Tools for assessing exposure to land transport emissions
July 2011

Ian Longley, Gustavo Olivares, Sharleen Harper and Kreepa Shrestha

NZ Transport Agency research report 451

ISBN 978-0-478-38023-1 (print)
ISBN 978-0-478-38022-4 (electronic)
ISSN 1173-3756 (print)
ISSN 1173-3764 (electronic)

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

Longley, I, G Olivares, S Harper and K Shrestha (2011) Tools for assessing exposure to land transport emissions. *NZ Transport Agency research report 451.* 110pp.

National Institute of Water & Atmospheric Research Ltd
41 Market Place, Auckland Central, Auckland 1010
Private Bag 99940, Newmarket, Auckland 1149, New Zealand
Phone +64-9-375 2050, Fax +64-9-375 2051
www.niwa.co.nz

Keywords: air quality, environment, modelling, particulate matter, pollution

An important note for the reader

Acknowledgements

Some of the modelling results presented were based on traffic modelling output generated by Beca Infrastructure and provided to us courtesy of the NZ Transport Agency. Auckland air quality monitoring data and artificial meteorological dataset were provided courtesy of Auckland Regional Council.

One of the instruments vital for this research (TSI P-Trak condensation particle counter) was kindly loaned by Auckland Regional Council.

Abbreviations and acronyms

AADT	average annual daily traffic
AAQG	ambient air quality guidelines
AER	air exchange rate
APEX	air and particles exchange model
CPC	condensation particle counter
ELM	emission-link model
GIS	geographic information system
HCV	heavy commercial vehicle
I/O	internal/external concentration ratio
ITEM	integrated transport exposure model
IVEM	in-vehicle exposure model
LUR	land-use regression
NES	National Environmental Standards for Air Quality
NIWA	National Institute of Water & Atmospheric Research
NZTER	New Zealand Transport Emissions Rate database
OSPM	operational street pollution model
PM	particulate matter
PNCs	particle number concentrations
RCM	roadside corridor model
UFP	ultrafine particles
VEPM	vehicle emission prediction model
VOC	volatile organic compounds
WRR	Western Ring Route

Contents

Executive summary

In its investigation of tools for assessing exposure to land transport emissions, this research project focused on two main 'modes' of exposure:

1 The local elevation in concentrations of ambient air pollutants in the vicinity of major roads

2 Exposure to road traffic emissions while travelling in road vehicles.

This report is correspondingly divided into two parts.

Part One describes the principal output – the roadside corridor model (RCM) – what it is and does, and how it differs from alternative and existing approaches. The report also illustrates a number of potential applications of the RCM, and discusses potential future developments. Appendix A describes the RCM and its development in detail.

The roadside corridor may be considered to be the zone within which concentrations of air pollutants are significantly elevated because of road traffic emissions. Defining its extent and the degree of impact is conventionally approached in one of two ways. For the purposes of conservative screening applications simple generalised dispersions are available, such as those provided in the *Good practice guide for assessing discharges to air from land transport* (MfE 2008). Where greater accuracy and consideration of local features is required (as in detailed roadside assessments for regulatory purposes) a more complex combination of emission and dispersion modelling is often used. However, greater accuracy is not guaranteed as this approach is very demanding in terms of expert skills to use, review and audit relatively high operational costs, high data demands and multiple sources of uncertainty. The RCM was intended to fill the gap between these two approaches providing intermediate accuracy in a simple and accessible package. The model was also specifically intended for use in health impact assessment situations for which the existing methods were often unsuitable.

The RCM was based on the parameterisation of thousands of results from the emission-dispersion model pairing of the vehicle emission prediction model (VEPM) and the AusRoads model (the two main modelling tools recommended for regulatory use in New Zealand) applied to the (differing) wind climates of Wellington and Auckland.

The resulting model consists of two look-up tables which specify the values of two parameters needed to mathematically describe the dispersion curves that form the core of the RCM. The look-up tables have been integrated into a spreadsheet interface. A demonstration GIS-based web interface was also developed to inform potential future software development.

The RCM was designed to predict long-term roadside elevations in concentrations of passive contaminants (for example CO, PM_{10}, NO_x, but not NO_2) in urban areas. The model makes independent predictions for either side of the road to account for biases in the local wind climate.

The use of the RCM in specifying a general or average roadside corridor has shown how this task is highly sensitive to the choice of an air quality threshold to which the edge of the corridor corresponds. There is considerable uncertainty surrounding what these thresholds should be, which currently inhibits the definition of general corridor widths.

This project aimed at assisting with the identification of locations where the impact of transport emissions is significant. The RCM, when used in conjunction with monitoring, can assist in differentiating major traffic sources from other sources. This has been illustrated for several Auckland Regional Council monitoring stations where the RCM predicted that adjacent roads contributed 2%–16% to the annual mean PM_{10}.

The RCM, in its current state, is unvalidated. The model may under-perform in central business districts, in the vicinity of large or tall buildings, at intersections and in areas of complex terrain. The current version of RCM relies on emission rates from the VEPM. This emission model depends heavily on European emission factors, but the majority of our light vehicles were built to Japanese emission standards. The emissions of these Japanese vehicles are estimated in VEPM by drawing 'equivalences'. This approach may introduce large uncertainties that have yet to be quantified. Moreover, the emission factors in the VEPM require updating.

Part Two considers exposure to emissions while travelling in cars. Relatively little is currently known about exposure to traffic emissions while inside vehicles, and off-the-shelf assessment tools did not exist at the time of the research.

Most of the studies on the health effects of air pollution have been based on the assumption that the level of air pollution at a person's home represents their personal exposure. However, personal exposure to traffic pollution is strongly influenced by the time spent within close proximity to busy traffic, including the time spent in cars.

Our research on in-vehicle exposure conducted within this project consisted of three principle activities:

1 Collection of observational data

2 Analysis of this data

3 Initial development of an initial air and particles exchange model (APEX).

Concentrations of ultrafine particles were measured and modelled due to their direct health relevance, strong association with vehicle exhaust, the availability of sensitive, portable instrumentation, and to represent exhaust emissions in general.

Measurements of ultrafine particles were conducted in vehicles driving over a range of roads in inter-peak traffic periods across Auckland during a period of five days. Journey-mean concentrations varied in magnitude between 'trips'. We estimated in-vehicle concentrations were greater on average than ambient 'urban background' levels (ie at non-trafficked urban locations in Auckland). Concentrations within the vehicle cabin were typically 80% of those immediately outside the cabin. This fraction was reduced to 24%–30% if the vehicle's vents were set to recirculate the air. In-vehicle exposure was generally higher during periods of motorway driving, although we did not have enough data to comment on how consistent or systematic this effect was.

An APEX model was derived to predict concentrations of air pollutants inside a vehicle moving in traffic if the external on-road concentrations were known. Elevations in on-road concentrations were related to periods of motorway driving, passing through busy intersections, encounters with gross emitting vehicles and uphill gradients.

APEX requires a single parameter – the vehicle's air exchange rate (AER). Two experimental studies were conducted to quantify the AER for two cars. We found the AER was influenced by:

- opening windows
- ventilation fan speed
- vehicle cabin volume and/or air tightness
- closing air vents
- vehicle speed
- air conditioning.

Exploratory analysis using APEX suggested opening windows had only a minor influence on in-car exposure, but recirculating air could substantially reduce exposure. The amount of reduction appeared to be sensitive to the traffic characteristics (volume, speed, flow intermittency) of the route taken. Further research is required to quantify these relationships.

In each case, elevated in-car exposures arose from polluted air being injected into the vehicle cabin and then 'trapped', with internal concentrations exceeding external values for up to a few minutes (the magnitude and duration being dependent upon the AER).

The APEX model is a research tool which, on its own, is neither suitable nor intended for informing policy directly. However, it is designed to form the technical underpinning of a policy-relevant tool to be developed outside the scope of this project.

Abstract

Exposure to elevated concentrations of road traffic air pollutants mainly occurs within a few hundred metres of major roads, or while travelling in road vehicles.

Existing roadside air assessment tools are either crude and conservative, or are complex and demanding with no guarantee of improved accuracy. Neither approach is well suited to health risk assessment and both present substantial uncertainty for regulatory use.

A third approach was developed aimed at delivering moderate accuracy in a simple, accessible package, better suited to health risk applications. The roadside corridor model is a parameterised implementation of a more complex emission-dispersion model. It is implemented as a spreadsheet, suitable for integration into a GIS-based tool. Several practical applications of the model are demonstrated, including road project assessment, risk 'corridor' definition, and disaggregation of local and remote sources in roadside air quality monitoring data.

Much less is currently known about exposure inside vehicles. Ultrafine particles were measured in cars on Auckland roads. Concentrations varied over an order of magnitude between 'trips' and were larger than outdoors at non-trafficked locations. Concentrations within the vehicle cabin were reduced when the air was recirculated. Exposures were generally higher during periods of motorway driving.

PART 1: ROADSIDE CORRIDOR MODELLING

1 Introduction to roadside corridor modelling

1.1 The project

In its investigation of tools for assessing exposure to land transport emissions, this research project focused on two main 'modes' of exposure:

1. The local elevation in concentrations of ambient air pollutants in the vicinity of major roads (the roadside corridor)
2. Exposure to road traffic emissions while travelling in road vehicles.

The key aim was to develop and demonstrate a general method for exposure prediction in both of these situations, and to encapsulate the methods in simple, semi-empirical predictive models. The intention was that these models could be embedded in more comprehensive multi-modal exposure assessment tools in the future.

The project forms part of an ongoing interest in developing an operational integrated transport exposure model (ITEM) to predict, explain and analyse exposure to traffic emissions for the purposes of:

- epidemiological research
- health impact assessment
- transport and land-use project, policy and strategy assessment
- transport and land-use modelling and planning
- vehicle and roadway design.

There are several existing methods for roadside corridor assessment. However, these methods suffer several limitations, which can manifest in inconsistency and (in some cases) excessive and unhelpful complexity. The first aspect of the project focused on developing a new approach for roadside corridor assessment based on simplicity and robustness, with an emphasis on supporting health risk assessment and public health research in the future.

In contrast, much less is currently known about exposure to traffic emissions while inside vehicles, and off-the-shelf assessment tools did not exist at the time of the research. For this reason, our research on in-vehicle exposure (the second aspect of this project) was focused at an earlier stage of development – principally gathering observational data to inform a first attempt at deriving a general in-vehicle exposure model (IVEM).

1.2 Aim and content of Part 1

The first part of the report summarises the results of the roadside corridor research. It describes the road corridor model (RCM), the philosophy behind the approach and how it differs from alternative and existing approaches. Part 1 also illustrates a number of potential applications of the RCM in the form of simple case studies and demonstrations and finally discusses potential future developments.

Appendix A describes the RCM and its development in detail.

2 What is the roadside corridor?

2.1 The influence of major roads on local air quality

Over the last few decades numerous research studies have investigated the impact of road traffic on air quality. Vehicle emissions are rapidly dispersed and diluted, so that concentrations downwind reduce relatively rapidly. The rate at which the concentration reduces with distance is highly variable, depending especially on meteorological conditions, but also on the aerodynamic properties of the environment (eg whether it is open, vegetated or built up).

Many studies have tried to describe typical or indicative dispersion curves which predict the reduction in concentrations as a function of perpendicular distance to a road. This can be achieved in two ways through:

- analysis of roadside monitoring data
- dispersion modelling.

The roadside corridor may be considered the zone within which concentrations are significantly elevated compared with what they would be if the road was not there, or compared with locations further from the subject road (ie urban background locations). Considering different pollutants can lead to different corridor widths due to instrumental limitations and the ease of identifying urban background concentrations.

Zhou and Levy (2007) reviewed 33 studies which provided a meta-analysis of the spatial extent to which roads affect local air quality, and noted this extent was dependent on the nature of the pollutant considered. They found median values of 140m for inert pollutants with a low background (eg carbon monoxide or elemental carbon), 350m for reactive pollutants formed in the atmosphere (eg nitrogen dioxide) and 175m for reactive pollutants removed from the atmosphere (eg number concentration of ultrafine particles).

Because of the dominant role of meteorology in dispersion, general dispersion curves derived for one climate are not necessarily directly applicable to a different climate. In New Zealand, generalised dispersion curves are provided, based on dispersion modelling, in the *Good practice guide for assessing discharges to air from land transport* (MfE 2008). These curves show concentrations beyond 100m from the road being less than 10% of those at the roadside, and concentrations at 200m from the road being effectively indistinguishable from urban background concentrations.

These analyses can be supplemented by literature on the health effects of living near major roads (or in some cases health effects on children attending roadside schools). Such literature has rapidly increased in volume in recent years, leading to several review papers. For example, Brugge et al (2007) reviewed literature on cardiac and pulmonary risks and concluded health effects had generally been reported to a distance of about 200m. They also noted some recent studies which considered smaller geographical areas and pollutants reflecting local gradients (such as particles from traffic sources, rather than more regional pollutants and sources) had greater associations for cardiovascular outcomes. Data for lung cancer is scarcer but also suggestive of an association.

Riedl (2007) reviewed studies on the effect of air pollution on asthma and allergy and noted recent progress in explaining the toxicological mechanisms which provide plausibility to the observed epidemiological associations. In particular, Riedl noted how ‘exposure to diesel exhaust particles may convert asymptomatic individuals to symptomatic ... by lowering the allergen threshold necessary to produce symptoms'.

Boothe and Shendell (2008) reviewed 29 road proximity studies which 'consistently reported statistically significant associations between residential proximity to traffic and at least one of the following adverse health effects: increased prevalence and severity of symptoms of asthma and other respiratory diseases, diminished lung function, adverse birth outcomes, childhood cancer, and increased mortality risks'. Furthermore, this review concluded 'The consistency of reported results across the studies we reviewed provided a "weight-of-evidence" finding, suggesting that residential proximity to traffic can be associated with adverse health effects and poses a public health threat'.

2.2 A brief review of need for road transport exposure tools

2.2.1 Health risk assessment

Research has highlighted that extended periods of time spent in close proximity to transport emissions is a major cause of illness (eg Brugge et al 2007), with a burden on society as great as that from road traffic accidents (reported in the *Health and air pollution in New Zealand* (or HAPINZ) report, Fisher et al 2007). This burden, however, remains only crudely quantified at present. For instance, the HAPINZ method derived its risk estimates from rather crude estimations of traffic-related PM_{10} which were empirically predicted by the total volume of traffic in a given census area. A large volume of international studies, especially in the field of epidemiology, have recently indicated how such a method may substantially underestimate the serious risk traffic emissions present to persons chronically exposed as a result of living within a major road's 'corridor of influence'. Underestimation of risk is likely to arise whenever estimates:

- are related to PM_{10} rather than the suite of pollutants more directly associated with traffic sources
- are related to acute (short-term) exposures rather than chronic exposure
- are restricted to acute health outcomes and ignore associations with chronic conditions
- do not account for strong concentration gradients within approximately 100m–200m of the road
- do not consider the effects of childhood exposure.

New risk assessment studies which incorporate some of these factors are now showing that risk underestimation can be addressed where long-term multi-pollutant micro-scale exposure assessment data is available (eg Künzli et al 2008).

2.2.2 Epidemiological/toxicological research

There is a large and rapidly expanding literature base regarding the epidemiological associations between living near major roads and adverse health effects. These associations are not necessarily causal, but the toxic effect of vehicle emissions is strongly suspected to have a major, if not dominant, role. However, substantial uncertainties remain and causality cannot yet be confidently ascribed to vehicle emissions.

The remaining uncertainties can be related to two major issues:

1. exposure misclassification
2. pollutant speciation.

Exposure misclassification refers to the errors inherent in exposure assessment. These errors arise due to limitations in accurately modelling vehicle emissions and their dispersion, indoor infiltration and the treatment of confounding from indoor sources. This research project dealt specifically with errors arising in dispersion modelling.

Pollutant speciation refers to the fact that vehicles emit a wide range of pollutants, whereas exposure assessments are often limited to a single pollutant (commonly nitrogen dioxide, black carbon or mass of particulate matter). Each pollutant predicts a different exposure resulting in a different risk, whereas the real exposure and risk relates to the combination of pollutants, and perhaps to the synergistic risk of multi-pollutant exposure (eg exposure to one contaminant increases the sensitivity or response to another). For these reasons more information is required on the general properties of roadside dispersion.

To date, all of the epidemiological evidence comes from outside New Zealand. It is reasonable to question whether data from abroad applies here. There are several factors that would suggest dispersion in New Zealand's cities may be different from foreign cities, especially those with different climates (such as Southern California, where much of the current epidemiological evidence originates). New Zealand's cities also have quite characteristic building designs and land use, and many have complex topography. Local exposure and health risk assessments should be based on locally relevant dispersion data.

2.2.3 Operational management of highways

One of the needs for which this research was relevant was to help identify locations where the impact of transport emissions is significant. The RCM should not be relied on to do this, but be used in conjunction with observational validation.

2.2.4 Project management and policy making

The NZTA recently developed a draft Standard for Producing Air Quality Assessments for State Highway Projects (NZTA 2010). The draft Standard requires new road projects (including upgrades of existing roads) to undergo a screening assessment. The Standard states:

> *The purpose of the screening assessment is to identify, as part of a general assessment of potential social and environmental issues, whether there are any air quality risks or opportunities arising from a proposed roading project.*

The Standard then requires the project to be assessed for comparison with a number of thresholds which will trigger a second tier of assessment (the scoping assessment). The thresholds are somewhat arbitrary at present. This research project was intended to lead to an improved definition of these thresholds.

In terms of policy making, beyond implementing vehicle emission standards, much of the health burden arising from exposure to vehicle emissions can, in principle, be reduced by targeting policies and schemes where exposure is greatest and by adopting exposure reduction as an indicator of sustainable transport policy. Anticipated future traffic growth suggests decisions to reduce this burden should be taken as soon as possible.

3 Approach and capabilities of the roadside corridor model

3.1 Existing approaches to modelling roadside air quality

A range of different modelling approaches and specific modelling tools can, and have, been used to predict roadside air quality. We briefly review these approaches here in terms of their suitability for the applications identified in section 2.2, and a more detailed technical review is provided in appendix A.

In brief, commonly available modelling options consist of (in order of decreasing complexity):

- emission-dispersion models
- regression models (spatial or temporal)
- simple banding or interpolation.

Emission models predict emissions arising from a given vehicle fleet (considering the mix of types, fuels and ages of vehicles comprising the fleet) with given traffic characteristics (such as speed and proportion of cold engines). Emission models such as the vehicle emissions prediction model (VEPM) incorporate a fleet model which permits prediction of emissions for a given year within a range. The fleet data the VEPM requires as input is often not available, although the VEPM does provide default values. The VEPM is complex and embodies numerous assumptions (which imply uncertainties) that are not immediately apparent to users.

Dispersion models predict the dilution of the vehicle emissions within the roadside corridor in response to a given meteorological condition. Ambient dispersion is complex, and sensitively dependent upon many meteorological and aerodynamic parameters. One of the major weaknesses of dispersion models is the necessary meteorological data and parameters are often not available, leading to uncertainties in model output.

Combined emission-dispersion model pairs are conventionally used for roadside air quality assessment in the case of regulatory assessments, but have been less frequently used in health research and assessment due to their being seen as ‘air quality’ rather than ‘public health’ tools, and requiring expert users. Their main advantages are the ability to make predictions about roads which do not yet exist, and their explicit physical consideration of meteorology. Their main disadvantages are their complexity (which implies the need for expert skills to use, review and audit), relatively high operational costs, high data demands and multiple sources of uncertainty.

Regression models (also known as correlation models) are relatively simple compared with emission-dispersion models. They empirically relate observed variation in concentrations (temporal or spatial) to easily available explanatory variables. For temporal regression models the explanatory data will often be hourly variations in traffic and meteorological parameters. For spatial regression models the explanatory data is often spatial distributions of traffic, land use or terrain/topography. Spatial regression models (or land-use regression models) have been rapidly adopted for health-oriented research and for screening purposes in recent years, enabled by developments in geographic information systems (GIS). They are generally considered not to offer sufficient accuracy for regulatory assessment.

The main advantage of regression models is their simplicity, requiring less expertise of users and providing fewer opportunities for inconsistencies to arise. Crucially, regression models are based on site-specific observational data. Their main weakness is they cannot automatically be applied with confidence to other locations without further observational validation (and often adjustment). As they do not explicitly consider emissions or dispersion, they cannot be used to consider alternative emission scenarios, or to consider temporal or spatial variations in meteorology. This limits their ability to make predictions about future projects. Also, spatial regression models typically focus on long-term averages rather than short-term peaks.

A further two model ‘types’ are currently available:

- parameterisations of emissions-dispersion modelling
- hybrid models.

A parameterised model considers a range of emission-dispersion modelling results and statistically analyses them. The simplest approach is to average all likely results, which is effectively the approach behind the general dispersion curves included in the *Good practice guide for assessing discharges to air from land transport* (MfE 2008). The RCM developed in this project is the same type of model, but defining 192 general dispersion curves rather than one.

Hybrid models combine some components of the types previously discussed. For example, several researchers have experimented with combining emission-dispersion modelling with land-use regression modelling to develop a land-use regression model which incorporates local meteorology. Further research has investigated incorporating temporal components into land-use regression models.

3.2 RCM technical objectives

Considering the needs identified in chapter 1, and the capability gaps discussed in section 3.1, the specific technical objectives of the RCM are to:

- focus primarily on prediction of long-term impacts as the primary need of health impact assessment
- create a model that is as simple as possible, relying on as few input variables as practical, and relying on widely available input data as much as possible
- represent the importance of prevailing wind climates by making independent predictions for either side of a given road and considering the road's orientation with respect to prevailing winds as an input variable
- enable prediction based on alternative emissions.

The need for simplicity combined with consideration of emissions and meteorology led us to develop the RCM following a parameterised emission-dispersion model approach.

3.3 Rationale for adopting an alternative approach to roadside modelling

The conventional approach of using emission-dispersion modelling to predict the impact of traffic emissions on roadside air quality involves a minimum of three models: a vehicle emission model to convert traffic operational data (particularly speed) into emission factors; a vehicle fleet model which weights these emission factors depending on fleet composition for a given year; and an atmospheric dispersion model which predicts the resulting ambient concentrations arising from the emissions at prescribed points known as receptors.

The conventional approach has a long history and is the default approach in most cases. It is particularly well suited to the study of a single or small number of relatively simple road links in data-rich environments, where detailed modelling products are required. It is less well suited to the study of large networks of links because it requires a new dispersion model 'run' for every required traffic/emission/ meteorological scenario. This can amount to tens or hundreds of cases, each with significant data demands and large volumes of data output. If the intention is to model many road links for multiple years, the demands on manipulating the resulting modelling input and output data can be considerable. There is also a great deal of unnecessary duplication involved in this case. The conventional approach is also less suitable when the demands on the modelling are relatively relaxed, for instance where easily achieved indicative results may be preferable to results which are detailed and accurate but come at a high premium in terms of data and effort required.

3.4 Our approach to model development

Our aim in this project was to develop an alternative approach better suited to applications where the demands on accuracy and detail were more relaxed, but a greater coverage might be required. Inherent in our approach was the desire to make the model as easy as possible to use, in order to allow a variety of end users to utilise it while minimising the quantity of input data required for the maximum possible number of model scenarios. This inevitably meant some fidelity had to be sacrificed in return for speed and accessibility.

The RCM was based on the concept that, in the long term, variation in many of the input variables utilised in conventional emission-dispersion modelling actually contributed very little to the variation in output results and the modelling outcome. We therefore sought to identify which variables were relatively trivial and which variables were crucial through sensitivity tests. Once several input variables had been discarded, only those variables with the largest influence on end results remained. Furthermore, random and systematic variation was averaged at several stages during the model development, thus sacrificing information about variation from the average in the interest of simplicity.

3.5 Objectives arising from the limitations imposed by our modelling approach

The alternative approach described above imposes limitations on the capability and applicability of the RCM model. These limitations circumscribe, and inform, the appropriate modelling objectives:

1. The model is aimed at predicting **long-term impacts**. This typically means annual averages. As the model is based on generalised average dispersion curves it is inherently unable to describe short-term deviations from the norm, such as peak impacts. However, by extending the complexity of the model, it could be extended to address **typical** short-term impacts.
2. The model applies to **urban areas only**. This allows us to remove one of the determining variables of dispersion (aerodynamic properties of the surface, including vegetation, buildings etc). The ability to consider rural locations could be added in the future, at the cost of increasing model complexity.
3. Predictions are restricted to **passive contaminants** only, for which emission rate information is available[1]. This means the model does not predict concentrations of contaminants which are likely to undergo chemical or physical transformation within the corridor width (up to a few hundred metres), also to maintain model simplicity. The model is not suitable, for example, for predicting nitrogen

[1] The RCM is currently formulated to gather emission data from the VEPM which limits predictions to PM_{10}, NO_x, CO, VOC.

dioxide (NO_2) and ultrafine particles. The National Institute of Water & Atmospheric Research (NIWA) is conducting research outside this project to inform how these species could be added to the RCM.

4 The model is necessarily limited to describing 'typical' dispersion in 'typical' areas. Atypical areas for which the model (in the form in which it was developed within the scope of this project) is not designed, and where its performance may be weak, include central business districts (CBD) or any location characterised by tall or large buildings and street canyons, areas with significant or complex terrain and locations subject to local air flows and micro-climates (such as coasts, isolated ridges and basins etc). One consequence of this is the model is not intended to provide accurate predictions for a specific small-scale location (eg a single road) or for intersections.

3.6 Additional objectives

Even a brief inspection of wind roses from any location shows most are subject to systematic patterns in wind speed and direction which are characteristic of that location. There is clearly the potential for this to introduce distinct directional biases on pollutant dispersion in the long term. To take an extreme example, if a given location with a north-south oriented road only ever experienced westerly winds, then the impact of that road on air quality would be restricted to its easterly side only. Previous research abroad has shown modelled long-term exposure estimates can be improved if long-term bias in wind patterns is taken into account (eg Arain et al 2007). For this reason, an additional modelling objective for the RCM is to be able to incorporate the effect of local wind climate and provide independent estimates of exposure for both sides of the road.

4 Overview of the roadside corridor model

4.1 Project outputs

The roadside corridor model (RCM) is not a new model, nor a development of an existing model. It is truer to describe the RCM as a parameterisation of the output from combining a suite of existing models. The combination we have chosen is not the only possible combination (see below), and in the future the applicability and fidelity of the RCM may be improved by substituting alternative model components.

The core of the RCM is a generalised dispersion curve which is described by two parameters (a coefficient and an exponent). Values for these parameters depend upon the road orientation and the time period (24 hours, am peak, pm peak or inter-peak), and are stored within the model in a look-up table for various combinations. The look-up table permits specification of the dispersion curve for each combination of road orientation and time period, but the curve alone does not provide an exposure estimate. Determining an exposure estimate requires weighting of the curve by emissions and traffic data; a spreadsheet interface has been developed which permits this step for a single road as well as providing for user input and output of numerical results.

NIWA has also developed a demonstration GIS-based web interface. This interface has partial functionality, and its main purpose has been to inform future development of a user interface for the RCM.

4.2 Current and possible future model architecture

The key components of the RCM are:

- a core dispersion model
- an emissions model
- a user interface.

The RCM requires the following input data:

- meteorological data
- traffic volume data (annual average daily traffic ((AADT)) and average am peak, pm peak and inter-peak volumes)
- other traffic data as required by the emissions model (at least fleet composition and average speed in the case of the VEPM).

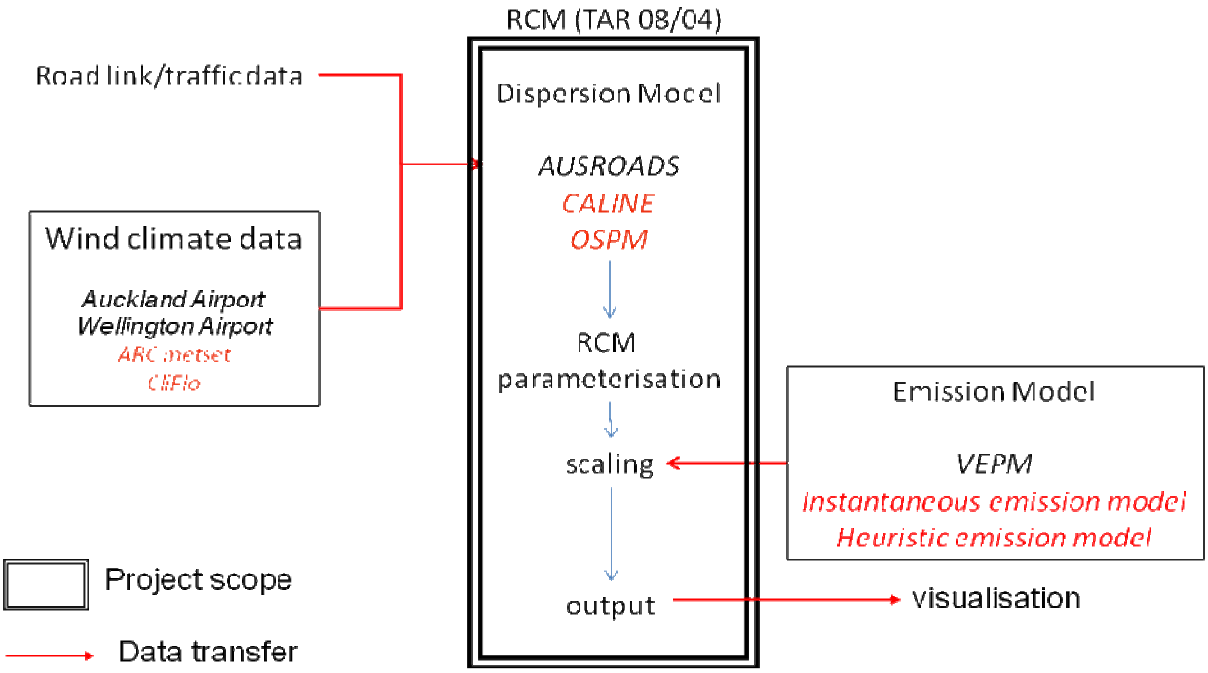
Figure 4.1 illustrates the model architecture. The version developed within this project is configured using as the dispersion model and VEPM as the emissions model, and is based on two meteorological datasets of observations at Auckland and Wellington airports. Alternative components are highlighted in the red text in figure 4.1, and are discussed further below.

Dispersion model: The AusRoads model is effectively the default choice for roadside dispersion modelling in Australia and New Zealand. The model was developed by EPA Victoria and is freely available. It is based on the American CALINE model, which has no user interface but is probably more suitable for future model development. The operational street pollution model (OSPM) is a street canyon semi-empirical dispersion model, and its future inclusion into the RCM should extend the RCM use to CBD locations. Other dispersion models are also available which could be substituted for the AusRoads model.

Emission model: The VEPM was recently developed and is the default emissions model for use in New Zealand. At present, use of the RCM requires the VEPM to be run offline and the data to be transferred to RCM manually. Further software development could automate this process making RCM much faster to use. The VEPM is designed to predict emissions over a drive-cycle average, not on certain types of roads. If road-specific emissions are required the VEPM could, in principle, be replaced by a more appropriate model. Such a model might be one of two types – an instantaneous model (predicting emissions on a second-by-second basis), or a heuristic model (a self-validating, learning model based on detailed in situ observational data). Neither of these model types is currently available to describe the New Zealand fleet.

The VEPM depends heavily on European emission factors, but the majority of our light vehicles were built to Japanese emission standards. The emissions of these Japanese vehicles are estimated in VEPM by drawing 'equivalences'. This approach may introduce large uncertainties that have yet to be quantified. Moreover, the emission factors in the VEPM require updating.

Figure 4.1 Current (black) and possible alternative (red) configurations of RCM components



Wind climate data: For the purposes of developing the RCM we selected two meteorological datasets to explore the sensitivity of the output to dataset choice. Meteorological data is generally available in two forms: i) observational data, and ii) 'artificial' data (ie data derived from detailed meteorological modelling). A large volume of meteorological data is hosted by NIWA's CliFlo climate database. Allowing the user unchecked choice of data, however, could lead to considerable inconsistency and it is for this reason artificial datasets are often generated. Auckland Regional Council, for example, has funded the creation of a standard dataset for Auckland. Engineering an interface with meteorological datasets, fully exploring the sensitivity of RCM output to the choice of dataset, and ensuring validity and consistency of the dataset are all substantial projects with significance and application well beyond the scope of this project.

4.3 Development of the RCM

The process of developing the RCM is described in full in appendix B. In brief, AusRoads was used to model the response of emissions from a single road to the Auckland and Wellington meteorological datasets at receptors on either side of the road for 24 different road orientations. Sensitivity tests were then conducted to assess the influence of:

- orientation
- one year versus two years of meteorological data
- 'motorway' versus 'arterial' diurnal traffic profiles
- differences between am peak, pm peak, inter-peak and full 24-hour results
- Wellington versus Auckland meteorology.

The results showed orientation, time of day and location (Auckland/Wellington) were the most significant explanatory variables. The model also succeeded in predicting significantly different concentrations on either side of the road (for certain orientations).

4.4 The RCM parameterisation

The core of the RCM is a pair of general power-law dispersion curves (one for each side of the road) which predict the long-term average pollutant concentration resulting from the road as a function of perpendicular distance from its centreline. This gives an estimate of the concentration elevation on either side as a result of the road (ie the concentration over and above what would be observed if the road were not there).

Each dispersion curve is described by two parameters (a coefficient *A* and an exponent *B*, each of which depend upon the orientation, time of day and location) and is weighted by the traffic volume (*V*) and emission factor (*EF*). The equation describing each curve is:

$$\text{predicted concentration} = V \times EF \times A \times \text{distance}^{B} \qquad \text{(Equation 4.1)}$$

Values of *A* and *B* for each side of the road for various orientations, time periods and locations are determined by the AusRoads modelling and are listed in look-up tables in the RCM. These tables are reproduced in appendix B. The emission factor (*EF*) is determined by the VEPM.

