14.7	19.8	23.7	29.2	33.5	38.8
Bus	$PM_{2.5}$ (µg/m^3)	3.7	7.9	14.6	25.2	29.2	36.6	39.7
Bike	$PM_{2.5}$ (µg/m^3)	10.4	10.7	11.7	14.6	18.4	23.8	37.9
Car	$PM_{2.5}$ (µg/m^3)	9.9	10.4	14.6	18.7	21.6	24.1	29.2
Bus	PM_1 (µg/m^3)	2.5	6.1	11.3	18.0	21.7	26.3	34.0
Bike	PM_1 (µg/m^3)	6.0	6.7	8.0	9.5	11.7	15.5	18.1
Car	PM_1 (µg/m^3)	6.6	6.9	9.3	12.4	14.8	21.1	26.5

A2 Pollution ratios

Table A3 Descriptive statistics of pollution ratios of trips in Christchurch

Ratio	Pollutant	Min	Percentile 05	Percentile 25	Median	Percentile 75	Percentile 95	Max
Car : off-road bike	CO	-0.1	0.1	1.3	2.5	7.5	26.1	37.3
On-road bike : off-road bike	CO	0.2	0.4	0.9	1.6	2.4	4.1	25.1
Car : on- road bike	CO	-0.1	0.0	0.9	1.5	3.0	12.2	46.9
Car : bus	CO	-0.4	0.2	1.0	1.8	2.6	10.4	96.4
Bus : on-road bike	CO	-0.1	0.3	0.6	1.0	1.2	4.2	14.9
Bus : off-road bike	CO	-0.1	0.4	0.8	1.4	2.0	4.2	29.5
Car : off-road bike	PM_{10}	0.4	0.4	0.6	0.8	1.0	1.4	1.4
On-road bike : off-road bike	PM_{10}	0.4	0.5	0.8	0.9	1.2	1.6	1.6
Car : on-road bike	PM_{10}	0.5	0.5	0.5	0.6	0.7	0.7	0.7
Car : bus	PM_{10}	0.6	0.6	0.7	0.8	1.0	1.6	1.6
Bus : on-road bike	PM_{10}	0.7	0.8	0.9	1.1	1.4	2.5	3.7
Bus : off-road bike	PM_{10}	0.7	0.7	0.9	1.0	1.4	2.5	3.0
Car : off-road bike	$PM_{2.5}$	0.5	0.5	0.8	0.9	1.2	2.2	2.2
Car : bus	$PM_{2.5}$	0.6	0.6	0.7	0.8	0.9	1.4	1.4
On-road bike : off-road bike	$PM_{2.5}$	0.3	0.3	0.9	1.0	1.1	1.5	1.8
Car : bus	$PM_{2.5}$	0.6	0.6	0.7	0.8	0.9	1.4	1.4
Bus : on-road bike	$PM_{2.5}$	0.7	0.9	1.0	1.1	1.5	2.6	2.9
Bus : off-road bike	$PM_{2.5}$	0.7	0.7	0.9	1.1	1.3	2.5	2.7
Car : off-bike	PM_1	1.0	1.0	1.1	1.4	2.1	4.4	4.4
On-road bike : off-road bike	PM_1	0.3	0.3	1.1	1.2	1.4	1.9	1.9
Car : on-road bike	PM_1	1.9	1.9	1.9	2.1	2.3	2.3	2.3
Car : bus	PM_1	0.6	0.6	0.8	0.8	1.0	1.3	1.3
Bus : on-road bike	PM_1	0.7	0.8	1.0	1.2	1.6	2.8	3.0
Bus : off-road bike	PM_1	0.7	0.7	1.3	1.5	2.1	3.4	3.4
Car : off-road bike	UFP	0.2	1.3	1.5	2.1	3.2	5.2	5.4
On-road bike : off-road bike	UFP	0.2	0.4	1.3	2.2	2.8	4.1	4.9

Ratio	Pollutant	Min	Percentile 05	Percentile 25	Median	Percentile 75	Percentile 95	Max
Car : on-road bike	UFP	0.5	0.6	0.7	1.1	1.8	2.5	2.8
Car : bus	UFP	0.4	0.4	0.8	1.0	1.2	1.9	1.9
Bus: on-road bike*	UFP	1.0	1.0	1.1	1.2	1.5	2.2	2.2

* UFP levels on bus and off-road bike were never measured simultaneously, so this ratio has been omitted.

