

Detailed observations and validated modelling of the impact of traffic on the air quality of roadside communities

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Executive summary

The research described in this report was designed to serve two key and related needs:

- 1 To assist in the enhancement and development of modelling tools to assess the local air quality impacts of road transport emissions
- 2 To provide data and models that will enable the development of New Zealand-based estimates of health effects of those impacts on local air quality.

Recent international research shows that the air quality impact of major roads is significant up to a 250m distance and possibly further. Associated research reveals that a substantial health burden arises from the long-term chronic exposure to emissions in roadside communities, which is not represented by the national environmental standards for air quality (NES), and ignoring this burden, as is currently the norm, leads to an undervaluation of the resulting health-care costs attributable to traffic (Künzli et al 2008). A rational judgment of the cost-benefit of mitigation of traffic emission exposure requires that its full impact be quantified. However, before this project was carried out there was no New Zealand-based data while overseas data describes different emissions, climate and urban fabric.

The objectives of this project were:

- 1 To conduct highly detailed observations of air quality in a typical New Zealand roadside residential community.
- 2 To generate a dataset suitable for the validation of roadside dispersion models, and use of that dataset to provide a validation of 1) the vehicle emission prediction model (VEPM) + Ausroads combination, 2) the National Institute of Water & Atmospheric Research (NIWA)/NZ Transport Agency (NZTA) roadside corridor model, 3) the semi-empirical site-optimised (SOSE) model, and 4) a land-use regression (LUR) model in a motorway setting.
- 3 To compare the performance of these four models and make recommendations on their use.
- 4 To enable and extend general health impact analysis of emissions from particular roads.

We established a unique monitoring design that allowed the contribution of the road to local air quality to be estimated from the difference in concentrations between downwind and upwind sites. Our main study area was Otahuhu East in Auckland, a residential area bisected by the Auckland southern motorway (SH1). At this location the motorway carries 112,000 vehicles per day. This area was chosen for technical and logistical reasons rather than any pre-existing concerns about air quality or health outcomes there.

An observational programme was conducted which comprised of three layers:

- 1 A unique dataset of monthly monitoring of nitrogen dioxide at 32 sites across the study area, and an additional 28 sites in the similar nearby community of Mangere, for a year (2010-11).
- 2 Intensive, partly-simultaneous continuous monitoring of particulate matter, oxides of nitrogen, ozone, carbon monoxide and meteorological parameters at a motorway kerbside site and two sites set back up to 250m either side of the motorway through autumn and winter 2010.
- 3 An additional campaign of 'mobile' particulate matter, carbon monoxide and black carbon using both a car and a bicycle on selected days in winter 2010.

Four modelling evaluation studies were also conducted. These were:

- 1 The first New Zealand-based evaluation of the common regulatory model Ausroads (an atmospheric dispersion model designed for roadside applications) when used with VEPM.
- 2 An in-depth evaluation of the New Zealand-originated empirical SOSE model, and extension of its capabilities to provide a new method for establishing background air quality.
- 3 An extension of the popular land-use regression technique to finer scales than previously attempted.
- 4 An evaluation of the new roadside corridor model developed by NIWA with funding from the NZTA.

Our observational campaigns revealed that air quality in our study area was fully compliant with the NES at all times. The contribution of emissions from vehicles on the motorway to fine particulate matter (PM_{10})¹ was small (10% on average) and PM_{10} levels only became elevated on cold evenings with low winds due to domestic heating emissions. Motorway emissions made a much larger contribution to roadside concentrations of nitrogen dioxide (approximately half) but the margin of compliance with the NES was greater than for PM_{10} . The average NO_2 concentration at the motorway's edge was $24\mu g m^{-3}$, or 60% of the WHO guideline of $40\mu g m^{-3}$. The probability of NES exceedance at roadside locations is dominated by the baseline, ie the concentrations due to all other sources, including surrounding roads, at any given site. The relatively large margin of compliance for NO_2 (World Health Organisation guideline and NES) in our study area was due to the relatively low levels of non-motorway traffic.

Concentrations of traffic-related air pollutants at the kerbside site were approximately double the concentrations at the setback sites. Our observational findings were broadly similar to the findings from international studies, but also indicative of higher emissions rates from the Auckland vehicle fleet than from the US fleet. Our study indicated that variability in the decay rate of pollutant concentrations with distance from a major road seen in international studies may be explained by the previously underestimated significance of emissions from minor roads.

This study has shown how passive monitoring can identify localised variations in concentrations (hot-spots and differences with side of the motorway) that would have remained unknown if only the fixed site continuous monitoring or conventional emission-dispersion modelling had been used.

The modelling studies indicated that the commonly used regulatory dispersion model Ausroads may be conservative. We found that it underestimated dispersion (and hence overestimated concentrations) downwind of our study motorway. This has not previously been reported, but is consistent with the known limitations of the model formulation.

Although this project was not explicitly designed to evaluate the VEPM emission model, indirect evidence suggested that VEPM was performing well in predicting emission rates for oxides of nitrogen. However, we found that the greatest contribution to uncertainty in the regulatory application of these models was likely to come from the selection of data representing background air quality. We found that non-local sites were relatively poor in describing background air quality.

Our research confirmed advice established for non-roadside applications, that dispersion modelling should be based on meteorological data collected from sites that are well-suited to the task of representing air flow over the whole area of interest. In practice this means taller masts (ideally 10m high) are preferred to shorter ones, and masts subject to any form of local sheltering are to be avoided.

The use and applicability of the empirical modelling approaches SOSE and LUR was successfully extended. We showed that SOSE provides a reliable and robust method for predicting temporal patterns in roadside

¹ PM_{10} : particulate matter smaller than $10\mu m$ in diameter

air quality, as well as providing an independent estimate of local background air quality. We showed that LUR models can be used to describe subtle street-by-street variations in air quality at resolutions not attempted before, providing a relatively low-cost means of understanding and assessing the impact of micro-scale infrastructure design on air quality. NIWA's road corridor model (RCM) was shown to be a promising tool, capable of describing the large-scale impact of the motorway on local air quality. However the RCM, and the roadside dispersion model upon which it is based (Ausroads), appeared to slightly exaggerate the width of the corridor within which exposure is increased. This could lead to a small positive bias (overestimate of risk) in health assessments. Technical difficulties associated with its validation indicate that further development is required if this tool is to be used for fine-grained analysis in which accuracy is essential.

This is not a health outcomes study. However, it is designed to enable and extend health impact and epidemiological study of emissions from both specific roads and from road traffic in general. Our study showed that PM_{10} is poorly suited to representing additional risks from air quality within roadside corridors. NO_2 appears to be a much more practical and sensitive measure.

Previous assessments used estimates of NO_2 concentrations at census area unit level to investigate the relationship between traffic pollution and mortality in Auckland (Scoggins et al 2004). That methodology did not include localised roadside corridor effects, which our study showed to be substantial. There is great potential for the health burden associated with traffic pollution to be substantially underpredicted because of this, a point mentioned in the updated HAPINZ report (Kuschel et al 2012). This raises the possibility that previously derived exposure-response relationships for both PM_{10} and NO_2 may not be suitable within the roadside corridor.

In general terms we have concluded that:

- roadside corridor impacts in Auckland appear to be highly localised, but not trivial
- PM_{10} is a poor (insensitive) measure of the potential health impact of vehicle emissions on air quality
- assessments in urban areas should not rely on considering a single dominant road only, but also the local network of roads
- NO_2 exhibits distinct spatial gradients which could be related to spatial patterns in traffic volumes.

The database generated provides a platform for extended research which could include:

- the reasons for (and hence solution to) the model weaknesses identified
- the use of proxies to understand a wider range of pollutants (eg CO, black carbon and ultrafine particles)
- the development and validation of novel modelling approaches or products.

Abstract

Detailed observations of air quality and local meteorology were conducted on either side of a stretch of the Auckland southern motorway, and in the surrounding residential neighbourhood. The data revealed emissions from motorway traffic contributed, on average, to a 10% elevation in concentrations of particulate matter at a roadside site relative to a setback site (150m away or more) and to a doubling in concentrations of nitrogen dioxide. National environmental standards for air quality were not exceeded, but international health research indicates that the spatial variation in traffic-related air pollutants observed in this study represent a risk that is not currently accounted for in risk assessments in this country.

The observational data captured was used to evaluate four different roadside air quality modelling approaches. The most commonly used model in regulatory context (Ausroads) was found to be conservative, but its effectiveness could be undermined by the use of inappropriate (particularly off-site) estimates of background air quality. Less commonly used assessment methods (passive monitoring, semi-empirical and regression modelling) were shown to offer several advantages for assessment.

1 Introduction

1.1 Project background

This chapter describes the background and origins of this research project, the project objectives and scope and an overview of the layout of this report.

1.1.1 Purpose

The research described in this report was designed to serve two key and related needs:

- 1 To assist in the enhancement and development of modelling tools to assess the impact of road transport emissions on local air quality
- 2 To provide data and models to enable the development of New Zealand-based estimates of the health effects resulting from the impact of road transport emissions on local air quality.

Our aims were to:

- improve confidence in current models and develop alternative models for use in a wider range of situations
- provide both data and models for health effects studies that are currently not possible.

The intended outcome was an improved understanding of the effects of traffic pollution and an ability to mitigate and manage those effects thus reducing the health cost burden to New Zealand.

1.1.2 Policy context

This project was conceived shortly after the publication of the New Zealand Transport Strategy 2008 (NZTS).² One of its objectives was 'ensuring environmental sustainability'. One of the desired outcomes related to this objective was 'Negative impacts of transport are reducing in terms of the human and natural environments'.

Another objective was 'protecting and promoting public health'. Progress towards this objective was to be measured in part by the target: 'Reduce the number of people exposed to health-endangering concentrations of air pollution in locations where the impact of transport emissions is significant'.

This research was designed to assist in both objectives by addressing a major gap in the identification of such locations and quantification of 'health-endangering concentrations'.

Many New Zealand towns and cities have air quality that leads to breaches of the national environmental standards (NES), and concentrations of many air pollutants are generally elevated up to a few hundred metres around major roads. Airsheds with more than 10 exceedences of the NES limit value for PM₁₀ a year must meet three exceedences by 1 September 2016, and one exceedence by 1 September 2020. Airsheds

² The New Zealand Transport Strategy (NZTS) is a non-statutory document released by a previous government that uses a planning horizon to 2040 that has been largely superseded in the short term by subsequent policy decisions. The current government supports the overall intent of the NZTS; however, it is less relevant as a practical guide to how New Zealand is responding to the issues facing the transport sector in the immediate term. Connecting New Zealand (MoT 2011) is a more relevant non-statutory document summarising the government's transport policy and intentions over the next decade and is informed by other government non-statutory documents that have different time horizons, including the National Infrastructure Plan that has a planning horizon of 2030.

with fewer than 10 exceedences a year must meet one exceedence by 1 September 2016. Being able to quantify the contribution from major roads to PM_{10} levels will be of great value to councils now and into the future: in many areas it is likely that domestic emissions will fall while road transport vehicle kilometres travelled rise, further increasing the relative importance of road transport emissions.

1.1.3 Background and research need

The NZTS contained the target: 'Reduce the number of people exposed to health-endangering concentrations of air pollution in locations where the impact of transport emissions is significant'. However, our ability to identify such locations, and quantify concentrations of air pollution and hence the number of people exposed to them is currently quite limited.

Recent international research shows that the air quality impact of major roads is significant up to a 250m distance and possibly further (Zhou and Levy 2007). Negative health outcomes are increasingly being related to a traffic emission source. In New Zealand, exceedences of NES nitrogen dioxide have been recorded alongside major roads in Auckland. Exceedences of the standard for PM_{10} are more frequent and widespread in urban areas throughout New Zealand. This is mainly related to domestic heating emissions, although the contribution of motor vehicle emissions to local PM_{10} air quality is important where background levels are elevated. However, many observed health impacts are associated with chronic long-term exposure at times when standards are not breached.

These observations made overseas describe different emissions, climate and urban fabric. Before this project was carried out there was no New Zealand-based data (and very little worldwide) on several key exhaust pollutants (eg ultrafine particles (UFP), black carbon (BC)). Health impact assessment is currently biased to multi-source pollutant mixes and acute effects of regional air pollution episodes, not the chronic impacts that relate specifically to traffic sources.

Modelling approaches exist to predict some of these impacts, but were either unvalidated in New Zealand or needed to be parameterised for New Zealand application using local data. There was no New Zealand evidence-based advice available to air quality practitioners, and those who hire them, on how to select and use such models and interpret the result. This has led to inconsistencies and uncertainties. This research was designed to provide robustness and scientific rigour to the modelling methodologies, giving a better understanding of their capabilities and limitations. Hopefully, this will help avoid unnecessary argument and possible litigation.

Air quality impacts are generally judged in terms of the likelihood of exceedence of the NES – based on daily or hourly peaks in contaminant concentrations. International research is revealing that the greater health burden due to traffic arises from the long-term chronic exposure to emissions in roadside communities, which is not represented by the NES. Ignoring this burden, as is currently the norm, leads to an undervaluation of the resulting health-care costs attributable to traffic (Künzli et al 2008). A rational judgement of the cost-benefit of mitigation of traffic emission exposure requires that its full impact be quantified.

Several major road projects are currently planned around the country. There is an immediate demand for appropriate and validated tools to reveal how the negative impacts of road traffic and road design can be minimised.

1.1.4 Relationship with similar research

The NZ Transport Agency (NZTA) previously funded a National Institute of Water & Atmospheric Research (NIWA) project which included development of a roadside air quality exposure model (Longley et al 2011). This model was restricted to the primary pollutants: PM_{10} and carbon monoxide. At the end of that project,

the model remained largely unvalidated because of the lack of available observational data in New Zealand suitable for this purpose. The Ministry of Science & Innovation funds a NIWA-led air quality research programme which includes a focus on vehicle emissions and their impacts. Co-funding from that programme has substantially enhanced this project, allowing additional instrumentation and analysis to be introduced.

Our approach is similar and complementary to several research campaigns that have been carried out elsewhere. The present project was designed specifically to provide data that can be compared with this study and will feed back directly into international studies of the health impacts of road traffic.

1.2 Project objectives

The objectives of this project were to:

- conduct highly detailed observations of air quality in a typical New Zealand roadside residential community
- generate a dataset suitable for the validation of roadside dispersion models, and use of that dataset to provide a validation of 1) the vehicle emissions prediction model (VEPM) + Ausroads combination; 2) the NIWA-NZTA roadside corridor model; 3) the semi-empirical site-optimised (SOSE) model and; 4) a land-use regression (LUR) model in a motorway setting
- compare the performance of these four models and make recommendations on their use
- enable and extend general health impact analysis of emissions from particular roads.

1.3 Deliverables

- A range of datasets (meteorological, traffic, emission rates, specification of line source and receptor geometry) used as inputs to the modelling.
- Quality controlled air quality (PM_{10} , CO, NO, NO_2 , NO_x , O_3) and meteorological (wind speed, direction, temperature, relative humidity, solar radiation, rainfall) observations from three fixed sites as hourly averages.
- Complete passive monitoring dataset for nitrogen dioxide.
- Mobile monitoring dataset.
- Initialised and validated LUR model for nitrogen dioxide covering the study area.
- Initialised and validated site-optimised semi-empirical model for PM_{10} , CO and NO_x applicable to the study's fixed monitoring sites.
- A peer-reviewed technical report (appendix A of this report).
- A peer-reviewed 'user's guide' report (the main body of this report).

1.4 Limitations of the project:

- This project was designed to enable direct evaluation of roadside dispersion models. It was **not** designed to enable direct evaluation (or validation) of the models listed below. However, the datasets generated may assist in their evaluation outside the scope of this project:
 - traffic models (eg EMME)

- emission models (eg VEPM)
- non-roadside dispersion models (eg CALPUFF, TAPM)
- artificial meteorological datasets (eg Auckland Regional Council (ARC) meteorological dataset).
- This project did not compare Ausroads to alternative dispersion models (such as ADMS-Roads, Breeze, HI-WAY, etc). However, the datasets generated are designed to enable their inter-comparison outside the scope of this project.
- This project could not enable the direct validation of VEPM. This is because it was not technically feasible to conduct local observations of emissions rates within the resources available. This implies that the sources of errors contributing to emission modelling uncertainty cannot be reliably assessed. This dataset may assist in the partial validation of VEPM outside the scope of this project.
- Assessment of uncertainties in traffic data, and the implications for emission modelling results (such as how fleet composition is estimated) were not conducted within this project.
- LUR models and site-optimised semi-empirical models are inherently local and strictly valid only in the area for which they were validated. Applicability elsewhere cannot be assumed without investigation.
- This project was not intended to directly assess the contribution of non-motorway sources to local air quality.

2 Study design and methods

2.1 Study design

We developed our study design, taking into account end-user needs and by preparing a literature review. Our study area needed to be a suburban location in a mainly or entirely residential area, and contain one of New Zealand's busiest roads. The core of our design was the establishment of three fixed continuous monitoring sites along a trajectory aligned with the predominant wind direction, so that one site was upwind and two were downwind, one at the kerbside and two at setback sites on either side of the road. This enabled us to estimate the contribution of the road to local air quality from the difference in concentrations between downwind and upwind sites.

Relatively few people live as close as 150m to a major road. Beyond this distance, some previous studies have indicated smaller but detectable elevations in air pollutant concentrations, whose significance is also related to the much larger exposed population. To investigate the wider significance of roadside air quality for roadside communities we included monitoring over a wider area as part of our study design. Coverage of this wider area was not practical using continuous monitoring. Instead, we adopted a network of low-cost passive monitoring to provide long-term spatial coverage with low temporal resolution, supported by sporadic mobile monitoring with high temporal resolution.

We included modelling techniques in our project in order to:

- evaluate their performance in reproducing, generalising and explaining the observed air quality
- investigate ways in which modelling could be improved
- demonstrate and extend the use of temporal and spatial regression modelling techniques for roadside air quality assessment.

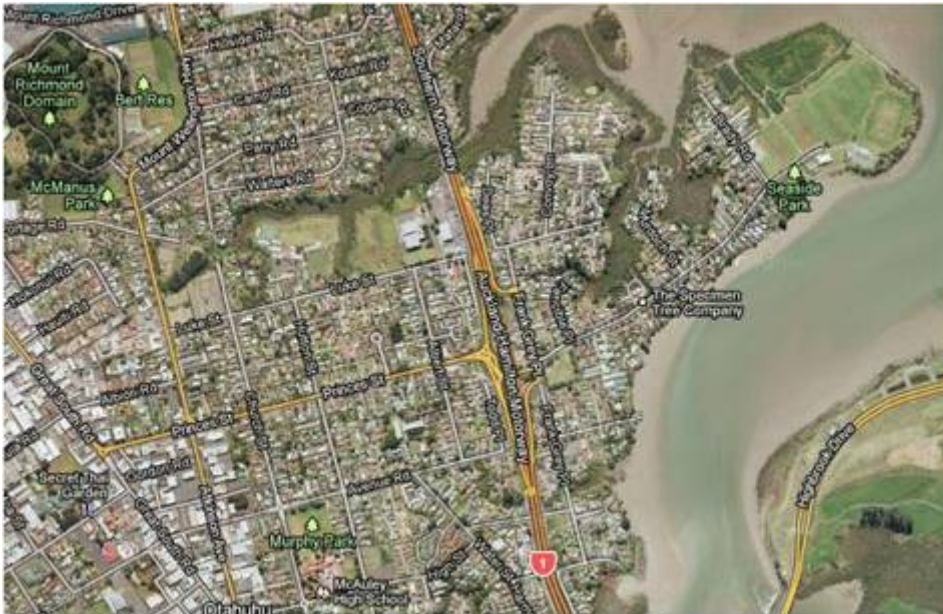
Although road traffic is known to be responsible for the emissions of many different air pollution species, we chose oxides of nitrogen (NO_x) to be the primary indicator and main focus of our study. This is because technology for monitoring NO_x is well developed and widely used, road traffic is the dominant source of NO_x emissions in most urban areas, and NO_x instruments have a high sensitivity relative to the concentrations and concentration gradients observed in roadside locations. We also included assessments of PM_{10} and nitrogen dioxide (NO_2) due to the existence of NES for both, and the widespread use of both indicators in health risk analysis. We also made measurements, where practical, of carbon monoxide (CO), particle number concentrations (PNC, often used as an indicator of UFP) and BC during mobile monitoring, and ozone (O_3) at the fixed sites, as alternative indicators of road traffic emissions which have been associated with adverse effects on human health.

Passive monitoring was to be conducted over a year to provide data across a full seasonal cycle, and due to its relatively low cost. During that year, continuous monitoring was planned for approximately three months, the shorter period being related to budgetary constraints. We deliberately chose the autumn-winter period to straddle the commencement of the home heating season, which begins approximately in May) and has a strong influence on particulate air quality in Auckland. Mobile monitoring was designed to represent a range of morning, afternoon, evening and night periods scattered through the monitoring campaign. As far as practicalities allowed we aimed to maximise the occurrence of simultaneous monitoring using the different modes (passive, continuous, mobile).

2.2 Study area location

The study area selected was in Otahuhu East, Auckland (figure 2.1), a low-lying residential area bisected by the Auckland southern motorway (SH1) carrying ~120,000 vehicles per day on average. We estimate that only 24km of motorway in Auckland have volumes of this magnitude or greater.

Figure 2.1 Satellite image of the study area. Most of the area is residential, with commercial land use to the west



2.3 Overview of the datasets generated

2.3.1 Passive monitoring

Passive monitoring of nitrogen dioxide (NO₂) was conducted at 32 sites across the study area in order to:

- provide far greater spatial coverage than could practically be achieved using continuous methods
- relate continuous measurements at distances less than 250m from the motorway to air quality at greater distances
- provide an input to LUR modelling
- provide a validation dataset for future modelling assessments.

Additional passive monitoring of nitrogen dioxide (NO₂) was conducted at 28 sites across the nearby and similar Mangere area in order to investigate the generality of LUR modelling (see modelling studies below) derived from passive monitoring data.

2.3.2 Full continuous observations

Continuous measurements were conducted at three sites, one to the west and two to the east of the Auckland southern motorway as illustrated in figure 2.2, during autumn, winter and spring 2010. The sites were:

- on undeveloped land at 38 **Luke Street**, ~250m west of SH1

- on **Deas Place Reserve**, immediately adjacent to the southbound Princes Street off-ramp of SH1
- in the rear yard of a private property-- **25 Deas Place** ~150m east of SH1.

At each of these three sites, measurements of CO, NO, NO₂, NO_x and PM₁₀ (plus O₃ at Deas Place Reserve and Luke Street) were conducted alongside meteorological observations. Details of all instrumentation can be found in appendix A (section A5). During the campaign meteorological conditions were typical for the season with a general preponderance of south-westerly winds.

Figure 2.2 The three fixed monitoring sites (boxes). From left to right (west to east): Luke Street, Deas Place Reserve and 25 Deas Place



Figure 2.3 The kerbside monitoring station on Deas Place Reserve



3 Outcomes of the observational study

3.1 Continuous monitoring

3.1.1 Comparison with the national environmental standards and air quality guidelines

Two pollutants for which NES exist in New Zealand were measured – particulate matter (PM₁₀) and nitrogen dioxide (NO₂). Observed air quality was fully compliant with these NES at all times. Table 3.1 shows the highest recorded concentrations of each and the percentage of the limit reached for each averaging period.

Table 3.1 Observed maximum concentrations ($\mu\text{g m}^{-3}$) relative to national environmental standards and air quality guidelines for PM₁₀ and NO₂.

NES/AQG ^(a)	Limit value	Deas Place Reserve (kerbside)	25 Deas Place (100m setback)	Luke Street (250m setback)
PM ₁₀ (24 hr)	50	43 (86%)	44 (88%)	43 (86%)
NO ₂ (1 hr)	200	86 (43%)	55 (28%)	68 (34%)
NO ₂ (24 hr)	100	44 (44%)	28 (28%)	30 (30%)

^(a) AQG = air quality guidelines

The highest recorded 24-hour PM₁₀ concentration – 43 $\mu\text{g m}^{-3}$ – was recorded on 17 June 2010. Twenty-four hour PM₁₀ concentrations above 25 $\mu\text{g m}^{-3}$ were only observed when the daily minimum one-hour temperature was less than 8°C and daily mean wind speed was less than 2 m s⁻¹, indicating that peak concentrations were associated with a domestic heating source.

3.1.2 Effect of the road on concentrations

The average PM₁₀ concentration was 17 $\mu\text{g m}^{-3}$ at both of the setback sites and 18.7 $\mu\text{g m}^{-3}$ at the kerbside site. Therefore, on average, the motorway contributed 1.7 $\mu\text{g m}^{-3}$ (or 10%) to PM₁₀ at the motorway's edge. We estimated that on days when domestic heating sources and low wind speeds led to peaks in 24-hour PM₁₀ concentration, the absolute kerbside increment was 2.1 $\mu\text{g m}^{-3}$, but that the relative contribution was reduced to 7% due to higher overall concentrations.

Mean NO₂ concentrations at the kerbside site were 7–10 $\mu\text{g m}^{-3}$ higher than at the setback sites. The average NO₂ concentration at the motorway's edge was 24 $\mu\text{g m}^{-3}$, or 60% of the WHO guideline of 40 $\mu\text{g m}^{-3}$.

Mean NO_x concentrations at the kerbside were approximately double those at both setback sites. The diurnally averaged difference in NO_x concentrations between the kerbside site and the setback sites (figure 3.1) clearly resembled the diurnal cycle in traffic volume, and peaked during the morning traffic peak at around 180 $\mu\text{g m}^{-3}$. Background NO_x concentrations were comparable in our study and a similar study in Las Vegas (both ~90 $\mu\text{g m}^{-3}$) but absolute roadside concentrations were higher. This observation is consistent with the Auckland vehicle fleet producing higher NO_x tailpipe emissions compared with the American fleet, due to the former's higher proportion of older vehicles.

Both NO₂ and NO_x show distinctive elevated roadside concentrations as seen in figure 3.1. However, PM₁₀ concentrations at the roadside are almost indistinguishable from the setback sites (figure 3.2).

Figure 3.1 Diurnal average hourly NO_2 and NO_x at the three sites during the intensive observational period

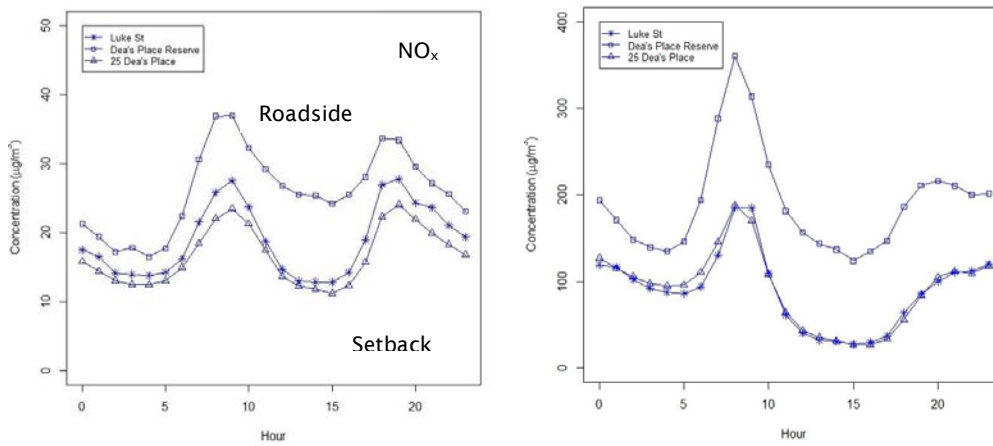
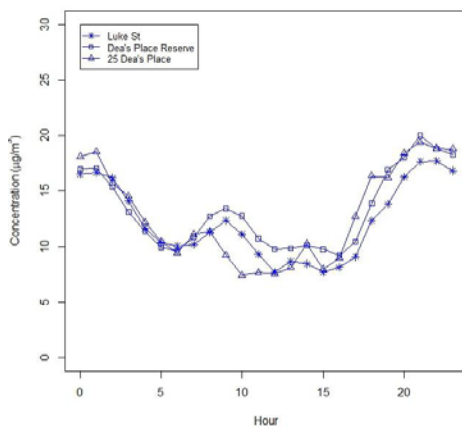


Figure 3.2 Diurnal average hourly PM_{10} at the three sites during the intensive observational period



3.2 Passive monitoring

Annual average NO_2 across the study area ranged from $12\mu\text{g m}^{-3}$ to $26\mu\text{g m}^{-3}$ (or $22\mu\text{g m}^{-3}$ when one outlier influenced by a busy intersection was removed). This represents at least a two-fold variation over this relatively small study area.

Figure 3.3 Campaign-mean NO_2 concentrations for all deployments in Otahuhu overlaid onto a satellite image



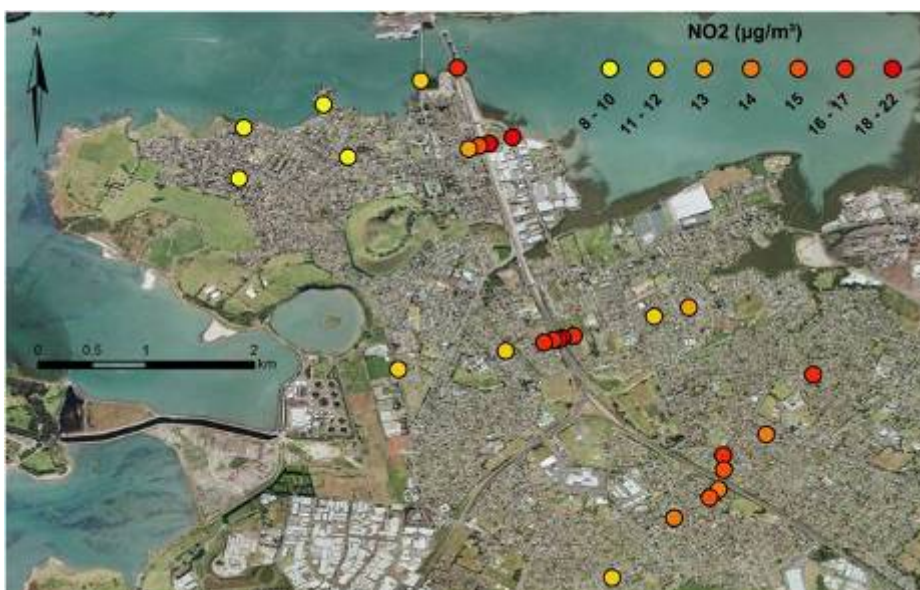
The data reveals:

- strong localised gradients within ~200m of the motorway
- localised gradients in close proximity to the other two significant roads in the study area: Atkinson Ave (on the far west of the study area) and Princes Street (running east-west through the centre of the study area)
- a larger scale gradient with slightly higher concentrations in the west and south compared with the north and east.

Analysis suggests that the larger scale gradient may be related to general traffic density, or to differences in dispersion characteristics, maybe due to the influence of the creeks bordering the north and east side of the study area.

Additional monitoring was conducted at 28 sites in the nearby suburb of Mangere (figure 3.4) which is also dissected by a busy motorway, in order to investigate the generality of findings regarding motorway-dominated suburbs. There was a similar range in concentrations and roadside gradients as that observed in the Otahuhu study area.

Figure 3.4 Campaign mean NO_2 concentrations for all deployments in Mangere overlaid onto a satellite image



4 Outcomes of modelling studies

4.1 Project findings regarding emission-dispersion modelling (Ausroads+VEPM)

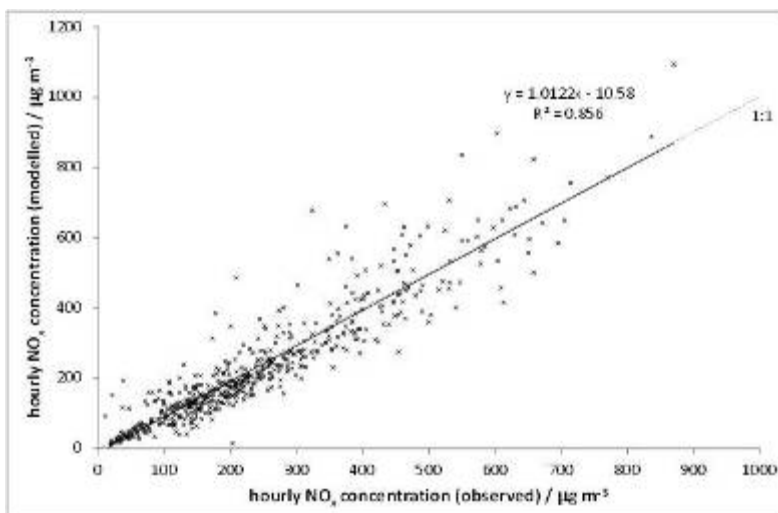
The pairing of the New Zealand vehicle emission model VEPM and the atmospheric roadside dispersion model 'Ausroads' is often the default choice for predicting the impact of a road on roadside air quality. These models are commonly used in a regulatory context for assessing a proposed road project which may be a new build, or a change to an existing road. In this study we evaluated the ability of the model pair to predict the air quality impact as observed in our study area. This evaluation included specifically assessing Ausroads independently of VEPM.

Multiple scenarios were modelled using the common dispersion/emission model pairing of Ausroads+VEPM. The 'baseline' scenario modelled a single line source with atmospheric stability and mixing height assumed to be constant, using on-site meteorological observations. For all scenarios, Ausroads was found to overpredict concentrations in the 150m zone by a factor of one to three times, with the factor tending to three in the daytime. This over-estimation was found to be independent of the use of VEPM.

4.1.1 Modelling NO_x

- Using the baseline scenario, Ausroads+VEPM was generally successful at predicting NO_x concentrations at the motorway's edge (figure 4.1).

Figure 4.1 Estimated hourly NO_x concentrations (Ausroads/VEPM prediction + observed background concentration from Luke Street) versus observed hourly NO_x for kerbside site (Deas Place Reserve) in westerly winds only



- We found no evidence of VEPM contributing to modelling error for NO_x. Indeed, analysis of the kerbside data suggests VEPM was predicting NO_x emissions to a high degree of accuracy within the context of the typical modelling performance.

4.1.2 Modelling PM₁₀

- Hourly PM₁₀ at 150m distance was overpredicted by 0–15µg m⁻³ in the morning. Modelling overprediction appeared to be offset by the influence of un-modelled sources/processes in the afternoon and evening.
- Combining Ausroads + VEPM output for PM₁₀ with various background data led to an overprediction of absolute concentrations by 0–20µg m⁻³ (5–10µg m⁻³ on average).
- Due to the relatively low sensitivity of the PM₁₀ metric to road traffic emissions, and the inherent variability of data reported by the monitors used in the regulatory context, the sensitivity of predictions to the source of background data was not as great for PM₁₀ as it was for NO_x.
- Our conclusions regarding PM₁₀ are necessarily tentative due to large underlying uncertainties. In general, our modelling estimates led to an overprediction of PM₁₀ at the kerbside, relative to observations. Given the good performance in predicting NO_x, we assume that Ausroads is not contributing to prediction error at the kerbside. In that case, the overprediction in PM₁₀ can be attributed to either:
 - overprediction of particulate matter (PM) emission rates by VEPM, or
 - invalidity of the assumption that the setback Luke Street site adequately represented the background, ie all sources affecting the kerbside site except the motorway.

4.1.3 Sensitivity to model input

- Use of hourly estimated atmospheric stability using a well-established algorithm increased model overprediction at night.
- Using off-site meteorological data led to underestimates of concentration attributable to the motorway of 22% to 35% in five out of eight cases – the specifics of the site and dataset were significant.
- The error introduced by using longer ‘historic’ meteorological datasets (as opposed to observations made during our campaign) was small in comparison to this spatial error and acted to increase the net error for three out of four off-site datasets.
- Use of the artificial meteorological dataset created on behalf of the ARC for regulatory assessments reduced the ‘error’ relative to using a locally observed dataset to < ~10%.
- Relative to using our baseline meteorological dataset, using data from two other (shorter) on-site masts led to a fairly consistent over-estimation of mean concentrations at all receptors in three out of four cases.
- Modelling all eight lanes on the motorway as independent line emission sources led to a prediction of concentrations up to 22% lower than modelling the motorway as a single line source.

4.2 Project findings regarding site-optimised semi-empirical (SOSE) modelling

SOSE is a modelling tool which parameterises the relationship between meteorology, traffic and concentrations at a given point. It takes existing roadside observational data as an input and predicts the time-series of concentrations at the same point as an output. This study was the first time SOSE has been proven:

- for a motorway site
- to provide credible disaggregation of roadside data into road and background contributions.

4.2.1 Basic SOSE modelling

(Figure 4.2): In general, the model performed well for all pollutants and all sites. However, it tended to underpredict peak concentrations and slightly overestimate low concentrations (as is the norm for regression models). Weekday predictions were better than weekend predictions. The model performed better for the gaseous pollutants than for PM_{10} , probably due to the road dominating the source for the gaseous pollutants more so than for PM_{10} .

4.2.2 Using SOSE to estimate background concentration

(Figure 4.3): Very good agreement was found between predicted and observed background concentrations, especially for NO_2 . The results suggest that the motorway contributes a large fraction of the CO and NO_x at night. Approximately 20% of the PM_{10} originates from the motorway. The background dominates during the evening hours for PM_{10} whereas for NO_x , the motorway dominates consistently throughout the day.

Figure 4.2 Time series of one week of observed and SOSE-predicted concentrations for the Deas Place Reserve site a) CO b) PM₁₀ c) NO₂ d) NO_x

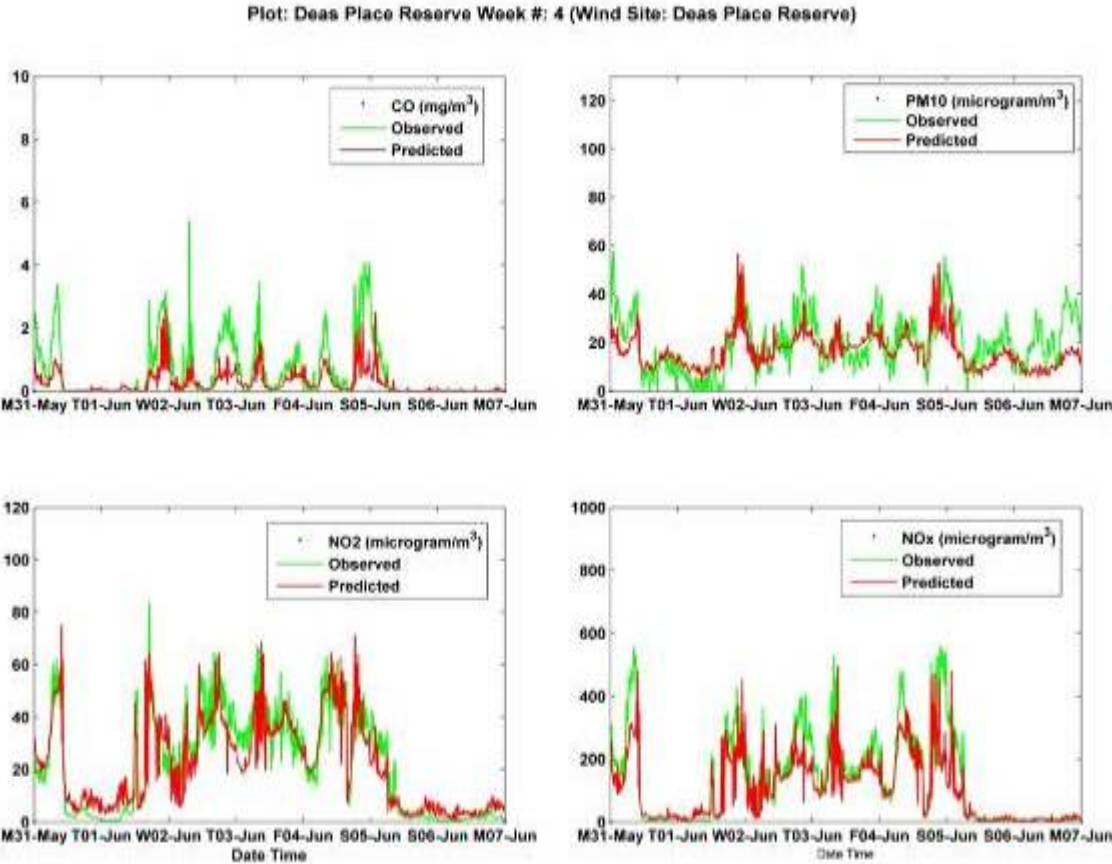
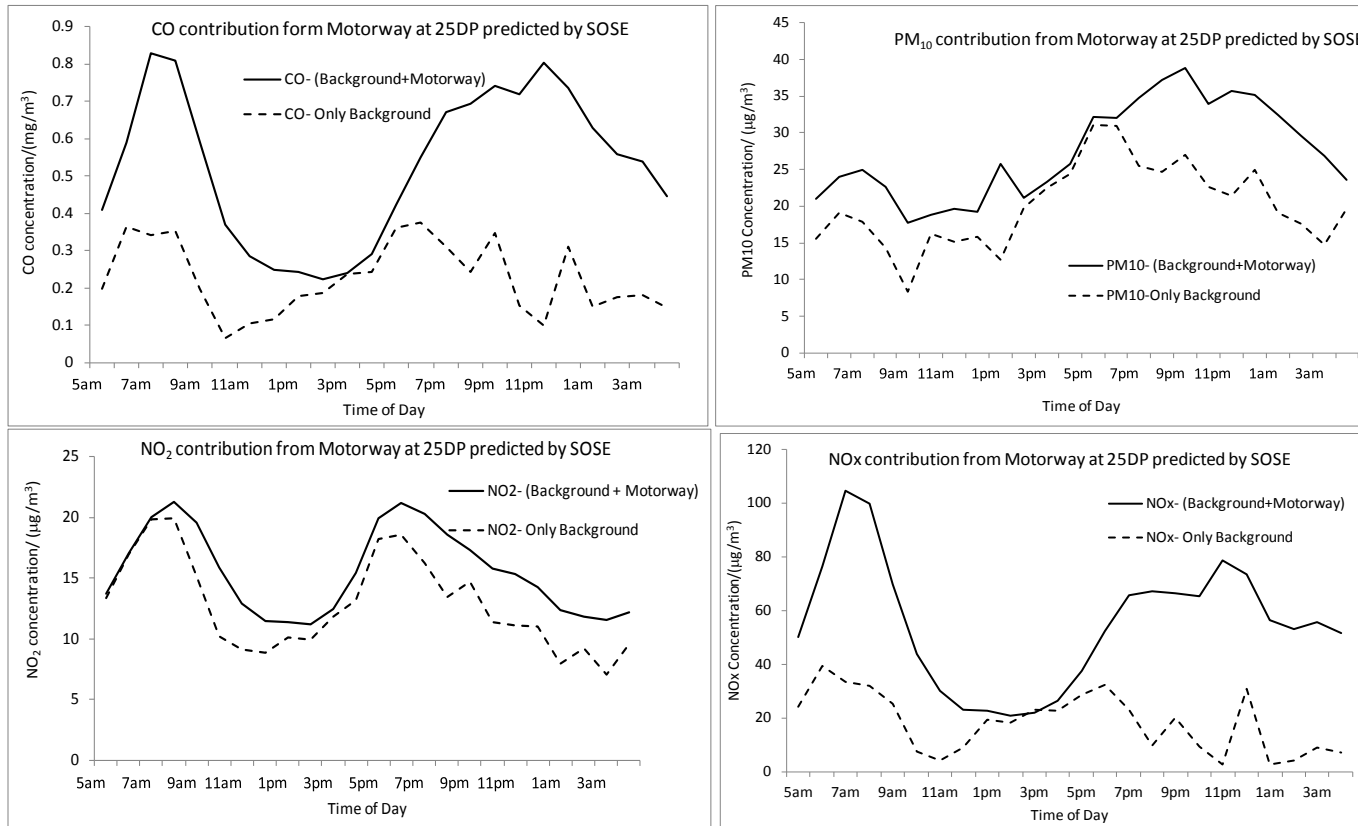


Figure 4.3 Daily profile of the contribution of the motorway and the background to the total daily concentration profile for each pollutant for westerly winds a) CO b) PM_{10} c) NO_2 d) NO_x



4.2.3 Choice of meteorological monitoring site

The SOSE model, like dispersion models, is dependent upon a source of representative meteorological data. Sensitivity tests showed that model performance was strongest when using data from:

- 1 a high meteorological tower (10m better than 6m)
- 2 a meteorological tower in an open area (more representative of the regional flow)
- 3 a meteorological observation site as close as possible to the air quality monitoring site, as long as (2) is not compromised.

4.3 Project findings regarding land-use regression modelling

LUR models were generated to describe the spatial pattern in long-term average nitrogen dioxide across the study area, and the Mangere area (independently), at 20m resolution (figure 4.4).

Performance of the Mangere model was good by international standards (R^2 of 0.77). Performance of the Otahuhu model was weaker (R^2 of 0.49) (see figure 4.5).

The weaker performance of the Otahuhu model may be due to the very fine spatial scale of the study area within which there is limited geographical variability in land use with which to predict spatial variation in air pollution. Nevertheless, model performance was still acceptable with the main features of observed spatial variation adequately described.

Figure 4.4 Map of NO_2 predicted by land-use regression model for Otahuhu

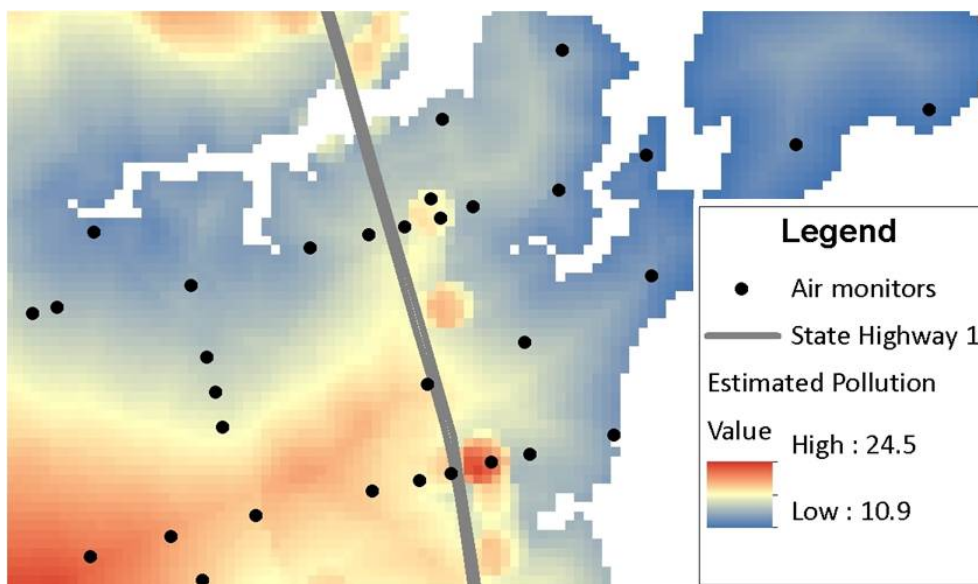
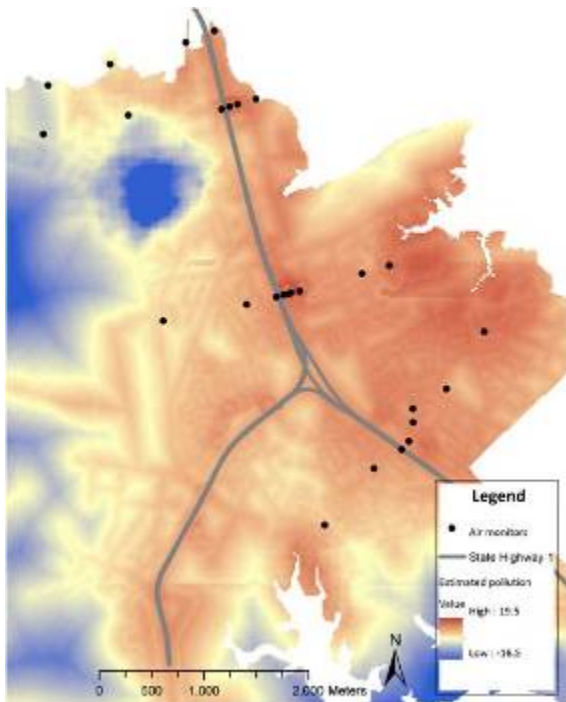


Figure 4.5 Map of NO₂ predicted by land-use regression model for Mangere



4.4 Project findings regarding the roadside corridor model

- A roadside corridor model (RCM) was recently developed by NIWA (Longley et al 2011) This model was intended to occupy a middle ground between the complex physically based models and simple empirical models. Until now, the RCM remained unvalidated.
- Direct evaluation of the RCM predictions is not possible as motorway-only concentrations cannot be measured. However, a partial model evaluation was attempted by constructing a pseudo-observed time series of motorway-only concentrations at the project's fixed continuous monitoring sites. These exploited the presence of an 'upwind' site which we assumed represented non-motorway contributions. This evaluation showed good predictive power for NO_x at the kerbside in the am peak, and an overprediction when a full 24-hour period was modelled. The model underpredicted the observed dispersion between the kerbside and setback sites, consistent with the similar underprediction by Ausroads, upon which the RCM is based.
- Evaluation of the RCM's predictions for PM₁₀ was hampered by the low sensitivity of the PM₁₀ metric to vehicle emissions, and probably confounded by near-field domestic woodsmoke sources breaking the assumption that upwind sites represented all non-motorway sources.
- The small degree of overprediction in using the RCM is likely to make it suitable for screening applications for which it was intended where a small degree of conservatism is acceptable or desirable.

5 Recommendations for users conducting health risk analysis and epidemiological studies

5.1 Introduction

This is not a health outcomes study. However, it was designed to enable and extend health impact and epidemiological study of emissions from both specific roads and from road traffic in general. This section explains how the results from this project can be used in health-based studies.

5.1.1 Health impact analysis

In the context of our study we view health impact or health risk analysis to have two purposes:

- 1 The quantification of predicted health outcomes enabled through **established** exposure-response³ relationships for the purposes of risk assessment and management
- 2 The establishment of **new** exposure-response relationships, either in cases where none have existed before (eg in the case of new or emerging risks), or to develop New Zealand-specific relationships if it is considered that the New Zealand population may exhibit a different response from that established abroad; ie for the purposes of advancing epidemiological knowledge.

In the context of general air quality, health risk assessments have previously been conducted in New Zealand using well-established exposure-response relationships derived outside New Zealand. These relationships have been based primarily on measures of 'airshed-scale' PM₁₀ derived from an urban air quality monitoring station as the indicator of exposure. Additionally foreign relationships for NO₂ and CO have also been applied in New Zealand. It is questionable whether these relationships are appropriate for the New Zealand population, which has a particular vulnerability profile (eg high rates of asthma and cardiovascular disease).

In the specific case of near-road exposure to road vehicle emissions, true exposure is qualitatively different from that observed at many monitoring stations due to the altered physico-chemical composition of pollutants, compared with both the climates in which much foreign data originates (eg high ozone levels in California), and to non-roadsite sites even in the same city. Furthermore, PM₁₀ is a relatively poor indicator of traffic pollution, due to its large bias towards non-traffic sources. Toxicological studies strongly imply that the toxic potential of exhaust particles is much more strongly represented by measures that emphasise the smallest, fresh combustion products which are under-represented by PM₁₀. Thus, PM₁₀ measures smooth out the significance of localised sources and exhibit low spatial variation across a city. We conclude from this that using existing international general exposure-response relationships to characterise roadside exposure is potentially flawed.

The combination of the weak ability of PM₁₀ to represent the toxic potential of traffic emissions relative to more representative metrics, and its inability to reveal fine-scale gradients in health-relevant exposure is borne out by the substantial recent growth in research revealing that intra-urban gradients in health impacts arising from traffic emissions are greater than inter-urban gradients (Jerrett et al 2005). Road-

³ The term 'exposure-response' is used here to represent the statistical association between exposure to air pollution (the term 'dose' is often used in health literature) and the probability of an increase in the rate of a specifiable health outcome (eg mortality, disease prevalence or exacerbation) within a population.

specific exposure-response relationships now exist relating multiple health outcomes to residential proximity to major roads, generally relating the exposure to a traffic-specific pollutant, such as NO_2 , NO_x , CO, BC or PNC (Boothe and Shendell 2008). Nearly all of these relationships are derived from observational data gathered in North America or Europe, ie in different climates to New Zealand, with different vehicle fleets, road and land-use layouts, and different populations with potentially different sensitivities.

Considering this context, our project aimed to contribute to the following long-term aims:

- 1 To ensure that when general exposure-response relationships based on PM_{10} , CO or NO_2 are used in New Zealand, the exposure is specified as accurately as possible.
- 2 To establish the ways in which road emissions, roadside dispersion and roadside air quality differ or are similar in locations where roadside exposure-response relationships have been derived abroad, so that those relationships can be applied with greater confidence in New Zealand (on the assumption that the population sensitivity is the same), or can be adapted to account for New Zealand-specific conditions.
- 3 To characterise roadside air quality in New Zealand in sufficient detail to enable the investigation of New Zealand-specific exposure-response relationships.

Thus, in general, this project seeks to deliver improvements to the 'exposure' half of the exposure-response relationship.

5.2 The advances in exposure assessment for health risk analysis provided by this project

5.2.1 Exposure scales

A health risk analysis is usually conducted by determining the exposure of a subject population, using an exposure-response relationship to estimate a related probable health risk outcome and scaling that by the size of the population, adjusting for vulnerability if possible. Assessments are often comparative, comparing with a reference population or comparing a future scenario with a baseline scenario.

A key issue is the scale of aggregation. Both exposure assessment and exposure-response functions are scale specific. Ideally both should be conducted at the same scale (eg city, airshed, census area unit, census meshblock or road-corridor-exposure zones) but often they are not. Our study provided data that has helped to better define the scales of exposure and how exposure may be quantified, as well as indications as to how exposure-response relationships should be applied and can be improved.

5.2.2 PM_{10} exposures

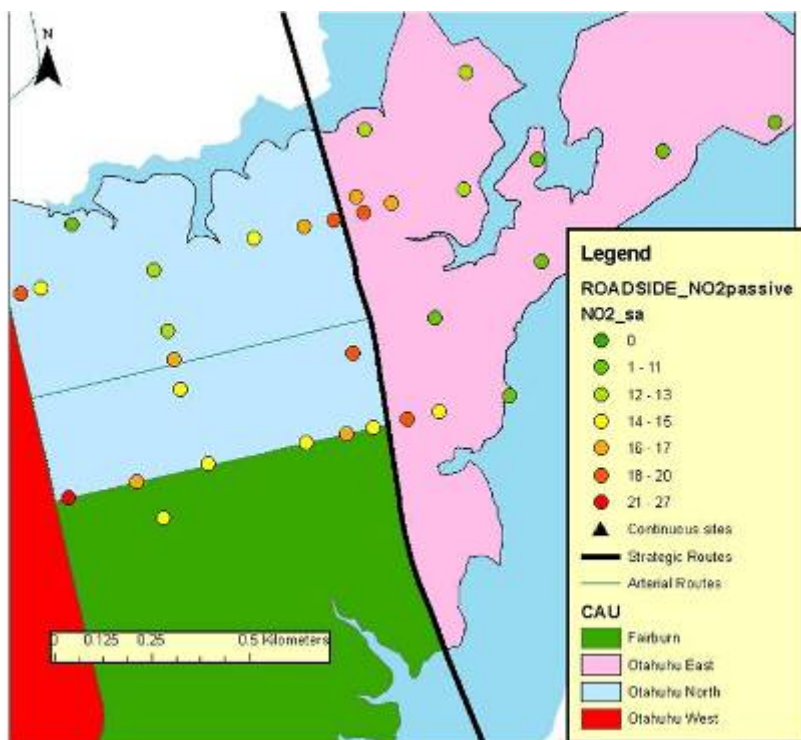
Our study showed that the roadside increment in PM_{10} is small relative to background concentrations, and the roadside corridor appears to be narrow (although the study also indicated that the relatively low sensitivity of PM_{10} as a measure of traffic emissions makes it difficult to accurately specify the roadside increment). The RCM was successful at predicting the enhancement at the kerbside site and we conclude that it can be used to describe the average roadside increment close to the road. However the roadside dispersion model Ausroads and RCM together, as used in this study, appeared to slightly exaggerate the corridor width. This could lead to a small positive bias (overestimate of risk) in health assessments.

5.2.3 NO₂ exposures

Our study showed that the influence of a road on NO₂ concentrations can extend further than for PM₁₀, and that the increment is relatively greater. RCM was able to reasonably predict the extent of the roadside impact when modelling NO_x. This is somewhat unexpected and may be due to opposing errors (underprediction of dispersion and ignoring secondary sources of NO₂) acting to cancel each other out. Further investigation of the observational and modelling datasets should provide further insight, and confirm whether RCM is an appropriate tool, or needs further development, for general use in describing roadside NO₂ gradients and in health risk analysis.

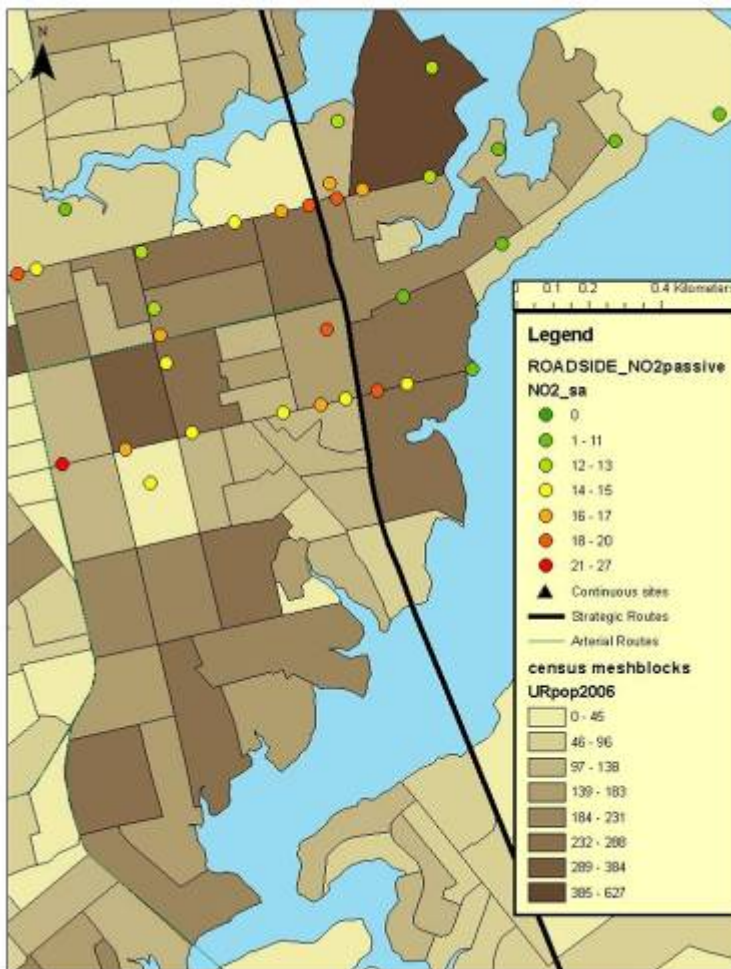
Beyond the roadside corridor, our study showed significant variation in NO₂ levels across the study area related to local traffic. This was detected using passive monitoring, and generalised using a LUR model. This means that, at the scale of census area unit (CAU), there was substantial variation (12–26µg m⁻³, see figure 5.1). Indeed, the variation within each CAU in the study area was comparable to the variation in NO₂ across the whole of Auckland.

Figure 5.1 Concentrations of nitrogen dioxide (observed by passive monitoring) across the study area overlaid on census area units



The census meshblock scale is more appropriate for aggregating variation in NO₂ as both have similar length scales (~10–100m). This is shown in figure 5.2. A LUR model, or other spatial interpolation or geographic information system (GIS) method, could be used to allocate mean NO₂ values to each meshblock in this (or any other) study area without substantial loss of accuracy. However, both figures 5.1 and 5.2 indicate that data collected along roads (lamp posts are regularly used for passive monitoring) often fall along the boundaries of CAUs or meshblocks which also use roads as borders. This complicates the process of allocating concentration values to particular meshblocks or CAUs.

Figure 5.2 Concentrations of nitrogen dioxide (observed by passive monitoring) across the study area overlaid on census meshblocks



Note: Brown shading indicates usually-resident population

5.2.4 Using PM₁₀ exposure-response relationships

The Health and Air Pollution in New Zealand (HAPINZ) study (Fisher et al 2007) applied a PM₁₀ exposure-response function of a 4.3% increase in mortality risk per 10 $\mu\text{g m}^{-3}$ increase in PM₁₀. This function was used following the influential assessment by Künzli et al (2008) who in turn chose this function after considering evidence from the Harvard Six-Cities (Dockery et al 1993) and American Cancer Society (Pope et al 1995) cohort studies. The same function has since been used in at least one New Zealand road project assessment (Kirkby 2010). It should be appreciated that this function is derived from city-scale monitoring and assessment of PM₁₀ from all sources. To apply such a function to much smaller scales, especially below the CAU scale, is highly questionable and implies substantial uncertainties.