To predict the corridor width on either side of the road for a given concentration elevation threshold, the formula above is re-arranged to give:

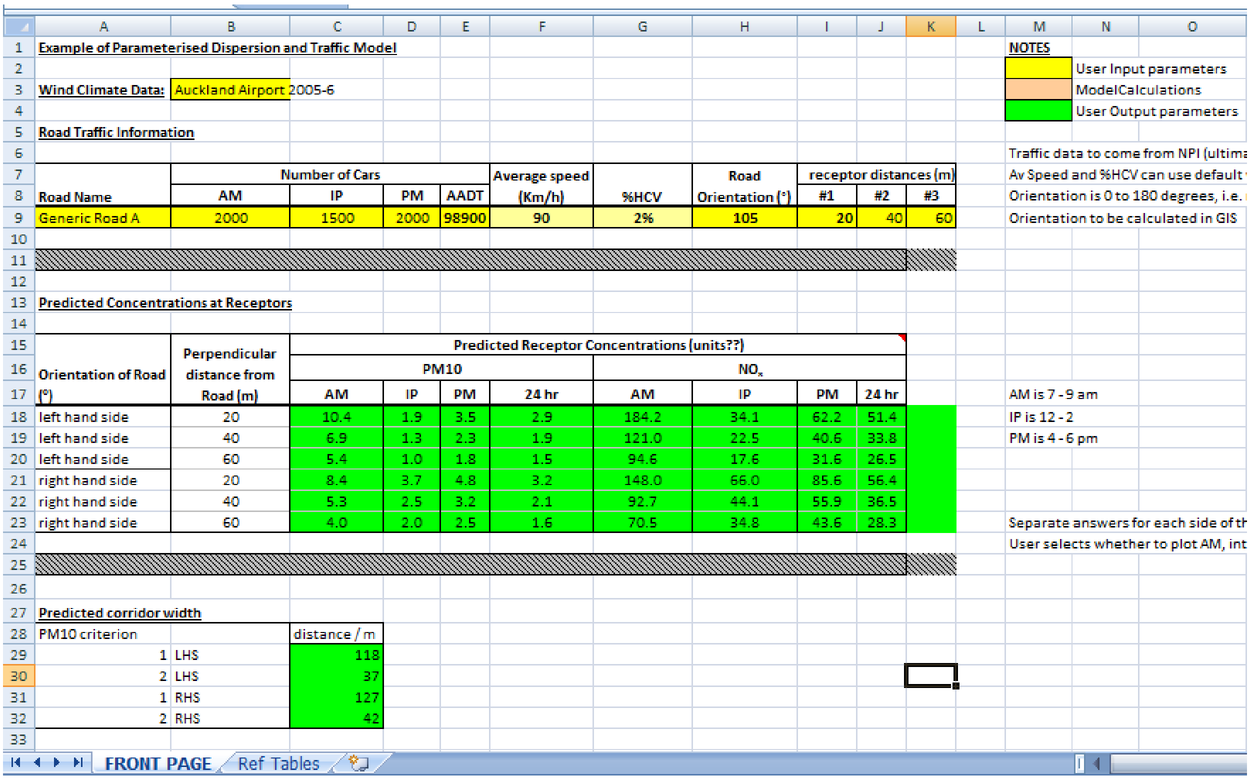
$$\text{corridor width} = \left(\frac{\text{threshold concentration}}{V \times EF \times A} \right)^{1/B} \qquad \text{(Equation 4.2)}$$

4.5 The host spreadsheet

Figure 4.2 shows a screenshot of the current version of the spreadsheet provided for user input to and output from the RCM. Note there is plenty of scope for improving the user interface. There are currently two forms of output (in green):

1. The predicted concentrations due to the road at certain distances
2. The predicted corridor widths for certain threshold concentrations.

Figure 4.2 Screenshot of the roadside corridor model implemented in a spreadsheet



Example of Parameterised Dispersion and Traffic Model

Wind Climate Data: Auckland Airport 2005-6

Road Traffic Information

Road Name	Number of Cars AM	IP	PM	AADT	Average speed (Km/h)	%HCV	Road Orientation (°)	receptor distances (m) #1	#2	#3
Generic Road A	2000	1500	2000	98900	90	2%	105	20	40	60

Predicted Concentrations at Receptors

Orientation of Road (°)	Perpendicular distance from Road (m)	Predicted Receptor Concentrations (units??) PM10 AM							
		PM10				NO_x			
		AM	IP	PM	24 hr	AM	IP	PM	24 hr
left hand side	20	10.4	1.9	3.5	2.9	184.2	34.1	62.2	51.4
left hand side	40	6.9	1.3	2.3	1.9	121.0	22.5	40.6	33.8
left hand side	60	5.4	1.0	1.8	1.5	94.6	17.6	31.6	26.5
right hand side	20	8.4	3.7	4.8	3.2	148.0	66.0	85.6	56.4
right hand side	40	5.3	2.5	3.2	2.1	92.7	44.1	55.9	36.5
right hand side	60	4.0	2.0	2.5	1.6	70.5	34.8	43.6	28.3

Predicted corridor width

PM10 criterion		distance / m
1	LHS	118
2	LHS	37
1	RHS	127
2	RHS	42

NOTES
User Input parameters
ModelCalculations
User Output parameters
Traffic data to come from NPI (ultima
Av Speed and %HCV can use default
Orientation is 0 to 180 degrees, i.e.
Orientation to be calculated in GIS
AM is 7 - 9 am
IP is 12 - 2
PM is 4 - 6 pm
Separate answers for each side of th
User selects whether to plot AM, int

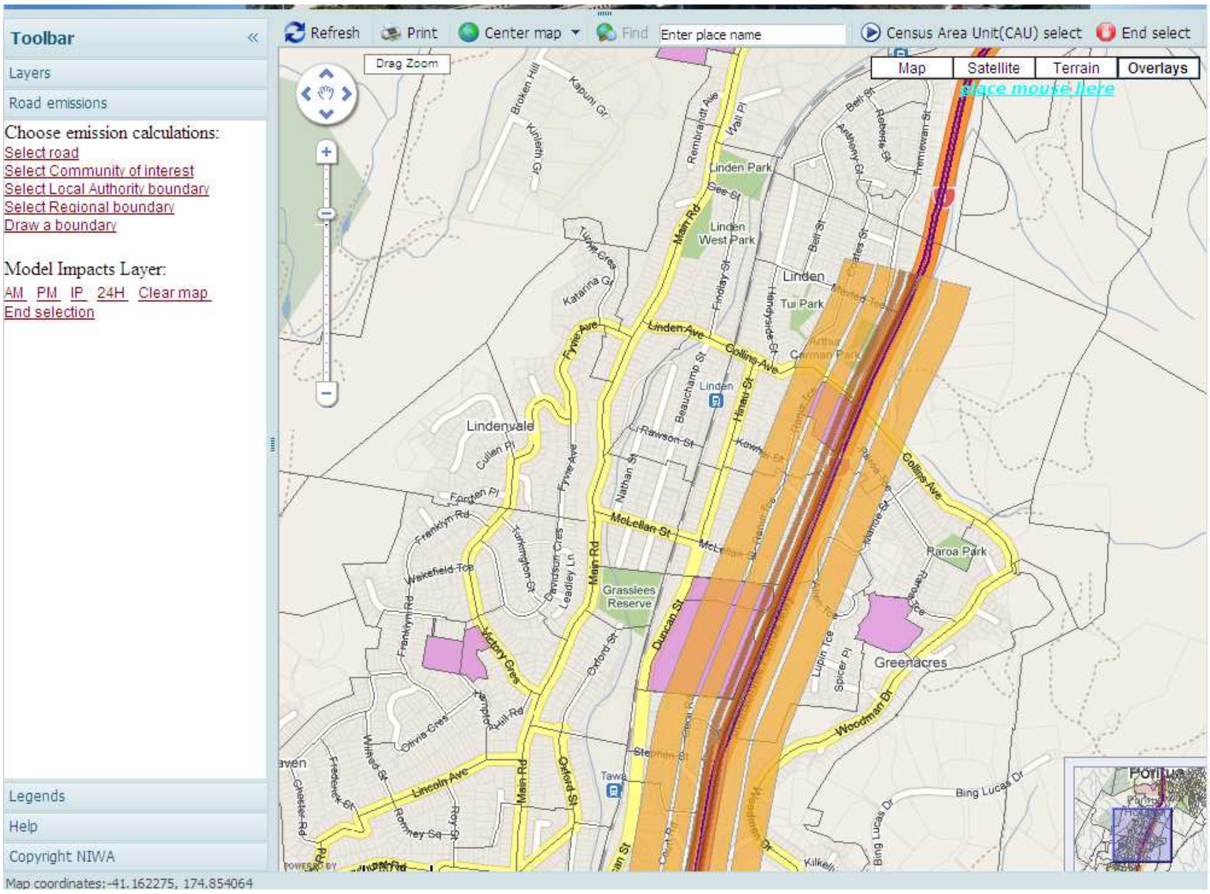
FRONT PAGE Ref Tables

4.6 A demonstration GIS/web-based model host

NIWA has developed a demonstration GIS-based web interface for the RCM. The tool does not possess full functionality at present; it allows the user to select one, or multiple road sections, and plot coloured regions representing predicted concentration elevation bands (of a user-selected pollutant) due to the selected roads. Figure 4.3 shows a screenshot of the demonstration web interface.

The main potential for this tool is to overlay the predicted corridor for a given concentration elevation threshold onto other GIS-based layers, particularly population data, but also land-use data (such as schools and other sensitive receptors). Further integration can also be achieved if the traffic data is hosted on the same system.

Figure 4.3 Example screenshot of the demonstration GIS-based web interface. This example shows coloured regions around SH1 corresponding to three different PM_{10} concentration elevation bands



5 Project-specific applications and case studies

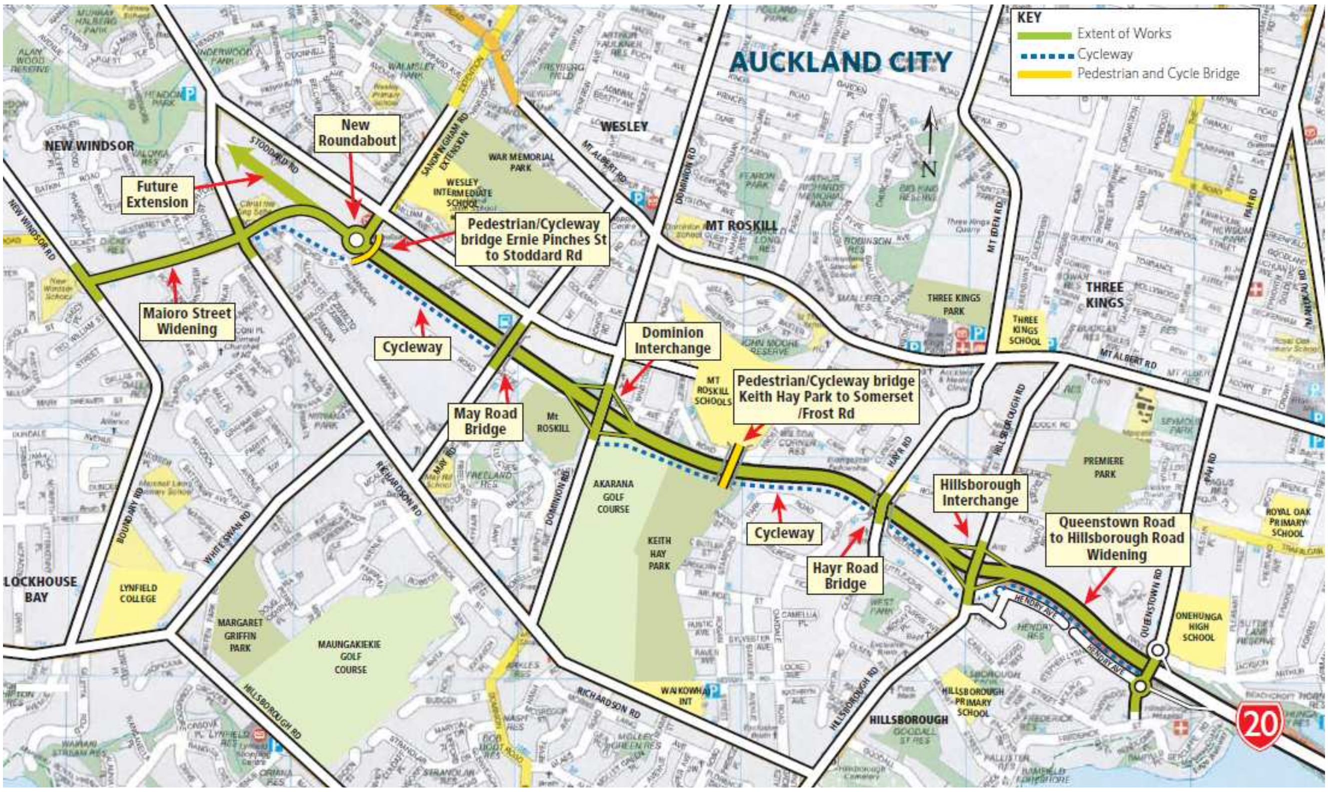
5.1 Road project case study: SH20 Mt Roskill Extension

5.1.1 Overview of the case study

In 2009 the SH20 Mt Roskill Extension in Auckland opened to traffic. This 3km section of the South Western Motorway begins at Hillsborough Road, and currently ends at a roundabout where two spurs connect the motorway to the Sandringham Road Extension and Maioro Street/Richardson Road (figure 5.1). The extension has been designed to form part of the Western Ring Route (WRR), which will be enabled by the continuation of the motorway to the north-west via the Waterview Connection linking the current terminus of SH20 to SH16 at Waterview. For the purposes of this case study we have also included the section of motorway between Hillsborough Road and Queenstown Road. Although this section existed before the extension it was substantially upgraded as part of the extension project.

This case study illustrates the application of the RCM to the SH20 Mt Roskill Extension, using projected traffic data for the year 2016 which was supplied to NIWA by Beca Infrastructure with the permission of the NZTA. This projected traffic data was created for the purposes of the air quality assessment for the SH20 Waterview Connection. We compare the RCM output for two scenarios related to the Western Ring Route: i) the ‘do minimum’ scenario (in which the Waterview Connection is not built), and ii) the Western Ring Route or WRR scenario (in which the Waterview Connection is built).

Figure 5.1 Map of the SH20 Mt Roskill Extension



5.1.2 RCM inputs

The RCM calculates concentration elevations for each side of a road section based on that section's orientation, average traffic volume, and fleet composition and speed. This implies road sections need to be straight (in order to specify an orientation) and have consistent traffic characteristics (ie not be broken up by intersections). The SH20 Mt Roskill Extension is mostly straight but includes two intersections (in addition to the Sandringham Rd and Queenstown Rd intersections which define the ends of the project). For this exercise the extension had already been split into eight separate links in the road/traffic shape file provided by Beca, and we used these links as the road sections. The meteorological dataset used was the Auckland Airport dataset, the traffic data used were those supplied by Beca for 2016 (as described above), and the input values used for the emission modelling component of the RCM (ie the VEPM) were the same as those specified by Beca for the Waterview Connection air quality assessment.

5.1.3 The 2016 do minimum scenario – RCM predictions

In this scenario (where the Waterview Connection is not built), the RCM was used to predict the corridor widths on either side of the SH20 Mt Roskill Extension for certain PM_{10} concentration elevation thresholds. The AADT value provided for this scenario was 67,100. A summary of the predicted corridor widths is given in table 5.1.

Table 5.1 Summary of predicted corridor widths (defined by PM_{10} elevation thresholds) on either side of the Dominion Rd to Hillsborough Rd section of the SH20 Mt Roskill Extension under the 2016 do minimum scenario

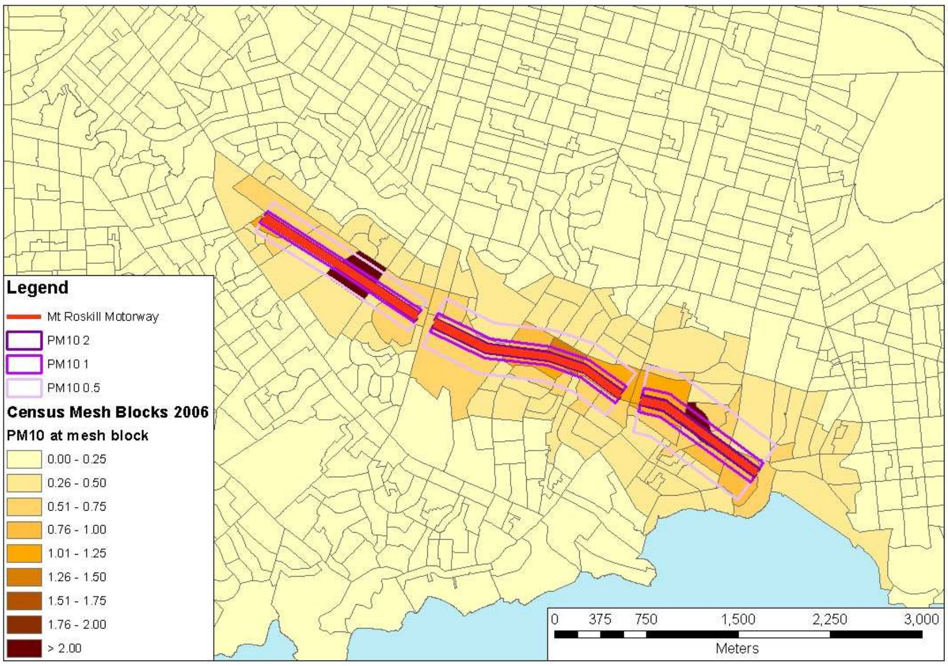
	Corridor width (m) for a PM_{10} threshold of		
	2 μg m^{-3}	**1 μg m^{-3}**	**0.5 μg m^{-3}**
South-west side	23	69	207
North-east side	20	62	195

The GIS-based RCM interface was used to plot the predicted corridor widths (shown as the purple lines parallel to the motorway in figure 5.2) and also to overlay the results on a population census meshblock layer. For each meshblock:

- the perpendicular distance from the motorway to the meshblock centroid was extracted and input to the RCM to predict the long-term average PM_{10} concentration elevation at that point, then
- the long-term average PM_{10} concentration elevation predicted at the meshblock centroid was ascribed to the entire meshblock.

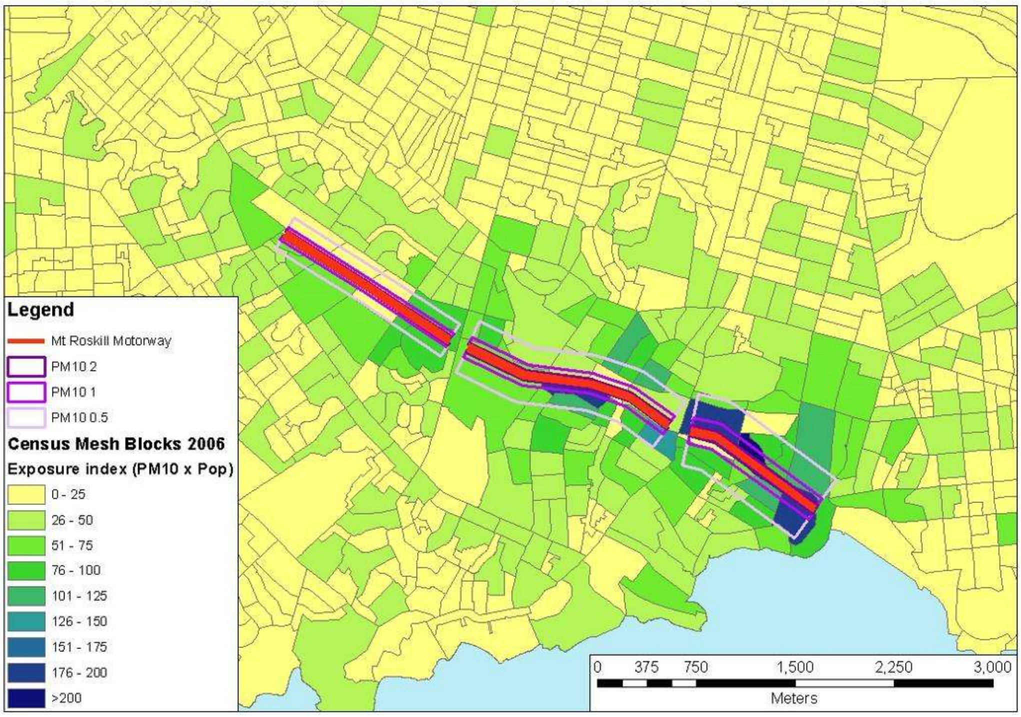
The population census meshblocks are shown outlined in grey in figure 5.2 and are coloured according to their RCM-predicted PM_{10} concentration elevation.

Figure 5.2 RCM output for the SH20 Mt Roskill Extension case study, 2016 do minimum scenario, showing elevation in annual mean PM_{10} ascribed to census meshblocks. Also shown (as purple lines) are the corridor widths for PM_{10} thresholds of 0.5, 1 and 2μg m^{-3}



When the RCM output can be overlaid with meshblock-level data using GIS then extended exposure analysis can be conducted. The residential population is described in census data, and several of the census meshblocks in the case study area have zero or very low populations (the motorway passes through an industrial/commercial corridor). Figure 5.3 shows the PM_{10} concentration elevation predicted for each meshblock multiplied by the normally resident population, effectively providing a population-weighted exposure estimate. By comparison with figure 5.2, it can be seen high PM_{10} concentration elevations at the western end of the motorway have a much diminished significance when population-weighted due to the industrial nature of those areas.

Figure 5.3 Population-weighted PM_{10} exposure (due to the SH20 Mt Roskill Extension, 2016 do minimum scenario) per census meshblock



5.1.4 The 2016 WRR scenario – RCM predictions

The 2016 WRR scenario (in which the Waterview Connection is built) leads to a substantial increase in traffic volumes for the SH20 Mt Roskill Extension (AADT = 91,500) but no substantial changes in fleet composition or average speed. Table 5.2 compares the RCM-predicted corridor widths on the south-west side of the SH20 Mt Roskill Extension under the WRR scenario with the corresponding values from the do minimum scenario. Figures 5.4 and 5.5 then show the PM_{10} concentration elevations ascribed to the population census meshblocks under the WRR scenario and the corresponding population-weighted PM_{10} exposure.

Table 5.2 Summary of the predicted corridor widths (defined by PM_{10} elevation thresholds) for the south-west side of the Dominion Rd to Hillsborough Rd section of the SH20 Mt Roskill Extension, comparing the do minimum and WRR scenarios

Scenario	AADT	Corridor width (m) for a PM_{10} threshold of		
		2 µg m^{-3}	1 µg m^{-3}	0.5 µg m^{-3}
Do minimum	67,100	23	69	207
WRR	91,500	37	112	339

Figure 5.4 RCM output for the SH20 Mt Roskill Extension case study, 2016 WRR scenario, showing elevation in annual mean PM_{10} ascribed to census meshblocks. Also shown (as purple lines) are the corridor widths for PM_{10} thresholds of 0.5, 1 and 2 μg m^{-3}

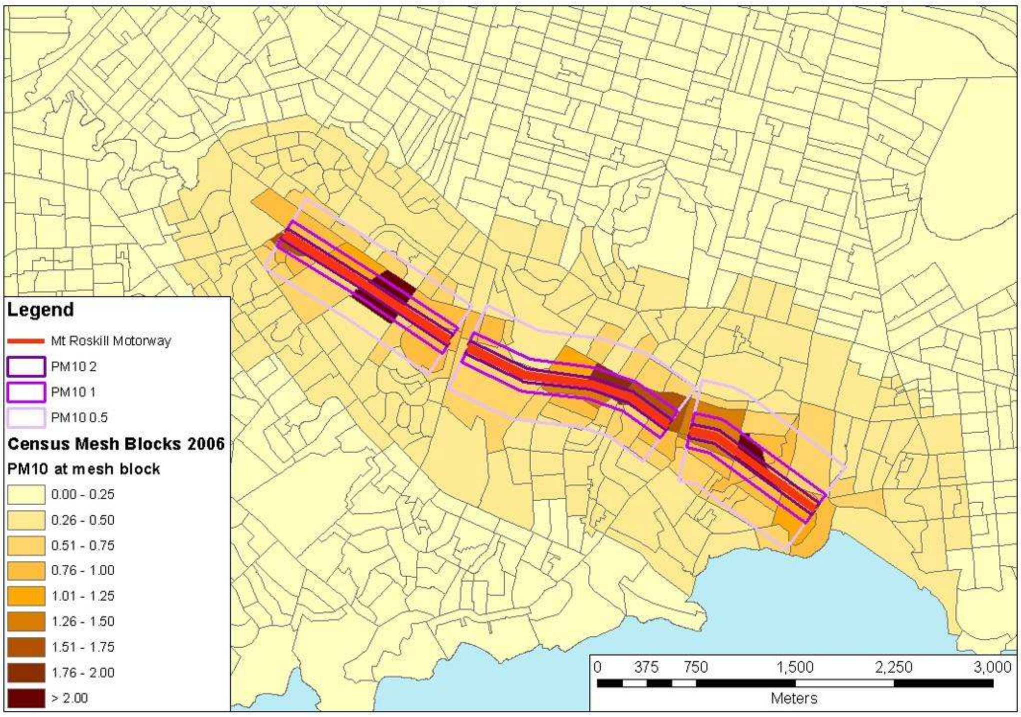


Figure 5.5 Population-weighted PM_{10} exposure (due to the SH20 Mt Roskill Extension, 2016 WRR scenario) per census meshblock

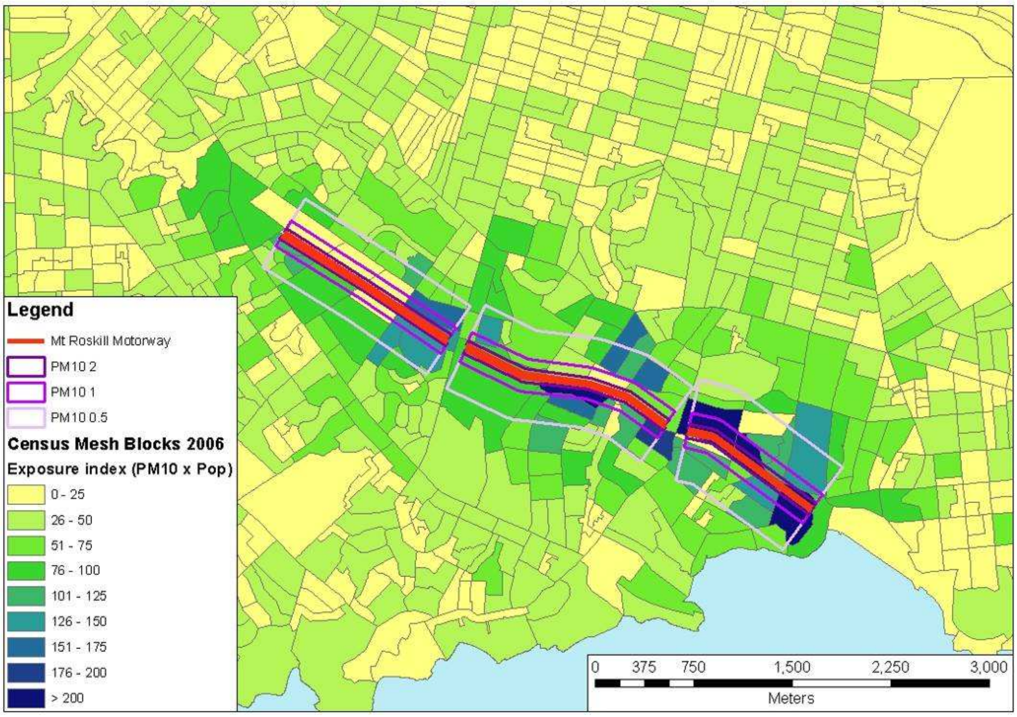


Figure 5.6 illustrates the population exposure distribution into various PM_{10} elevation bands for the two scenarios. Note the logarithmic scale for the exposed population (y-axis). The 0–0.25μg m^{-3} elevation band is not shown, as it generally relates to very distant meshblocks and the exposure in this band may be considered insignificant. Figure 5.7 shows the number of people in the population who would experience a change in exposure (ie move from one elevation band to another) between the two scenarios. For instance, as a result of the Waterview Connection being built (ie the WRR scenario), 1278 people are predicted to experience an increase in PM_{10} concentration of between 0.25 and 0.5μg m^{-3}, 183 people are predicted to experience an increase of between 0.5 and 0.75μg m^{-3}, and no one is expected to experience an increase larger than 0.75μg m^{-3}. Also, in this scenario no one is expected to experience a decrease in PM_{10} concentration; however, this would not necessarily be the case in every situation. For example, once other roads have been considered (especially those relieved of traffic by the project), increases due to the project may be offset by decreases due to the other roads.

Figure 5.6 Population exposure distributions into various PM_{10} concentration elevation bands for the SH20 Mt Roskill Extension, comparing the do minimum and WRR scenarios

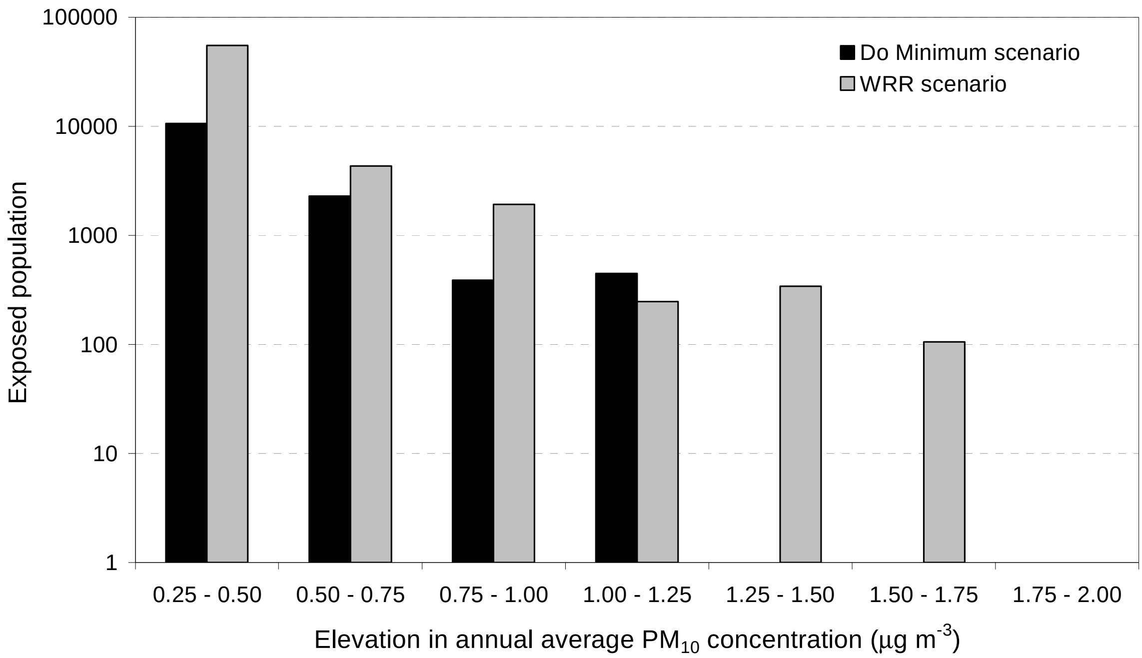
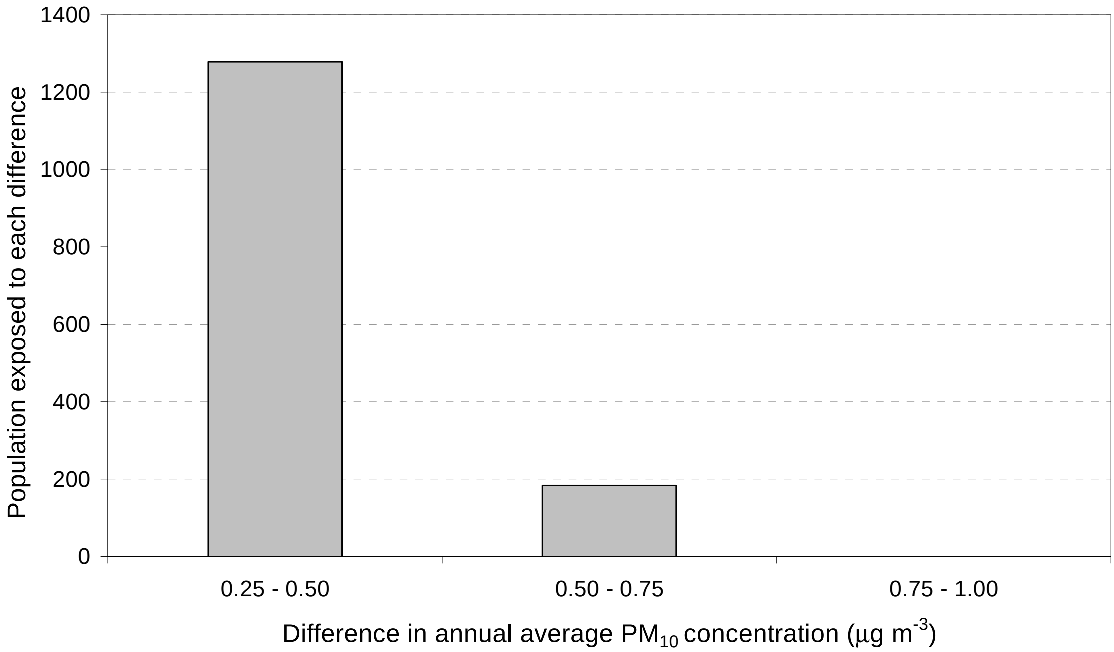


Figure 5.7 Differences in population exposure distribution between the do minimum and WRR scenarios



5.2 Traffic management: alternative emission profiles

Traffic management schemes or other actions, land-use changes or developments, and economic factors that lead to changes in traffic fleet composition or traffic speeds are all likely to lead to changes in emissions. Although these factors do not alter the dispersion characteristics, the resulting changes in emissions can be predicted using the VEPM. The impacts of such changes on roadside communities can be evaluated using the RCM, and GIS-based visualisation of the results can be informative.

An illustrative example is provided based on the section of Auckland's Southern Motorway (SH1) passing through Otahuhu. We compare a realistic scenario (based on 2006 traffic modelling data provided by Beca) with a hypothetical scenario in which total traffic volume remains unchanged (118,400 AADT) but the proportion of diesel heavy commercial vehicles (HCVs) is increased four-fold. Table 5.3 summarises the results for the east side of the motorway. The effect of this high HCV scenario is a two-fold increase in the PM_{10} tailpipe emission factor translating into a 2.8 times increase in the width of the roadside PM_{10} corridors.