Table A4 Descriptive statistics of pollution ratios of trips in Auckland

Ratio	Pollutant	Min	Percentile 05	Percentile 25	Median	Percentile 75	Percentile 95	Max
Car : train	CO	1.9	2.1	3.9	4.8	5.3	6.7	10.0
Train : bike	CO	0.2	0.2	0.4	0.6	0.6	0.9	1.1
Car : bike	CO	1.1	1.2	1.7	2.4	2.8	3.7	4.2
Car : bus	CO	1.3	1.3	1.8	2.2	2.7	3.7	72.4
Bus : bike	CO	0.0	0.5	0.9	1.0	1.3	1.5	1.7
Bus : train	CO	0.1	1.4	1.7	2.1	2.3	3.1	3.3
Car : bike	PM_{10}	0.6	0.6	0.7	1.0	1.3	1.5	1.6
Car : bus	PM_{10}	0.6	0.7	0.8	1.0	1.5	2.1	2.1
Bus : bike	PM_{10}	0.4	0.5	0.7	1.0	1.2	1.6	1.6
Car : bike	$PM_{2.5}$	0.6	0.7	1.0	1.1	1.4	1.9	2.0
Car : bus	$PM_{2.5}$	0.5	0.5	0.6	0.7	1.4	2.6	2.6
Bus : bike	$PM_{2.5}$	0.4	0.5	0.9	1.5	1.9	2.6	2.9
Car : bike	PM_1	0.6	0.7	1.0	1.3	1.6	2.8	3.1
Car : bus	PM_1	0.4	0.4	0.5	0.8	1.4	3.0	3.1
Bus : bike	PM_1	0.4	0.5	1.0	1.6	2.2	3.0	3.4

A3 Journey leg pollution values

Table A5 Descriptive statistics of journey leg pollution values for CO, PM_1 and UFP in Christchurch

Mode	Journey leg	CO (ppm)			PM_1 (µg/m^3)			UFP (count)		
		Mean	SD*	Median	Mean	SD	Median	Mean	SD	Median
Bus	Outside bus stop	1.23	1.24	0.87	11.02	6.53	9.80	43,997	59,953	21,590
	Travel	1.07	1.11	0.80	14.26	14.06	11.00	84,335	66,712	63,883
	Wait	0.90	1.05	0.66	10.25	6.52	8.40	50,187	78,034	23,593
	Inside bus station	0.51	0.48	0.36	14.40	8.31	12.20	102,864	64,241	91,095
Car	Travel	2.83	3.26	2.27	8.86	5.80	6.60	79,872	89,002	45,133
	Wait	0.80	1.06	0.49	9.58	6.98	6.70	22,758	19,619	16,307
	Enclosed car-park	3.06	4.78	1.37	12.36	6.00	11.40	61,292	72,176	37,946

*SD = standard deviation

Table A6 Descriptive statistics of journey leg pollution values for CO and PM_1 in Auckland

Mode	Journey leg	CO (ppm)			PM_1 (µg/m^3)**		
		Mean	SD*	Median	Mean	SD	Median
Train	Wait	3.12	0.78	2.95	–	–	–
	Travel	3.36	0.85	3.09	–	–	–
	Enclosed central city station	3.10	0.48	3.01	–	–	–
	Train station	3.78	1.37	3.20	–	–	–
Bus	Wait	2.82	1.18	2.61	11.86	6.89	9.94
	Travel	3.84	1.45	3.60	18.17	11.60	15.15
	Outside bus stop	3.15	1.29	2.81	16.62	11.64	13.95
Car	Wait	3.78	1.57	3.10	7.19	3.75	6.90
	Enclosed car-park	6.07	2.37	6.08	6.95	4.66	5.50
	Travel	7.97	2.85	7.43	14.60	15.22	9.40

* SD = standard deviation

** Only CO was successfully measured in the train.

Appendix B Abbreviations and acronyms

BTEX:	benzene, toluene, ethyl benzene and xylenes
CO:	carbon monoxide
CO_2:	carbon dioxide
DMU:	diesel multiple unit
EEM:	*Economic evaluation manual*
FAQMS:	fixed air quality monitoring system
GPS:	global positioning system
IEA:	International Energy Agency
NIWA:	National Institute of Water and Atmospheric Research
NO:	nitric oxide
NO_2:	nitrogen dioxide
NO_x:	mononitrogen oxides
NZTA:	NZ Transport Agency
NZTS:	New Zealand Transport Strategy
O_3:	ozone
PAH:	polycyclic aromatic hydrocarbons
PM:	particulate matter
ppb:	parts per billion
ppm:	parts per million
SO_2:	sulphur dioxide
UFP:	ultrafine particles
US:	United States
VOC:	volatile organic compounds