The updated HAPINZ study (Kuschel et al 2012) has used a function specifically derived in New Zealand (Hales et al 2012) of a 7% increase in the risk of mortality per 10 $\mu\text{g m}^{-3}$ of PM₁₀. This function was derived using a CAU-scale exposure model (Kingham et al 2008), but aggregated into quintiles. Again a mismatch in scales between the exposure of interest (eg roadside increments) and exposure-response function could introduce major uncertainties into any assessment.

In both of these cases, all PM is assumed to carry equal weight in terms of impact on mortality. Toxicological evidence suggests that it is highly unlikely that traffic emissions are as toxic as other emission sources (eg Cavanagh et al 2009), and that fresh emissions (as found in the roadside corridor) are toxicologically different from aged emissions (as found in more background locations, Yanosky et al 2012). Consequently, we conclude that PM₁₀ exposure-response relationships are poorly suited to quantifying the health risk associated with roadside locations.

5.2.5 NO₂ exposure-response

There is less literature generally on exposure-response relationships for NO₂. The HAPINZ study used a function derived specifically for Auckland by Scoggins et al (2004). In this study airshed dispersion modelling was used to predict mean NO₂ levels for each CAU across Auckland. This was linked to mortality data to derive a relationship of 0.13% increase in mortality risk per 10µg m⁻³ of NO₂. When combined with population and baseline mortality data this led to a similar estimate of traffic-pollution mortality burden as in a previous assessment by Fisher et al (2002) using PM₁₀ data and exposure-response functions. This indicates that neither PM₁₀ nor NO₂ alone are causal factors in the mortality, but that both act as proxies for the causal factor. The advantage in using NO₂ is that being more directly associated with traffic sources the impacts can be more confidently ascribed to traffic rather than to other sources, and that spatial and local variations can be investigated with greater confidence.

The method of Scoggins et al (2004) did not include localised roadside gradients, which our study showed to be substantial. There is great potential for the health burden associated with traffic pollution to be substantially underpredicted because of this, especially if the potentially greater toxicity of roadside air and synergistic effects of other pollutants is taken into account. This raises the possibility that the exposure-response relationship may not be suitable within the roadside corridor.

5.2.6 Traffic proximity exposure-response

There is substantial and growing evidence that the compositional mix of roadside air is different from other air pollution, accompanied by toxicological evidence that roadside air elicits different biological responses (eg Yanosky et al 2012). This implies that new exposure-response relationships specific to the particular pollutant mixture present in roadside air should be developed and investigated.

It is unclear at present whether such a dose-response relationship can be developed based on a single pollutant metric. We have shown how PM₁₀ (and by implication PM_{2.5} also) is poorly suited to do this. NO₂ has potential, but other studies have shown that PNC and BC are more sensitive to traffic emissions and might be better suited to this job. It is for this reason that PNC and BC measurements were added to the project (ie the mobile monitoring). On the other hand, a new compound multi-pollutant metric might be necessary.

In lieu of such a new roadside air pollution exposure-response relationship, many other studies have identified clear exposure-response associations based on either proximity to roads, or road traffic, or road/traffic density in the surrounding area. These findings have been repeated widely and are considered to be robust, and were one of the major motivators for this project. However, because these relationships do not consider air pollution, they may be strongly confounded by other stressors. These definitely include noise, but potentially also other factors influencing poor health outcomes near major roads, such as psycho-social stress associated with community severance, and elevated vulnerability associated with poverty (roadside communities are often poor communities).

In general, risks associated with roadside corridors have been associated with morbidity more than mortality. The key study in this area is McConnell et al (2006) which reported that children living within

75m of busy roads are more likely to have asthma than those living further away. This finding came from a study of over 5000 children in Southern California. The finding was used to represent a chronic health risk and compared with the health burden arising from asthma exacerbations due to nitrogen dioxide and other causes by Künzli et al (2008). This study found that 20% of children in the city of Long Beach lived within this 75m risk zone, and this accounted for 9.3% of all asthma cases, and that the total health burden was 40% greater if roadside corridor exposure was considered.

The major obstacle to reproducing such assessments elsewhere is that it cannot be assumed that traffic proximity relationships can be applied in locations other than those where the data originated, without introducing error. This is because the risk from air pollution '150m from a freeway' in California for instance, is likely to be different in New Zealand for several reasons, such as:

- American freeways are often wider
- American vehicles have different emission rates
- background air pollution levels are substantially different
- the climate is different, affecting dispersion curves.

The Künzli et al (2008) study was made possible and credible by the existence of locally generated exposure (traffic proximity)-response relationships. However, given the new data captured in our project, it should now be possible to account for many of the differences listed above.

When developing a city-scale exposure assessment it is reasonable to assume that most people spend most of their time in the city and so the assessment applies to them most of the time. For smaller spatial scales this becomes more problematic. In a roadside zone, strong concentration gradients, in which metres matter, mean that exposure errors are almost guaranteed – the gradient in concentration within a single property can be significant. When census meshblock data is used to provide population data, assignment of the population to the meshblock centroid would probably underestimate the number of residents living close to roads. In the assessment by Künzli et al (2008) census meshblock data was used to allocate subjects (children) evenly across each meshblock volumes, taking care not to place subjects within 60m of the centrelines of freeways, 45m of other highways, 30m of major arterials or 10m of other roadways.

Many people will find themselves in the roadside zone for only part of their daily lives, or for short or sporadic periods of time. Research is ongoing using the project database to understand the significance of personal time-activity profiles in weighting the impact of the roadside corridor on residents' exposure.

5.3 Recommendations for the conduct of roadside health risk assessment

Based on our findings within this study we make the following recommendations:

1 Determine objectives.

Many of the decisions to be made in conducting an assessment are related to the assessment objectives. Clearly determining those objectives at the start of the process is highly recommended. Amongst the questions to be addressed are:

- a Will the assessment be aimed primarily at relative or absolute measures – eg comparison of alternative scenarios (in which the accuracy of absolute burdens is less important), or on assessing the absolute impact (to determine cost burden or significance)?

- b Is the assessment considering a single road, a single community, or all roads (in a city/region/country)?
- 2 Choice of exposure metric:
- a Traffic-related air pollution is a complex mixture for which there is no single universal measure, and any given measure is therefore incomplete and limited in some way.
 - b Data availability, quality, bias, representativeness and/or the ease of acquiring data will be major considerations which are covered in detail in chapter 6.
 - c PM_{10} (and $PM_{2.5}$) have a substantial history, backed by robust, verified and widely used exposure-response relationships. However, those relationships have not been developed for use in roadside corridors and their applicability in this zone is currently unknown. We have also shown that PM is a relatively insensitive measure for roadside impacts. This makes PM suitable only for relative measures or for large populations where small-scale detail is not significant.
 - d Nitrogen dioxide (NO_2) is much more sensitive to traffic, and better suited for detecting localised and subtle variations in exposure and attributing them to traffic sources. However, there is much less literature quantifying the exposure-response relationships allowing fewer morbidity outcomes to be assessed.
 - e Traffic/road proximity or density measures allow access to a rapidly growing literature on a very wide range of health outcomes, including asthma, effects on children and adverse birth outcomes. However, these measures can be vague and when transferred from one setting to another (eg country of origin to New Zealand), errors can be introduced which can only be minimised by considering differences in emissions and dispersion.
- 3 Exposure assessment:
- a This is dealt with in detail in chapter 6. However, in the context of health risk assessment we recommend caution over allocating populations to CAU or meshblock centroids as this is likely to underestimate the population affected by localised concentration gradients.
 - b This is not a health outcomes study and as such it has not produced any new exposure-response relationships. However, the study has been designed to enable this in the future (see recommendations for further research below).

5.4 Research recommended to further the options for health risk assessment

We recommend that the data gathered in this project is used to relate the pollution exposure risk implied by roadside corridor studies elsewhere (eg the risk of asthma prevalence from living within 75m of a southern California freeway) to the equivalent in New Zealand by quantifying the differences in:

- vehicle emission rates (both modelled and computed from inverse dispersion modelling)
- atmospheric dispersion and transformation.

Further study to generalise the observed NO_2 gradients and resolve the divergence found in this study between observed and modelled roadside gradients should be progressed to validate whether the RCM (or some alternative) is an appropriate tool for roadside risk analysis.

5.5 The advances in exposure assessment for epidemiology provided by this project

There are three main needs for new epidemiological research:

- 1 Generation of exposure-response relationships specific to traffic-related air pollution
- 2 Generation of exposure-response relationships specific to New Zealand
- 3 Generation of exposure-response relationships specific to the roadside corridors within which air pollution composition and toxicity are distinctly different from other environments.

This project was designed (within available resources) with these three needs in mind. It is our hope that the data collected, especially regarding NO_2 , NO_x , UFP and BC, in combination with modelling techniques which allow near-traffic and background contributions to be disaggregated, will be helpful in providing the exposure assessments needed to progress all three needs.

The project has provided a valuable database which could be used to characterise exposure in the study area in great detail and thus provide the basis of a health outcomes study. Although not covered within the scope of this project report, our study was supplemented by particulate sampling for toxicological analysis permitting further investigation of how composition and toxicity in the roadside corridor differed from the urban background.

The modelling developments within this project, and those which could be based upon it in the future, are also designed to extrapolate our findings to other roadside corridors across the country. We recommend that a study is conducted to revisit the Auckland-specific exposure-response functions estimated by Scoggins et al (2004) by both updating the available data, and by incorporating the roadside corridor which was excluded in that study. Such a study should further compare the power of NO_2 as a measure of roadside impacts, relative to PM_{10} .

5.6 Further science that can now be developed as a result of the project outputs

- Insight into how the RCM, LUR and SOSE can be combined into a general, consistent traffic emissions exposure model for short and long-term impacts of single roads, road networks and whole cities.
- Relationships between short-term and long-term air quality risks at roadside locations.
- Enabling long-term forecast, hindcast and scenario evaluation into roadside air quality assessment tools.
- Development of tools for attribution of roadside air quality (particles and trace gases) to near-road, far-road and other sources (ie identification of transport contribution to roadside air quality).
- Generalised roadside curves for UFP and BC.
- Data to further investigate methods for enabling reliable geo-transferability of road traffic LUR models
- Guidance for application of foreign exposure-response relationships to New Zealand
- Roadside risk assessment for New Zealand.
- Vehicle emission factors for UFP.

6 Recommendations for users conducting roadside air quality assessments

6.1 Introduction

This section takes users through the decision-making process to be followed when undertaking an air quality assessment (regardless of the purpose or scale of assessment), in relation to the research described in appendix A.

6.2 Tools available

6.2.1 Relating to objectives

The selection of assessment tools – whether they be means of observing or modelling air quality – should be guided by the principle of ‘fitness for purpose’ and involve no more complexity than is necessary for the task in hand. This implies achieving clarity over the following considerations.

- 1 Is the focus on compliance with standards and guidelines, population exposure or other?
- 2 Is the focus on ‘typical’ or ‘extreme’ conditions, or both? Over what time scale?
- 3 Will multiple alternative scenarios be tested?
- 4 What degree of accuracy is required? What degree of uncertainty will be tolerated? Is conservatism required? If so, by how much?

6.2.2 Monitoring

Continuous monitoring (ie methods that provide continuous data at hourly or finer resolutions) is best-suited to a focus on temporal patterns, peak and atypical conditions. Although not covered in this project, work elsewhere (eg Sherman and Fisher 2007; Longley et al 2008; Mitchell 2012) has shown how peak and annual mean concentrations of PM_{10} and NO_2 can be empirically related, but there are inconsistencies so that a nationally applicable relationship has not yet been formulated. Nevertheless, these studies show that low-resolution passive monitoring may be used in a ‘screening’ mode to indicate the probability of occurrence of certain levels of peak concentrations.

Passive monitoring (or other methods that provide data with a low temporal resolution) is generally better suited to describing spatial patterns. This is especially true for nitrogen dioxide with which spatial variation in traffic-related air pollution can be monitored at relatively low cost. This study has shown how passive monitoring can identify localised variations in concentrations that would have remained unknown if only the fixed site continuous monitoring had been used. These features included localised hot spots associated with local roads and intersections, and a difference in NO_2 levels either side of the motorway, probably related to differences in residential and/or traffic density. It is worth noting that an emission-dispersion model also failed to identify these local variations as they were related to emission sources not included within the modelling.

Low-cost technologies capable of high resolution continuous monitoring are still in the development stage. Thus, at present, a combination of simultaneous continuous and passive monitoring offers the best (simple and reliable) way of covering variation in both space and time.

Passive or screening monitoring of PM is currently limited to instruments like Hi-Vol samplers or black smoke filters. New low-cost dust sensors are entering the market but are unlikely to be suitable for project assessment applications for some years. None of these technologies were tested within the scope of this project.

Mobile monitoring is a promising alternative that potentially provides detailed spatial information and some limited temporal information. It also offers a means of gathering detailed information about PM. However, the technique is not well established. We included mobile monitoring in this project and while we generated substantial amounts of data, we found that techniques for processing and analysing the complex data generated require further development before the information contained within this data can be fully presented and understood.

6.2.3 Empirical modelling

In this project we demonstrated the use of two empirical models. Empirical models generalise the patterns of variation found in an observational dataset (often termed the ‘training’ dataset). We used SOSE as an example of a temporal model and LUR as an example of a spatial model. SOSE takes continuous monitoring data as its input, whereas a LUR takes spatially varying monitoring data (usually passive) as its input. Consequently, both models describe, and partially explain, air quality ‘here and now’. These models offer several features beyond the observational data on which they are based, such as the ability:

- to interpolate between data points
- to offer a partial explanation of the variability
- to provide new information about the underlying properties of the model domain
- to identify localised, temporary or trending processes from residuals (divergence of observations and predictions).

Table 6.1 provides some specific examples.

Table 6.1 Capabilities of empirical models

	Temporal model (eg SOSE)	Spatial model (eg LUR)
Interpolation	Fill in gaps in monitored time series	Predictions for unmonitored locations
Explaining variability	How traffic volumes, wind speed and direction combine	Role of traffic proximity, density and volume, and other land uses
Properties	Background air quality	Background air quality
Localised processes	Step-changes or trends in emissions	Locations of localised hot spots

Empirical models provide the means to extrapolate data patterns beyond the training dataset. Thus temporal models like SOSE can make predictions of concentrations days, weeks, or years before or after the duration of the monitoring. Similarly, spatial models, like LUR, can be used to make predictions outside the spatial extent of the modelling domain. However, great care must be taken with these procedures. The lack of observational data means that validity of the predictions cannot be assured and it may not be possible to even estimate uncertainties (although this is true of deterministic models also).

In the case of SOSE, extrapolations to the same season in the following year (and previous year) have been shown to be credible (Dirks et al 2006). On the other hand, great care should be taken in extrapolating a winter training dataset to summer, for instance.

The relatively poor performance of LUR models when extrapolated (ie used to make predictions in locations outside the domain of the training dataset) is a well-documented weakness of LURs. Previous studies have taken models trained in one city and used them to make predictions in other cities. In our study we found that models developed for similar communities only 5km from each other were substantially different. This does not mean that LURs cannot be extrapolated over this short distance. Our modelling approach specifically optimised each model for local performance, not for transference. Future research should investigate changing the optimisation criteria to see if LUR extrapolation can be improved within a city.

6.2.4 Deterministic modelling

In this project we demonstrated the use of a deterministic model-pair (the dispersion model Ausroads and the emissions model VEPM), as well as a much simpler parameterised version of that pairing (RCM). These models are deterministic in the sense that they require no air quality observations as input. This means they are the natural choice for:

- predictions of current or historic air quality where no observational data exists, or
- predictions of future air quality or changes in air quality, including predictions relating to currently unbuilt roads.

Additionally, when air quality observations do exist, these models can be used to disaggregate that data into contributions from the subject road(s) and other sources.

A major point of difference between these models and the empirical models discussed above is that deterministic models predict the effect only of those emission sources explicitly described in the model, whereas empirical models (like the observations they are based on) describe the cumulative effect of **all** emission sources (although empirical models may allow some source disaggregation). Thus, to provide predictions of cumulative effect, predictions using deterministic models must be combined with other data describing non-modelled sources, which we typically describe as 'background air quality'. This background data can come from local monitoring, from more deterministic modelling (eg an airshed model) or from empirical modelling.

6.2.5 Model availability, access and alternatives

Four models were featured in this study representing four model 'types'. For each type, alternative models exist, as listed in table 6.2. Furthermore, other types of model also exist but were not featured in our study.

An Ausroads software package for Windows is available from the Victoria EPA (www.epa.vic.gov.au). It includes a graphical user interface and requires a meteorological dataset (time series) as input. An emissions dataset (time series) can also be provided as input, or emissions can be set to unity and the model output scaled by emission rates.

The VEPM is available from Auckland Council (www.aucklandcouncil.govt.nz) in the form of a macro-enabled Excel spreadsheet.

Site-optimised semi-empirical modelling (SOSE) is not available as an on-the-shelf product. Rather it is a method described in peer-reviewed literature (Dirks et al 2002; 2003). For further details, contact Dr Kim Dirks (k.dirks@auckland.ac.nz).

Similarly, LUR modelling is a method rather than a specific model. The basis of the method is described in Briggs et al (1997). For further details, contact Dr Simon Kingham (simon.kingham@canterbury.ac.nz).

The roadside corridor model (RCM) is available as an Excel spreadsheet from NIWA. Its basis is described in detail in *NZTA research report 451* (Longley et al 2011).

Table 6.2 Models used in this study, and alternative models

Model type	Example featured in this study	reference	Alternative models of this type	reference
Empirical (temporal)	SOSE	Dirks et al (2002)	Generalised additive model (GAM) Artificial neural network (ANN)	Carslaw et al (2007) Singh et al (2012)
Empirical (spatial)	LUR	Hoek et al. (2008)	Hybrid models	Wilton et al (2010)
Roadside dispersion	Ausroads	www.epa.vic.gov.au	CALINE HI-WAY Breeze ADMS-Roads CAR-FMI	www.arb.ca.gov www.epa.gov www.breeze-software.com www.cerc.co.uk en.ilmatietaenlaitos.fi/
Vehicle emission	VEPM	Kar et al (2008)	NZTER COPERT micro-scale	www.emisia.com/copert/ Smit and McBroom (2009)
Parameterised emission-dispersion	RCM	Longley et al (2011)	Reduced-form emission-dispersion model (RFEDM)	Batterman et al (2010)
Other	None		Quick urban and industrial complex dispersion model (QUIC)	Nelson and Brown (2010)

6.3 The importance of background air quality and how to assess it

6.3.1 Overview of background air quality

In the following discussion 'background' refers to pollution concentrations due to sources other than the road being assessed. This is equivalent to concentrations at locations which are beyond the direct influence of the subject road. 'Baseline' refers to existing air quality at all locations, roadside or otherwise, before a project or other change is progressed.

Any assessment of cumulative effects (total air quality) – which is essential for assessing compliance with the NES and ambient air quality guidelines (AQG) – requires the inclusion of an assessment of background air quality. Background air quality data can be generated from:

- on-site monitoring
- off-site monitoring
- airshed modelling
- background-focused or large-scale empirical spatial modelling
- roadside empirical modelling.

Off-site monitoring (ie monitoring more than ~200m from the subject road, and more specifically in a location likely to be affected by sources which do not affect the subject receptors) is commonly used if it is available. This data is often provided by regional council monitoring sites. The key consideration for use of such data is its representativeness of the study receptors. A generalised analysis of representativeness was beyond the scope of this project. However, we have shown how use of Auckland Council monitoring sites (more than 5km from our study area) introduced substantial error into an assessment of air quality in our study area. The error is likely to be greater when assessing short-term effects (such as NES-compliance) compared with long-term effects. The potential error is also substantially larger for NO₂ than PM₁₀. This is because vehicle emissions make a much larger contribution to NO₂ than PM₁₀ making NO₂ measurements more sensitive to their location vis-à-vis road traffic sources.

6.3.2 Background PM₁₀

Our study confirmed the findings of many others that roadside PM₁₀ concentrations are dominated by the background component rather than the road emissions component. This implies that where the objective is the assessment of cumulative effects with reference to the NES, or annual PM₁₀ guideline, compliance is going to be mostly dependent upon that background, and this should be the main focus of assessment efforts. Where the assessment objective is the comparison of alternative scenarios, however, attention should be paid to whether the background should be the same or different for each scenario.

In principle, spatial variation in PM₁₀ (ie variation between background and roadside sites, as well as other source of variation) can be described using screening monitoring approaches and empirical spatial modelling, such as LUR. However, this is limited at present by the lack of suitable low-cost monitoring technology, and was thus not included in our study. Instruments such as mini-vols, micro-vols and black smoke filters have been used for this purpose elsewhere. However, in the New Zealand context, such technologies are much more likely to identify variation related to non-transport sources and as such their applicability to roadside assessment is currently unknown.

Mobile monitoring has been used to provide training data for LUR models in a small number of research projects. Research is ongoing at NIWA and the University of Canterbury to determine if the mobile data captured during this study can be used in this way, and how.

Other tools available in New Zealand to estimate background PM_{10} include the original HAPINZ model (Kingham et al 2008), although this only predicts annual mean PM_{10} , and the updated HAPINZ model (Kuschel et al 2012).

6.3.3 Recommended research on background PM_{10} for improving assessments

- Develop spatial representativeness of monitors and decision criteria for when new local monitoring is needed.
- Implement PM screening/passive measurement technology and spatial modelling for roadside applications.
- Develop means of compensating for temporal bias and guidance for temporal extrapolation of monitored data.
- Validate the updated HAPINZ model.
- Add short-term prediction to the updated HAPINZ model.

6.3.4 Background NO_2

Our study confirmed the findings of many others that the enhancement in NO_2 concentrations in the roadside corridor (~200m) can be of a similar magnitude to, and therefore often greater than, the local background concentration. This is in considerable contrast to PM_{10} and is due to traffic emissions being the dominant source of NO_2 , but not PM_{10} . The implications of this are that spatial gradients in NO_2 are much steeper and therefore spatial variability much greater for NO_2 . Consequently this makes monitored NO_2 data much more sensitive to the details of the monitoring site vis-à-vis its relation to traffic. It also means that the spatial representativeness of any NO_2 monitoring site is likely to be much lower than for PM_{10} , especially if the site is influenced by traffic. The relatively low background levels of NO_2 in Auckland⁴ also mean that this phenomenon is even more significant here than in many other international cities.

In terms of assessment this means that substantial spatial variation in NO_2 concentrations (a factor of two would be a reasonable rule-of-thumb) should be expected across any urban area. Consequently, in terms of an assessment, one needs to decide whether to focus on the peak values, and accept that such a value will be conservative for most other receptors, or the average, or whether assessing the variability is required. In most areas of New Zealand, the substantial margin of compliance with the NES and AQG will mean that the simplicity of the conservative approach is acceptable. There are two potential weaknesses of this approach. First, it requires the location of peak concentrations to be identified, or at least estimated *a priori* of monitoring, and, when adopted, it must be accepted that the assessment may fail to identify the peak sites. Second, receptors near busy roads or intersections may have highly localised concentrations close to, or exceeding the NES or AQG which, if extended to the whole domain, may lead to excessive and misleading conservatism. These problems were highlighted by our study in which the highest monitored NO_2 concentrations were not alongside the subject motorway, but at an intersection nearly 1 km from the motorway. Here the mean NO_2 concentration was 60% higher than the nearest other site, only 180m away.

⁴ And probably elsewhere in New Zealand, although there is minimal data at present to confirm this.

In health risk and exposure applications, it is more common to require a moderately accurate description of concentration variation. Peak sites may be less relevant, especially if they do not coincide with exposure-relevant locations (eg intersections or non-residential sites).

Consequently, most measurements of NO₂ in urban areas are influenced by nearby roads to some degree. This makes the definition and use of the term 'background' imprecise. In our study, mean concentrations of NO₂ at locations more than 200m from the motorway or the other busy road in the area varied from 12 – 19µg m⁻³ due to the varying influence of local roads. Therefore one could reasonably define the background level as the minimum, average or maximum of this range. This clearly has implications for the placement of a monitoring site, or the use of data from an existing monitoring site. In the case where a major road could influence a monitoring site, the influence can be broadly estimated using the RCM. It is important to note, however, that the RCM was designed to predict gradients of conserved pollutants (eg NO_x), not reactive pollutants such as NO₂. Ongoing research at NIWA is using the observational data from this study to improve the RCM for NO₂. Furthermore, our study has shown how SOSE can be used as an alternative (more complex) method for determining roadside bias in a continuous NO₂ dataset, providing suitable local meteorological data is available.

These factors, combined with the relative paucity of continuous NO₂ monitoring in New Zealand compared with PM₁₀, support the use of a network of passive NO₂ monitoring as is currently undertaken by the NZTA, complemented by more intensive passive monitoring on a campaign basis, if required, for specific projects. One of the outstanding weaknesses of the NZTA national network is that, although most sites have been chosen to represent roadside influence, the degree of that influence is unknown and likely to be inconsistent, and sites do not necessarily represent peak concentrations in any given area. This means that the spatial representativeness, and hence degree of conservatism in terms of exposure assessment, cannot be assessed. This could be addressed and improved in three ways:

- 1 Using RCM (or a similar tool) to disaggregate measured NO₂ concentrations into road component and background component
- 2 Adding background monitoring near all the existing (and historic) roadside monitoring sites (this could be on a short-term campaign basis)
- 3 Developing a spatial regression model to extrapolate the data to the surrounding neighbourhoods.

NIWA is currently exploring the third option, but this will be greatly aided by additional background monitoring.

Dense passive monitoring and LUR modelling are ideal when:

- spatial variation is likely to be present
- uncertainty in background levels needs to be addressed
- it is suspected that available monitoring data is biased by roadside location and is over-conservative
- when receptor-specific background information is required.

As shown in our study, a network of passive monitoring across an assessment domain can reveal detail not available from other methods, provide ground-truthing for any deterministic modelling and be available at low cost. Although our monitoring lasted a year, the spatial patterns identified were relatively consistent. Further research could be conducted using this and other NZTA datasets, to further explore how consistent and robust information can be gained from less than a year's worth of data.

Above and beyond the data from a passive monitoring network, a LUR model provides a means of interpolating to unmonitored locations. This is useful in the following cases:

- Monitoring points and assessment receptors are in different locations (a difference of as little as 10m might be significant).
- There are many more receptors than monitoring points.
- More spatial variation is anticipated than can be covered by a passive monitoring network (eg assessment domain is large and complex).
- It is desirable to be able to compare and assess alternative traffic or land-use scenarios.
- There may be unmonitored hotspots which need to be identified.
- Assessment is required for fine-scale population exposure assessment (eg at meshblock or at individual properties).

If a LUR model is anticipated the passive network needs to be designed accordingly. There is a rapidly expanding literature on this relationship. At present, there are no modelling tools equivalent to HAPINZ to provide background NO₂ estimates across the country. NIWA has been scoping how this could be done.

6.3.5 Recommended further research on background NO₂ for improving assessments

- Develop a robust and general method for temporal extrapolation of NO₂ data (eg from short-term data to annual mean concentrations), including compensation and correction for atypical meteorological conditions.
- Implement background monitoring campaigns to supplement the NZTA national network.
- Develop a national background NO₂ model.

6.4 Predicted effects associated with the road

6.4.1 New road(s)

Where it is required to predict the effect of a new road on roadside air quality a deterministic model is usually required. Several roadside dispersion models exist. It was not the purpose of this project to compare these models and we are therefore unable to comment on specific model recommendations. Instead we took what is probably the most commonly used example of this class in New Zealand (Ausroads) and compared its predictions to our observations. We found that model predictions were substantially sensitive to the source of meteorological and geometric input data. We also found, in a default modelling scenario using high-quality locally observed meteorological data, the model overpredicted the degree to which roadside concentrations were elevated during the daytime compared with setback concentrations was by approximately a factor of three. Consequently, we found Ausroads (coupled with VEPM) to be generally conservative. Whether this conservatism is acceptable or requires addressing will be dependent upon the assessment objectives. In general, however, we note that the uncertainty related to predictions of cumulative effects from emission-dispersion modelling was small compared with the uncertainty related to the assessment of the background contribution.

Where less temporal detail is required (eg annual mean but not peak concentrations) the RCM can provide a reasonable estimate of the contribution of a future road to local air quality.

Although not investigated within this study, in a situation in which a future road is planned within an assessment area that also includes a comparable existing road, a LUR model could be used to predict the long-term impact of the new road.

6.4.2 Changes to existing road(s)

In addition to the use of Ausroads and RCM, assessments around changes to an existing road can be enhanced by the use of empirical approaches. The formulation of SOSE assumes that temporal patterns in emissions are relatively constant. A step-change in emissions, or any variation in this consistent pattern can be applied in the modelling which will lead to a prediction of the air quality response to those changes. Similarly, a LUR model which includes traffic volume (or density) as an independent variable can be used to produce long-term predictions based on changes in the value of that variable.

6.4.3 Recommended further research on modelling of predicted effects

- Extend analysis of sensitivity of modelling outcomes to input data and parameter selection in the context of different modelling applications.
- Compare Ausroads with alternative dispersion models.

6.5 Assessment process

Based on our findings within this study we recommend the following process for aiding tool selection and use.

1 Determine objectives.

Many of the decisions to be made in conducting an assessment are related to the assessment objectives. Clearly determining those objectives at the start of the process is highly recommended. Among the questions to be addressed are:

- a Is the assessment solely interested in compliance with standards, guidelines or targets, or is the degree of compliance important? Is predicting the number of exceedences important? Is predicting the timing of exceedences important?
- b Is the assessment interested in the variation in concentrations within the domain? If so, at what scale? Census units? Streets? Individual properties?
- c Are absolute (or cumulative, all-source) concentrations required, or are estimates of the contribution from traffic sources, or a single road alone sufficient?
- d Are long-term measures (annual mean concentrations) sufficient or is shorter-term detail required (eg peak 24-hour, diurnally averaged or other concentrations)?
- e What is the desired balance between conservatism and accuracy in the assessment? Will excessive conservatism be unhelpful?
- f How important are hot spots in the assessment? How small does a hot spot become before it is irrelevant? Is a road intersection a relevant receptor?
- g Are non-residential or low/zero-exposure locations relevant to the assessment?
- h Will the assessment include (or might it include in the future) assessment of alternative scenarios?

2 Identify existing data sources:

- a A data search should determine whether air quality observations from any site are accompanied by co-located or nearby meteorological data.
- b We urge caution against using non-local NO₂ if possible, unless substantial uncertainty in the assessment is acceptable. This may be the case in a compliance-oriented assessment in a domain

with a large margin of compliance, and/or where small-scale hot spots are considered not to be relevant.

3 Evaluate existing data for bias:

- a Where monitored PM_{10} is available it may over-represent background PM_{10} if it is obtained at a location influenced by a road (for example, by ~10% in our study). Our study shows it is difficult to accurately assess the spatial extent of such an influence. However, as a first approximation our data suggests that 150m is a reasonable estimate for a major road such as the motorway studied (annual average daily traffic volumes of 120,000). Such a location may be suitable from the point of view of compliance (ie assessing the 'worst-case baseline'), but may over-represent concentrations at receptors in most of the assessment domain. A conservative approach is often unsuitable for exposure and health-risk applications.
- b How far away a monitoring site can be from a receptor before it fails to be representative of that receptor is complex and beyond the scope of this study. However, we found that assessments based on Auckland Council monitors more than 5km from our study area performed relatively poorly, and that the deterioration in performance was not linked to distance from the study area alone. Reductions in performance were linked primarily to decreased correlation implying more impact on the prediction of peaks than on average concentrations.
- c Although not studied in this project, PM_{10} monitoring sites can be substantially biased by topographically induced effects such as valley flow and sheltering. These effects tend to be more acute in meteorological conditions in which peak concentrations occur. Monitoring data from sites in shallow valleys and dips should be treated with caution.
- d Our study included a limited assessment of the impact of using off-site monitoring data to represent background NO_2 in our study area. The results showed that the use of off-site data substantially reduced assessment performance to a much greater degree than for PM_{10} . This is due to the lower spatial representativeness and higher local bias of any NO_2 monitoring site, including roadside sites in particular, and sites on the urban edge or coast (with urban sources in certain wind directions only). Although not studied systematically, we suggest that NO_2 monitoring from more than 1km away, or close to roads not included in the assessment, is probably inappropriate for roadside assessments.

4 Compensate for data bias

- a Due to the strong sensitivity of NO_2 concentrations to source proximity and strong spatial gradients, we recommend that all NO_2 data is assumed to be biased in some way, unless there are good reasons to assume otherwise. Whether (and how) this bias is compensated for depends upon whether the assessment's focus is on compliance (in which case the acceptability of conservatism might mean that the bias is acceptable), whether hot spots are of interest or whether a risk of a lack of conservatism (unmonitored sites may exhibit greater 'bias') is important.
- b In contrast to NO_2 , our study showed that the contribution of a major road to PM_{10} is quite small compared with background (~10% on average in the case of the road studied in our research). It may therefore be considered that for roads with less traffic, the bias will be smaller still and negligible in some instances (depending upon the assessment objective).
- c Where a positive bias (data over-represents concentrations in the assessment domain) is expected compensation for that bias might not be deemed necessary in the case of compliance objectives (including the bias is conservative), or for exposure objectives (where describing relative differences in concentrations across the domain is more important than absolute concentrations).

- d If it is suspected that monitored data is biased by a local road source in this way then RCM can be used to estimate the magnitude of this bias on an annual mean basis, thus subtracting the bias from observed data to estimate the background.
 - e If more temporal detail is required (important temporal variation in bias is suspected) then SOSE can be used to estimate the mean bias as a diurnal, weekly, monthly or seasonal average, depending on the scope of the data, so long as suitable nearby and simultaneous meteorological data is available.
 - f Further detail still can be determined if a continuous NO₂ dataset is accompanied by local meteorological data. In this way the degree of bias can be related to the influence of local sources on the monitoring station as a function of wind direction and wind speed.
 - g SOSE can potentially be used to fill any gaps in air quality time series, if high-quality meteorological data is available from a nearby site. However, it should be noted that SOSE in its current form is optimised for predicting average concentrations, not peaks and SOSE should not be expected to accurately estimate unmonitored peaks.
 - h In principle, SOSE can be used to temporally extrapolate a dataset to time periods outside of the training dataset, so long as the training dataset covers similar meteorological conditions and suitable meteorological data is available.
- 5 Commission new continuous monitoring:
- a When/why?
 - i. When there is reason to believe that the available PM₁₀ data/site is insufficiently representative (in space or time) of the assessment domain (representativeness is likely to be reduced at times of peak concentrations).
 - ii. Insufficient data exists to adequately compensate for bias in existing data.
 - iii. When a localised risk of NES exceedance for NO₂ is suspected and needs to be confirmed.
 - b How?
 - i. Include high-quality local meteorological monitoring if any form of modelling is likely to be included in the assessment.
 - ii. For NO₂ aim to determine whether the site represents peak concentrations, or the upper, mid or lower range of the local background concentrations.
- 6 Commission new passive/screening monitoring
- a When/why?
 - i. When substantial spatial variation is likely to be present and quantifying the variation is consistent with the objectives.
 - ii. When uncertainty in background levels needs to be addressed.
 - iii. When it is suspected that available monitoring data is biased by roadside location and could be excessively-conservative.
 - iv. When receptor-specific background information is required.

b How?

- i. Monitoring for a whole year provides a robust means of determining annual mean concentrations, but is not necessary (and indeed not common internationally) if a suitable robust method for temporal extrapolation is adopted.
- ii. Measurement sites should aim to cover the full range of anticipated concentrations across the assessment-relevant parts of the domain from maximum to minimum.
- iii. Networks should include intersections if they are believed to be relevant to the assessment (it may be considered they are not relevant for exposure).

c If the consequent development of a LUR model is likely (see below) the following should also be considered

- i. As a broad rule of thumb, a minimum of 40 measurement sites are normally used located either in a grid pattern, or in such a way that they span the range of anticipated independent variables or anticipated spatial gradients in concentrations.
- ii. The authors of this report are able to offer expert advice on the preparation of a model-suitable monitoring network.

7 Developing a LUR**a When/why?**

- i. Monitoring points and assessment receptors are in different locations (a difference of as little as 10m might be significant).
- ii. There are many more receptors than monitoring points.
- iii. More spatial variation is anticipated than can be covered by a passive monitoring network (eg assessment domain is large and complex).
- iv. It is desirable to be able to compare assess alternative traffic or land-use scenarios.
- v. There may be unmonitored hotspots which need to be identified.
- vi. Assessment is required for fine-scale population exposure assessment (eg at meshblock or at individual properties).

7 Relationship of this project to ‘Stocktake of transport related air pollution (TRAP) research in NZ’

We have reviewed the document *Stocktake of transport related air pollution (TRAP) research in NZ*, version 2’ (Kuschel and Bluett 2010) to find research needs identified by that report’s authors (in bold below, alongside need priority specified in that report) and match them up with the research being conducted in this project and the subsequent research we feel it could enable.

New Zealand good practice guide (MfE 2009) for roadside and tunnel air quality compliance monitoring (need = high)

Chapter 6 of this report provides some guidance regarding the siting and influence of the siting on air quality monitoring.

New Zealand roadside and tunnel air quality standards (need = low)

Standard-setting is usually based on an understanding of the current status or air quality and applicability of available health evidence. The RCM model, especially if combined with SOSE and LUR (or equivalents) in the near future, should be able to make extensive predictions of present and future roadside air quality across the country. The study is also aimed at interpreting the applicability of roadside health evidence to New Zealand.

Validate emission models eg VEPM (need = high)

Although our project did not consider this directly, the data is available for inverse modelling of vehicle emissions on the southern motorway.

Improved understanding and implications of traffic flow on emissions (need = medium)

Although this not a direct focus of our study, traffic speeds did systematically vary on the southern motorway during our study indicating both persistent and sporadic congestion. It is possible that data mining will provide further insight into the effect of this congestion on emissions rates.

Method to identify hot spots where the impact of transport emissions is significant (need = medium)

This research has shown the contribution of one of the country’s busiest stretches of road to roadside air quality. Our results can be extrapolated to other sections of road around the country, and we have also demonstrated how RCM, SOSE, LUR and passive monitoring can also be used to provide further information about the contribution of background sources. Partly in response to the findings from this study, additional NIWA research has led to the development of a ‘traffic impact model’ which aims to describe long-term variation in traffic-related air pollutants across the country.

Guidance on standard method for exposure assessment (need = medium)

As this is a key goal of the project the project could form a substantial input into such guidance.

Updated HAPINZ study incorporating latest population and air quality data (need = high)

Although such a project has recently been completed (Kuschel et al 2012), it will not have had the opportunity to incorporate the outputs from our project which are specifically aimed at providing improved transport-specific exposure assessments for health risk analyses, such as HAPINZ. Specifically, the Updated HAPINZ report states:

...not being able to robustly assess NO₂ exposure means that the results of this update most likely under-estimate the health impacts of motor vehicle-related air pollution.

Moreover...

*...it is likely that there are separate and independent health effects resulting from other air pollutants – in particular exposure to NO₂. There would be value in designing and undertaking a **pilot NO₂ exposure assessment** in an area already identified as being impacted by motor vehicle emissions given that roadways are also the dominant source of NO₂ emissions. The most obvious location would be in the urban area of Auckland where existing continuous NO₂ monitoring data are already available...*

Our study has now provided a substantial improvement in NO₂ exposure assessment which can enable these gaps in the HAPINZ assessment to be addressed.

Guidelines for separation distances from major roadways and mitigation options (need = medium)

Our project database and resulting models should be the ideal tools upon which to base guidance around separation distances. This is covered in brief in Longley et al (2011).

Good practice guide on how to deal with background/baseline concentrations (need = high)

Our study design was specifically intended to determine background and the data can be analysed further to provide generalised guidance for roadside sites. Furthermore the LUR implicitly determines background and can be used for baselines, as recently demonstrated for the Waterview Connection AEE. Chapter 6 in this report provides significant information about how background/baseline concentrations can be determined.

Relationship and linkages between management/mitigation of vehicle noise and air pollution (need = medium)

Although not part of this project directly, project investigator Kim Dirks has conducted noise perception surveys in the ROADSIDE study area as part of a separate study funded by the Health Research Council.

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Appendix A: Technical report

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A1 Introduction

A1.1 Layout of the technical report

The project consisted of three layers of observations (long-term passive measurements, medium-term continuous measurements and short-term mobile measurements), and four layers of modelling (emission-dispersion modelling, semi-empirical temporal modelling, land-use regression (LUR) modelling and semi-empirical spatial modelling).

This appendix first describes the study design (chapter A2) and study area (chapter A3). It then covers each of the data layers independently in the order listed above (chapters A4 to A7 cover monitoring and A8 to A11 cover modelling).

Chapter A12 synthesises the results from the various layers, compares them (where appropriate) with previous studies, and indicates their wider significance and applicability in New Zealand.

This is followed by recommendations for further research and overall project (technical) conclusions.

The mobile monitoring dataset which was captured but not used in the analysis or modelling is described in annex A.

Box and whisker plots are employed throughout the results in chapters A4 to A12, and are all formatted as described below.

Each box and whisker consists of a box bisected by a horizontal line that represents the median value. The box itself defines the lower and upper quartiles and thus contains 50% of the data. The whiskers extend to the most extreme data point which is no more than 1.5 times the size of the box. More extreme data values are suppressed. Owing to the skewed nature of the data, the outermost data points are often beyond the maximum axis value. The value of the maxima for each box and whisker (be it expressed, suppressed or beyond the axis limits), is printed directly above it. Each box and whisker also displays a red triangle at the mean value of the data. The difference between the mean and median values gives an indication of the skewedness of the data, with a greater distance between the mean and median values indicating a less statistically normal distribution. The number of data points for each box and whisker is indicated below the x axis.

A2 Study design

We developed our study design, taking into account end-user needs and by preparing a literature review.

Our study was designed around the following over-arching concepts:

- 1 To be applicable (through up/down-scaling if necessary) to a majority of urban roadside locations in New Zealand which would have exposure to a large number of people over extended periods of time.
- 2 To manage complexity by limiting the number of influencing variables, eg avoiding complex terrain, changes in surface cover, etc.

To align with these concepts a study area was required that would meet the following criteria:

- a mainly or entirely residential area
- a suburban location
- containing a busy road, ie as close to the upper range of traffic volumes found in New Zealand as practical and other limitations allowed, to maximise the 'signal' being observed and cover the range of concentrations likely to be experienced at most other locations in the country.

The core of our design was the establishment of three fixed continuous monitoring sites, one at the kerbside and two at setback sites, one on either side of the road. The three sites were to be established, as far as practicalities allowed, to roughly represent three points along a trajectory aligned with the predominant wind direction, so that one site was upwind and two were downwind. No previous continuous monitoring has been conducted simultaneously on two sides of a motorway in New Zealand. We made the assumption that there was no source or sink between the upwind and downwind sites other than the road source. This imposed a logistical requirement to seek a location which satisfied, as near as possible, this assumption. The result of this design is that the contribution of the road to local air quality can be estimated from the difference in concentrations between downwind and upwind sites.

Continuous fixed monitoring is expensive and our site criteria meant we could not specify exactly how far the instruments should be from the road. Furthermore, we could not know in advance how the distance of the sites from the road would affect the results. We sought a kerbside site which was as close to the road as logistically practical in order to maximise the road signal. We sought setback sites in the range 50m to 250m from the road as a result of the findings of the literature review that suggested air quality impacts from a road should be detectable up to ~150m for passive pollutants.

Relatively few people live as close as 150m to a major road. Beyond this distance, some previous studies have indicated smaller but detectable elevations in air quality, to which a much larger section of the population would be exposed. To investigate the wider significance of roadside air quality for roadside communities we included monitoring over a wider study area in our design. Coverage of this wider area was not practical using continuous monitoring. Instead, we adopted a network of low-cost passive monitoring to provide long-term spatial coverage with low temporal resolution, supported by occasional mobile monitoring with high temporal resolution.

We included modelling techniques in our project in order to:

- evaluate their performance in reproducing, generalising and explaining the observed air quality
- investigate ways in which modelling could be improved
- demonstrate and extend the use of temporal and spatial regression modelling techniques for roadside air quality assessment.

Although road traffic is known to be responsible for the emissions of many different air pollution species, we chose oxides of nitrogen (NO_x) as the primary indicator and main focus of our study. This is because:

- technology for monitoring NO_x is well developed and widely used
- road traffic is the dominant source of NO_x emissions in most urban areas
- NO_x instruments have a high sensitivity relative to the concentrations and concentration gradients observed in roadside locations.

We also included assessments of particulate matter (PM₁₀) and nitrogen dioxide (NO₂) due to the existence of national environmental standards (NES) for both, and the widespread use of both indicators in health risk analysis. Furthermore we made measurements, where practical, of carbon monoxide (CO), particle number concentrations (PNC, often used as an indicator of ultrafine particles (UFP)) and black carbon (BC) (often used as an indicator of diesel vehicles) as alternative indicators of road traffic emissions associated with adverse effects on human health.

Passive monitoring was to be conducted over a year to provide data across a full seasonal cycle, and due to its relatively low cost. During that year, continuous monitoring was planned for ~three months, the shorter period being related to budgetary constraints. We chose the autumn-winter period so as to straddle the commencement of the home heating season, which begins approximately in May and has a strong influence on particulate air quality in Auckland. Mobile monitoring was designed to represent a range of morning, afternoon, evening and night periods scattered through the monitoring campaign. As far as practicalities allowed we aimed to maximise the occurrence of simultaneous monitoring using the different modes (passive, continuous, mobile).

A3 Study area

A3.1 Study area selection criteria

Considerable attention was paid to the selection of the study area, in order to meet the requirements of the methodology and effectively constrain the variables influencing the measured and predicted pollutant concentrations. These could be meteorological, surface-based or emission source in character.

The study area needed to contain three sites roughly in line along the predominant wind direction at appropriate distances from each other and the emission source, namely a busy motorway. No other emission sources could be close by, either large point sources or small area sources. For this reason, a residential area was most appropriate, although this did not preclude the influence of domestic heating emissions which might compromise the analysis of the PM and CO data. Roadside atmospheric dispersion models tend not to permit local variations in the land cover and topography to be considered⁵ and so we attempted to find an area in which these were uniform, so that the three sites had consistent surface properties.

Ideally, the emission source should be a motorway at grade, with consistent traffic levels and real-time traffic data available for the study period to maximise the air quality 'signal' which was the subject of our study, and to reduce the influencing factors which are typically unmodelled or difficult to model.

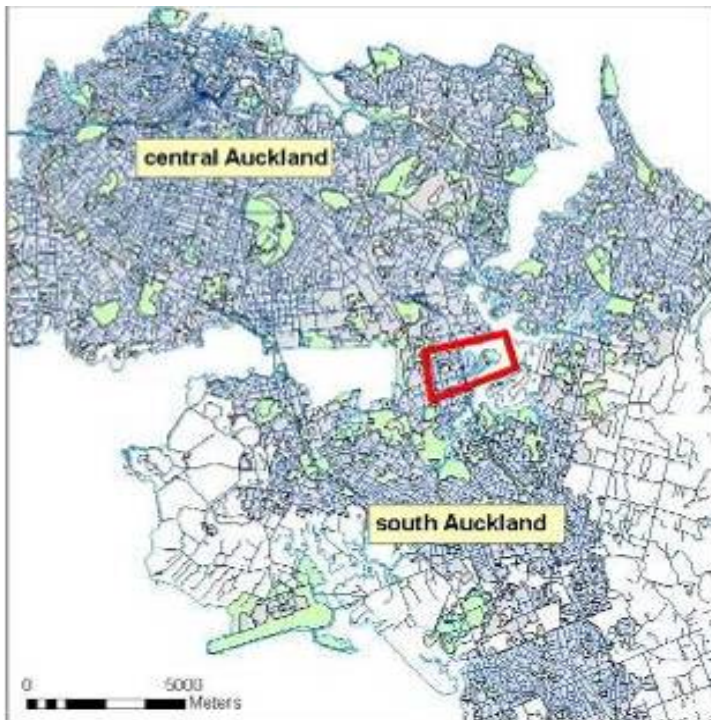
A3.2 Study area location and physical features

The study area selected was in Otahuhu East. This area is located on a narrow ~3km-wide isthmus connecting central Auckland with south Auckland (figure A3.1). The isthmus is defined by the Mangere Inlet to the west and the Tamaki River to the east (figure A3.2). The study area is relatively flat with an altitude of <20m (figure A3.3).

The study area was centred on a central reference point (used in the modelling, see the following chapters) as specified in table A3.1.

⁵ Except land-use regression models

Figure A3.1 Study area location (bold box)



Note: Shaded areas represent built-up areas and lines represent the road network

Figure A3.2 Study area location (bold box) showing relationship to local water bodies

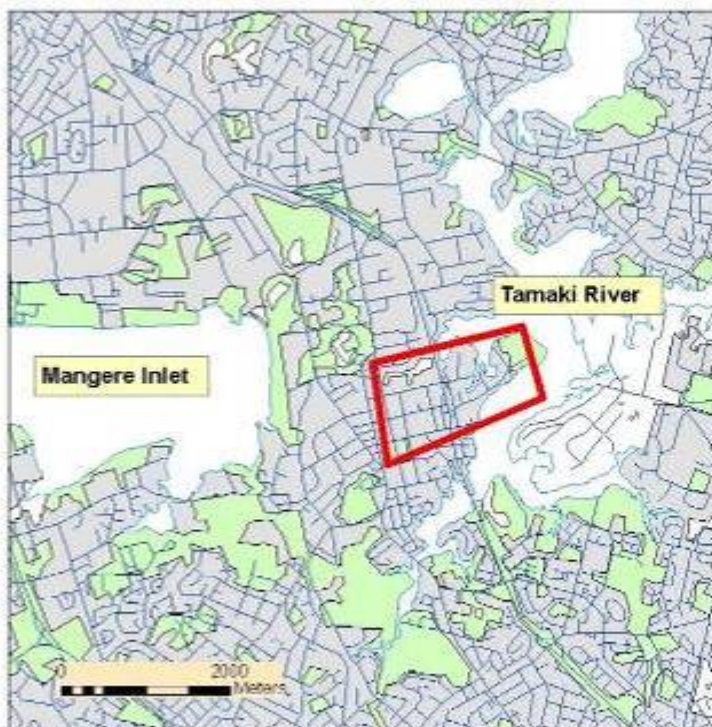
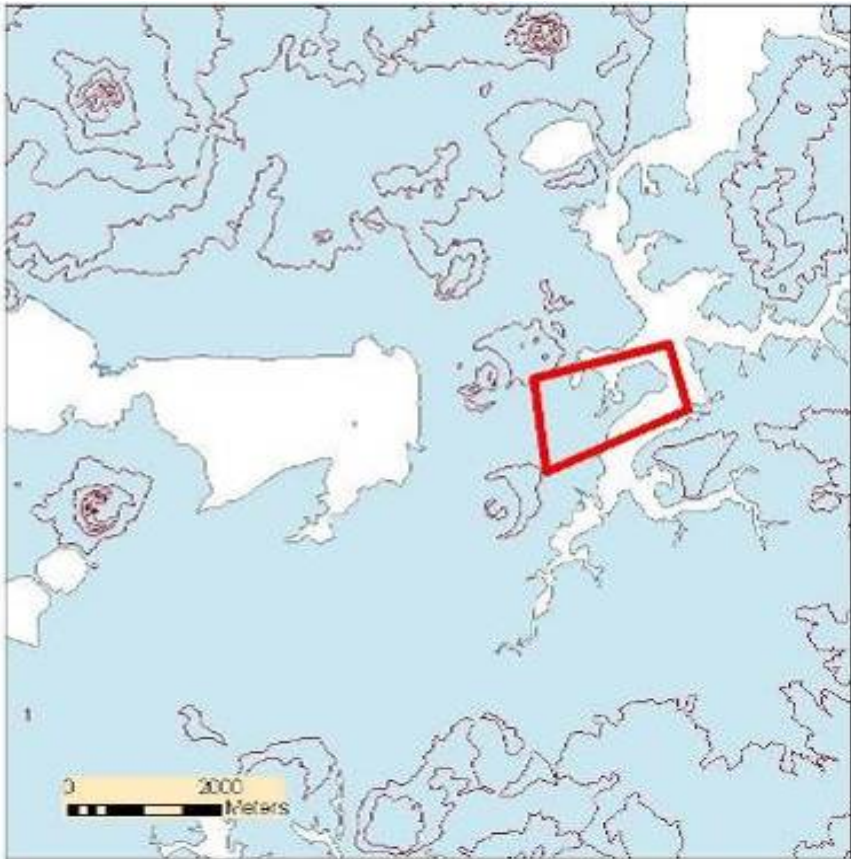


Figure A3.3 Physical context of study area location (bold box)



Note: Shade = land, white = water, lines = altitude contours in 15m steps

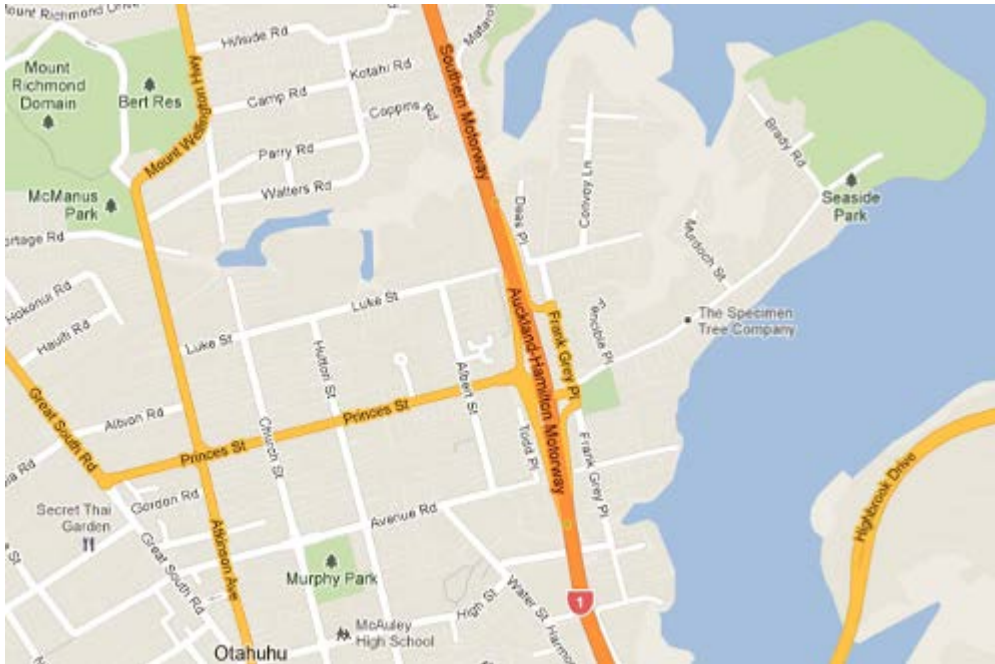
Table A3.1 Coordinates of study area reference point

New Zealand Transverse Mercator (NZTM)	1764979E	5910608N
New Zealand Map Grid (NZMG)	2675390E	6472290N
WGS84	-36.93611°	174.85252°

A3.3 Study area land-use and built environment

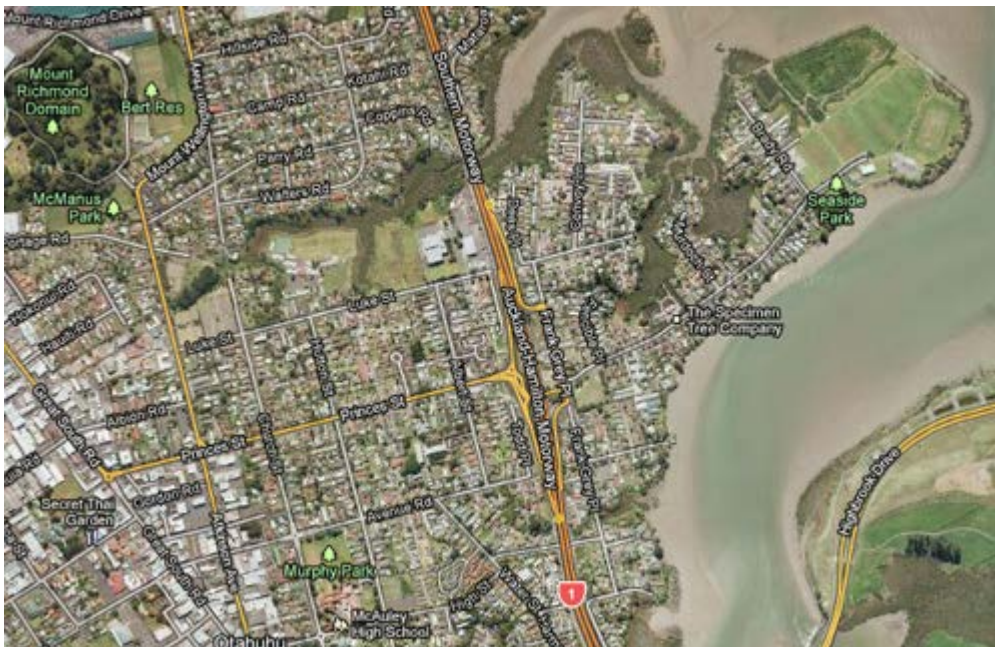
The Otahuhu East area is dissected by the Auckland southern motorway (SH1) which travels approximately north to south (see figure A3.4). The local area is linked to the motorway by a single interchange with Princes Street. Princes Street is laid out perpendicular to the motorway and links with the main alternative north-south road in the area - Atkinson Avenue. Atkinson Avenue also forms the western edge of Otahuhu’s town centre area which contains a busy local shopping area.

Figure A3.4 Road network in the study area



Note: Otahuhu's town centre lies mainly in the triangle formed by Atkinson Avenue, Princes Street and Great South Road (bottom left)

Figure A3.5 Satellite image of the study area



Note: Most of the area is residential, with commercial land use to the west

According to traffic count data collected by the NZ Transport Agency (NZTA), annual average daily traffic (AADT) on the motorway through Otahuhu was 116,000 north of the Princes Street interchange, and 122,000 south of the interchange in 2010. We estimate that only 24km of motorway in Auckland has volumes of this magnitude or greater.

According to traffic count data accessed from Auckland Transport, there are only two other roads in the area with significant levels of traffic. These are Atkinson Avenue (AADT ~18,000) and Princes Street (AADT ~13,000). On all other roads in the study area AADT is <2000.

A3.4 Study area size

The study area covered approximately 3km east-west by 2km north-south. The east-west scale (perpendicular to the motorway) was selected to:

- encompass the anticipated extent of identifiable dispersion of emissions from the motorway
- incorporate some 'background' locations beyond this area
- incorporate the potential influence of Atkinson Avenue and the Otahuhu town centre.

The north-south scale (parallel to the motorway) was selected to capture any significant impact of Princes Street due to its role as the local access link to the motorway.

A3.5 Auckland climate

A long-term wind rose for Auckland Airport, 10km south-west of the study area, is shown in figure A3.6. It can be seen that the predominant wind direction is south-westerly, with a secondary mode of north-easterly winds. However, in the case of calmer winds, which are more significant for air quality, the prevalence of south-westerly and north-easterly winds is more equal.

For the purposes of this project, observational meteorological data was sourced from four observational sites. These long-term permanent sites are listed in table A3.2 and their locations are shown in figure A3.7. Data was retrieved from the National Climate Database, maintained by NIWA.