Table 5.3 Summary of RCM results for the 2006 SH1 (Otahuhu) case study

Scenario	% diesel HCV	Corridor width (m) for a PM_{10} threshold of		
		2 µg m^{-3}	1 µg m^{-3}	0.5 µg m^{-3}
Realistic	12%	87	270	838
High diesel HCV	48%	247	768	2380

6 General applications and case studies

6.1 Assessing the general width of motorway corridors

6.1.1 The question

The RCM is designed to predict the width of a corridor of degraded air around specific road links. However, there are numerous situations where a general corridor width (a single typical value, or simple formula) that is applicable across a whole city, or the whole country, would be useful. Examples include land-use guidelines for the management of reverse sensitivity (such as exist in California and British Columbia), and a screening approach for the assessment and identification of the location and scope of mitigation options.

6.1.2 Thresholds

The case studies in chapter 5 have indicated the definition of a corridor width requires the choice of an air quality threshold. The thresholds illustrated in chapter 5 were road-related elevations in annual PM_{10} of 0.5, 1 and 2μg m^{-3}. Choosing between these three options (particularly the first two) leads to substantial differences in the resulting corridor width.

There are several issues surrounding the choice of thresholds, particularly consistency and the restrictiveness (conservatism or precautionary nature) or leniency of the threshold, and the scientific basis (or otherwise) of the threshold.

Roadside corridor thresholds are in some ways analogous to the significance criteria specified in the *Good practice guide for assessing discharges to air from land transport* (MfE 2008). These criteria are used to indicate whether a project's impact on air quality is *likely* to be sufficiently significant to justify a detailed (Tier 3) assessment. The criteria are suggested as a guide only. They are based on 10% of the National Environmental Standards for Air Quality (NES) in the case of CO and NO_2. However, criteria are set at 5% of the NES for PM_{10}, and 5% of the *Ambient air quality guidelines* (MfE 2002) for $PM_{2.5}$ and NO_2 (24-hour average), with the guide stating: 'Longer-term exposure to these pollutants has a more serious health effect, and is consequently set at a relatively low percentage'.

In general, limited justification is provided for these percentages. None of the criteria specifically relate to annual means and so are not directly comparable to the output of the RCM. However, if the same percentage criteria are applied, then 5% of the ambient air quality guideline for annual PM_{10} (20μg m^{-3}) is 1μg m^{-3}.

The very same criteria are adopted in the NZTA (2010) draft Standard for Producing Air Quality Assessments for State Highway Projects with the same limited justification given.

The main differences between a 'restrictive' and a 'lenient' threshold are summarised in table 6.1.

Table 6.1 Summary comparison between lenient and restrictive thresholds for defining roadside corridor widths

Lenient	Restrictive
eg 10% of the NES/AAQG	eg 5% of the NES/AAQG
'robust'	'precautionary'
Leads to a relatively robust corridor definition	Leads to a relatively uncertain corridor definition
More likely to be detectable by monitoring	Possibly too subtle to be detected by monitoring
	Consistent with California and British Columbia guidelines

An alternative approach to thresholds is to base these on pollutant-dependent scientific criteria. A practical option is to define criteria based on common measurement technologies.

For instance, a threshold could be based on the increment which a commonly available instrument can confidently distinguish. For example, in the case of PM_{10}, NES-compliant instrumentation cannot confidently distinguish differences smaller than 2μg m^{-3}, so this could be a credible threshold. A related approach is to determine a threshold which could be observationally attributed to a specific road, perhaps through a combination of observational analysis and expert elicitation. The disadvantages of both these instrumental/observation-based approaches is that the threshold (and hence corridor width) could change (perhaps dramatically) if new measurement technology becomes available. A further variant on an observational-based approach is to define a threshold based on its detectability (verification) using low-cost passive monitoring.

An alternative and perhaps more ideal approach is to base thresholds on epidemiological/toxicological evidence. Such evidence is not available in New Zealand at present, and data that exists abroad is also highly uncertain (which is part of the motivation of this research project – see chapter 1) due to the many confounding factors involved in toxic response and illness and the practical difficulties in gathering clear evidence of this nature.

In summary, while moderately consistent corridor widths may be defined, they are based on relatively arbitrary thresholds. We recommend further research is conducted to consider the options for improving consistency between approaches for defining thresholds.

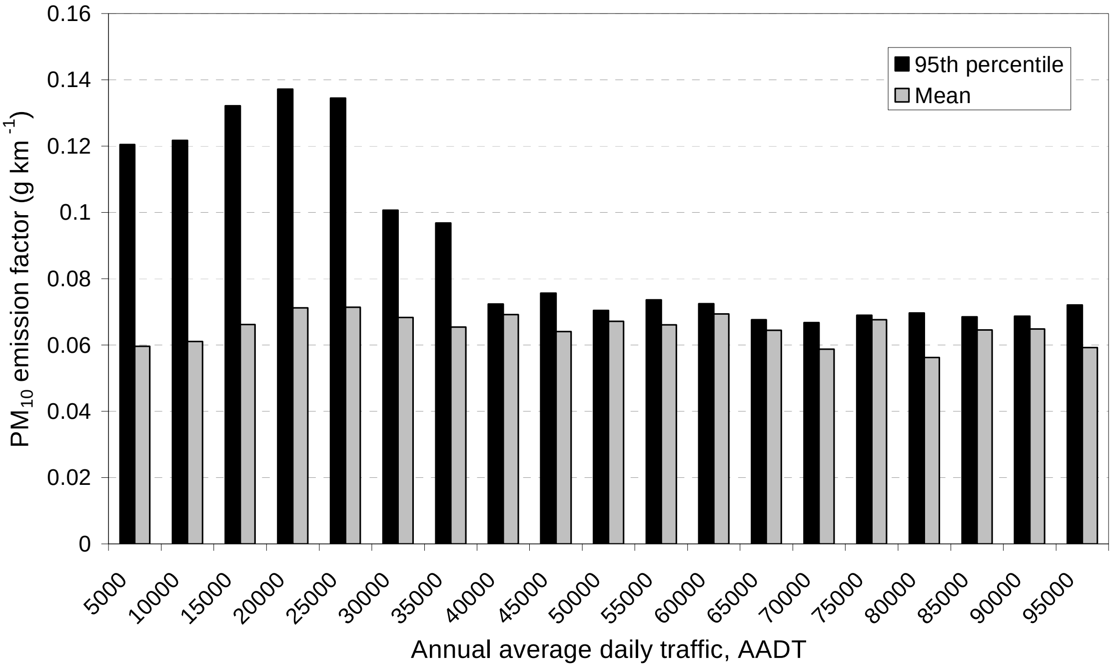
6.1.3 An example analysis of general corridor width for Auckland

The following illustrative analysis aims to estimate generic roadside corridor widths for Auckland and indicate the sensitivity of the results to the choice of threshold, and the difference between conservative and average treatment of emissions.

For this analysis we have initially assumed that a threshold of 5% of the annual PM_{10} guideline (ie 1μg m^{-3}) is appropriate. We have also used for this analysis the traffic emissions predicted for the purposes of the Waterview Connection technical assessment (by Beca Infrastructure). The traffic emissions modelling for this assessment covered approximately 18,000 road links in the Auckland region, predicting road-specific fleet-average emission factors based on modelled traffic volumes, speeds and fleet composition. For the purposes of this illustrative exercise we focus on the emission factors predicted for traffic characteristics in the inter-peak period, based on the 2006 scenario.

Figure 6.1 shows the median and 95th percentile fleet-weighted PM_{10} emission factors as a function of AADT over the approximately 18,000 links. It shows that, for AADT > 40,000 (which corresponds to motorways), the PM_{10} emission factor is effectively independent of AADT with 90% of the values falling between 0.052 and 0.076 g km^{-1} (0.064 g km^{-1} on average). Though not shown here, our analysis also revealed that at AADT < 40,000 the PM_{10} emission factor lies below 0.08 for 80% of the links. For the remaining 20% of the links, the PM_{10} emission factor is elevated to approximately double. Inspection of the data suggests the elevated emission factors on these links are primarily explained by low traffic speeds (< 30km/h).

Figure 6.1 Two statistical properties of PM_{10} emission factors per road link as a function of AADT on that link based on traffic and emissions modelling of ~18,000 road links in the Auckland region

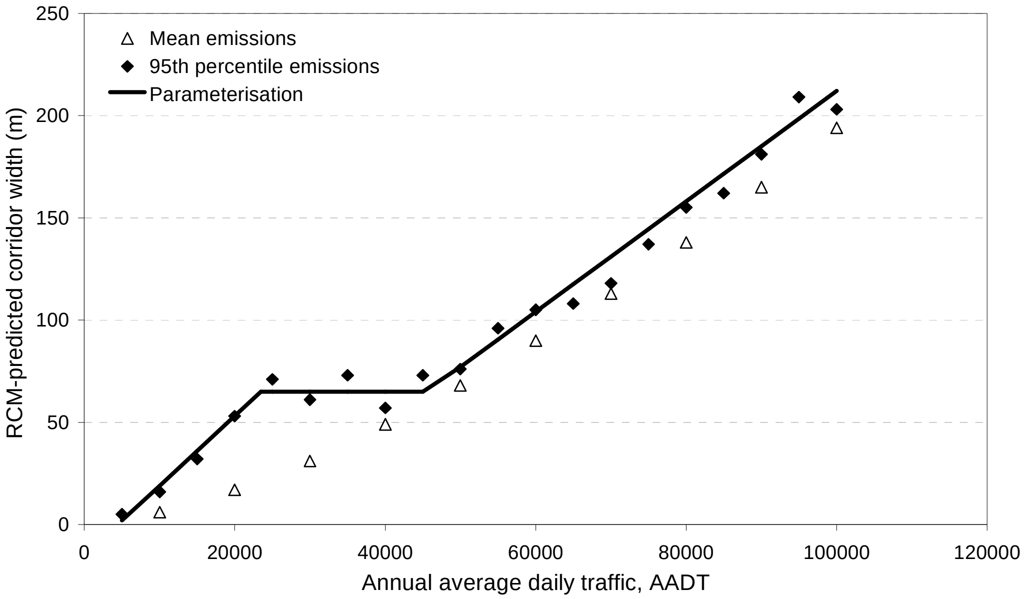


We used two sets of emission factors as inputs to the RCM: i) a constant value of 0.065g km^{-1} for all AADT, and ii) the 95th percentile for each 5000 band, as shown in figure 6.1. We set the road orientation to 75°, ie the orientation which, in Auckland, would give the maximum corridor width. These values were entered into the RCM, and the corridor width (average of both sides) was calculated for each AADT band. The results are illustrated in figure 6.2. The white triangles show the predicted corridor width based on the constant (typical) emission factor. The relationship for this choice of emission factor is well described by a power law of the form

$$\text{corridor width (m)} = \left(5 \times 10^{-6}\right) \times \text{AADT}^{1.5} \qquad \text{(Equation 6.1)}$$

The black diamonds show the predicted corridor width based on the 95th percentile emission factor for each band, and the black line is a parameterised fit to that data. Using the parameterised fit there is a plateau where the corridor width is a constant 65m for AADT values between 23,000 and 45,000. Some other predicted corridor widths are shown in table 6.2.

Figure 6.2 Road corridor width predicted by the RCM (based on threshold of 5% of the annual PM_{10} guideline) as a function of AADT in Auckland based on two approaches to characterising the distribution of emissions on the road network



The analysis above was based on a threshold of 5% of the annual PM_{10} guideline (ie 1μg m^{-3}). Figure 6.3 compares the median emission factor curve with a curve based on a 10% threshold, and some of the results are also included in table 6.2. This data shows just how sensitive the results are to the choice of threshold.

Figure 6.3 Comparison of road corridor widths predicted by the RCM as a function of AADT in Auckland (based on median emissions) calculated for thresholds of 5% and 10% of the annual PM_{10} guideline

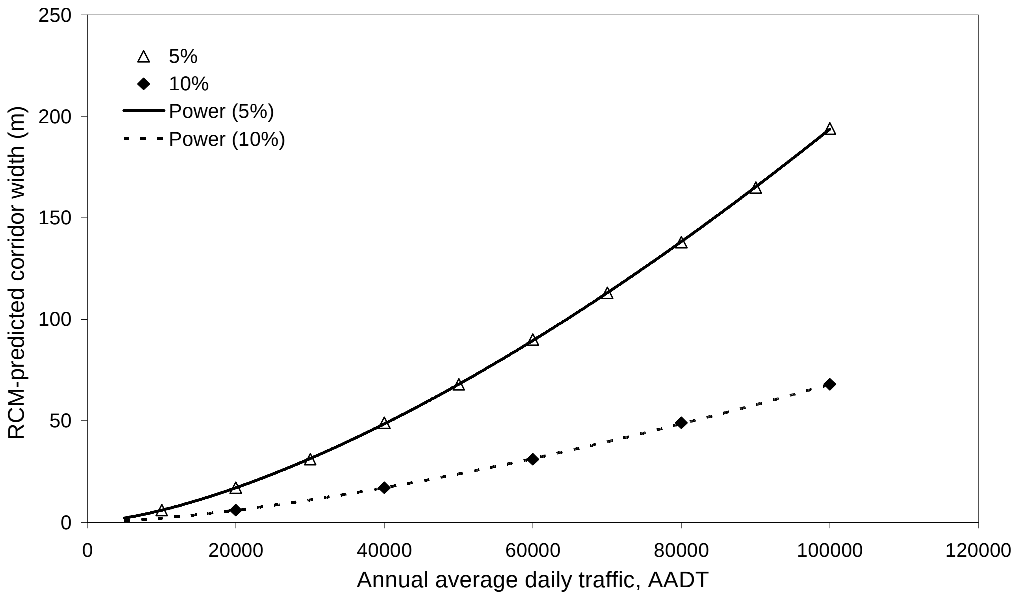


Table 6.2 Predicted generalised road corridor widths for Auckland based on median emissions for thresholds of 5% and 10%, and 95th percentile emissions for a threshold of 5%

AADT	Corridor width (m)		
	Median emissions		95th percentile emissions
	5% threshold:	10% threshold:	5% threshold
10,000	6	2	19
20,000	17	6	53
30,000	31	11	65
40,000	49	17	65
60,000	90	31	104
100,000	194	68	212

These values are provided as an illustrative example. They are based on both Auckland-specific traffic data and Auckland-specific dispersion (ie the Auckland meteorological dataset). It is likely the results would be slightly different for other cities. There is a small amount of inherent conservatism due to modelling the road orientation (for Auckland) which gives the largest values. The results are also based on 2006 traffic volumes, characteristics and emissions (as predicted by the VEPM). For future years corridor widths could decrease (due to reductions in emission factors) or increase (due to growth in traffic volumes or congestion). It should be noted the VEPM depends heavily on European emission factors, but the majority of our light vehicles were built to Japanese emission standards. The emissions of these Japanese vehicles are estimated in VEPM by drawing ‘equivalences’. This approach may introduce large uncertainties that have yet to be quantified. Moreover, the emission factors in the VEPM require updating.

6.2 Source apportionment of long-term roadside monitoring data

6.2.1 Urban background assessment and its applications

When combined with roadside monitoring, the RCM enables decomposition of source contributions because the difference between observed (all source) concentrations and modelled (road only) concentrations is equal to the contribution of all other sources. This latter contribution may be termed the ‘urban background’. When used with the RCM, the urban background means the concentration related to all emission sources other than the road (or roads) the RCM is modelling.

The output from the RCM alone cannot be used for comparison with air quality standards and guidelines because it does not include the urban background contribution. The RCM alone provides only relative assessments of air quality between different locations with different degrees of roadside influence. If the urban background can be quantified, however, then a measure of absolute air quality can be gained for which comparison with standards and guidelines is valid.

Other situations where using the RCM to assess the urban background might be of value are in assessments of changes to traffic emissions on a road, or network of roads, that will not affect the urban background, eg traffic management schemes or road upgrades. Furthermore, it may be assumed as a first approximation, that the urban background concentration represents concentrations outside the roadside corridor predicted by the RCM. It therefore represents concentrations at the majority of urban residences. This application is of particular value to studies of population exposure and public health impacts.

6.2.2 Application to Auckland monitoring sites

As an illustrative example we have applied the RCM to model annual mean PM_{10} contributions from adjacent roads at five roadside Auckland Regional Council (ARC) monitoring stations (based on 2006 traffic data). The results are shown in figure 6.4. The predicted values have been compared to the measured annual mean PM_{10} concentration (for 2006) at each site to estimate the percentage contribution the adjacent roads make to that annual mean as shown in figure 6.5.

Figure 6.4 Contribution of emissions from adjacent road predicted by RCM (black), and estimated contribution of all motor vehicle sources predicted by receptor modelling (white, Davy et al 2009), to annual mean PM_{10} at several ARC roadside monitoring sites

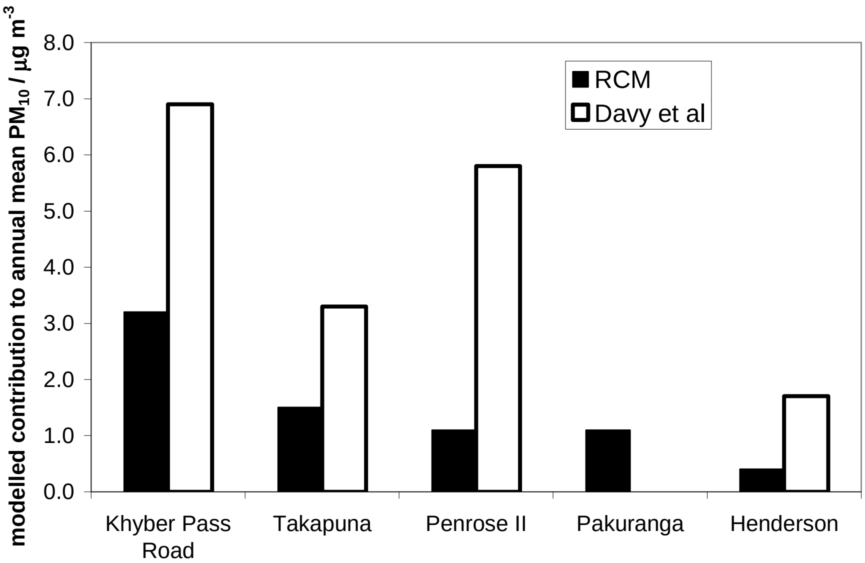
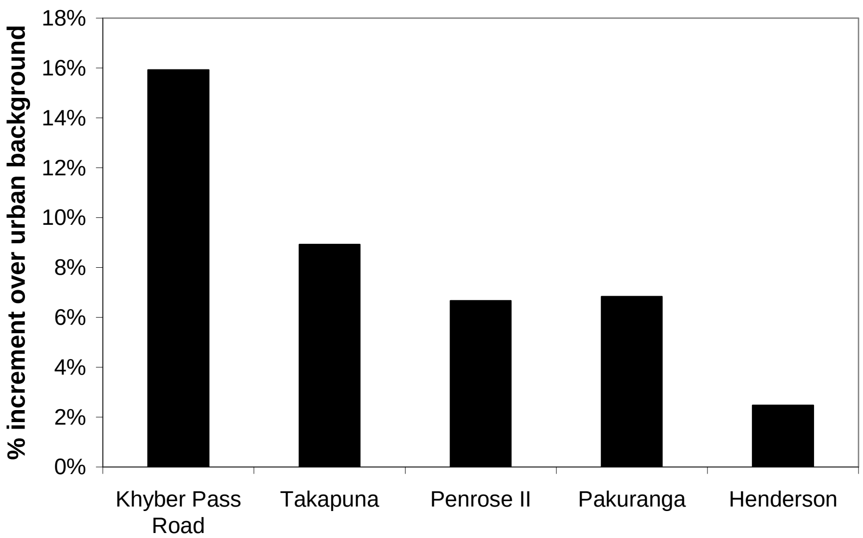


Figure 6.5 RCM predictions of the elevation in long-term PM_{10} above the local urban background attributable to adjacent roads at several ARC monitoring sites



6.2.3 Method limitations and comparison with alternative methods

The analysis presented above is a relatively crude form of source apportionment. There are several other methods of source apportionment, but any comparisons should bear in mind the ways in which this method differs. Rather than segregating traffic from non-traffic sources, it segregates the near-field dominant traffic source (adjacent roads). More distant (ie un-modelled) roads and traffic emissions are classed as part of the urban background. It is primarily for this reason that this method predicts lower percentages of 'road' contributions than methods such as receptor modelling based on chemical analysis of PM filters, or other chemically based approaches. This is illustrated in figure 6.4 where RCM predictions have been displayed alongside results of source apportionment analyses derived from chemical analyses of filter samples collected in 2006 and 2007 (Davy et al 2009). Taking the example of Khyber Pass Road, the RCM predicts the contribution to PM_{10} from the adjacent roads modelled (Khyber Pass Road, Mountain Road and the Southern Motorway) is 3μg m^{-3}, whereas the Davy et al study predicts a total motor vehicle contribution of ~7μg m^{-3}. If we assume both methods are accurate, then we conclude the difference (4μg m^{-3}) is derived from other more distant motor vehicle sources in Auckland (ie those not included in the RCM modelling).

The analysis above has applied the RCM to specific sites, despite the caveats expressed in chapter 2 about the RCM providing general results for typical locations but being less suited to providing site-specific results. This is particularly the case for the Khyber Pass Road site, which suffers from a degree of flow sheltering and recirculation, and is at a complex and busy intersection with complex gradients. The degree of error involved in applying the RCM to this site is currently unknown. Furthermore, the Pakuranga monitoring site is in a dip in which we hypothesise that pollutants (particularly woodsmoke) accumulate in certain meteorological conditions. The degree to which this also introduces an error has not, at present, been determined.

Despite these caveats, the method has the potential to be of great value. We recommend the potential of combining the RCM approach with other analytical techniques is explored in future.

7 Future modifications and enhancements

7.1 Overview

There are three directions in which further research could progress in order to ensure the value and applicability of the RCM:

1 Observational validation

2 Extension of its capabilities and applicability

3 Software development.

7.2 Observational validation

In its current state the RCM is unvalidated, though, in our opinion, the concentrations and corridor widths predicted by the model are credible and consistent with expectation.

The credibility of the model would, however, be substantially enhanced if it was observationally validated. Model limitations likely to lead to uncertainty or error in its predictions include those:

- inherent in the choice of the VEPM to describe emissions (including inherent uncertainties and potential errors inherent in applying this 'fleet average' model to individual road links)
- inherent in the choice of CALINE to describe dispersion (including assuming a minimum wind speed of 0.5m/s)
- related to simplified assumptions detailed in this report (such as the assumption of flat terrain and applicability of the meteorological datasets).

The VEPM depends heavily on European emission factors, but the majority of our light vehicles were built to Japanese emission standards. The emissions of these Japanese vehicles are estimated in VEPM by drawing 'equivalences'. This approach may introduce large uncertainties that have yet to be quantified. Moreover, the emission factors in the VEPM require updating.

It is not possible to validate the RCM directly against observational data from a single roadside monitoring site, due to the fact that a monitor observes concentrations arising from all emission sources, not just the adjacent road. Although methods exist to independently apportion roadside monitoring data into near-field (adjacent road) sources and other sources (other than using the RCM itself, as demonstrated in chapter 5), such as filtering and detailed dispersion modelling, these methods are themselves subject to additional uncertainties and require their own validation.

A more satisfactory approach can be applied if monitoring data either side of the road is available. This permits the RCM prediction (contribution from the road only) to be directly compared with the difference in concentrations recorded by the two (upwind and downwind) monitoring sites. However, even this method suffers from the weakness that the observations are being compared with the output not just of the RCM, but the combination of the RCM and the emissions model (and the traffic model if the predictions are based on modelled rather than observed traffic). The ideal design of an observational dataset would be to have a minimum of three observational sites – one upwind and two downwind – one at kerbside and one more set back. In this way the upwind site permits subtraction of other (background) source contributions, the kerbside site permits validation of the traffic and emissions models, and the observed dilution between the kerbside and setback sites permits validation of the dispersion component (ie the RCM) independently of the traffic and emission models.

An alternative (or additional) approach to validation is to reformulate the models into a 'heuristic' or 'learning' mode. This means monitoring data is continually (or regularly) supplied to the model and used to adjust or tune its parameters based on these observations. This approach can be applied independently to either the dispersion component or the emissions model. It is likely to be more appropriate for the emissions model as emissions (unlike dispersion) are likely to change over time.

7.3 Extending the model's capabilities and applicability

7.3.1 Selection of meteorological data and artificial meteorological datasets

Appendix B documents how the RCM has been developed, to date, on the basis of two meteorological datasets – observations from Auckland Airport for the period 2004–2005, and observations from Wellington Airport for the whole of 2006. Though not shown, these datasets led to different parameters for the dispersion curves, representing the different wind climates in the two cities. This suggests, if the RCM is to be applied to any location with a significantly different wind climate, a locally-representative meteorological dataset should be used. Significantly different wind climates could occur in other cities, but also in locations within Auckland or Wellington which are subject to locally variant climates, especially micro-climates such as coastal land-sea breeze systems (which also affect harbours) and valley/ridge flows.

Selection of an appropriate meteorological dataset for dispersion modelling is recognised as a challenging problem that has the potential for substantial inconsistency. Validating the applicability of any given dataset is also a substantial undertaking. By locking the RCM to particular datasets we would be implying a NIWA endorsement of the validity of that dataset. Providing such validation or endorsement is well beyond the scope of this project.

This issue is widely recognised in New Zealand. It is for similar reasons the ARC has recently commissioned the creation of an 'official' meteorological dataset for dispersion modelling in Auckland. NIWA is aware of projects and plans to create similar datasets for other airsheds.

In the future it may be appropriate for the RCM to source its input data from such artificial datasets. To enable this to happen, it would be appropriate to investigate the sensitivity of the RCM output to the way the meteorological datasets are used, especially if the datasets allow the user to select data for multiple locations within the given domain.

7.3.2 Variations in land use, topography and the built environment

The core models on which the RCM is based (principally AusRoads, but also the VEPM), and the parameterisation approach necessarily limit the RCM's applicability. It is unclear at present whether the AusRoads/VEPM combination is able to make valid predictions for locations with sloping or complex terrain, variations in land use and built topography (eg warehousing, parkland, parking lots) and especially locations subject to flow sheltering and recirculation (eg street canyons). This model combination is not well suited to intersections or locations dominated by intermittent or congested traffic.

In several of these cases an alternative core model or dispersion-emission model combination may provide better performance. For example, the OSPM is a widely used street canyon model which would be well suited to being embedded in the RCM as an option (or providing alternative parameterised dispersion curves for use in street canyons). Also, CAL3QHC is a version of CALINE (upon which the AusRoads model is based) designed for application to intersections – this could also potentially be adopted in the RCM. It is at intersections and in congested traffic in particular that the VEPM may be out-performed by an emissions model expressly designed for such locations.

7.3.3 Non-passive pollutants

The RCM currently predicts dispersion of 'passive' or inert pollutants, ie ones which do not undergo any chemical or physical transformation during dispersion. On the relevant timescale (dispersion over tens to hundreds of metres, ie minutes) carbon monoxide and key volatile organic compounds (VOCs), like benzene) are effectively inert so that the only process which the model needs to capture is dilution.

Total NO_x (oxides of nitrogen) is approximately inert, but the split between NO and NO_2 is highly dynamic and these individual components cannot be considered to be 'passive' in this context. Consequently, the current RCM predicts NO_x, but not NO_2. Although there are many complex interacting factors influencing the split between NO and NO_2 there is some predictability. Understanding the relevant processes and capturing them in an operational model is the subject of active research both abroad and at NIWA. We recommend the RCM be extended to incorporate predictions of NO_2 as soon as the New Zealand-based research data is available to support it.

To a first approximation, tailpipe PM_{10} can be considered passive on the relevant timescales. However, coarse particles are known to have high deposition rates relative to finer particles, although there is very limited observational data, especially in New Zealand, to quantify the effect of this on roadside PM_{10} levels. Ultrafine particles (UFP) are not passive as they are subject to extra physical and chemical processes (such as nucleation, coagulation, adsorption, evaporation and condensation), especially in the first few seconds/metres after emission. The role of these processes at the roadside is also a very active area of international research. The limited understanding to date indicates climate plays a substantial role in the processes affecting UFP, to the extent data from other climates should not be assumed to apply in New Zealand without validation.

The implementation of predictions for NO_2, coarse and UFP is further limited by the lack of vehicle emission factors for these species. Neither primary NO_2 nor UFP are included in the VEPM and coarse particles are represented in the VEPM by brake and tyre wear particles. Other international data sources are available for NO_2 and UFP, but their applicability to the New Zealand vehicle fleet has not been assessed.

7.4 Software development

7.4.1 Selection and ingestion of input data

Traffic data may be available as traffic model output (sometimes expressed as GIS shape files), as observed traffic counts, or in the form of user-defined scenarios. The validity of RCM predictions are limited by the validity of the traffic input data. Previous experience has shown us modelled traffic volumes can differ substantially from observed traffic counts. The provision of verified traffic data by an appropriate agency would assist in deriving consistent results from the RCM.

The usability of the RCM will be enhanced if a software application is developed to handle a range of traffic input data, especially if linked directly to the data's source (eg traffic model output). Furthermore, at present, the RCM depends on emission rates estimated by the VEPM on a case-by-case basis (ie 'manually' computed for every case of traffic data). It would be highly desirable to automate the process of deriving emission rates from traffic data.

7.4.2 GIS interface and meshblock output

GIS-based applications of the RCM have been demonstrated in chapter 4, including merging the output with meshblock-based data (such as population). This is probably the key to the RCM's use in exposure assessments, and for public health assessment and research in particular. There are a number of technical software challenges to be overcome in order to ensure RCM data used for application at meshblock level is valid and easy to generate. The demonstrations provided in chapter 4 were generated manually. The process could be automated with further investment in software development.

While a web-based demonstration interface has been developed, at present it suffers from a number of technical difficulties which result in visibly apparent errors. However, NIWA's GIS specialists are confident that the underlying technical problems can be resolved if modest resources are allocated to developing this tool.

PART 2: IN-VEHICLE EXPOSURE

8 Introduction to in-vehicle exposure

8.1 Aim and content of Part 2

Part 2 summarises the results of our research on in-vehicle exposure. The main body of this part provides an overview of several observational studies of in-vehicle exposure and highlights key findings. It also describes the initial development of an air and particles exchange model (APEX), as well as the need for and intended use of such a model.

Appendix C provides a literature review for Part 2 and appendices D to G contain further details describing the observational studies and the APEX model.

8.2 Project activities and future directions

Our research on in-vehicle exposure consisted of three principle activities:

1 Collection of observational data

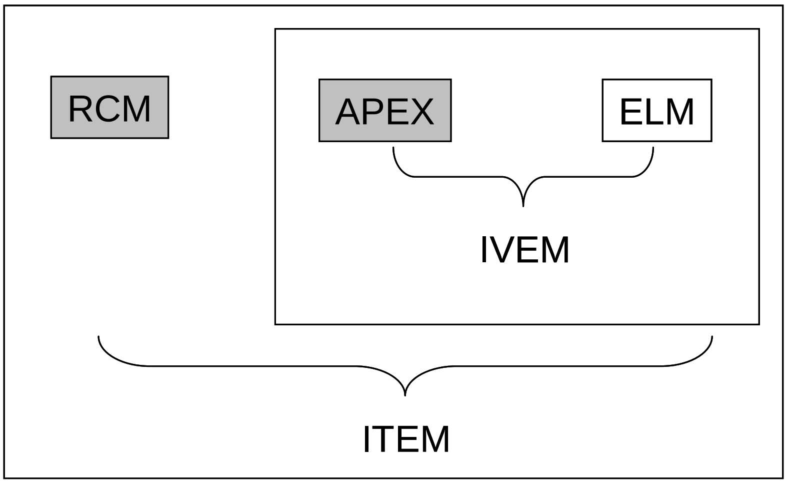
2 Analysis of this data

3 Initial development of the APEX model.

APEX predicts concentrations of air pollutants inside a vehicle moving in traffic if the external concentrations are known. The model is currently composed as an equation which can be implemented as a spreadsheet.

Our long-term vision for the models developed in this project was to combine them with a third model (which we termed an emission-link model or ELM). The APEX model and ELM should be viewed as sub-models describing two different factors which contribute to and control in-vehicle exposure: the role of the vehicle and its ventilation (APEX) and the role of the route chosen and the traffic upon it (ELM). This split is maintained in this report in which these two influences are described separately. The combination of these two factors, represented by the APEX and ELM models, would enable a generalised operational in-vehicle exposure model (IVEM). The IVEM would allow the investigation and prediction of in-vehicle exposure for any given route, trip or traffic conditions (within the limitations of the ELM formulation). Finally, our vision was to combine the IVEM with the roadside corridor model (RCM) developed within this project (described in Part 1) into an integrated exposure assessment package (as shown in figure 8.1). Unfortunately, limited resources meant it was not possible to progress development of the ELM within the term of this project. However, the observational data and analysis presented in this report provide a sound foundation upon which development of an ELM (and hence IVEM and ITEM) can be based.

Figure 8.1 Intended eventual model structure. The grey boxes show components developed within this project. The air and particles exchange model (APEX) is intended to combine with the emission link model (ELM) to form the in-vehicle exposure model (IVEM). IVEM combines with the roadside corridor model (RCM) to form the integrated transport exposure model (ITEM)



9 Exposure to traffic emissions while travelling in cars

9.1 Significance of in-vehicle exposure

Most of the studies of health effects of air pollution have been based on the assumption that the level of air pollution at a person's home represents their personal exposure (Fisher et al 2002; 2007). However, a small but growing number of studies show our exposure to traffic pollution is strongly influenced by the length of time we spend in close proximity (a few metres) to busy traffic. For most people this means the time we spend in cars, especially for those who do not live or spend long periods in buildings by major roads, and for younger children who walk and cycle less (MoT 2008). Several studies have shown traffic pollutant levels are many times higher in cars than they are in the home (Di Marco et al 2005). Together, this evidence indicates that, in previously observed associations with adverse health effects, place of residence may be acting as a simple proxy for the type, number and duration of journeys made from that location as well as the level of traffic air pollution at that location. Neglecting the short-term high exposure of a mobile population is one of the greatest limitations in further determining the role of traffic pollution in the development, and exacerbation, of respiratory and other adverse health outcomes.