Figure A3.6 Long-term wind rose for Auckland airport

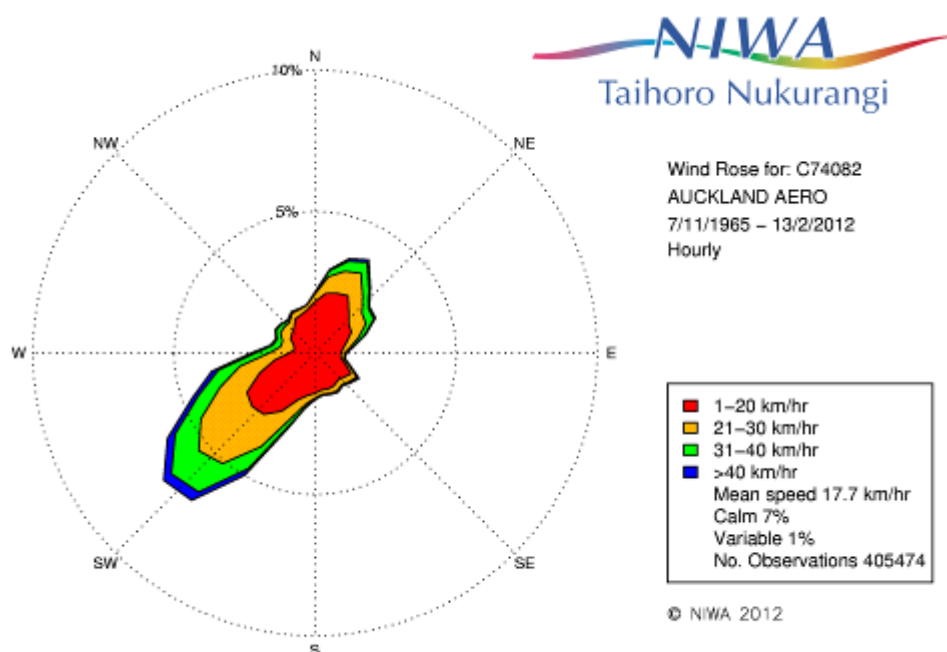


Table A3.2 Details of permanent meteorological stations used as data sources for dispersion modelling in this project

Name	Coordinates (NZTM)	Co-ordinates (NZMG)	Distance from study reference point	Mast height
Airport	1759165 5902930	2669565 6464622	9.7km	10m
Onehunga	1760436 5911538	2670853 6473228	4.7km	10m
Wiri	1766415 5904322	2676818 6466000	6.4km	10m
Mangere	1758007 5908097	2668417 6469792	7.4km	10m

Figure A3.7 Locations of the four permanent meteorological stations used as data sources in this project (Onehunga, Mangere, airport and Wiri)



Note: The main additional meteorological station (Luke Street) was set up during the main observational campaign period of the project (see chapter A5). The red box defines the study area. Contours are altitude (15m).

A3.6 Other local emissions

Other than road traffic sources, the only known emission sources in the study area were domestic heating and cooking. Domestic heating sources in Auckland are largely from wood-burning appliances, although gas and coal heating may also be found. Domestic home heating sources generally have a significant impact on PM_{10} levels between 6pm and 6am, with levels peaking around 11pm to midnight. In Auckland this signal is often indistinct except on colder nights with lower winds. The impact of cooking sources on ambient concentrations of air pollutants has not been documented in Auckland.

More distance sources included an industrial zone >1.6km west of the central reference point, and a gas turbine electricity generating station 1.7km southeast of the central reference point.

A4 Passive observations

A4.1 Overview

Passive measurements of NO₂ were conducted using Palmes diffusion tubes at 32 sites across the study area, plus an additional 28 sites in the neighbouring and similar community of Mangere. The data covered a period of one year, with two breaks. Exposure periods were two weeks during the intensive observational period (described in chapter A5) and four weeks at other times. The data was used to explore the long-term spatial pattern in NO₂ across the study area.

A second passive monitoring campaign was also conducted in the nearby area of Mangere to investigate the transferability of LUR models from one location to another similar location (see chapter A10 for more details).

A4.2 Sites and siting rationale

Passive monitoring was conducted across the study area in order to:

- provide far greater spatial coverage than could practically be achieved using continuous methods
- relate continuous measurements at distances less than 250m from the motorway to air quality at greater distances
- provide an input to LUR modelling
- provide a validation dataset for future modelling assessments.

Monitoring sites were chosen to meet a range of criteria:

- a degree of redundancy to allow for monitor losses and failures, which are relatively common for passive monitors
- avoidance of highly localised influences likely to bias the representativity of the measurements (sheltered locations, close proximity to emission sources unrepresentative of the general location, etc)
- relatively even coverage of an anticipated range in concentrations across the study area (see the 'monitoring demand' concept below).

The 'monitoring demand' concept is well established in the international literature (eg Kanaroglou et al 2005). In brief, sites were selected based on *a priori* 'best guesses' about spatial gradients in concentrations. We anticipated the following gradients:

- large-scale spatial gradients in an east-west direction due to a higher density of traffic emission sources in Otahuhu town centre to the west and the absence of sources in the Tamaki River to the east
- medium-scale spatial gradients around the creeks defining the northern, eastern and south-eastern edges of the study area, due to the lack of emission sources
- fine-scale spatial gradients around the motorway, the interchange and other significant roads.

Once the anticipated gradients were established, sites were allocated to provide higher coverage in areas of stronger gradients. We aimed to avoid locations that might give inconsistent results, such as those that were highly sensitive to uncontrollable or unobserved conditions, as set out in the siting criteria above.

This process resulted in the identification of 32 sites based on a grid system aligned with the street network. This permitted three straight-line parallel east-west gradients to be constructed, while also allowing the investigation of north-south gradients. The grid was described using a system of letters for north-south 'columns' and numbers for east-west 'rows', as shown in figure A4.1.

Figure A4.1 Location of project passive monitoring sites (diffusion tubes)

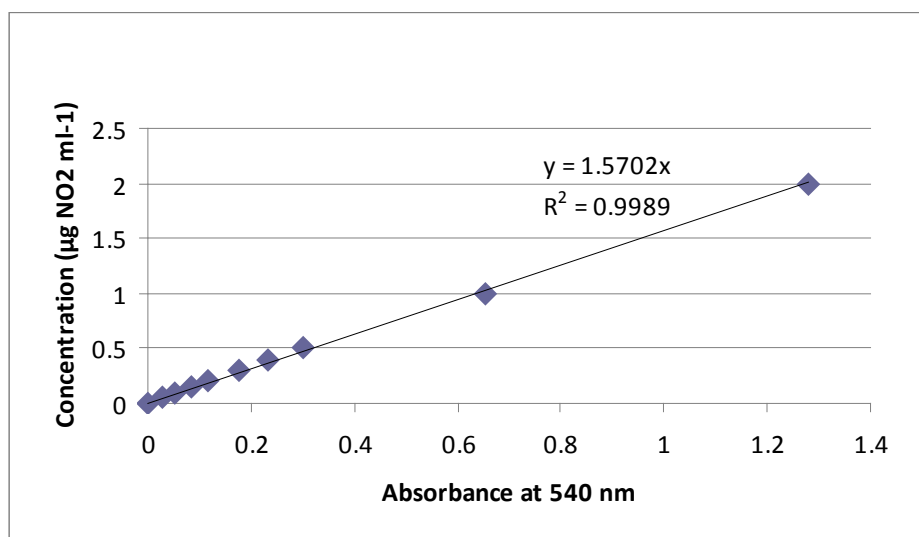


A4.3 Methods

The Palmes' diffusion tube method was employed for passive NO_2 monitoring. Palmes' tubes are widely used in air quality monitoring around the world as a cheap surrogate for expensive reference monitors.

Tubes were prepared no earlier than two weeks prior to dispatch, using the original standard mixture of 50% triethanolamine and 50% acetone ($\text{C}_3\text{H}_6\text{O}$) (Palmes 1976). When not exposed in the field, the tubes were kept refrigerated. To ensure there was no opportunity for samples to degrade, analysis was carried out within two weeks of field collection. The analytical procedure involved the preparation of a reagent comprising *N*-1-naphthylethylenediamine dihydrochloride, orthophosphoric acid (H_3PO_4), sulphanilamide ($\text{C}_6\text{H}_8\text{N}_2\text{O}_2\text{S}$) and deionised water. Prior to every analytical session, a calibration curve was derived from nine standards consisting of a stock nitrite solution and reagent (figure A4.2). If the coefficient of determination was $<.995$, the reagent was discarded and a fresh batch made. Absorbance of the calibration standards and samples was measured on a spectrophotometer at a wavelength of 540nm. Results were then calculated as units of $\mu\text{g}/\text{m}^3$ using Microsoft Excel. Preparation and analysis was carried out in a laboratory at the University of Canterbury.

Figure A4.2 Calibration curve of absorbance for concentration of NO₂ obtained for Palmes' diffusion tubes in this project



A4.4 Timing, data coverage and quality assurance

Passive monitoring was planned to take place on a fortnightly basis during the intensive observation period and then reduced to a four-week cycle. After three rounds of deployments, the fourth was deployed but erroneously not removed for a further seven weeks. This fourth deployment was considered to have yielded unreliable data, which was discarded from the analysis. Due to manpower limitations the diffusion tube surveys were not recommenced until late November. Deployment 8 commenced late and was shortened to only 15 days as a consequence of restrictions on manpower, laboratory and instrument access in the aftermath of the Canterbury earthquakes. The timing of each deployment completed is provided in table A4.1.

Table A4.1 Timing and duration of diffusion tube deployments

Deployment #	Start	End	Duration/days
1	26 May 10	9 Jun 10	14
2	9 Jun 10	23 Jun 10	14
3	23 Jun 10	7 Jul 10	14
4	24 Nov 10	21 Dec 10	27
5	21 Dec 10	19 Jan 11	29
6	19 Jan 11	16 Feb 11	28
7	16 Feb 11	18 Mar 11	30
8	31 Mar 11	15 Apr 11	15
9	15 Apr 11	11 May 11	26
10	11 May 11	9 Jun 11	29
11	9 Jun 11	7 Jul 11	28
12	7 Jul 11	3 Aug 11	26
13	3 Aug 11	23 Aug 11	20

Deployment #	Start	End	Duration/days
14	23 Aug 11	21 Sep 11	29
15	21 Sep 11	19 Oct 11	28
16	19 Oct 11	16 Nov 11	28
17	16 Nov 11	14 Dec 11	28

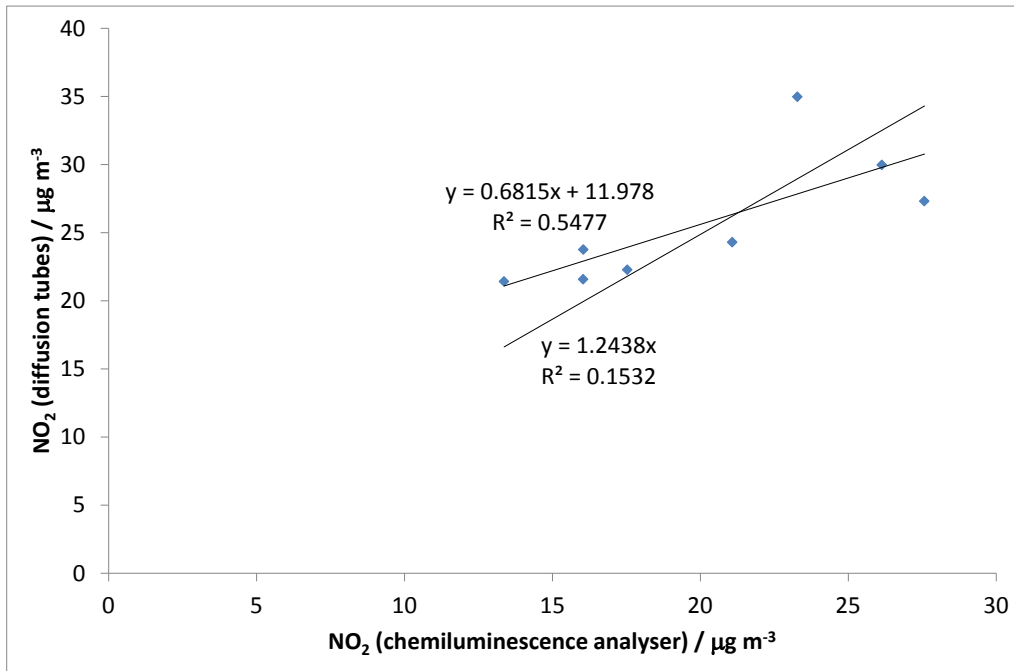
To provide a degree of quality control, NO₂ diffusion tubes were deployed in triplicates at the three fixed monitoring sites. At all other sites diffusion tubes were deployed in pairs. Field blanks were also deployed and the average of these blanks deducted from the calculated concentrations of non-blanks.

A number of tubes were lost due to vandalism. An average of three sites (9%) per deployment were vandalised, with a total of 34 site losses or 8.3% of all site measurements. To reduce losses at problem sites, an effort was made to increase the height of tube brackets on posts. No samples were lost during laboratory analysis.

Duplicates or triplicates which rendered an error greater than 30% were removed from the results. Error was calculated by means of percent relative standard deviation (RSD). For total duplicate/triplicate data, there were three (2%) and 15 (10%) cases for each of these removal criteria, respectively. RSD for duplicates and triplicates fell within the range of 9% to 13%, with a mean RSD of 11%. This was in agreement with previous findings by Heal et al (1999), where sampling occurred for a similar duration (mean RSD = 7.9%).

Co-location with chemiluminescent reference monitors has previously shown Palmes' tubes tend to either overestimate or underestimate within a range of ~10% for two to four-week exposure periods, with greater uptake more likely at unsheltered sites (Campbell et al 1994; Bush et al 2001). This further increases the degree of error associated with this type of sampling. A comparison between data from nine diffusion tube cycles and mean NO₂ recorded at the same site (API model 200 chemiluminescence analyser) rendered only a weak-moderate relationship (figure A4.3). These results gave an approximate 31% total overestimation by the Palmes' tubes compared with our reference instrument. Factors such as wind direction, wind velocity, tube exposure and the difference between active and passive sampling mechanisms would have played a role in the resulting disagreement.

Figure A4.3 Relationship between NO₂ recorded by chemiluminescence analyser and diffusion tubes at Deas Place Reserve



A4.5 Results

A4.5.1 Net spatial pattern

Data pooled from all the deployments was used to derive an estimate of the campaign-mean concentration of NO₂ for each site, as plotted in figures A4.4 and A4.5, and provided in table A4.2.

Figure A4.4 Campaign-mean NO₂ concentrations for all deployments

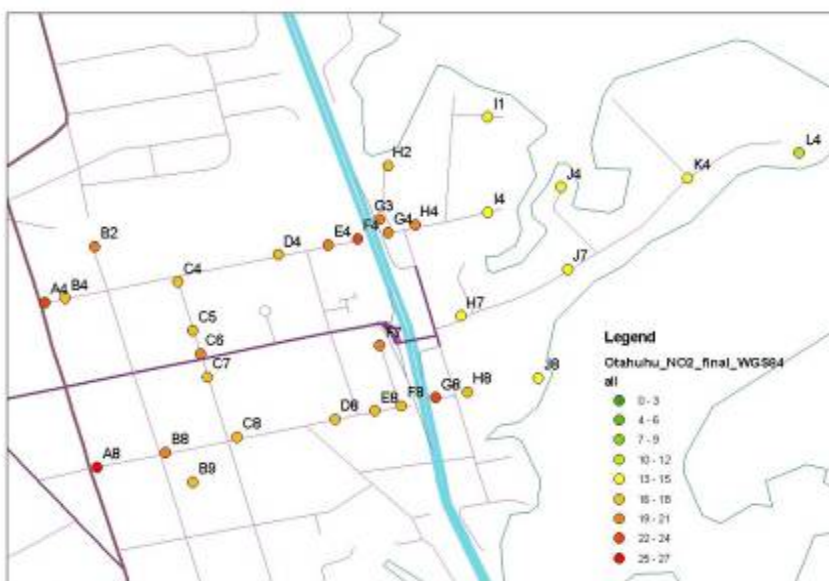


Figure A4.5 Campaign-mean NO₂ concentrations for all deployments overlaid onto a satellite image

With the exception of site A8 (bottom left, described in more detail below), the estimated mean NO₂ concentrations varied from 12 to 22 µg m⁻³, ie more than two-fold over this relatively small study area. From this data, we identified at least three spatial features:

- 1 Strong localised gradients (approximately a factor of 2) within ~200m of the motorway
- 2 Localised gradients in close proximity to the other two roads in the study area with significant levels of traffic: Atkinson Ave (on the far west of the study area) and Princes Street (running east-west through the centre of the study area)
- 3 A larger scale gradient with slightly higher concentrations (2 to 4 µg m⁻³) in the west and south compared with the north and east. This larger scale gradient may be related to general traffic density, or to differences in dispersion characteristics, possibly due to the influence of the creeks bordering the north and east side of the study area.

Table A4.1 Campaign-mean NO₂ concentrations for each site

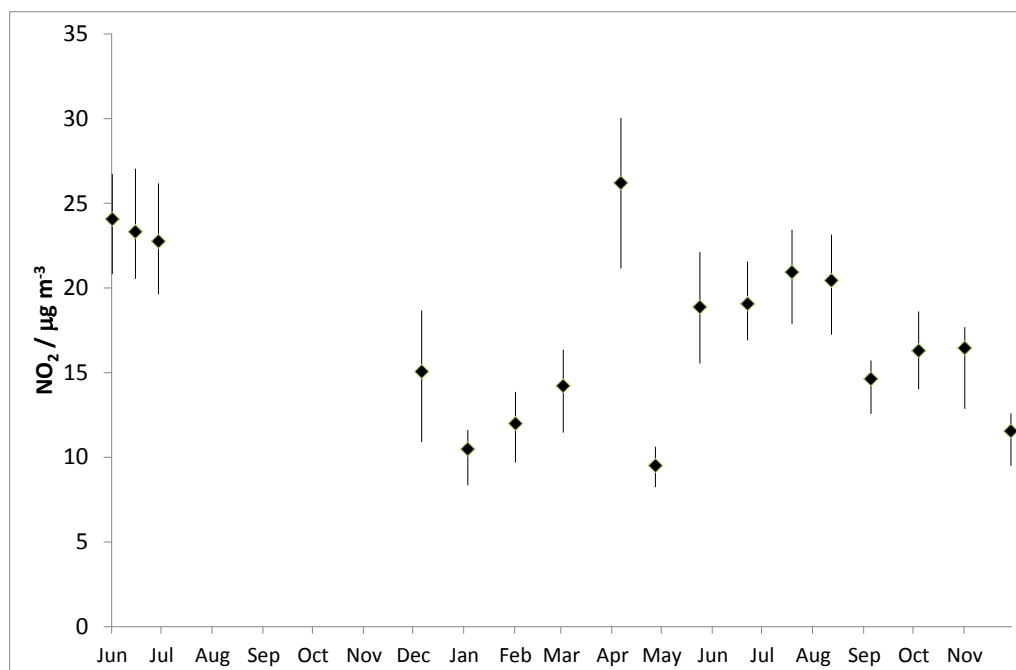
Site	NO ₂ /µg m ⁻³	Site	NO ₂ /µg m ⁻³
A4	21.1	F7	18.7
A8	30.4	F8	17.3
B2	13.5	G3	21.2
B4	16.1	G4	21.7
B8	19.0	G8	21.8
B9	16.8	H2	16.0
C4	17.7	H4	18.8
C5	15.5	H7	13.7
C6	18.8	H8	17.0
C7	17.4	I1	14.6
C8	17.1	I4	13.1
D4	15.8	I8	13.7
D8	17.0	J4	12.1

Site	NO ₂ /μg m ⁻³	Site	NO ₂ /μg m ⁻³
E4	18.1	J7	12.0
E8	17.1	K4	13.5
F4	22.6	L4	12.0

A4.5.2 Seasonal trend

Previous work (Longley and Gadd 2010) identified an underlying consistent seasonal pattern in NO₂ concentrations in Auckland. This pattern consisted of peak concentrations in mid-winter and minimum concentrations in mid-summer. The same pattern was identified at both roadside and urban background sites across the city, and was consistent with trends seen in long-term datasets collected at fixed continuous monitoring sites by Auckland Regional Council (ARC). Figure A4.6 presents the mean and range of concentrations across all our study sites according to date.

Figure A4.6 Interquartile range and mean concentrations across all sites for each deployment showing general seasonal trend and meteorologically driven deviations from the trend



Broadly, the data follows the expected seasonal trend of elevated concentrations at all sites in winter (deployments 1-3 and 10-13) and reduced concentrations in summer (deployments 4-7 and 16).

Deployments 8 and 9 stand out as anomalously high and low respectively. In an attempt to understand these anomalies we analysed wind data recorded at Auckland airport during each deployment, as summarised in table A4.3. Deployment 8 can be seen to have coincided with an extended period of low winds (8 to 15 April 2011). Deployment 9 coincided with a higher than average prevalence of easterly winds. Any impact of relatively high winds during deployment 6 is not immediately apparent.

Table A4.3 Wind conditions during each passive monitoring deployment

Deployment	Wind speed/m s ⁻¹		Prevalence of easterly winds
	Mean	Interquartile range	
1	4.2	2.1 - 6.0	52%
2	5.0	2.6 - 7.2	39%
3	3.8	2.0 - 5.3	47%
4	4.7	2.8 - 6.3	36%
5	4.5	2.5 - 6.2	45%
6	5.5	3.7 - 9.1	31%
7	3.7	1.8 - 5.4	51%
8	3.8	1.3 - 5.9	34%
9	4.3	2.2 - 5.9	74%
10	4.7	2.3 - 6.9	41%
11	4.4	2.7 - 5.7	42%
12	5.1	2.5 - 7.4	34%
13	4.3	2.1 - 5.7	37%
14	4.7	1.8 - 6.7	30%
15	4.7	2.5 - 6.7	52%
16	5.2	3.5 - 6.6	27%
17	6.4	4.7 - 8.1	44%

A4.5.3 Seasonal adjustment

We estimate that the current dataset is seasonally biased because of the uneven coverage of the data with respect to the seasonal trend discussed above. Seasonal bias in NO₂ concentrations in Auckland has previously been investigated by the NIWA research team. Seasonal adjustment factors were derived based on three to four years of passive NO₂ monitoring data collected around the Auckland region by the NZTA (Longley and Gadd 2010). By applying the seasonal adjustment factors derived at that time to the Otahuhu dataset, we estimate that seasonal bias has led to the site means (as presented in figure A4.4) overestimating the annual mean concentrations by approximately 2µg m⁻³ across all sites.

A4.5.4 Mangere campaign

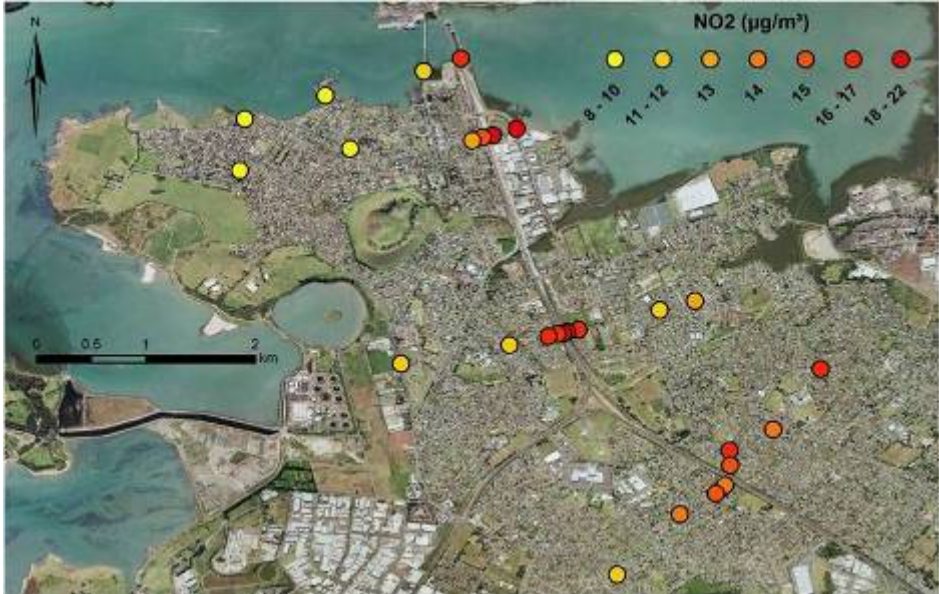
One of the purposes of the passive monitoring was to provide input 'training' data for a land-use regression model, or LUR (chapter A10). An established weakness of LUR models is their limited ability to describe spatial patterns in locations other than where their training data originated. An additional campaign was conducted in Mangere, approximately 3km to 7km west or south west of the Otahuhu study area. This additional campaign was motivated by the desire to test the similarity and geographical transferability of LUR models based on data from similar but different locations. It also provided the opportunity to explore increasing the scale of modelling by covering a larger area with a similar number of observations. The Mangere area has many similarities to Otahuhu East, including being a largely flat, largely residential area, bisected by a motorway (the Auckland south-western motorway, SH20). A network of 28 sites was established around four transects of the motorway. Sampling began on 21 December

2010, with all subsequent deployments being simultaneous with the Otahuhu deployments. In total, 50 weeks of sampling were conducted.

Table A4.4 Summary statistics of NO₂ measurements in Mangere

Mangere						
Site	Mean	Median	St Dev	Min	Max	Missing data
MA1	16.5	16.1	5.8	8.3	31.2	0
MA2	14.9	14.8	6.8	4.6	32.2	1
MA3	16.7	16.0	6.2	8.7	33.3	0
MA4	16.4	14.5	7.0	9.9	30.8	5
MA5	14.1	13.4	6.3	7.9	31.1	1
MA6	11.9	9.9	4.9	7.9	22.5	4
MA7	13.9	14.0	6.7	4.4	31.7	1
MA8	11.9	10.4	5.2	6.6	25.4	1
MA9	12.1	11.5	4.2	7.5	21.8	3
MB0	14.3	13.0	6.9	8.0	29.6	5
MB1	12.4	10.8	5.3	8.1	24.1	5
MB2	16.7	15.7	5.6	11.5	32.0	1
MB3	18.5	18.1	6.2	11.4	33.3	0
MB4	23.4	20.1	12.5	12.2	41.1	9
MB5	15.7	13.8	7.5	8.8	37.5	1
MB6	16.9	16.7	8.2	8.5	39.3	2
MB7	10.1	9.4	2.7	7.6	14.6	7
MB8	11.7	11.0	5.3	7.3	26.1	0
MC3	16.3	15.9	5.0	9.5	24.3	1
MC4	18.4	19.8	4.6	10.5	26.3	0
MC5	15.0	12.0	8.3	9.1	39.3	1
MC6	13.2	10.9	7.9	6.7	36.5	1
MC7	9.1	8.0	5.1	3.9	23.8	0
MC8	7.6	6.0	5.6	2.7	25.1	0
MD5	15.0	10.7	9.8	9.2	42.5	2
MD6	11.8	10.0	7.4	6.6	33.9	1
MD7	9.8	9.0	6.7	4.0	30.7	0
MD8	8.9	7.2	5.5	3.7	24.9	0

Figure A4.7 Campaign-mean NO₂ concentrations for all deployments in Mangere overlaid onto a satellite image



A5 Full continuous observations

A5.1 Overview

Continuous measurements of the basic meteorological parameters, PM_{10} , NO_x , NO_2 , CO and O_3 were conducted at three sites, one to the west and two to the east of the Auckland southern motorway during autumn, winter and spring 2010. Not all sites were operating at all times, nor were all parameters measured at all times due to logistical and resource constraints, and instrument failures. For completeness, this chapter describes the summary features of the whole dataset, whereas the subsequent chapter (A6) describes an ‘intensive observational period’ (IOP) defined as a period during which all sites were reporting all parameters. The IOP has been defined to permit comparison between sites in accordance with the core objectives and study design concept. The full dataset will be of value for other purposes and analyses.

A5.2 Sites

The main observational campaign began on 2 April 2010 and finished on 29 September 2010. Fixed monitoring sites were installed at three locations, one on the west of SH1 and two on the east side, as illustrated in figure A5.1. The sites were:

- on undeveloped land at 38 **Luke Street**, ~250m west of SH1
- on **Deas Place Reserve**, immediately adjacent to the southbound Princes Street off-ramp of SH1
- in the rear yard of a private property - **25 Deas Place** ~150m east of SH1.

Within the project these sites took on different ‘roles’ depending on wind direction, as summarised in table A5.1.

Table A5.1 Designations of the role of each fixed monitoring site

Wind direction	Site		
	Luke Street	Deas Place Reserve	Deas Place
Westerly	Background	Kerbside (downwind)	Near-setback
Easterly	Far-setback	Kerbside (upwind)	Background

Figure A5.1 The three fixed monitoring sites (boxes), from left to right (west to east): Luke Street, Deas Place Reserve and 25 Deas Place



Figure A5.2 The kerbside monitoring station on Deas Place Reserve



A5.3 Methods

At each of these three sites, measurements of CO, NO, NO₂, NO_x and PM₁₀ (plus O₃ at Deas Place Reserve and Luke Street) were conducted in accordance with the *Good practice guide for air quality monitoring and data management* (MfE 2009). Furthermore each site had its own independent meteorological observations, including meteorological masts at heights 9.5m at Luke Street, 60m at Deas Place Reserve and 7.5m at 25 Deas Place. In our analysis the wind data was not adjusted for these differences in height.

Table A5.2 Instruments designated to fixed-point monitors

Contaminant	Instrument
CO	API model 300 analyser
PM10	Thermo FH62C14 beta attenuation monitor
NO ₂ , NO	API model 200 chemiluminescence analyser
O ₃	API model 400 analyser
Wind	Vector A101M & W200
T & RH	Vaisala 50Y
Solar radiation	Licor L200
Rainfall	Ota tipping bucket 0.2mm tip

A5.4 Timing, data coverage and quality assurance

Table A5.3 details the timeframe of the campaign. Commissioning and decommissioning of the three fixed sites was staggered for logistical reasons. All three sites were operational from late May until early August 2010 (although with some gaps due to technical failures, as noted in table A5.3). This was the IOP during which passive and mobile monitoring was also conducted (see annex A).

The raw data from the CO, NO_x, O₃ monitors and the beta attenuation monitors, as well as from the meteorological sensors was recorded on Campbell CR10 data loggers. The data was downloaded from the loggers via cell phone telemetry and checked each working day. Data was collected over a 10-minute averaging period.

Instrument calibrations were performed on-site at the start and end of the campaign. Instrument intercomparisons were conducted after the campaign.

The CO monitor at Deas Place Reserve reported a fault very early in the campaign. The fault could not be diagnosed at the time. During the quality assurance process the fault was diagnosed but it was determined that the data recorded did not meet the quality standards required, nor could it be corrected. Consequently, we are unable to report any CO data from the kerbside Deas Place Reserve site.

Table A5.3 Timing of fixed site monitoring by parameter

Site	Parameter	Logging began	Logging ended	Data gaps
Deas Place Reserve	Meteorology	1 Apr	29 Sep	None
	NO _x	20 Apr	25 Sep	None
	PM ₁₀	2 Apr	29 Sep	None
	O ₃	6 May	17 Sep	None
Luke Street	Meteorology	7 Apr ^(a)	24 Aug	None
	NO _x	27 Apr	24 Aug	None
	PM ₁₀	9 Apr	24 Aug	None
	CO	27 Apr	24 Aug	None
	O ₃	19 Apr	24 Aug	None
25 Deas Place	Meteorology	11 May	6 Aug	None
	NO _x	14 May	3 Aug	10-30 Jun

Site	Parameter	Logging began	Logging ended	Data gaps
	PM ₁₀	27 May	4 Aug	20–30 Jun
	CO	14 May	3 Aug	None

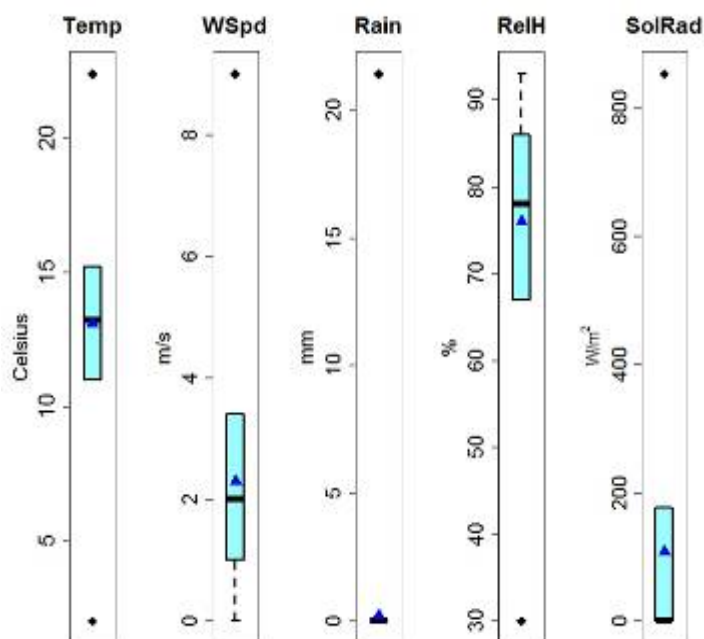
(a) Wind direction from 22 April

A5.5 Meteorological conditions during the campaign

Figures A5.3, A5.4 and A5.5 summarise the meteorological conditions during the campaign. A general preponderance of south-westerly winds was observed, consistent with the long-term climate (figure A3.6). Some deviation can also be observed between the three local sites, especially Deas Place Reserve (kerbside), at which a higher frequency of weak westerly winds was observed.

Rainfall was observed during 658 hours, or 15% of the total dataset.

Figure A5.3 Meteorological summary statistics of hourly average data for the full campaign (based on Deas Place Reserve)



Note: Boxes represent interquartile range

Figure A5.4 Windroses for the campaign meteorological sites during the period of continuous observation

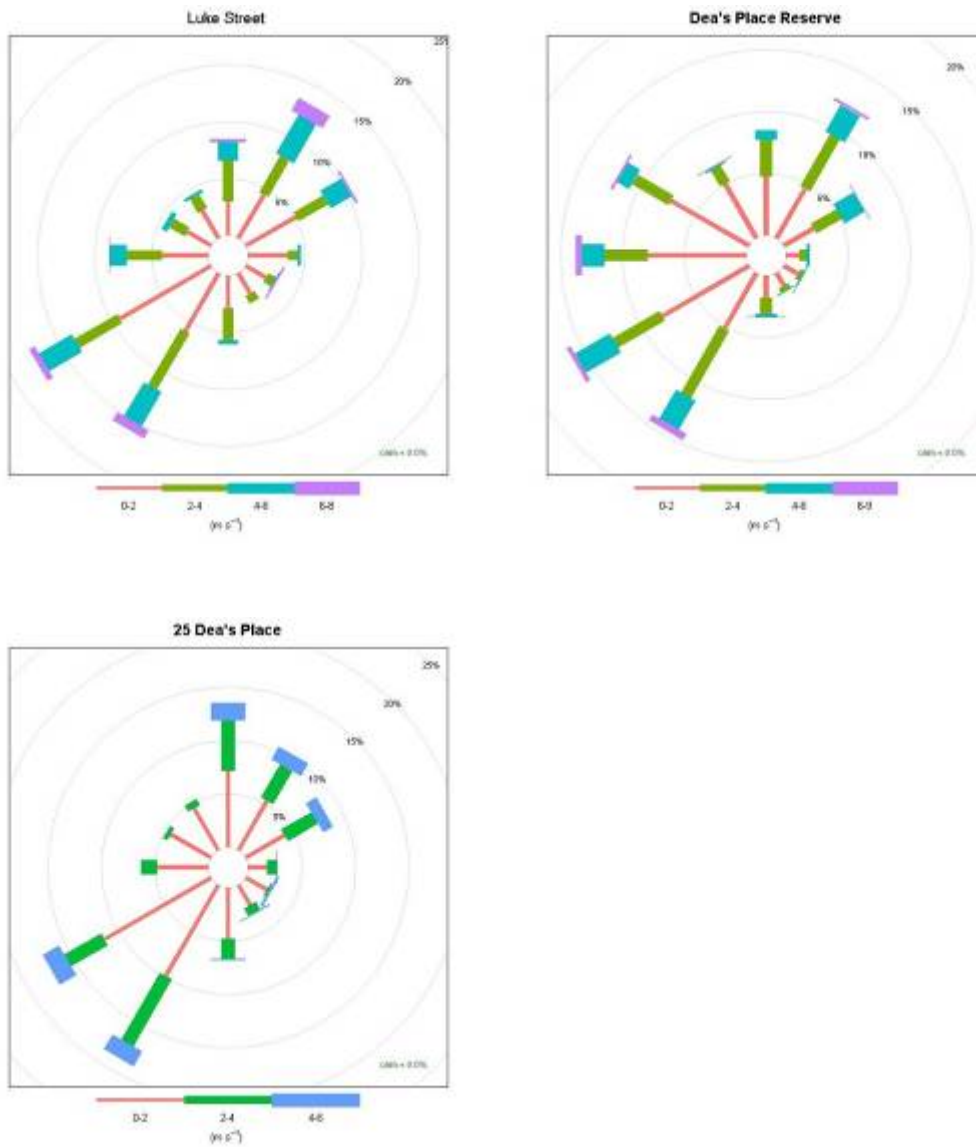
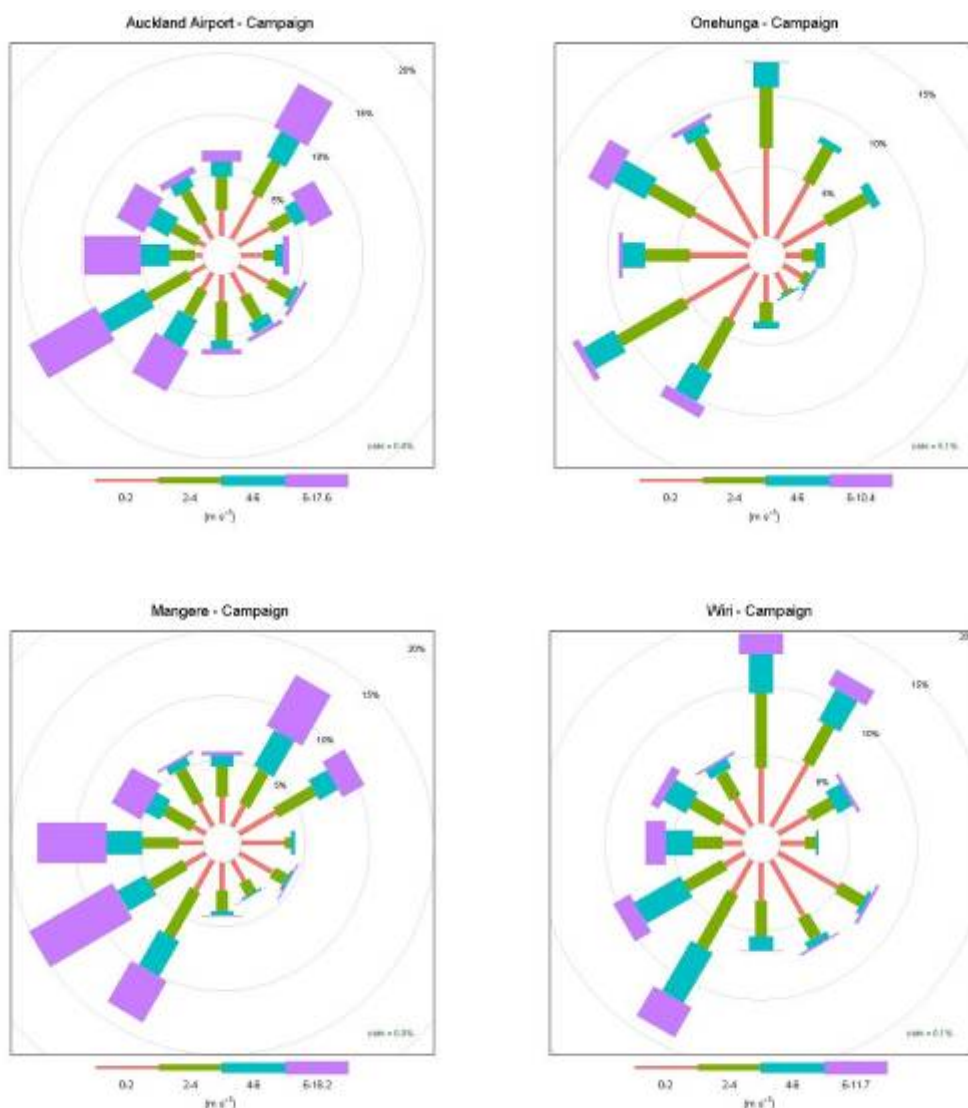


Figure A5.5 Windroses for the off-site permanent meteorological sites during the period of continuous observation



A5.6 Summary statistics of hourly air quality concentrations

Table A5.4 Summary statistics of hourly average concentrations ($\mu\text{g m}^{-3}$) for the full campaign at Deas Place Reserve (kerbside)

Parameter	NO _x	NO ₂	PM ₁₀	O ₃
n	3784	3784	4305	3561
Mean	158	24.3	17	23.5
Min	0	0	0	0
Median	114	25.0	15	21.4
Max	1228	86.4	94	67.3
Interquartile range	26 - 224	10.7 - 35.2	9 - 22	4.2 - 40.1

Table A5.5 Summary statistics of hourly average concentrations ($\mu\text{g m}^{-3}$) for the full campaign at 25 Deas Place (150m setback)

Parameter	NO _x	NO ₂	PM ₁₀	CO
n	1400	1400	1373	1939
Mean	92	16.6	19	0.3
Min	0	0	0	0
Median	43	16.9	15	0.1
Max	850	54.9	9	5.4
Interquartile range	12 - 124	7.2 - 24.0	9 - 25	0.0 - 0.5

Table A5.6 Summary statistics of hourly average concentrations ($\mu\text{g m}^{-3}$) for the full campaign at Luke Street (250m setback)

Parameter	NO _x	NO ₂	PM ₁₀	O ₃	CO
n	2843	2843	3294	3037	2851
Mean	84	18.4	16	29.3	0.7
Min	0	0	0	0	0
Median	34	17.1	14	28.9	0.2
Max	1024	67.9	94	92.1	8.1
Interquartile range	18 - 92	10.1 - 25.6	10 - 20	10.8 - 46.8	0.1 - 0.8

A5.7 Summary statistics of 24-hour average air quality concentrations

To facilitate comparison with the NES for PM₁₀ and the AQG for NO₂ we have calculated the daily averages for these two parameters according to the regulatory requirements (midnight to midnight).

Table A5.7 Summary statistics of 24-hour average concentrations ($\mu\text{g m}^{-3}$) for the full campaign at Deas Place Reserve (kerbside)

Parameter	NO ₂	PM ₁₀
n	134	151
Mean	24	17
Min	1	3
Median	26	16
Max	44	43
Interquartile range	16 - 32	12 - 21

Table A5.8 Summary statistics of 24-hour average concentrations ($\mu\text{g m}^{-3}$) for the full campaign at 25 Deas Place (150m setback)

Parameter	NO ₂	PM ₁₀
n	58	56
Mean	16	19
Min	1	5
Median	16	18
Max	28	44
Interquartile range	9-19	14-21

Table A5.9 Summary statistics of 24-hour average concentrations ($\mu\text{g m}^{-3}$) for the full campaign at Luke Street (250m setback)

Parameter	NO ₂	PM ₁₀
n	118	137
Mean	18	16
Min	4	4
Median	18	15
Max	30	43
Interquartile range	14-23	12-19

Figure A5.6 Time series of 24-hour average PM₁₀ for the full campaign

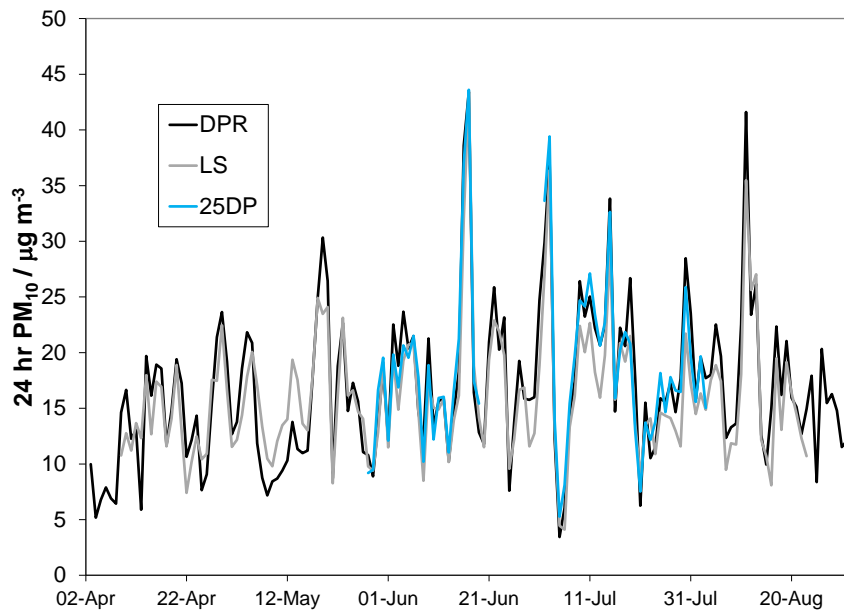
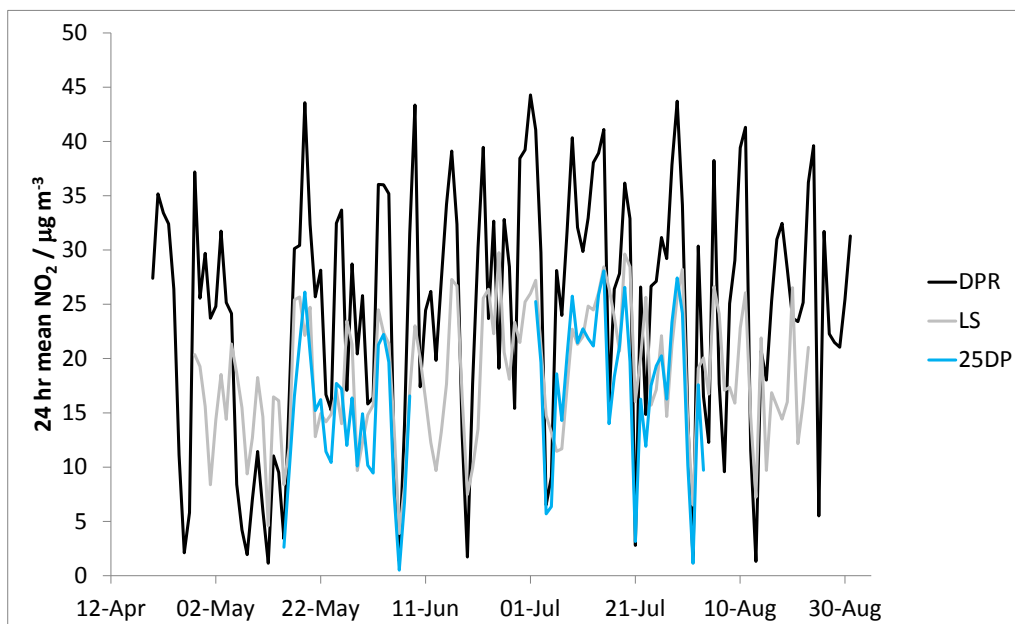


Figure A5.7 Time series of 24-hour average NO₂ for the full campaign



A5.8 Results compared to standards and guidelines

Table A5.10 summarises air quality observed during the campaign with respect to the NES for PM₁₀ and NO₂, and the AQG for NO₂. The Ministry for the Environment also maintains a guideline for annual average PM₁₀ of 20µg m⁻³. We cannot directly compare our data to this guideline as we do not have a full year's worth of data.

Table A5.10 Observed maximum concentrations (µg m⁻³) relative to national environmental standards and air quality guidelines for PM₁₀ and NO₂

NES/AQG	Limit value	Deas Place Reserve (kerbside)	25 Deas Place (100m setback)	Luke Street (250m setback)
PM ₁₀ (24 hr)	50	43	44	43
NO ₂ (1 hr)	200	86	55	68
NO ₂ (24 hr)	100	44	28	30

A6 Intensive observational period (IOP)

A6.1 Overview

This chapter repeats the analysis of chapter A5, but applies only to the IOP, ie the period from 14 May (27 May for PM_{10}) until 3 August during which all three fixed sites were fully operational. This permitted unbiased comparison between sites and the IOP dataset was used for much of the project analysis.

A6.2 Summary observed meteorological conditions during the IOP

Figures A6.1 to A6.4 show that meteorological conditions during the IOP were not significantly different from the campaign as a whole, except for a slight reduction in temperature (as the IOP was in mid-winter), and a slight reduction in the prevalence of NE winds.

Figure A6.1 Meteorological summary statistics of hourly average data during the IOP at Deas Place Reserve

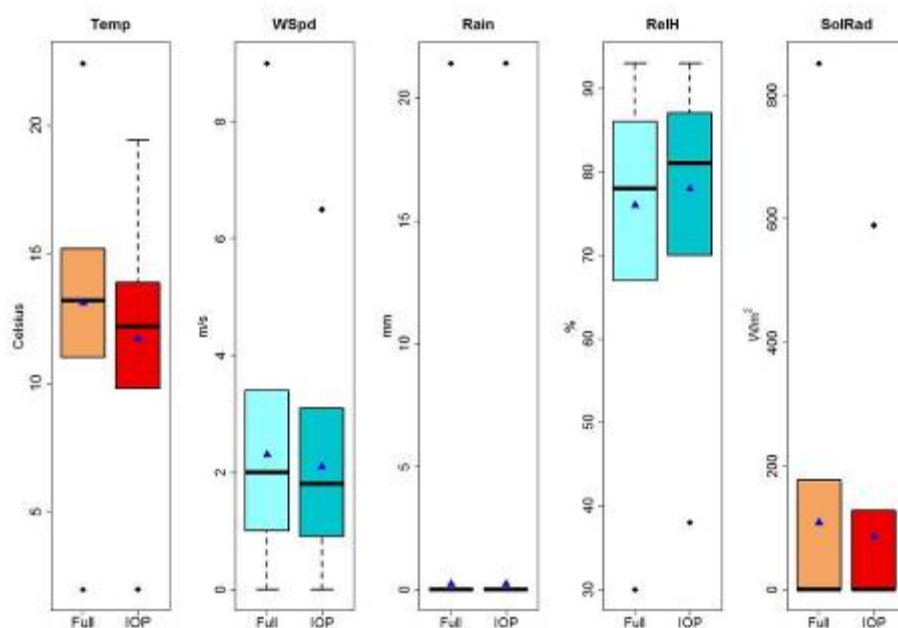


Figure A6.2 Meteorological summary statistics of hourly average data during the IOP at Luke Street

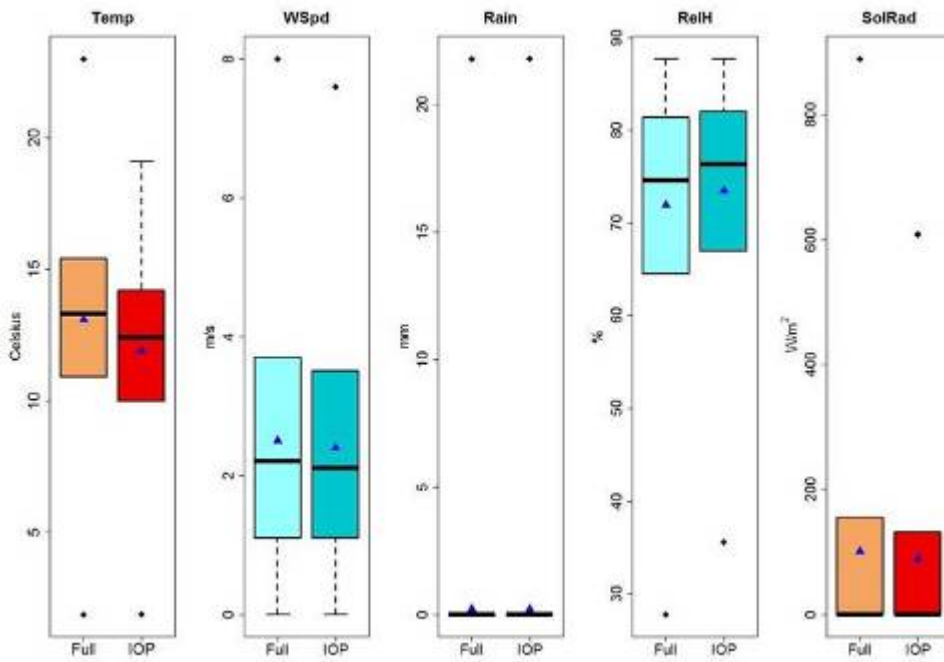


Figure A6.3 Meteorological summary statistics of hourly average data during the IOP at 25 Deas Place

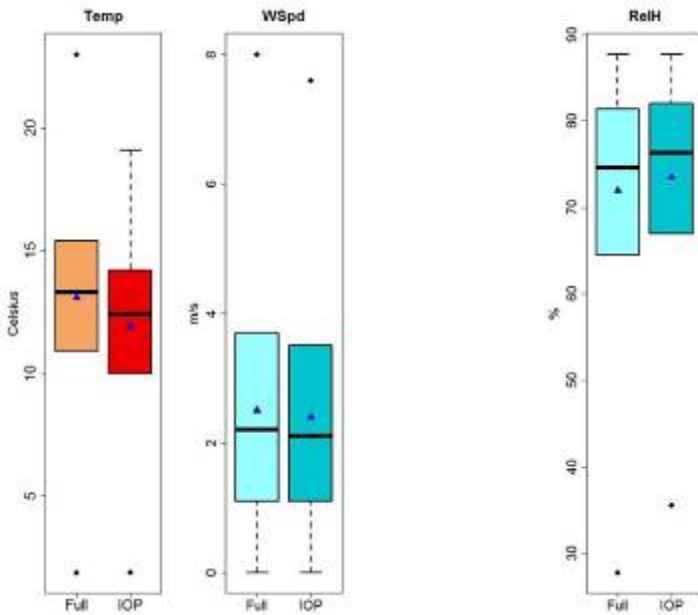


Figure A6.4 Wind roses from the campaign meteorological sites for the IOP and the full campaign

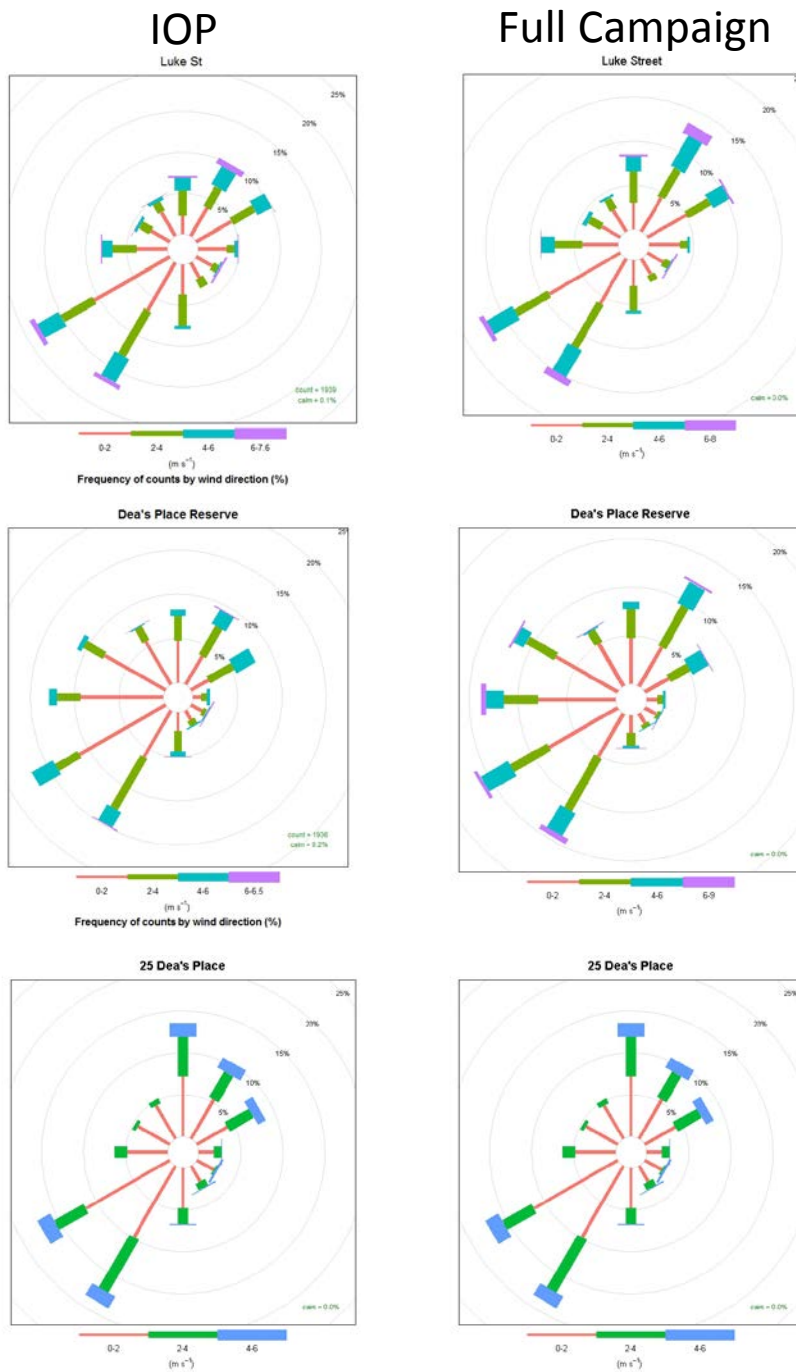


Table A6.1 Hours of observed rainfall

Period	Hours during which rainfall was observed
Full campaign	658 (15%)
IOP	335 (17%)

A6.3 Inter-site comparison of pollutant concentrations

A6.3.1 Overview

In this section, we compare the summary statistics between sites for each pollutant. All analysis is conducted on hourly averages. In the box and whisker plots, the box represents the interquartile range, the bar is the median and the triangle is the mean. Whiskers represent minimum and maximum values. Maxima are often not shown for clarity (due to highly skewed distributions), but maximum values are displayed at the top of each figure.

Table A6.2 Mean hourly concentrations ($\mu\text{g m}^{-3}$)

Pollutant	Site		
	Deas Place Reserve (kerbside)	25 Deas Place (near-setback)	Luke Street (far-setback)
NO _x	188	92	90
NO ₂	26.5	16.5	19.2
O ₃	20.6	n/a	27.5
PM ₁₀	18.4	20.6	16.9

A6.3.2 Oxides of nitrogen

Figure A6.5 Summary statistics of hourly average NO_x concentrations for the IOP

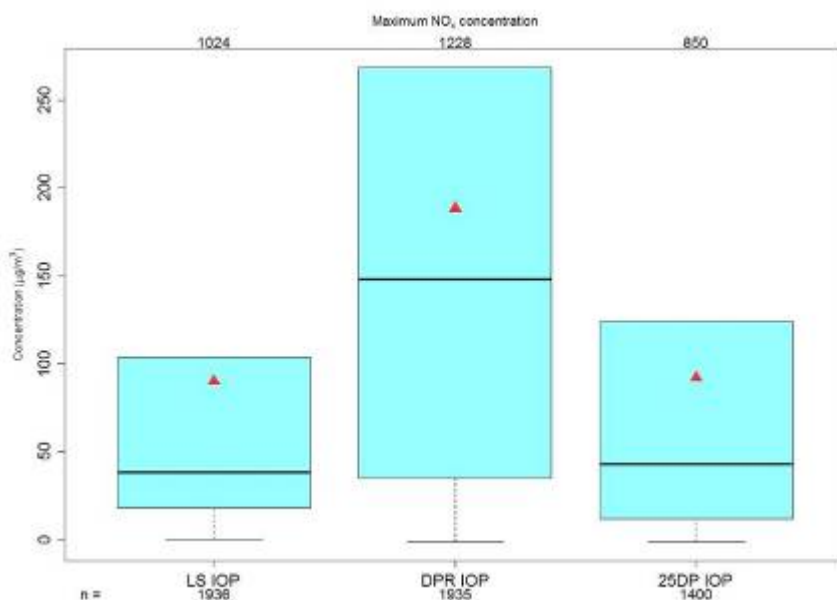
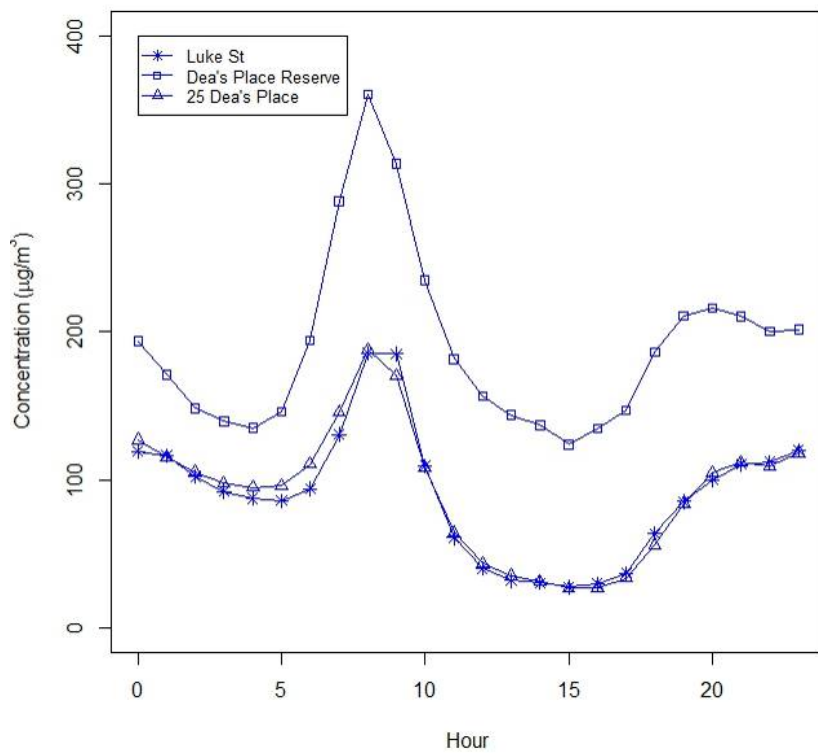


Figure: A6.6 Diurnal average hourly NO_x at the three sites during the IOP



A6.3.3 Nitrogen dioxide

Figure A6.7 Summary statistics of hourly average NO₂ concentrations for the IOP

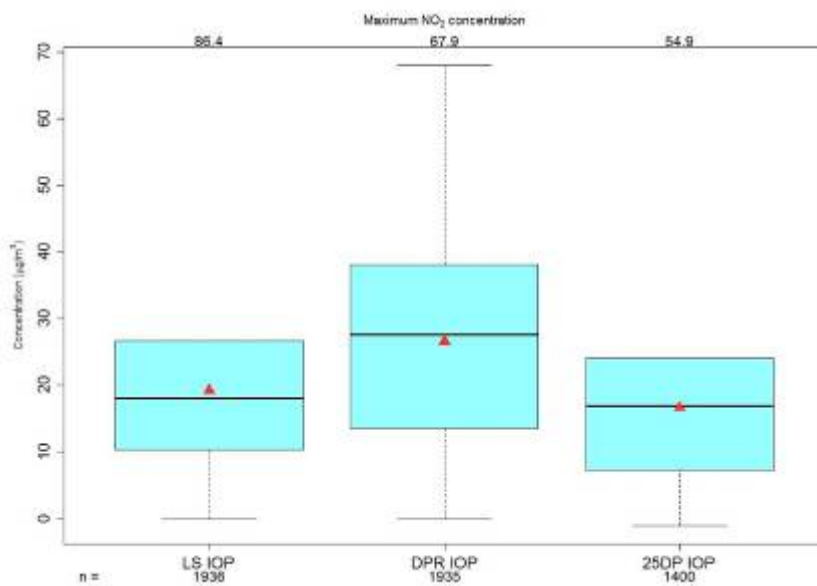
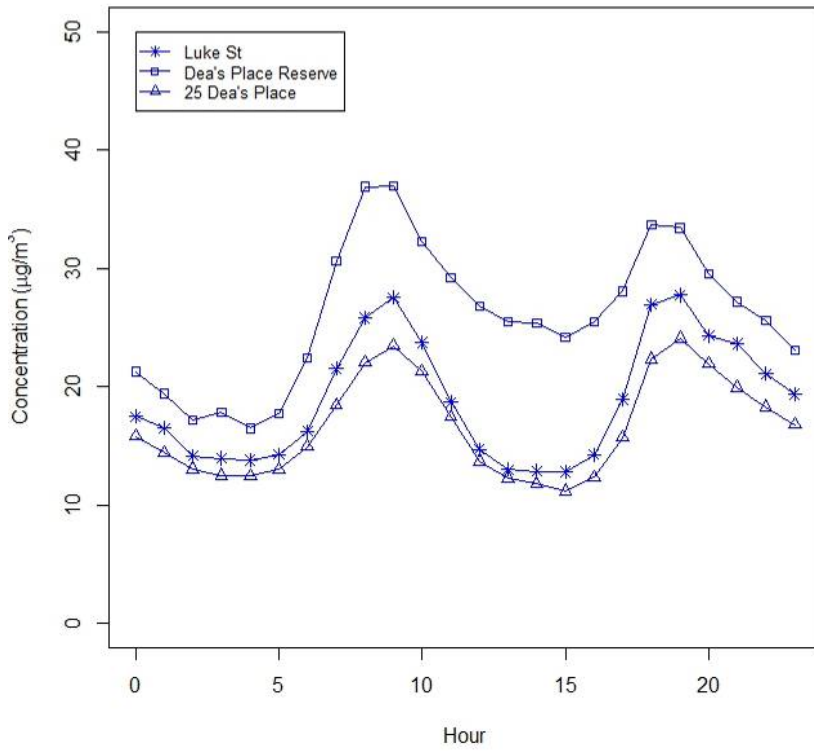


Figure A6.8 Diurnal average hourly NO₂ at the three sites during the IOP



A6.3.4 Ozone

Figure A6.9 Summary statistics of hourly average O₃ concentrations for the IOP

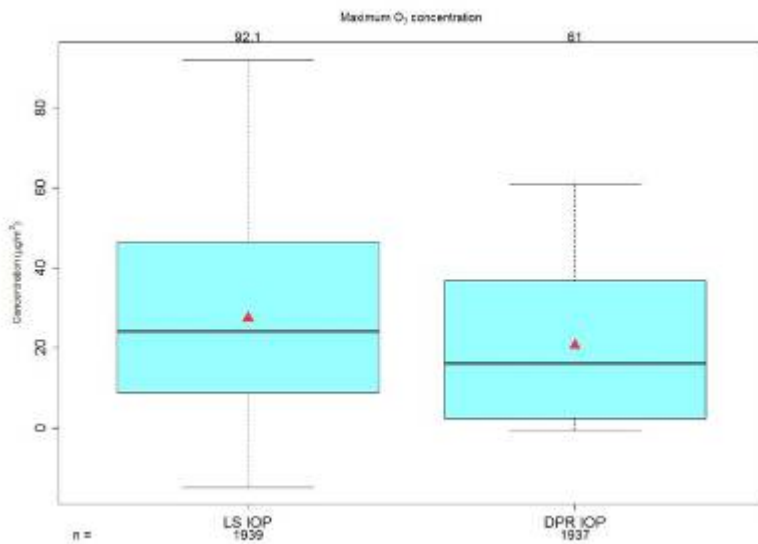
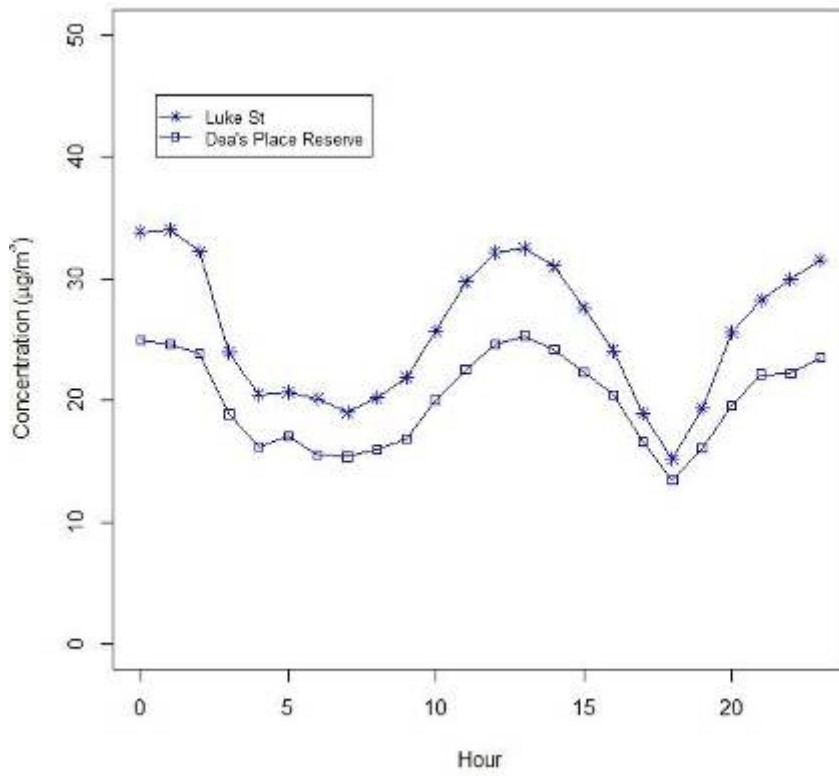


Figure A6.10 Diurnal average hourly O₃ at the kerbside and one setback site during the IOP



A6.3.5 PM₁₀

Figure A6.11 Summary statistics of hourly average PM₁₀ concentrations for the IOP

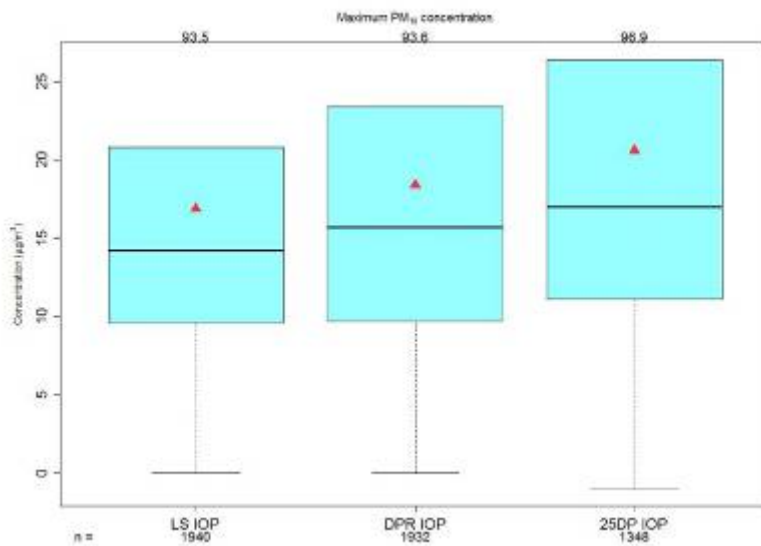
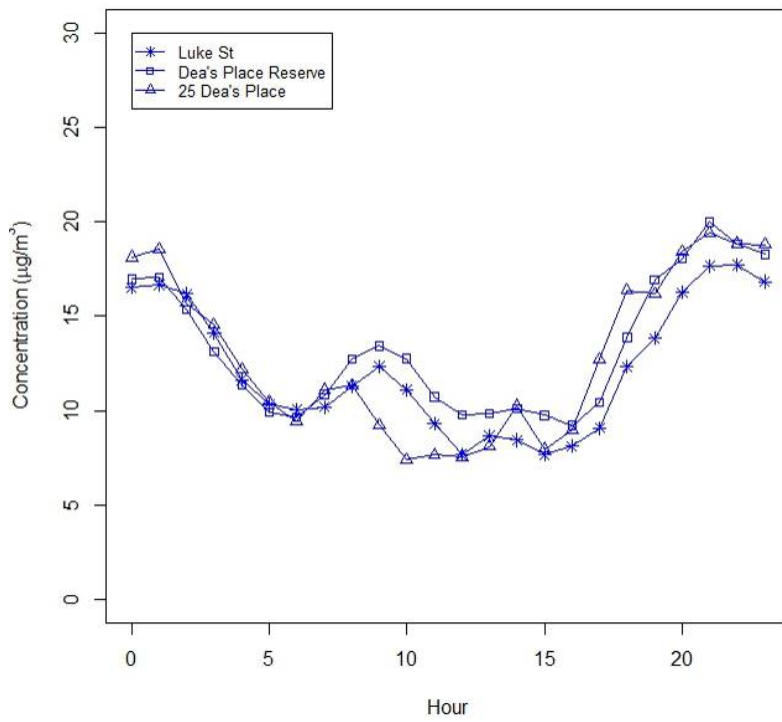


Figure A6.12 Diurnal average hourly PM₁₀ at the three sites during the IOP



A7 Interpretation of observational results

A7.1 Average contribution of the motorway to roadside concentrations

A7.1.1 Contribution of the motorway to roadside concentrations – method

Key to our study design was the measurement of the same parameters using the same instruments simultaneously both upwind and downwind of the motorway. To estimate the contribution of the motorway to roadside concentrations, we made the assumption that the only significant emission source common to the three monitoring sites was the traffic on the motorway. The motorway contribution was then equal to the difference between upwind and downwind concentrations.

The following analysis also relied on all three sites reporting valid data, ie during the IOP only. The IOP covered the periods of 14 May to 10 June and 30 June to 3 August for NO_x , and 27 May to 20 June and 30 June to 4 August for PM_{10} .

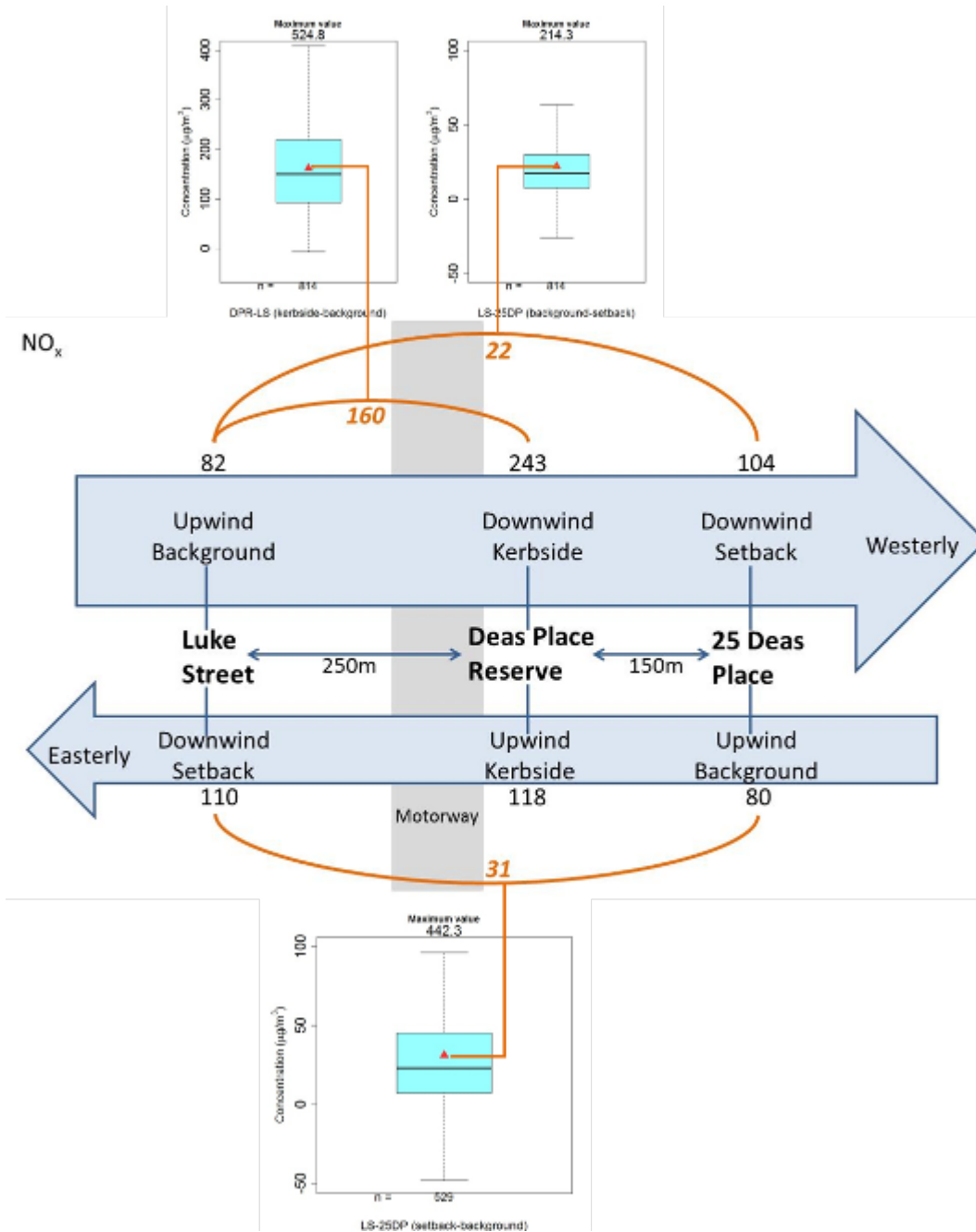
In order to qualify the up/downwind location of a site, data was segregated into westerly and easterly winds. To qualify for inclusion in the 'westerly' subset, we only selected data for which all three meteorological sites reported wind directions in the range 180° to 330° . Similarly data was selected for the 'easterly' subset only if all three sites reported wind directions in the range 0° to 150° .

The three sites performed different roles depending on whether the wind was from the west or east. In westerlies, Luke Street was the background site, Deas Place Reserve was directly downwind of the motorway source and 25 Deas Place was downwind at a distance of approximately 150m. Thus in westerlies the difference in concentration between Luke Street and Deas Place Reserve was the contribution of the motorway. The difference in concentration between Luke Street and 25 Deas Place represented the contribution of the motorway after 150m of dispersion. In easterlies, 25 Deas Place was upwind – the background site, Deas Place Reserve was directly upwind of the motorway source and Luke Street represented the contribution of the motorway after 250m of dispersion.

Figures in the following sections show the mean concentrations at each site under these two conditions and the statistical distribution of the incremental concentrations between the sites, to show the range of contribution the motorway can make. These comparisons were made for NO_x , NO_2 and PM_{10} .

A7.1.2 Contribution of the motorway to roadside concentrations of NO_x

Figure A7.1 Box plots of difference in NO_x concentrations (µg m⁻³) between upwind and downwind sites in westerly winds (above) and easterly winds (below). Central panel depicts mean concentrations and differences between the three sites



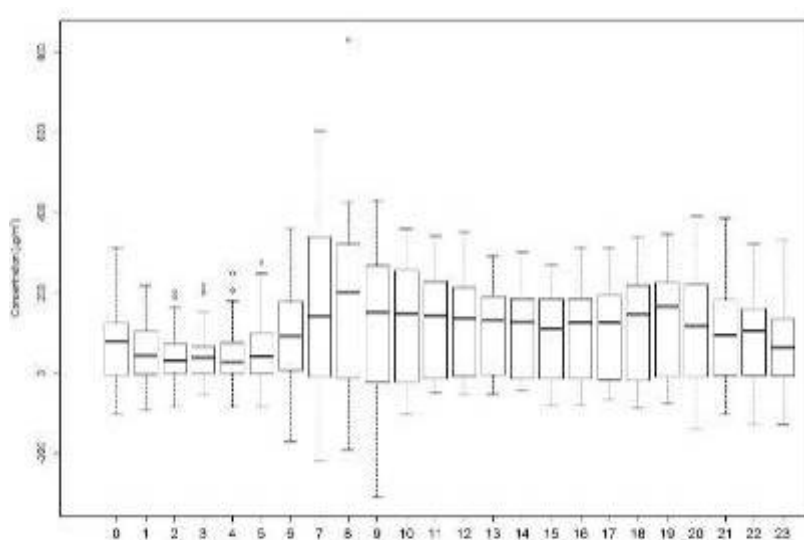
A7.1.2.1 Net effect (all wind directions)

When all IOP data was considered together, mean NO_x concentrations at Deas Place Reserve were approximately double those at both Luke Street and 25 Deas Place.

Table A7.1 Mean hourly concentrations of NO_x at the three sites for the whole IOP, including the difference between sites

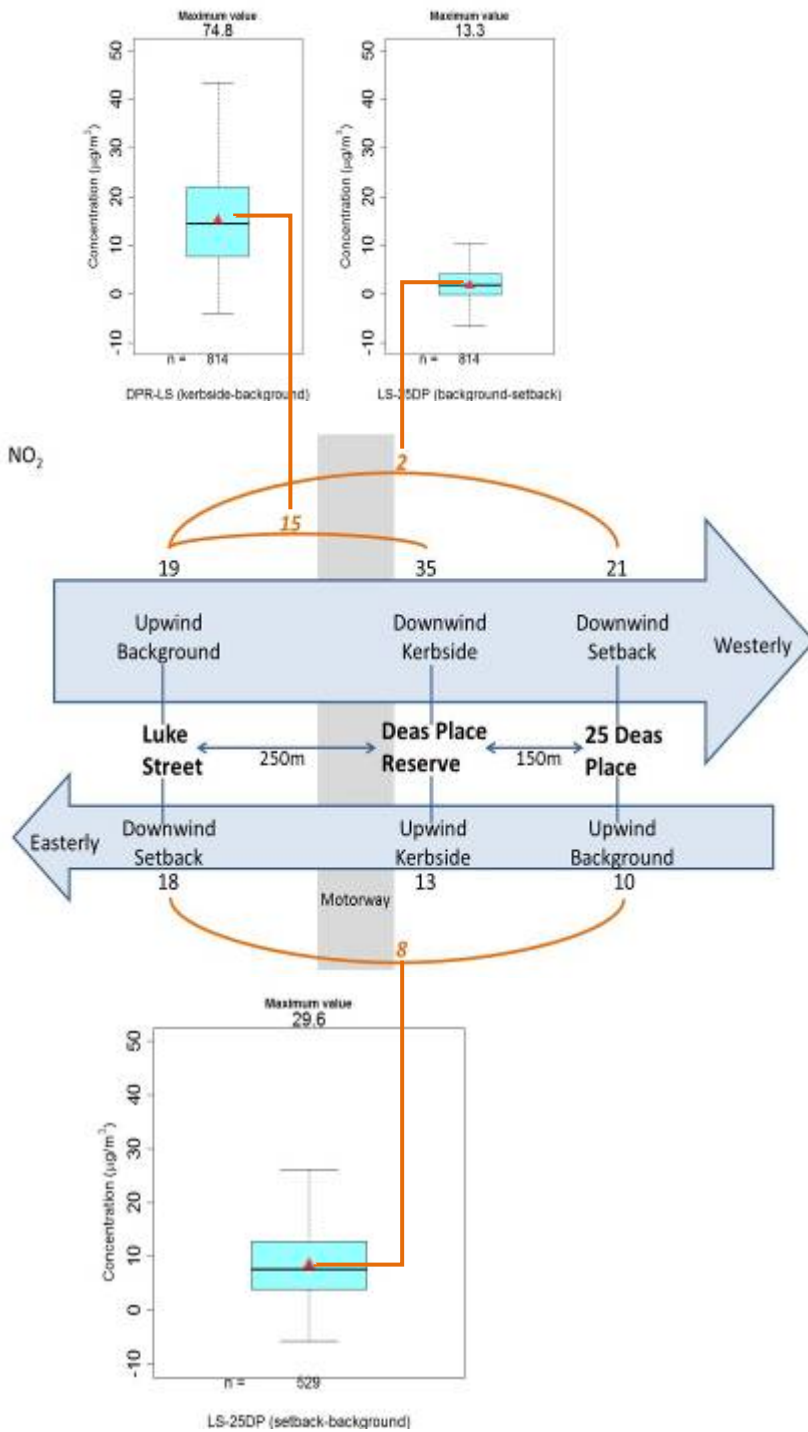
Site	Mean NO _x /μg m ⁻³
25 Deas Place	92
Deas Place Reserve	191
Luke Street	90
Deas Place Reserve - Luke Street	100

The diurnal average hourly concentrations are shown in figure A6.6 where it can be seen that the hour-by-hour concentrations at Luke Street and 25 Deas Place are generally very similar. The diurnally averaged difference in NO_x concentrations between the kerbside site of Deas Place Reserve and Luke Street is shown in figure A7.2. The cycle clearly resembles the diurnal cycle in traffic volume and peaks during the morning traffic peak at around 200 μg m⁻³.

Figure A7.2 Diurnal average of the difference in hourly concentrations of NO_x between Deas Place reserve (kerbside) and Luke Street (setback)

A7.1.3 Contribution of the motorway to roadside concentrations of NO₂

Figure A7.3 Box plots of difference in NO₂ concentrations (µg m⁻³) between upwind and downwind sites in westerly winds (above) and easterly winds (below). Central panel depicts mean concentrations and differences between the three sites



A7.1.3.1 Net effect (all wind directions)

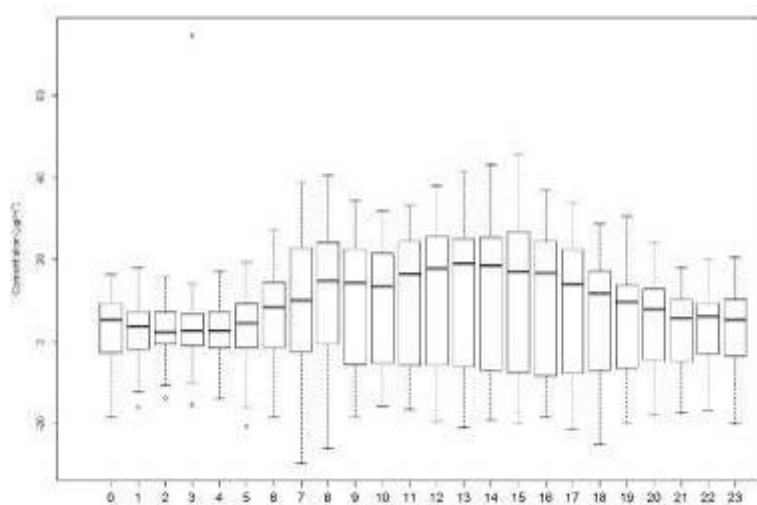
When all IOP data was considered together, mean NO_2 concentrations at Deas Place Reserve were $7\mu\text{g m}^{-3}$ higher than at the Luke Street far setback site and $9\mu\text{g m}^{-3}$ higher than at the 25 Deas Place near setback site.

Table A7.2 Mean hourly concentrations of NO_2 at the three sites for the whole IOP, including the difference between sites

Site	Mean $\text{NO}_2/\mu\text{g m}^{-3}$
25 Deas Place	17
Deas Place Reserve	26
Luke Street	19
Deas Place Reserve – Luke Street	7

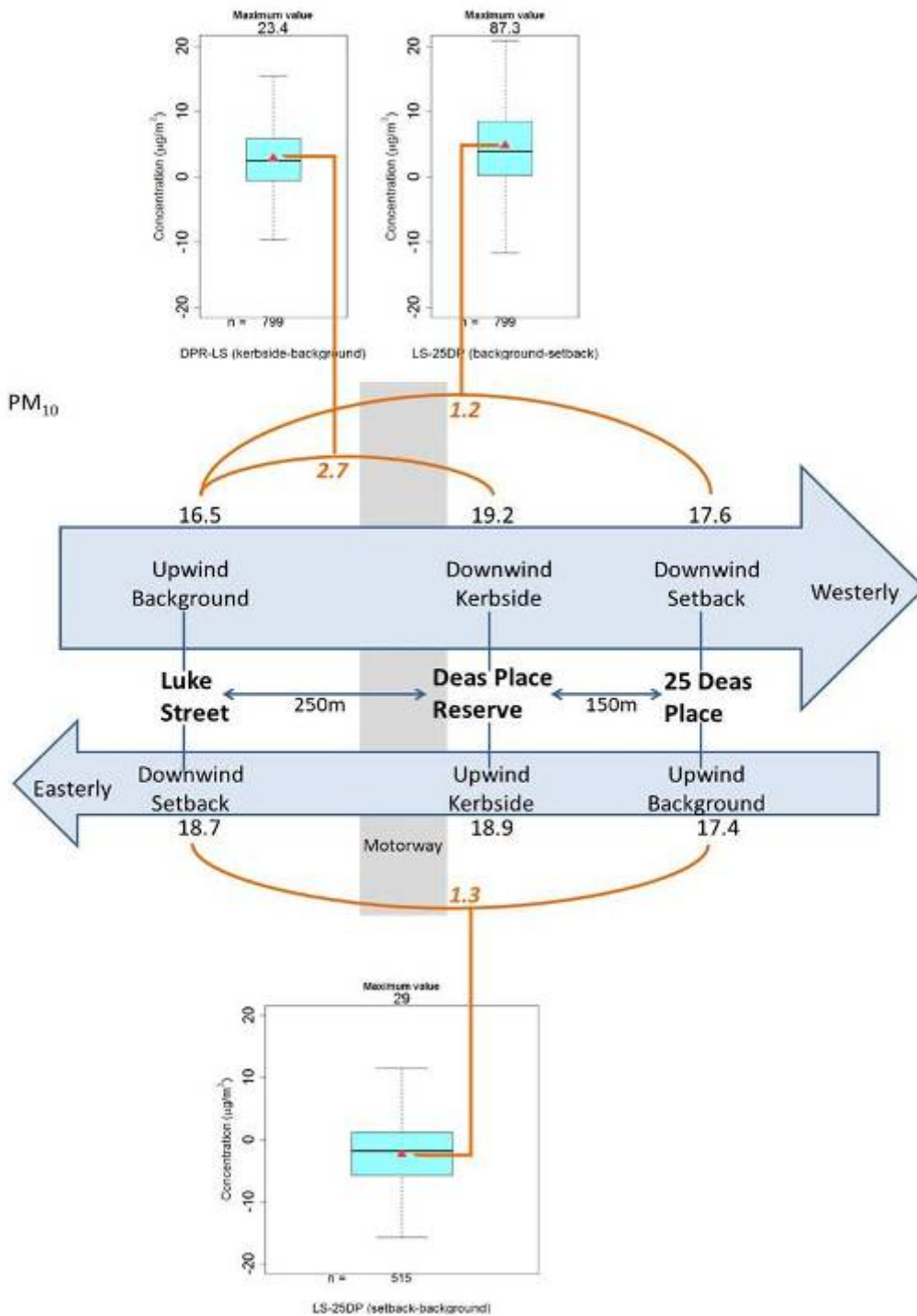
The diurnal average hourly concentrations are shown in figure A6.8 where it can be seen that those at Luke Street are, on average, 1 to $5\mu\text{g m}^{-3}$ higher than at 25 Deas Place. The diurnally averaged difference in NO_2 concentrations between the kerbside site of Deas Place Reserve and Luke Street is shown in figure A7.4. The cycle partially resembles the diurnal cycle in traffic volume, but peaks during the early afternoon at $18\mu\text{g m}^{-3}$.

Figure A7.4 Diurnal average of the difference in hourly average concentrations of NO_2 between Deas Place reserve (kerbside) and Luke Street (setback)



A7.1.4 Contribution of the motorway to roadside concentrations of PM₁₀

Figure A7.5 Box plots of difference in PM₁₀ concentrations (µg m⁻³) between upwind and downwind sites in westerly winds (above) and easterly winds (below). Central panel depicts mean concentrations and differences between the three sites



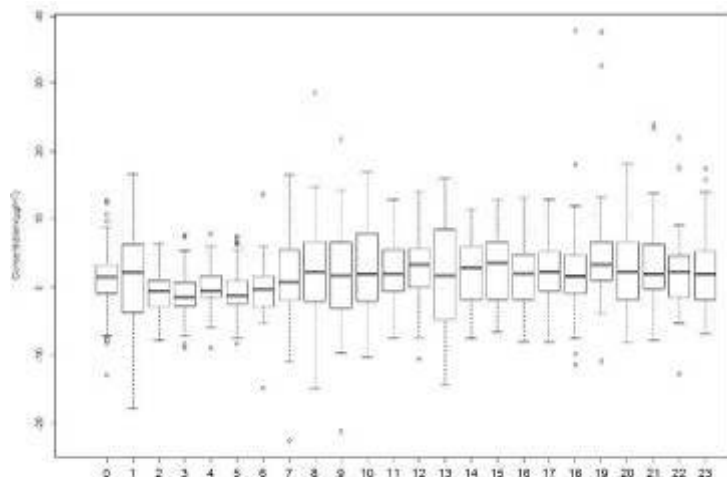
A7.1.4.1 Net effect (all wind directions)

When all IOP data was considered together, mean PM₁₀ concentrations at Deas Place Reserve were 1.7µg m⁻³ higher than at both the setback sites.

Table A7.3 Mean hourly concentrations of PM₁₀ at the three sites for the whole IOP, including the difference between sites

Site	Mean PM ₁₀ /μg m ⁻³
25 Deas Place	17.0
Deas Place Reserve	18.7
Luke Street	17.0
Deas Place Reserve – Luke Street	1.7

The diurnal average hourly concentrations are shown in figure A6.12. The diurnally averaged difference in PM₁₀ concentrations between the Deas Place Reserve kerbside site and the Luke Street setback site is shown in figure A7.6, and can be seen to average 2.4μg m⁻³ during the daytime (7am to 6pm). The cycle partially resembles the diurnal cycle in traffic volume, but with the difference in PM₁₀ between the two sites persisting beyond the evening traffic peak and remaining above zero until 2am.

Figure A7.6 Diurnal average of the difference in hourly average concentrations of PM₁₀ between Deas Place reserve (kerbside) and Luke Street (setback)

A7.2 Peak contribution of the motorway

A7.2.1 Contribution to absolute hourly NO₂ concentrations

The peak contribution of the motorway to absolute concentrations of NO₂ was evaluated using the 99.9th percentile hourly concentration for the IOP. The largest observed contribution was 53μg m⁻³, or ~one quarter of the NES.

Table A7.4 99.4th percentile hourly NO₂ concentration (equivalent to 99.9th percentile over a year) at each site, and the difference between sites in westerly and easterly winds, and for the whole IOP

Site	99.4th percentile NO ₂ /μg m ⁻³		
	Westerly winds	Easterly winds	All IOP
25 Deas Place	43	36	41
Deas Place Reserve	68	50	65
Luke Street	43	43	44
Deas Place Reserve – 25 Deas Place	35	24	32

Site	99.4th percentile NO ₂ /µg m ⁻³		
	Deas Place Reserve – Luke Street	37	24
25 Deas Place – Luke Street	12	5	12

A7.2.2 Contribution to absolute 24-hour PM₁₀ concentrations

The peak contribution of the motorway to absolute concentrations of PM₁₀ was evaluated by calculating the daily 24-hour average PM₁₀ at each site. No breakdown by wind direction was meaningful for 24-hour averaged data, so the IOP was considered as a whole. Table A7.5 shows the mean, maximum and 99.9th percentile of the measurements.

It should be noted that the mean difference between the kerbside and setback of 25 Deas Place is negative. In fact, on 80% of the days of the IOP the setback site had larger PM₁₀ concentrations than the kerbside site, indicating the influence of a larger, non-motorway source. By contrast, the Luke Street setback site experienced PM₁₀ concentrations higher than the kerbside site for 26% of days in the IOP.

The largest observed difference between the kerbside and either setback site was 6.8µg m⁻³, or 14% of the NES. The difference exceeded 5µg m⁻³ at Luke Street on nine occasions. On each of these occasions the daily mean wind speed was less than 2m s⁻¹.

Table A7.5 Observed 24-hour-average PM₁₀ on the five dates identified as peak PM₁₀ episodes

Site	24-hour average PM ₁₀ (µg m ⁻³) – all IOP		
	Mean	Max	99.9 th ile
25 Deas Place	20.5	45.4	45.2
Deas Place Reserve	18.5	43.4	43.1
Luke Street	16.8	43.5	43.1
Deas Place Reserve – 25 Deas Place	-1.9	4.1	4.0
Deas Place Reserve – Luke Street	1.7	6.8	6.8
25 Deas Place – Luke Street	3.6	8.3	8.4

A7.2.3 Contribution to absolute 24-hour PM₁₀ concentrations on calm winter nights

Table A7.6 lists the five days during the IOP when the 24-hour average PM₁₀ exceeded 25µg m⁻³ at all three of our operating monitoring sites. The same dates corresponded to four out of the five highest 24-hour averages recorded during the IOP at the following Auckland Council/MfE monitoring sites:

- Henderson
- Penrose⁶
- Takapuna
- Pakuranga
- Glen Eden
- Patumahoe

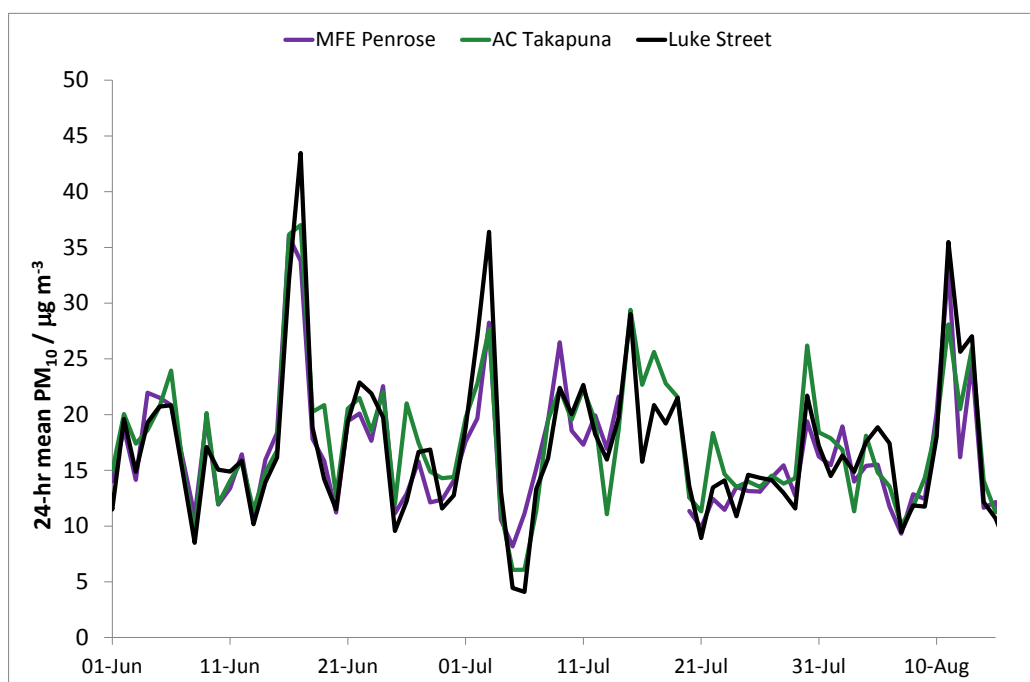
⁶ Using PM_{2.5} data. No PM₁₀ data was reported from Penrose on 15 July.

The time series of 24-hour PM_{10} at our Luke Street setback site, and at the Penrose and Takapuna motorway-influenced sites (5km and 19km respectively from the study area) for the mid-winter period are shown in figure A7.7, clearly indicating the peak events detected at all three sites. These observations strongly indicate that the five dates were affected by regional-scale reduced dispersion conditions.

Table A7.6 Observed 24-hr average PM_{10} on the five dates identified as peak PM_{10} episodes

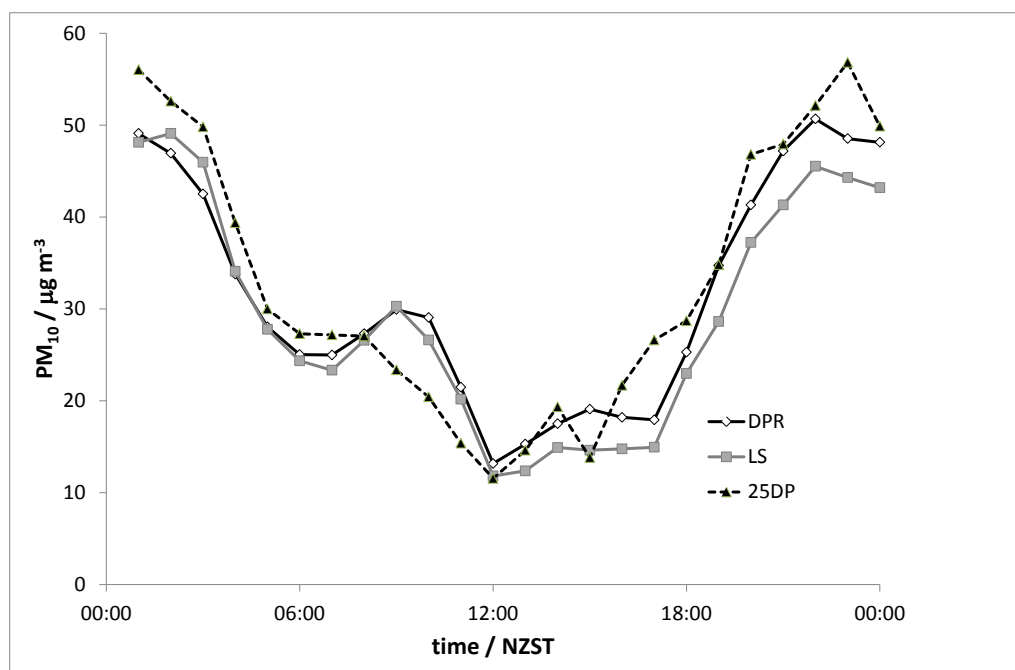
Date	Luke Street	Deas Place Reserve	25 Deas Place
16 June	39	32	37
17 June	43	43	44
2 July	27	30	34
3 July	36	37	39
15 July	29	34	33

Figure A7.7 24-hr average PM_{10} observed in the study area (at Luke Street), and at two other monitoring stations within 150m of the motorway sections of SH1



In figure A7.8 we show the diurnally averaged hourly PM_{10} concentrations from all three campaign sites during the five days identified in table A7.6. This figure indicates that the difference in PM_{10} between the three sites remained small throughout the day and peak concentrations clearly arose a few hours either side of midnight, when traffic on the motorway was rapidly falling towards a minimum.

Figure A7.8 Diurnally averaged hourly PM_{10} concentrations at the three study area sites during the five episodes identified above



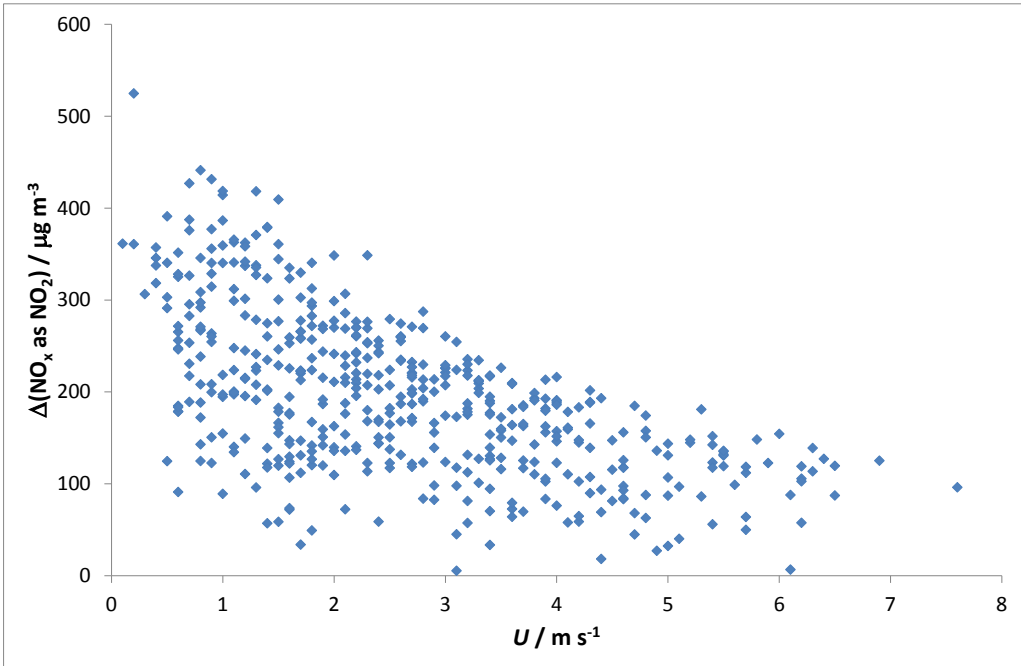
DPR = Deas Place Reserve, kerbside; LS = Luke Street, far setback; 25DP = 25 Deas Place, near setback

This suggests that during times of peak concentrations, the motorway's contribution to PM_{10} concentrations was 'swamped' by a source much greater both spatially and in emissions. That source was most likely domestic wood-burning.

A7.3 Influence of wind speed on motorway contribution

Figure A7.9 shows the relationship between differences in hourly NO_x concentrations between Deas Place Reserve and Luke Street (ie the contribution of the motorway) as a function of wind speed (in westerly winds only). Only daytime data (6am to 8pm) is shown during which time the traffic volume and hence the emission rate was consistently elevated (see figure A8.5). This indicates a clear and expected inverse relationship and sensitivity of this contribution to wind speed. An analysis of the impact of wind speed on PM_{10} found no clear relationship (figure not shown).

Figure A7.9 Hourly values of the difference in NO_x concentrations between kerbside (Deas Place Reserve) and upwind background (Luke Street)



This demonstrates the contribution of the motorway to NO_x at the kerbside, as a function of wind speed. Only data captured during westerly winds during daytime (6am to 8pm) is shown.

A8 Emission-dispersion modelling

A8.1 Aims

Emission-dispersion modelling was conducted in order to:

- evaluate the 'default' modelling tools available in New Zealand for roadside air quality assessment, especially in the regulatory context
- test the performance of dispersion modelling, independent of emission modelling
- extend the spatial aspect of the measurements by relating continuous measurements at distances <250m from the motorway to air quality at greater distances
- inform the understanding the relationships between conserved pollutants (eg NO_x) and reactive pollutants (eg NO₂, UFP)
- help identify weaknesses in emission-dispersion modelling
- provide an inter-comparison dataset for future assessments of alternative modelling tools or approaches.

A8.2 Model selection

We chose to use the dispersion model Ausroads, which is a re-packaging of the model CALINE (see below), developed by the Victoria Environment Protection Authority (EPA). CALINE is probably the world's most widely used tool for estimating air quality downwind of a road. It originated in the early 1970s, with version 4 available and has been in use since the early 1990s. CALINE4's predictions have been tested against numerous datasets by different researchers (eg Nagendra and Khare 2002) and it has been found to be generally one of the better performers among roadside dispersion models, albeit with a tendency to overpredict for parallel wind conditions and in low winds (Holmes and Morawska 2006). It also does not take into account the presence of buildings and other obstacles to dispersion. CALINE does not include any modelling of aerosol dynamics or explicit nitrogen chemistry, although NO₂ concentrations can be modelled using the discrete parcel method – a highly simplified scheme in which NO₂ concentrations are independent of available oxidant or sunlight.

Despite these limitations CALINE4 and HIWAY2 are widely used due to their relative computational simplicity, which implies modest data requirements and short run-times on a PC. Other widely used roadside models include ROADWAY, BREEZE, CAR-FMI and ADMS-Roads. These models have been extensively reviewed by Nagendra and Khare (2002) and Holmes and Morawska (2006).

Although no model is specifically recommended for regulatory use in New Zealand, the good practice guides released by MfE mention AusRoads first and devote more coverage to this model than to alternatives.

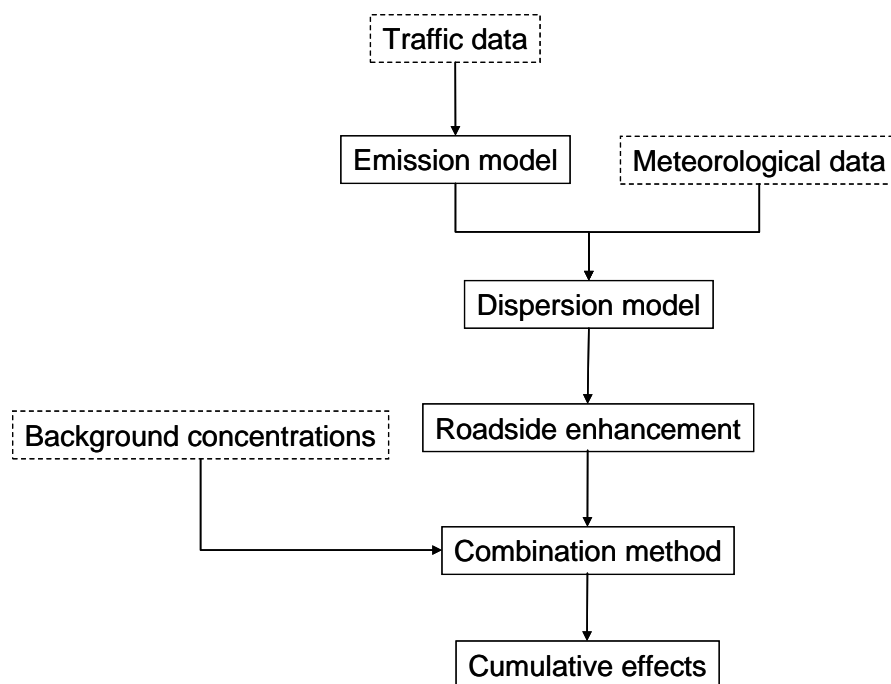
For emission modelling, we chose to use the vehicle emission prediction model (VEPM). It is problematic to apply foreign emission models to New Zealand conditions due to the incompatibility of the New Zealand fleet. It is for this reason that VEPM was recently developed in New Zealand (Kar et al 2008). 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. VEPM's performance is still being evaluated. The main alternative is the New Zealand traffic emission rate model which VEPM has largely replaced.

A8.3 Concept and method

A8.3.1 Overview of how the observational study design permits model validation

Figure A8.1 summarises the components of roadside air quality modelling using an emission-dispersion modelling technique. The technique predicts air quality alongside a road in response to a given traffic profile under certain meteorological conditions. The traffic and meteorological conditions may be specified using observed or artificial data (ie modelled or scenario). These conditions may represent current conditions (ie observed simultaneously), or expected conditions (eg data from one period used to represent conditions in another period, as in using historical data to generate ‘typical’ conditions). Each step may introduce intrinsic errors to the assessment (arising from model or observational limitations and uncertainty). Extrinsic errors may also arise due to choices made by the modeller (choice of model, parameter values, approach to dealing with missing data, etc).

Figure A8.1 Components of roadside air quality assessment based on emission-dispersion modelling.



Note: Dashed boxes represent data sources which may be derived either from observations, or may be the output of modelling activities

Our observational campaign was designed to remove, constrain or reduce a number of the error sources associated with the steps shown in figure A8.1 and listed in table A8.1. Specifically:

- the background concentrations
- the roadside enhancement
- the cumulative effects
- most of the required meteorological data (exceptions explained below)
- most of the required traffic data (exceptions explained below)

were all directly observed on site at high resolution and concurrently. This means that:

- the performance of the dispersion model (with meteorological data) can be investigated independently of the emission model (and traffic data)
- an estimate of emission rates can be determined independently of an emission model from inverse dispersion modelling – this estimate can then be compared with emission modelling to provide partial validation of an emission model.

Alternative meteorological datasets are available, enabling sensitivity tests of the modelling results to dataset selection. Specifically these datasets are:

- three study observational sites (Luke Street, Deas Place Reserve and Deas Place)
- concurrent observations from four nearby permanent sites – airport, Mangere, Wiri and Onehunga
- historic (eg ‘typical year’ or long-term climatology) observations from the same four nearby permanent sites
- the use of additional meteorological data from which boundary layer parameters can be estimated
- the Auckland Council artificial meteorological dataset.

By conducting such sensitivity tests it will be possible to quantify the contribution of many intrinsic and extrinsic model errors to assessment uncertainty.

Table A8.1 Error sources in the roadside observational study design

Step	Error type	Comment
Traffic data	Volume	Observed and modelled
	Speed	Observed
	Fleet composition	Derived from observation (vehicle length)
	Cold start	No data
Emission model	Detailed fleet composition	No data
	Local bias	No data
	Emission model errors	No data
Meteorological data	Spatial representativeness	Observed (locally)
	Temporal representativeness	Observed (concurrently)
	Intrinsic model error	Observed
	Intrinsic observational error	Observed (ie quantifiable)
	Missing/assumed data	No data; alternative assumptions may be tested
Dispersion model	Geometry	Alternative descriptions modelled
	Land-use	No variation
	Topography	No variation
	Intrinsic error	No data
Background concentrations	Low resolution	Observed (at high resolution)
	Spatial and temporal representativeness	Observed (locally and simultaneously with traffic and meteorology)
	Instrumental/model error	Data available
Combination method	Temporal coincidence	Observed (locally and simultaneously with traffic and meteorology)

A8.3.2 Limitations of our study design

Our design does **not** present opportunities for investigating errors due to the treatment of topography. This is simply because the resource limitations constrained the study to one location. Our approach to this limitation was to select a flat study area to prevent complex 'topography' from interfering with other results (ie constraining the error associated with topography by removing it). This is consistent with the chosen roadside dispersion model which does not allow the user to specify topography as it is assumed not to be significant over spatial scales <1km.

For similar reasons our design does **not** permit an investigation of errors associated with land use, as land use cannot be varied in our dataset. Our study area was explicitly selected to encompass residential land use of a form common in urban New Zealand (single-storey detached houses). In the case of Ausroads, the lack of land-use variations is desirable as the model is not capable of representing different surfaces (described as an aerodynamic roughness length) within the domain. For the SOSE model, the local topography and differences in land use are incorporated into the statistically optimised model parameters.

A8.3.3 Comment on missing meteorological data

It is well established that the state of the atmospheric boundary layer is a major factor in dispersion near the surface. 'State' is multi-faceted and difficult to define. Several meteorological parameters can be used to describe this state. In dispersion modelling, two of the most commonly used parameters are the mixing height and the atmospheric stability. Unfortunately these parameters, unlike wind speed etc, are very difficult to observe, especially in urban areas. In dispersion modelling applications this data is often missing and is either estimated using reasonable assumptions, or using non-local data. It was not possible within the resources available for our study to conduct the observations necessary to gather local data on boundary layer state. However, it is possible to use our dataset to compare alternative assumptions or estimates. Using SOSE, it is possible to estimate the average daily distribution of the 'box height'. However, these estimates in box height are an average over the study period, and, as such, they are not useful for estimating the day-to-day variability in the surface conditions.

A8.3.4 Validation of emission models

Our study design was not intended to permit the direct validation of emission modelling because no direct observations of emission rates were made. However, the dataset does permit a degree of indirect validation.

The approach to validation described above allows the errors relating to meteorological data and dispersion modelling to be constrained. After these errors have been accounted for, the residual errors must be attributed to uncertainties in the traffic data and emission modelling. Our study design includes local, concurrent traffic observations as volume and speed in four vehicle length classes. VEPM on the other hand handles nine classes of vehicles and therefore the dataset does **not** include the full observations of traffic fleet and speed required by VEPM. This prevents the dataset from being used to quantify errors involved in converting the observed traffic data into the data required as an input to VEPM. Furthermore, our study design does **not** include observations of the emission standard and age of all vehicles on the southern motorway, and therefore did not permit us to attribute emission model errors between the fleet model and the per-vehicle class emission model.

Nevertheless, our study design **does** permit two forms of partial validation. First, the total error related to traffic data and emission modelling can be quantified (when all other error sources are constrained). Second, the unobserved total emission rate can be estimated through inverse modelling. This is where the modelling process outlined in figure A8.1 is executed in reverse, (ie the modelling question is framed as 'what emissions would have led to the observed concentrations for the observed meteorological

conditions?'). The emission rates estimated in this way can be directly compared to those predicted by VEPM for the aggregated fleet, although, as the method is dependent on dispersion modelling and meteorological data, it is subject to the uncertainties associated with these components. Also, using SOSE, though it is not possible to say anything about the absolute values of the emissions rates, it is possible to determine whether the emission rates between pollutants are in proportion to each other (their ratio).

A8.4 Modelling inputs

A8.4.1 General

Ausroads was used to model the impact of emissions from the southern motorway only on local air quality in the study area. In all cases, all other emission sources, including the non-motorway sections of the Princes Street interchange, were ignored.

Multiple modelling runs were conducted. However, in all cases, the aerodynamic roughness length of the surface was held constant at 0.4m.

As indicated above, the dispersion ratio is independent of the emission strength, therefore Ausroads runs were conducted with constant unit emissions for every hour. Given the linear relationship between emissions and concentrations for Gaussian models (like Ausroads) emission estimates were used at a later stage to amplify the dispersion results in order to obtain the estimated contribution from the motorway traffic.

A8.4.2 Meteorological datasets

Several input meteorological datasets were created in order to systematically investigate the influence of the following five issues on the modelling outcomes:

- 1 The impact of micro-scale differences between the three met sites within the study area
- 2 The difference between using local on-site data and data collected remotely (several kilometres away)
- 3 The difference between using data collected during the campaign and using a larger, historical dataset
- 4 The difference between assuming constant mixing height and stability class compared to hourly estimated values
- 5 The difference between using observed data and artificial data.

Table A8.2 lists the 15 datasets created and how they were combined in order to investigate each issue. In general, the Luke Street dataset was used as a reference dataset. We decided to do this as the Luke Street meteorological mast was the tallest of the three on-site campaign masts (at 9.5m); it was also located in an open field with no buildings within 30m. In this way, it more closely represented the kind of site recommended by MfE for meteorological data for modelling purposes, compared with the 25 Deas Place and Deas Place Reserve sites (MfE 2004).

Table A8.2 Combinations of meteorological datasets used to address five dispersion modelling issues

Issue	Subject datasets	Reference datasets
Local influences on met site	Deas Place Reserve 25 Deas Place	Luke Street
Local versus remote met site	Airport Wiri Onehunga Mangere	Luke Street

Issue	Subject datasets	Reference datasets
Concurrent versus historical met set	Airport_historic Wiri_historic Onehunga_historic Mangere_historic	Airport Wiri Onehunga Mangere
Boundary layer parameters	Luke Street_estimated_H Luke Street_estimated_stability	Luke Street
Observational versus artificial metset ^(a)	ARC_metset	Luke Street

^(a) metset = meteorological dataset

The on-site datasets and sites (Luke Street, Deas Place Reserve and 25 Deas Place) are described in chapter A4. The off-site datasets were generated from four nearby permanent meteorological stations, as summarised in table A8.3. Their locations are indicated in figure A8.2.

The artificial dataset was a subset of the ARC artificial meteorological dataset for Auckland. Specifically, a time series for 10m altitude was extracted from the CALMET output for the grid point closest to the study area reference point.

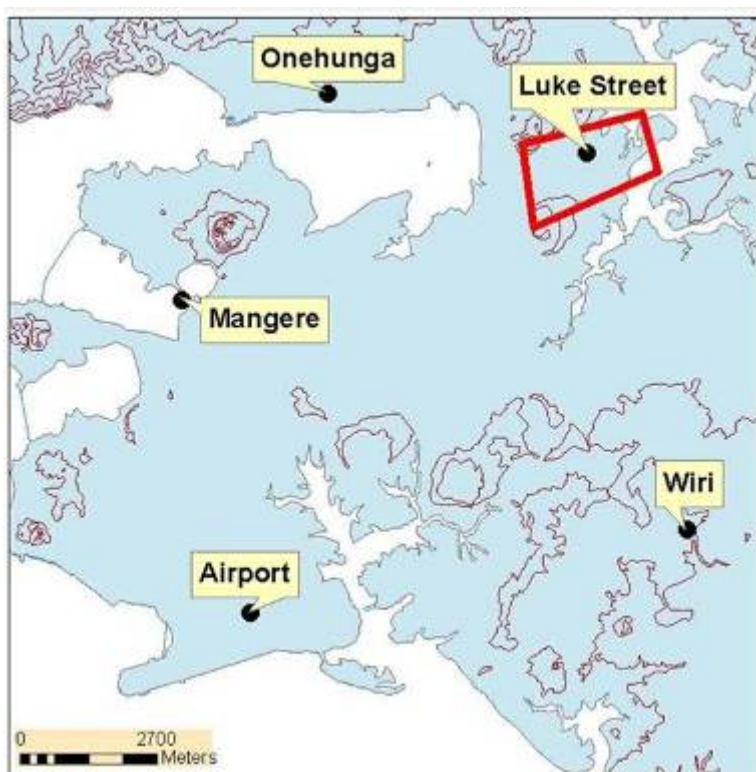
The local metsets covered the full duration of available wind data from each site. The remote metsets covered the full duration of the campaign. The historic metsets covered the full years 2005–07. The ARC metset covered the whole of 2007 (this artificial metset did not provide more recent data).

For all datasets except the ARC metset and those with estimated boundary layer parameters, each dataset consisted of hourly records of wind speed and direction derived directly from the hourly average observed data at each site.

Table A8.3 Details of permanent meteorological stations used as data sources for dispersion modelling within this project

Name	Coordinates (NZTM)	Coordinates (NZMG)	Distance from study reference point	Mast height
Airport	1759165 5902930	2669565 6464622	9.7km	10m
Onehunga	1760436 5911538	2670853 6473228	4.7km	10m
Wiri	1766415 5904322	2676818 6466000	6.4km	10m
Mangere	1758007 5908097	2668417 6469792	7.4km	10m

Figure A8.2 Locations of meteorological stations used as data sources for dispersion modelling.



Note: Red box defines study area. Contours are altitude (15 m)

Table A8.4 Details of meteorological datasets created for dispersion modelling within this project

Metset	Start	End
Luke Street (including Luke Street_estimated_H and Luke Street_estimated_stability)	23 Apr 2010	24 Aug 2010
Deas Place Reserve	1 Apr 2010	29 Sep 2010
25 Deas Place	11 May 2010	6 Aug 2010
Airport Wiri Onehunga Mangere	1 Apr 2010	29 Sep 2010
Airport_historic Wiri_historic Onehunga_historic Mangere_historic	1 Jan 2005	31 Dec 2007
ARC_metset	1 Jan 2007	31 Dec 2007

A8.4.3 Mixing height

The good practice guide (GPG) for atmospheric dispersion modelling (MfE 2004) states the mixing height has a critical role in determining the air pollution potential of an area, and refers to four methods for determining mixing height, including direct measurements (sonic detection and ranging), derivation from

upper air or surface measurements, or through using a prognostic meteorological model. The GPG states that heights less than 50m contain a ‘significant degree of uncertainty’ (MfE 2004). Indeed many models (Ausroads included) set an absolute minimum mixing height of 50m.