The *International study of asthma and allergies in childhood* (ISAAC 1998) found New Zealand was among those countries with the highest childhood asthma prevalence in the world. Asthma is also the most common cause of hospital admission for children in New Zealand (Holt and Beasley 2001). Furthermore, New Zealand has one of the highest car ownership rates in the world, and a relatively undeveloped urban public transport infrastructure. New Zealand's vehicle fleet contains a high proportion of large cars with engines above three litres (13%, compared with 4% in the UK, for example), and the fleet is also relatively old (average age 12 years), with a large number of vehicles built to vintage or no emission standards. Consequently, the New Zealand fleet is relatively high polluting compared with those of Europe or North America, where most of the studies cited in this review were conducted.

The negative impacts of transport emissions on health (and the associated economic costs) and on the wider environment are incorporated into policy and strategic analysis and project assessment in either a highly simplified manner, or not at all. When assessment is made it is usually based on the effect of the net emissions across a whole airshed (typically a city) on typical ambient concentrations across that airshed. What is not currently assessed is the effect of elevated exposures in vehicles on individual exposures which, if aggregated across a whole city, could be equally or more substantial than the effect on 'urban background' air quality. Associated, though currently unquantified, 'costs' may arise from the negative personal experience (whether conscious or not) of being exposed to traffic emissions while in vehicles.

It is plausible in-vehicle exposure and the negative impacts of road traffic emissions in general can be mitigated through policies other than vehicle emission technologies. This may include traffic management, transport planning and road design, and urban design and planning, for it is these functions that determine where, when and how one travels. Enabling such analysis requires a micro-scale understanding of, first, how in-vehicle exposure occurs and how significant it is, and, second, what are the geographical and transport infrastructural factors that influence this exposure.

9.2 Why can't we use the data and models we already have?

There are currently no models or tools which explicitly predict in-vehicle exposure, either for individual journeys, or for whole populations.

As a first approximation, one might assume in-vehicle exposure is related to the duration of time spent in a vehicle, and indicative data can be extracted from travel surveys. However, data from this project (and others) clearly shows it is the exhaust emissions of other vehicles, especially those in front, to which we are principally exposed in a vehicle. This indicates travel times will need to be weighted in some way by some measure of traffic on the same route at the same time. The challenge before us is to determine how that weighting should be applied. The weighting could be related to the typical volume of traffic on the route of our journey – data which is generally readily available. However, choosing this type of weighting means exposure becomes quite sensitive to the route chosen, the time of day, day of the year, etc. Furthermore, research clearly indicates exposure due to infiltration of emissions into the vehicle is strongly influenced by the vehicle design, how it is ventilated, and the speed at which it is driven (see appendix C for further details). Choosing a weighting based on traffic volume may not necessarily be appropriate, as the vehicle immediately in front of the exposed vehicle is probably disproportionately responsible for that exposure. Routes where there is a high probability of following a bus or truck may lead to substantially higher exposures (or increased risk of higher exposures) than a route with similar volumes, but no buses or trucks.

Observational data is scarce, the number of potential explanatory variables is large, and the number of different permutations (different routes with different traffic volumes, speeds, fleet compositions, different times of day, different vehicles, different users with different driving styles and preferences) is so much greater still, that observational studies provide only a glimpse of the full range of exposures. There is also the possibility these studies may be biased. Observations alone cannot possibly provide the information we seek. We conclude, therefore, that simple 'guesses', such as exposure being predicted by journey time alone, are inadequate and to be avoided. Instead, we believe the correct way to proceed is through a multi-stage research programme in which observational data is gathered to develop a process-based model of the exchange of polluted air between the roadway atmosphere and the vehicle cabin interior. This can then be used to create a more generalised understanding of how travel is related to exposure.

10 In-vehicle exposure: observations in Auckland

10.1 The use, meaning and significance of particle number concentrations

The exposure data presented throughout this report is expressed in terms of particle number concentrations (PNCs). It must be noted this is not the same as, or in any way equivalent to, particle mass concentrations, such as PM_{10} or $PM_{2.5}$. It is well established that mass-based measures of particulate matter are biased towards the more 'massive' particles, such as aged accumulation mode particles (those observed days or weeks after their emission), and coarse particles (such as mineral dust and brake, tyre and road wear dusts), but under-represent ultrafine particles (such as fresh vehicle emissions). PNCs, however, count only the number of particles in a given volume of air, regardless of their mass. PNCs are representative of the more 'numerous' particles, which in urban atmospheres and transport microenvironments are always freshly emitted ultrafine particles[2]. Ultrafine particles are known to deposit much more deeply and efficiently in the lungs and are suspected to be more strongly toxicologically related to adverse health outcomes. Furthermore, most mass-based instruments do not respond to ultrafine particles. In summary, measurements of particle numbers are far more suitable for the purposes of investigating in-vehicle exposure to traffic emissions.

Until very recently, PNCs had not been measured in the ambient urban atmosphere in New Zealand. Regional councils do not monitor PNCs at their regulatory monitoring stations, which means comparing the exposures reported here to alternative exposures (eg in non-trafficked locations) is not a simple matter. NIWA has recently begun a cross-project approach to limited monitoring of PNCs in a wide range of locations and environments (of which this project forms part). Over the next few years the amount of PNC data available (initially in Auckland) will expand. In the meantime, indicative measures of 'urban background' PNCs have been extracted from our observational data and are reported below.

10.2 Overview of observational campaigns

The analysis in this report is based on two observational campaigns. The following discussion provides a summary. Further details of methods and data capture are set out in appendices D and E. The results discussed below are based on a pooling of data from these two campaigns.

10.2.1 NIWA's November 2007 pilot study

The first campaign was a pilot study funded by NIWA, carried out over three days in November 2007. A single vehicle was used throughout (a 2006 Honda Civic Hybrid, 1.3L CVT) in normal driving conditions on a range of roads in Auckland. Briefly, internal and external PNCs were simultaneously captured over 13 journeys during the weekday inter-peak period across the Auckland isthmus, with the journeys lasting an average of 17 minutes each. Figure 10.1 shows a map of all of the routes. The routes were chosen to encompass motorways (including the Central Motorway Junction), major roads (such as Great North Road), a suburban high street (Avondale), and quiet suburban roads. For the first six journeys the car's ventilation was set to a 'normal' setting (air conditioning on, vents open, fan on low, all windows closed). For the last seven journeys these ventilation settings were varied.

[2] The term 'ultrafine particles' is generally used to denote particles whose size (eg diameter) is smaller than 100nm (0.1μm).

The results of the pilot study raised questions which led to the genesis of the research project and the inclusion of the results in this report. The pilot study is described in more detail in appendix D.

10.2.2 January 2009 campaign

The second campaign was conducted within the scope of this NZTA project and was carried out during three days in January 2009. The objective of the second campaign was to gather additional observational data, including some on alternative routes to the pilot study, to begin to address some of the questions raised by the pilot study.

This study included two further observational mini-studies conducted in normal traffic. Two instrumental configurations were used in these tests:

1. 8 January 2009: Two instruments sampling internal air, but with the car's windows fully open so they were effectively sampling external air. This was strictly an untested assumption, but was entirely consistent with our measurements of air exchange rate (see appendix G). This configuration was adopted to perform instrument inter-comparisons. A Toyota Camry was used on this day.
2. 13 January 2009: The same configuration (and vehicle) as the pilot study (one instrument sampling internal air and an identical instrument simultaneously sampling external air through a sampling inlet, with all windows closed, air conditioning on, vents open and fan on low speed).

On 8 January, tests were conducted over four trips of ~20 minutes duration and one of 33 minutes duration in the Auckland isthmus. All trips were conducted in normal traffic during the weekday inter-peak period. The trips covered some of the same roads as the pilot study, as well as additional roads in the CBD and Eastern Bays (a map is provided in figure E.1 in appendix E).

On 13 January, three trips were conducted, two between Mairangi Bay (North Shore) and the Auckland CBD via East Coast Road, and one return trip via the Northern Motorway. These trips were also conducted in normal traffic during the weekday inter-peak period.

Appendix E has further details of how the tests were conducted.

10.2.3 General comment about observational data

Care should be taken when reviewing the observational data gathered. The condensation particle counters (CPCs) used in this project are not subject to any agreed quality assurance protocols or good practice guides for use, unlike regulatory particulate matter (PM_{10}) monitors, for instance. Concentrations reported by CPCs are known to be highly sensitive to the ability of the individual instrument to detect the smallest particles, especially those below 50nm in diameter. For this reason, the concentrations presented in this report should not be directly compared with those measured by other CPC models elsewhere. In future research we aim to better characterise the performance of the instruments used in this project and improve inter-comparability.

10.3 Journey mean exposures

Table 10.1 presents a summary of the average internal concentrations observed over the whole journey for each of the trips in the two observational campaigns.

Table 10.1 Summary of journey mean internal PNCs

Date	Trip #	Route	Duration /mins	Ventilation	Mean PNC/ cm^{-3}
15/11/07	1	Titirangi – NEWMARKET	18	normal	30,700
21/11/07	2	NEWMARKET – Avondale	14	normal	59,800
21/11/07	3	Avondale – NEWMARKET	30	normal	49,000
21/11/07	4	NEWMARKET – Meadowbank	17	normal	23,700
21/11/07	5	Meadowbank – NEWMARKET via CBD	39	normal	31,000
28/11/07	6	NEWMARKET – Avondale	13	normal	45,500
28/11/07	7	Avondale – NEWMARKET	17	high fan	86,400
28/11/07	8	NEWMARKET – Avondale	15	window open	51,100
28/11/07	9	Avondale – NEWMARKET	19	recirc, a/c on	17,900
28/11/07	10	NEWMARKET – Meadowbank	12	window open	64,700
28/11/07	11	Meadowbank – NEWMARKET	10	mid fan	42,100
28/11/07	12	NEWMARKET – Meadowbank	8	recirc, a/c on	14,300
28/11/07	13	Meadowbank – NEWMARKET	9	a/c & fan off	37,300
8/1/09	1	CBD – Meadowbank	20	window open	15,000
8/1/09	2	Meadowbank – St Heliers	18	window open	13,200
8/1/09	3	Okahu Bay – CBD	15	window open	35,000
8/1/09	4	CBD loop	20	window open	19,000
8/1/09	5	CBD loop – Greenlane – CBD	33	window open	15,800
8/1/09	6	CBD – Greenlane – CBD	17	window open	13,000
13/1/09	1	Mairangi Bay – CBD via East Coast Rd	24	normal	13,700
13/1/09	2	CBD – Mairangi Bay via SH1	17	normal	31,400
13/1/09	3	Mairangi Bay – CBD via East Coast Rd	15	normal	10,200

The main findings were:

- All of our journeys except one began or ended in the CBD or Newmarket. The average journey time was 17 minutes. Nineteen out of 22 journeys included at least some time on a motorway. Hence it seems reasonable our data encompassed a large number of car journeys which took place in Auckland.
- Despite this commonality in all trips, in-vehicle mean exposures were quite variable. The minimum was 10,200 cm^{-3} (Mairangi Bay – CBD, via East Coast Road), while the maximum was 59,800 cm^{-3} with normal ventilation or 86,400 cm^{-3} with high fan ventilation (Avondale – Newmarket, via SH16). The average was 33,000 cm^{-3}.

10.4 Comparison of mean in-vehicle exposure with ambient urban background concentrations

The in-vehicle concentrations reported above cannot be compared with ambient levels from monitoring stations. At the time of these campaigns there were no fixed stations monitoring PNCs anywhere in

New Zealand. However we have estimated urban background concentrations from our data by assuming there are periods in our external concentration data when there are no other vehicles in the vicinity (and there are periods in our data when we can confirm this to be true, including rest periods between trips). We assume that, at these times, we are measuring the urban background concentration.

The determinants of ambient PNCs and the variation in typical urban concentrations are complex, beyond the scope of this report, and currently unknown for New Zealand. However, it is plausible and consistent with findings from outside New Zealand (Morawska et al 2008), that PNCs will vary across Auckland, with potentially higher values in the CBD. We may also expect PNC to vary substantially from day to day (primarily in response to meteorological variation).

Table 10.2 summarises our estimates of urban background concentrations during our campaigns.

Table 10.2 Summary of estimated urban background PNCs during the observational campaigns and the range of the ratio of mean in-vehicle concentration to urban background

Date	Estimated urban background PNC/ cm^{-3}	Estimated where?	Ratio in-vehicle/ background
15 Nov 2009	1000	New Lynn	30
21 Nov 2007	4000	Meadowbank, Avondale	6–15
28 Nov 2007	7700	Avondale, Remuera	2–11
8 Jan 2009	4400	Eastern suburbs	3–8
13 Jan 2009	2800	Mairangi Bay	5–11

Consequently, our mean in-vehicle concentrations were between 2 and 30 times greater than the corresponding estimated urban background concentrations. Over the whole of our observational dataset the mean in-car concentration was typically greater than the urban background concentration.

10.5 Ventilation characteristics as a determinant of exposure

We reviewed the impact of the ventilation characteristics of the cars by pooling data with common ventilation settings and considering the mean I/O ratio (the term I/O is conventionally used to represent indoor/outdoor – here it is referring to internal/external concentrations):

- With normal ventilation settings the mean I/O ratio was 0.81, although this varied between 0.68 and 0.97 between trips.
- Altering the ventilation settings significantly altered the I/O ratio. With vents set to recirculate, the ratio was reduced to 0.30 or 0.24.
- I/O ratios greater than one were recorded on only one trip, which had high fan ventilation settings (Avondale – Newmarket, high fan ventilation, windows closed, air conditioning on). Whether this was a typical or atypical result is not clear without a detailed inspection of the dataset and repeat observations.
- I/O ratios of approximately zero were observed on two trips when the vehicle windows were open. Consequently, fully open windows appeared to indicate the occupant was fully exposed to external concentrations.

10.6 Traffic emissions as a determinant of exposure

We briefly review here the influence of traffic volumes as a determinant of in-vehicle exposure. In table 10.3 we have reported the mean in-vehicle PNCs on each day (28 Nov 2007 not included as ventilation settings were changed on a regular basis, preventing like-for-like inter-comparison). On the remaining four days exposures were higher during motorway driving.

Table 10.3 Mean in-vehicle PNCs for whole days of observations, segregated by motorway and non-motorway driving

Date	Mean in-vehicle PNC/cm^{-3}	
	Motorway	Other
15 Nov 2007	43,300	47,600
21 Nov 2007	57,100	38,400
8 Jan 2009	20,600	18,000
13 Jan 2009	31,800	12,900

10.7 Conclusions and questions arising

In-vehicle PNCs were 2 to 30 times higher than estimated urban background values. Exposures during motorway driving appeared to be higher, although not consistently so.

Typical I/O ratios of approximately 0.8 indicate the car's cabin and ventilation system offer some degree of protection against full exposure to traffic emissions, relative to the absence of a cabin. We may speculate this means the occupants of open-top cars, motorbikes and cars with open windows experience a relatively greater exposure. Whether this applies to cyclists cannot be answered with this dataset. However, we speculate cyclists' exposure might be quite different as they often occupy a different part of the road space and are less liable to follow specific vehicles.

The results from the observational campaigns raised a number of significant questions, namely:

- Do vehicle/traffic/route characteristics have any systematic influence on exposure?
- How representative is our data (of other vehicles, other routes)?
- Are there any other influences on in-vehicle exposure that need to be considered in a generalisable model?

These questions can be equally applied to three contexts:

1. The influence of vehicle characteristics on exposure (to be addressed by the APEX model)
2. The influence of traffic on exposure (to be addressed by a future emission-link model)
3. The interaction between these variables.

11 In-vehicle exposure: analysis and modelling

11.1 Analytical aims

The aim of this section is to explore the observational data in detail, in order to seek insight into the processes determining exposure and to inform the proposed in-vehicle exposure model. This is achieved by detailed data analysis, and through the development and exploitation of an air and particles exposure model (APEX).

11.2 Observational data – detailed analysis of individual journeys

Observational data in normal traffic was gathered at one-second resolution, with logging of position via GPS. In contrast to many previous studies (appendix C) this permits highly detailed analysis of the determinants of in-vehicle exposure, and hopefully permits us to develop an understanding of the relevant physical processes to inform a generalisable predictive model.

Figures 11.1 and 11.2 show an example of data from trip 1 on 13 January 2009 (Mairangi Bay – Auckland CBD via East Coast Road). These figures show many of the features typical in all of the trip datasets:

- External concentrations (on a one-second basis) vary significantly. In the example in figure 11.1 the range is from 2000 to 200,000 cm^{-3}, ie two orders of magnitude.
- The distribution of external concentrations is skewed, with lower concentrations being more common, interspersed with briefer elevations and occasional ‘spikes’.
- Internal concentrations vary over a smaller range (approximately over a single order of magnitude) and less rapidly than external concentrations.
- Internal concentrations broadly track trends in external concentrations, albeit with a damped and/or lagged response.
- The internal response to ‘spikes’ in external concentrations is suggestive of a relatively rapid (ie seconds) infiltration of particles from outside to inside the vehicle cabin.
- Immediately following such spikes internal concentrations are often higher than external concentrations. This suggests the injection of particles into the cabin volume where dilution is further inhibited, so particle-laden air is relatively ‘trapped’ in the cabin. Such periods of trapping tend to be up to a few minutes duration.

Figure 11.1 Interior and exterior PNCs observed during trip 1 on 13 January 2009 (Mairangi Bay to Auckland CBD). The large, brief elevation in external concentrations after three minutes represents the East Coast Road/Constellation Drive intersection

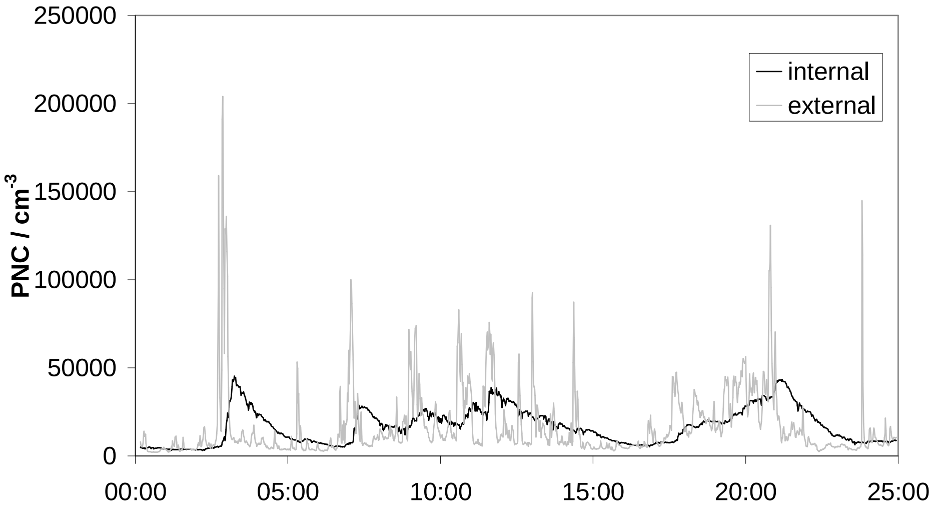
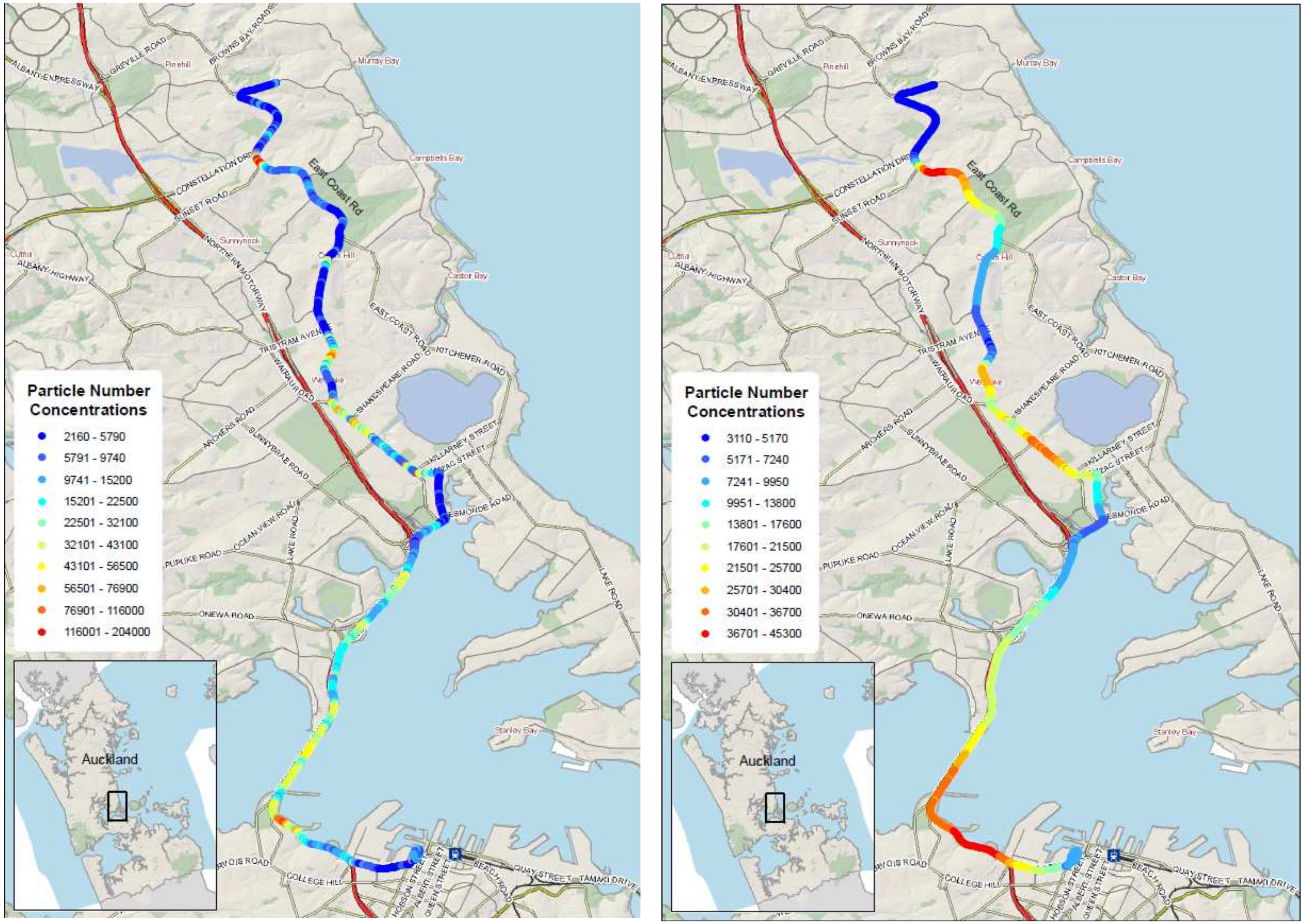


Figure 11.2 Exterior (left) and interior (right) PNCs (red = high, blue = low) observed during trip 1 on 13 January 2009 (Mairangi Bay to Auckland CBD). It can be seen that external concentrations peak sporadically and briefly, whereas internal concentrations are elevated following the external peaks



11.3 Influence of car ventilation on exposures and I/O ratios

11.3.1 Initial air and particles exchange (APEX) model

Data from the pilot study (28 Nov 2007) showed the ventilation settings of the test car had a significant impact on I/O ratios, and hence on in-vehicle exposure (see below). However, we need to be able to generalise and explain this variation so predictions for other routes in other vehicles can also be made. It is not practical to make multiple observations in a wide range of vehicles; hence a modelling approach is required.

The APEX model developed is described in detail in appendix F. In brief, it predicts in-cabin concentrations of an air pollutant on a second-by-second basis (or a few seconds), if the external concentrations are known at the same time scale. The model requires just a single parameter (the air exchange rate) which can be derived from multiple sources (see appendices C and G).

The model formulation is:

$$\frac{dN}{dt} = \lambda\left(Nout - Nin\right) \qquad \text{(Equation 11.1)}$$

where N_{in} and N_{out} are the internal and external particle number concentrations, respectively (cm^{-3}), and λ is the air exchange rate.

Note that this initial formulation of the model assumes the only process contributing to changes in the internal concentration is infiltration and exfiltration of particles with the air that enters and leaves the cabin. There are other factors such as filtration of particles by the air conditioning system, deposition of particles on internal surfaces and respiration into the lungs; however, the mechanisms behind these processes are more complicated and are not considered in the initial model at present.

11.3.2 Estimating air exchange rate from normal observational data

The air exchange rate (or AER) is a parameter which describes the rate at which internal and external air is exchanged. More specifically, it is the number of times the internal volume of air is changed per unit time. It is likely to be related to vehicle speed, ventilation settings and the inherent leakiness of the vehicle (and therefore a property of any individual vehicle and the way it is used – see appendix C for a review of previous studies).

The AER is conventionally assessed for a vehicle using a controlled method. However, our initial need was to characterise the net effect of all relevant processes (infiltration, exfiltration, deposition etc) as they occur in real driving conditions. For this reason, we initially attempted to extract an estimate of the AER from our observational data under normal conditions, assuming it is primarily a function of speed. The procedure involved extracting segments of data during which the car was travelling at a relatively constant speed for a minute or more. We then used the APEX model to iteratively adjust the AER values to obtain a best fit between modelled and observed internal concentrations during that period (see appendix G for further description).

This procedure was undertaken for trips from the pilot study on 15 and 28 November 2007, all describing data from the Honda Civic in normal driving. Normal ventilation settings were used on 15 November and four separate ventilation settings were used on 28 November:

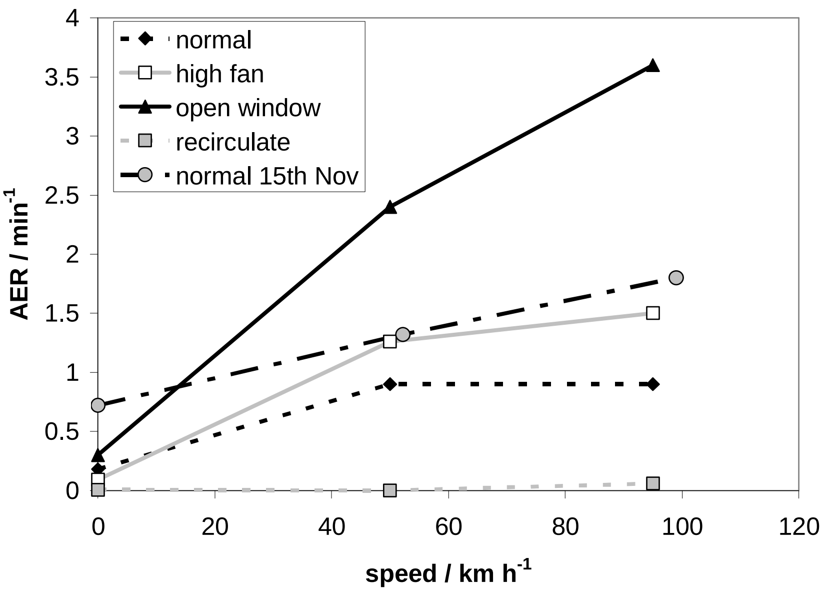
- a single fully open window

- high fan (windows closed)
- normal (windows closed, low fan setting)
- recirculation.

The procedure used to estimate the AER is difficult and prone to errors. These are explored further in appendix G, but, in brief, are primarily due to the complications arising from multiple uncontrolled influences in normal driving, particularly the rapid changes in the relevant processes (infiltration, deposition, changes in speed etc). Data from 21 Nov 2007 proved particularly complex and relatively resistant to the satisfactory estimation of the AER (which requires extended periods of near-constant speed).

Figure 11.3 shows the AER values derived for each ventilation setting at the three selected speeds.

Figure 11.3 Air exchange rates estimated for three vehicle speeds and four ventilation settings from pilot study data from 28 November 2007 (and one set from 15 November 2007). The data refers to a Honda Civic



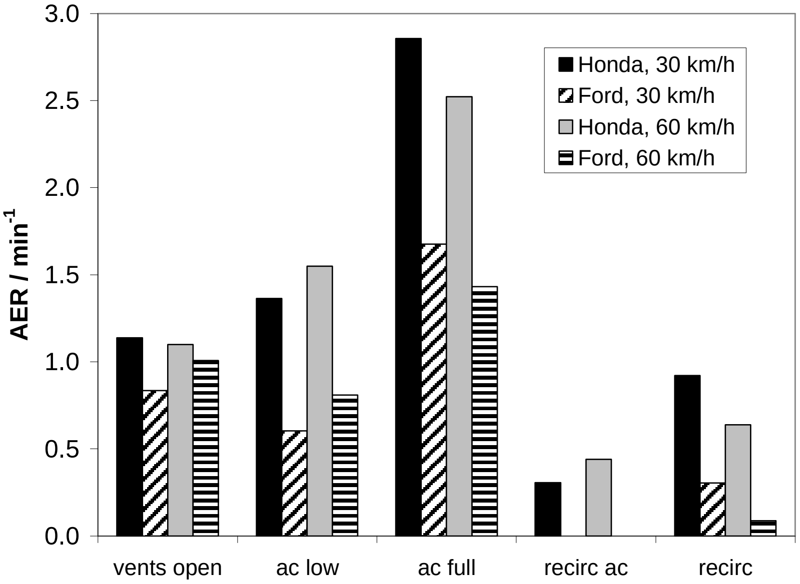
In general, normal driving conditions led to an estimated AER in the region of approximately 1min^{-1} (0.02s^{-1}). Air recirculation led to much lower AER values at all speeds, whereas opening the window increased AER values by a degree related to vehicle speed (higher speed leads to more turbulent air exchange through the open window).

11.3.3 Estimating air exchange rate from semi-controlled observational data

Because of the difficulties in extracting AER estimates from 'normal' driving data during the pilot study, we subsequently set up two semi-controlled ventilation tests with two vehicles: the same Honda Civic used in the pilot study and a Ford Falcon. The method and results are described in detail in appendix G. The main findings are reproduced here.

Figure 11.4 summarises the main relationships found in the data.

Figure 11.4 Dependence of estimated AER on ventilation setting (the Ford Falcon used did not allow air conditioning in recirculating mode)



We broadly found the following variables had an influence on the AER, roughly in order of descending significance:

- opening windows (approximately 10-fold increase)
- ventilation fan speed (1.5–3-fold increase)
- vehicle (the Honda Civic had roughly double the AER of the Ford Falcon – may be related to differences in cabin volume, and/or inherent degree of air tightness
- closing vents (ie recirculating air; led to 20%–90% decrease in the AER)
- vehicle speed (20%–45% increase when vents open, reduction in some cases with recirculating air)
- air conditioning (10%–40% increase, some decreases with recirculating air).

In the more common conditions (windows closed, vents open, fan speed low, vehicle speeds moderate) we found the AER varied between 0.6 and 1.6min^{-1} in our two vehicles.

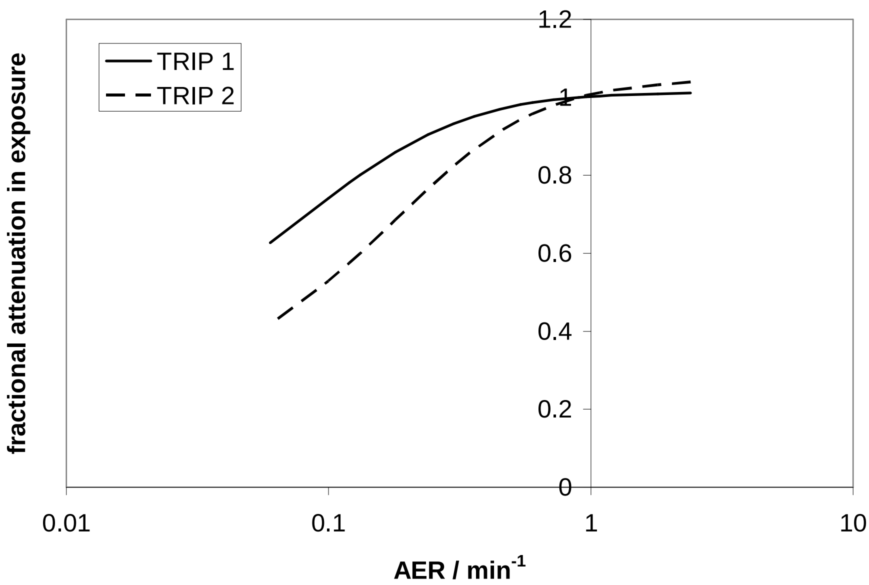
11.3.4 Using APEX to investigate the dependence of in-vehicle exposure on the AER

We used the APEX model for trips 1 and 2 on 13 January 2009 (Mairangi Bay to CBD) in order to investigate the effect of alternative quantifications of the AER. First, we assumed a fixed AER of 0.9 min^{-1} for each trip (independent of vehicle speed), determined by the best fit between modelled and observed internal concentrations. We then systematically varied the AER and used the APEX model to determine the journey mean I/O ratio.

Trips 1 and 2 differ in several ways. Trip 1 was a southbound trip mostly via East Coast Road, encountering low levels of traffic for most of the journey. Trip 2 was a northbound trip with 11 minutes (out of 17) on the Northern Motorway. The external concentrations were considerably more skewed in trip 1 than trip 2 with a stronger bias towards a small number of brief high concentrations (intersections, single vehicle encounters) compared with trip 2, which was more characterised by extended periods of elevated concentrations (relating to the multi-vehicle encounters on the motorway). In terms of air exchange, this meant that trip 1 was dominated by exfiltration (63% of records showed internal concentrations higher than external) whereas trip 2 was more evenly split between infiltration and exfiltration.

These differences between trips 1 and 2 correspond to a slightly different response to changes in the AER. The results for both trips are summarised in figure 11.5. This shows an increase in the AER (ie represented by opening of the windows) has almost no effect on mean exposure. A decrease in the AER (making the vehicle more sealed, eg by recirculating air, or having a larger cabin volume) to 0.1 min^{-1} decreased exposure by one quarter in the case of trip 1 and a half for trip 2. It must be noted, however, the model is likely to perform less well for low AERs, as the volumes of air exchanged are small, making the predictions more sensitive to errors in capturing the air exchange process (and errors arising from ignoring other processes such as deposition).

Figure 11.5 Fractional attenuation in journey-mean in-vehicle concentration as a function of the AER (relative to an AER of 0.09 min^{1}) based on data from the 13 January 2007, trip 2 only (note: logarithmic scale for the AER)



Although based on very limited data, we tentatively conclude, compared with a base case of normal ventilation, increased air exchange makes comparatively little difference to exposure. Decreased ventilation, however, decreases exposure especially on trips that are less skewed, when traffic is heavier, emissions more even and there is less opportunity for polluted air to remain trapped in the cabin following its injection.

The dependence of the relationship between the AER (ie vehicle characteristics) and in-vehicle exposure upon route/traffic characteristics indicates the need for further investigation of the role of route and traffic characteristics, and suggests the two influences cannot be addressed in isolation. This indicates to us the need for the development of the complementary ELM to complete a predictive IVEM.

11.4 Variation in exposure between routes

11.4.1 Introduction and the emission-link model

Observational data showed in-vehicle exposure varied significantly between routes, and between repeat trips over the same route. We seek to generalise and predict this variation for these and other routes, as it is not practical to make multiple observations on every typical journey, hence a modelling approach is required.

11.4.2 Motorways

What is immediately striking from the results summarised in table 11.1 is the significantly higher concentrations (more than double) observed on the Mairangi Bay to CBD trips utilising the full stretch of the SH1 Auckland Northern Motorway (13 Jan 2009, trip 2) compared with the alternative relatively low-traffic route along East Coast Road (trips 1 and 3 on the same day). These three trips were conducted between 12.30 and 1.30pm. This clearly suggests the time spent in busy traffic on the motorway is a significant determinant of exposure. Although this brief amount of data can only be indicative, it is notable the low-exposure route of East Coast Road was marginally quicker than the motorway route in trip 3, although not in trip 1, perhaps indicating greater travel time variability on the non-motorway route.

We pooled all the external concentration data captured during the pilot study and the 2009 observations on normal roads. The mean external concentrations measured on each day for motorway and non-motorway segments are shown in table 11.1. In four out of five cases higher concentrations were observed during motorway driving.

Table 11.1 Mean external PNCs observed per day, segregated by periods spent in motorway driving and other roads

Date	Mean external PNCs/cm^{-3}	
	Motorway	Other
15 Nov 2007	44,900	51,900
21 Nov 2007	91,900	45,000
28 Nov 2007	83,700	44,700
8 Jan 2009	20,600	18,000
13 Jan 2009	46,000	12,500

The challenge for future research is to quantify and explain this variation, and determine if motorway exposures are systematically higher.

'Motorway' is a rather crude categoric variable. It may be more informative to consider what features of motorways lead to higher concentrations. We could speculate vehicle speed and/or volumes might be the key factors. However, visual inspection of our observational data indicated some of the highest concentrations recorded during all of our campaigns were captured during congested conditions at and around the Central Motorway Junction, and on Great North Road in Avondale (which is effectively Avondale's 'high street'). It appears congested stop-go traffic is the common feature in these two locations and this observation suggests congestion may be a more important explanatory factor than traffic volume.

Our observational dataset was mostly limited to inter-peak periods. We do not yet have sufficient data to compare sections of motorway at different times of the day and different levels of volume and congestion.

We suggest further observational data spanning the peak and inter-peak periods is required to investigate the potentially independent effects of volume, speed and congestion.

11.4.3 Road gradient

Some of the features of the observational data are suggestive of a role of road gradient. For instance, table 11.2 compares the mean external concentrations observed on three roads in Auckland's CBD on 8 January 2009.

Table 11.2 Comparison of combined data from three major roads in Auckland's CBD with varying gradients

Road	Total duration of data/mins	Mean external PNC/cm^{-3}
Symonds Street (uphill)	9.5 (trips 4 and 5)	22,800
Hobson Street (uphill)	5.4 (trips 5 and 6)	14,600
Nelson Street (downhill)	9 (trips 3, 4 and 6)	9500

All three roads are major traffic routes carrying over 20,000 vehicles per day on average according to Auckland City Council. Hobson Street and Nelson Street carry one-way traffic, with Hobson Street (southbound) having an uphill gradient and Nelson Street (northbound) having a downhill gradient. Symonds Street has two-way traffic, and our vehicle travelled in the southbound direction, which has an uphill gradient. Symonds Street is also a major bus route. We therefore consider it plausible an uphill gradient on busy roads with several intersections (see below) leads to enhanced emissions from accelerating vehicles and engines under high load, and recommend this possibility is investigated further.

11.4.4 Intersections

All the datasets reveal very brief elevations which can often be associated with particular intersections. For instance this can be seen in figures 11.6 and 11.7 where a large but short-lived elevation in external concentrations occurred at the East Coast Road/Constellation Drive intersection. Externally the elevation in concentration lasts 18 seconds. However, the response inside the vehicle is to initially increase concentrations seven-fold. Internal concentrations then remain elevated over external concentrations for almost four minutes in which time the car has travelled a further 3.5km. Thus although the intersectional hot-spot is highly localised, its influence on exposure potentially extends over several kilometres.

Figure 11.6 Interior and exterior PNCs observed in the few minutes before and after crossing the East Coast Road/Constellation Drive intersection during trip 1 on 13 January 2009 (Mairangi Bay to Auckland CBD)

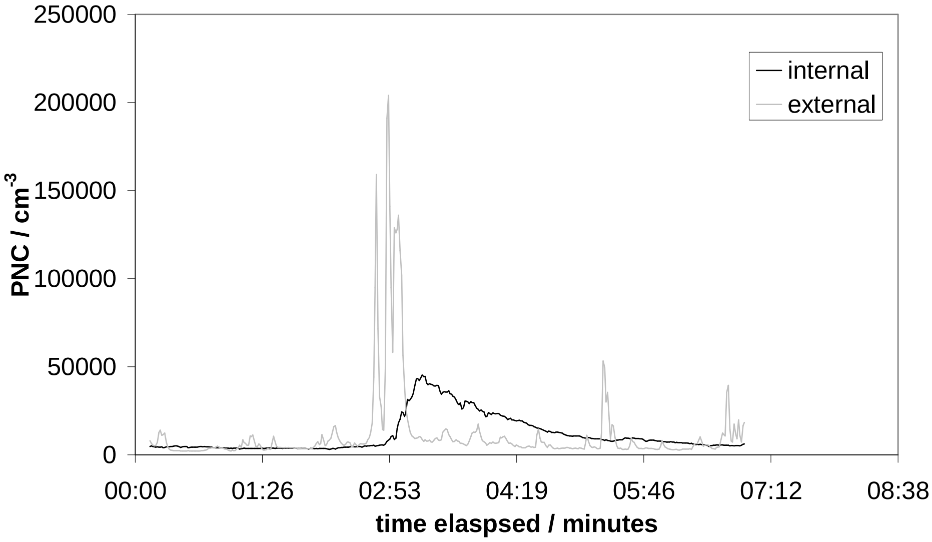
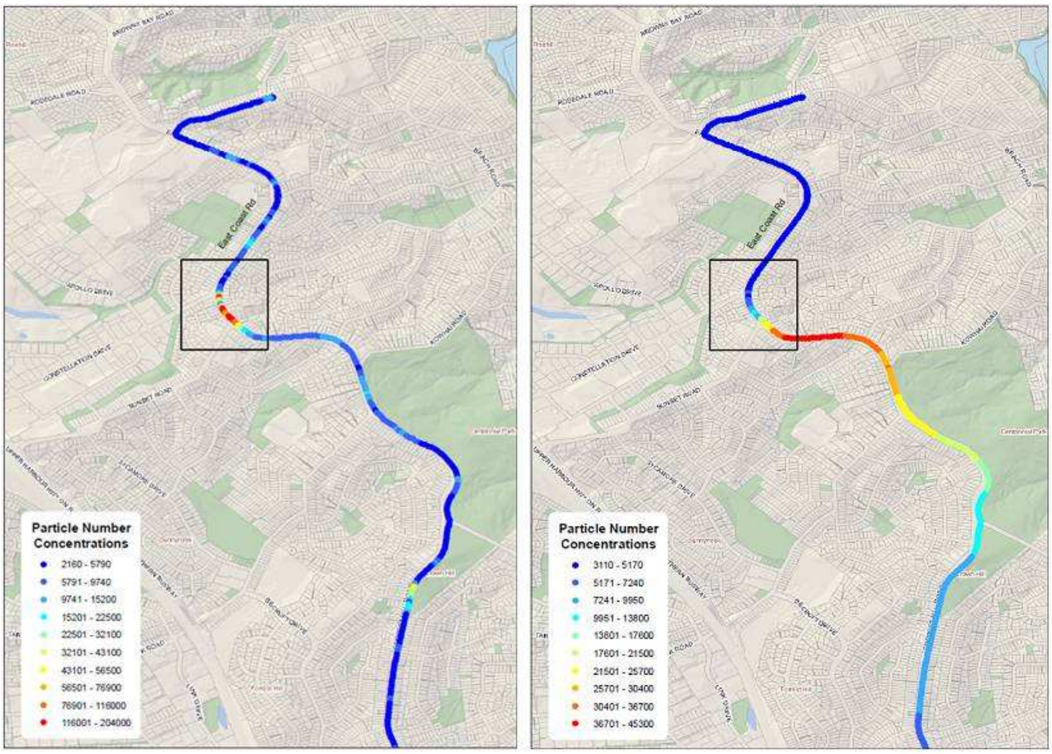


Figure 11.7 Detail of the impact of passing the Constellation Drive/East Coast Road intersection (intersection identified by box). External PNCs shown left, internal PNCs shown right. Colour indicates PNCs (red = high)

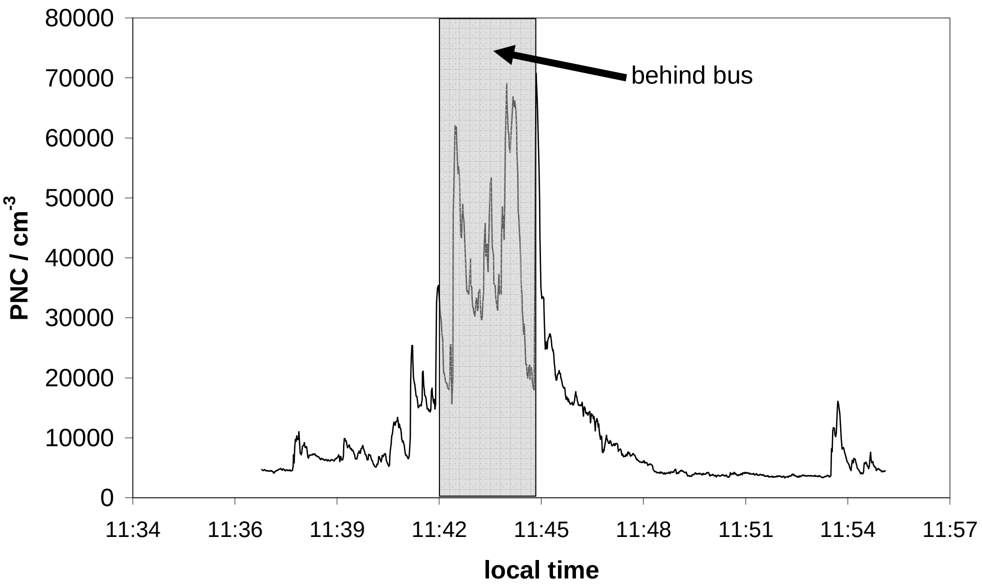


As passages through intersections happen briefly, at different speeds and over different durations, we may expect a significant degree of random variation in the observed response. The limited data we have captured suggests intersections may play a disproportionately large role (relative to their time spent in them) in exposure and therefore feel an extended effort needs to be made to incorporate and generalise their impact into an in-vehicle exposure model. We propose this will require more observational data including the systematically repeated travel through a limited range of intersections to characterise the nature and probability of particle injections as illustrated above.

11.4.5 Gross emitters

Visual inspection of the high-resolution data indicates other brief periods of elevated concentrations are also evident. Data shows relatively brief elevations in external concentrations can have a significant influence on internal concentrations (and hence exposure) due to the trapping of particles in the vehicle cabin. Another identified cause of elevated concentrations was the result of spending approximately three minutes behind a bus on St Johns Rd (8 January 2009, trip 2, figure 11.8). We estimate this led to the trip 2 average concentration being elevated to almost double what it would otherwise have been (~7000 cm^{-3}, compared with 13,200 cm^{-3}).

Figure 11.8 Detail of PNCs from 8 January 2009, trip 2, showing the three-minute period during which the test car was behind a bus



12 Conclusions

PNCs were measured inside three typical cars (a Honda Civic, a Ford Falcon and a Toyota Camry) over 21 journeys on a range of Auckland roads. The average journey duration was 17 minutes. All journeys except one began or ended in Auckland's CBD or Newmarket. Nineteen out of 22 journeys included some time on a motorway. Nine journeys consisted of a normal ventilation setting (windows closed, vents open, air conditioning on, low fan speed). Six journeys included windows fully open. The seven remaining journeys involved variations in ventilation.

Journey-mean in-vehicle concentrations were quite variable. The minimum was 10,200 cm^{-3} (Mairangi Bay – CBD via East Coast Road) while the maximum was 59,800 cm^{-3} for normal ventilation or 86,400 cm^{-3} for high fan ventilation. The mean for all journeys was ~33,000 cm^{-3}.

We estimated the concurrent urban background concentration from our data. This varied between 1000 and 4400 cm^{-3}. In-vehicle concentrations were estimated to typically be an order of magnitude higher than the urban background concentrations.

The mean I/O ratio (ie ratio of internal to external concentrations) was 0.81 for normal ventilation. For recirculating air the ratio reduced to 0.24–0.30. A mean I/O ratio of greater than one was observed only once.

A typical mean I/O ratio of ~0.8 indicates the car's cabin and ventilation system offers some degree of protection against full exposure to traffic emissions, relative to the absence of a cabin. We may speculate this means occupants of open-top cars, motorbikes and cars with open windows experience a relatively greater exposure.

The high resolution observational data was examined to investigate the role of the vehicle (including ventilation) and traffic characteristics on in-vehicle concentrations. The data was used to develop a simple empirical APEX model to investigate the role of the vehicle. Data analysis was also performed to enable the future development of a model to capture the role of the surrounding traffic emissions.

The APEX model is a simple model (based on empirically relating the AER to vehicle speed and ventilation) which successfully captured inter-journey variability in in-vehicle concentrations, although with a tendency towards over-estimation of absolute concentrations.

Data from normal driving and from a semi-controlled air exchange test indicated the AER for our 2006 Honda Civic was ~1 min^{-1} for common driving conditions (windows closed, vents open, intermediate vehicle speeds).

By conducting a semi-controlled experimental study on two vehicles (a Honda Civic and Ford Falcon) we experimentally found the following variables had an influence on the AER, roughly in order of descending significance:

- opening windows (approximately 10-fold increase)
- ventilation fan speed (1.5 to 3-fold increase)
- vehicle (the Honda Civic had roughly double the AER of the Ford Falcon – this may be related to differences in cabin volume, and/or inherent degree of air tightness)
- closing vents (ie recirculating air; led to 20%–90% decrease in the AER)
- vehicle speed (20%–45% increase when vents open, reduction in some cases with recirculating air)
- air conditioning (10%–40% increase, some decreases with recirculating air).

Exploratory analysis using the APEX model suggested an increase in the AER (ie represented by opening of the windows) had a minimal effect on increasing mean exposure. A decrease in the AER (making the vehicle more sealed, eg by recirculating air, or having a larger cabin volume) was found to decrease exposure.

Exploratory analysis of observational data suggested important determinants of exposure might be traffic volume (represented by time spent on busy motorways), passage through busy intersections, encounters with gross emitting vehicles and uphill gradients.

13 Recommendations for future developments

13.1 APEX model

In general we note the APEX model is deliberately a very simple model based on several assumptions. At present APEX has only a very limited basis in physical processes, and a more physically realistic model could be developed, possibly from our existing observational dataset. However, a key question, in our view, would be whether a more complex model such as this is necessary in terms of its required use.

In our conception, the principal requirement of the model is to predict journey-mean in-vehicle exposure. We have noted in this report the APEX model appears to successfully predict inter-journey variability in exposure, but with some over-estimation in absolute concentrations. This conclusion is based on a limited validation exercise. Further observational data will permit a more thorough validation. A key question for this validation would be to determine if the over-estimate is consistent, or whether there may be some systematic bias associated with one or more variables. If such a bias is found then the underlying assumptions of the APEX model should be re-evaluated with reference to the high resolution data captured in this project. It may be possible the APEX model can be further developed to explain and account for any bias.

13.2 Emission-link modelling

The research conducted in this project has been intended to inform the development of a general model to describe the influence of traffic and route characteristics on external concentrations and hence (via APEX) to in-vehicle exposure. The analysis in this report gives a strong indication the route may influence exposure through:

- traffic volume and/or speed (which may be acting as proxies for total emissions on the route)
- traffic density, ie the number of vehicles within a certain ‘zone of influence’ around the subject vehicle (the zone to be defined by further analysis)
- the probability of encountering other traffic at intersections, especially during acceleration
- the probability of encountering gross emitting vehicles and the duration of such encounters
- uphill gradients.

The data we have collected is probably insufficient for a systematic examination of these factors. An examination of the first factor requires data coverage of a wide range of traffic-emission characteristics, whereas the latter two factors are inherently probabilistic and hence require substantial datasets to cover the likely variability.

This branch of research is likely to be enhanced by further research on characterising the statistical distribution of emission rates from gross emitting vehicles, their use, geographical distribution and travel patterns.

13.3 Observational data

Our observational dataset is relatively limited in terms of routes covered, and the range of vehicles. It is also limited in the sense there are multiple variables influencing exposure, in particular, variables subject to considerable random variation (such as the probability of encountering gross emitting vehicles, and the duration of the encounters that occur). For these reasons the validity and representativeness of our dataset will be greatly enhanced if the dataset is supplemented.

Further validation and improvement of the APEX model (or equivalent) will be supported by further observations in normal driving covering:

- older and newer vehicles
- larger and smaller vehicles (in terms of cabin volume).

It is likely the following data will be valuable if the assumptions of the APEX model are to be examined and a model is required which is more closely representative of the relevant physical processes incorporated in 'air exchange':

- particle size distributions (preferably internal and external, to probe physical size-dependent particle processes)
- measurement at multiple points in the vehicle cabin (to investigate differential exposure and the role of air mixing)
- measurement of passive trace gases (to eliminate the influence of particle-specific processes)
- detailed characterisation of instrument response to traffic aerosols
- detailed characterisation of inlet response.

Further experimental investigation of the AER is justified in order to provide the data to operationalise the APEX model. Observational studies should pay attention to:

- a wider range of vehicles (including cabin volumes and ages)
- repeated observations to overcome uncertainties and errors
- improved methods for determining the AER in recirculating air
- more attention to low vehicle speeds
- consideration of the presence or otherwise of cabin air filters
- data to test the APEX assumption that infiltration and exfiltration can be described by the same AER.

Development of an accompanying ELM or another approach to modelling the influence of route/traffic characteristics on exposure will be supported by further observations covering:

- more routes or route types (including longer and shorter journeys, non-CBD routes)
- peak-hour traffic and congestion
- identification and/or characterisation of the vehicle in front (for extended analysis of gross emitters and emission activity at intersections).

14 References

Arain, MA, R Blair, N Finkelstein, JR Brook, T Sahsuvaroglu, B Beckerman, L Zhang and M Jerrett (2007) The use of wind fields in a land use regression model to predict air pollution concentrations for health exposure studies. *Atmospheric Environment 41*: 3453–3464.

Boothe, VL and DG Shendell (2008) Potential health effects associated with residential proximity to freeways and primary roads: Review of scientific literature, 1999-2006. *Journal of Environmental Health 70*: 33–41.

Brugge, D, JL Durant and C Rioux (2007) Near-highway pollutants in motor vehicle exhaust: A review of epidemiologic evidence of cardiac and pulmonary health risks. *Environmental Health 2007*, doi:10.1186/1476-069X-6-23.

Davy, P, B Trompetter and A Markwitz (2009) Source apportionment of airborne particles in the Auckland region: 2008 update*. GNS science consultancy report 2009/165, June 2009 final.*

Di Marco, GS, S Kephalopoulos, J Ruuskanen, and M Jantunen (2005) Personal carbon monoxide exposure in Helsinki, Finland. *Atmospheric Environment 39*: 2697–2707.

Fisher, GW, KA Rolfe, T Kjellstrom, A Woodward, S Hales, AP Sturman, S Kingham, J Petersen, R Shrestha, D King (2002) *Health effects due to motor vehicle air pollution in New Zealand*. Report to the Ministry of Transport.

Fisher, G, T Kjellstrom, S Kingham, S Hales and R Shrestha (2007) *Health and air pollution in New Zealand. Final report*. Prepared for Health Research Council, Ministry for the Environment and Ministry of Transport.

Holt, S and R Beasley (2001) *The burden of asthma in New Zealand*. Report produced for the Asthma and Respiratory Foundation of New Zealand.

ISAAC (1998) Worldwide variations in the prevalence of asthma symptoms: the international study of asthma and allergies in childhood (ISAAC). *European Respiratory Journal 12*: 315–335.

Künzli, N, L Perez, F Lurmann, A Hickro, B Penfold and R McConnell (2008) An attributable risk model for exposures assumed to cause both chronic disease and its exacerbations. *Epidemiology 19,* no.2: 179–185.

Ministry for the Environment (MfE) (2002) Ambient air quality guidelines. Updated 2002. Accessed May 2011. *www.mfe.govt.nz/publications/air/ambient-air-quality-may02/ambient-guide-may02.pdf*

Ministry for the Environment (MfE) (2008) Good practice guide for assessing discharges to air from land transport. Accessed May 2011 from *www.mfe.govt.nz/publications/air/assessing-discharges-land-transport-jun08/*

Ministry of Transport (MoT) (2008) *New Zealand household travel survey: comparing travel modes*. v.1.4, Updated January 2008.

Morawska, L, Z Ristovski, ER Jayaratne, DU Keough and Z Ling (2008) Ambient nano and ultrafine particles from motor vehicle emissions: Characteristics, ambient processing and implications on human exposure. *Atmospheric Environment 42*: 8113–8138.

New Zealand Transport Agency (NZTA) (2010) *Draft Standard for Producing Air Quality Assessments for State Highway Projects*. Wellington: NZTA.

Riedl, MA (2007) The effect of air pollution on asthma and allergy. *Current Allergy and Asthma Reports 8*: 139–146.

Zhou, Y and JI Levy (2007) Factors influencing the spatial extent of mobile source air pollution impacts: a meta-analysis. *BMC Public Health* 7, 89 doi:10.1186/1471-2458/7/89.

Appendix A: Review of existing tools for assessing roadside air quality

A.1 Emission-dispersion modelling

A.1.1 Introduction

In regulatory contexts, the most common approach to modelling roadside air quality impacts is the combination of emission and dispersion modelling. The inputs to the emission model are traffic volumes, with some models also requiring data such as average speed, fleet composition (from two classes – light and heavy duty – up to 13 or more classes), percentage cold start, fuel formulation etc. In many cases this data will be derived from traffic modelling, leading to a three-part model in total (traffic-emissions-dispersion). Operational emission models generally incorporate a vehicle fleet model which essentially scales up emissions from individual vehicle types, based upon the predicted proportion of each type in the fleet. In New Zealand, until recently the New Zealand Transport Emissions Rate database (NZTER) was the only available emission/fleet model for this purpose. However, from 2008, NZTER was superseded as the default choice of vehicle emission model in New Zealand by the vehicle emissions prediction model (VEPM). VEPM essentially matches the New Zealand fleet model to the UK vehicle emissions model by mapping the mostly Japanese-built fleet to UK/European equivalents. It is currently unvalidated, but a programme of validation is being conducted by NIWA and several collaborators.

Dispersion models predict the effect of wind and turbulence on the dispersion of emissions and predict concentrations at downwind receptors. By combining an emission and a dispersion model for a single set of input data (a certain fleet travelling at a certain speed, combined with certain meteorology) a single set of outputs (concentrations at given receptors) is generated. This combination is well suited to investigating scenarios, such as worst cases. However, determination of ‘normal’, ‘typical’ or ‘average’ cases over a period of time is more demanding. A simple approach is to run multiple simulations for a range of likely inputs.

Emission-dispersion models demand a considerable amount of input data and have numerous stages and assumptions, all of which have inherent errors and uncertainties. Their performance is dependent upon the degree to which their inherent assumptions are violated (eg flat terrain and no obstacles are two common assumptions that are often violated), the amount and type of input data that is unknown and the demands made of the precision and accuracy of the model.

A.1.2 Emission models

Smit et al (2008a) defined three categories of emission models, based on the method by which congestion is incorporated:

- **Type A models**: These require driving pattern data as input. They are fully capable of modelling the effects of congestion, as data input explicitly takes congestion into account.
- **Type B models**: Driving pattern data is generated during the emission modelling process, from macroscopic variables such as traffic volume, traffic density and road infrastructure characteristics. Congestion-related variables are included in the modelling, so congestion is explicitly accounted for.
- **Type C models**: Emission factors are developed using measured driving pattern data. However, levels of congestion are usually implicit to the data and are not able to be changed by the model user.

Moreover, the level of congestion implicit to the model development is usually not obvious (ie model users are unaware whether emission factors were developed for free-flowing, partially congested or fully congested conditions).

Whereas type C models are better suited to large spatial scales (eg entire urban areas), type A and type B models are well suited to small scale analyses (eg neighbourhood level or individual road links).

Of these categories, type C models are the most commonly used. In a review of 58 studies using traffic emission models, Smit et al (2008a) reported 81% used type C models, while 16% used type A and only 3% used type B models. Most type C models were average speed models (Smit et al 2008a). This included New Zealand's VEPM.

However, average speed is not a universally satisfactory indicator of congestion for the purposes of determining traffic emissions. The relationship between speed and emissions is non-linear, and is influenced by vehicle fleet composition (National Roads Authority and DEFRA 1992/2007). Using chassis dynamometer tests, Ntziachristos and Samaras (2000) showed a low correlation between emission factors and mean travelling speed, which was at least partly due to differences in vehicle age and state of repair. The profile of vehicle types is also important. For example, large, heavy vehicles are more likely to cause congestion than small, light vehicles because they are slower to accelerate and demand more road space.

Using average speed is likely to underestimate the emissions caused by accelerations and decelerations (Lin and Ge 2006). Similarly, up to a four-fold difference in vehicle emissions can be experienced for the same average speed, depending on the variability of speed due to factors including congestion (EC 1995, cited in Smit et al 2008a).

Augmenting modelling results with experimentally derived traffic data, including the detection of traffic congestion, has been shown to reduce errors of modelled emission rates (Kuhlwein and Friedrich 2005). In the absence of experimental data for input, speed post-processing methods may be applied to improve emission estimates.

A different approach is to use modelled average speed distributions for traffic streams. Smit et al (2008b) noted the use of speed distributions did not explicitly account for the effect of different driving patterns, but did implicitly include driving dynamics. Errors caused by incomplete accounting of driving dynamics for specific roadways were considered to cancel out when aggregated into networks, so Smit et al (2008b) recommended the mean speed distribution modelling approach was suitable for application to large road networks.

While congestion is usually implicitly incorporated in average-speed models, the accuracy of emission estimates can be greatly improved by including a congestion algorithm (Smit et al 2008). At the very least, Smit et al (2008) recommended average-speed models should provide information on the level of congestion in the driving patterns that form the basis of the models, along with a recommendation on applications the models were suitable for.

A.1.3 CALINE

Probably the most widely used tool for estimating air quality downwind of a road is the dispersion model CALINE. It originated in the early 1970s, with version 4 (CALINE4) available and in use since the early 1990s. CALINE4's predictions have been tested against numerous datasets by different researchers (eg see the review by Nagendra and Khare 2002). It is generally one of the better performers among roadside dispersion models, albeit with a tendency to over-predict for parallel wind conditions and in low winds (Holmes and Morawska 2006). The performance of CALINE4, along with other similar deterministic models, deteriorates very close (order 10m) to the road (eg Levitin et al 2005). Nagendra and Khare (2002) concluded deterministic models such as CALINE4 were best suited for evaluating long-term averages. CALINE4 has been found to underestimate concentrations in non-smooth, heterogeneous driving conditions, such as congested driving or around intersections (eg Lin and Ge 2006). It also does not include the presence of buildings and other obstacles to dispersion, and it is therefore not surprising its performance also deteriorates in more urbanised, densely built-up locations (eg street canyons) (Yura et al 2007). CALINE4 does not include any modelling of aerosol dynamics or explicit nitrogen chemistry.

Despite these limitations CALINE4 (and its commonly used and similar rival, HIWAY2) are widely used due to their relative computational simplicity, which implies modest data requirements and short run-times on a PC. The mathematical core of CALINE4 is adopted in the AusRoads model developed by Victoria EPA in Australia, which is widely used in Australasia.

A.1.4 Other roadside dispersion models

Roadside dispersion models other than CALINE have been developed and are widely used. These include HIWAY-2, ROADWAY, CAR-FMI and ADMS-Roads. These models were extensively reviewed by Nagendra and Khare (2002) and Holmes and Morawska (2006). HIWAY-2 is a gaussian model developed by the USEPA, very similar in form to CALINE. CAR-FMI was developed in the 1990s by the Finnish Meteorological Institute and has been evaluated against experimental datasets (eg Kukkonen et al 2001). When recently compared with CALINE4, CAR-FMI gave a lower index of agreement (Levitin et al 2005). Only limited comparison of ADMS-Roads against CALINE has been performed (Ellis et al 2001).

A.1.5 Other roadside dispersion modelling approaches

Most widely used roadside dispersion models either fail to take account of the extra turbulent mixing induced by vehicles themselves, or incorporate a highly simplified treatment which is not road (or vehicle) specific. Several modelling approaches have been developed to account for this extra mixing, which is most critical for near-field impacts (eg Sahlodin et al 2007) but these are currently only at the 'research' stage and have yet to be widely adopted, tested and used in regulatory or policy contexts.

The quick urban and industrial complex model (QUIC) has been developed over the last decade at the Los Alamos National Laboratory in the United States (www.lanl.gov/projects/quic/index.shtml). It is intended for rapid use to model dispersion in complex urban areas with minimal data input. It is still in a developmental stage and has yet to be widely adopted for routine air quality management purposes.

A.2 Land-use regression and GIS models

Land-use regression (LUR) models are necessarily local. They are explicitly based on identifying which land use features influence spatial patterns in a specific pollutant at a specific place. Different pollutants have different sources, sinks and transport properties and hence their patterns are influenced by different variables to different degrees. Similarly, different locations have different densities of land-use data, eg one location may have large areas of parkland with deciduous trees, while another may be densely urban. Two cities, or different parts of the same city, might have quite different road widths and typical building heights. In these examples, the importance of variables such as parkland and building density, road width or distance from the road centreline will all influence pollutant variability in different ways, and a variable which is a powerful predictor in one area, may be a weak predictor in another.

This geographical specificity can often prevent model portability and intercomparison. LUR models are also often developed for a single pollutant due to logistical and financial limitations on data collection.

NO_2 is often the favoured pollutant due to the relatively inexpensive and easy deployment of diffusion tubes. Clougherty et al (2008) compared the ability of a wide range of variables describing traffic to describe daily NO_2, elemental carbon and $PM_{2.5}$ concentrations (averaged up to one mean concentration per site per season) for 44 homes across Boston. The traffic measures included weighted and unweighted traffic densities at five buffer distances, density of urban roads, total roadway length, total AADT $\times$ length, distance to the nearest road over various traffic thresholds, distance to truck routes, and AADT, diesel fraction and truck flow on nearest road. This study noted that actual traffic counts for minor roads are sparse and more uncertain, and length measures were more stable than ADT-based indicators. Total roadway length produced the strongest concentration estimates in the neighbourhoods studied, although it was noted that in areas with better traffic data, factors incorporating traffic density may fare better.

Henderson et al (2007) conducted a similar study based on the Greater Vancouver area. They found different regression variables describing traffic had different power for different pollutants, with road length performing better for NO, but vehicle density performing better for absorbance (a proxy for black carbon), although the margins were very small. The authors concluded that overall, vehicle density performed better than road length, but both types of variable were 'equally able to explain small-scale variability in pollutant concentrations' – although we need to take care over the interpretation of 'small scale' in the context of a model covering 2200km^2.

LUR models are based on exploiting pre-existing geographical datasets, especially those that can be expressed in GIS, as predictors of long-term average spatial patterns in air pollution. An inherent weakness of this approach is that it is unable to incorporate the effect of local winds, and especially the bias they introduce as any given location is likely to have a predominant wind direction and different degrees of dispersion in different directions. Arain et al (2007) have trialled the inclusion of wind data into a LUR model with a large 150 $\times$ 150km domain including the cities of Toronto and Hamilton. They incorporated data from 38 weather stations and isolated the typical winds at various times of day. Data from the evening traffic peak was found to provide the best model performance, and this was attributed to evening winds being more stable and representative of periods when traffic pollution levels are highest. Inclusion of wind data slightly improved the ability of the model to reproduce observed NO_2 variability. Recognising the lack of spatial coverage of observational wind data, relative to the spatial variability, this study also incorporated the use of regional weather model data. However, this data was weak at reproducing the local wind flows at a scale pertinent to the objective of identifying flows at 100m (or less) scale.

A.3 References

Arain, MA, R Blair, N Finkelstein, JR Brook, T Sahsuvaroglu, B Beckerman, L Zhang and M Jerrett (2007) The use of wind fields in a land use regression model to predict air pollution concentrations for health exposure studies. *Atmospheric Environment 41*: 3453–3464.

Clougherty, JE, RJ Wright, LK Baxter and JI Levy (2008) Land use regression modelling of intra-urban residential variability in multiple traffic-related air pollutants. *Environmental Health* 7: 17.

Ellis, K, C McHugh, D Carruthers and A Stidworthy (2001) Comparisons of ADMS-Roads, CALINE4, and UK DMRB model predictions for roads. *7th International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes*. Belgirate (Lake Maggiore), Italy, 28–31 May 2001.

Henderson, SB, B Beckerman, M Jerrett and M Brauer, M (2007) Application of land-use regression to estimate long-term concentrations of traffic-related nitrogen oxides and fine particulate matter. *Environmental Science and Technology 41*: 2422–2428.

Holmes, NS and L Morawska (2006) A review of dispersion modelling and its application to the dispersion of particles: An overview of different dispersion models available. *Atmospheric Environment 40*: 5902–5928.

Kuhlwein J and R Freidrich (2005) Traffic measurements and high-performance modelling of motorway emission rates. *Atmospheric Environment 39*: 5722–5736.

Kukkonen, J, J Häkönen, J Walden, A Karppinen and K Lusa (2001) Evaluation of the CAR-FMI model against measurements near a major road. *Atmospheric Environment 35*: 949–960.

Levitin, J, J Häkönen, J Kukkonen and J Nikmo (2005) Evaluation of the CALINE4 and CAR-FMI models against measurements near a major road. *Atmospheric Environment 39*: 4439–4452.