Directly observed estimates of the mixing height were not available for our observational campaign. For this reason we derived two scenarios for the time series of mixing height:

- 1 Assumed constant ‘typical’ value
- 2 A conservative, conditions-dependent minimum value based on the ARC artificial meteorological dataset.

The ARC artificial meteorological dataset includes surface data extracted from the CALMET grid for sites where meteorological data is observed. This data includes a local estimate of mixing height for every hour of 2005 and 2007. Our scenario estimates have been based upon this artificial time series representing the location of the Wiri meteorological station (dataset H8 according to the ARC dataset’s documentation).

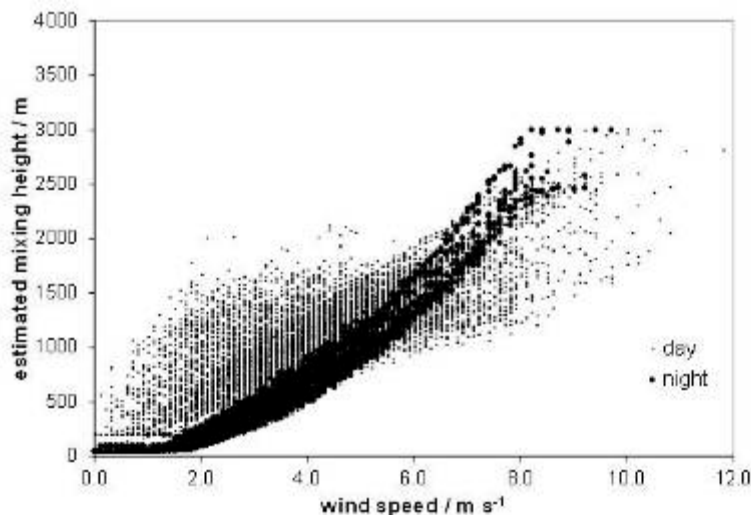
First, we adopted a modelling convention of 1000m for our assumed constant value. In terms of hourly variation, figure A8.3 shows that the estimated mixing height has a reasonably well-defined curvi-linear relationship with wind speed during the night but a greater linear relationship with more scatter during the day. Dispersion is most sensitive to low mixing heights and less so to greater mixing heights. Therefore, assuming that the night-time relationship is also valid during the daytime is a conservative, hence it give a lower estimate of the daytime mixing height. Consequently, we derived the following empirical relationship between minimum mixing height (H_{min}) and hourly mean wind speed (U):

If $U < 1.8 \text{ m s}^{-1}$: $H_{min} = 50\text{m}$

If $1.8 < U < 5.6 \text{ m s}^{-1}$: $H_{min} = 18.9U^{2.4}$

If $U > 5.6 \text{ m s}^{-1}$: $H_{min} = 1000\text{m}$

Figure A8.3 Relationship between hourly estimated mixing height (using CALMET, from the ARC artificial dataset for Wiri) and hourly mean wind speed observed at Wiri for 2005 and 2007



A8.4.4 Boundary layer stability

As with the mixing height, the related phenomenon of boundary layer thermal stability is well known to have a substantial influence on dispersion. Ausroads provides the user with the option of supplying

stability information on an hourly basis in the form of the Pasquill-Gifford (P-G) class. The P-G class ranges from A (highly unstable) to F (highly stable). P-G class cannot be measured directly, but can be defined using various algorithms. In this project we used the Victoria EPA algorithm which requires data on wind speed and either solar radiation (for daytime) or cloud cover (for night-time). Neither solar radiation nor cloud cover data was directly available for the campaign from the study area, but was available from the airport, 9km away. P-G class is specified in the ARC artificial dataset.

In common with the approach to mixing height, we derived two scenarios:

- 1 Assuming constant P-G class of D (neutral)
- 2 Using the Victoria EPA algorithm to estimate P-G class for each hour based on locally observed wind speed, plus radiation and cloud data from Auckland airport.

A8.4.5 Source and receptor geometry

Two source geometries were modelled:

- 1 A single straight continuous line source, extending 5km in either direction from the study reference point (coordinates 0,0 in Ausroads), aligned with the motorway's centreline.
- 2 Eight line sources representing three northbound lanes, three southbound lanes a southbound off-ramp and northbound on-ramp; with through lanes also extending 5km in both directions.

Receptors were placed along Luke Street (perpendicular to the motorway) at the following distances (in metres) from the motorway centre line (some receptors were chosen to coincide with locations of diffusion tubes):

- 10, 15, 20, 25, 30, 36, 40, 60, 80, 100, 115, 130, 150, 170, 190, 210, 250, 523, 819 on the western side
- 10, 15, 20, 25, 30, 40, 46, 60, 80, 100, 120, 130, 150, 170, 190, 210, 313, 519, 840, 1142 on the eastern side.

A8.4.6 Traffic data

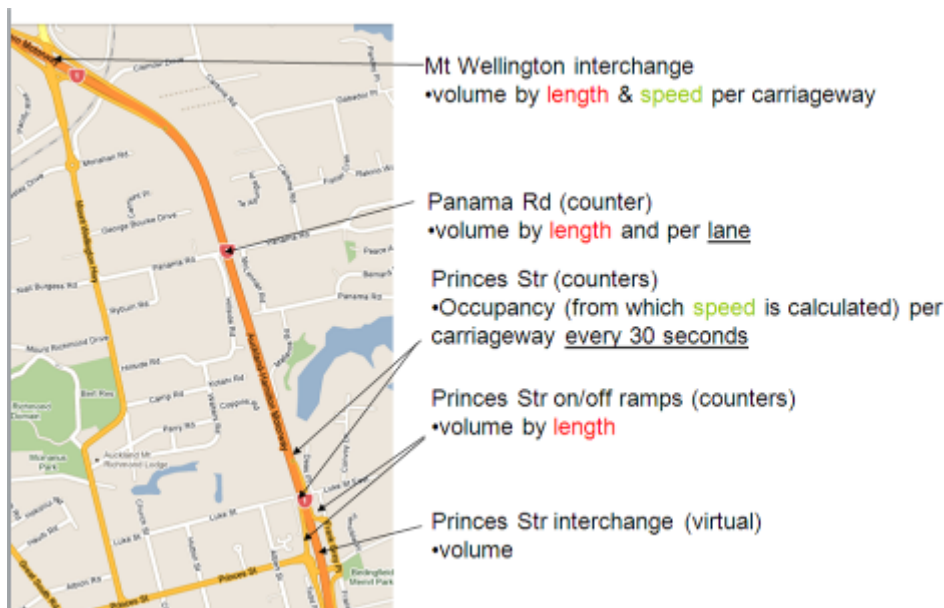
Emission modelling was conducted using VEPM. As an input, VEPM requires:

- fleet composition in nine vehicle classes
- average speed for cars, light commercial vehicles (LCVs) and heavy commercial vehicles (HCVs)/buses.

Other data required to estimate cold start emissions, degradation and fuel specification can be provided as an option. VEPM predicts fleet average emission rates per vehicle. Thus, estimation of total emission rates also requires vehicle volume rates.

The full range of data required was not available at this site. However, an unusually detailed dataset was available from a range of automated traffic counting sites on the section of motorway passing through the study area, as shown in figure A8.4.

Figure A8.4 Sources of traffic observations along the stretch of the southern motorway (SH1) passing through the study area



The high degree of traffic counting detail in this area permitted us to derive hourly averaged volume, fleet and speed profiles for each lane (and the two ramps at the Princes Street interchange) for weekdays, Saturdays and Sundays. In constructing these profiles we assumed that, except for periods of congestion (see below), traffic volumes, speeds and fleet composition at any given hour did not vary substantially between weekdays, or between one week and another.

The resulting profiles had the following properties, some of which are illustrated in figures A8.5 and A8.6.

Traffic volume:

- There were distinct morning and evening peaks in volume.
- Volumes in each direction were approximately the same between mid-day and 3pm, seven days a week.
- There was a relatively rapid fall in volume around 7pm to 8pm.
- Volumes varied slightly by lane, with lane 3's share falling substantially between 8pm and 5am.
- Daytime volumes on the on/off-ramps were 6% to 9% of the total SH1 volumes.
- On- and off-ramps had slightly different diurnal volume patterns compared with the main carriageway, with less pronounced northbound morning peak and southbound afternoon peaks, but a more pronounced southbound morning peak.

Figure A8.5 Diurnal average traffic volume on SH1 at Panama Rd for Jan – Aug 2010 inclusive

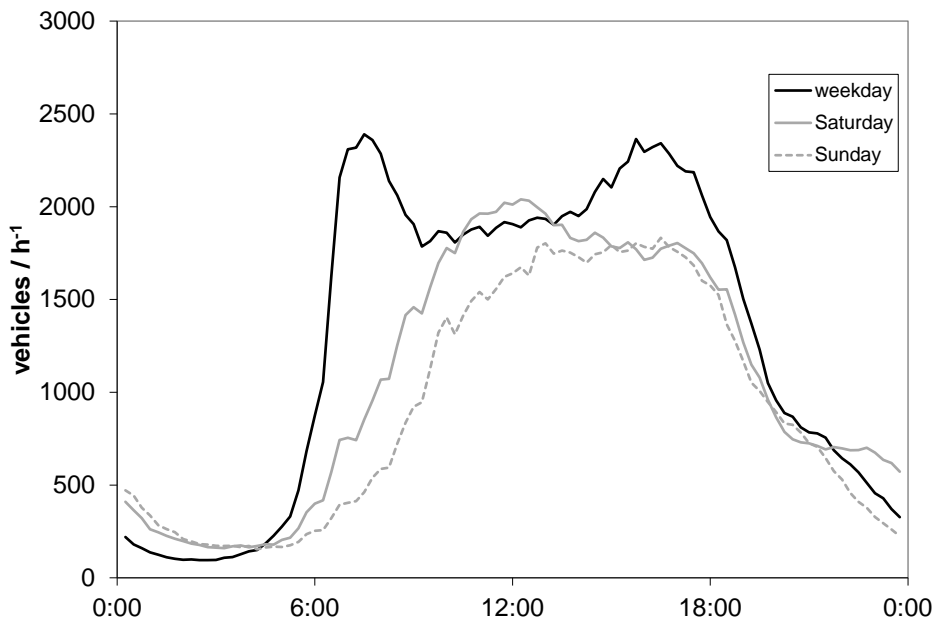
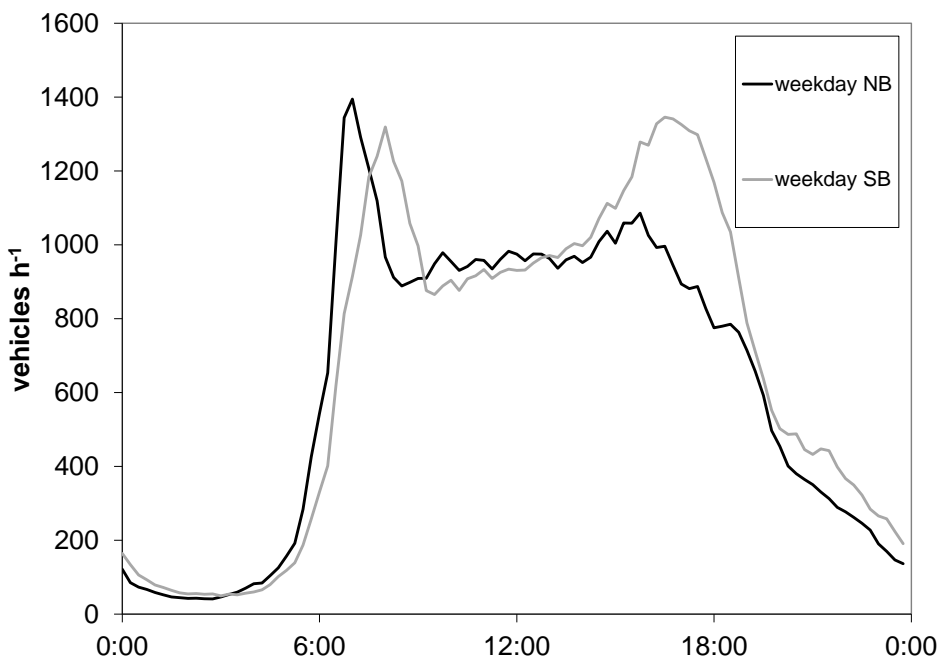


Figure A8.6 Diurnal average weekday traffic volume on SH1 for northbound and southbound traffic at Panama Rd for Jan – Aug 2010 inclusive



Traffic speed:

- Speed was relatively bi-modal, being mainly in the range 70km/h to 100km/h⁻¹ or 5km/h to 30km/h⁻¹.
- Congestion on the motorway was mainly limited to the northbound carriageway on weekdays. Congestion (speed reduction to <30km/h⁻¹) commenced on most weekdays around 7am. The return to normal speeds, however, was more unpredictable occurring between 8am and 10.30am. Evening northbound congestion was more random, but could occur between 4pm and 6.30pm.

- Congestion on the southbound carriageway was relatively rare and limited to weekdays during the 4pm to 6.30pm period.

Fleet composition:

- Fleet composition was not directly available from observations, which reported vehicle length in four classes. We inferred that vehicles longer than 5.5m equated to HCVs, as defined by VEPM (ie >3.5 tonnes).
- HCVs represented < 5% of volume during the daytime, but up to 30% per hour at night on weekdays.
- HCVs tended not to use lane 3.

A8.4.7 Emission rates

Emission rates were modelled using VEPM v3.0, with the following assumptions:

- The proportion of all vehicles which were HCVs (as defined by VEPM) equalled the proportion of vehicles longer than 5.5m in the traffic profiles, as derived from observations (see above).
- The petrol/diesel split was per VEPM defaults for 2010.
- The car/LCV split was per VEPM defaults for 2010.
- The split of HCVs between the five classes specified in VEPM was per VEPM defaults for 2010.
- Average speed was the same for all vehicles.
- All optional model inputs were set to the VEPM default values.

Emission rates were calculated for each of eight traffic lanes for every hour of a 'standard' weekday, Saturday and Sunday.

A8.5 Meteorological sensitivity tests

A8.5.1 Comment on presentation of results

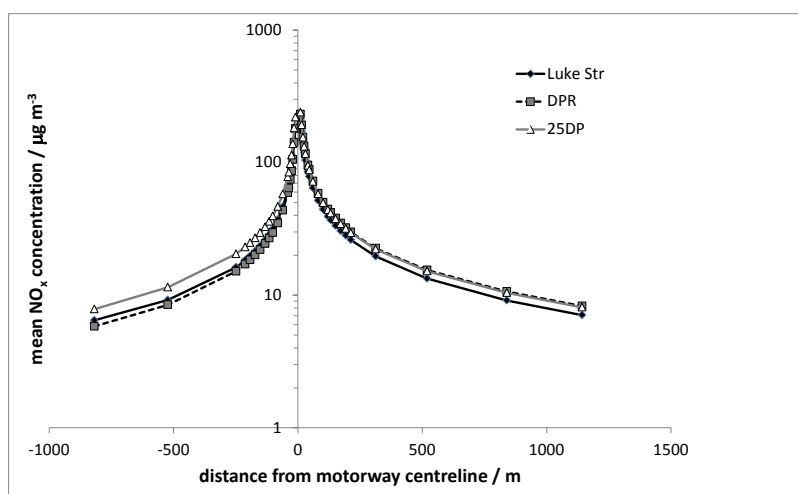
The sensitivity tests generated a vast amount of data. For clarity and simplicity, we only present results here for receptors at +/-150m from the motorway centreline. We also only present the sensitivity of mean concentrations to choice of metset. Other results (ie for other receptors or other statistics) are in the project database and are available on request.

Ausroads was run with unit emissions at all times. Results were then weighted using the modelled hourly NO_x emission rates in order to identify the impact on modelled concentrations.

A8.5.2 Comparison between three local meteorological datasets

For this test we compared predicted mean concentrations using the three local metsets for the period of the IOP only (ie the period during which all three sites were reporting meteorological data) as shown in figure A8.7. Relative to the Luke Street metset, using the 25 Deas Place dataset led to an average overestimate of concentrations of 13% at 150m on the east side and 26% at 150m on the west side. Use of the Deas Place Reserve dataset led to an average overestimate of concentrations of 14% at 150m on the east side and an underestimate of 6% at 150m on the west side. These relative errors were generally consistent at all receptors for each side of the motorway.

Figure A8.7 modelled mean NO_x concentrations for the IOP based on three alternative metsets. Note log scale on y-axis



A8.5.3 Comparison of off-site to local meteorological datasets

Compared with the Luke Street dataset, the campaign off-site datasets led to a wide range in results, from consistent overprediction when the Onehunga dataset was used to 20% to 35% underprediction when the Mangere or airport datasets were used. The wide range in results prevented identification of a better performing remote dataset.

Table A8.5 Relative difference in predicted mean concentration of NO_x for the IOP using off-site metsets relative to the reference (Luke Street) metset

Metset	West side (150m)	East side (150m)
Onehunga	+33%	+9%
Wiri	+4%	-26%
Mangere	-24%	-28%
Airport	-22%	-35%

A8.5.4 Comparison of historic to campaign meteorological datasets

Note that these metsets cover not only different time periods but also different durations. Results from historical metset were compared with results from the corresponding campaign metset at the same site. For Wiri and the airport the differences were small (<10%). The substantially different result for eastern and western receptors from the Onehunga dataset further indicated the likelihood of bias associated with wind direction for that mast.

Table A8.6 Relative difference in predicted mean concentration of NO_x for the IOP using campaign-period off-site metsets relative to the historic off-site metsets

Metset	West side (150m)	East side (150m)
Onehunga	-5 %	-40 %
Wiri	-4 %	-4 %
Mangere	-14 %	-18 %
Airport	+7 %	-9 %

A8.5.5 Comparison of constant to estimated boundary layer parameters

Compared with the Luke Street dataset in which mixing height was artificially set to a constant 1000m and stability to P-G class D, including an hourly estimate of stability class led to an overestimate of 15% to 16% on both sides of the motorway. Including an estimate of mixing height made no difference at all.

Table A8.7 Relative difference in predicted mean concentration of NO_x for the IOP using estimated stability and mixing height, relative to constant assumptions, using the reference (Luke Street) metset

Metset	West side (150m)	East side (150m)
Estimated stability	+16%	+15%
Estimated mixing height	No change	No change

A8.5.6 Comparison of artificial to observational meteorological datasets

Note that these metsets cover different time periods and durations (see table A8.2).

Compared with the Luke Street dataset, the ARC metset led to an overestimate of 10% on the east side and an underestimate of 7% on the west side.

Compared with the Luke Street dataset which had estimated boundary layer parameters, the ARC metset led to an underestimate of 4% on the east side and an underestimate of 20% on the west side.

A8.5.7 Meteorological sensitivity tests – Conclusions and discussion

Figure A8.8 summarises the results of the meteorological sensitivity tests, presenting the percentage impact on predictions of concentrations at the receptors 150m east and west of the motorway, relative to using the reference (Luke Street with constant boundary layer parameters) dataset.

Overall, the largest source of deviation from the reference was the use of off-site datasets. This led to underestimates of concentration from 22% to 35% in five out of eight cases. Use of data from Onehunga led to relative overestimates of concentrations, indicating that the specifics of the site and dataset were significant.

Internationally, it is common practice for dispersion modelling to be conducted using meteorological data from a nearby climate station. Despite the Auckland airport meteorological mast having a height of 10m and the Luke Street mast a height of 9.5m, during the campaign, the mean wind speed recorded at Luke Street was only 58% of that observed at the airport. This is not unexpected due to the significant change in surface roughness between the largely open and flat space around the airport, and the urbanised surface of our study area.

Data from the Mangere, Onehunga and Wiri sites was similarly affected, but to a lesser extent respectively, ie the Wiri site provided the smallest deviation from the reference (Luke Street) results. One could qualitatively argue that the land use around the Wiri site was more similar to our study area than any of the other off-site locations. The magnitude of the deviation from using the Onehunga site was similar to the others, but the sign was opposite – ie use of the Onehunga metset led to a relative overprediction rather than underprediction of concentrations. In light of the large variation and inconsistency in these results, a more detailed analysis of the observational and modelling datasets would be warranted. Dealing with this spatial variation in meteorology is one of the purposes of the ARC artificial dataset. In our test, use of this dataset reduced the ‘error’ relative to using the Luke Street dataset to < ~10%.

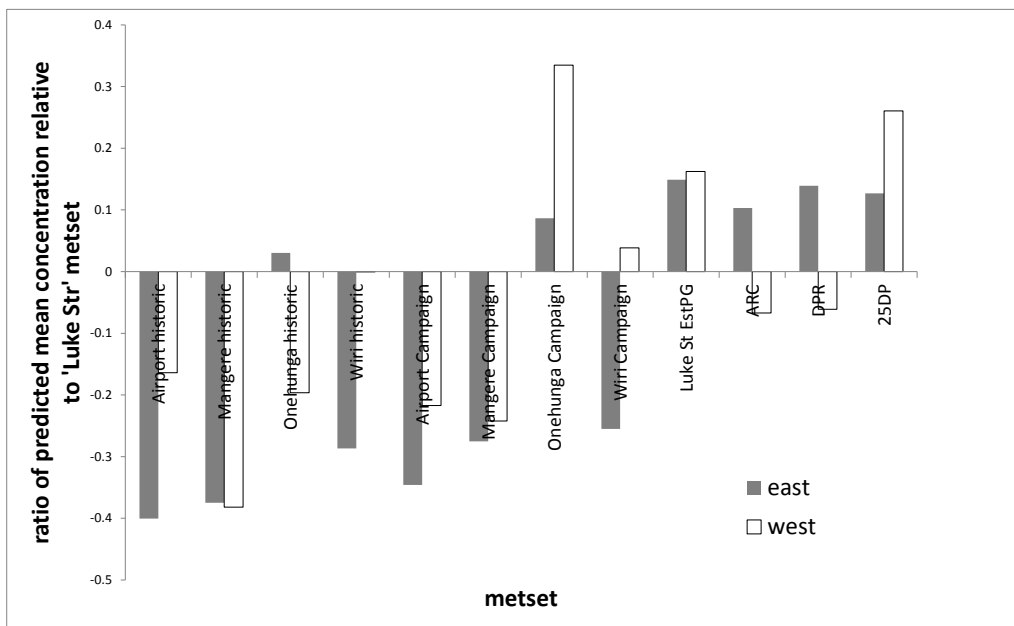
The error introduced by using the longer ‘historic’ datasets was small in comparison with this spatial error and acted to increase the net error in the case of the airport, Mangere and Wiri datasets.

Relative to using the Luke Street dataset, using data from either of the two other (shorter) on-site masts led to a fairly consistent overestimation of mean concentrations at all receptors in three out of four cases. This could be interpreted as a consequence of an underestimation of wind speed when using data from less ideal masts. The effect of using the Deas Place Reserve (kerbside) site varied between overestimation of concentrations for eastern receptors and underestimation for western receptors. This could be related to the fact that a nearby tree sheltered the anemometer in north-easterly winds (our detailed analysis, not shown, revealed a 20% reduction in the ratio of wind speeds between Deas Place Reserve and Luke Street in the 30° - 40° wind direction).

Although the MfE guidance followed in our study for the siting of meteorological masts is clear and consistent, we have shown that artefacts or local anomalies can still affect data collected in this manner. Thus, following the GPG is not enough to ensure meteorological data does not introduce errors that will influence dispersion modelling output. Evaluation of the data is necessary before and concurrently with dispersion modelling.

The effect of estimating stability introduced a relative overestimation of concentrations similar to using data from the shorter on-site masts. This result implied that the inclusion of estimated stability was conservative for our campaign relative to our (commonly used) assumption of neutral stability at all times.

Figure A8.8 Ratio of predicted mean concentrations at 150m receptors on east and west side for a range of metsets, relative to using the Luke Street metset



A8.6 Geometry sensitivity tests

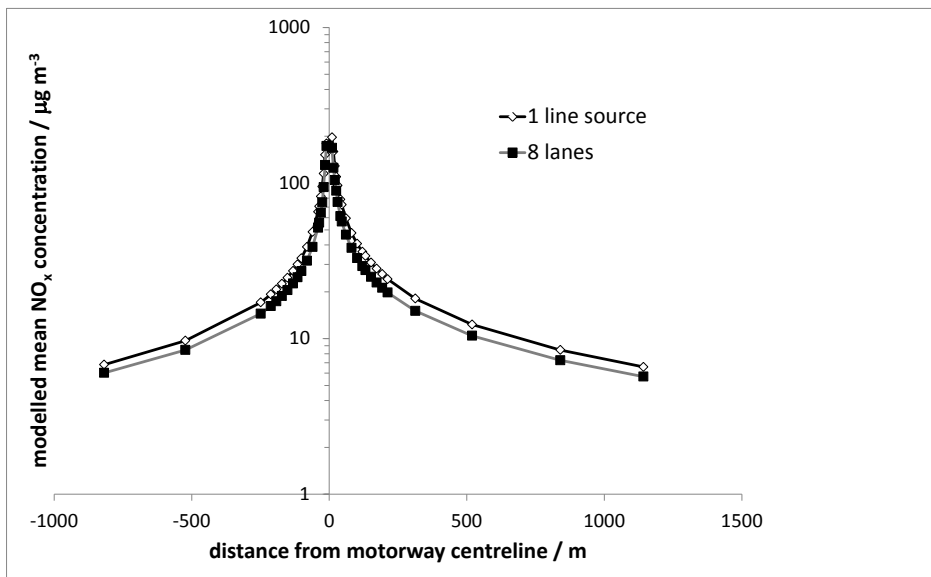
A8.6.1 Inputs

For this test, the roadway geometry was altered while keeping all meteorological inputs constant. Throughout, the Luke Street (constant mixing height and atmospheric stability) dataset was used.

A8.6.2 Results

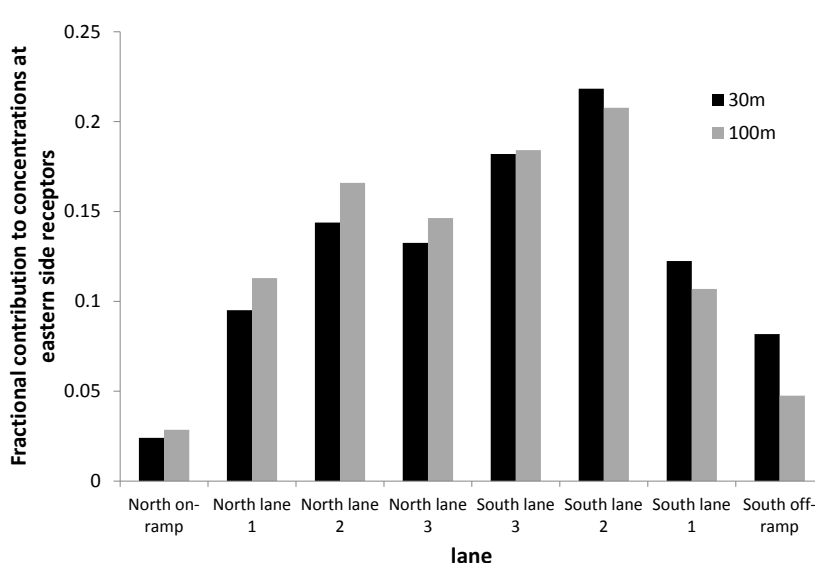
Modelling all eight lanes independently led to a prediction of concentrations up to 22% lower than modelling a single line source. The underprediction peaked in the 20m to 60m range on both sides of the motorway. This probably occurred because modelling a single line source does not account for the different dilution of emissions from the closer nearside lanes relative to the emissions on the far-side carriageway lanes. Figure A8.9 shows the contribution of emissions from each lane to modelled mean concentrations at 30m and 100m to the east of the motorway. It can be seen that emissions from the middle southbound lane (lane 2) dominate. The on and off-ramps together contribute 11% at 30m and 8% at 100m.

Figure A8.9 Modelled mean NO_x concentrations for the IOP based on two alternative geometries (one line source and eight line sources representing each lane of traffic)



Note log scale on y-axis

Figure A8.10 Modelled fractional contribution of emissions from each lane to mean NO_x concentrations for the IOP at receptors 30m and 100m east of the motorway centreline



A8.7 Evaluation of the Ausroads model performance

A8.7.1 Methods

In this section we describe the evaluation of the Ausroads model performance independently of emission modelling and assessment of background air quality. The Luke Street (constant mixing height and stability) meteorological dataset was used throughout to model the motorway as a single line source. All other model parameters were identical to the tests described above.

The observational campaign was explicitly designed to permit validation of dispersion modelling independently of all other assessment components (traffic, emission, meteorological data or modelling). This was achieved by combining three measurement points. The concept was based on the assumption that the only significant physical process occurring between the kerbside and setback sites was dilution (ie dispersion), with zero emission or removal. Over the ~100m between our two sites this assumption was reasonable (traffic on Deas Place, a residential cul-de-sac within this space, is minimal). Therefore, during prevailing SW winds, we defined a dispersion ratio (using C to denote concentration of any pollutant emitted by traffic and passive over the ~500m spatial scale of the study area) as:

$$dispersion_ratio = \frac{C_{setback} - C_{background}}{C_{kerbside} - C_{background}}$$

This ratio is independent of the emissions on the motorway, and therefore independent of errors in emission or traffic data. The dispersion ratio is directly available from our observations at high temporal resolution, and can therefore be compared in detail with the prediction of any relevant dispersion model.

For the model's evaluation we created a validation dataset from the fixed observations. The criteria for inclusion in the dataset was:

- all three fixed sites operating and reporting quality assured data
- westerly winds only (so that Luke Street represented background, Deas Place Reserve represented downwind kerbside, and 25 Deas Place represented downwind setback).

A8.7.2 Dispersion ratio results

The observed hourly dispersion ratio generally exhibited lower values than the modelled ratio as well as a wider range. In the observations the interquartile range was 0.07 – 0.19 with a mean of 0.11, whereas Ausroads predicted an interquartile range of 0.31 – 0.34 with a mean of 0.33 (see figure A8.11), ie on average Ausroads overpredicted the contribution of motorway emissions at the setback site threefold.

The range in the observed dispersion ratio was found to vary with both time of day and wind speed (which are physically related to each other), with an increase in dispersion ratio associated with low wind speeds at night. Model results did not reproduce this influence. The consequence was that the degree of overprediction in modelled setback concentrations was greater (and more consistent) during the daytime, and reduced (albeit with a larger range) at night (see figure A8.12). We did not find the dispersion ratio to be dependent on wind direction in either the observational or modelled datasets.

Figure A8.11 Hourly dispersion ratio (modelled and observed) as a function of wind speed

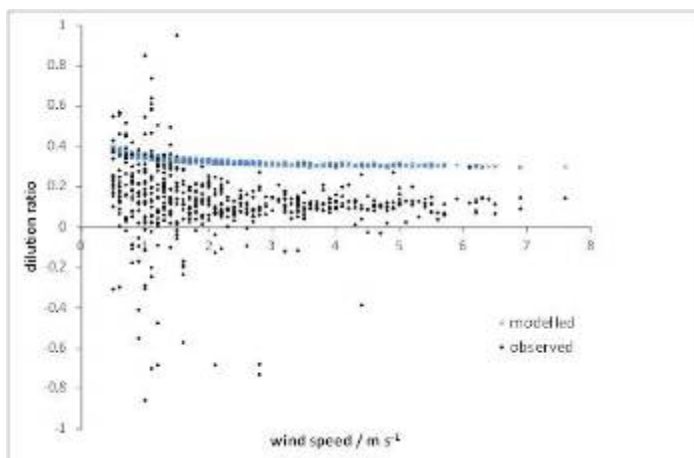
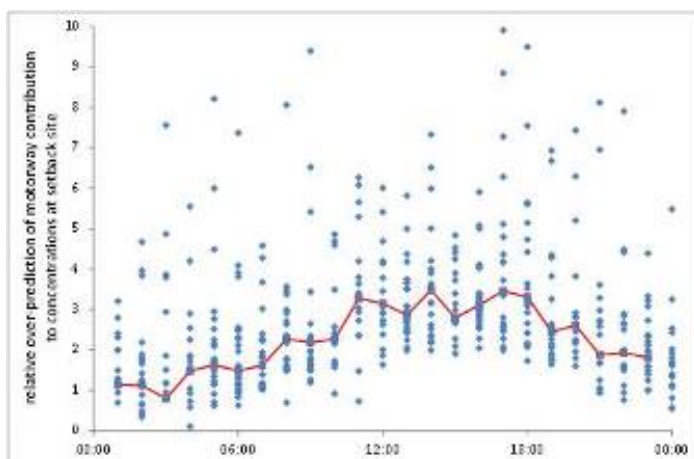


Figure A8.12 Ratio of modelled to observed hourly dispersion ratio as a function of time of day (blue dots), with diurnal hourly mean (red line)



A8.8 Evaluation of emission-weighted Ausroads results incorporating observed background data

A8.8.1 Methods

The ambient concentration estimated by Ausroads was calculated by weighting the **unit emissions** results by the emission estimates calculated using VEPM for the traffic fleet. This gave an estimate of the contribution of the motorway to downwind concentrations for each hour of the validation dataset. Taking advantage of the fact that concentrations upwind of the motorway were measured, the modelled impact of the motorway downwind could be added to the background concentrations and obtain cumulative concentrations. This allowed us to make a direct comparison with observed air quality at our downwind sites.

Selection of data to represent background air quality is a potential source of substantial error in assessments of roadside air quality. It is for this reason that our study design required the collection of simultaneous concentrations at setback sites on both sides of the motorway, so that we would have a highly local indication of background concentrations in all wind directions (Luke Street in the case of westerly winds and 25 Deas Place in the case of easterly winds). However, in most other situations (eg

modelling assessments for road projects) such highly local data is not available and background data is often taken from the nearest available monitoring site.

At the time of writing Auckland Council was preparing draft guidance for the identification and use of background data for industrial resource consents (Metcalf and Rolfe 2011), and the NZTA was also preparing national guidance for road project assessments. The Auckland Council guideline proposes acquiring data from one of a pre-selected list of Auckland Council permanent monitoring sites. The site chosen for the assessment should be 'representative of the assessment location'. The Penrose and Takapuna sites are recommended for 'urban roadside' locations, and the Henderson and Glen Eden sites for other 'urban' locations (and Musick Point for NO₂). Consequently we have used data from all five sites in our assessment.

The observational results presented in chapters A4 to A7 make it clear that the impact of motorway emissions on observations at the setback sites (Luke Street and 25 Deas Place) was relatively small, especially for PM₁₀. Therefore our evaluation mainly focused on the kerbside Deas Place Reserve receptor.

We executed two evaluation scenarios: a) in westerly winds only (so that the kerbside receptor was always downwind) and b) for all wind directions. Isolating easterly winds did not provide meaningful information because in easterly winds the contribution of the motorway to concentrations at Deas Place Reserve was modelled as zero and the observed local background was effectively the same as at the subject receptor, yielding a meaningless 100% prediction performance.

We present below a basic evaluation of the resulting performance of our predictions relative to observed hourly concentrations. Numerous statistical tests have been used to perform model evaluations each with their own limitations (eg Willmott et al 1985; Levitin et al 2005; Olesen and Chang 2010). An extensive or systematic model evaluation was beyond the scope of this project and we include here only scatter plots and some basic statistics, although the full dataset is available for extended analysis.

A8.8.2 NO_x, Westerly winds only

Figure A8.13 Estimated hourly NO_x concentrations (Ausroads/VEPM prediction + observed background concentration from Luke Street) versus observed hourly NO_x for kerbside site (Deas Place Reserve) in westerly winds only

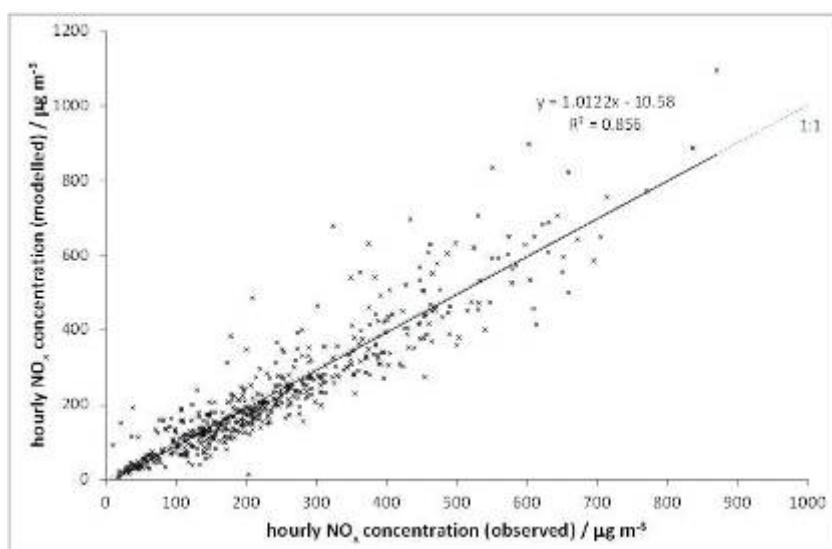


Figure A8.13 indicates what appears to be relatively good prediction performance. Most of the datapoints cluster around the 1:1 line. Closer inspection (not shown) reveals a slight tendency towards overprediction

in the daytime and underprediction at night, in line with the results presented earlier in terms of the **dispersion ratio**. However, the effect is relatively small and the most significant occurrences of underprediction are limited to low wind conditions in the daytime.

Figure A8.14 Estimated hourly NO_x concentrations (Ausroads/VEPM prediction + observed background concentration from Luke Street) versus observed hourly NO_x for setback site (25 Deas Place) in westerly winds only

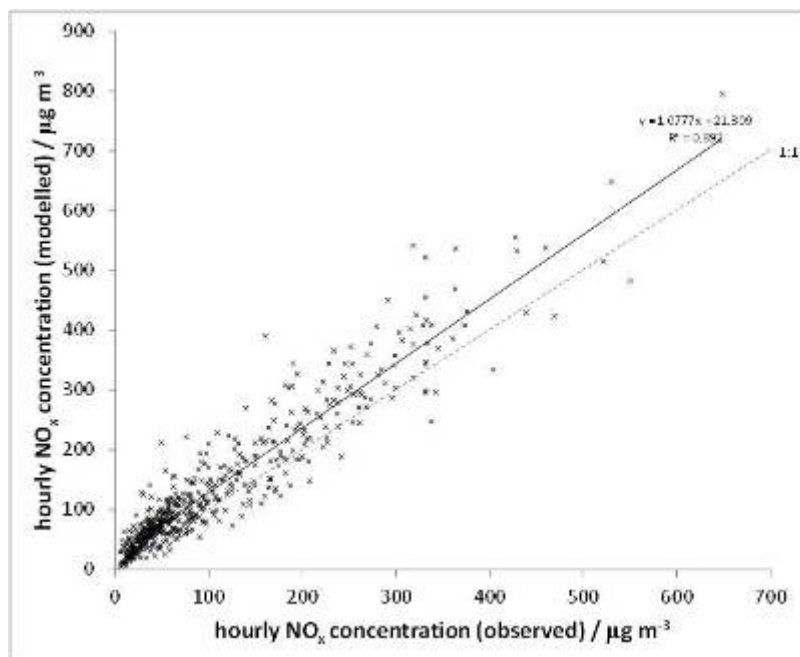
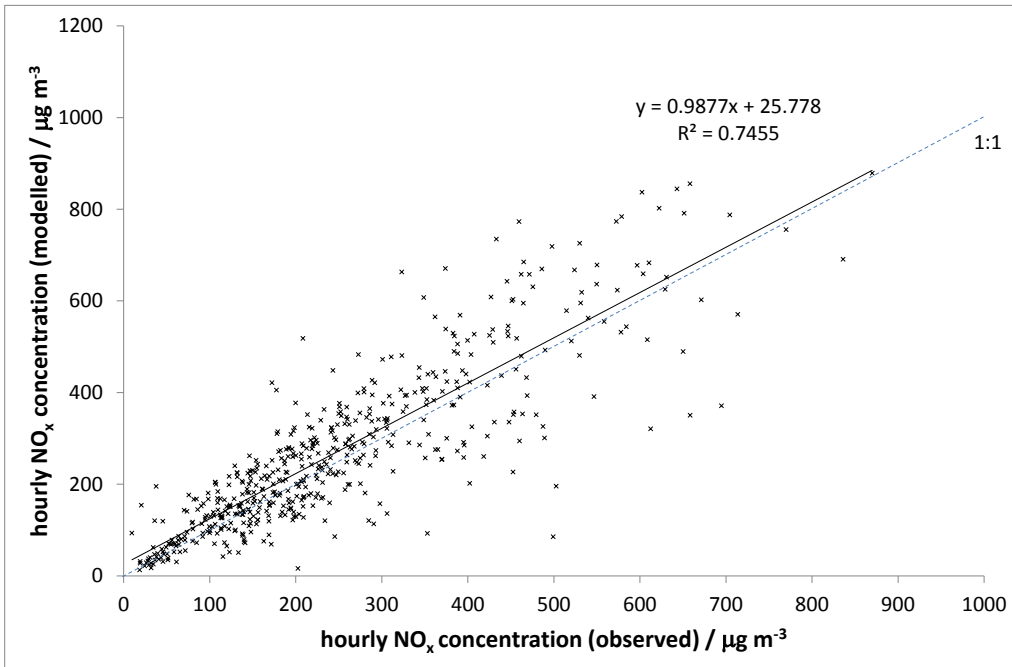


Figure A8.14 indicates slightly weaker performance compared with the prediction for the kerbside site (above), with a persistent tendency towards overprediction across the whole concentration range. Further analysis (not shown) indicates that over-prediction error is relatively moderate (0 – 50 μg m⁻³) in wind speeds above 3 m s⁻¹. At lower wind speeds the error tends to be larger in the daytime, or smaller (even underpredicting) at night. There is no clear evidence of any influence of wind direction, rainfall, solar radiation or humidity on prediction error.

The best performing non-local site for background NO_x was Penrose. Figure A8.15 shows predicted NO_x at Deas Place Reserve constructed from hourly NO_x concentrations at Penrose added to be the Ausroads+VEPM modelled contribution of the motorway, plotted against observed NO_x at Deas Place Reserve. Compared with using Luke Street as background (figure A8.13 above), the slope of the best fit has changed little, but the scatter has increased, lowering the R² from 0.86 to 0.75. Closer inspection reveals a tendency to overprediction at lower concentrations consistent with 'double counting' (ie the supposed 'background' data from Penrose contains a limited impact from the motorway in westerly winds).

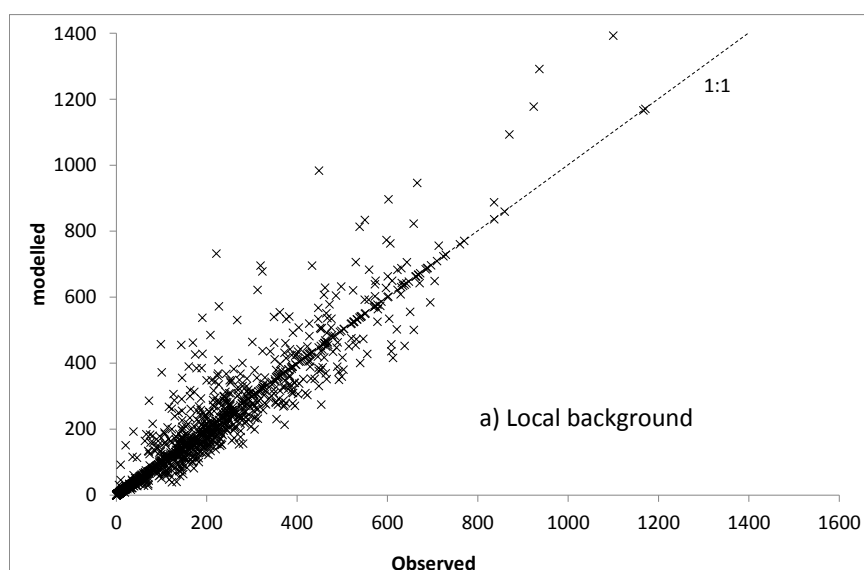
Figure A8.15 Estimated hourly NO_x concentrations (Ausroads/VEPM prediction + observed background concentration from Penrose) versus observed hourly NO_x for kerbside site (Deas Place Reserve) in westerly winds only



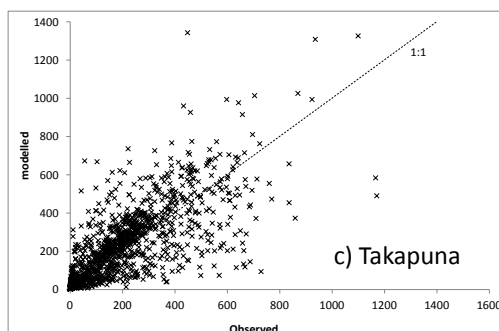
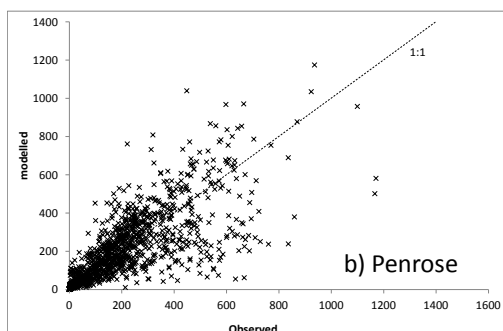
A8.8.3 NO_x, all wind directions

Figure A8.16 shows predictions of NO_x at Deas Place Reserve using: a) local background; b) and c) background from sites classed as urban roadside by Auckland Council; and d) and e) background from sites classed as urban by Auckland Council.

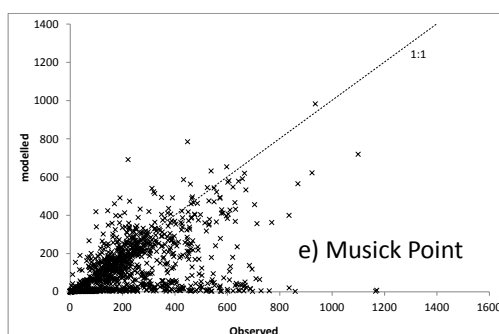
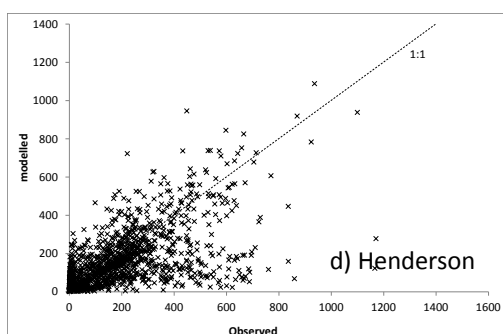
Figure A8.16 Estimated hourly NO_x concentrations (Ausroads/VEPM prediction + background concentration) versus observed hourly NO_x for kerbside site (Deas Place Reserve)



“Urban Roadside” sites



“Urban” sites



Note: Background data is sourced from a) study area (Luke Street in westerly winds and 25 Deas Place in easterly winds), b) Penrose, c) Takapuna, d) Henderson, e) Musick Point.

The Penrose and Takapuna sites were recommended by Auckland Council to represent ‘urban roadside’ receptors. Penrose is 5km from our study area and Takapuna is 20km. The sites are respectively 106m and 60m east of major motorways, ie they experience ‘downwind’ conditions relative to the motorway in roughly the same wind directions as each other, and our subject receptor of Deas Place Reserve. Figure A8.16 shows that using data from either of these sites to represent background reduces prediction

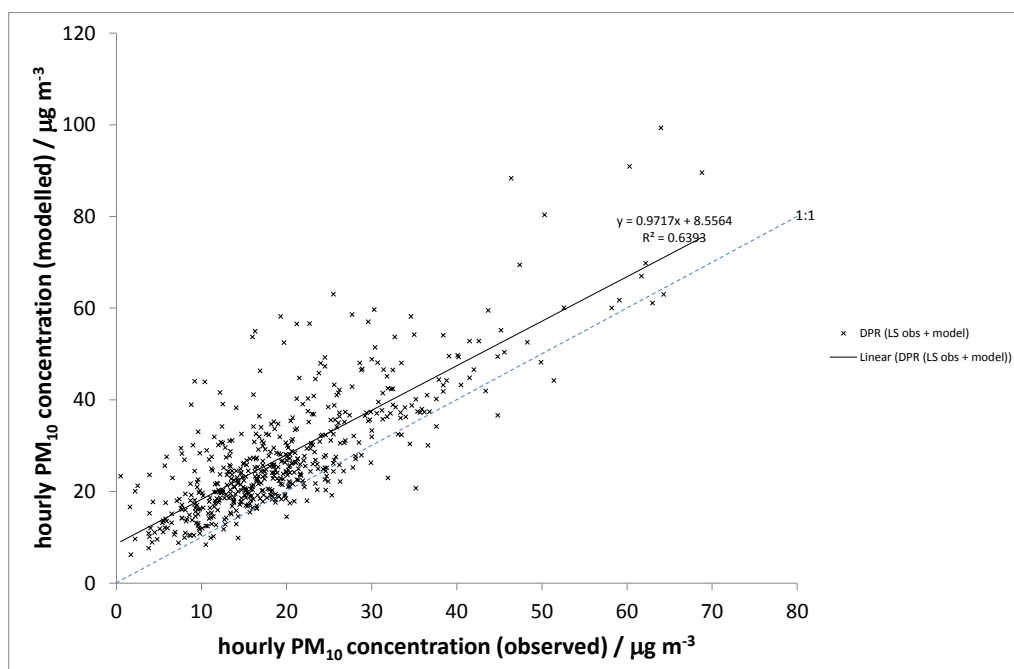
performance relative to using local data, with substantial underprediction at higher concentrations. There is somewhat greater scatter using data from the more distant Takapuna site.

Compared with the use of local background data, the use of data from Henderson substantially worsens prediction performance. A major reason for this is that the site is immediately to the west of a significant road source (Lincoln Road), so that in easterly winds concentrations are substantially elevated at Henderson whereas they are not at Deas Place Reserve due to it being upwind of the motorway.

Figure A8.16 e) shows the result of using data from Musick Point to represent background. This site, while only 10km from our study area, and therefore closer than either Takapuna or Henderson, provides very poor performance for a large portion of the dataset. This is related to Musick Point's geographical location on the coast in that it represents regional (oceanic) rather than urban background on easterly winds.

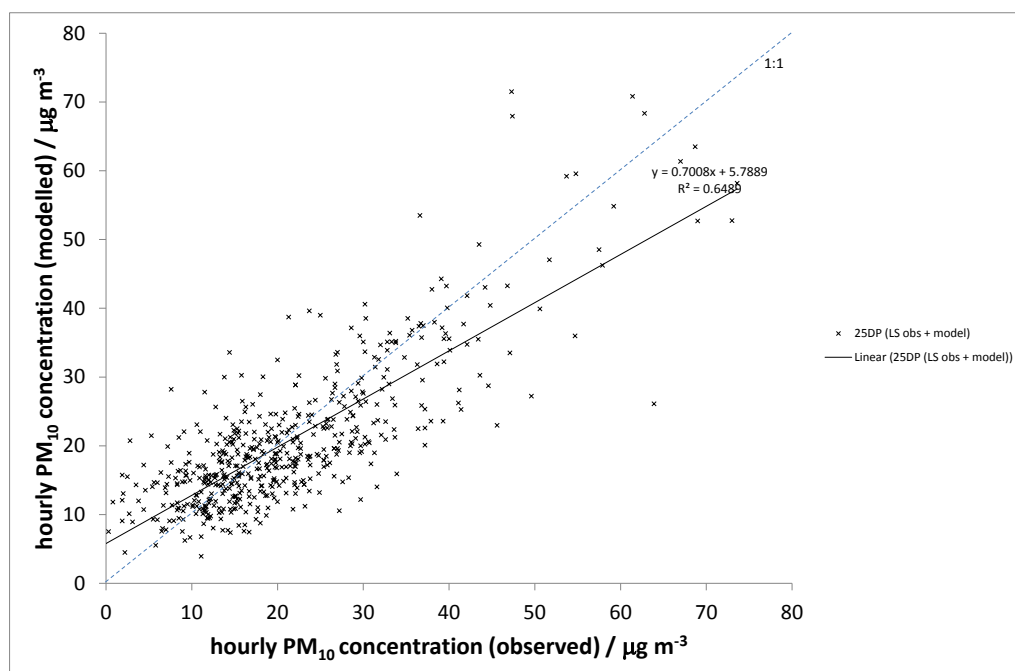
A8.8.4 PM_{10} , westerly winds only

Figure A8.17 Estimated hourly PM_{10} concentrations (Ausroads/VEPM prediction + locally observed background concentration from Luke Street) versus observed hourly PM_{10} for kerbside site (Deas Place Reserve)



PM_{10} was overpredicted by our modelling approach typically by 0 – 20 $\mu\text{g m}^{-3}$ (5 – 10 $\mu\text{g m}^{-3}$ on average). There was a tendency for the overprediction to be slightly increased in low winds, especially in morning peak hours. During the morning peak, there was a consistent absolute overprediction error of 10 – 40 $\mu\text{g m}^{-3}$, at a time when observed concentrations could range from 10 – 70 $\mu\text{g m}^{-3}$. Assuming Ausroads is performing well for the kerbside site (as suggested by the NO_x results above) this could imply an overprediction by VEPM of PM emissions on the motorway.

Figure A8.18 Estimated hourly PM₁₀ concentrations (Ausroads/VEPM prediction + locally observed background concentration from Luke Street) versus observed hourly PM₁₀ for easterly setback site (25 Deas Place)



Compared with the kerbside site, it can be seen that the model predictions for the setback site were relatively poor. We could find no clear relationship between the absolute error and wind direction. However, there was a clear relationship with time of day, with overestimation occurring particularly in the morning and with a weak tendency towards underestimation in the late afternoon and evening.

This error pattern is distinctly different from the corresponding pattern for NO_x, with the main difference being the tendency to underpredict after mid-day, in contrast to overpredicting for NO_x. This implies that in the evening the overestimation of PM₁₀ was being compensated by an additional source of PM₁₀ that was not being modelled.

Figures A8.19 and A8.20 show predicted PM₁₀ at Deas Place Reserve based on Penrose and Pakuranga PM₁₀ data respectively. The R² has deteriorated from 0.64 using Luke Street data, to 0.29 and 0.39 using Penrose and Pakuranga data respectively. In fact, better performance is provided by using PM₁₀ data from Takapuna, a road-influenced site 20km to the north of the study area, as shown in figure A8.21.

Figure A8.19 Estimated hourly PM₁₀ concentrations (Ausroads/VEPM prediction + observed background concentration from Penrose) versus observed hourly PM₁₀ for kerbside site (Deas Place Reserve)

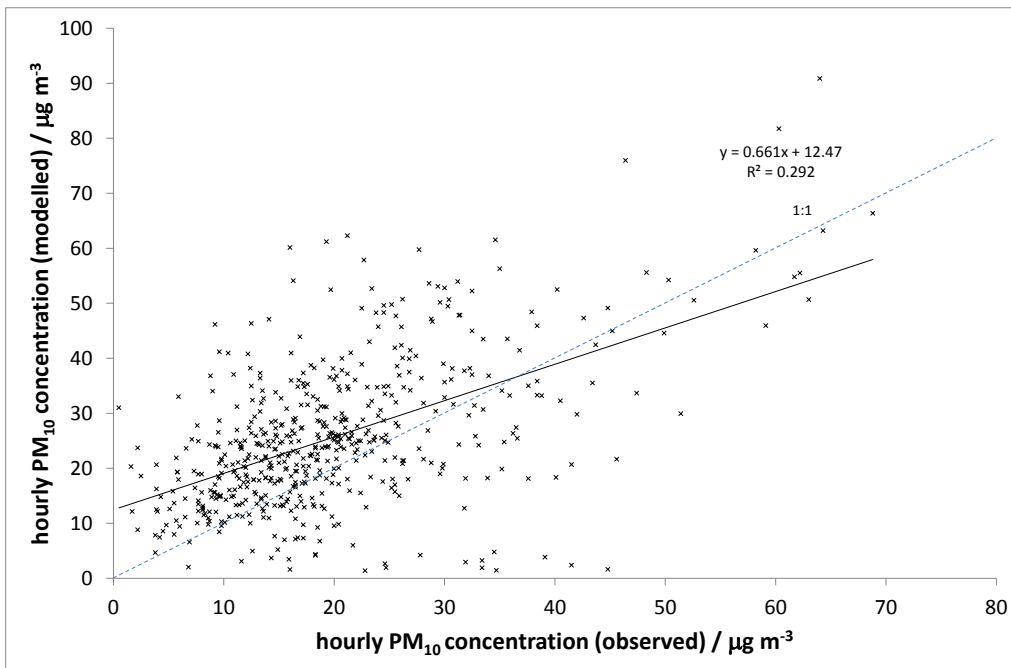


Figure A8.20 Estimated hourly PM₁₀ concentrations (Ausroads/VEPM prediction + observed background concentration from Pakuranga) versus observed hourly PM₁₀ for kerbside site (Deas Place Reserve)

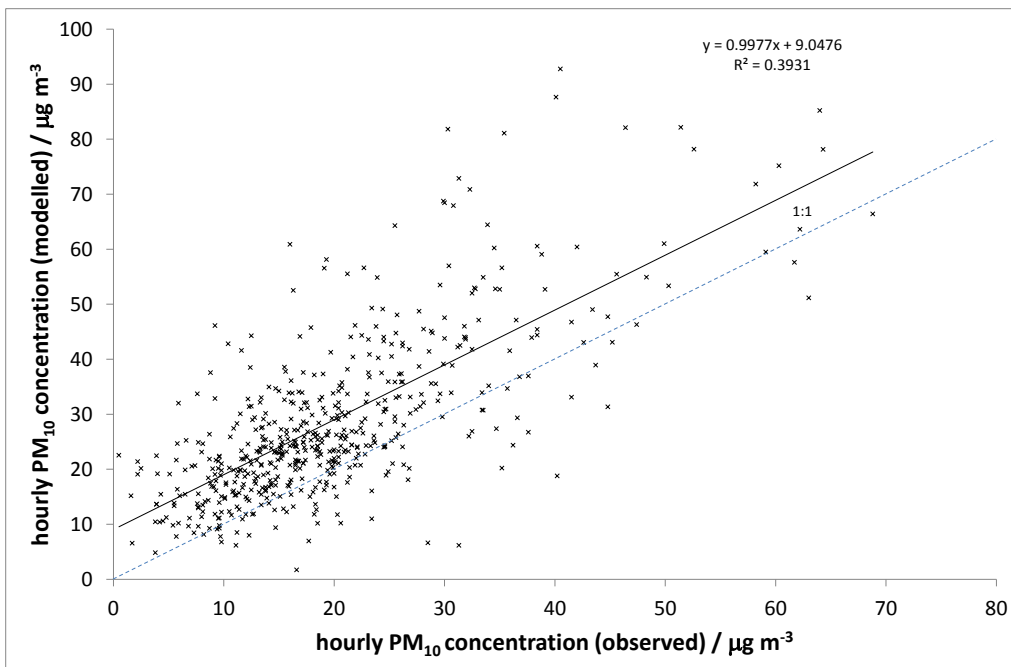
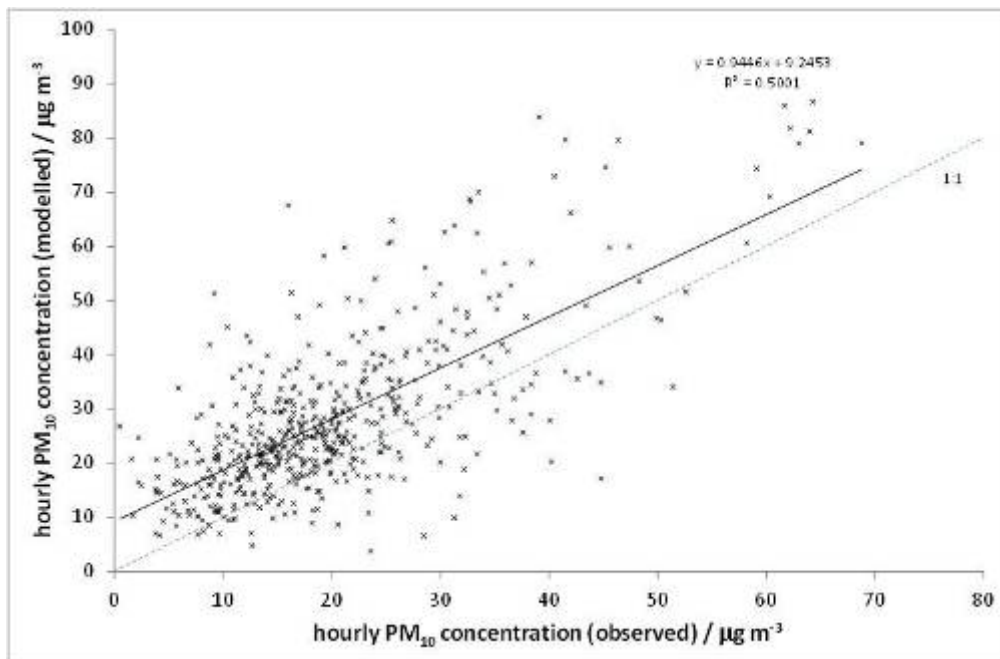
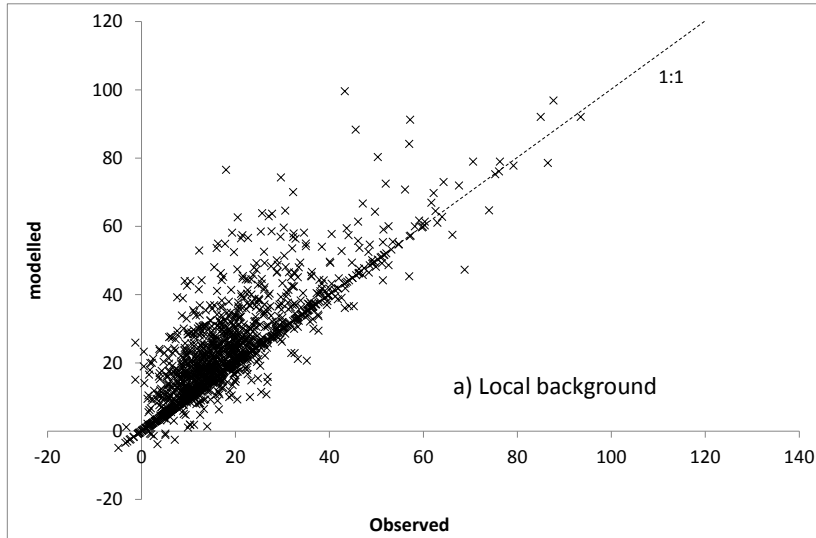


Figure A8.21 Estimated hourly PM_{10} concentrations (Ausroads/VEPM prediction + observed background concentration from Takapuna) versus observed hourly PM_{10} for kerbside site (Deas Place Reserve)

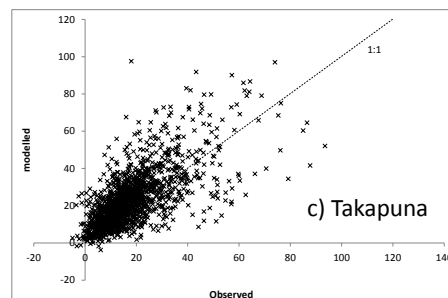
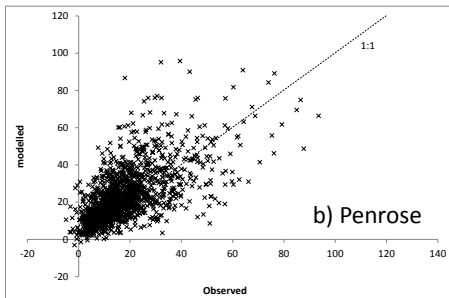


A8.8.5 PM₁₀, all wind directions

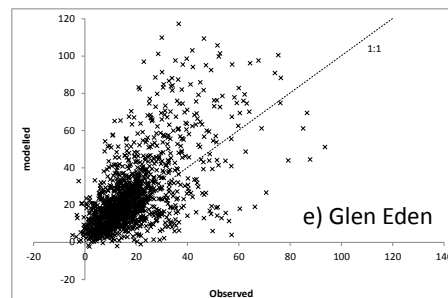
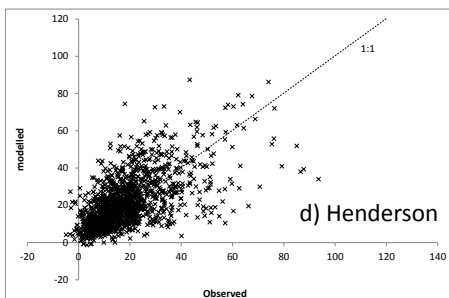
Figure A8.22 Estimated hourly PM₁₀ concentrations (Ausroads/VEPM prediction + background concentration) versus observed hourly PM₁₀ for kerbside site (Deas Place Reserve)



“Urban Roadside” sites



“Urban” sites



Background data was sourced from a) study area (Luke Street in westerly winds and 25 Deas Place in easterly winds), b) Penrose, c) Takapuna, d) Henderson, e) Glen Eden.