Lin, J and Y Ge (2006) Impacts of traffic heterogeneity on roadside air pollution concentration. *Transport Research Part D 11*: 166–170.

Nagendra, SMS and M Khare (2002) Line source emission modelling. *Atmospheric Environment 36*: 2083–2098.

Ntziachristos, L and Z Samaras (2000) Speed-dependent representative emission factors for catalyst passenger cars and influencing parameters. *Atmospheric Environment 34*: 4611–4619.

Sahlodin, AM, R Sotudeh-Gharebagh and Y Zhu (2007) Modelling of dispersion near roadways based on the vehicle-induced turbulence concept. *Atmospheric Environment 41*: 92–102.

Smit, R, AL Brown and YC Chan (2008a). Do air pollution emissions and fuel consumption models for roadways include the effects of congestion in the roadway traffic flow? *Environmental Modelling & Software 23*: 1262–1270.

Smit, R, M Poelman and J Schrijver (2008b) Improved road traffic emission inventories by adding mean speed distributions. *Atmospheric Environment 42*: 916–926.

Yura, EA, T Kear and D Niemeier (2007) Using CALINE dispersion to assess vehicular PM2.5 emissions. *Atmospheric Environment 4*1: 8747–8757.

Appendix B: Model development

B.1 Our modelling concept and approach

The modelling conducted within this project was based upon the concept that the dispersion modelling could be de-coupled from the emission modelling, due to the processes of dispersion being dependent upon meteorology alone and independent of emissions. Thus, we altered the conventional sequence and interdependencies of the various modelling components (figure B.1) so the dispersion of pollutants was 'pre-modelled' independently, and emission modelling was used to scale the results (figure B.2). Our intentions in doing so were to:

- reduce unnecessary duplication in dispersion modelling
- permit clarity over the relative roles of dispersion and emissions in the results (and the contribution of each to total uncertainties).

Figure B.1 The conventional approach to roadside atmospheric dispersion modelling

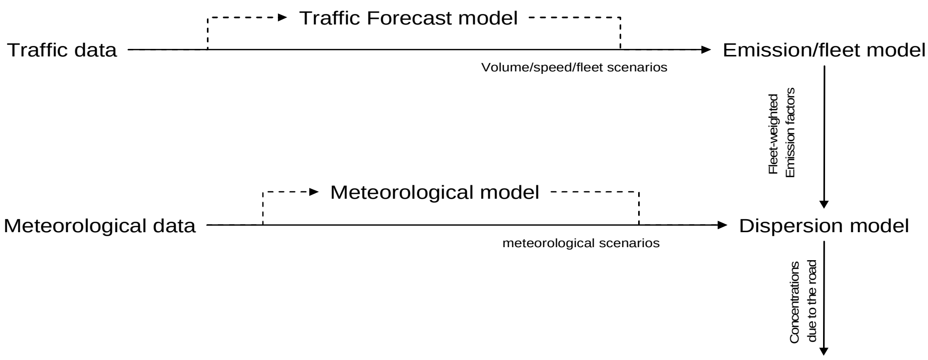
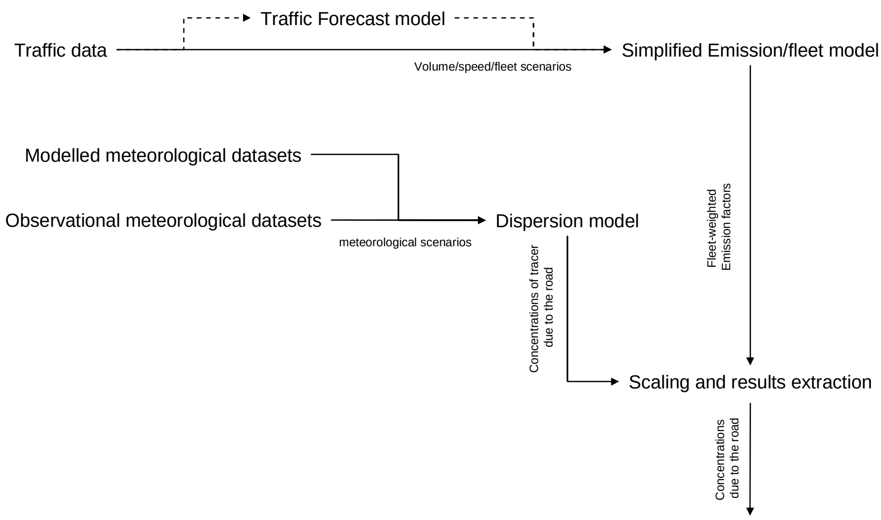


Figure B.2 A schematic representation of the approach to roadside modelling adopted in this project



To convert the generic dispersion results into road-specific results the emission modelling was conducted separately. Fleet-weighted emission factors and activity factors were calculated for each specific road link of interest. These results were then used to scale the generic dispersion modelling results in order to generate road-link-specific, long-term average pollutant concentration gradients.

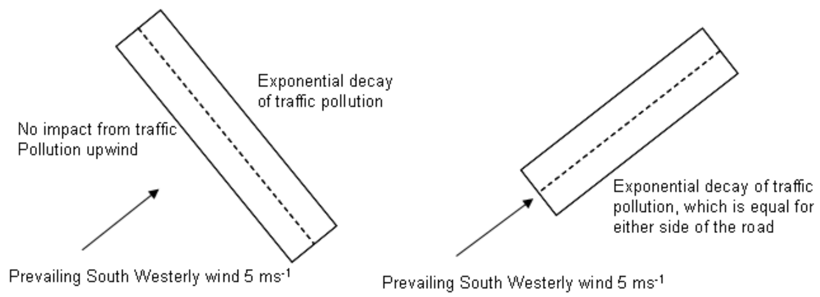
We hypothesised the long-term pattern of dispersion around any line emission source, such as a road, was principally a function of the local climate with other secondary variables of less importance. If we therefore set the emission source strength to unity, the resulting long-term pattern of dispersion was determined by how the long-term pattern of winds interacted with the orientation of the road. Secondary variables would be variations in mixing height, topography, land use and obstacles, such as buildings, earthworks and vegetation. Consistent with our objective of wanting to predict general long-term patterns of suburban dispersion, and a willingness to sacrifice detail for simplicity, we made the following simplifying assumptions:

- Topography is not significant (ie all land is flat).
- Variations in land use and the built environment are not significant (ie a uniform roughness length is applied in all cases).

Options to address and improve on these simplifications and the errors they might introduce were discussed in chapter 6 of this report.

If one accepts these assumptions then it becomes clear, for any given orientation of a road, the pattern of dispersion is determined by the meteorological dataset only. For illustrative purposes, imagine a hypothetical scenario in which the wind blows at 5 m s^{-1} from a south-westerly direction at all times (see figure B.3). For a road which is oriented perpendicular to this prevailing wind, no air quality impact upwind would be expected, while, downwind, the concentration of traffic pollutants would be expected to decay with distance. Alternatively, for a road which is oriented parallel to the prevailing wind, the concentration of traffic pollutants would decay equally with distance on both sides of the road, though the rate of decay would be different from that for the perpendicular road.

Figure B.3 An example of the expected dispersion patterns from a generic road orientated perpendicular and parallel to the prevailing wind direction



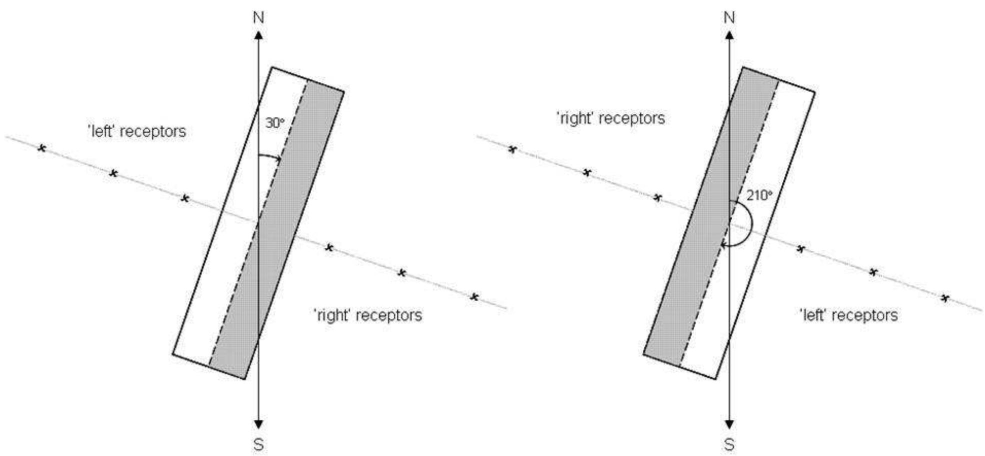
B.2 Dispersion modelling runs

Our modelling concept was to pre-model the pollutant concentration at various distances for all orientations of a generic road (assuming an emission factor of unity), in order to parameterise the long-term average pollutant concentrations in response to a given climate.

The dispersion modelling in this project was conducted using AusRoads, which is a roadside dispersion modelling package developed by, and available from, the Victoria EPA in Australia. The mathematical core of AusRoads is the long-established CALINE4 model (see appendix A). The CALINE suite of models is probably the most widely used tool internationally for estimating air quality downwind of a road.

The layout of the road link and receptors for the modelling is illustrated in figure B.4. We defined the road orientation as the clockwise angle between due north and the centre-line of the road. Receptors were placed at equal distances on either side of the road, along a line which was perpendicular to the centre-line. We included receptors at distances of 1, 2, 4, 6, 8, 10, 15, 20, 25, 30, 40, 50, 60, 80 and 100m on either side. Here, the 'side' of the road is defined as either 'left' or 'right' and is relative to its orientation. For example a road oriented at 30º (as shown in figure B.4) has left receptors to the north-west and right receptors to the south-east. Note, however, the same road could also be defined as having an orientation of 210º, in which case the receptors are swapped over with the left receptors lying to the south-east and the right receptors to the north-west.

Figure B.4 The generic road link and receptor layout for which pollutant concentrations have been calculated



The dispersion modelling for development of the RCM was carried out using two separate (hourly) observational meteorological datasets:

1 Observations from Auckland Airport for the two-year period 2004–2005

2 Observations from Wellington Airport for the whole of 2006.

These meteorological datasets included ambient temperature, wind speed, wind direction, Pasquill-Gifford stability class and mixing height. Table B.1 shows the set of parameters used in the AusRoads runs.

The road was initially laid out with an orientation of zero (ie south to north). We then re-oriented the road by rotating it clockwise in 15º intervals until a full 360º rotation had been completed. AusRoads was run for each orientation, so the overall results were hourly predicted concentrations at each receptor for each orientation in response to the given meteorological dataset.

Table B.1 Parameters used in the AusRoads simulations

Parameter	Value	Comment
Anemometer height	10m	
Met site roughness height	0.3m	Default for climate stations
Horizontal dispersion	Pasquill Gifford	As available in the meteorological dataset
Wind exponent	Irwin Urban	Recommended for non-rural areas
Averaging times	1 hr	
Surface roughness	0.4m	Residential default

B.3 Modelling and sensitivity test results

B.3.1 Hourly weighting of results to account for emissions bias

The dispersion model results are based on the assumption of unit emissions at all hours. Long-term averaging of this data risks predicting unrealistic results. Wind speed and direction have in-built temporal biases, as do traffic emission rates. For instance, emission rates in the early morning are low, so dispersion results at these times would be over-represented if this bias was not accounted for. Similarly, wind speeds are often lower in the morning and higher in the afternoon, so the effect of morning emissions is often amplified in terms of concentrations.

To account for such biases we weighted the raw dispersion results using a generic diurnal emissions profile. Two kinds of hourly traffic weightings were trialled based on traffic count observations for 2007 from SH1 at Takapuna/Mt Wellington for 'motorway' and counts conducted for NIWA on East Coast Road, Forrest Hill, North Shore, also in 2007, for 'arterial'. No significant difference was found in the dispersion results arising from the motorway and arterial profiles, therefore the arterial road traffic weightings were used as the default.

Within the traffic weightings for the specific road type the AusRoads output was corrected by applying an emission factor for that particular time of day in order to generate a diurnally corrected output. In addition, the day of the week is important as weekdays, Saturdays and Sundays have different profiles. Public holidays were considered to have Sunday profiles. Our approach did not allow for different diurnal profiles for different parts of the fleet (as scaling the results by a fleet-specific emission factor was done in the project-specific modelling).

After applying the hourly emissions weighting, mean concentrations were calculated for each receptor distance and orientation for every hour in the dataset.

B.3.2 Sensitivity to orientation

Figure B.5 shows the predicted long-term average concentration at three distances (10, 20 and 80m) on the right-hand side of the road for each road orientation. Generally it shows the greatest concentration elevations occur at orientations of around 60° and 210°, and the smallest concentration elevations occur around 0°.

For each distance, we compared the average of the predicted concentration on the left- and right-hand sides of the road for each orientation with the average concentration over all orientations. This procedure showed road orientations between 15° and 90° (or, equivalently, between 205° and 270°) gave above average concentrations, and road orientations between 105° and 180° (equivalently, between 285° and 0°)

gave below average concentrations. Road orientations of 45° and 105° gave the most symmetrical impacts, and road orientations of 0°, 15° and 165° gave the most asymmetrical impacts. These results were weakly related to receptor distance, with some minor variations at the most distant receptors.

Note that the results in this section are specific to the meteorological dataset used, which, in this case was Auckland. The generic pattern of dispersion could be different when using alternative meteorological datasets.

Generalised dispersion curves representing the maximum and minimum concentration over all orientations for each receptor distance, as well as the mean, are shown in figure B.6. At any given distance, the difference between the maximum and minimum concentration for any orientation (ie the maximum variation introduced by changes in orientation) was 24%–28%. Hence, orientation is an important variable and should be retained in the modelling package.

Figure B.5 The influence of road orientation on predicted long-term concentrations due to unit emissions as a function of perpendicular road distance (left-hand side)

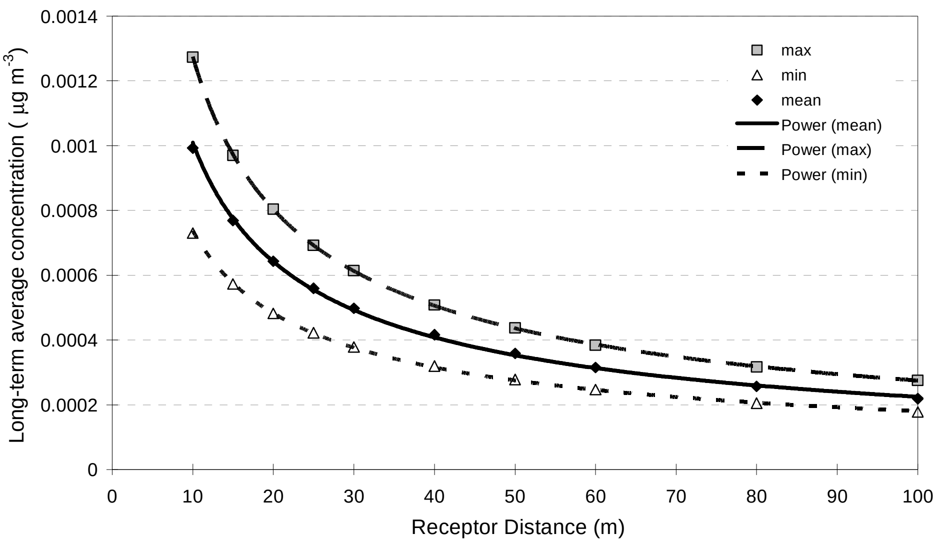
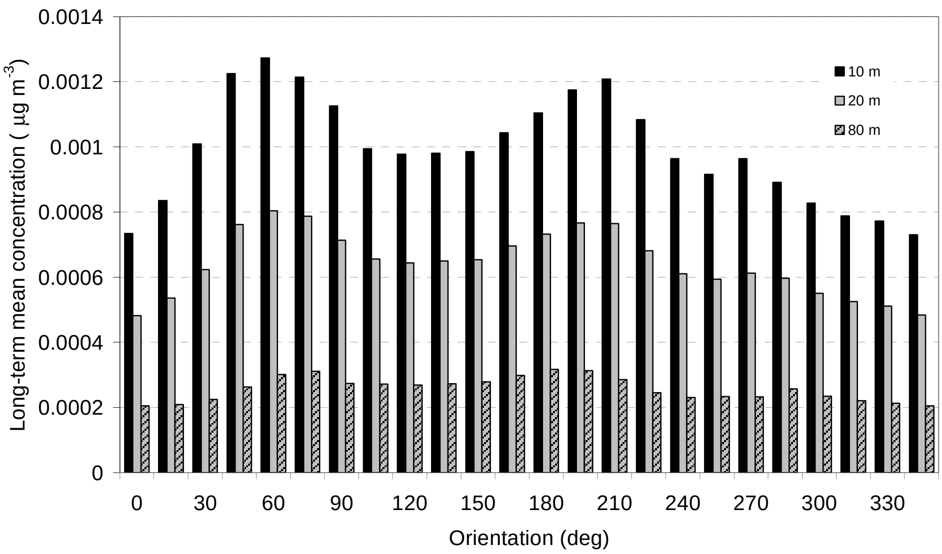


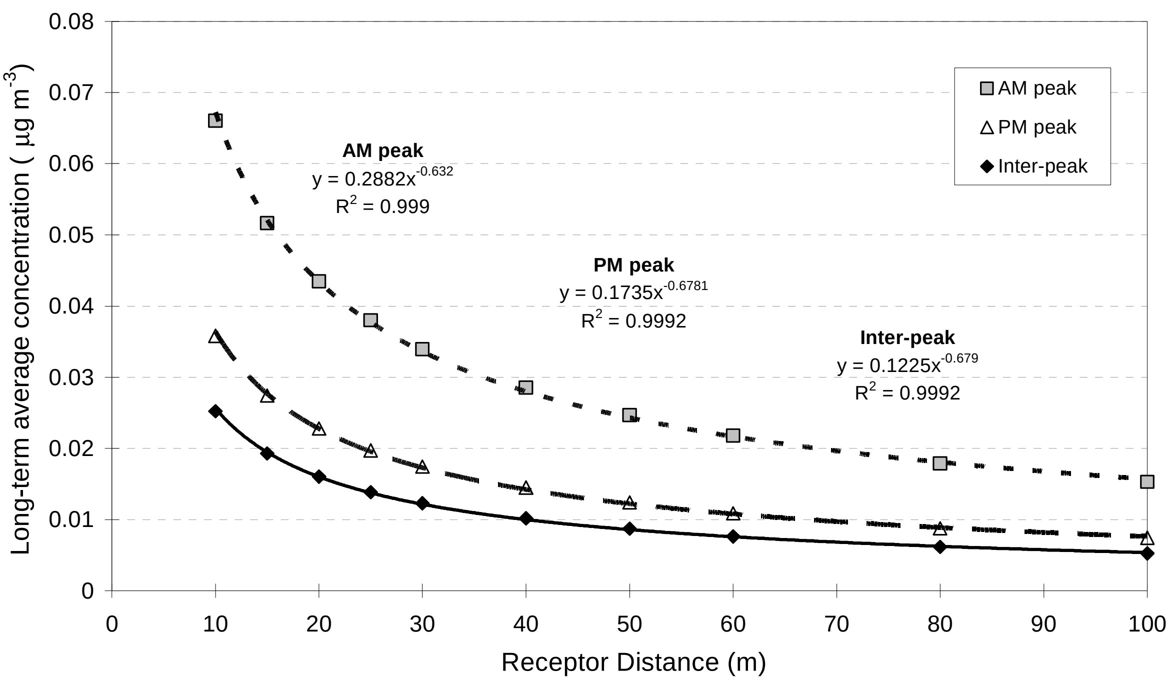
Figure B.6 Maximum, mean and minimum concentrations of all orientations for the full dataset



B.3.3 Significance of time of day

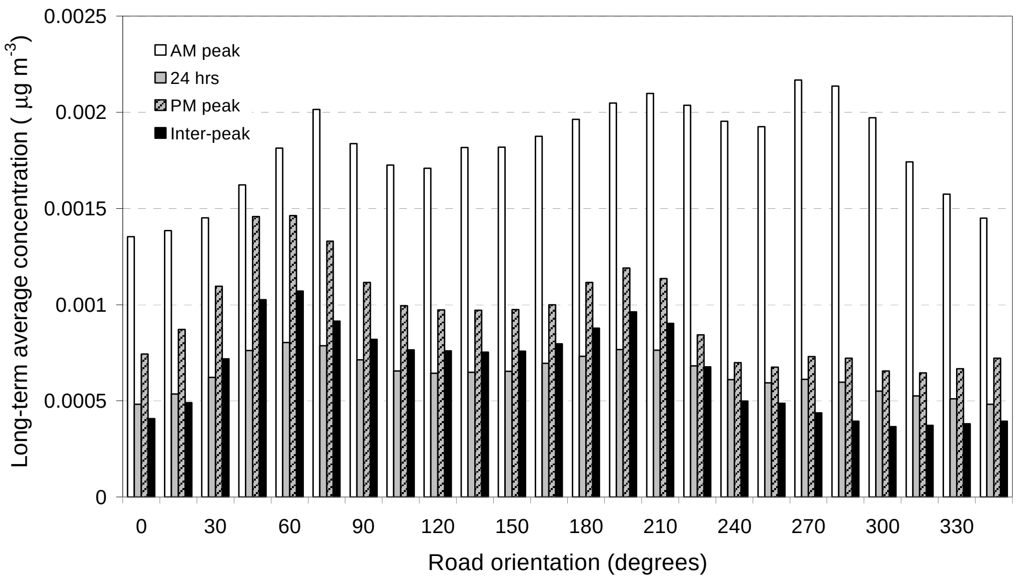
Long-term average concentrations were also calculated independently for all am peak (7am to 9am), pm peak (4pm to 6pm) and inter-peak (midday to 2pm) periods. The generalised dispersion curves representing the long-term average concentration (averaged over all orientations) based on the Auckland meteorological dataset are shown in figure B.7. The results for the inter-peak period are indistinguishable from the full 24-hour results and thus, for the sake of clarity, are not shown. It can be clearly seen that the predicted concentrations for the am peak exceeded those for the other periods. The am peak emissions were approximately 1.4 times greater than the 24-hour average emissions (based on the traffic weighting factors), but the am peak concentrations were approximately 2.8 times greater than the 24-hour average. This indicates a 40% elevation in the am peak concentrations was attributable to emissions weighting, and a 100% elevation was due to systematically poor dispersion conditions coinciding with the am peak period.

Figure B.7 Long-term average emission-profile-weighted concentration averaged over all road orientations using the Auckland meteorological dataset. Receptors closer than 10m to the road are not shown for clarity (also due to breakdown of power-law fit)



The relationship between concentrations and orientation was quite different in the am peak results from any other period. This is illustrated in figure B.8, which shows the predicted concentrations at 20m on the right-hand side of the road for the full range of orientations. In most cases the am peak predictions were between 1.6 and 3.3 times those for the inter-peak period. However, this ratio rose above 3 (peaking at 5) for orientations between 240° and 345°.

Figure B.8 Long-term average concentration as a function of perpendicular distance and road orientation with a traffic emission profile of an arterial road



For any given distance, the difference between the maximum and the minimum long-term average concentration for any orientation, ie the maximum variation introduced by orientation, was also found to be sensitive to the time of day. The maximum variation was 19% to 24% for the am peak results, 49% to 64% for the inter-peak results and 45% to 60% for the pm peak results.

B.3.4 Sensitivity to meteorological dataset

The effect a specific meteorological dataset, in particular the dataset year, has on the predicted long term pollutant concentration at each receptor was investigated by following the same procedures as described above. In this case, however, only output data produced from AusRoads for either the 2004 or 2005 Auckland Airport meteorological dataset was used for further processing.

The predictions made using the single year meteorological datasets were compared with those made using the 2004/2005 combined dataset in order to determine any significant difference caused by the use of a particular year of meteorological data. The results are summarised in table B.2.

Table B.2 Mean percentage difference in predicted long-term concentrations at receptors between those made using an individual meteorological year dataset and those made using the combined years meteorological dataset for the different traffic emission scenarios

Model scenario	24-hour average	am morning peak	inter-peak period	pm evening peak
Constant traffic emission profile	± 4.60	N/A	N/A	N/A
Diurnally weighted traffic emission profile	± 4.60	± 4.90	± 7.1	± 2.59

Table B.2 shows that, on average, there was a 3% to 7% difference between those predicted long-term concentrations made by AusRoads using a single year meteorological dataset and those made using the combined one. Consequently, the use of a different meteorological dataset can affect the dispersion modelling predictions made using AusRoads for a generic road; the effect is small compared with orientation, but may be sufficiently significant to justify further investigation.

The largest percentage differences in predicted concentrations from using different meteorological datasets were found to increase as the perpendicular receptor distance from the road increased. Certain road orientations were also shown to be more affected by the use of the different meteorological datasets (table B.3).

Table B.3 The road orientations that produced the largest percentage difference in long-term predicted concentrations between the individual meteorological year dataset and the combined meteorological year's dataset

Model scenario	24-hour average	am morning peak	Inter-peak period	pm evening peak
Constant traffic emission profile	120°, 135°, 330° and 345°	N/A	N/A	N/A
Diurnally weighted traffic emission profile	165°, 120°, 300° and 345°	165°, 180°, 300° and 315°	105°, 210°, 285° and 315°	225°, 240°, 315° and 345°

Table B.3 shows road orientations to the south-east and north-west tended to be most sensitive to changes in the specific year of the Auckland Airport meteorological dataset used within the dispersion modelling. Thus, in addition to affecting the predicted concentrations for an increasing perpendicular receptor distance, the concentrations for particular road orientations are also be affected by the use of a different meteorological dataset. However, as noted above, these differences are only relatively slight for a comparison of two years of different meteorological data. A more extensive sensitivity study would be required in order to ascertain if the differences observed above are significant and representative of other meteorological years. A commonly adopted approach would be to assess the 30-year norm and extremes within a 30-year period.

B.3.5 Conclusions resulting from the AusRoads sensitivity studies

Some points to note from the AusRoads sensitivity studies:

- The time of day (am, pm or inter-peak) is an important variable in determining receptor concentrations.
- Orientation is even more important when considered in conjunction with the time of day.
- The specific meteorological dataset being used is important and can affect the extent by which both road orientation and perpendicular distance from the road influence predicted concentrations.

B.4 Parameterisation of AusRoads results

B.4.1 Approach

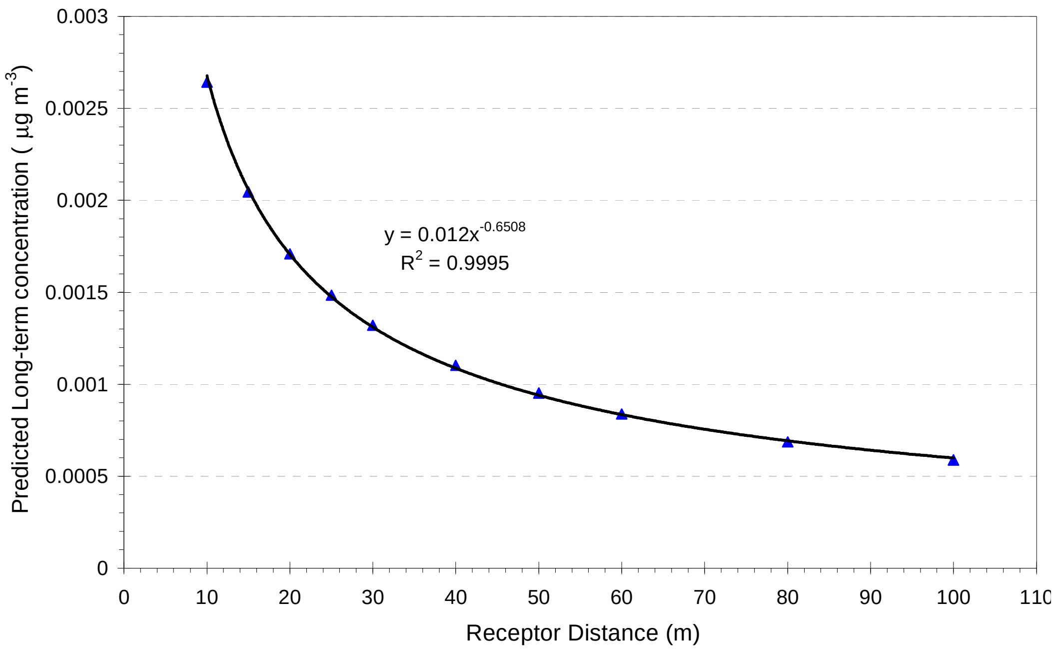
Each of the concentration gradients predicted from the AusRoads modelling for each road orientation can be approximated by a two-parameter power-law decay curve (as shown in figure B.9), especially if the first 10m are discarded. Thus, the equation for each concentration gradient (assuming an emission factor of unity) is of the form:

$$\text{predicted concentration} = A \times \text{distance}^{B} \qquad \text{(Equation B.1)}$$

where the parameters *A* (coefficient) and *B* (exponent) depend upon orientation, and also upon time period and location (ie meteorological dataset). Values of *A* and *B* for each orientation, time period and location are stored in look-up tables in the RCM (reproduced as tables B.4 and B.5 in this appendix). At present the parameterisation is based on the average concentration (rather than the maximum, for instance) due to the model's intended application in describing long-term impacts.

With regard to discarding the first 10m from the road, AusRoads is known to perform poorly within this distance (eg Levitin et al 2005), which is partly due to weaknesses in the model formulation and also due to the potential influence of other factors external to the dispersion model (eg vegetation). Furthermore, results within 10m have limited application, as receptors relevant for long-term assessment are unlikely to be so close to a major road.

Figure B.9 Example of a power-law decay curve used to parameterise the long-term average concentration gradient predicted by AusRoads for an Auckland road in the am period, oriented at 120º



B.4.2 Look-up tables for RCM parameters

Table B.4 RCM look-up table for Auckland metset

	Parameterised values for each time period							
	exponent				co-efficient			
Road orientation	AM	IP	PM	24 hr	AM	IP	PM	24 hr
0	-0.592907	-0.646035	-0.62907	-0.613527	0.007998	0.00283	0.004904	0.003035
15	-0.645028	-0.725635	-0.709762	-0.669944	0.009545	0.00432	0.007288	0.003978
30	-0.672465	-0.833273	-0.751905	-0.725408	0.010865	0.008673	0.010406	0.005464
45	-0.652268	-0.846913	-0.803779	-0.74945	0.011409	0.012842	0.016043	0.007146
60	-0.650478	-0.73076	-0.708196	-0.698309	0.012725	0.009557	0.012222	0.006505
75	-0.687061	-0.626295	-0.654647	-0.662484	0.015706	0.005976	0.009437	0.005718
90	-0.7387	-0.606652	-0.641577	-0.681528	0.016691	0.005042	0.007613	0.005485
105	-0.674399	-0.581635	-0.613983	-0.628123	0.012941	0.004372	0.006248	0.004297
120	-0.650771	-0.60379	-0.607168	-0.622688	0.011986	0.00463	0.005992	0.004151
135	-0.643139	-0.60536	-0.618855	-0.618763	0.012423	0.004616	0.006189	0.004135
150	-0.587794	-0.596978	-0.626319	-0.606617	0.010576	0.004531	0.006357	0.004021
165	-0.586126	-0.58724	-0.607245	-0.604484	0.010839	0.004634	0.00618	0.004249
180	-0.572319	-0.602811	-0.637084	-0.599911	0.010912	0.005356	0.00754	0.004424
195	-0.5964	-0.670958	-0.714	-0.639332	0.012208	0.007188	0.010086	0.005199
210	-0.61991	-0.78487	-0.797439	-0.698778	0.013408	0.009462	0.012349	0.006192
225	-0.612753	-0.90278	-0.849634	-0.720059	0.012733	0.009971	0.010659	0.005863
240	-0.623451	-0.804296	-0.80961	-0.691793	0.01262	0.005537	0.007856	0.004842
255	-0.635656	-0.713321	-0.693964	-0.662581	0.012904	0.004116	0.005401	0.00431
270	-0.702876	-0.655869	-0.687891	-0.688768	0.017698	0.003126	0.005727	0.004809
285	-0.607164	-0.60115	-0.616971	-0.603691	0.013173	0.002397	0.004582	0.00364
300	-0.622405	-0.641715	-0.587854	-0.606917	0.012695	0.002501	0.003811	0.003387
315	-0.621963	-0.660689	-0.603094	-0.614769	0.011176	0.002687	0.003919	0.0033
330	-0.591558	-0.652787	-0.618884	-0.622613	0.009257	0.002689	0.004259	0.003297
345	-0.598207	-0.658809	-0.632279	-0.612239	0.008684	0.002833	0.004791	0.003026

Table B.5 RCM look-up table for Wellington metset

	Parameterised values for each time period							
	exponent				co-efficient			
Road orientation	**AM**	**IP**	**PM**	**24 hr**	**AM**	**IP**	**PM**	**24 hr**
0	-0.531569	-0.595175	-0.577826	-0.553995	0.062607	0.03269	0.039168	0.054795
15	-0.555852	-0.586606	-0.57272	-0.548637	0.072148	0.032113	0.040413	0.056319
30	-0.558416	-0.629498	-0.615809	-0.580596	0.07753	0.041051	0.053599	0.067262
45	-0.629621	-0.659536	-0.672619	-0.635934	0.111116	0.057	0.072053	0.089122
60	-0.750981	-0.716763	-0.720095	-0.735216	0.169809	0.092143	0.102875	0.139021
75	-0.823121	-0.819726	-0.802942	-0.815273	0.18564	0.165049	0.157586	0.187326
90	-0.85596	-0.904015	-0.859251	-0.839927	0.183229	0.183456	0.152316	0.170837
105	-0.791736	-0.774183	-0.766689	-0.751284	0.108976	0.112512	0.106165	0.105387
120	-0.719019	-0.670806	-0.707081	-0.688524	0.077101	0.06725	0.085924	0.081357
135	-0.619798	-0.584305	-0.602158	-0.636642	0.044428	0.044306	0.056762	0.06585
150	-0.576866	-0.547245	-0.564145	-0.578795	0.033961	0.035415	0.045847	0.051996
165	-0.599206	-0.557097	-0.549389	-0.585652	0.034275	0.035729	0.042054	0.049528
180	-0.559058	-0.551549	-0.553306	-0.56367	0.028393	0.036471	0.043519	0.045902
195	-0.56195	-0.540478	-0.55548	-0.540336	0.029662	0.038013	0.044876	0.042827
210	-0.620577	-0.591599	-0.569205	-0.577135	0.043954	0.049504	0.049856	0.053072
225	-0.679886	-0.669723	-0.672826	-0.658857	0.062628	0.068136	0.0787	0.080465
240	-0.823543	-0.749819	-0.707361	-0.731691	0.130364	0.083273	0.088593	0.111672
255	-0.861629	-0.868751	-0.791004	-0.783288	0.205724	0.10304	0.112631	0.144846
270	-0.783626	-1.033357	-0.866104	-0.793168	0.186798	0.199661	0.16491	0.173788
285	-0.688132	-0.832211	-0.817647	-0.727901	0.139697	0.116512	0.156946	0.150852
300	-0.622341	-0.734926	-0.701759	-0.645208	0.101192	0.080776	0.089908	0.102522
315	-0.597342	-0.633682	-0.613373	-0.598586	0.087365	0.051856	0.058241	0.076257
330	-0.540254	-0.602209	-0.593474	-0.566953	0.066267	0.040872	0.047767	0.061596
345	-0.537114	-0.606753	-0.603985	-0.596608	0.062174	0.036833	0.045283	0.065089

Appendix C: Literature review of in-vehicle exposure

C.1 In-vehicle exposure observations

C.1.1 Types of study

Several in-vehicle studies have been conducted, but are not considered in this review. This is principally for two reasons. Firstly, we have excluded studies in which measured concentrations have been compared to alternative transport modes. This is not pertinent to our project as we have focused on road vehicle emissions and exposure in cars so we may capture the exposure of the majority. However this is the subject of another NZTA-funded project led by Simon Kingham (University of Canterbury). Secondly, we have excluded consideration of studies in which only a single journey-average concentration has been reported, as this does not permit any mechanistic insight into the causes of that concentration and its potential variation. An overview of all available studies (up to 2005) is provided in the review by Kaur et al (2007).