Due to the relatively low sensitivity of the PM₁₀ metric to road traffic emissions, and the inherent variability of data reported by beta attenuation monitors, the variation within figure A8.22 is not as visually apparent as it is for NO_x. A simple linear regression analysis indicates the highest R² for Takapuna (0.46), with slightly lower R² values for Penrose (0.40), Glen Eden (0.39) and Henderson (0.37).

A8.8.5.1 Limitations in analysing PM_{10}

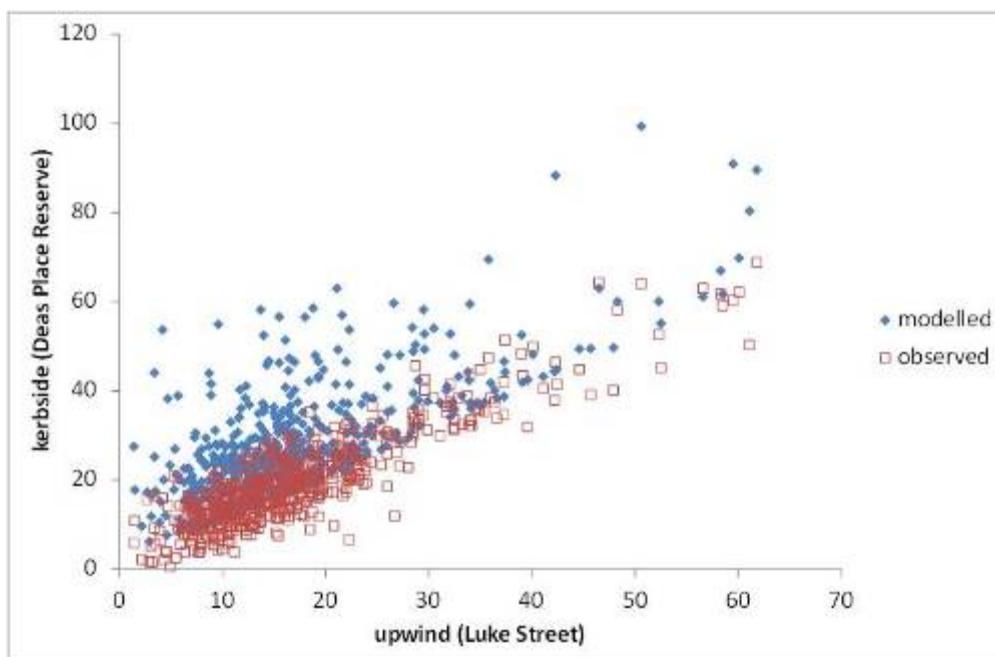
Our study design assumption of no emission sources or sinks within the dispersion micro-study area (formed by the space between the kerbside and setback sites) was more problematic for PM_{10} compared with NO_x . We also had less confidence in the ability of Luke Street to represent 'background' concentrations. In both cases this was primarily due to the probability of the presence of intermittent domestic heating and other small point sources in the area. For instance, domestic refuse burning was anecdotally observed by project researchers at various times during the campaign.

This is illustrated by the large number of hourly observations within the IOP for which the implied assumption in westerly winds that PM_{10} (Deas Place Reserve) $>$ PM_{10} (25 Deas Place) $>$ PM_{10} (Luke Street) is not true. For example, PM_{10} at the 'upwind background' site, Luke Street, was greater than at the downwind kerbside of Deas Place Reserve for 29% of hourly observations. This implies that Luke Street may have been affected by additional sources of PM_{10} to which the other sites were not exposed, or were exposed to a lesser degree. This may also apply to the other two sites.

These problems were compounded by the low sensitivity of PM_{10} instruments, relative to NO_x instruments. Although these were the main reasons why the experiment and much of the analysis was primarily based on observations of NO_x , generally in New Zealand air quality assessments consider assessment of PM_{10} to be a priority.

The results are illustrated in figure A8.23. This shows considerable scatter in the relationship between upwind and downwind (kerbside) observed concentrations of PM_{10} , and it is clear that upwind concentrations can often be higher than downwind. The scatter in the modelled relationship is not substantially greater (except for a few outliers). Hence the uncertainty in the assessment of PM_{10} arises as much from the observations as from the modelling exercise.

Figure A8.23 Scatter plot of hourly PM_{10} (in $\mu g m^{-3}$) at the kerbside versus the upwind sites as modelled (solid diamonds) and observed (open squares) in westerly winds only



A8.9 VEPM/Ausroads modelling – discussion of results

A8.9.1 Overview

Our major findings from this modelling exercise were as follows.

- 1 Ausroads underpredicted roadside dispersion at a distance of 30m – 150m from the road.
- 2 Using off-site meteorological datasets generally (but not consistently) led to underprediction of the contribution of the motorway to local air quality, relative to the use of a local dataset. An extract from the ARC artificial dataset representing our study area gave results closer to the local dataset than the off-site datasets.
- 3 The Ausroads+VEPM combination performed well in predicting kerbside NO_x.
- 4 The Ausroads+VEPM combination overpredicted kerbside PM₁₀.
- 5 Having local upwind data to represent background concentrations substantially improved overall prediction performance.

A8.9.2 Source of meteorological data

The substantial impact of modelling with off-site meteorological data (leading to lower predicted concentrations at both kerbside and setback receptors relative to using local data) is indicative of a representativeness problem for sites far from the assessment area. In general, in situations where local meteorological data is not available, the use of off-site data, which is often derived from deliberately selected flat and open sites, will lead to the overprediction of dispersion and underprediction of concentrations, especially if the subject area has a rougher surface or more complex urban topography.

A8.9.3 Underprediction of dispersion by Ausroads

Our first finding would suggest that Ausroads is generally conservative when predicting concentrations at roadside receptors in the ~100m range. This conservatism appeared to apply in the daytime hours only, when the contribution of the motorway was overpredicted by approximately a factor of 3. That, and the lack of dependence on wind direction, suggests that the error was related to a form of enhanced mixing which was not included or accounted for in the Ausroads formulation. The error could not be attributed to the use of the Luke Street site rather than the airport site as a source of meteorological data for modelling. Introducing an estimate of atmospheric stability (rather than the default scenario of assumed constant neutrality) made a small improvement in the modelled dispersion, but the error essentially remained.

We used the clues provided by the relationship between the error and both wind speed and daylight hours to speculate that the sources of unmodelled mixing might be

- traffic-induced turbulence
- enhanced local solar-driven thermal turbulence
- enhanced mechanical turbulence due to roughness elements (mainly houses) which was inadequately captured by the constant roughness length used in Ausroads.

Even though it is not possible with our dataset to evaluate the performance of Ausroads in the 0m – 30m range, it is well documented that Gaussian plume models, like Ausroads, do not perform well at this range because a number of fundamental assumptions are not valid.

Consequently, we cannot determine whether Ausroads was contributing errors to modelling concentrations at the kerbside. However, if we assume that, due to the short distances involved (our kerbside site was ~2m from the edge of the nearest lane of traffic) that Ausroads was contributing minimal error, then the good performance at predicting NO_x at the kerbside suggests that the emission model VEPM was not contributing significant error either (although the corollary that both models were, on average, contributing equal and opposite errors cannot be ruled out).

A8.9.4 Assessment of net (cumulative) effects

The significance of the findings regarding Ausroads depends on the objectives of the assessment being performed. Commonly, the aim is to assess the net impact of both the subject road, and other 'background' sources, ie absolute concentrations. As illustrated in chapters A6 and A7, the motorway's contribution to net concentrations was substantially larger for NO_x and NO₂ than for PM₁₀. Thus predictions of net effects were likely to be more sensitive to aspects pertaining to vehicle emission and roadside dispersion modelling for NO_x than they were for PM.

Given the good performance in predicting NO_x, we assumed that Ausroads was not contributing to prediction error at the kerbside. In that case, the overprediction in PM₁₀ could be attributed to either

- overprediction of PM emission rates by VEPM, or
- invalidity of the assumption that the Luke Street site represented the background, ie all sources affecting the kerbside site except the motorway.

Furthermore, our ability to assess the performance of the Ausroads+VEPM combination to predict PM₁₀ was severely limited by the low sensitivity of the regulatory technology for measurement of PM₁₀ relative to the range of inter-site concentration differences being observed. Consequently, our conclusions surrounding PM₁₀ were necessarily tentative due to large underlying uncertainties. In general, our modelling estimates led to an overprediction of PM₁₀ at the kerbside, relative to observations. The magnitude of model overprediction was not constant during the day, but was reduced, on average, in the afternoon and evening. This could imply the presence of a systematic additional source affecting the downwind sites. Further analysis is required to tease out the characteristics of this potential source. At present we speculate that it might include domestic woodsmoke differentially affecting the setback 25 Deas Place receptor due to it being embedded among residences, or re-suspension of road dust from the motorway. Although VEPM includes a prediction of brake and tyre wear emission rates, neither VEPM nor Ausroads describes the dependence of re-suspension of dust from the road surface on both traffic speed, volume or wind speed, the latter two of which are typically elevated in the afternoon.

Interference from other PM₁₀ sources affected our three measurement sites to different and varying degrees in a way that could not be modelled. Researchers on the ground reported sporadic outdoor burning, domestic woodsmoke emissions, and the potential for smoking and emissions from stationary cars biasing our measurements.

We anticipate that full analysis of the mobile datasets (see annex A) will help to establish the likelihood and magnitude of any error regarding bias at Luke Street.

Our limited analysis of the use of alternative sources of background data indicated that:

- using a local dataset substantially improved model prediction performance relative to an off-site source
- using data from closer sites (eg Pakuranga, Botany, Penrose) did not necessarily improve performance relative to more distant sites (eg Takapuna)

- predictions for NO_x appeared to be more sensitive to selection of a background site than PM₁₀. However, this was due not to a weakness in the modelling procedure, but to the low sensitivity of PM₁₀ as a measure of traffic emissions and low variability in PM₁₀ across Auckland, relative to NO_x.

Further analysis is required to explain and generalise the role of different sources of background data in roadside assessments.

A9 Site-optimised semi-empirical modelling

A9.1 Overview

Site-optimised semi-empirical (SOSE) modelling is a simple approach which takes a time series of air pollution and wind observations at a given location and summarises the way that, on average, the pollution concentrations are related to changes in local winds and emission rates at the location. It can be used to generalise, interpolate and extrapolate limited time series observations. SOSE modelling has been gradually developed over the last decade and this development is likely to continue in the near future. The rationale for its inclusion in this project was to evaluate its performance at a motorway site for the first time, compare its performance with the more physically based but complex emission-dispersion approach (such as VEPM+Ausroads), and explore its ability to provide information about background concentrations and emission rates for the first time.

In this section, we present the modelling results for the SOSE model. The model has been tested for many different environments internationally but this was the first time the model was tested specifically for a motorway source and the first time the model was tested comprehensively for any site in Auckland. All analyses, unless otherwise indicated, were carried out on 10-minute averaged data. For the majority of the results, the week 31 May to 7 June 2010 was chosen as an example. This week was chosen based on completeness of the dataset and because of its representativeness of the dataset as a whole.

- 1 The model was tested using meteorological data that was *not* co-located with the air quality monitoring. The relative merits of co-located and not co-located (but better sited) meteorological data as an input into SOSE modelling were compared.
- 2 Estimates of the background concentration were made and compared with observations. The intercept regression parameter from SOSE was the background concentration. Using the fact that data had been collected on both sides of the motorway concurrently, it was possible to verify the average daily profile of the background concentration with observations on the opposite side of the road.
- 3 Estimates of the daily profile of the 'box height' were made. This was achieved by including the emission data (output from VEPM) into the model, so that the slope of the regression term in SOSE became the inverse of the 'box height'. Given that all pollutants are subject to the same meteorology, it can be expected that the box heights profiles for each of the pollutants will be the same. Any systematic difference can be attributed to differences in the relative emissions estimates from VEPM.

A9.2 Background

A9.2.1 What is SOSE?

In its simplest form, the SOSE model assumes that all variability in air pollution concentration at a particular site at a particular time of day, day of the week and season is determined entirely by the wind speed and direction (whether windward or leeward of the source). Roadside sites are particularly suited to the latter assumption in that traffic volumes, composition and speeds (and hence emissions) tend to follow very predictable temporal patterns, especially at sites with higher volumes. The roadside application is also very suited to the approach in that monitored concentrations are very sensitive to whether the site is upwind or downwind of the line source, and it is relatively simple (compared with point or area sources) to determine the upwind/downwind status of a monitoring site.

The model can be run for any of the major urban pollutants (CO, NO_x, PM₁₀) and for any combination of meteorological station and air pollution monitoring station (either co-located or not). The model requires training data (three to four weeks minimum to cover a sufficiently wide range of meteorological conditions) and can then be used to provide predictions for the same date range as the training set or for independent data. Once trained, the model requires meteorological data from the same site as was used for the training.

The model had previously been tested and was found to be reliable for urban and suburban sites (major arterials) in Hamilton, New Zealand, for relatively flat terrain, and for major roads in both flat and complex terrain in urban areas in Italy. At the time of our study, the model had yet to be tested on sites adjacent to a motorway and tested comprehensively in Auckland.

A9.2.2 Brief description of SOSE (and similar models)

Most physically based models require detailed information about the site geometry, both in terms of the nature and layout of the sources, as well as the surface topography that modulates the surface wind flows. In urban and suburban environments and in areas with significant natural topography, this can be quite complex. While physically based models show some success in predicting air pollution concentrations, it is unrealistic to expect them to be able to account fully for these complexities due to our limited understanding of dispersion processes in such environments. In addition, in situations of multiple sources of pollution (road, home heating, natural etc) it is unrealistic to expect these models to be able to account for air pollution concentrations, both from the point of view of the source and the subsequent dispersion.

An alternative approach is to use empirically based models, including artificial neural network (ANN) models, which have been shown to produce reasonably reliable estimates of roadside pollution concentrations in a variety of meteorological conditions (Gardner and Dorling 1999) and in complex topographical environments. However, in the case of entirely empirically based models, the mathematical relationships between the model input and output is not revealed (they are, to a large extent 'black-box' models). As such, they may not be able to be used to answer many physically based questions that are of interest from both a scientific and regulatory point of view.

As an alternative to entirely physically based models and ANN modelling, a SOSE model has been developed. SOSE is a box model based on a simple linear regression of the concentration on the wind speed (plus an offset term) (Dirks et al 2002; 2003). The slope of the regression (the regression coefficient) is proportional to the emission rate from the road and inversely related to the 'box height' (a measure of the vertical spread of the plume), and the intercept represents the background concentration, the concentration leeward (upwind) of the road of interest. Central to the model is that the data is sorted by time of day, and also sorted into leeward and windward wind conditions in relation to the line source (the motorway in this instance), and, in its simplest form, by weekdays and weekends, to account for differences in emissions that generally occur between these two day types.

The advantage of this type of modelling approach is that it has limited data requirements (only the wind speed and direction are needed as input), and it tends to produce very good results in a wide range of topographical and emissions environments as well as for most of the major vehicle pollutants (CO, NO, NO₂, PM₁₀ and BTEX). It also has some physical basis in that one can learn about the physical environment in a way that is not possible using empirical models such as ANN. For example, it is possible to make estimates of the impact of a road on the local air quality of the region as well as estimates of the daily profile of the 'box height' (the vertical mixing throughout the day).

A disadvantage of semi-empirical modelling is that it requires on-site training data. It also tends to underpredict peak concentrations, consistent with all regression-based models and as such is not ideal for predicting extreme events such as exceedances, unless they occur regularly within the dataset.

For these reasons, SOSE complements physically based models (falling somewhere between physically based models and ANN models). In addition, SOSE also complements LUR models as it provides high-time-resolution information for a specific site. In contrast, LUR models provide high-resolution spatial information but limited temporal resolution. In this way, a combination of LUR and SOSE provides a semi-empirical alternative to the physically based models such as the roadside corridor model.

	Physical	Semi-empirical
Temporal	Gaussian dispersion model (eg Ausroads, RCM)	SOSE
Spatial		LUR

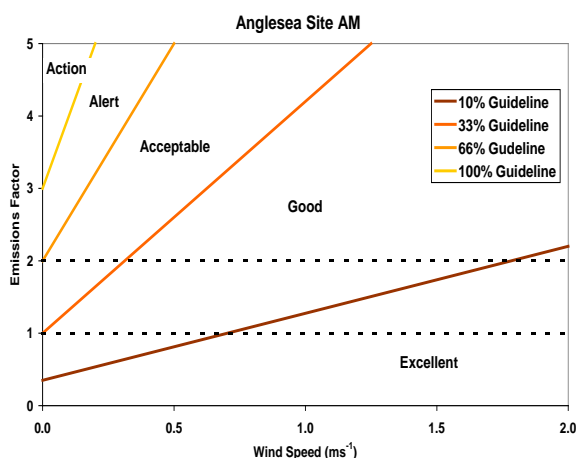
A9.2.3 SOSE's history and applications

SOSE was originally developed in 1995 for Environment Waikato (then Hamilton Regional Council) which was in need of a simple approach to modelling carbon monoxide for each of its two air quality monitoring sites. In its original form, the model was able to reliably interpolate for missing data by training the model on existing data. This has obvious benefits in order to prevent biases in the reporting of long-term averages when there are significant periods of missing data. It was also possible to use the model to investigate the conditions (wind speed and emission factor relative to current levels) required to reach each of the environmental performance indicator classifications proposed at the time (see table A9.1 and figure A9.1). A description of the basic model based on data from Environment Waikato is provided in Dirks et al (2002) and Dirks et al (2003).

Table A9.1 Proposed environmental performance indicator classifications (MfE 1997)

Category	Maximum Measured Value	Comment
Action	Exceeds the guideline	Completely unacceptable, by national and international standards
Alert	66-100% of guideline	A warning level, which can lead to exceedances if trends are not curbed
Acceptable	33-66% of guideline	A broad category, where maximum values might be of concern in some sensitive locations but generally at a level which does not warrant dramatic action.
Good	10-33% of guideline	Peak measurements in this range are unlikely to impact air quality
Excellent	<10% of guideline	Of little concern, if maximum values are less than one tenth of the guideline, average values are likely to be much less.

Figure A9.1 Emission factor (relative to current value) required to reach each of the proposed environmental performance indicator classifications for the Anglesea site during morning rush hour

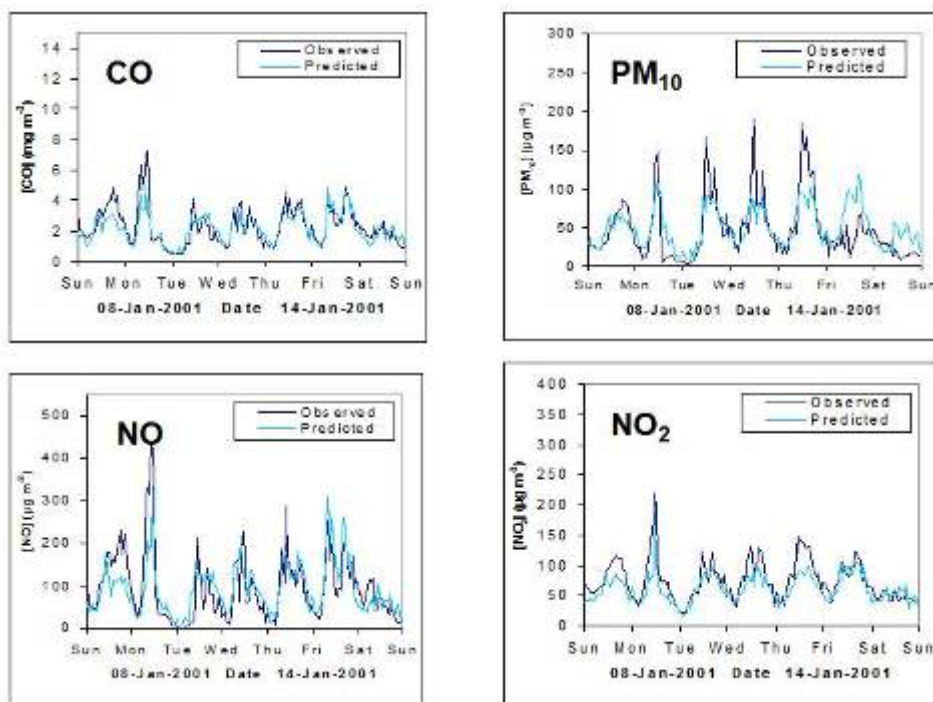


Source: Dirks 2001

The model was originally designed and tested on carbon monoxide as the sole pollutant and for Hamilton which is essentially flat terrain. At the time, only short periods of data were available for testing the model, so it was not possible to test the consistency of the model parameters from one year to the next. The modelling approach was therefore developed further NO, NO₂ and PM₁₀ in addition to CO. Figure A9.2 is an example of one week of winter data for the Aosta Valley. The model predictions are generally good (better for the gaseous pollutants than for PM₁₀) but there is a tendency to underpredict peak concentrations, as mentioned previously.

It is also significant that the model parameters are consistent from year to year. This means the model parameters from one year can be used to predict concentration from another year, in the absence of any significant changes in the emissions patterns (see figure A9.3). It also demonstrates that predictions based on independent data are very similar to those made on non-independent data. The model was also shown to perform well in an area of complex topography so was not restricted to flat environments (Dirks et al 2006).

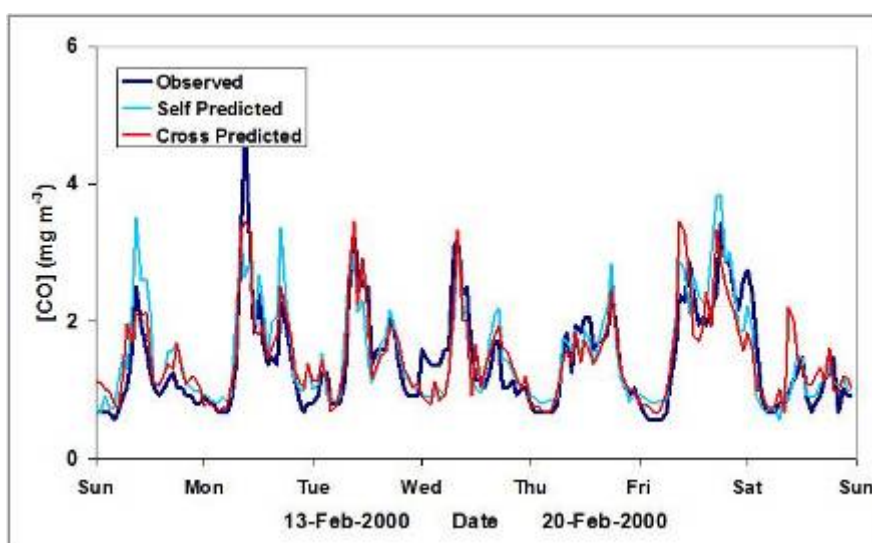
Figure A9.2 SOSE model output for winter-time in the Aosta Valley



Source: Dirks et al 2006

The SOSE model has since been implemented for roadside sites in several cities worldwide including for busy intersections in India (Gokhale and Pandian 2007), congested streets in Hong Kong (He et al 2009) and also for roadways in Greece (Kassomenos et al 2004). The model has been found to be effective in predicting CO, NO, NO₂ and PM₁₀ concentrations (Dirks et al 2003) as discussed above, and more recently, for BTEX, pollutants for which road traffic emissions represent a significant source (unpublished).

Figure A9.3 SOSE model predictions for CO for the Aosta Valley with model parameters from one year used to predict the other



This is an example of one week's observations and predictions.

An important application of this model, and one that was tested for the first time in this study, is the ability of the model to estimate background concentration. While estimates have been made in the past for other cities, it has never been possible to verify the estimates as this requires simultaneous monitoring on both sides of a road. Once this is known, it is possible to estimate the contribution of the road to the overall air quality of the region after the road is already in place. It can also be used to predict the pollutant concentrations for a range of what-if scenarios associated with changes in emissions patterns, taking into account the existing background concentrations. It is also possible to estimate what the air quality of the region would have been like in the absence of the road source. This has useful implications for road planning.

A9.3 Methods

A9.3.1 Model formulation

In order to optimise the model (and hence determine the values of the parameters needed to make predictions), linear regressions of pollutant concentrations on the inverse of the wind speed were performed for each time of day according to the sorting outlined above. A small offset was applied to the wind speed in order to avoid severe overpredictions in very light wind speeds. The regression coefficient (slope of the regression) represents the ratio of the source strength and the box height while the intercept of the regression represents the background concentration.

In the SOSE model, the emission rate Q ($\mu\text{g m}^{-1} \text{s}^{-1}$) is assumed to be constant along a road and the pollutants are mixed uniformly within a two-dimensional box of height Δz (m) (see figure A9.4). The horizontal wind speed (m s^{-1}), which is assumed to be uniform within the layer, removes the pollutants through advection. At the same time, pollutants are introduced into the box through advection of the background concentration. The concentration C ($\mu\text{g m}^{-3}$) under steady-state conditions and for conditions when the monitor is leeward (downwind) of the road is given by:

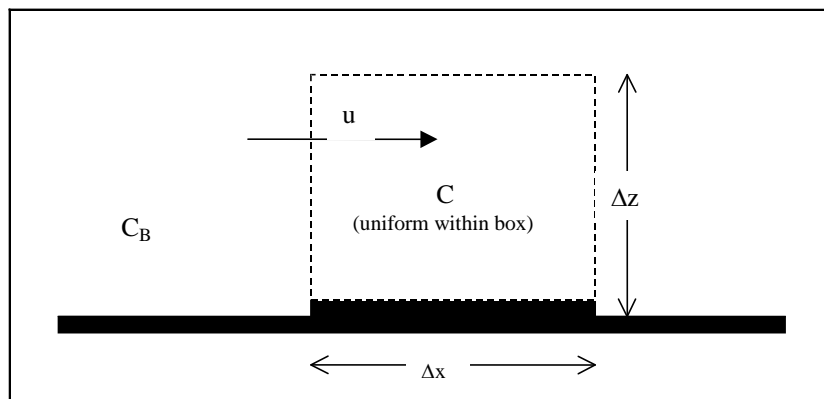
$$C = \frac{Q_l}{\Delta z(u + u_o)} + C_l$$

While for downwind conditions (when the monitor is upwind of the road), the air pollution concentration is given by:

$$C = \frac{Q_w}{\Delta z(u + u_o)} + C_w$$

where the subscripts l and w represent leeward and windward conditions, respectively. The u_o term is the 'wind speed offset' (in m s^{-1}), which is included to avoid severe overprediction in very light wind speeds and is determined empirically through the minimisation of the model root-mean-squared error (RMSE).

Figure A9.4 Schematic of a box model



In the absence of any emissions data, the optimum model parameters ($Q\Delta z^{-1}$, C_i , $Q_w\Delta z^{-1}$ and C_w) are found by performing linear regressions of air pollutant concentration (C) on the wind speed function $(u + u_0)^{-1}$ for leeward and windward conditions separately for each time interval throughout the day across all days in the dataset. If the strength of the line source is known, then the regression parameters become Δz^{-1} and C_i for the leeward conditions. This has the advantage that the box height may be estimated and there is no longer a need to separate weekdays from weekends. A full description of the model is given in Dirks et al (2002; 2003).

A9.3.2 Role of SOSE modelling in the study design

The SOSE modelling was introduced into the study design for the following reasons:

- The model had never been tested for a motorway. The model validation would be unique in this regard.
- Until our study, the model had only been tested on data from single sites. In particular, while the model was able to predict empirically the background component to the pollution, there had never been an opportunity to compare this empirical prediction with the observed concentration on the other side of the road. Being able to estimate (reliably) the background concentrations had important implications as it allowed the impact of the motorway (over and above the background) to be quantified.
- A consistency in the box height between pollutants allowed us to verify that the emissions estimates were in proportion to each other. If this was the case, it might allow the model to be transferred from one site to another without the need for recalculating the model parameters. Model output could be obtained using wind data as well as emissions estimates for the new site using VEPM.
- The model had always been run with co-located meteorological data. With the current dataset consisting of three meteorological sites and three air pollution monitoring (all in quite different locations in terms of land-use areas, it was possible to identify which site provided the most 'representative wind information for the purpose of modelling. SOSE does not require the wind measurements to be 'correct', merely that the wind is consistent in terms of how it influences pollution concentrations.

A9.4 Inputs

SOSE generalises the variability in an air quality time series from a specific fixed site. We therefore used the IOP data from the three project fixed sites (Luke Street, Deas Place Reserve and 25 Deas Place), as described in chapters A5 and A6, as the 'subject' and input dataset for the modelling. Wind data was also taken from the same three sites. Emission rates were the same as those generated using VEPM for the purposes of emission-dispersion modelling as described in chapter A8.

A9.5 Results

A9.5.1 Basic model output

In the first instance, the basic model was run based on training data that consisted of the whole of the IOP time period using co-located data (ie Luke Street meteorological data to model the air pollution concentrations at Luke Street). No emissions data was included as an input: the model predicted concentrations based solely on the wind speed and direction (and the time of day and day of the week- whether weekday or weekend).

Figures A10.5 to A10.10 are examples of one week of model predictions for each of the sites and pollutants, plotted both as a time series as well as predicted versus observed concentrations. The week was chosen based primarily on dataset completeness. Table A9.2 presents the model statistics for the whole of the dataset for each pollutant and each site.

Based on these results and similar graphs for the other weeks, the following observations were made:

- Model performance was generally good in that the predicted concentrations, to a large extent, captured the temporal variability in the observed concentrations most of the time for all three monitoring sites and across all pollutants. The more the regression line deviated from the 1:1 line, the less the model captured the temporal variability.
- There was a tendency to underpredict peak concentrations and overpredict low concentrations. This is a consequence of all regression-based models and represents a limitation associated with this modelling technique.
- The SOSE model performed better for the gaseous pollutants than for PM_{10} . This was observed across all three sites. The model performance was significantly better for NO_x than for CO.
- There were periods of significant divergence between model-predicted and observed concentrations. The model performed particularly poorly for CO at Deas Place Reserve ($r = 0.54$) compared with the other sites ($r = 0.73$ and $r = 0.67$ for Luke Street and 25 Deas Place, respectively).

Figure A9.5 Time series of one week of observed and SOSE-predicted concentrations for the Luke Street site: a) CO, b) PM₁₀, c) NO₂, d) NO_x

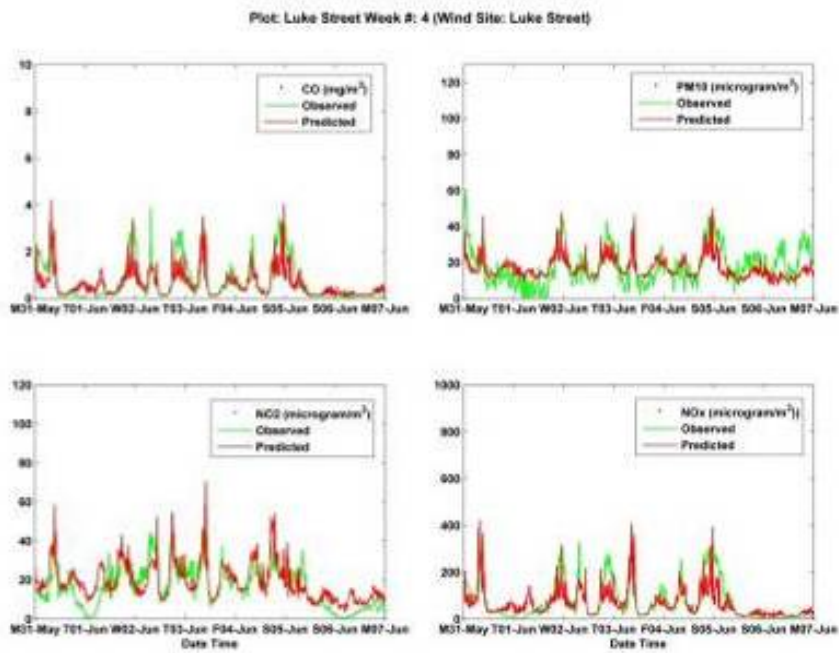


Figure A9.6 One week of observed versus SOSE-predicted concentrations for the Luke Street site: a) CO, b) PM₁₀, c) NO₂, d) NO_x

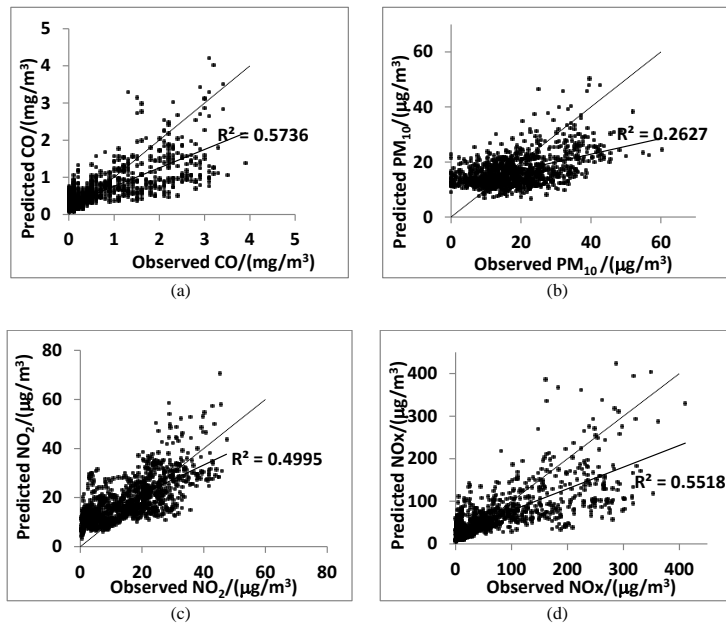


Figure A9.7 Time series of one week of observed and SOSE-predicted concentrations for the 25 Deas Place site: a) CO, b) PM₁₀, c) NO₂, d) NO_x

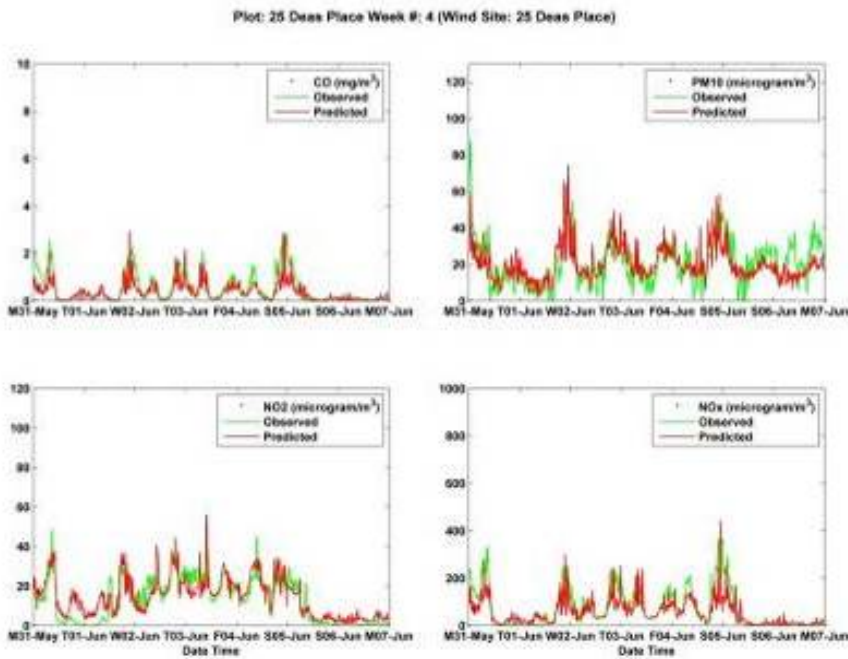


Figure A9.8 One week of observed versus SOSE-predicted concentrations for the 25 Deas Place site: a) CO, b) PM₁₀, c) NO₂, d) NO_x

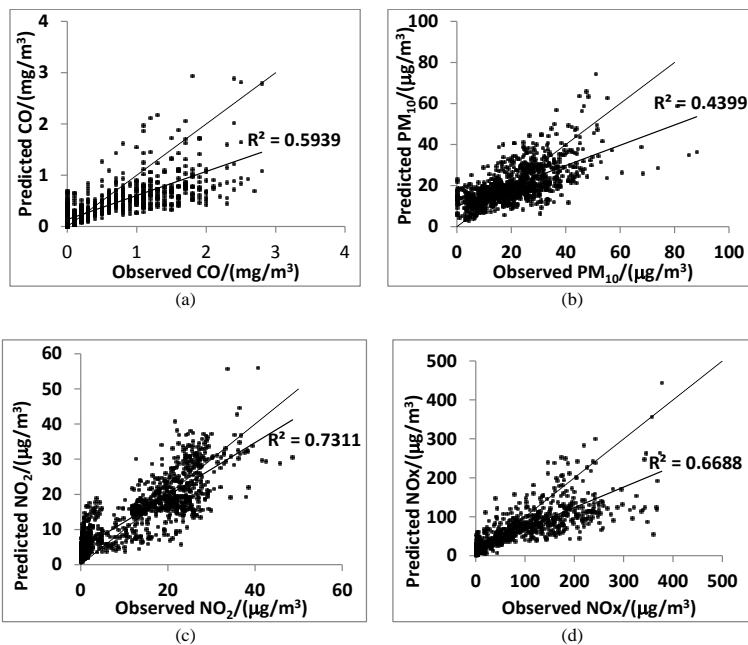


Figure A9.9 Time series of one week of observed and SOSE-predicted concentrations for the Deas Place Reserve site: a) CO, b) PM₁₀, c) NO₂, d) NO_x

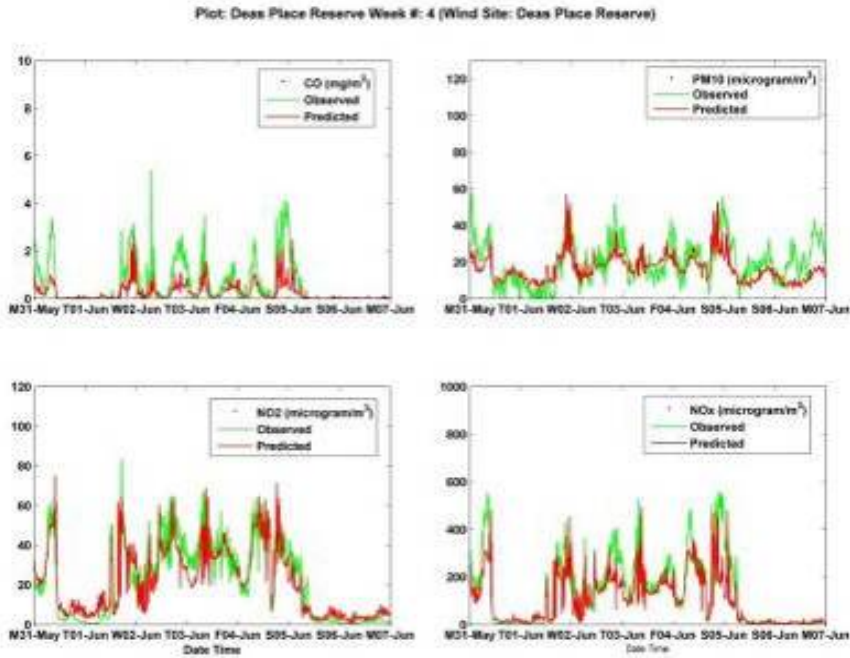


Figure A9.10 One week of observed versus SOSE-predicted concentrations for the Deas Place Reserve site: a) CO, b) PM₁₀, c) NO₂, d) NO_x

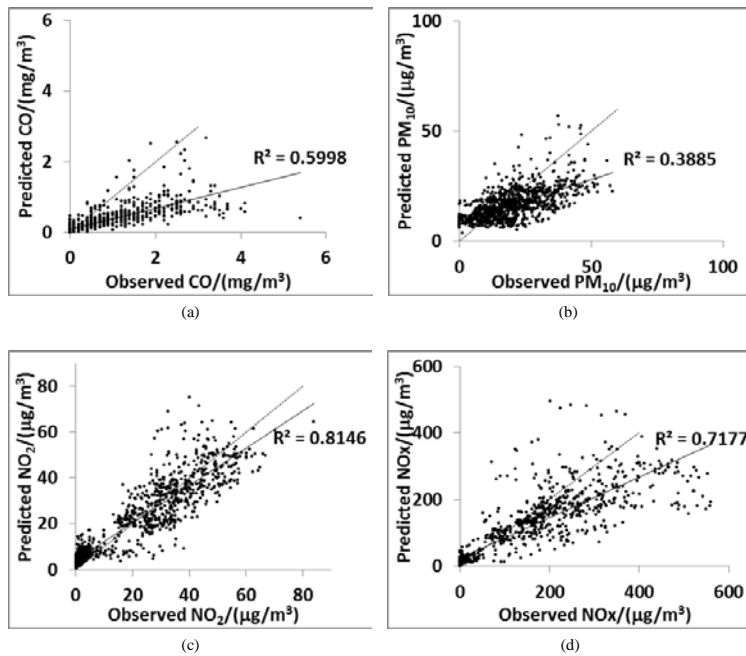


Table A9.2 Model statistics – basic model output for SOSE

Site	Pollutant	Model evaluation statistic			
		r	r ²	MAE ^(a)	RMSE ^(b)
Luke Street	CO (mgm ⁻³)	0.73	0.53	0.33	0.56
	PM ₁₀ (µgm ⁻³)	0.54	0.29	6.76	9.28
	NO ₂ (µgm ⁻³)	0.70	0.49	5.98	7.66
	NO _x (µgm ⁻³)	0.73	0.54	32.02	54.88
25 Deas Place	CO (mgm ⁻³)	0.67	0.45	0.28	0.46
	PM ₁₀ (µgm ⁻³)	0.65	0.42	8.45	11.69
	NO ₂ (µgm ⁻³)	0.80	0.64	4.97	6.58
	NO _x (µgm ⁻³)	0.77	0.59	32.77	52.49
Deas Place Reserve	CO (mgm ⁻³)	0.54	0.30	0.27	0.52
	PM ₁₀ (µgm ⁻³)	0.51	0.26	7.49	10.40
	NO ₂ (µgm ⁻³)	0.84	0.70	6.94	9.17
	NO _x (µgm ⁻³)	0.75	0.56	46.43	78.32

(a) mean absolute error

(b) root-mean-squared error

A9.5.2 Independent data predictions

In the basic model configuration, all of the data was used as training data and the model predictions were based on the same dataset, and not on independent data. The training dataset was used every week except the week for which predictions were made. The model was therefore rerun with each week removed in turn. This was done for all three sites and all four pollutants. Figures A9.11, A9.12 and A9.13 are time series of one week of observed and predicted concentrations for Luke Street, 25 Deas Place and Deas Place Reserve, respectively, using independent data.

Based on these results and similar graphs for the other weeks of observation, the model predictions were found to be very similar to the results based on all of the data (not independent data). Ideally, the model would have been trained based one on season of data and tested on the same season the following year (and vice-versa) for year-to-year consistency of the model parameters, as was carried out for the Aosta Valley dataset as mentioned previously (Dirks et al 2005). This was not possible with the current dataset. However, one can expect the model parameters to be consistent from one year to the next, providing there are no significant changes in the vehicle fleet nor significant changes in the built environment (building structures, vegetation, topography) in the immediate vicinity of the site.

Figure A9.11 Time series of one week of observed and SOSE-predicted concentrations for the Luke Street site: a) CO, b) PM₁₀, c) NO₂, d) NO_x. Results are based on ‘independent’ data (all data except the week used for training)

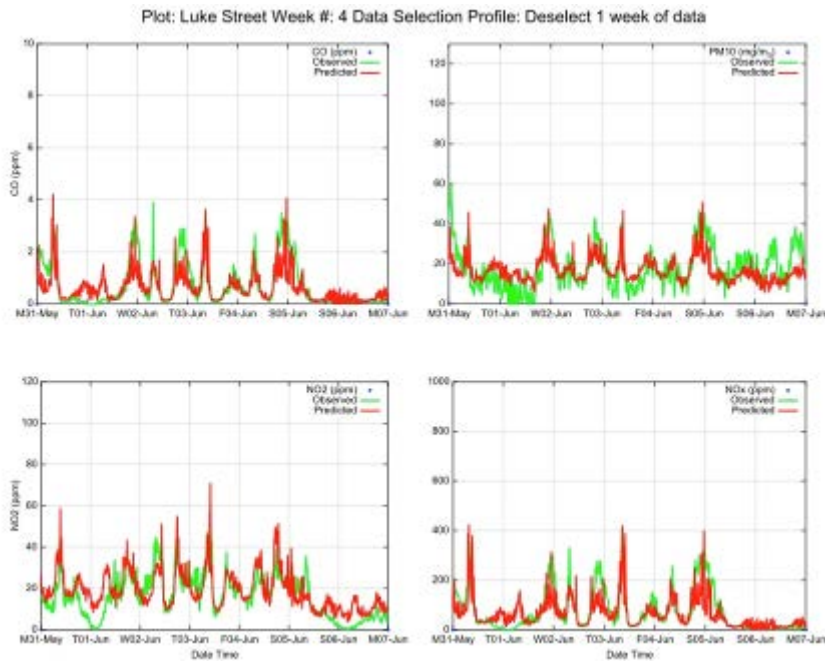


Figure A9.12 One week of observed versus SOSE-predicted concentrations for the 25 Deas Place site: a) CO, b) PM₁₀, c) NO₂, d) NO_x. Results are based on ‘independent’ data (all data except the week used for training)

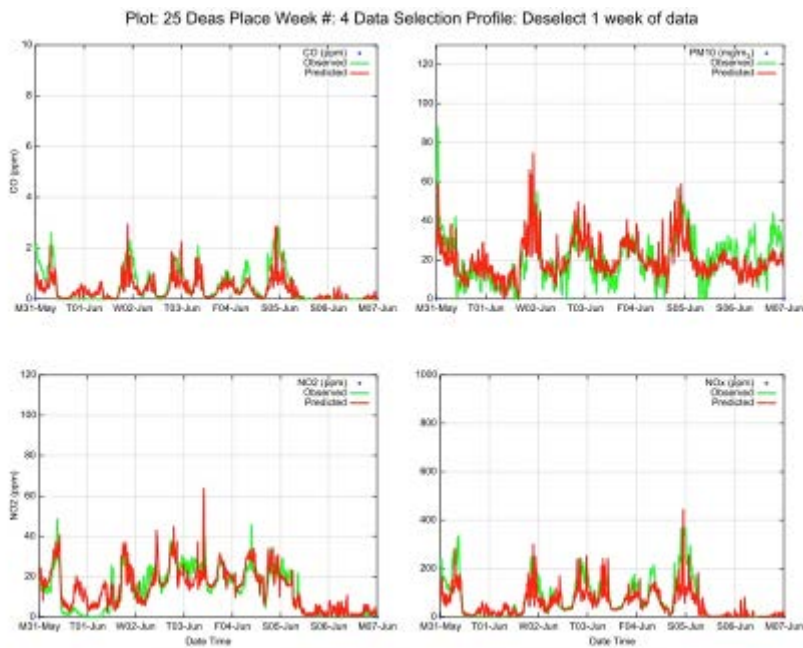


Figure A9.13 One week of observed versus SOSE-predicted concentrations for the Deas Place Reserve site: a) CO, b) PM₁₀, c) NO₂, d) NO_x. Results are based on 'independent' data (all data except the week used for training)

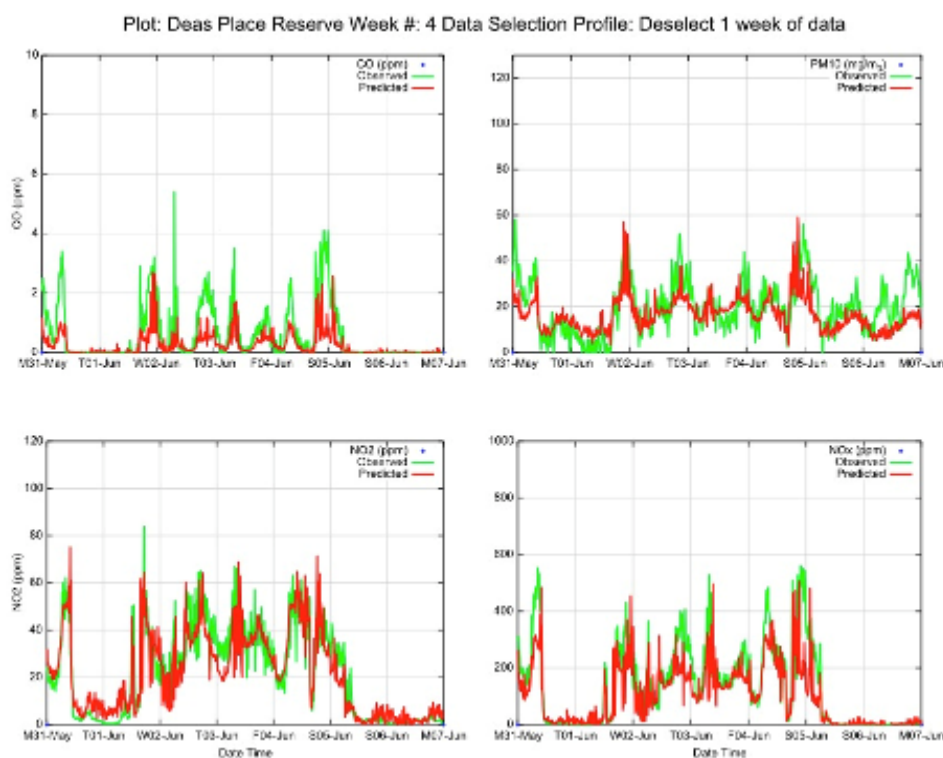


Table A9.3 Model statistics – for SOSE based on independent data

Site	Pollutant	Model evaluation statistic			
		r	r ²	MAE	RMSE
Luke Street	CO (mgm ⁻³)	0.75	0.55	0.36	0.58
	PM ₁₀ (µgm ⁻³)	0.56	0.31	6.82	8.74
	NO ₂ (µgm ⁻³)	0.74	0.55	5.91	7.52
	NO _x (µgm ⁻³)	0.74	0.54	36.34	57.80
25 Deas Place	CO (mgm ⁻³)	0.77	0.60	0.26	0.43
	PM ₁₀ (µgm ⁻³)	0.67	0.45	7.04	9.31
	NO ₂ (µgm ⁻³)	0.85	0.73	4.38	5.82
	NO _x (µgm ⁻³)	0.82	0.67	30.65	51.51
Deas Place Reserve	CO (mgm ⁻³)	0.77	0.59	0.4	0.75
	PM ₁₀ (µgm ⁻³)	0.64	0.41	7.39	9.51
	NO ₂ (µgm ⁻³)	0.9	0.82	5.89	7.96
	NO _x (µgm ⁻³)	0.85	0.72	47.62	81.08

A9.5.3 Choice of meteorological data

Prior to this study, the SOSE model had only ever been tested using meteorological data collected from a meteorological station co-located with the air quality monitoring station, at heights ranging from

approximately 4m to 10m and in a wide range of levels of obstruction, such as within street canyons. This was done on the assumption that only co-located meteorological stations would be able to adequately capture the detailed wind flows within the complex surface layer in such a way that they were sufficiently 'representative' for reliable model predictions to be made. However, often other longer-term meteorological data is available, such as data from airport sites, which may be able to be used for modelling. These sites are generally chosen to be 'representative' of the local area. As such, the World Meteorological Organisation (WMO) requires that such measurements be made at a height of 10m and in an area unobstructed by buildings or natural features (Wieringa 1996).

Little is known about the relative merits of co-located meteorological data compared with meteorological data that is compatible with WMO siting requirements and may be located some distance from the air pollution monitoring station. In our study, meteorological data was available from all three sites, each representing distinct surface environments: Luke Street was the most compatible with the WMO requirements, Deas Place Reserve was within the region directly affected by vehicle-generated turbulence, while 25 Deas Place was located with a suburban environment, immediately surrounded by relatively dense single-storey housing and small trees. In this section, we compare SOSE model performance for all three sites and all pollutants for all combinations of meteorological monitoring stations. The purpose is to determine the relative merits of the choice of meteorological monitoring site in SOSE model performance.

Figure A9.14 compares one week of observed and predicted concentrations for NO_x and CO for Luke Street using meteorological data from each of the three sites. Figures A9.15 and A9.16 are the equivalent graphs for 25 Deas Place and Deas Place Reserve. The model performance statistics are presented in table A9.4.

Note that for all three sites, the best model performance is generally achieved using meteorological data from the Luke Street site, the site most compatible with WMO requirements, irrespective of co-location. This suggests that, providing the distance between the air quality monitor and meteorological tower is less than about 1km, capturing the regional flow (as would be expected from a WMO-compatible site) is more important than co-location. Further analysis based on an additional nine meteorological stations around Auckland suggests that in the range of 1km to 10km, model performance drops off with increasing separation. Beyond this, model performance is steady out to separations of about 30km (analysis not included).

While the meteorological tower located at Deas Place Reserve is affected directly by vehicle-generated turbulence (particularly that from large trucks travelling at high speed along the motorway, and on the southbound off-ramp which is a mere 4m from the measurement site) the meteorological data was nonetheless sufficiently robust to allow for good model predictions for all three sites.