C.1.2 Contribution of in-vehicle exposure to total exposure

Within the large European EXPOLIS study, 201 volunteers in Helsinki carried personal CO monitors for two consecutive weekdays, as reported by Di Marco et al (2005). On average the participants spent only 8.1% of their time in transport, compared with 55.2% at home and 88.0% indoors. Those using cars spent 4.8% of their time on average in the car. Mean concentrations of CO in the cars used were 3.2mg m^{-3}, compared with <2 in every other micro-environment. This study indicated for central urban residents CO exposure occurred through much of the day. For suburban commuting residents the CO exposure occurred mostly during commuting but the total effect on the daily dose was similar for each group.

C.1.3 Comparison of in-vehicle exposure to ambient exposure

The California Air Resources Board equipped an electric Toyota RAV4 as a mobile monitoring platform which has been used in field studies in Los Angeles. Westerdahl et al (2005) reported that concentrations of particle number concentration (PNC), black carbon (BC), NO and CO were an order of magnitude greater than those observed on residential streets. Further data was analysed by Fruin et al (2008) showing freeway concentrations were dominated by emissions from diesel vehicles, whereas residential PNCs were associated with hard acceleration of petrol vehicles with concentrations roughly one-third of those on freeways.

C.1.4 Routes, duration and contribution of vehicle-use to total exposure

Fruin et al (2008) combined data recorded from a mobile platform with other data observed in micro-environments typically occupied by Los Angeles citizens and concluded ‘33%–45% of total ultra-fine particle (UFP) exposure for Los Angeles residents occurs due to time spent travelling in vehicles’.

In the school-commuting study of Ashmore et al (2000), data derived from questionnaires distributed to destination schools identified that journey times were shorter by car. This implied integrated doses between car and pedestrian journeys were comparable and indicated the importance of the way exposure was reported (mean, peak or integrated exposures). The authors concluded the home location, rather than the mode of travel, was the crucial determinant of exposure, and the route selection and journey time determination were crucial factors in travel exposure assessment.

C.1.5 Inter-modal comparison

Ashmore et al (2000) equipped volunteers (acting as surrogates for schoolchildren travelling to schools) with personal CO monitors and GPS receivers. Four routes were travelled simultaneously by car and on foot. The car journeys were found to lead to consistently higher CO exposures than walking.

Gulliver and Briggs (2004) compared PM concentrations recorded in a car with pedestrian exposure on the same two routes in Northampton (UK). Relatively little variation was reported in average PM_1, $PM_{2.5}$ and PM_{10} concentrations between mode. We do not find this result surprising considering the choice of an integrating optical nephthelometer (OSIRIS) to determine concentrations. With a lower particle size cut of 0.4mm this instrument failed to observe the majority of fresh exhaust particles. It had too low a flow rate to observe coarse particles with sufficient statistical significance for the short observational durations in the study, and was biased to long-range and/or secondary particles. The study did indicate correlations with a fixed PM_{10} monitoring site (1km away) were weak. Gulliver and Briggs (2004) concluded 'efforts to reduce car usage, by encouraging people to take short journeys on foot, may actually increase exposures to air pollution. This is because the longer journey times on foot compared with those by car would lead to raised overall exposures'. We believe this is over-simplistic and cannot be a general conclusion, as explained further below.

A much larger follow-up study was conducted in London along 48 routes in London (Briggs et al 2007). Crucially, as with the studies cited above, routes were chosen to be representative of journeys to school, ie they were predominantly short and in suburban areas. Routes were also chosen so walking or car were reasonable modal choices. Monitoring of PM using the OSIRIS was supplemented with a PTrak portable condensation particle counter (CPC) reporting PNC. Average exposures during walking were consistently higher than in-vehicle exposure, with the excess being greater for coarse particles and smaller for PNC. This result was largely related to the filtration of particles entering the vehicle cabin space.

C.1.6 Observational studies – critique and conclusions

The above studies seek to draw general conclusions based on journey-average ratios (whether between modes or considering vehicle internal/external concentration ratios). We feel this can be misleading and cannot be easily generalised. Briggs et al (2007) conceded their consultation of the literature (as ours, below) indicated the ventilation of the vehicle (including air conditioning, installed filtration, vent speeds etc) had a major influence on how internal concentrations responded to external concentrations. In the studies of Briggs et al (2007) a single fixed setting was employed (windows closed, air conditioning off, fans set to medium) which could correspond to a relatively high air exchange rate (see below). When combined with a suburban route emission profile this could lead to relatively lower ratios compared with what might have been observed with the same vehicle, but with settings leading to lower air exchange, and a more intense external emission profile (eg on a motorway). Briggs et al (2007) also noted whereas in-car concentrations of coarse particles tended to decay over the duration of a journey (presumably due to deposition and reduced infiltration), PNC tended to rise over time, implying accumulation. Briggs et al noted in parentheses that this occurred 'albeit still not to concentrations equivalent to those experienced by the walker'. This may only be true, however, because of the short duration of journeys. For longer journeys, and those involving occasional encounters with high external concentrations (such as gross emitting vehicles) we may speculate accumulation will continue and could easily exceed external or pedestrian concentrations, and (unpublished) data collected by NIWA shows this to be a regular occurrence. These inconsistencies show the need for a more deterministic approach to generalising in-vehicle exposure.

C.2 Health effects of normal in-vehicle exposure

Our review revealed no literature on the effects on health of normal in-vehicle exposure.

Riediker et al (2004) reported a unique study of a cohort who spent longer amounts of time than average in cars. North Carolina State Highway Patrol troopers took part in a thorough study among healthy young non-smoking men to assess the effects of $PM_{2.5}$ in vehicles during a nine-hour shift. Physiological monitoring was undertaken and monitoring of $PM_{2.5}$ in the vehicles, although associations with fixed ambient and roadside $PM_{2.5}$ were also considered. The in-vehicle concentrations were generally lower than the concentrations recorded at the outdoor fixed sites. This is not unexpected as the vehicles spent only a limited period of the shift driving in busy traffic, and due to the partial filtering of particles of outdoor source afforded by the vehicle envelope. Mean nine-hour exposure was 24μg m^{-3} of $PM_{2.5}$

A few hours after exposure undesirable effects arose on vagal activity (ectopic beats), peripheral blood inflammatory markers (C – reactive protein) and coagulation markers (fibrinogen). Despite the lower in-vehicle concentrations, these effects were more strongly associated with in-vehicle $PM_{2.5}$ than external. The largest effect on heart-rate variability was seen on waking the morning after the in-vehicle exposure. This study is significant because it heralds cardiovascular effects involving inflammation, coagulation and cardiac rhythm among a group at otherwise low risk for such outcomes.

C.3 Tools for the assessment of in-vehicle exposure

C.3.1 GIS-based techniques

Gulliver and Briggs (2005) presented a GIS-based system for modelling human journey-time exposures to traffic-related air pollution called STEMS (space-time exposure modelling system). This system combined a traffic model (SATURN), a dispersion model (ADMS) and a background air quality model to produce hourly pollution maps for a local domain. Time-activity data and personal monitoring was then combined to plot the exposures of persons moving through the mapped air pollution domain, whether on foot or in a car. Although the modelling approach had great promise, the demonstration provided in the paper was severely hampered by the choice of PM_{10} as the pollutant to be modelled. This pollutant was dominated by background sources with relatively small local gradients. The observational data upon which the model was derived (pedestrian and in-car PM_{10} data) was also weak and did not allow for a mechanistic understanding of the roles of road dust, the enclosure and filtration provided by the cabin. As with most empirical models, this particular model was very local in its applicability, and the key exposure assessments were based on a limited range of journey types (routes, origins, destinations and durations) with applicability to other types of journey being untested. The authors themselves noted that 'Choice of route… has a considerable effect on journey-time exposures'. It was noted the results were very sensitive to the origin of meteorological data (ie observation site), especially for morning journeys when the meteorological conditions most relevant for dispersion were most changeable.

C.4 A mechanistic understanding of in-vehicle exposure

C.4.1 Factors influencing infiltration

Air exchange rate

The concentrations of traffic pollutants arising inside a vehicle as it drives down a road are dependent upon the rate at which particles enter the vehicle, and the rate at which they leave it. The net effect of these processes can be described by the air exchange rate (AER) of the vehicle. The AER, in turn, is determined by:

- whether the windows are opened or closed
- the operation of the vents and air conditioning
- the leakiness of the bodywork
- the speed of the vehicle
- the external wind speed and level of air turbulence.

Other processes which may be significant include the removal of pollutants within the car, principally by deposition to internal surfaces (including the lungs of occupants). This will be most significant for the most rapidly depositing pollutants – ultrafine particles. Deposition (by interception and impaction more than by diffusion) may be expected to preferentially remove coarser particles. Ultrafine particle number concentrations may also be reduced by coagulation.

Stationary/passive ventilation

Quantifying the AER is the key to predicting and quantifying the exposure of vehicle occupants on a mechanistic basis. As experiments on stationary vehicles are simpler to conduct than for moving vehicles, there has been (initially at least) more data available on stationary than on moving vehicles. Fletcher and Saunders (1994) conducted systematic tests on five stationary vehicles. They found a power law relationship between the AER and wind speed (the power varying from 1.05 to 1.33, with a general value of 1.24 proposed) with wind direction relative to the car playing a minor role. With the vents closed the AER varied negligibly between the five vehicles. With the vents open the AER was higher, and inter-vehicle differences were observable, but the power law relationship with wind speed remained. The AER typically varied over the range 0.4 to 10 hour^{-1}.

One of Fletcher and Saunders' test vehicles was studied in motion with the vents closed. A similar power law with respect to speed (in this case vehicle speed on the assumption it was much faster than wind speed) was found, but with the AER approximately 40% higher. Rodes et al (1998) reported AERs of 13.5–39h^{-1} at 55mph with windows closed and vents on low. Ott et al (2008) tested the Fletcher and Saunders model and concluded it successfully predicted the relationship between the AER and vehicle speed for any vehicle with passive ventilation. The model predicted AERs of around 20h^{-1} at 40mph, approaching 40h^{-1} at 70mph.

Effect of opening windows

Rates of 1–4h^{-1} were measured in stationary cars with windows closed, by Park et al (1998), but this rose to 13–26h^{-1} with windows open. Ott et al (2008) found opening one window by 3in dramatically increased the AER (by approximately 10-fold).

Active ventilation

Active ventilation (fans) probably increases the AER in a way that is likely to be more difficult to generalise. Ott et al (2008) observed values of 20–40h^{-1} at 20–60mph. Setting the air conditioning to maximum severely reduced the AER to less than 10h^{-1} (if all windows were closed).

Other observations of the AER

Zhu et al (2007) estimated rates of 1–2min^{-1} in a car with windows closed, but with various ventilation settings, during freeway driving in Los Angeles, averaging 50–60mph. Batterman et al (2006) reported 1.5min^{-1} in a car at 100km/h^{-1} under low-to-medium vent conditions.

Filtration

Zhu et al (2008) conducted 32 sets of two-hour measurements on Los Angeles freeways in a specially equipped van. The van was designed to deliver filtered or unfiltered air to an exposure chamber in which a wide range of gas and particle measurements were made. When air was passed through the HEPA filter 97% of the particles (expressed as an ultrafine number concentration) were removed on average.

Qi et al (2008) conducted detailed studies on the relative roles of the vehicle cabin and cabin air filters on in/out concentration ration and particle removal. They found each to be significant, such that the presence of cabin filtration needs to be considered when trying to make general predictions across a vehicle fleet. The tested air filter exhibited a classical size-dependent behaviour with minimal filtration around 400nm, and was less effective for ultrafine particles with increased fan speed. Its general effectiveness was illustrated when the fan was off, leading to a gradual decay in in-cabin concentrations, implying a combination of deposition and a lack of infiltration.

This study highlighted the significance of vehicle ventilation design. Each of the two models tested had the filter in a different part of the ventilation system so one filtered recirculating air while the other did not. The study also reported how some new vehicles periodically interrupted air recirculation with a brief injection of ambient air.

As with the inter-modal studies discussed above, this study used a simple measure with which to make judgements (average exposure reduction over a fixed length of time) which we believe is insufficiently general to be of transferrable use, as it is dependent on too many factors, including external concentration profile. Results from different on-road tests could not be validly compared because of the differing external profiles (ie forcings to the filter) encountered on each journey. This was admitted in the fan off case (with minimum air exchange) in which this setting would be considered best or worst case depending on the initial internal concentration (low or high, respectively). The winter part of this study was also conducted when cabin temperatures of 22ºC contrasted sharply with external temperatures of -5ºC, yet the effect of this on particle dynamics, which could be considerable for ultrafines, was barely discussed.

Larger vehicles

Vehicles with larger internal cabin volumes might be expected to have different AERs. A study of these in a horse trailer (Purswell et al 2006) indicated a maximum AER of 1.42 min^{-1} at 97km/h with all windows and vents open. In extensive controlled tracer studies in school buses, exchange rates of 0.2–0.8 min^{-1} have been measured at 40km/h with windows closed and 1.6–5 min^{-1} with windows open (Fitz et al 2003). Experimental studies on in-service buses in the UK have led to estimated rates of 0.14 min^{-1} (windows closed) and 0.35 min^{-1} (windows open) (Leavey and Longley, 2007). These values are substantially lower than those above for smaller cabins, and presumably reflect the larger volume of air to be mixed.

C.4.2 Conclusions

These studies may not all be immediately compared as different approaches were used to derive the air exchange rate. However, in summary it appears that for vehicles with small cabin values of 0.5–2 min^{-1} may be typical, perhaps biased towards lower values where driving with closed windows is more common.

Variability between vehicles and between journeys can be large, however. Operation of the fan and its settings, air conditioning, opening of windows and the presence of filtration are all very significant variables, and no study has yet comprehensively managed to model all of these factors together.

C.5 References

Ashmore, MR, K Batty, F Machin, J Gulliver, A Grossinho, P Elliott, J Tate, M Bell, E Livesley and D Briggs (2000) Effects of traffic management and transport mode on the exposure of schoolchildren to carbon monoxide. *Environmental Monitoring and Assessment 65*: 49–57.

Batterman, S, CR Jia, G Hatzivasilis and C Godwin (2006) Simultaneous measurement of ventilation using tracer gas techniques and VOC concentrations in homes, garages and vehicles. *Journal of Environmental Monitoring 8*, no2: 249–256.

Briggs, DJ, K deHoogh, C Morris and J Gulliver (2007) Effects of travel mode on exposures to particulate pollution. *Environment International 34*: 12–22.

Di Marco, GS, S Kephalopoulos, J Ruuskanen, and M Jantunen (2005) Personal carbon monoxide exposure in Helsinki, Finland. *Atmospheric Environment 39*: 2697–2707.

Fitz, DR, AM Winer, S Colome, E Behrentz, LD Sabin, SJ Lee, K Wong, K Kozawa, D Pankratz, K Bumiller, D Gemmill and M Smith (2003) *Characterizing the range of children's pollutant exposure during school bus commutes.* Final report. Prepared for the California Air Resources Board.

Fletcher, B and CJ Saunders (1994) Air-change rates in moving and stationary vehicles. *Journal of Hazardous Materials 38:* 243–256.

Fruin, S, D Westerdahl, T Sax, C Sioutas and P Fine (2008) Measurements and predictors of on-road ultrafine particle concentrations and associated pollutants in Los Angeles. *Atmospheric Environment 42*: 207–219.

Gulliver, J and DJ Briggs (2004) Personal exposure to particulate air pollution in transport microenvironments. *Atmospheric Environment 38*: 1–8.

Gulliver, J and DJ Briggs (2005) Time-space modelling of journey-time exposure to traffic-related air pollution using GIS. *Environmental Research 97*: 10–25.

Kaur, S, MJ Nieuwenhuijsen and RN Colvile (2007) Fine particulate matter and carbon monoxide exposure concentrations in urban street transport microenvironments. *Atmospheric Environment 41:* 4781–4810.

Leavey, AL, ID Longley, MW Gallagher, R Agius and CL Martin (2007) Insights into ultrafine particle exposure to bus commuters gained from measurements in Manchester, UK. *Proceedings of the Air and Waste Management Association's Annual Conference and Exhibition, AWMA 3*: 1705–1710.

Ott, W, N Klepeis and P Switzer (2008) Air change rates of motor vehicles and in-vehicle pollutant concentrations from secondhand smoke. *Journal of Exposure Analysis and Environmental Epidemiology 18*: 312–325.

Ott, W, P Switzer and N Willits (1994) Carbon monoxide exposures inside an automobile travelling on an urban arterial highway. *Journal of the Air and Waste Management Association 44:* 1010–1018.

Park, JH, JD Spengler, DW Yoon, T Dumyahn, K Lee and H Ozkaynak (1998) Measurement of air exchange rate of stationary vehicles and estimation of in-vehicle exposure. *Journal of Exposure Analysis and Environmental Epidemiology 8*: 65–78.

Purswell, JL, RS Gates, LM Lawrence, JD Jacob, TS Stombaugh and RJ Coleman (2006) Air exchange rate in a horse trailer during road transport. *Transactions of the American Society of Agricultural and Biological Engineers 49*: 193–201.

Qi, S, N Stanley, DYH Pui and TH Kuehn (2008) Laboratory and on-road evaluations of cabin air filters using number and surface area concentration monitors. *Environmental Science & Technology 42*: 4128–4132.

Riediker, M, WE Casclo and TR Griggs (2004) Particulate matter exposure in cars is associated with cardiovascular effects in healthy young men. *American Journal of Respiratory and Critical Care Medicine 169*: 934–940.

Rodes, C, L Sheldon, D Whitaker, A Clayton, K Fitzgerald, J Flanagan, F DiGenova, S Hering and C Frazier (1998) *Measuring concentrations of selected air pollutants inside California vehicles: Final report*. ARB contract no.95-339, prepared for California Environmental Agency & South Coast Air Quality Management District.

Westerdahl, D, S Fruin, T Sax, PM Fine and C Sioutas (2005) Mobile platform measurements of ultrafine particles and associated pollutant concentrations on freeways and residential streets in Los Angeles. *Atmospheric Environment 39*: 3597–3610.

Zhu, YF, A Eiguren-Fernandez, WC Hinds and AH Miguel (2007) In-cabin commuter exposure to ultrafine particles on Los Angeles freeways. *Environmental Science & Technology 41*: 2138–2145.

Zhu, YF, DC Fung, N Kennedy, WC Hinds and A Eiguren-Fernandez (2008) Measurements of ultrafine particles and other vehicular pollutants inside a mobile exposure system on Los Angeles freeways. *Journal of the Air Waste Management Association 58*: 424–434.

Appendix D: NIWA's observational pilot study

D.1 Note on instruments used

In all of the observational studies described in this report, particle number concentrations (PNCs) were measured using one or two TSI 'P-Trak' Model 8525 portable condensation particle counters (CPCs) (one of which was kindly loaned by the Auckland Regional Council).

The manufacturers do not quantify the instrument's accuracy. However, independent research has compared this model's performance against a more sophisticated CPC for which accuracy is specified. Co-location experiments, including at a busy roadside environment showed the P-Trak was reliable and yielded comparable results (Matson et al 2004). The manufacturers claim the instrument responds to particles in the size range 20nm–1μm in diameter. Independent research has indicated a practical lower size cut of 25–30nm (Zhu et al 2006). This limits the instrument's ability to sample fresh traffic emissions in which substantial numbers of particles below 30nm are found (Zhu et al 2006). This prevents our data from being directly comparable to PNCs measured elsewhere or with different CPC models. However, comparison with other locations or environments was not the principal purpose of our research. Our purpose was to make comparisons between identical instruments in the same microenvironment in the same city and to provide informative rather than definitive data.

Location was continuously recorded using a GPS receiver, providing speed also.

D.2 Introduction and study objectives

A pilot study was funded by NIWA and conducted on 15, 21 and 28 November 2007. The study consisted of simultaneous measurements of PNCs inside and outside a 2006 Honda Civic IMA 13.L CVT on several routes in central Auckland.

The objectives were to:

- gather some indicative PNCs both in and outside a vehicle during typical traffic and driving conditions
- primarily gather data describing a typical ventilation setting, but also to gather indicative data for common alternative ventilation settings
- cover a range of road types from quiet minor suburban roads to busy motorways, while also representing typical journeys.

D.3 Study design

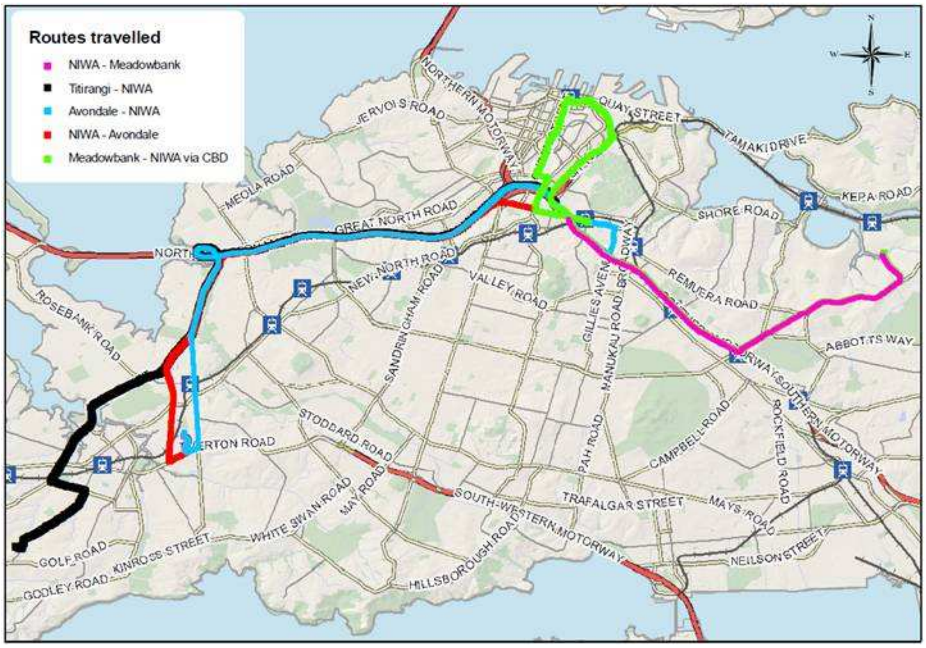
PNCs were measured using two TSI 'P-Trak' Model 8525 portable CPCs. The instruments were placed on the back seat of the car. One instrument drew air from the exterior of the car via a 1/8in diameter copper inlet which was passed through a small gap at the top of the rear window. The instruments were logged at their fastest available resolution (1 second).

D.4 Routes

The routes selected are illustrated in figure D.1. The routes included parts of the Southern and North-Western Motorways, the Central Motorway Junction, a major arterial road (Great North Road/Ash

Street/Rata Street), less busy main roads (Greenlane East, Remuera Road, city centre streets and minor suburban roads.

Figure D.1 Routes travelled during the Auckland pilot study, at which time the NIWA base was in Newmarket, close to the Central Motorway Junction. Black = Titirangi - NIWA, Red = NIWA – Avondale, Blue = Avondale – NIWA, Purple = NIWA – Meadowbank, Green = loop through CBD



The tests were conducted as detailed in table D.1. As a whole the tests constituted 3 hours and 45 minutes of driving, of which 2 hours and 11 minutes were with 'normal' ventilation (air conditioning on, all windows closed, fan on lowest non-zero speed).

Table D.1 Summary of pilot study trips

Date	Trip #	Route	Ventilation	Duration/minutes
15/11/07	1	Titirangi – NIWA	normal	18
21/11/07	2	NIWA – Avondale	normal	14
21/11/07	3	Avondale – NIWA	normal	30
21/11/07	4	NIWA – Meadowbank	normal	17
21/11/07	5	Meadowbank – NIWA via CBD	normal	39
28/11/07	6	NIWA – Avondale	normal	13
28/11/07	7	Avondale – NIWA	high fan	17
28/11/07	8	NIWA – Avondale	window open	15
28/11/07	9	Avondale – NIWA	recirc, a/c on	19
28/11/07	10	NIWA – Meadowbank	window open	12
28/11/07	11	Meadowbank – NIWA	mid fan	10
28/11/07	12	NIWA – Meadowbank	recirc, a/c on	8
28/11/07	13	Meadowbank – NIWA	a/c & fan off	9

D.5 References

Matson, U, LE Ekberg and A Afshari (2004) Measurement of ultrafine particles: a comparison of two handheld condensation particle counters. *Aerosol Science and Technology 38*:487–495.

Zhu, Y, N Yu, T Kuhn and WC Hinds (2006) Field comparison of P-Trak and condensation particle counters. *Aerosol Science and Technology 40*:422–430.

Appendix E: 'Normal driving' observational study

E.1 Note on instruments used

In all of the observational studies described in this report, particle number concentrations (PNCs) were measured using one, or two TSI 'P-Trak' Model 8525 portable condensation particle counters (CPCs) (one of which was kindly loaned by Auckland Regional Council). The manufacturers claim the instrument responds to particles in the size range 20nm–1μm in diameter. The limitations of this instrument are discussed in appendix D.

Location was continuously recorded using a GPS receiver, which provided speed also.

E.2 8 January 2009 observational dataset

E.2.1 Introduction and study objectives

A short study was conducted on 8 January 2009 between 10am and 4pm. The study consisted of measurements of PNCs in a 2001 Toyota Camry 2.2GL wagon, on several routes in central Auckland.

The primary objective was to conduct instrument intercomparison tests when exposed to traffic emissions (ie for purposes other than this project). However, this test also gave us an opportunity to gather additional external concentration data. We therefore re-covered some of the same roads used in the pilot study and also extended the coverage to additional roads.

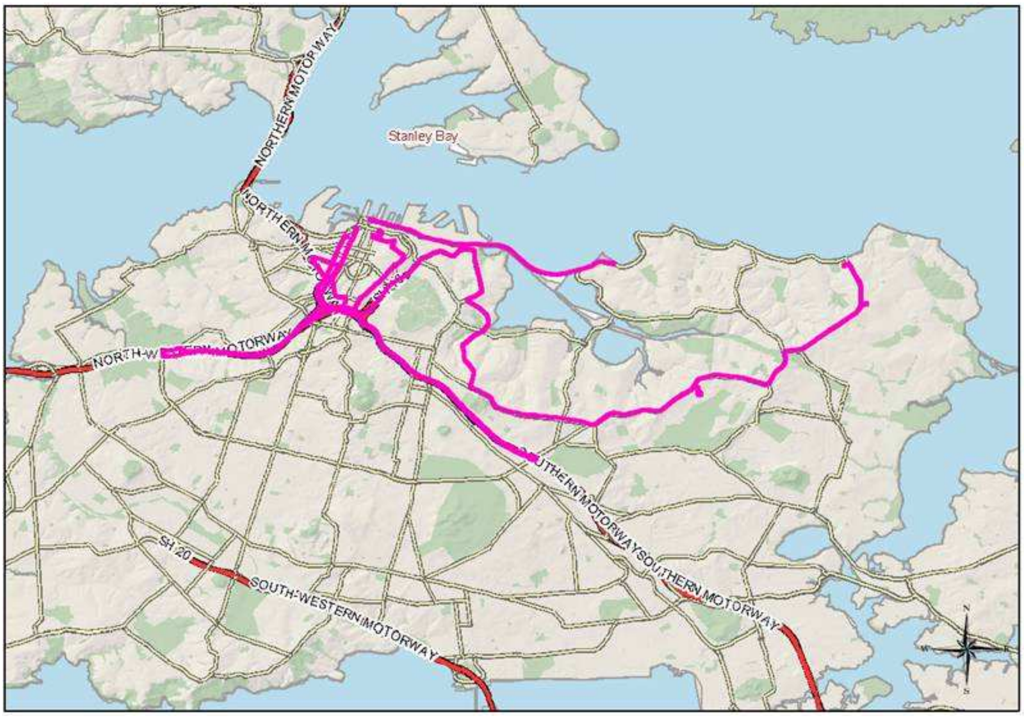
E.2.2 Study design

In contrast to the pilot study, instruments were all placed on the passenger seat and all windows were fully open. Previous data (appendix D) indicated this condition led to a PNC indoor/outdoor ratio of ~1, resulting in internal concentrations ≈ external concentrations. No inlet was used. The instruments were logged at their fastest available resolution (1 second).

E.2.3 Routes

The routes selected are illustrated in figure E.1. They included parts of the Southern and North-Western Motorways, the Central Motorway Junction and several roads in the CBD (including Symonds Street, Nelson Street and Quay Street). Previously un-surveyed roads included St Heliers Bay Road and Tamaki Drive and a low-traffic route through Parnell.

Figure E.1 Routes taken (pink lines) during the 8 January 2009 study



The tests were conducted as detailed in table E.1. As a whole the tests constituted two hours and three minutes of driving.

Table E.1 Summary of trips taken on 8 January 2009

Trip #	Route	Via	Duration /minutes
1	CBD – Meadowbank	Quay Str, Parnell, Remuera Rd	20
2	Meadowbank – St Heliers	St Heliers Rd	18
3	Okahu Bay – CBD	Tamaki Drive, SH16, Nelson Str	15
4	CBD – CBD	CBD loop	20
5	CBD – CBD	CBD, then SH1 to Greenlane and return	33
6	CBD – CBD	SH1 to Greenlane and& return	17

E.3 Mairangi Bay (13 January 2009) observational dataset

E.3.1 Introduction and study objectives

A short study was conducted on 13 January 2009 between noon and 2pm. The study consisted of measurements of PNCs in the same Honda Civic used for the pilot study, between Mairangi Bay (North Shore) and Auckland CBD.

The objectives were to:

- gather additional indicative PNCs both in and outside a vehicle during typical traffic and driving conditions on a previously unsurveyed route
- gather data describing a typical ventilation setting only

- compare two routes linking the same origin and destination – one with relatively higher traffic than the other – in the same traffic and meteorological conditions.

E.3.2 Study design

The instruments were set up in an identical fashion to the pilot study (see above).

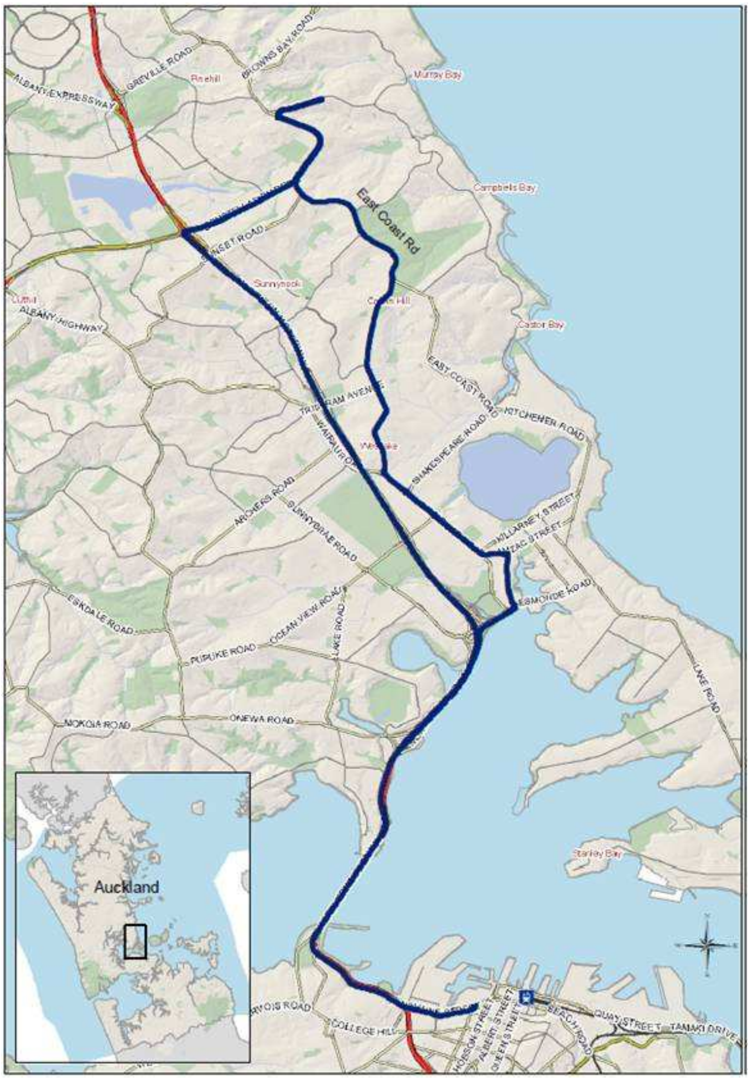
E.3.3 Routes

The route driven comprised three trips between Sunrise Avenue (adjacent to Wisteria Way) in Mairangi Bay, North Shore, Auckland, to the NIWA office on Market Place in Auckland's CBD, as described in table E.2 and figure E.2. The traffic contrast was provided by trip 2 travelling via the Northern Motorway and trips 1 and 3 via East Coast Road (although these routes require use of the motorway to cross the Waitemata Harbour).

Table E.2 Summary of trips undertaken on 13 January 2009

Trip #	From	To	Via
1	Mairangi Bay	CBD	East Coast Road
2	CBD	Mairangi Bay	Northern Motorway
3	Mairangi Bay	CBD	East Coast Road

Figure E.2 Routes travelled on 13 January 2009



Appendix F: Air and particles exchange model (APEX)

F.1 A brief explanation and derivation of the model

The APEX model is intended to address the role the physical construction and design of a vehicle, and the way it is ventilated, play in the in/out exchange of particles and hence in-vehicle exposure. It is a mechanistic model which is parameterised with observational data. It can then be used to investigate the implications of alternative air exchange characteristics on hypothetical exposures.

The APEX model as described in this report is an initial formulation. It assumes the only process contributing to changes in the internal concentration is infiltration and exfiltration of particles in the air which enters and leaves the car. Other factors, such as particle filtration by the air conditioning system, particle deposition onto internal surfaces, and respiration into the lungs are assumed to have negligible effect. At the time of the research, APEX was a ‘research’ model that could not be applied to operational uses. However, this project provided a solid foundation and rich dataset from which the model could be further developed towards an operational model which is more fully inclusive of the processes involved, and which is informative for transport and land-use planning, and public health risk assessment and research.

The APEX model is designed to predict near-instantaneous in-cabin concentrations of an air pollutant on a second-by-second basis (or a few seconds), if the external concentrations are known at the same time-scale. The results provided are particle number concentrations (PNCs). It is essentially an air-exchange model. In principle, the model should apply to any air pollutants for which there are no sources inside the vehicle (and indoor sources could potentially be added to the model in the future).

Input information required by the model is as follows:

- external PNCs (ie in the air immediately ahead of the car) for at least 10 seconds in advance
- the air exchange rate (AER) of the car.

The AER is itself dependent upon:

- ventilation settings (windows, vents and fans)
- speed (including wind speed)
- the car itself (its internal volume and ‘leakiness’).

AER values are highly variable. For any given car, literature values can be assumed to apply (see appendix C), or, alternatively, a ‘rough cut’ AER estimate can be experimentally established based on modelling assumptions (see appendix E).

For the APEX model, the features of the pilot study observational data led us to adopt a simple first-order mass-balance approach, where the net exchange of pollutant between the exterior and interior was assumed to occur at a rate directly proportional to the concentration difference. This simple formulation was ‘blind’ to the actual exchange mechanism, the net effects of which were parameterised. Furthermore, this formulation assumed particle removal processes (such as filtration by the air conditioning system, deposition onto internal surfaces and respiration into the lungs) had negligible effect.

The model derivation below considers the ‘pollutant’ to be particles, described by the PNC, and makes the following key assumptions:

- The PNC is uniform throughout the car at all times.

- Particles which enter the car mix instantly throughout the cabin's interior.
- The particles do not influence each other (eg there is no coagulation).
- Infiltration and exfiltration are described by the same AER.

The number concentration of particles in the car is denoted N_{in} (cm^{-3}). Assuming the volume of air in the car is V (cm^{-3}), and the particles are uniformly distributed throughout the car, the total number of particles in the car is *NV*.

Since the volume of the car is constant, the rate of change in the total number of particles with respect to time is

$$V\frac{dNin}{dt} \quad \text{(Equation F.1)}$$

This is assumed to equal the rate of particle infiltration from the exterior, minus the rate of particle exfiltration from the interior, ie

$$V\frac{dNin}{dt} = q\left(Nout - Nin\right) \quad \text{(Equation F.2)}$$

where q is the flow rate of air between the interior and exterior of the car ($cm^3\ s^{-1}$) and N_{out} is the exterior PNC (cm^{-3}).

Dividing through by V gives

$$\frac{dNin}{dt} = \lambda\left(Nout - Nin\right) \quad \text{(Equation F.3)}$$

where $\lambda = q/V$ is the AER (s^{-1}). This is the initial formulation used for the APEX model.

Appendix C summarises previous research on the AERs of vehicles, and appendix E describes in detail how we empirically derived estimates of the AERs for two vehicles from our observational data.

F.2 Application of APEX to a sample of pilot study data

Figure F.1 shows a sample of the pilot study data. This figure illustrates PNCs observed both inside and outside a Honda Civic travelling between Titirangi in west Auckland and Newmarket. The noticeable step increase in both concentrations approximately one-third of the way into the journey represents a transition from minor to major roads.

The APEX model was applied to this data in a spreadsheet, implemented as a finite-step iterative solution. The observational data was available at a time resolution of one second, and so the model was iterated in one-second steps also. Figure F.2 shows the same observed interior data (with the exterior data removed for clarity) and the modelled interior data using APEX. Figure F.3 plots the modelled versus observed interior PNCs for this journey.

Figure F.1 Exterior (grey) and interior (black) PNCs from pilot study trip 1 (Titirangi to Newmarket). The step increase one third of the way into the journey represents a transition from low-traffic to high-traffic roads

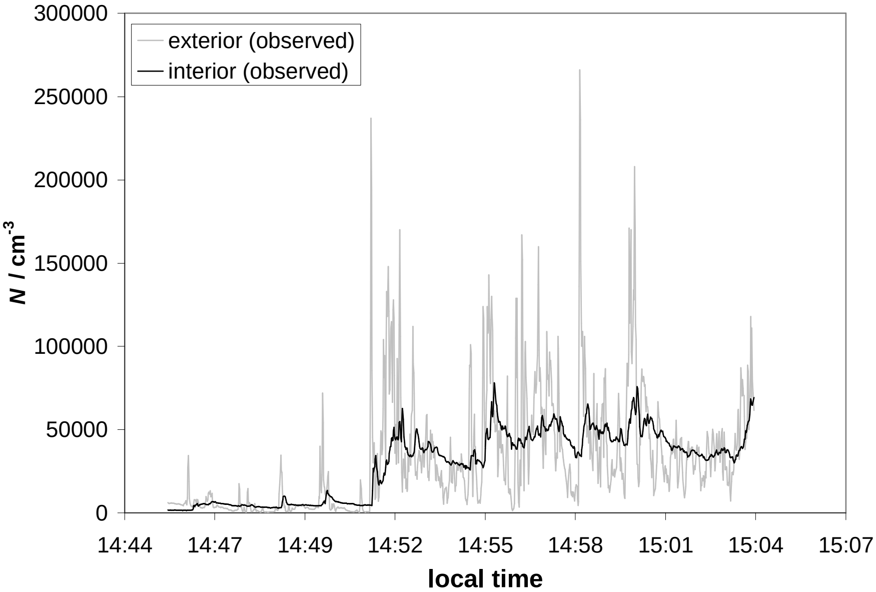


Figure F.2 Interior PNCs from pilot study trip 1 (Titirangi to Newmarket) showing observed concentrations and modelled (using APEX) concentrations

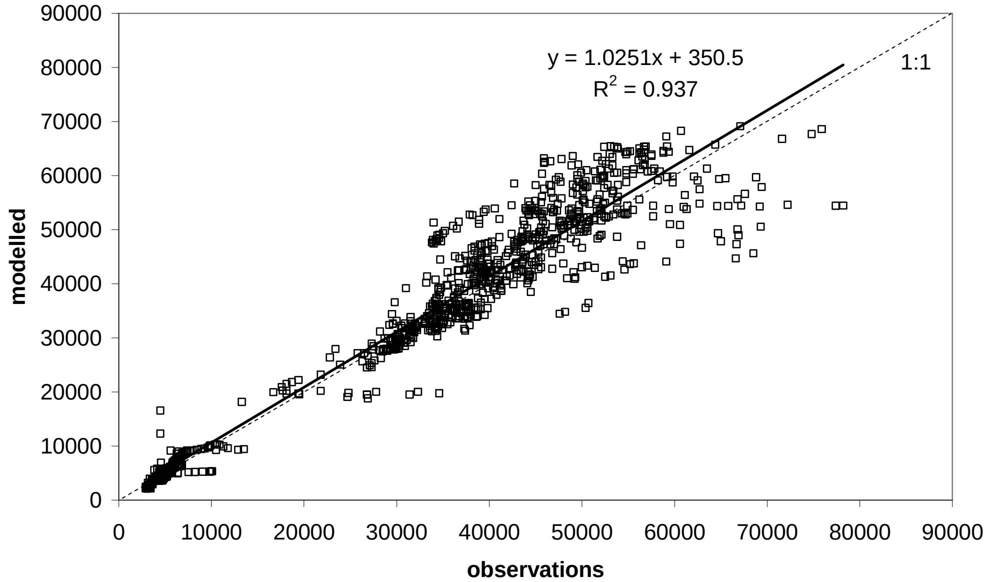
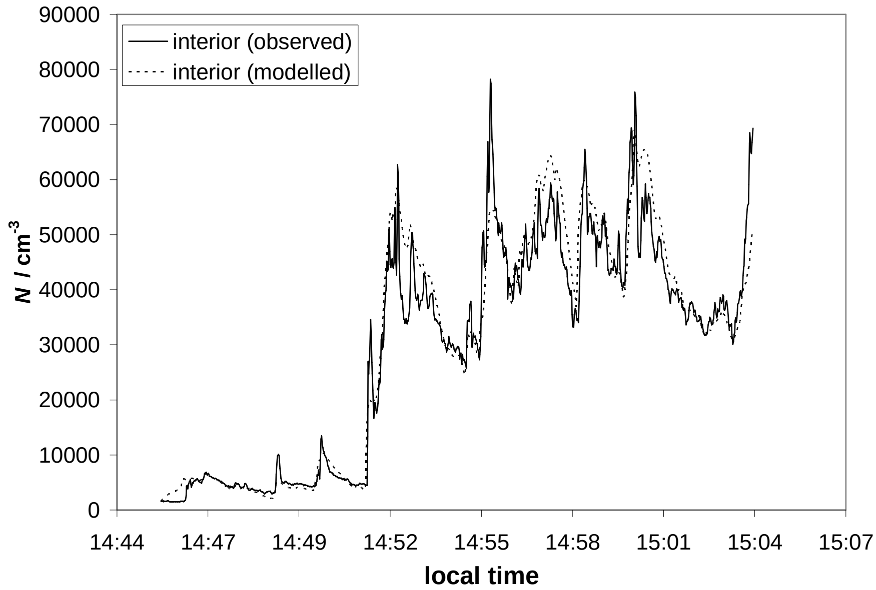


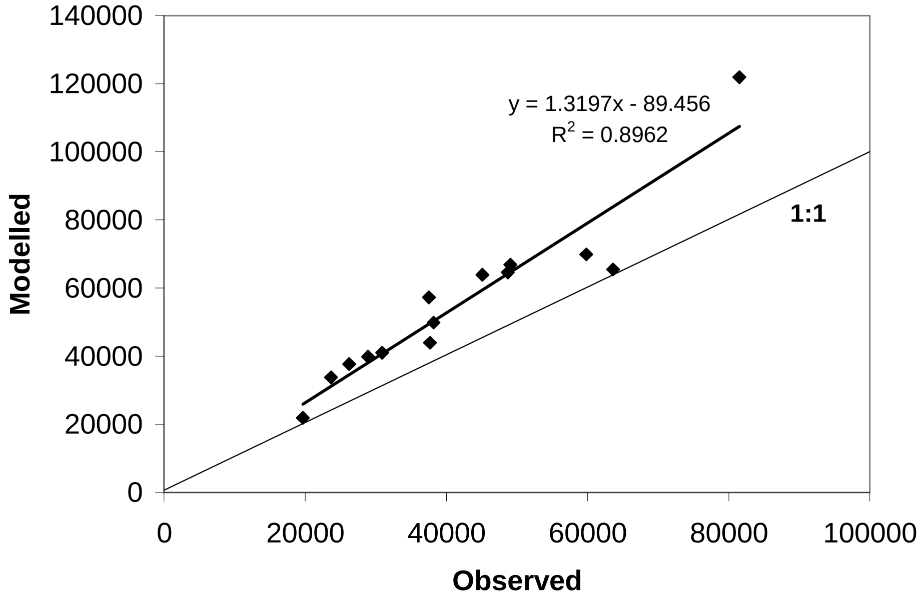
Figure F.3 Plot of modelled (APEX) versus observed interior PNCs from pilot study trip 1



The model appears to have performed well on this dataset. The model performed somewhat less well in certain cases, however, with the model over-estimating internal concentrations. This is illustrated in figure F.4, which shows the mean interior concentrations (both modelled and observed) for 14 trips conducted on 21 Nov 2007.

On average the model is shown to over-estimate internal concentrations by 32%. However, it is clear the correlation is strong. The main purpose of the model is to investigate comparisons between actual or hypothetical trips, and for this purpose the strength of the correlation is the most important criterion of model success. The consequences of the over-estimation are that the model output should only be used to compare with itself, and not be used to make absolute comparisons with observational data (without correction). Further inspection of our dataset should inform whether there is any systematic bias in the over-estimation (eg related to certain kinds of route, traffic, driving or ventilation conditions). If any such bias is identified then the underlying assumptions of the APEX model should be re-evaluated with reference to the high-resolution data captured in this project. It may be possible to further develop the APEX model to explain and account for any bias.

Figure F.4 Modelled (APEX) and observed trip-average (mean) internal PNCs for 14 trips conducted on 21 November 2007 (1:1 line also shown)



Appendix G: Empirical investigation of car air exchange rates

G.1 Parameterising the air exchange rate using APEX and the pilot study data

The air exchange rate (or AER) is a model parameter which is likely to be related to vehicle speed, ventilation settings, and the inherent 'leakiness' of the vehicle (and therefore a property of any individual vehicle and the way it is used – see appendix C for a review of previous studies).

For the purpose of developing the APEX model we used observational pilot study data to parameterise the AER as a function of vehicle speed for three different speeds and four ventilation settings. During the pilot study a range of vehicle speeds was observed, but extended periods of relatively constant speed tended to occur in the following three cases:

1 'Cruising' on the motorway at an average of 95–100km/h

2 Steady travel on arterial roads at an average of 50–55km/h

3 Stationary (at intersections and pedestrian crossings).

Data was extracted for continuous periods when vehicle speeds were in each of these three bands, assuming the AER would be constant in each band. We then used the APEX model formulation (appendix F) to iteratively adjust a value of the AER to obtain a best fit between modelled and observed internal concentrations during that period.

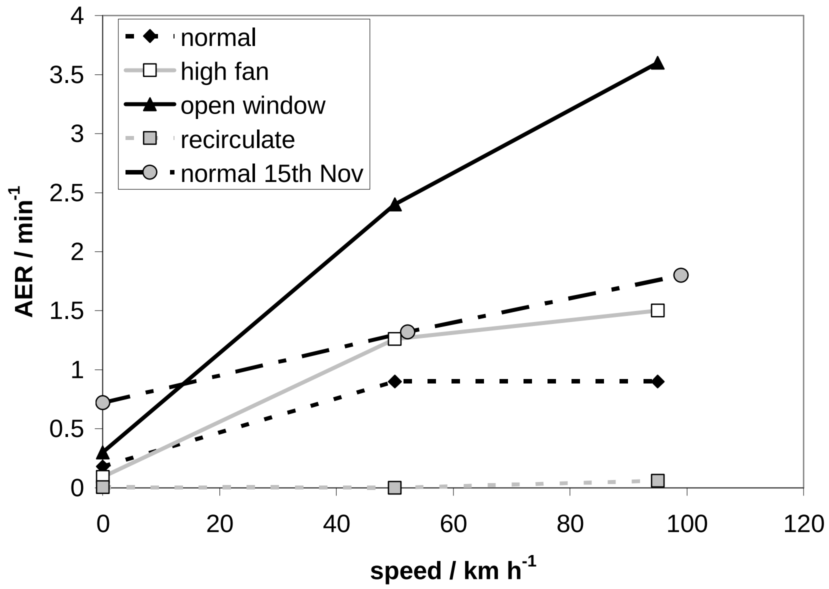
Pilot study data was available from 15 and 28 November 2007 for a 2006 Honda Civic 1.3L CVT. 'Normal' ventilation settings were used on the 15th and four different ventilation settings were used on the 28th:

- a single fully-open window
- high fan (windows closed)
- normal (windows closed, low fan setting)
- recirculation.

Data from 21 November 2007 proved particularly complex and relatively resistant to the satisfactory estimation of the AER (which requires extended periods of near-constant speed) and was therefore not used.

Figure G.1 shows the AER estimates derived for each setting in the three selected speed bands.

Figure G.1 AERs estimated for 3 vehicle speeds and 4 ventilation settings from pilot study data from 28 November 2007 (and one set from 15 November 2007). The data refers to a Honda Civic



This method requires the vehicle speed to be relatively constant for approximately 30 seconds or more, which is not always the case. We found it relatively easy to extract such periods in our data for high speeds (motorway cruising) and for zero speed (at intersections), but it was much harder to achieve for intermediate speeds. Furthermore, the success of the technique is dependent upon the assumption that the AER is constant in each speed band being true. Closer inspection of the data has suggested this assumption is in fact not true. Furthermore, the estimates obtained by using APEX are based on the modelling assumptions (ie other removal processes such as deposition have no effect). Realistically, however, the observational data will be influenced by these other removal processes, and this will in turn influence the AER estimate. Further study, and consideration of the implications of these factors, will be conducted outside of the scope of this project.

As this method can lead to significant errors, we do not have sufficient confidence to quote figures to decimal places. However, in general, normal driving conditions led to an estimated AER value in the region of approximately 1 min^{-1}. Air recirculation led to much lower air exchange rates at all speeds, whereas opening the window increased the AER by a degree related to vehicle speed (higher speed leads to more turbulent air exchange through the open window). The method adopted does not permit us to quantify uncertainties around these values. However, qualitatively we expect uncertainties to be larger as the AER decreases, as the quality of empirical fit will be weaker as the amount of air exchange diminishes. Therefore, we are less confident about our results in the ‘recirculate’ mode.

G.2 Semi-controlled air exchange rate study

G.2.1 Introduction and objectives

A subsequent series of tests were conducted to investigate car AERs under semi-controlled conditions, and the variables (speed, ventilation, cabin volume) influencing these rates. These tests were conducted on 13 January 2009 using a 2006 1.3L CVT Honda Civic Hybrid, and on 15 January 2009 using a 2004 Ford Falcon Futura station wagon.

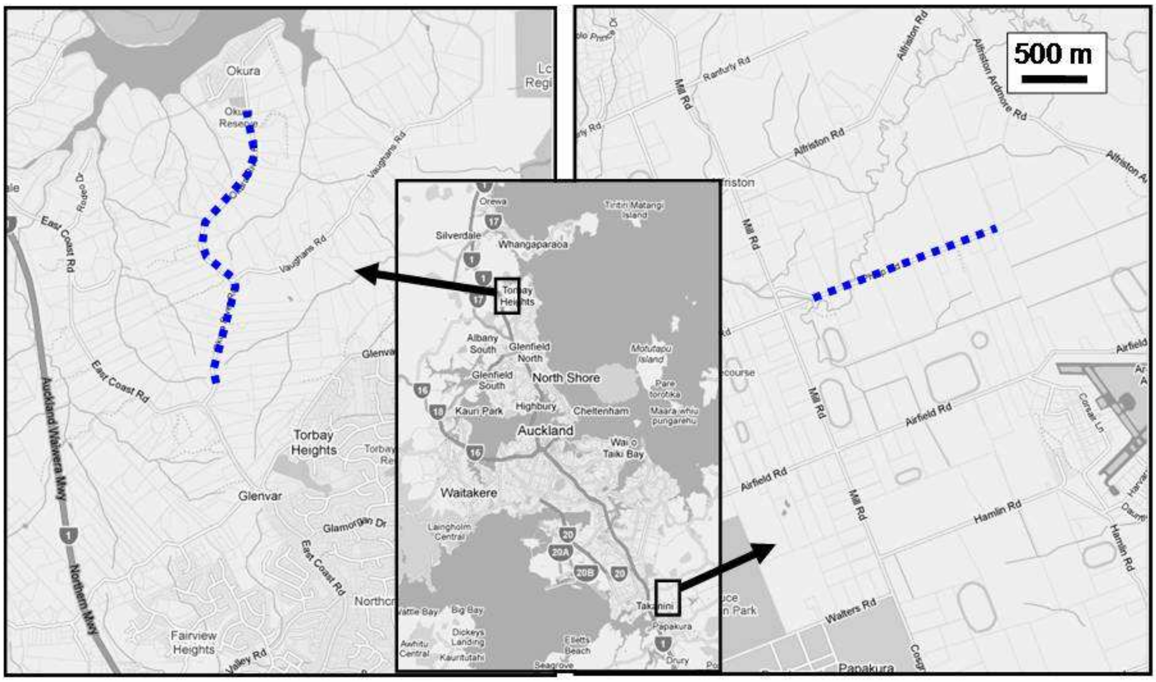
G.2.2 Study design and instrumental set-up

The general concept for this study was to temporarily introduce a 'tracer' smoke into the vehicle cabin and then observe the particle number concentration (PNC) decay as the smoke was removed. Ventilation conditions and vehicle speed were systematically changed to investigate the influence on the decay.

To minimise external influences two test roads were selected with minimal traffic (dead ends) and minimal curves. This minimised the chances of encountering other traffic, thus reducing the impact of uncontrolled external sources, and made it easier to achieve and maintain a constant speed. Based on pilot study data it was considered each test should ideally have a duration of one to two minutes, implying a section of road of a minimum length of 1km at 60km/h. Given these criteria it was not possible to find an appropriate stretch of road for which tests could be conducted at 100km/h. Consequently, test speeds of 30 and 60km/h were selected.

The roads chosen were Okura River Road to the north of Albany and Philip Road in Alfriston, north of Papakura, shown in figure G.2.

Figure G.2 Test routes for the semi-controlled ventilation tests. Left: Okura River Road. Right: Philip Road



The study assumptions included:

- The AERs derived from particles being removed from the cabin are similar to those describing particles entering the cabin from outside.
- The decay in internal concentrations is dominated by air exchange rather than deposition or coagulation.

Air (and hence PNCs) were reasonably well mixed in the vehicle cabin so our measurement point was not influencing our comparative results.

Measurements were made with the same pair of TSI Model 8525 P-Trak portable condensation particle counters used for all the observational studies mentioned in this report. One instrument sampled external air through a copper inlet passed through a gap in the rear passenger door. The other instrument was placed in an elevated position on the rear passenger seat.

G.2.3 Procedure

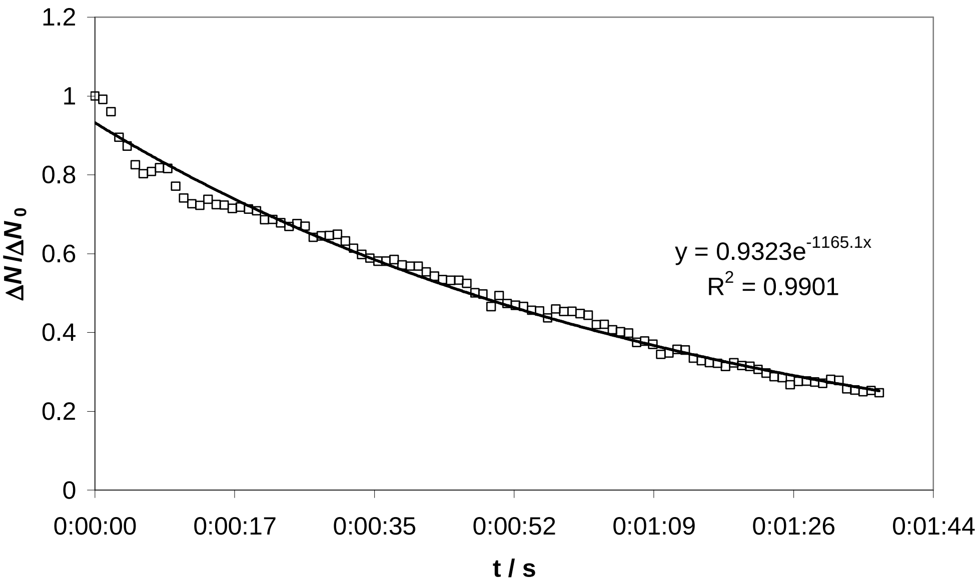
At the start of each test, smoke was introduced into the vehicle interior for 10 seconds. The car was then accelerated to the test speed and driven to the end of the test road. At this point the car doors were opened to fully remove any remaining smoke. The next test was not started until the internal and external concentrations had equalised.

Analysis consisted of fitting an exponential decay curve to the time series of the difference in concentration (ie internal – external, denoted ΔN) normalised by the initial concentration difference ΔN_0, where $t = 0$ is the visually determined start of the decay. The fit is described by the equation

$$\left(\frac{\Delta N}{\Delta N0}\right) = e^{\lambda t} \qquad \text{(Equation G.1)}$$

where λ is the AER. One such fit is illustrated in figure G.3.

Figure G.3 Experimental data of normalised PNC difference (internal – external) as a function of time, following an injection of smoke particles into the vehicle cabin, with exponential decay fit. This example is test 12 on the Ford Falcon (a/c on, 60km/h)



The test ventilations settings were slightly different for the two cars (see tables G.1 and G.2). This is because the Civic had an integrated air conditioning and fan control, whereas the Falcon had a single air conditioning switch (on or off) and an independent fan speed control.

G.2.4 Test results

Table G.1 List of test settings and results for the Honda Civic (13 January)

Test #	Speed/km/h	Vents	a/c	Fan	Windows	AER/min^{-1}
1	30	Open	Off	Off	Fully open	11.2
2	30	Open	Off	Off	Open 3cm	0.63
3	30	Open	On	Min	closed	1.36
4	60	Open	Off	Off	Fully open	13.0
5	60	Open	Off	Off	Open 3cm	1.44
6	60	Open	On	Min	closed	1.55
7	30	Open	Off	Off	closed	1.14
8	30	Open	On	Full	closed	2.86
9	60	Open	Off	Off	closed	1.10
10	60	Open	On	Full	closed	2.52
11	30	Recirculate	Off	Off	Closed	0.92
12	30	Recirculate	On	Min	Closed	0.30
13	60	Recirculate	Off	Off	Closed	0.64
14	60	Recirculate	On	Min	Closed	0.44

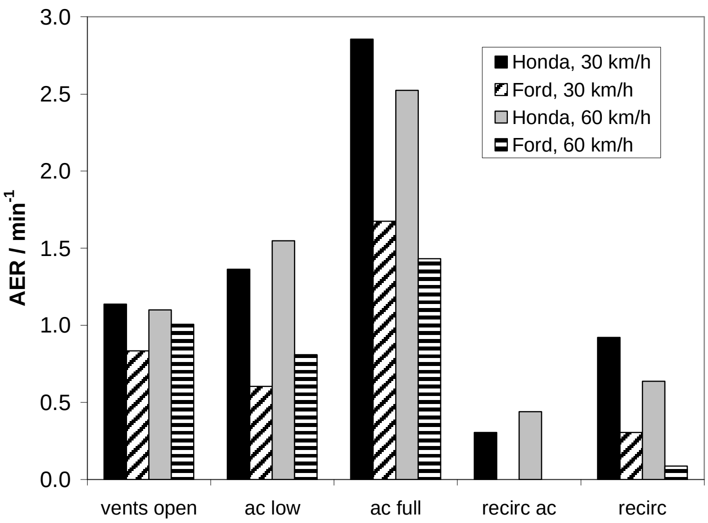
Table G.2 List of test settings and results for the Ford Falcon (15 January)

Test #	Speed-km/h^{-1}	Vents	a/c	Fan	AER/min^{-1}
1	30	Open	Off	Low	0.83
2	60	Open	Off	Low	1.01
3	30	Recirculate	Off	Low	0.30
4	60	Recirculate	Off	Low	0.09
5	30	Recirculate	Off	Half	0.28
6	60	Recirculate	Off	Half	0.88
7	30	Recirculate	Off	Full	0.43
8	60	Recirculate	Off	Full	0.65
9	30	Open	Off	Half	0.78
10	60	Open	Off	Half	1.12
11	30	Open	On	Low	0.60
12	60	Open	On	Low	0.81
13	30	Open	On	Half	0.86
14	60	Open	On	Half	1.13
15	30	Open	On	Full	1.68
16	60	Open	On	Full	1.43

G.2.5 Results – general patterns

We have tried to summarise the role of each of the parameters on the estimated AER from the data, partly illustrated in figure G.4. However, low AERs are subject to greater uncertainty due to the lower rate of decay. We also noticed low AER data was further subject to increased scatter and random variation, presumably due to reduced air mixing in the vehicle cabin.

Figure G.4 Dependence of estimated AER on ventilation setting (the Ford Falcon used did not allow air conditioning in recirculating mode)



Effect of window opening:

- In the Honda, fully opening windows increased the AER by approximately an order of magnitude, compared with fully closed (with the air conditioning on).
- With this limited dataset we were unable to consistently determine the effect of opening the window by 3cm.

Effect of speed:

- In the Ford, the AER was 20%–45% higher (33% on average) at 60km/h compared with 30km/h, except when the vents were set to recirculate at low fan speeds, when the AER was two to three times higher at 60km/h.
- In the Honda the data was more complex. In typical driving (windows closed, vents open) speed had a small and varied effect on the AER. It was reduced by 30% at 60km/h compared with 30km/h when air was recirculating and the air conditioning off.

Effect of air conditioning:

- The results were complex and deserve further investigation. In general, results suggested the air conditioning increased the AER in both vehicles by 10%–40% (the change being greater at 60km/h relative to 30km/h). However, in recirculation mode, air conditioning appeared to reduce the AER by one to two thirds in the Honda.

Effect of fan speed:

- Increasing fan speed from low to maximum led to a two to three times increase in the AER in the Ford and 1.5 to 3 times in the Honda (depending on vehicle speed).

Effect of recirculation:

- Recirculation decreased the AER by 20%–40% in the Honda when the fan was off, and 70%–80% when the fan was running on low speed. In the Ford the reduction was 20% (half fan, 60km/h), 65% (half fan, 30km/h), 70% (low fan, 60km/h) and 90% (low fan, 30km/h).

Effect of vehicle:

- AERs measured for the Ford Falcon were approximately half the nearest equivalent values for the Honda Civic. The internal volume of the Ford was approximately double that of the Honda.

G.2.6 Summary

We broadly found that the following variables had an influence on the AER, roughly in order of descending significance:

- windows open/closed
- ventilation fan speed
- vehicle (ie cabin volume, and/or inherent degree of air tightness)
- vents open or closed (ie air recirculating)
- vehicle speed
- air conditioning.

G.3 Implications for using and developing APEX

Our APEX model requires a formulation of the AER to predict internal concentrations. In developing APEX we have assumed the AER is a function of vehicle speed, based on our literature review. The research reported here in this appendix generally supports this assumption, although it is difficult to quantify the relationship with any certainty due to our limited datasets and inherent uncertainties in the methods.

Within APEX we suggest two alternative formulations for the AER which should be compared in terms of inherent uncertainties and fitness for purpose.

Our initial formulation is to assume a relationship with speed of the form

$$\lambda = Av + B \qquad \text{(Equation G.2)}$$

where the parameters A and B are constants which depend on the characteristics of the vehicle (although they may also be influenced by wind speed and direction – see appendix C). Estimates from 15 November 2007 indicate values of A of 0.0007 and B of 0.012. Data from 28 November 2007 indicates a non-linear relationship. However, if a linear bit is forced A is 0.0005 and B is 0.0047.

Given the variation in these values, an alternative formulation is to assume a fixed, speed-invariant value for the AER. In normal driving conditions (windows closed, vents open, fan speed low) the AERs estimated in the semi-controlled test varied between 0.6 and 1.6 min^{-1} in the range 30 to 60km/h. Estimates derived from our 'normal driving' observational campaign for similar ventilation conditions varied from 0.8 to 1.8 min^{-1} between 50 and 100km/h. As a representative estimate we propose that an AER of 1 min^{-1} for a car of volume and age similar to the Honda Civic in our tests at typical urban driving speeds is appropriate.

Appendix H: Technical recommendations for further development of APEX

H.1 APEX over-estimation

Appendix F describes how the APEX model was found prone to over-estimation of in-vehicle concentrations. This over-estimation was generally considered indicative of the lack of particle removal (such as filtration by the air conditioning system, deposition on internal surfaces, and respiration into the lungs) in the model formulation. Model errors were regularly triggered by rapid falls in external concentrations leading to a corresponding fall in internal concentrations that the model was under-estimating. This error remained uncorrected and was ‘carried through’ the simulation. This implies an un-modelled process removing particles from the cabin, which becomes relatively more significant when concentrations are falling. As this phenomenon was associated with large falls in external concentrations, we hypothesise it was related to the injection of a large number of fresh exhaust particles due to a close encounter with a gross emitting vehicle. We further hypothesise this could lead to a momentary increase in particle deposition. For instance, fresh exhaust emissions are likely to be dominated in number by ultra-fine particles, and these small particles have increased depositional speeds due to Brownian diffusion. Furthermore, ultra-fine particles are known to deposit more efficiently in the lungs.

A further failing appears to be that the model reacts too fast to increases in the external concentration, ie the model predicts a single time-step response (within one second) to any external increase. The observational data, however, indicated a time delay in the internal response to external changes. This suggests the model needs to include a time lag, at least if being used at a one-second resolution.

H.2 Improved model formulation

Particle removal processes which are not included in the current formulation of the model, but which we consider may be significant, include particle filtration by the air conditioning system, and particle deposition (which we consider to include deposition onto internal surfaces and in the lungs of occupants). A possible alternative model formulation for APEX which includes these processes would be

$$\frac{dNin}{dt} = \lambda\left(1-p\right)Nout - \left(\lambda+k\right)Nin \qquad \text{(Equation H.1)}$$

where λ is the AER, *p* is the fraction of particles removed by the air conditioning system (which may be a function of particle size), and *k* is the fraction of particles which are removed by deposition per unit time (which will also be a function of particle size).

Given this formulation it would also be helpful to have some information on particle size in the measurement of PNCs. If there is no practical method of measuring particle size, then recording the particle mass concentration as well as the corresponding particle number concentration with the P-Traks will at least give some indication.