Figure A9.14 Time series of one week of observed and SOSE-predicted concentrations for the Luke Street site for CO and NO_x based on wind speed data from each of the three sites: a) CO and b) NO_x with Luke Street meteorological data, c) CO and d) NO_x with 25 Deas Place meteorological data and e) CO and f) NO_x with Deas Place Reserve meteorological data

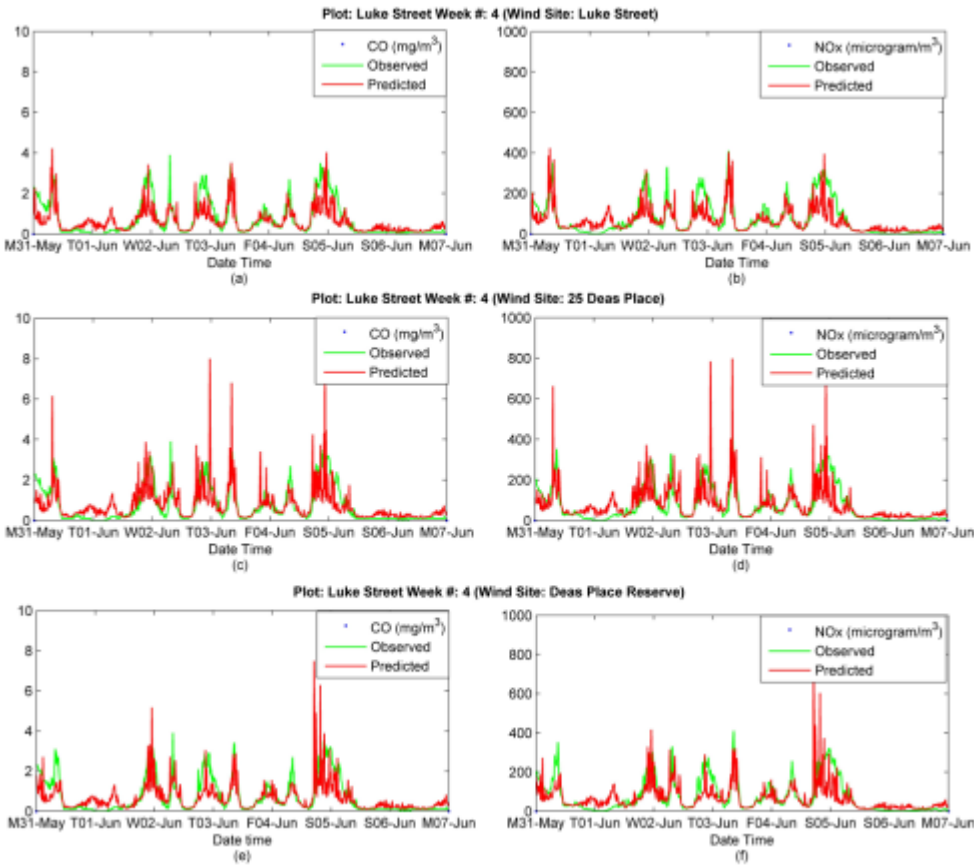


Figure A9.15 Time series of one week of observed and SOSE-predicted concentrations for the 25 Deas Place site for CO and NO_x based on wind speed data from each of the three sites: a) CO and b) NO_x with Luke Street meteorological data, c) CO and d) NO_x with 25 Deas Place meteorological data and e) CO and f) NO_x with Deas Place Reserve meteorological data

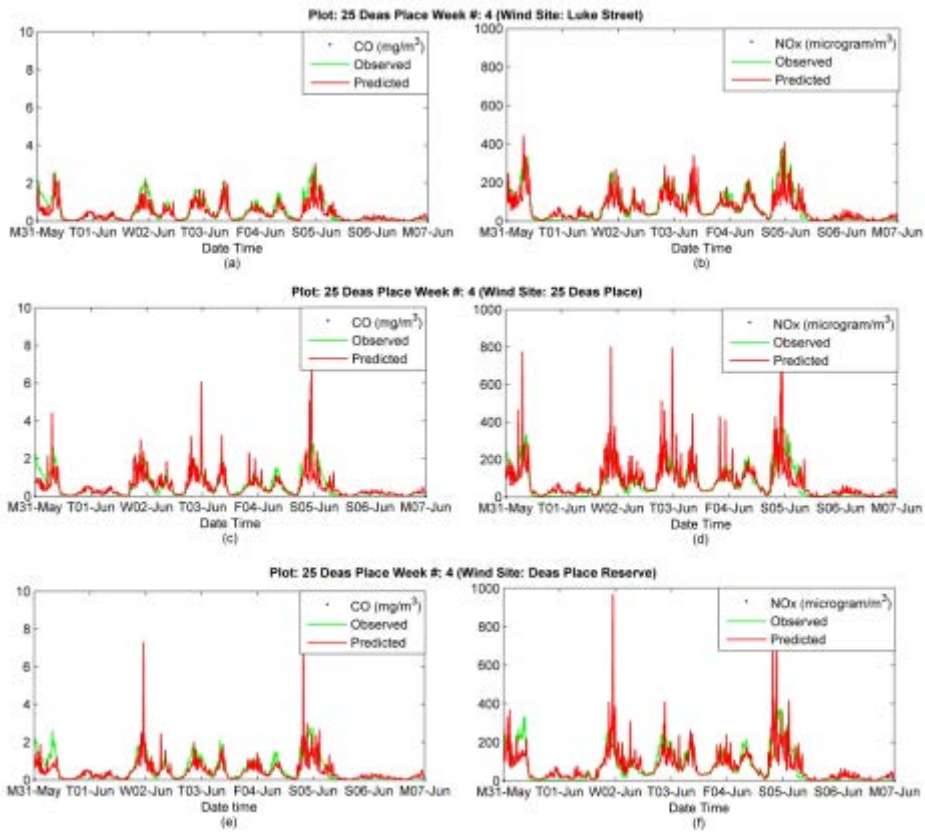


Figure A9.16 Time series of one week of observed and SOSE-predicted concentrations for the Deas Place Reserve site for CO and NO_x based on wind speed data from each of the three sites a) CO and b) NO_x with Luke Street meteorological data, c) CO and d) NO_x with 25 Deas Place meteorological data and e) CO and f) NO_x with Deas Place Reserve meteorological data

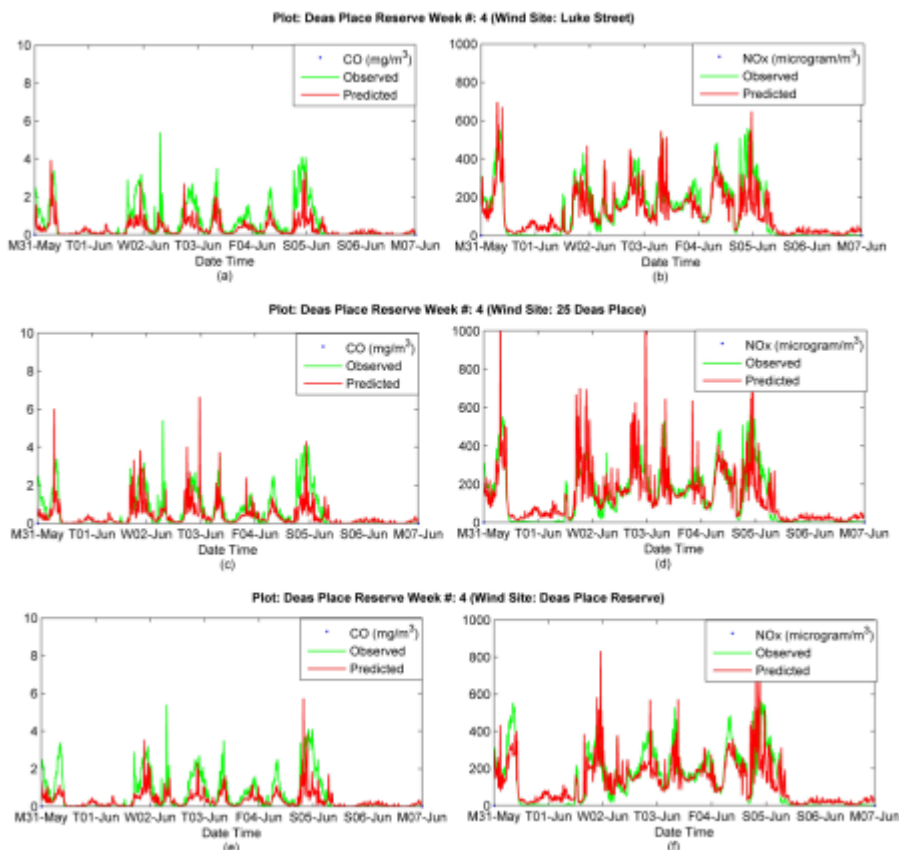


Table A9.4 Model statistics –for SOSE for all combinations of meteorological and air quality monitoring sites

Air quality site	Met site	Pollutant	Model evaluation statistic			
			R	r ²	MAE	RMSE
Luke Street	25 Deas Place	CO (mgm ⁻³)	0.69	0.48	0.38	0.61
		PM ₁₀ (µgm ⁻³)	0.57	0.33	7.01	9.56
		NO ₂ (µgm ⁻³)	0.71	0.50	6.13	7.87
		NO _x (µgm ⁻³)	0.71	0.50	35.69	59.34
	Deas Place Reserve	CO (mgm ⁻³)	0.71	0.51	0.34	0.57
		PM ₁₀ (µgm ⁻³)	0.50	0.25	6.74	9.34
		NO ₂ (µgm ⁻³)	0.83	0.70	7.03	9.21
		NO _x (µgm ⁻³)	0.72	0.51	32.59	56.38
25 Deas Place	Luke Street	CO (mgm ⁻³)	0.74	0.54	0.27	0.42
		PM ₁₀ (µgm ⁻³)	0.66	0.44	8.41	11.50
		NO ₂ (µgm ⁻³)	0.80	0.65	5.03	6.52
		NO _x (µgm ⁻³)	0.78	0.61	32.49	51.46

Air quality site	Met site	Pollutant	Model evaluation statistic			
			R	r ²	MAE	RMSE
	Deas Place Reserve	CO (mgm ⁻³)	0.70	0.48	0.27	0.44
		PM ₁₀ (µgm ⁻³)	0.65	0.42	8.35	11.73
		NO ₂ (µgm ⁻³)	0.82	0.68	4.70	6.24
		NO _x (µgm ⁻³)	0.75	0.56	32.98	54.34
Deas Place Reserve	25 Deas Place	CO (mgm ⁻³)	0.62	0.39	0.31	0.56
		PM ₁₀ (µgm ⁻³)	0.60	0.36	7.71	10.61
		NO ₂ (µgm ⁻³)	0.84	0.71	7.42	9.58
		NO _x (µgm ⁻³)	0.78	0.62	53.53	81.14
	Luke Street	CO (mgm ⁻³)	0.62	0.38	0.30	0.53
		PM ₁₀ (µgm ⁻³)	0.57	0.33	7.61	10.49
		NO ₂ (µgm ⁻³)	0.83	0.69	7.42	9.67
		NO _x (µgm ⁻³)	0.78	0.60	51.05	79.59

A9.5.4 Background concentration

One of the challenges in many air quality modelling studies is the lack of information on the background concentrations. This information is necessary in order to quantify the contribution of a road source to the overall air pollution of a region for a road that is already in operation. One option is to monitor pollutants on both sides of a road and compare the concentrations between the two sites under various different wind conditions. The limitation is that, in many situations, due to budget constraints, only data from one site (and therefore one side of the road) is available.

One of the potentially useful applications of the SOSE model is that it can provide such estimates based on data from one site alone. The SOSE model is a regression-based model for which the intercept of the regression is assumed to be the background concentration when the winds are such that the pollution monitoring site is downwind of the road. The set of parameters produced as an output of the model include the (SOSE-predicted) average daily profile of the background concentration.

While this daily profile of the background concentration has been predicted in other studies, including that of the Aosta Valley (Dirks et al 2005), its physical basis could not be verified due to a lack of a dataset consisting of air quality monitoring carried out simultaneously on both sides of the road. The present dataset allows for such a comparison.

Figures A9.17 and A9.18 compare the SOSE-predicted and observed background concentrations for westerly and easterly winds, respectively. For westerly winds, the observed background concentrations are from Luke Street for westerly winds while the SOSE-predicted concentrations are for 25 Deas Place for westerly winds only. For easterly winds, the observed background concentrations are for 25 Deas Place for easterly winds and the SOSE-predicted concentrations are for Luke Street for easterly winds.

In general there is a good agreement between the observed and SOSE-estimated average daily profile of the background concentration for all of the pollutants. The best predictions are for NO₂ for both wind sectors (westerly and easterly). Also, it is worth noting that CO and NO_x show very similar daily profiles quite distinct from those of PM₁₀ which are relatively flat throughout the day.

Figures A9.19 and A9.20 show the contribution of the motorway and the background to the total concentration for the westerly and easterly wind sectors, respectively. The results suggest that the motorway contributes a large fraction of the CO and NO_x at night. The NO₂ is mainly not from the motorway, except during the middle of the day. Approximately 20% of the PM₁₀ originates from the motorway.

Note that most of the NO₂ and NO_x originates from the motorway whereas it only contributes a small fraction of the PM₁₀. The background dominates during the evening hours for PM₁₀ whereas for NO_x, the motorway dominates consistently throughout the day.

Figure A9.17 Comparison between estimates of the average daily profile of the background concentration predicted by SOSE for westerly winds at 25 Deas Place and the observed concentrations from Luke Street for westerly winds: a) CO, b) PM₁₀, c) NO₂, d) NO_x

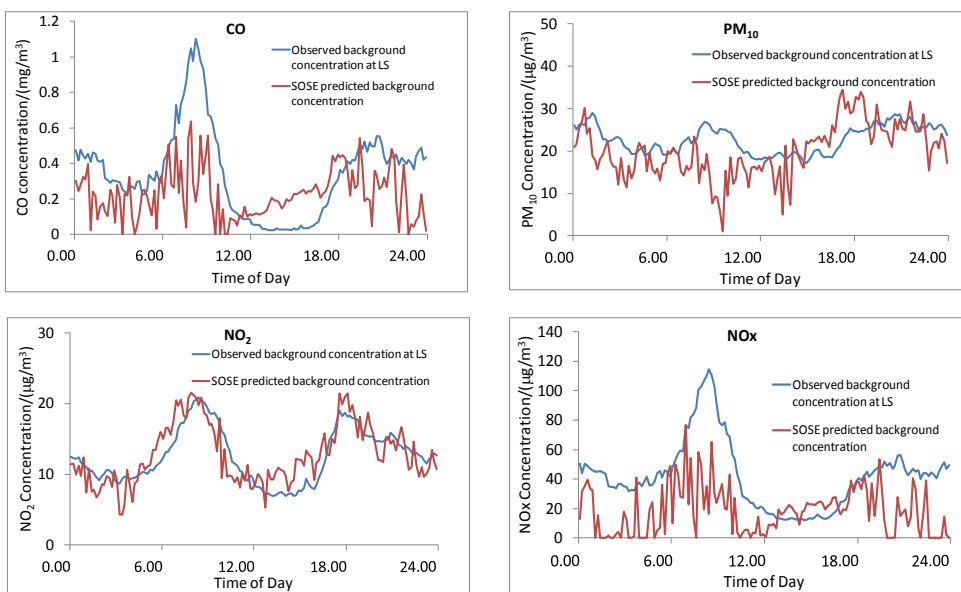


Figure A9.18 Comparison between estimates of the average daily profile of the background concentration predicted by SOSE for easterly winds at Luke Street and the observed concentrations from 25 Deas Place Luke Street for easterly winds: a) CO, b) PM₁₀, c) NO₂, d) NO_x

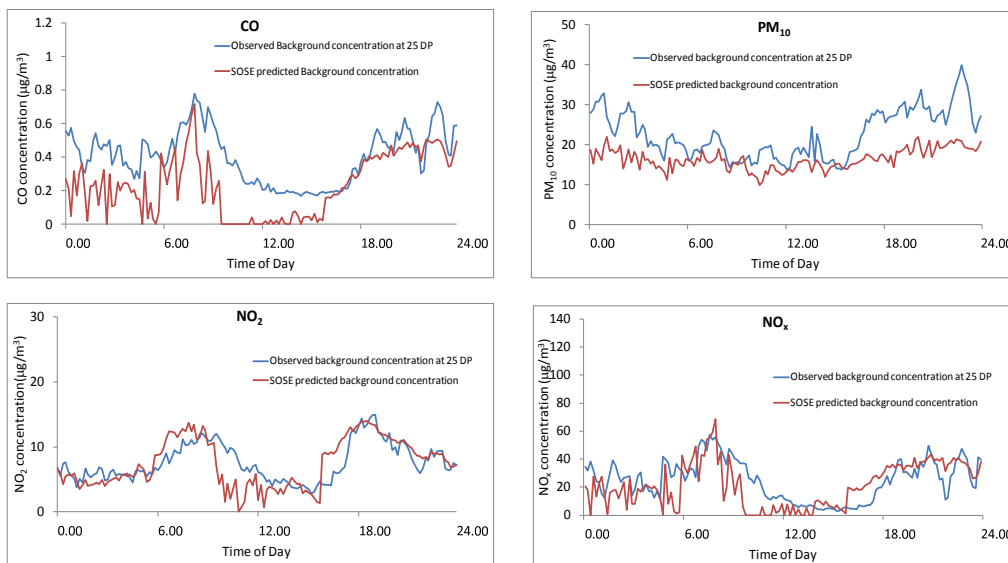


Figure A9.19 Daily profile of the contribution of the motorway and the background to the total daily concentration profile for each pollutant for westerly winds: a) CO, b) PM₁₀, c) NO₂, d) NO_x

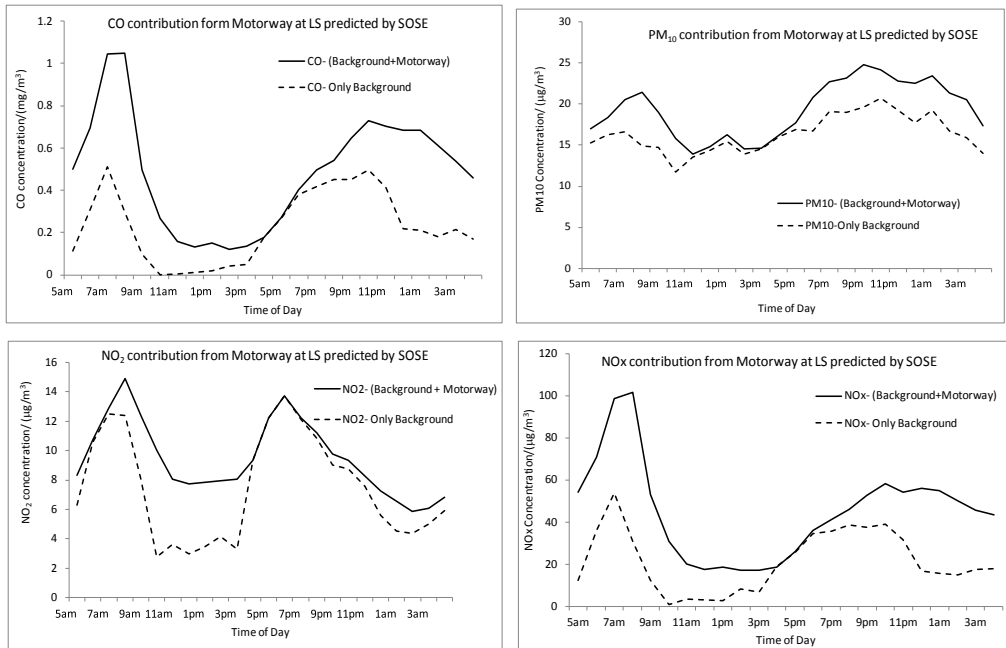
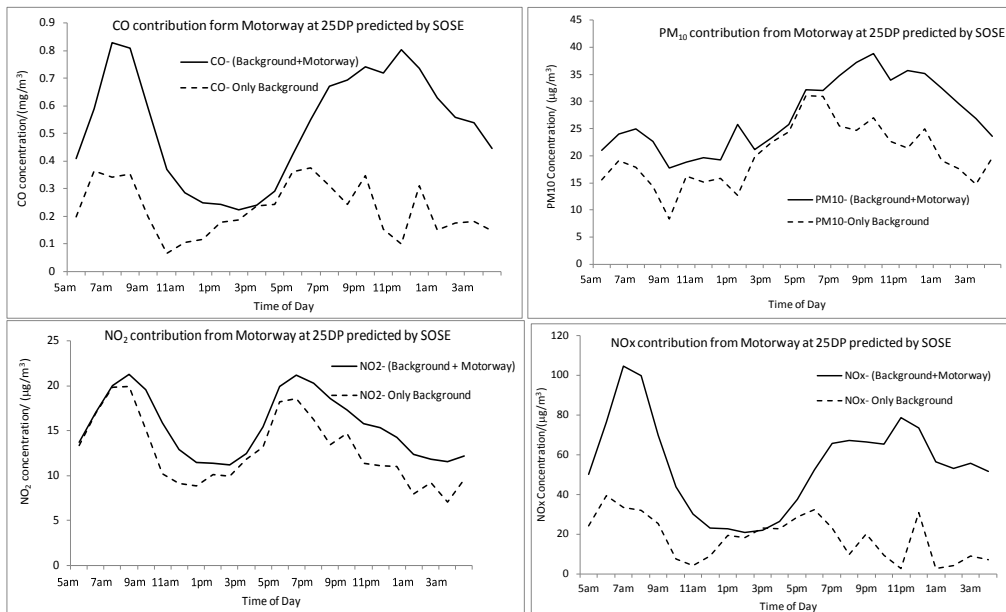


Figure A9.20 Daily profile of the contribution of the motorway and the background to the total daily concentration profile for each pollutant for easterly winds: a) CO, b) PM₁₀, c) NO₂, d) NO_x



A9.5.5 Box height estimates

Previously no emissions estimates have been used. Model predictions have been made solely based on meteorology as input and the model trained on air pollution observations. The slope parameter derived from the regression is given by the product of the emissions rate and the inverse of the box height ($Q\Delta z^{-1}$). The addition of Q (output from VEPM) into SOSE allows the box height to be isolated as the slope parameter

becomes Δz^1 . This means that it is possible to derive a daily profile of the box height for each pollutant at each site. This is of interest from the point of view of its potential contribution to developing an understanding of the surface meteorology in urban and suburban environments. However, it can also be used to verify VEPM estimates. Given that the box height can be expected to be the same regardless of the pollutant (in this sense, the meteorology is isolated), one can expect the box heights to be the same for each of the pollutants. Any systematic difference in the box height estimates would indicate that the proportions of the pollutants are not correct (note: it is not possible to say anything about their absolute values).

Figures A9.21 and A9.22 are the box height estimates for CO, PM₁₀ and NO_x for Luke Street and 25 Deas Place, respectively. The following observations can be made:

- The model-predicted box heights based on CO and NO_x are physically plausible, varying from less than about 10m at night to greater than 100m during the day, often becoming very high and unstable during the late afternoon. While it is possible that this is a mathematical artefact, it is consistent with instabilities (and very deep mixing layers) that can occur at this time of the day.
- The box heights for 25 Deas Place are consistently higher during the day than for Luke Street. This can be expected due to the differences in land use between the two sites. The Luke Street site is much more open with a shallower surface layer than for 25 Deas Place which consists of densely packed single-storey housing.
- The box height estimates for CO and NO_x are similar and in proportion to each other. This suggests that the same can be said for the VEPM emissions estimates for CO and NO_x.
- While the box height estimates for CO and NO_x are comparable, they are very different from those of PM₁₀. It is reasonable to conclude that the model predictions for PM₁₀ are too poor to allow a comparison to be made. This may simply be due to the fact that the motorway contributes little to the total pollution compared with CO and NO_x. It is therefore not possible to say whether the VEPM estimates of PM₁₀ are in proportion to those of NO_x and CO.

Figure A9.21 Box height estimates for the Luke Street site for weekdays: a) CO, b) PM₁₀, c) NO_x

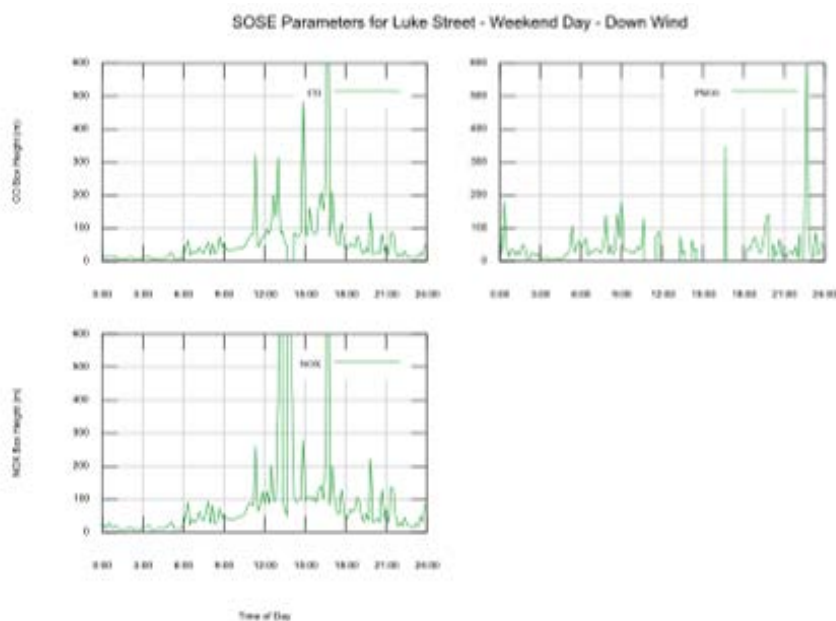
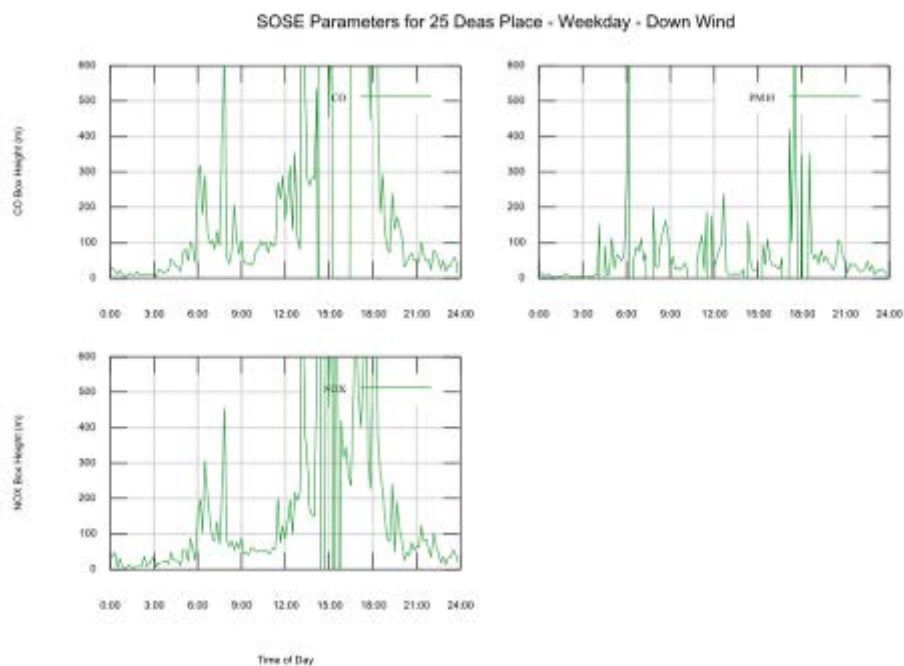


Figure A9.22 Box height estimates for the 25 Deas Place site for weekdays: a) CO, b) PM₁₀, c) NO_x

A9.6 Key findings

- The SOSE model worked well for motorways emissions. This was not unexpected.
- The model captured most of the diurnal variability in concentrations of the pollutants. This was more so for NO_x for which the model performed best, and was worst for PM₁₀.
- The PM₁₀ predictions were somewhat smoother than the observed concentrations. This could be partly explained by the monitoring method used to measure the PM₁₀ concentrations.
- Though not entirely apparent from the week selected as an example, the model tended to miss peaks in concentration. As such, the model, as currently formulated, is not suitable for predicting exceedences and other extreme events.
- While not shown directly here, the model parameters from one year could be used to predict concentrations for the same site for other years providing meteorological monitoring is ongoing.
- The results of this study suggested it was not necessary to have co-located meteorological data to model reliably using SOSE. In fact, model results suggested that better model performance might be achieved by using meteorological data from a site more compliant with WMO specifications (10m measurements and a site not obstructed by topography) than by using data from a co-located meteorological tower. This has important implications in the context of the 'representativeness' of meteorological monitoring for the mesoscale modelling of surface wind flows needed for air pollution forecasting. Despite the Deas Place Reserve site being sufficiently close to the motorway to be affected directly by the vehicle-generated turbulence, the model generally performed well using either co-located meteorological data or data from either of the other two sites.

A9.7 Implications for the use of SOSE

This research, in conjunction with previous research, implies that SOSE has a number of strengths and weaknesses relative to other approaches, as summarised below.

A9.7.1 Strengths of SOSE

- **Estimating the daily profile of the background concentration based on one air pollution monitoring station**

One of the most useful features of the model appears to be its ability to estimate the daily profile of the background concentrations based on measurements from one site. This is particularly the case for NO_x for which the motorway is a strong source and is not affected by contributions from home heating. The model is able to reliably predict the mean daily distribution of PM₁₀. However, a significant fraction of the PM₁₀ is background so one would expect a regression model to be able to estimate this reasonably well.

- **Estimating the contribution of a road to the air quality of an area, over and above the background concentration**

If the average daily distribution of the background concentration is known, then the average daily distribution of the road contribution can be isolated. This can be done based on measurements from one site and also after a road has already been in operation. In this way, the model may be used retrospectively.

- **Estimating the average daily distribution of the box height**

If the source strength is known, then the box height can be isolated (the regression coefficient is the inverse of the box height). This allows a comparison of box height predictions between pollutants for a given site. This has important implications for understanding dispersion mechanisms in regions of complex topography such as urban environments.

- **Identifying inconsistency in emission estimates between pollutants**

Given that all pollutants are subject to the same meteorology, if the emission estimates are in the correct proportion, one can expect that the box heights predicated by SOSE will be comparable. If not, then they suggest there is a mismatch in the proportion of the emission rates. The results of the present study suggested that the output from VEPM for NO_x and CO was consistent (the box height predictions were similar). Unfortunately, the model performance for PM₁₀ was sufficiently poor (and box height predications unstable) and it was not possible to comment reliably on the emission estimates for PM₁₀ relative to those of CO and NO_x.

A9.7.2 Limitations of SOSE

- **Predicting peak concentrations reliably**

The model is generally poor at predicting the occurrence of short-term peak concentrations. The regression parameters are heavily weighted towards the lower concentrations because there are significantly more of them (concentration distributions are not normally distributed). For this reason, the model is not suitable for use in investigating the likelihood of an exceedence or other extremes in concentration.

A10 Land-use regression modelling

A10.1 Introduction to land-use regression modelling

Models utilise a series of variables to estimate air pollution exposure at a local or national spatial scale. There are a number of types of models: dispersion models, receptor models, stochastic models, compartment or box models (Colls 2002) along with regression-based models, most commonly land-use regression (LUR) models. Most models need some understanding of meteorology coupled with some measure of emissions. Modelled data is usually then compared with monitored data to validate the model. Dispersion models mathematically simulate how air pollution disperses in the ambient environment by taking into account local wind patterns and air pollutants emitted from sources. The quality of the output of such models often rests on the quality and availability of the input data, which can often be lacking or of poor quality. In addition, mathematical dispersion models are also severely limited by our understanding of wind flows in complex environments, even with very sophisticated mathematics. As a result even if the data was extremely good, the models still might not work very well. A more recent tool in exposure assessment has been LUR modelling. First used in the late 1990s (Briggs et al 1997), LUR models use monitored levels of pollutants of interest as the dependent variable and variables, such as traffic, topography and other geographic variables as the independent variables in a multivariate regression model (Hoek et al 2008; Ryan and LeMasters 2007). The advantage of LUR models is that they use data that can be more readily collected or collated, often using GIS. They are now widely used and have been shown in urban areas to perform 'typically better or equivalent to geo-statistical methods such as kriging⁷ and conventional dispersion models' (Hoek et al 2008). LUR models use variables far wider than just those for land use but the term land-use regression is now widely used and understood (a point emphasised by Hoek et al 2008). Variables used include altitude, meteorology and various indicators of traffic emissions such as distance to road, and traffic volume.

Hoek's 2008 review identified 25 papers that had used LUR; this number will have increased since their review and LUR is now an established approach. Despite this, LUR-based approaches have rarely been used in New Zealand. The only published paper is the work of Kingham et al (2008a). Data sources used in the final model included census data on domestic heating, industrial emissions estimates, vehicle kilometres travelled and meteorological measurements. Results showed a good association between the model estimates and monitored data at locations where it had been used. Regression-based approaches often perform well compared with other modelling approaches by having input data available at a very fine spatial scale, and not having to rely on direct emissions data, which is often the Achilles heel of dispersion models. The results of this model have been subsequently used in a number of studies (Fisher et al 2007; Kingham et al 2008b; Richardson et al 2010; Kingham et al 2010; Hales et al 2010).

A10.2 Approach to modelling

There are multiple ways to calculate the best model, and no definitive right or wrong way. In our study, the modelling was carried out using R statistical software. The criteria for when to include or not include independent variables was based on the protocols developed as part of the ESCAPE project (ESCAPE 2010). We adopted the criteria that predictor variables would be included in the model if the increase in adjusted R² was greater than 1%, the coefficient conformed to the pre-specified direction, and the direction of effect for

⁷ Kriging is a geostatistical technique that estimates values between sampling sites by means of spatial interpolation

predictors already included in the model did not change. There was some debate about whether individual predictor variables should themselves be statistically significant. The ESCAPE project recommends variables remain only if $p < 0.10$. An alternative approach is to include variables if they improve the model irrespective of their own statistical significance. One approach to this is to use the Akaike information criterion (AIC) which is a measure of the relative goodness of fit of a statistical model. With this, the preferred model is the one with the minimum AIC value, and this was the approach we adopted.

A10.3 Dependent variable: NO₂ data

A full year of data (13 periods of 4 weeks) was used for the Otahuhu study area. We also used data from the Mangere study area for the same period (see chapter A4). Mean values for each site were used as the dependent variable. As noted in chapter A4 there are a number of data points missing. Taking the mean of the remaining data could result in an artificially high or low value as the missing data could have been during high or low pollution periods. Consequently, missing data was imputed, to provide a more meaningful mean value. This was done using the SAS statistical packages' multiple imputation procedure (SAS nd). This procedure assumes the data is from a continuous multivariate distribution and contains missing values on any of the variables. It replaces each missing value with a set of plausible values that represent the uncertainty about the correct value to impute. These plausible values are estimated based on other values in the rows (in this example these are the sampling sites) and the columns (in this example these are the sampling periods). An average is then taken of the estimated plausible values and this becomes the imputed value.

A10.4 Independent variables: land-use data

The following variables were calculated in a GIS and used as the independent variables in the analysis:

- 1 Distance to coast (metres)
- 2 Distance to nearest motorway (metres)
- 3 Distance to nearest road (metres)
- 4 Distance to nearest road for which we have traffic data (metres)
- 5 Altitude of monitor (metres above sea level)
- 6 Sum of road lengths within a 50m buffer
- 7 Sum of road lengths within a 5m to 100m ring⁸
- 8 Sum of road lengths within a 100m to 500m ring
- 9 Sum of road lengths within a 500m to 1000m ring
- 10 Sum of road lengths within a 1000m to 2500m ring
- 11 Sum of road lengths within a 2500 to 5000m ring
- 12 Sum of road lengths for which we have traffic data within a 50m buffer
- 13 Sum of road lengths for which we have traffic data within a 50m to 100m ring

⁸ Rings rather than buffers were used to enable multiple distances from the each sampling point, while avoiding issues of co-linearity. This approach is supported by the ESCAPE study.

- 14 Sum of road lengths for which we have traffic data within a 100m to 500m ring
- 15 Sum of road lengths for which we have traffic data within a 500m to 1000m ring
- 16 Sum of road lengths for which we have traffic data within a 1000m to 2500m ring
- 17 Sum of road lengths for which we have traffic data within a 2500m to 5000m ring
- 18 Total average daily traffic within a 50m buffer
- 19 Total average daily traffic within a 50m to 100m ring
- 20 Total average daily traffic within a 100m to 500m ring
- 21 Total average daily traffic within a 500m to 1000m ring
- 22 Total average daily traffic within a 1000m to 2500m ring
- 23 Total average daily traffic within a 2500m to 5000m ring

No census-based (eg population density) or land cover variables were used as the small spatial resolution of the two study areas meant there was not enough geographical variation in these types of variables to warrant inclusion (most studies using LUR are for much larger spatial areas).

A10.5 Results

The regression analysis was carried out in R and produced the following statistical results.

Otahuhu

<i>Coefficients</i>	<i>Estimate</i>	<i>Std. error</i>	<i>t value</i>	<i>Pr(> t)</i>
(Intercept)	1.090e+01	1.344e+00	8.110	7.89e-09***
Dist_Coast	1.318e-02	3.075e-03	4.287	0.000194***
TotAdt_b50	3.268e-05	1.256e-05	2.602	0.014662*
TotAdt_b100to500	2.705e-06	1.863e-06	1.452	0.157513

Residual standard error: 2.962 on 28 degrees of freedom.

Multiple R-squared: 0.4941, adjusted R-squared: 0.4399.

F-statistic: 9.117 on 3 and 28 DF, p-value: 0.0002262.

Thus producing the following regression equation:

$$NO_2 = 10.9 + (0.01318 * \text{distance to coast}) + (0.00003268 * \text{total average daily traffic within a 50m buffer}) + (0.000002705 * \text{total average daily traffic within a 100m to 500m ring})$$

Mangere

<i>Coefficients</i>	<i>Estimate</i>	<i>Std. error</i>	<i>t value</i>	<i>Pr(> t)</i>
(Intercept)	1.139e+01	2.469e+00	4.612	0.000151***
Dist_Motor	-2.441e-03	7.801e-04	-3.130	0.005062**
Dist_Rd	-1.460e-02	8.742e-03	-1.670	0.109776
Elev	-2.953e-01	8.037e-02	-3.675	0.001411**
TotAdt_b50to100	1.173e-05	4.754e-06	2.467	0.022314*
TotAdt_b1000to2500	1.161e-06	3.621e-07	3.206	0.004247**

Residual standard error: 1.751 on 21 degrees of freedom.

Multiple R-squared: 0.7722, adjusted R-squared: 0.718.

F-statistic: 14.24 on 5 and 21 DF, p-value: 3.835e-06.

Thus producing the following equation:

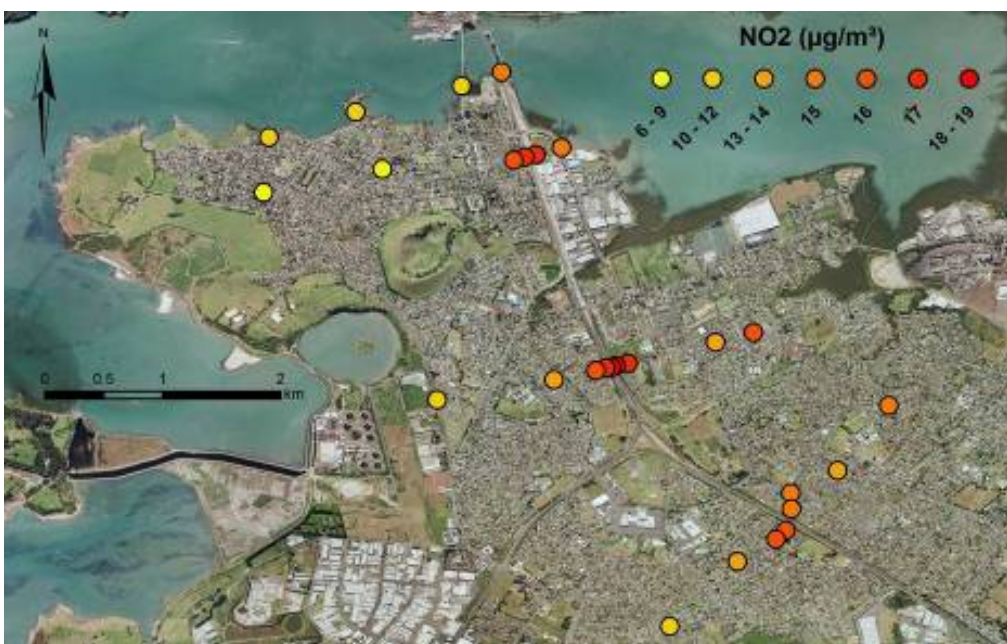
$$\text{NO}_2 = 11.39 - (0.002441 * \text{distance to motorway}) - (0.0146 * \text{distance to road}) - (0.2953 * \text{altitude of monitor}) + (0.00001173 * \text{total average daily traffic within a 50m to 100m ring}) + (0.000001161 * \text{total average daily traffic within a 1000m to 2500m ring})$$

The generated equations were applied to the individual monitoring sites producing modelled values for the two areas. These are mapped in figures A10.1 and A10.2. In both cases there are pollution gradients away from the major roads.

Figure A10.1 Modelled NO₂ values at monitoring sites for Otahuhu



Figure A10.2 Modelled NO₂ values at monitoring sites for Mangere



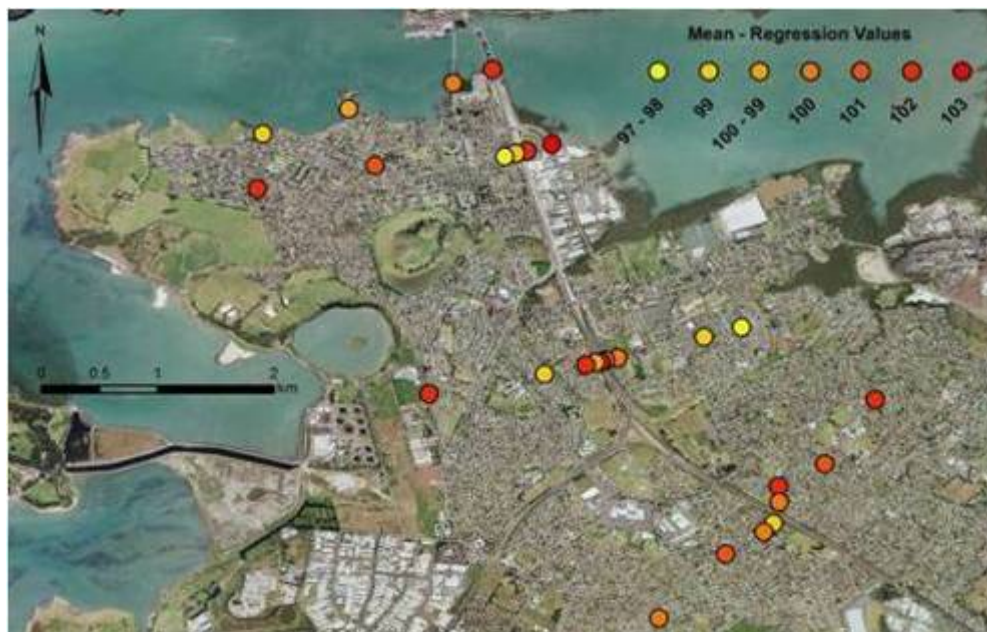
It is interesting how well the regression method presented here predicts the monitored annual concentrations. Figures A10.3 and A10.4 show the difference between observed (monitored) and expected (modelled values) for the two areas. There are no obvious spatial patterns as to where the model overpredicts (values over 100) or underpredicts (values under 100). For example the model does not seem to predict better or worse near or far from roads, the coast etc. This suggests that the difference between observed and expected is random and not associated with particular characteristics of the environment. It does signal that there may be other specific attributes that we should incorporate into our model.

Figure A10.3 Difference between actual mean measured and LUR predicted NO₂ values (Obs-Exp) at monitoring sites for Otahuhu



Note: A value of 100 indicates that monitored (observed) values equal modelled (expected) values. Values >100 indicate that monitored is greater than modelled and values <100 indicate that modelled is greater than monitored.

Figure A10.4 Difference between actual mean measured and LUR predicted NO₂ values (Obs-Exp) at monitoring sites for Mangere



Scatterplots of modelled against monitored data for the two sites are presented in figures A10.5 and A10.6. In both cases it can be seen that the model predicts similarly well at low and high values.

Figure A10.5 Scatterplot of monitored and modelled NO₂ for Otahuhu

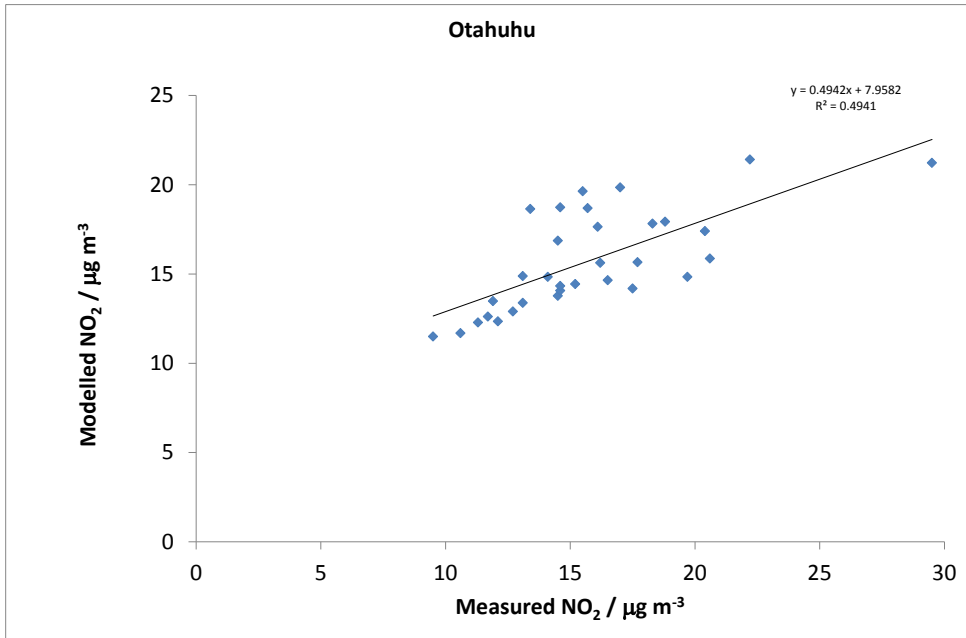
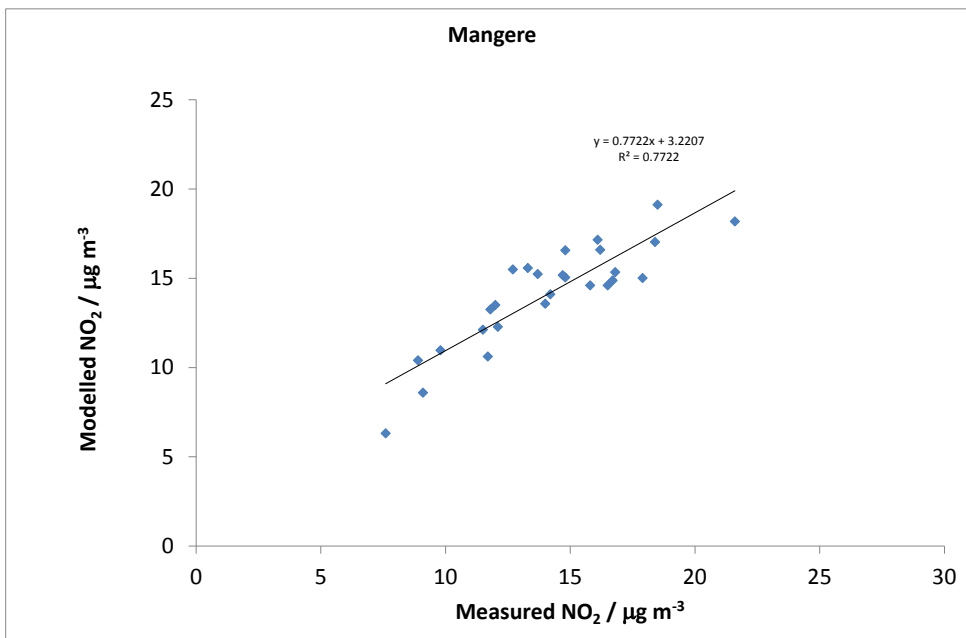


Figure A10.6 Scatterplot of monitored and modelled NO₂ for Mangere



The regression equations were applied to a grid of points every 20m producing pollution maps for the two areas (figures A10.7 and A10.8). In Otahuhu (figure A10.7) the importance of the coast can be seen with levels increasing with distance. This is a function of being included as an independent variable in the final equation. The motorway has a less significant impact. In Mangere (figure A10.8) the nature of the final equation means that proximity to the motorway is highly important.

Figure A10.7 LUR map for Otahuhu

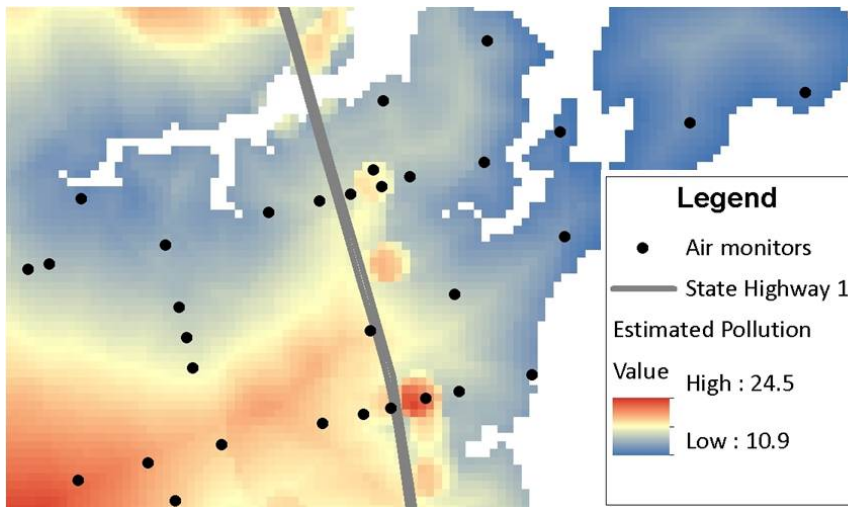
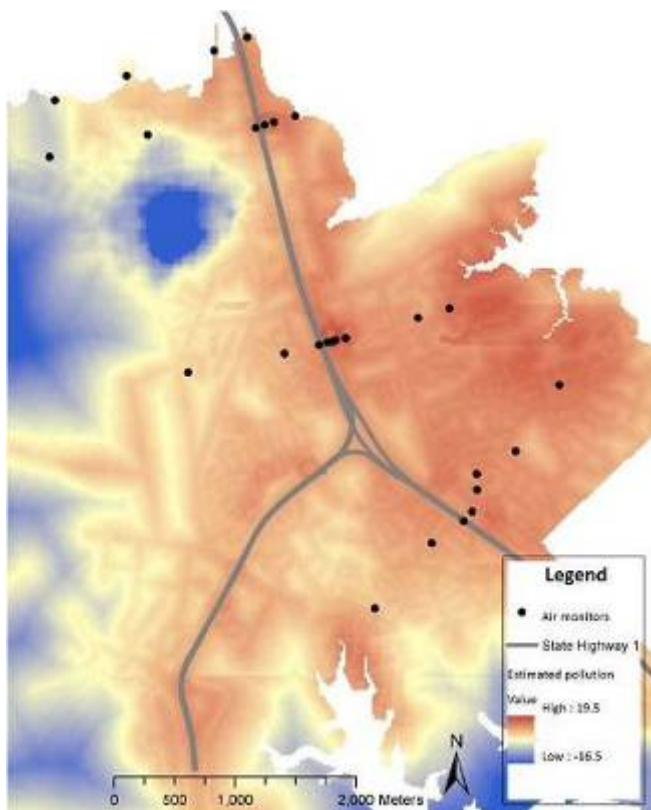


Figure A10.8 LUR map for Mangere



A10.6 Commentary

Model performance appears to be stronger in Mangere than in Otahuhu, as evidenced by the slope of the best fit line between modelled and observed being closer to 1, and a higher R^2 . The R^2 figures for Mangere (R^2 of 0.77) fit well with the R^2 values achieved in other studies. The R^2 figures for Otahuhu (R^2 of 0.49) are a bit lower. It can be seen that the model predicts equally well for high and low pollution areas. While

there are some differences between modelled and monitored values, they do not cluster at either high or low values but are located randomly through the dataset.

Studies of NO₂ in Europe (Briggs et al 1997; Briggs et al 2000) and the USA (Ross et al 2006) are comparable to the Mangere figures, although it should be noted that those studies were for citywide areas whereas this study has estimated values for smaller areas within a city. Using a similar approach, a multi-site study in Europe (Brauer et al 2003; Cyrus et al 2005) produced R² values of between 0.76 and 0.9 for predicting annual fine particle concentrations for their study sites in the Netherlands, Munich and Stockholm; again similar to the Mangere figures and slightly higher than those in Otahuhu. The only published study in New Zealand estimating PM₁₀ levels for the whole country achieved an R² value of 0.86. A review of 24 LUR models by Hoek et al (2008) found R² values for urban NO₂ ranging from 0.51 to 0.90. What can be concluded therefore is that this study produced similar R² values to those of other research.

The models for each area have some similarities, but are quite different. Neither model found that the 'sum of road length' variables were predictive. Both models rely on total average daily traffic data to predict NO₂, but use different distance buffers. Only the Mangere model used an explicit 'distance to road/motorway' variables (distance being implicit in the buffer variables in Otahuhu). Only the Otahuhu model used 'distance to coast', despite the Mangere data including coastal locations. Only the Mangere model used 'altitude', despite the low variability in altitude across the model area.

The relatively poor performance of the model in Otahuhu may be related to the size of the study area. The Otahuhu study area is 1.5km x 2.5km, the Mangere area approximately double that. It is hypothesised that in a smaller area there is not enough variability in the local geography used to predict the monitored values. It may also be the case that the 'distance to coast' variable is introducing some confounding. This variable is being used to explain the underlying west-east gradient so that higher concentrations are seen towards Otahuhu town centre on the western edge of the study area, and lower concentrations on the peninsula to the east of the motorway. However, we have hypothesised elsewhere (chapter A11) that this gradient may be due to differences in emission intensity between the western and eastern halves of the study area. This may explain the slight tendency for the model to overestimate concentrations in the south-west of the domain (ie where distance to the coast is greatest). The lack of a physical basis for a LUR model means that the 'cause' of the model error cannot be established using LUR alone. Further analysis of the modelling and the introduction of alternative independent variable formulations may further reveal whether domain size or variable specification is influencing model performance.

Overall, it has been shown that LUR works reasonably well in these two areas of Auckland, especially in the slightly larger study area of Mangere. This leads us to consider that there may be a size of study area below which LUR does not work as effectively, and that this area is somewhere in the range of 4km².

A11 Roadside corridor model

A11.1 Overview

The roadside corridor model (RCM) was developed by NIWA to meet a specific need. Its purpose is to describe the localised gradient in concentrations of traffic-related air pollutants within the 'roadside corridor', ie a parcel of land approximately ten to hundreds of metres either side of a busy road. The model was designed to represent an advance on the generalised dispersion curves provided in MfE (2008) by allowing consideration of three important factors:

- different dispersion at different times of day, especially during the morning traffic peak
- different wind climates, eg between Auckland and Wellington
- consideration of the relationship between road orientation and prevailing winds, so that dispersion may not be symmetrical on either side of the road.

It is important to note that the model describes:

- long-term (annual) average dispersion only
- passive (non-reactive) pollutants only (eg not nitrogen dioxide).

The model does not represent a new way of describing the processes of emission and dispersion. The model, in its present form, is based upon two existing models (Ausroads and VEPM) and is dependent upon their limitations.

Full details of the purpose, concept, development and formulation of the model is provided in *NZTA research report 451 'Tools for assessing exposure to land transport emissions'* (Longley et al 2011).

A11.2 Inputs

As input, the RCM requires only the average traffic volume (either AADT, am peak, inter-peak or pm peak), receptor distances and orientation of the road (eg a north-south aligned road has an orientation of zero degrees, and east-west road has an orientation of 90 degrees). Fleet-average emission factors are required. The RCM provides the option of treating the northbound and southbound carriageways independently or in combination.

Table A11.1 lists the input data we used to represent the southern motorway and our fixed monitoring stations. The traffic data was derived from traffic count data collected by the NZTA during the campaign, as reported in chapter A8. The fleet-average emission factors are averages derived from the emission modelling described in chapter A8.

Table A11.1 Inputs to RCM

Line sources	2, representing each carriageway	
Traffic volume	Southbound	Northbound
AADT	59,500	56,000
Am peak	7600	6800
Inter-peak	7600	7600
Pm peak	9300	7200
NO _x emission factor (g km ⁻¹)	1.088	1.065

Line sources	2, representing each carriageway	
Traffic volume	Southbound	Northbound
PM ₁₀ emission factor (g km ⁻¹)	0.077	0.078
Receptor distances	30, 150, 250 m	
Road orientation:	165 degrees	

A11.3 RCM output

Table A11.2 RCM results

	24 hr		Am peak	
	PM ₁₀	NO _x	PM ₁₀	NO _x
30m east (Deas Place Reserve)	3.4	47.0	10.1	141.5
150m east (25 Deas Place)	1.3	17.9	3.9	55.1
250m west (Luke Street)	1.3	18.8	3.8	53.1

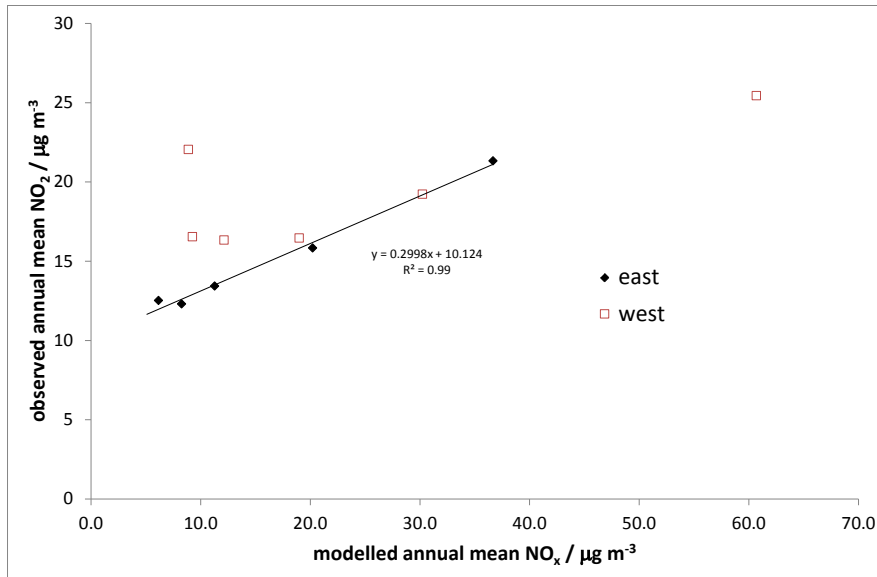
A11.4 RCM validation

A11.4.1 Roadside concentration gradients

A direct validation is not possible at this time. Full validation of a gradient requires more measurements at multiple distances from the road over the range for which the model is intended to be used. The RCM predicts roadside concentrations of passive pollutants for which VEPM provides emissions rates, ie PM₁₀, NO_x, VOCs and CO, not reactive or dynamic pollutants, such as NO₂ or PNC. Due to logistical and resource limitations, the only pollutant for which concentrations were observed continuously at multiple downwind locations was NO₂.

Figure A11.1 presents the predicted long-term mean concentration gradient of NO_x perpendicular to the motorway predicted by RCM, plus the average concentration gradient of NO₂ along 'row 4' of the passive monitoring network (see figure A4.1), also perpendicular to the motorway. It can be seen there is a strong correlation between modelled NO_x and observed NO₂ to the east of the motorway, meaning that RCM reproduces the shape of the concentration. The slope of the gradient (0.3) suggests a constant NO₂:NO_x ratio across this distance range. To the west of the motorway the observed concentrations are relatively greater than the modelled concentrations. This is most likely due to the presence of other road traffic sources in this sector, especially at ~ 1 km distance (Atkinson Ave and Otahuhu town centre).

Figure A11.1 Mean modelled (RCM) NO_x and mean observed NO₂ along a transect perpendicular to the motorway ('row 4' of the passive monitoring network – see figure A4.1 – equivalent to Luke Street)



Periodically, brief mobile measurements were made of CO, BC and PNC. These measurements were intended to be used for comparison with the RCM. However, as described in chapter A7, further analysis of this data is required to remove the influence of brief close encounters with vehicles on non-motorway roads in order to extract the underlying spatial patterns in the neighbourhood. For this reason, the mobile monitoring data is not currently suitable for comparison with the RCM output. Further research is planned to conduct this additional analysis.

A11.4.2 Concentrations at setback locations

Direct comparison of concentrations predicted by the RCM and concentrations measured at our continuous fixed sites is not possible. This is because the RCM predicts the contribution of the modelled roadway (SH1) only, whereas observed concentrations arise from the sum of all sources. However, the contribution from SH1 alone can be estimated by assuming that at any given time the upwind site represents the contribution of all other sources, and subtracting the hourly upwind concentration from observed concentrations at both upwind and downwind receptors.

We conducted this analysis providing a pseudo-observed time series of road-only concentrations for the whole IOP (ie period for which all three sites were reporting valid data), with the results shown in table A11.3 and directly compared with the RCM predictions in figures A11.2 to A11.5.

Table A11.3 Pseudo-observed mean concentrations attributable to motorway sources only (ie observed downwind concentrations with background concentrations subtracted)

	24 hr		Am peak	
	PM ₁₀	NO _x	PM ₁₀	NO _x
30m east (Deas Place Reserve)	2.7	97	2.8	153
150m east (25 Deas Place)	1.6	13	0.1	21
250m west (Luke Street)	-0.9	11	-0.2	27

Figure A11.2 Comparison of pseudo-observed and modelled mean NO_x concentrations at the three sites over the full 24-hour period

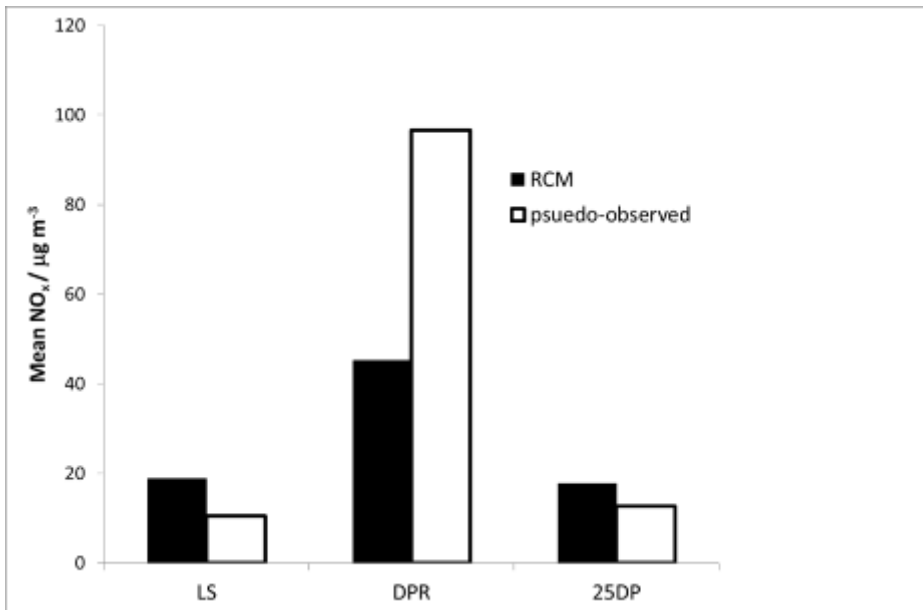


Figure A11.3 Comparison of pseudo-observed and modelled mean NO_x concentrations at the three sites over the am peak period only

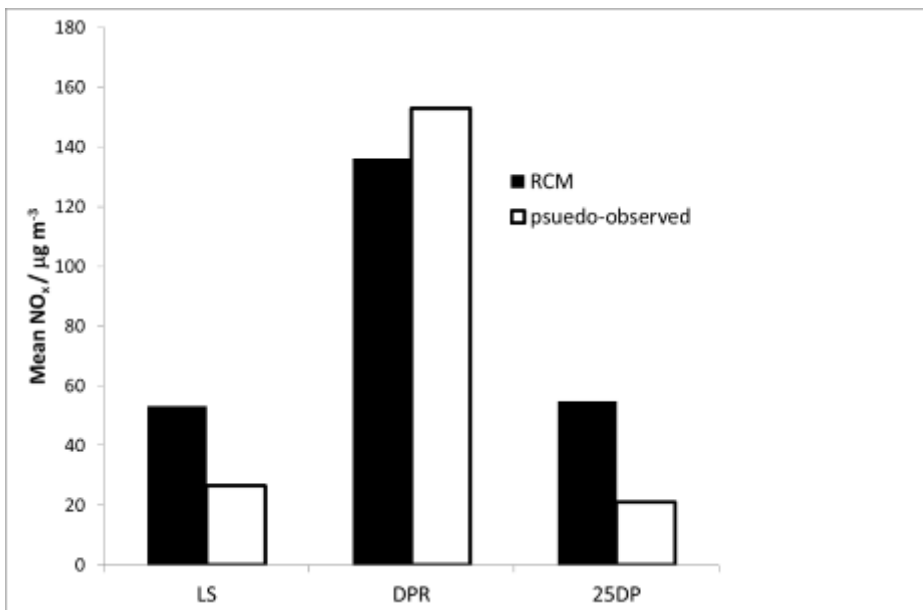


Figure A11.4 Comparison of pseudo-observed and modelled mean PM_{10} concentrations at the three sites over the full 24-hour period

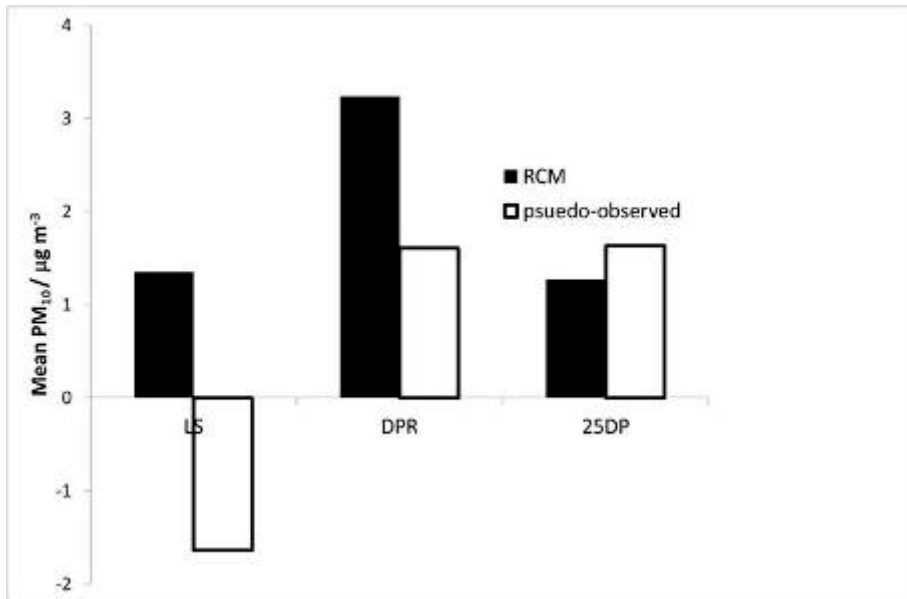
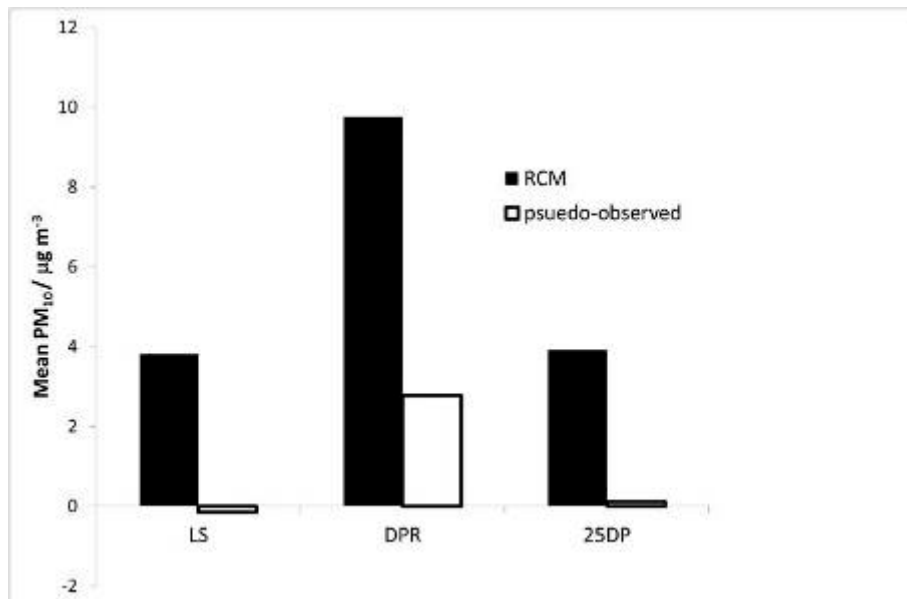


Figure A11.5 Comparison of pseudo-observed and modelled mean PM_{10} concentrations at the three sites over the am peak period only



These results show that

- The RCM performed relatively well at predicting NO_x impacts at setback sites on a 24-hour basis but underpredicted impacts at the roadside.
- The RCM predicted am peak NO_x at the kerbside well.
- The RCM appeared to underpredict absolute NO_x concentrations at the kerbside on a 24-hour basis.

- The RCM underpredicted the concentration gradient in NO_x (and by implication all traffic pollutants) between the roadside and setback sites, consistent with our findings for the performance of Ausroads, upon which the RCM is based.
- The RCM predicted the PM_{10} concentrations at the kerbside site would be $1.9\mu\text{g m}^{-3}$ higher than at the near setback site. This compared with an observed difference of $1.7\mu\text{g m}^{-3}$ between kerbside and near setback. RCM also predicted no difference between the near and far setback sites, which was also confirmed by observations.
- Further conclusions for PM_{10} were difficult to draw because of the difficulty in producing a valid comparative observational pseudo-dataset. This was revealed in the assignment of negative values for Luke Street (ie caused by lower PM_{10} at Luke Street when downwind than at the 'upwind site' of 25 Deas Place) indicating 'contamination' of the upwind PM_{10} concentration by near-field rapidly diluting non-motorway sources. Furthermore, the relatively low sensitivity of the instruments also hampered a robust comparison.
- Figure A11.5 appears to show the RCM overpredicting am peak PM_{10} concentrations. However, it is not possible to determine to what degree this is due to the difficulty in extracting the road-only signal from the observed PM_{10} data.

A11.5 Discussion

Evaluating the RCM is challenging for two major reasons:

- 1 The model predicted the impact of the motorway alone, whereas observations necessarily included all sources. This was exceptionally so for PM_{10} where non-motorway sources dominated and the impact of the motorway emissions were of a similar order of magnitude as the sensitivity of the instrumentation available to detect it. As distances from the motorway increased, the influence of local traffic sources became harder to avoid. Removing the contribution of non-motorway sources from observational data is difficult and prone to uncertainties. Ongoing NIWA research is planned to continue to reduce these uncertainties.
- 2 Near-road concentration gradients are very steep, with observational results from this project indicating that they are actually steeper than Ausroads/RCM predicted. This means that concentrations, both modelled and observed, were very sensitive to location or distance from the motorway. Small errors in defining source-receptor distances will lead to large differences in outcome. This is especially critical for the kerbside site of Deas Place Reserve, which is specified in the RCM as being 32m from the source (ie. the motorway centreline), yet is actually only ~2m from the edge of the southbound off-ramp.

A12 Synthesis

A12.1 This study in an international context

In a literature review conducted at the start of this project (available on request) we identified 18 major international observational campaigns of roadside air quality. In the intervening period, no other substantial studies have been reported in the peer reviewed literature, although we are aware of studies which have been conducted but are not yet reported (including the CAFEH study in Boston and a year-long campaign alongside the I-15 freeway in Las Vegas). However, in mid-2010 a new synthesis of observational roadside studies was published (Karner et al 2010), which we consider in this chapter.

The median duration of the studies in our review was seven days, and the maximum was four weeks. Thus our study, of 25 weeks in total length for the continuous observations, eight weeks for the IOP and 12 months for the passive observations, was substantially longer than all other studies identified.

Several studies have shown that findings, especially relating to NO₂ levels and gradients, can be dependent on the proportion of heavy duty vehicles on the road. The proportion of HCVs on the study section of the Auckland southern motorway (< 5% during daytime hours) was at the lower end of the range across those reported in previous studies.

A12.2 Roadside air quality in our study area

A12.2.1 What do the fixed sites represent?

Most roadside air quality studies are based on the concept of the roadside corridor – a strip of land around a major road in which concentrations of traffic-related air pollutants are elevated above background levels. Some studies have sought to explicitly evaluate the width of that corridor, while corridor width can also be inferred from other studies. Two recent reviews have sought to do this (Zhou and Levy 2007, Karner et al 2010) pooling data from 33 and 41 studies respectively. Both reviews have highlighted the technical difficulties inherent in doing this due to a) differences in study design and the way results are reported, b) difficulties in establishing what the background is, and c) sensitivity to the definition of the corridor edge, considering that that edge is gradual and continuous rather than distinct. Nevertheless, a value of around 150m for passive pollutants is consistently reported from studies in many different locations. Results for NO₂ are discussed separately below.

A key question in interpreting the data in our project was to establish whether the setback fixed monitoring sites, at 150m east and 250m west of the motorway, lay within or outside the motorway's corridor of influence. We could not rely on other continuous monitors at a greater distance to provide a comparison, as the next nearest monitors were >5km to the east. However, three sources of evidence were available:

- 1 On average, concentrations of NO_x at both setback sites were approximately equal, but substantially lower than at the kerbside site, despite the setback sites being different distances from the motorway (see figures A6.5 and A6.6).
- 2 Ausroads modelling (using the baseline scenario, see chapter A8) predicted that concentrations would have fallen to 20% of the on-road peak at ~110m, and 10% at ~275m; however, comparison with observations showed that Ausroads was underpredicting dispersion by a factor of ~3 in the daytime.

- 3 Complete analysis of the mobile data would provide further details regarding the rate of decay of carbon monoxide, BC and UFP concentrations up to 1 km either side of the motorway.

This combined evidence suggests that, consistent with international evidence, our fixed sites were outside the corridor of motorway influence, or at least on the outer edges, and thus represented 'setback' rather than 'roadside' locations in terms of passive pollutants (NO_x, particulate mass concentrations, CO).

Two studies have identified that the corridor may expand during night-time conditions (Zhu et al 2006) and especially in the pre-sunrise hours (Hu et al 2009). This is a meteorological phenomenon, and these two cited studies were both undertaken in Los Angeles which has a substantially different climate from Auckland. Further analysis could probe our dataset to determine whether periods of expanded corridor width were evident during our study.

Despite being 30m from the motorway centreline, our kerbside site was as close as it was possible to get to the motorway, with only ~4m between the sampling inlets and the edge of the southbound off-ramp. Compared with other sections of the motorway, the presence of the off-ramp increased the distance from the monitoring trailer to the motorway centreline. The off-ramp also carried decelerating (ie low-emitting) traffic. Nevertheless, this kerbside site was representative of several residential properties within the study area (ie properties within ~30m of the motorway centreline), and presumably, elsewhere on the Auckland motorway network.

A12.2.2 Roadside nitrogen dioxide concentration gradients

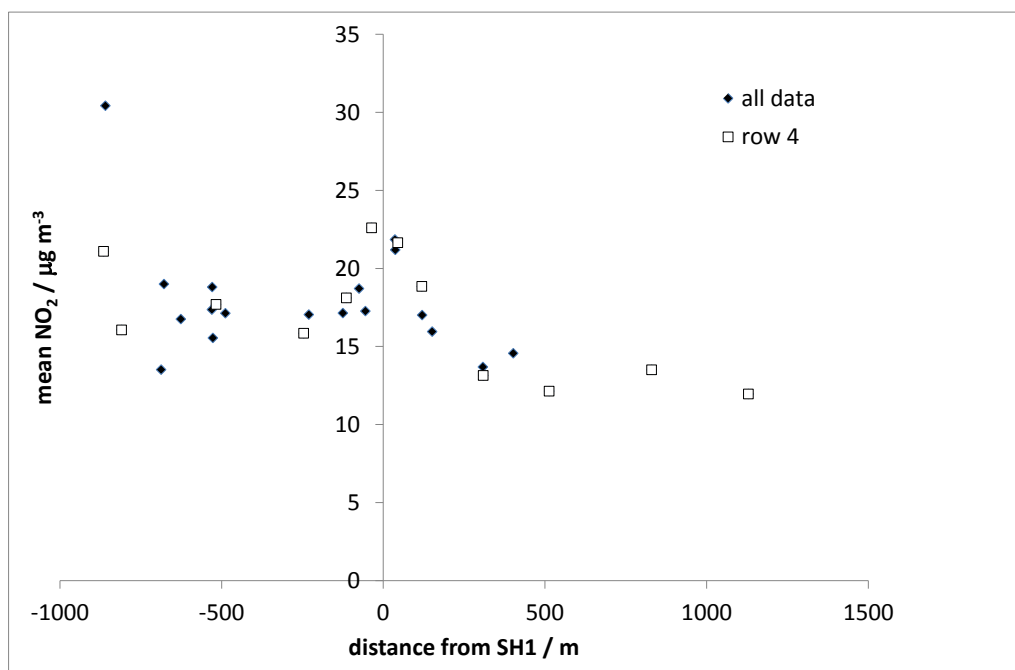
In their meta-reviews of roadside studies Zhou and Levy (2007) and Karner et al (2010) both noted that roadside concentration gradients were significantly different for NO₂ compared with passive pollutants, with generally less steep gradients and wider corridor widths. Zhou and Levy (2007) reported that roadside NO₂ levels decayed to background levels over a 200m to 500m distance, whereas Karner et al (2010) reported a continuous decay without an identifiable corridor edge.

The ARC conducted three campaigns of measurements of NO₂ along roadside transects (ARC 2007). The largest study was in 2006, consisting of four sites to the west and eight sites to the east (the predominantly downwind side) of Auckland's south-western motorway, approximately 5km west of our study area. Measurements were conducted up to 550m from the motorway. A gradient was seen on the east side to at least 200m, and potentially up to 500m. There was a quite different (steeper) gradient on the west (upwind side), and lower concentrations at each equivalent distance. Western side concentrations were taken to represent background and subtracted from concentrations on the eastern side, and a linear regression was fitted to the resulting values. Smaller studies had been conducted by the ARC on the eastern side only of the southern motorway at Penrose in 1997 and Otahuhu in 1998. A linear model was also fitted to the data at Penrose (up to 250m). The Otahuhu study consisted of four sites up to 250m along the same transect that made up row 8 in our study (see figure A4.1). A much steeper decay was seen than at either Penrose or Mangere, implying a much narrower corridor effect at Otahuhu than Penrose, despite it being the same motorway with similar volumes of traffic. The researchers found that a logarithmic function described the concentration decay in the results much better than a linear function.

Figure A1.1 shows the campaign-mean concentrations of NO₂ at each passive observation site in our study with respect to the shortest distance of the site to the centreline of the motorway. Row 4 - the longest cross-study-area transect (see figure A4.1) - is highlighted. It can be seen that a gradient in concentration with increasing distance from the motorway occurs up to ~400m. It can also be seen that there appears to be a 'background' value of ~ 12µg m⁻³ (10µg m⁻³ once corrected for seasonal bias, see section A4.6). These wider corridors of influence for NO₂ have significance for population exposure. Although concentration

elevations in the 100m to 500m range are substantially lower than in the first 100m, the exposed population is likely to be much larger.

Figure A12.1 Campaign-mean NO₂ concentrations (from the passive monitoring network) plotted against minimum distance to the motorway centreline (negative distances refer to the west side of the motorway)

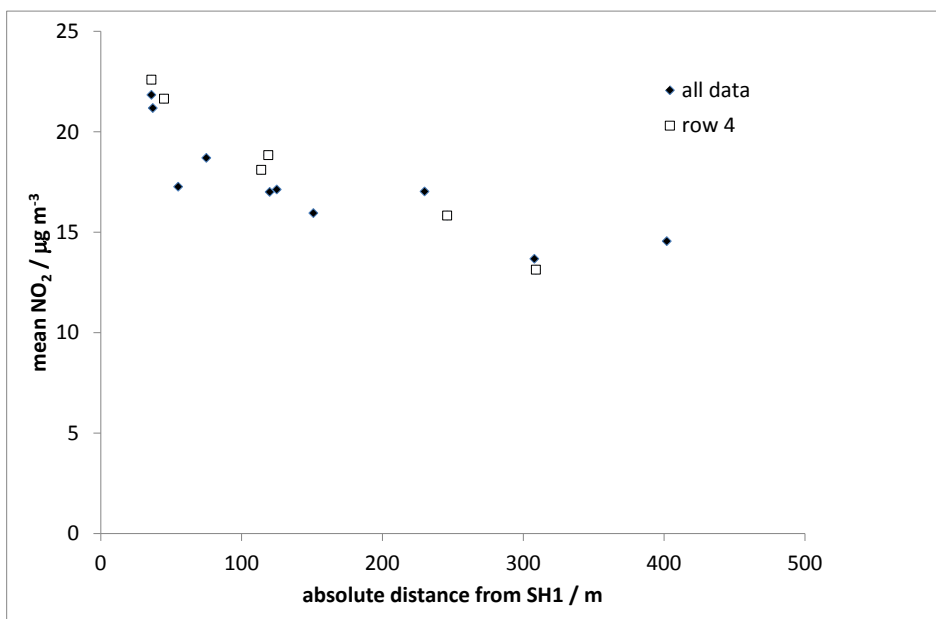


Note: Row 4 refers to a continuous transect across the study area passing through the study central reference point (see figure A4.1)

Figure A12.2 presents a 'zoomed in' view of the first 400m from the motorway against absolute distance (ie with sites from the west side (negative distance) projected onto the 'eastern' side).

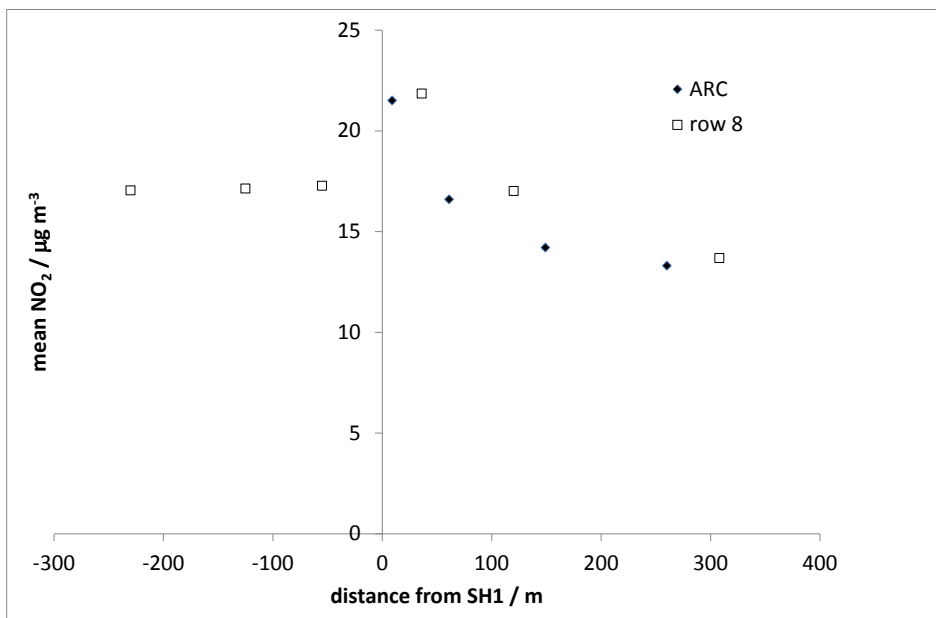
Figure A12.3 shows our data for row 8, along with data captured by the ARC along the same transect over three months in summer 1997–98. The strong similarity is apparent. We propose that the lower absolute concentrations in the ARC study are due partly to seasonal bias.

Figure A12.2 Campaign-mean NO₂ concentrations (from the passive monitoring network) plotted against minimum absolute distance to the motorway centreline up to 400m



Note: Row 4 refers to a continuous transect across the study area passing through the study central reference point (see figure A4.1)

Figure A12.3 Campaign-mean NO₂ concentrations against minimum absolute distance to the motorway centreline for row 8, including mean data from a study in summer 1997-98 (ARC 2007)

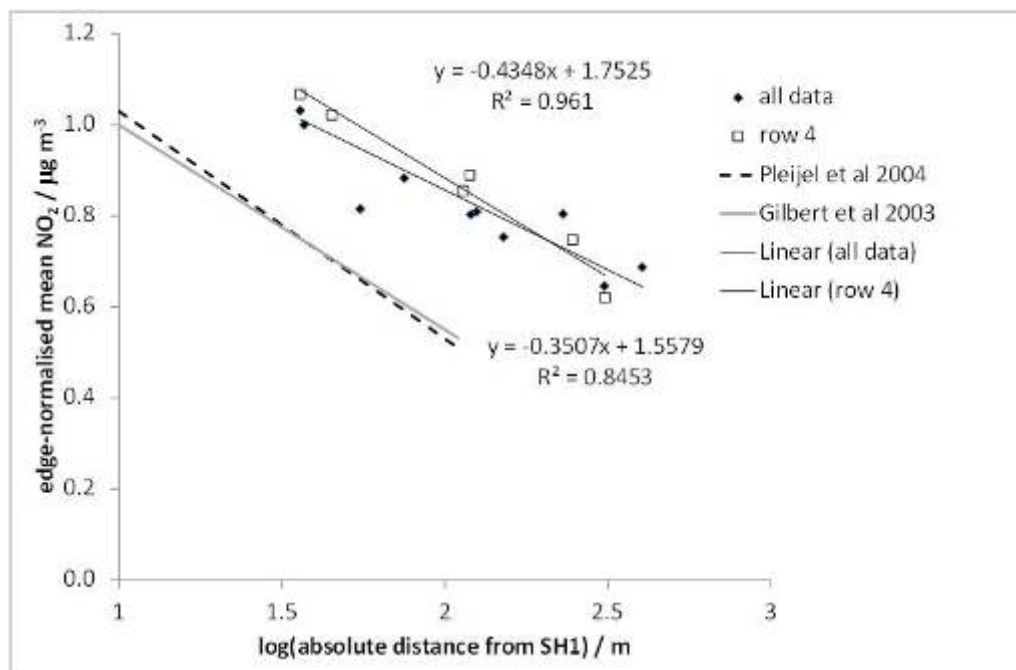


In studies in suburban Canada and rural Sweden, Gilbert et al (2003) and Pleijel et al (2004) used a logarithmic function to relate observed NO₂ concentrations (normalised to concentrations at 10m from the road) to the distance from the road (x), of the form $C_x = C_b + C_0 - k \log(x)$ where C_x is the concentration at distance x, C_b is the background concentration and C_0 is the concentration at distance = 0.

From our passive observations we excluded all data beyond 400m from the motorway, and then fitted a logarithmic relationship to both the whole remaining dataset and the row 4 dataset alone. The results are summarised in table A12.1 and figure A12.4.

For row 4 we found a very similar decay rate to that of Gilbert et al's (2003) study to which our study bears close resemblance. On the other hand, we found somewhat stronger decay rates than in the more rural studies of Pleijel et al (2004) and Clements et al (2009).

Figure A12.4 Campaign-mean edge-normalised NO₂ concentrations (from the passive monitoring network) plotted against the log of the minimum absolute distance to the motorway centreline up to 400m



Note: Row 4 refers to a continuous transect across the study area passing through the study central reference point (see figure A4.1). Also shown are the logarithmic decay fits to observational data from two similar previous studies (see text).

Table A12.1 Parameters describing decay of observed NO₂ with distance from the motorway from our study and three similar previous studies

Study	Location	Logarithmic decay parameter
Our study (all data)	Suburban Auckland	0.35
Our study (row 4)	Suburban Auckland	0.43
Gilbert et al (2003)	Suburban Montreal	0.45
Pleijel et al (2004)	Rural Sweden	0.5

A few studies have extended their analysis beyond the ~400m mark. Gilbert et al (2003), sampling up to a distance of 1310m, showed continued reduction in concentrations up to ~500m in a residential area of Montreal. In a follow-up experiment, Gilbert et al (2007) reported a logarithmic decay in NO₂ concentrations with distance persisting up to 3km in a study in Montreal. However, the authors conceded that they were not able to determine whether impacts from emissions from other roads (including intersections) in the suburban study area were contributing to this effect.

To resolve and explain the differences in NO₂ gradients between these studies we offer the following hypothesis. In many rural studies it is assumed that all of the observed NO₂ above background levels is related to emissions on the 'subject' road. This assumption has often been carried through to urban studies even though the assumption is more questionable due to the inevitable presence of other roads and emission sources. This has previously made it difficult to determine if the concentration gradients observed were representative of all roads, of rural/urban roads only, or were even more location specific.

First, it should be noted that the studies with steeper gradients (Clements et al 2009, Pleijel et al 2004) were conducted in rural areas in which the subject road was the sole emissions source. Our study, the previous ARC study in Otahuhu and the Montreal study (Gilbert et al 2003), which all displayed similar logarithmic decays, were all undertaken in suburban residential locations in cities with generally good air quality. Shallower gradients were found at Penrose and Mangere (ARC 2007), and in this study when we consider our larger dataset rather than just row 4. The Penrose and Mangere studies were undertaken in industrial areas, with the transect of the Mangere study in particular passing through a zone frequented by large diesel trucks. In our study non-motorway traffic volumes are minimal along row 4, but significant along row 7 (Princes Street, which provides Otahuhu's point of access to the motorway) and row 8.

Second, Gilbert et al (2003) reported significantly higher NO₂ concentrations downwind of the highway in Montreal compared with upwind, and the same phenomenon was reported by ARC (2007) in Mangere. If we similarly take 'upwind' to correspond to the west side of our study area and 'downwind' to correspond to the east side (based on prevailing winds as indicated by the wind roses for both the campaign and Auckland generally) then we find the opposite. There are slightly lower concentrations on the 'downwind' side for comparable distances. As indicated in figure A11.1 and figure A4.4, the lowest campaign-mean concentrations observed using our passive network were 12µg m⁻³ on the east side of the motorway and 14µg m⁻³ (or 16µg m⁻³ if one site in open fields, B2, is disregarded) on the west side.

Combining these pieces of evidence we propose that the variations in NO₂ gradients, especially in the 100m to 500m zone, are caused mainly by the variations in the density of local non-motorway traffic – a hypothesis further supported by the LUR modelling. In our study we estimate that emissions from this local non-motorway traffic are causing the 2 – 4µg m⁻³ difference in long-term NO₂ levels between either side of the motorway, related to the higher density of non-motorway traffic on the west side. This is also consistent with observed NO₂ levels from the passive network being lower than those predicted from modelling the emissions from the motorway alone.

We are aware of only one published study which has attempted to analytically isolate and quantify the impact of other sources on roadside air quality using a physically based approach. Henry et al (2011) used a method based on back trajectory analysis to estimate the contributions of a nearby airport and local industrial area on concentrations measured along the I-15 freeway in Las Vegas. The method was used to show that air parcels arriving at the freeway via the airport, which was 1+ km distant, carried a BC concentration of 1.3 – 1.4µg m⁻³, relative to the freeway's contribution of up to 1.1µg m⁻³. This is a promising technique which may be applied to our project database in future to help establish the degree to which our sites represented background air quality in general, how that status may depend upon wind speed and direction, and to quantify the impact of specific sources (such as non-motorway road traffic sources).

A12.2.3 Absolute contribution of motorway emissions to local air quality

On average, we estimated that the motorway contributed an additional 100µg m⁻³ to NO_x, 6µg m⁻³ to NO₂ and 1.7µg m⁻³ to PM₁₀ at the kerbside site above the setback sites. The estimate for NO₂ increased from 6 to a maximum of 10µg m⁻³ if we considered the wider corridor width for NO₂ and used the passive monitoring data.

Much of the existing literature on roadside studies does not attribute absolute concentrations to the subject road, so it is difficult for us to compare these values with other studies. For NO_x we were able to identify only one comparable study to ours, which was the study of the I-15 freeway in Las Vegas (Henry et al 2011). The traffic volumes in the Las Vegas study are described rather vaguely as 'in excess of 150, 000 vehicles [as an annual average]'. This should be compared with 116,000 in our study. The Las Vegas (I-15) study showed an average absolute contribution of the freeway to NO_x concentrations of ~80µg m⁻³ at 25m from the freeway's edge (55m from the centreline), 40µg m⁻³ at 135m (165m from the centreline) and 32µg m⁻³ at 300m (330m from the centreline). Alternatively this can be expressed as an increment of ~40µg m⁻³ between 25m and 135m. This can be compared to our observation of a difference of 100µg m⁻³ between the kerbside and setback sites. However, it should be noted that our kerbside site was much closer than in the Las Vegas study (~5m compared with 25m). Also we must assume that dispersion conditions are likely to have been different between Auckland in autumn/winter and Las Vegas. Although details are not provided by Henry et al (2011) they do note that 'Low wind speeds were quite common in these data: the distribution of wind speeds at station 2 was highly skewed with a peak (mode) at 1.3m s⁻¹'. Combining the expected higher traffic volumes and less efficient dispersion in Las Vegas implies that, on that basis alone, we would expect a higher contribution to NO_x concentrations from the I-15 than the Auckland southern motorway. The fact that we appear to observe the opposite may imply higher NO_x emission factors in Auckland, which is consistent with an expected older traffic fleet.

Comparisons for PM are available from more studies, but a strong caveat needs to be made that these studies involved not only further variability in local meteorology and traffic fleets, but also variability in measurement technology. We estimated an average contribution of the Auckland southern motorway to PM₁₀ of 1.7µg m⁻³, determined from the difference in concentrations between kerbside and setback sites. A similar increment of 2.1µg m⁻³ over a comparable distance was reported for PM_{2.5} by Zhu et al (2002) downwind of the I-405 freeway in Los Angeles. The I-405 is one of the busiest roads in the world with annual average daily volumes in excess of 300,000. Similarly, Reponen et al (2003) reported an increment of 2.0µg m⁻³ in PM_{2.5} between 80m and 400m downwind of the I-71 freeway in Cincinnati.

We found that the maximum contribution of the motorway as a 24-hour midnight-to-midnight average was 6 – 7µg m⁻³. We found no other studies reporting comparable statistics. We also found that peak values tended to occur on days with extended periods of low winds, but not all periods of low winds led to increased motorway contribution. This implies that timing is likely to be influential, ie whether the low winds coincided with periods of high or low emissions and whether such periods were confined to a single day, or split over two days, thus their contribution to 24-hour PM₁₀ being split between two days. Further analysis of the dataset could provide some insight into the climatology of PM₁₀ peaks.

A12.2.4 Relative contribution of motorway emissions to local air quality

On average, we estimated that the ratio of mean kerbside to setback concentrations was 2.1 for NO_x, 1.6 for NO₂ and 1.1 for PM₁₀, based on the fixed sites. When using the passive monitoring data (ie taking data from over 400m from the motorway to represent 'setback') the ratio for NO₂ varied from 1.5 (max site/average of all background sites) to 2.0 (max/min). A larger number of studies report the degree to which roadside concentrations are elevated relative to an assumed background level. Making comparisons is somewhat limited, however, by differences in the way the background level is estimated, and is also sensitive to the method and correct specification of the distance of the roadside site to the road. Also, it must be borne in mind that international and even inter-road comparisons may be telling us as much about variability in the background as variability in the subject road's contribution. Karner et al (2010) attempted to summarise the kerbside/background ratio for a large number of disparate studies finding values of 1.8 for NO_x, 2.9 for NO₂ and 1.3 for PM₁₀. In the I-15 Las Vegas study, the ratio for NO_x was 1.4

for comparable sites to our study, or 1.9 when background was taken from a predominantly upwind site. In our study the predominantly upwind and downwind setback sites reported the same mean concentration so that our ratio is the same using both sites. Background NO_x concentrations were comparable in our study and the Las Vegas study (both ~ 90 µg m⁻³) so the higher ratio in our study is consistent with higher NO_x emission factors in Auckland, as described above.

A12.2.5 Commentary on particulate matter

We estimated that emissions from the Auckland southern motorway contributed to a 1.7 µg m⁻³, or 10% increase, on average, in PM₁₀ concentrations at the roadside. We also estimated that on days when peak 24-hour PM₁₀ concentrations were observed, the absolute increment was little different at 2.1 µg m⁻³, but that the relative contribution was reduced to 7%. These values are below those summarised by Karner et al (2010). However, fair comparison is very difficult to achieve as many of the studies referenced by Karner et al (2010) featured much shorter campaigns than our study, a wide range of measurement technologies, and a wide range of traffic conditions. We have noted above that the contributions estimated were of a similar order of magnitude as the sensitivity and accuracy of the instruments we deployed (the beta attenuation monitor, the instrument used at most permanent regulatory monitoring stations in the Auckland Council network). Uncertainties in our estimate also arose from the difficulty in assuring that two of our key study design assumptions – that there were no significant sources or sinks between our three fixed monitoring sites – and that concentrations measured at our upwind site were representative of air masses arriving at our downwind sites – were valid for PM₁₀. The fact that our three sites could not be on the same trajectory for all wind directions, and that we were monitoring in an urban area with local roads, homes and businesses introduced the possibility that our upwind site was either over- or under-estimating ‘background’ concentrations and that the increment at the downwind site attributed to the motorway only might be due to other sources. We are unable at present to determine whether these errors may have introduced a net positive or negative error to the estimation of motorway contribution. On the other hand, the long duration of our study relative to others, and the three-site design, hold out the possibility that the errors in our study might be less than those in previous studies.

We were unable to distinguish between particles generated through wear and re-suspension processes (eg brake and tyre wear, road dust) and tailpipe emissions. Future analysis of UFP concentrations, plus filter samples taken at the roadside site during the campaign may provide some insight into the role of these two components.

In the context of health risk assessment, it is important to note that the relatively small contribution of motorway emissions to PM₁₀ identified in this study, and the fact that the peak observed concentrations were below the NES for PM₁₀ by a significant margin, do not necessarily translate into a negligible risk to the residents in our study area, and especially in the roadside corridor. Initial analysis of UFP and BC concentrations (outside the scope of this report) indicate a much greater contribution from the motorway, consistent with many other studies (see Karner et al 2010). Rather, our study corroborates a large body of international research showing that the PM₁₀ metric, and especially the technology conventionally used to measure it, is relatively insensitive to freshly emitted particles from road vehicle tailpipes due to their small size and hence low mass relative to more aged particles and particles derived from mechanical abrasion. Many of the particles emitted from traffic on the motorway will be below the limit of detection of the instruments we used (or alternatives we could have used), while other proto-particles could be in the gas phase within the roadside corridor, transitioning to the particle phase, and hence influencing PM₁₀, only sometime after they have advected away from the neighbourhood in which they were emitted. The consequences of the way particles are observed and quantified for health are still unclear and the subject of active research around the world.

A12.2.6 Observed air quality relative to New Zealand standards and guidelines

It is well recognised that exceedences of New Zealand's NES for PM_{10} ($50\mu\text{g m}^{-3}$ as a 24-hour average with one permitted annual exceedence) occur in many towns and cities, and that the majority of exceedences are observed at monitoring sites that are far less affected by near-field traffic emissions than our study monitoring sites (MfE 2009). Our observational results indicate that our study area was compliant with the standard by a significant margin (at least $7\mu\text{g m}^{-3}$), even at the predominantly downwind kerb of an eight-lane highway. On those days when PM_{10} was elevated we estimated that emissions from the highway contributed $\sim 2\mu\text{g m}^{-3}$, ie $\sim 5\%$. Peak 24-hour PM_{10} concentrations in our study area coincided with region-wide peaks in PM_{10} at both roadside and non-roadside monitoring sites. These observations indicate that the contribution of the motorway to the risk of NES exceedence was marginal.

A recent review (Longley et al 2008) found that the risk of exceedence of the NES for NO_2 ($200\mu\text{g m}^{-3}$ as a one-hour average with nine permitted annual exceedences) and the risk of exceeding the WHO guideline of $40\mu\text{g m}^{-3}$ as an annual average were roughly the same at most sites. Existing knowledge indicates that that risk is limited primarily to highly trafficked locations with restricted dispersion (eg street canyons) very close to busy intersections and close to very busy motorways in Auckland (Longley et al 2008; NZTA 2011). Our observational results indicated that our study area was compliant with the standard and WHO guideline by a substantial margin. However, in contrast to PM_{10} , the spatial variability in annual average NO_2 across the study area was much greater, ranging from $12\mu\text{g m}^{-3}$ to $26\mu\text{g m}^{-3}$, related directly to proximity to the major traffic sources of SH1 and Atkinson Avenue.

A12.2.7 Implications for other locations in Auckland and New Zealand

Average daily traffic volumes on the section of SH1 through our study area are $\sim 120,000$ per day. Based on traffic count data provided by the NZTA we estimate there is at most 30km of motorway in Auckland with a similar or greater volume of traffic. Traffic volumes in the range 150,000 to 200,000 have been reported between on the 5km stretch of SH1 between the Central Motorway Junction and Greenlane interchange. In other words, some locations outside our study area support traffic volumes up to 2/3 higher. If we assume that motorway traffic volumes correlate approximately with emissions, then the average contribution of a motorway to roadside PM_{10} levels around the busiest sections of Auckland's motorways could plausibly raise from $\sim 2\mu\text{g m}^{-3}$ to $\sim 3\mu\text{g m}^{-3}$, with marginal significance for NES compliance. However, the impact on NO_2 compliance could be much more significant. A simple 1:1 correlation between traffic and concentrations would suggest a potential increase in the contribution of a motorway to kerbside annual mean NO_2 from $14\mu\text{g m}^{-3}$ to $23\mu\text{g m}^{-3}$. Once added to a baseline value arising from non-motorway sources, which will be elevated in central Auckland, the risk of exceedence becomes significant.

An approximate 1:1 correlation between traffic and NO_2 concentrations within the study area is supported by the LUR modelling, and other modelling of NO_2 impacts conducted by NIWA independently of this project (Longley and Gadd 2010). Ongoing research based on this project's database will further clarify and quantify that relationship. The validity of assuming a 1:1 correlation between traffic volumes, emissions and concentrations and using that to draw conclusions about other locations, especially those which may be subject to more congested conditions cannot be assessed at this time and should be treated with caution as the relationship between congested motorway conditions and vehicle emissions is subject to considerable scientific uncertainty.

We know of no scientific reason why our broad findings should not be applicable outside Auckland. Variations in different cities are likely to be the result of different:

- background/baseline levels

- dispersion rates arising from more or less windy climates, or different dispersion conditions
- prevalences of congested traffic conditions.

A12.3 Roadside air quality assessment

A12.3.1 Performance of Ausroads and RCM

We found that the performance of Ausroads was acceptable during the night time but it underpredicted dispersion during the day time. Above $\sim 50\mu\text{g m}^{-3}$ of NO_x the impact of this on predictions of absolute NO_x concentrations at the 25 Deas Place setback site (ie once combined with a background estimate) was relatively insensitive to concentration, with an average overprediction of $36\mu\text{g m}^{-3}$. Expressed another way, for concentrations above $200\mu\text{g m}^{-3}$ the overprediction was an average of 16%. We are not aware of any other study which has been designed in such a way as to explore this phenomenon. Neither are we aware of any previous study which has reported similar overprediction using either Ausroads, or the CALINE suite of models upon which Ausroads is based.

As indicated before, one of the potential explanations of this underprediction of the dilution conditions is the impact of traffic-induced turbulence, particularly on the very near-field concentrations. However, we note that most widely used roadside dispersion models either fail to take this phenomenon into account, 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 can only be considered to be at the stage of 'research' at this point and have yet to be widely adopted, tested and used in regulatory or policy contexts.

We propose that further research is conducted to explore these issues. Further analysis of our extensive project database may reveal the causes of the overprediction and indicate potential solutions.

Since this project began, details of another modelling suite have been published which is based on a similar concept and need as the RCM, namely the 'reduced form emission/dispersion model' (RFEDM) developed at the University of Michigan (Batterman et al 2010). Whereas the RCM parameterises the results of combining Ausroads and VEPM, the RFEDM combines the dispersion model CALINE4 and the US-based vehicle emission model MOBILE6.2. Both RCM and RFEDM were developed independently without knowledge of the other, yet using remarkably similar methodologies. Predictions of the RFEDM were compared with data from two existing sites in Detroit where roadside air quality had previously been collected by the Michigan Department of Environmental Quality for regulatory purposes. However, the monitoring was not designed with model evaluation in mind and therefore the RFEDM evaluation cannot be compared directly to our study. Specifically, in Detroit only a single monitor was in place so that neither dispersion nor background could be independently assessed, only the net performance of the whole model package. Overall, there was a large degree of scatter in RFEDM predictions relative to observations. The RFEDM underpredicted mean concentrations by 33%, whereas our RCM overpredicted mean concentrations at 25 Deas Place by 35%. Batterman et al (2010) recognised that part of the reasons for relatively poor performance was due to the lack of time-varying background data. With the use of local time-varying background data our overall prediction performance using Ausroads and VEPM was far superior to the Detroit study.

A12.3.2 Background air quality

As described in chapter A8, model-based assessments of roadside air quality are usually based on combining the contribution of emissions from traffic on the 'subject' road with estimates of 'background' air quality. Although definitions of background air quality may vary depending on the context, it is often defined to

mean air quality arising from sources other than the subject road. It can also, more generally, be thought of as air quality beyond the influence of any single dominant source – ie beyond the roadside corridor. This latter interpretation is often termed ‘urban background’ while the former is referred to as ‘baseline’.

Minimal guidance and information is currently available on how to quantify background air quality in New Zealand. This has led to concerns over inconsistencies between regulatory assessments. At the time of writing both Auckland Council and the NZTA were developing consistent approaches to the preparation and use of data representing background air quality for the purposes of industrial resource consents and road project assessments respectively. Here we synthesise the findings of our project as they relate to informing this guidance.

First, we consider whether it is meaningful and appropriate to attribute a uniform background across our study area. We have noted at several points in this report how PM_{10} is a relatively insensitive measure to road traffic emissions from a single road.

The difference in PM_{10} levels between our three fixed measurement sites was, on average, small compared with the sensitivity of the measurement technology. This implies minimal long-term spatial variation in PM_{10} and lack of sensitivity to the precise location of the monitor. However, we did note that highly localised and temporary emission sources (such as domestic burning) may have been biasing individual monitoring sites, especially at the 25 Deas Place setback site, probably due to its location in a residential garden embedded within a group of houses. Further analysis of our mobile data may help to better understand the prevalence and impact of these localised impacts.

The passive monitoring network allowed us to determine that, beyond the NO_2 corridor of up to 400m from the motorway (and the immediate influence of Atkinson Ave on the study area’s western edge), there were relatively consistent long-term concentrations, but with some systematic variation. Campaign-mean concentrations varied from 12 – 14 $\mu g m^{-3}$ on the east side of the motorway and from 14 – 19 $\mu g m^{-3}$ on the west side, which we attributed to variations in local traffic density. Thus, a continuous monitor placed at location J4 for instance (where we recorded the lowest campaign-average concentrations of 12 $\mu g m^{-3}$) would under-represent urban background concentrations at all our other locations across the study area by up to 56% (or 26% on average). The minimum observed NO_2 concentration in the study area (10 $\mu g m^{-3}$ when corrected for seasonal bias) agrees well with the baseline NO_2 concentration predicted by the LUR model (10.9 $\mu g m^{-3}$) and with the value determined in the Waterview area of west Auckland using a hybrid regression model (10.4 $\mu g m^{-3}$, Longley and Gadd 2010).

In the case of regulatory assessments, locally observed background data is rarely available, and the preferred approach is to use non-local data. We tested the impact of this, and the choice of data, in chapter A8. Overall we found that assessments of NO_x were quite sensitive to the choice of background site, especially if the available sites were likely to have a directional bias, so that they were closer to an upwind source in a certain range of wind directions. Where that directional bias was similar to the bias anticipated at the assessment receptors, prediction performance could be good. However, as soon as the biases were misaligned, prediction power reduced considerably (see figure A8.16). This result emphasises how sites which are biased in this way are not particularly suitable to act as sources of background air quality data unless the bias is compensated for. Ideally efforts should be made to choose sites with a corresponding bias if possible, or for the bias to be analytically compensated for. The effect is less pronounced for PM_{10} due to its relatively poor sensitivity and hence poor prediction performance even with locally observed background data (figure A8.22).

SOSE and RCM potentially offer means of managing directional bias in observational data. Figures A10.18 to A10.21 show how SOSE was used to analytically remove the impact of emissions from the southern motorway at our fixed sites to reveal the underlying background contribution, and how the prediction

matched well with observations on a diurnal average basis. We have previously shown (Longley et al 2011) how RCM can be used to achieve the same thing in a simpler manner (ie long-term mean and am/pm peak mean contributions). Future research is intended to show how passive monitoring can be used to predict long-term background concentrations in pollutants other than NO₂ and also predict diurnal variation and the probability of short-term peaks in the background.

A13 Recommendations for further research

A13.1 Generalising observational data

Data gathered in this project should be sufficient to investigate and quantify the impact of variations in short-term meteorology on passive monitoring data, or any monitoring data captured at time scales longer than an hour. From such an analysis it should be possible to derive a 'correction' method for compensating for atypical meteorology leading to the estimation of annual means which are more typical and repeatable.

Data gathered in this project should also improve the understanding of the relationship between short-term and long-term concentrations of pollutants in roadside locations, such as the relationship between diurnal average concentrations, the probability of peak hourly concentrations and long-term mean concentrations.

SOSE has proved to be useful for predicting roadside concentrations once the model has received training data. This holds out the potential for SOSE to be used as a 'virtual' monitor, useful for replacing monitors which have malfunctioned, been decommissioned, or to interpolate periods of monitor absence in 'round-robin' campaign monitoring. We recommend that the model is specifically tested in such a scenario. SOSE is not the only model capable of performing this function. We also recommend that alternative models (such as generalised additive models or ANN models) are tested alongside SOSE.

The transferability of the results from this project to other locations in New Zealand or internationally is somewhat dependent on the assumption that different traffic volumes are directly proportional to emission rates and hence concentrations. Although we believe such an assumption to be plausible in most comparable locations in New Zealand (ie alongside large free-flowing roads with a 'typical' fleet mix) this simple 1:1 assumption may not hold at locations characterised by substantial and persistent traffic congestion. The relationship between congested driving and emissions is poorly understood, but is likely to be impacted by the 'nature' of the congestion (ie the degree to which it is characterised by stop-start or accelerate-brake cycles). If reducing the uncertainty around such locations is a priority then a new study focusing on these is recommended.

A13.2 Sources and gradients in particulate matter

Several substantial questions regarding PM have arisen during this project. Although we have noted several times in this report how the beta attenuation monitors used in the project, and the PM_{10} metric in general, have a low sensitivity to traffic emissions, the large amount of information available within this project may still be exploited to reveal new understanding regarding:

- the emission rates, dispersion, deposition and concentration gradients of coarse particles, particularly road dust including brake and tyre wear
- the contribution of domestic home heating to our observed concentrations and the degree to which this might have confounded our conclusions
- gradients in PM beyond the immediate roadside corridor across the study area
- the influence of regional air quality events on local gradients.

Additional analysis using our project database could include:

- analysis of the mobile data, including disaggregation of the data into foreground sources (close encounters with individual vehicles) and background sources
- determining roadside gradients for CO, PM, BC and UFP from the mobile data
- investigation of periods for which concentration gradients are systematically different (as found during the pre-sunrise hours in Los Angeles by Hu et al 2009)
- analysis of the differences between modelled concentrations (Ausroads, LUR) and observed concentrations (mobile and passive data)
- detailed time series analysis to detect localised emission sources (eg domestic burning)
- analysis of pollutant ratios to allow prediction of one from another and to provide insight into differential source/sink processes
- trial of the back trajectory technique for attributing background contributions as described by Henry et al (2011).

The issues of low sensitivity to PM measures, and the fact that PM₁₀ measures do not distinguish road dusts from tailpipe emissions could be addressed by using additional data observed concurrently with this project. This includes measurements of BC and particle size distributions at Deas Place Reserve, and hourly size-segregated particle filter sampling also conducted at Deas Place Reserve by Landcare Research.

A13.3 Emissions and sources of NO₂

Although we have described the observed gradients in NO₂ and compared them with other studies we have not yet been able to explore the role of the chemical reactivity of NO_x. Previous studies have indicated that secondary formation of NO₂ from NO is the main reason why NO₂ corridors are wider than those of passive or conserved pollutants. Recent studies have shown how the ability of roadside dispersion models to correctly model roadside NO₂ gradients is dependent on not only the correct specification of the upwind O₃ concentrations, but also the NO₂/NO_x ratio at the emission source (Chaney et al 2011; Wang et al 2011). Our ability to assess this in our study was limited by the lack of NO or NO_x concentrations across the study area. However, we do possess upwind and downwind NO, NO₂ and O₃ concentrations at high temporal resolution. We recommend that our dataset is analysed with this question in mind. Also, a more focused observational campaign is required to capture downwind NO or NO_x data. However, CO or UFP data from the mobile dataset may offer a useful proxy for NO_x, and we recommend exploring this possibility.

A13.4 Use of off-site data in assessments

We have conducted an initial study on the sensitivity of modelling results to the choice of site for meteorological data and background air quality data. At present the results of this study are specific to our study area. We recommend that these findings are further investigated with the aim of generalising how the relationships between the data site and the subject site influence assessment error, and in which way. This research would be valuable in providing guidance for the conduct of air quality assessments, for evaluating the purpose and value of long-term monitoring sites and guiding the siting criteria of new monitoring sites, for both long-term regulatory sites and project-specific sites.

A13.5 Model performance and improvement

Our extensive observational and modelling database presents numerous opportunities for a more detailed analysis and evaluation of Ausroads (or alternative models). We have presented sensitivity test results based on mean outputs, but the full database is freely available and permits evaluation of any relevant statistics.

The cause of the underprediction of dispersion by Ausroads remains unresolved. We speculate that the very close proximity of the kerbside site to the road has introduced analytical difficulties because the model formulation is not designed to provide optimum performance this close to the source. For the purposes of understanding the model a site slightly more distant from the road may have been better. However, assessments at kerbside are still required and useful for air quality research and management, and the kerbside site was dictated by the wider needs of the project.

The analysis of the mobile data will provide a new dataset for use in evaluating Ausroads and RCM.

LUR models were generated independently for two locations. The models were optimised for local performance, but not for geographical transferability, as is the norm for this class of model. This resulted in the models being significantly different and potentially highly local. However, extended research should investigate the transferability of each model to the other location, or to a third location. Models could be re-optimised for transferability and general applicability rather than local performance. This research could identify a more general formulation to give acceptable performance over a wider range of locations. The issue of whether 'distance to coast' introduces confounding to the model should be resolved if geographical transferability is desired.

A13.6 Alternative models

Roadside dispersion models other than Ausroads (and its base-model, 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 US Environmental Protection Agency, 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). Newer developments in roadside dispersion modelling include new approaches to improving the performance of CALINE in non-perpendicular winds (Briant et al 2011), and the 'reduced form emission/dispersion model' (Batterman et al 2010), which is very similar to RCM but based around the US vehicle emission model MOBILE6.2. Our dataset is suitable for the evaluation of all of these models.

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 considered to be a tool in a developmental stage and has yet to be widely adopted for routine air quality management purposes. We also believe that our dataset is suitable for the evaluation of QUIC in the New Zealand context.

Recently a new 'hybrid' form of spatial air quality model has emerged (Jerrett et al 2005). Hybrid models come in several forms, but usually combine some aspect of a dispersion model with a LUR model. Since the start of this project, the NIWA team has begun to develop a hybrid spatial air quality model (currently named the traffic impact model, or TIM) by combining the RCM with a reduced form LUR model for NO₂ developed initially for the regulatory assessment of the 'Waterview Connection' motorway project in

Auckland (Longley and Gadd 2010). The NIWA team plans to evaluate the TIM using the Otahuhu project database in the near future. The TIM is designed to predict long-term concentration gradients across the study area, but, unlike the LUR, will do this for a wider range of pollutants and will attribute concentrations to emissions on individual roads. Further research is planned to apportion concentrations to emissions from different vehicle types and on different journey profiles.

A13.7 Exposure impacts

One of our original aims in this project was to enable and extend health impact analysis. Through this project we have delivered a comprehensive description of air quality in the study area, and in particular its variation in space and time and across multiple pollutants. Furthermore we have developed and demonstrated modelling tools for assessing impacts that can be 'exported' to other locations.

The data and tools unlock the capability to explore exposure distributions across our study area and elsewhere. Concentrations of air pollutants can, potentially, be attributed to every residence individually for both long-term exposure, and on an hour-by-hour basis. The role of the motorway in contributing to the total potential exposure of all residents is now quantifiable, as is the role of local traffic. Potential exposure can be converted into actual exposure once personal time-activity patterns, time spent indoors, the prevalence and magnitude of indoor sources and infiltration rates into buildings can be included in the assessment. This is currently the objective of an ongoing PhD project by one of the research team members.

A14 Conclusions

A14.1 Overview

- We have successfully created the only air quality dataset gathered in New Zealand with simultaneous observations at three points (all within 250m of the same section of motorway) expressly designed to isolate emissions on a motorway from other sources.
- Observed air quality at the study site was compliant with NES and WHO guidelines.
- On average, PM_{10} concentrations were elevated by 10% at the kerbside relative to setback sites.
- On average, NO_2 concentrations were elevated by 38% to 100% at the kerbside relative to setback sites.
- We found that the commonly used regulatory tool combination Ausroads+VEPM overpredicted concentrations at a site 150m from the motorway. Given the technical limitations of the Ausroads model this result is not wholly unexpected, but it has not (to our knowledge) previously been reported. As Ausroads forms the basis of NIWA's RCM, this model is also affected.
- A unique micro-scale NO_2 passive monitoring network was established and generated a year's worth of data. This dense network provided additional insight revealing that road traffic emissions led to substantial but highly localised increases in concentrations within ~100m of the motorway, and smaller increases up to a distance of 400m. We believe this is the first study of its type able to distinguish the contributions of the motorway and local roads.
- Further analysis is required to understand:
 - the reasons for (and hence solution to) the model weaknesses identified
 - the relationships between PM_{10} , NO_2 , NO_x , CO, BC and UFP across the study area
 - how to formulate improved models which perform better than the existing suite.

A14.2 Technical details

A14.2.1 Roadside air quality observations:

- Observations began on 2 April and ended on 29 September 2010. All three sites were reporting data for ~1400 hours.
- The highest recorded 24-hour PM_{10} concentration was $43\mu g m^{-3}$ (17 June).
- 24-hour PM_{10} concentrations above $25\mu g m^{-3}$ were only observed when daily minimum one-hour temperature was less than $8^\circ C$ and daily mean wind speed was less than $2m s^{-1}$.
- The average PM_{10} concentration was $17\mu g m^{-3}$ at both of the setback sites and $18.7\mu g m^{-3}$ at the kerbside site. Therefore, on average the motorway contributed $1.7\mu g m^{-3}$ (or 10%) to PM_{10} at the motorway's edge. Given that the sensitivity of the beta attenuation monitors used by us (and widely used for regulatory monitoring) is $\pm 2\mu g m^{-3}$, the long-term contribution of the motorway to PM_{10} is on the borderline of detectability.
- We estimated that on days when peak 24-hour PM_{10} concentrations were observed the absolute kerbside increment was $2.1\mu g m^{-3}$, but the relative contribution was reduced to 7%. On average, the absolute kerbside increment peaked at $2.4\mu g m^{-3}$ during peak traffic hours (7am to 6pm).

- The largest observed contribution was $37\mu\text{g m}^{-3}$, or ~one fifth of the NES. The largest observed difference between the kerbside and either setback site $6.9\mu\text{g m}^{-3}$, or 13% of the NES. The difference exceeded $5\mu\text{g m}^{-3}$ on nine occasions. On each of these occasions the daily mean wind speed was less than 2m s^{-1} .
- The highest recorded one-hour NO_2 concentration was $86\mu\text{g m}^{-3}$ (17 June at 10 am) at the motorway's edge and $55\mu\text{g m}^{-3}$ at 150m distance.
- Mean NO_2 concentrations at the Deas Place Reserve kerbside site were 7 – $10\mu\text{g m}^{-3}$ higher than at the setback sites.
- The average NO_2 concentration at the motorway's edge was $24\mu\text{g m}^{-3}$.
- Mean NO_x concentrations at the kerbside were approximately double those at both setback sites.
- The diurnally averaged difference in NO_x concentrations between the kerbside site and the setback sites clearly resembled the diurnal cycle in traffic volume, and peaked during the morning traffic peak at around $180\mu\text{g m}^{-3}$.
- Background NO_x concentrations were comparable between our study and a similar study in Las Vegas (both $\sim 90\mu\text{g m}^{-3}$) but absolute roadside concentrations were higher, which was consistent with higher NO_x emission factors in Auckland.

A14.2.2 Emission-dispersion modelling (Ausroads+VEPM)

- Multiple scenarios were modelled using the common dispersion/emission model pairing of Ausroads+VEPM. The 'baseline' scenario modelled a single line source with atmospheric stability and mixing height assumed to be constant, using on-site meteorological observations.
- Using the baseline scenario, Ausroads+VEPM was generally successful at predicting NO_x concentrations at the motorway's edge (model = $1.01 \times$ observed, $R^2 = 0.86$).
- For *all scenarios*, Ausroads was found to underestimate dispersion in the 150m zone by a factor of 1 – 3 times, with the factor tending to 3 in the daytime. This underestimation was found to be independent of VEPM.
- The consequence of this was that absolute concentrations of NO_x at 150m distance were overestimated by typically 20 – $50\mu\text{g m}^{-3}$ during daytime, and this error increased in periods of low wind speeds.
- Hourly PM_{10} at 150m distance was overpredicted by 0 – $15\mu\text{g m}^{-3}$ in the morning. Modelling overprediction appeared to be offset by the influence of un-modelled sources/processes in the afternoon and evening.
- For hourly estimated atmospheric stability, a well-established algorithm increased model overprediction at night.
- Using off-site meteorological data led to underestimates of concentration attributable to the motorway of 22% to 35% in five out of eight cases - the specifics of the site and dataset were significant.
- The error introduced by using longer 'historic' datasets was small in comparison to this spatial error.
- Use of the ARC artificial dataset reduced the 'error' relative to using a locally observed dataset to $< \sim 10\%$.
- Relative to the Luke Street dataset, using data from either of the two other (shorter) on-site masts led to a fairly consistent overestimation of mean concentrations at all receptors in three out of four cases.

- Modelling all eight lanes independently led to a prediction of concentrations up to 22% lower than modelling a single line source.
- In conclusion, we found Ausroads, not VEPM, contributed most to a tendency of this model combination to overpredict roadside air quality impacts for NO_x beyond the kerbside.
- We found no evidence of VEPM contributing to modelling error for NO_x. Indeed, analysis of the kerbside data suggests VEPM was predicting NO_x emissions to a high degree of accuracy within the context of the typical modelling performance.
- The best performing non-local site for background NO_x was Penrose. Compared with using locally-observed background data, the slope of the best fit changed little, but the scatter increased, lowering the R² from 0.86 to 0.75.
- Cumulative PM₁₀ was overpredicted by our modelling approach typically by 0 – 20µg m⁻³ (5 – 10µg m⁻³ on average).
- Due to the relatively low sensitivity of the PM₁₀ metric to road traffic emissions, and the inherent variability of data reported by beta attenuation monitors, the sensitivity of cumulative predictions to the source of background data was not as great for PM₁₀ as it was for NO_x.
- Our conclusions regarding PM₁₀ are necessarily tentative due to large underlying uncertainties. In general, our modelling estimates led to an overprediction of PM₁₀ at the kerbside, relative to observations. Given the good performance in predicting NO_x, we assume that Ausroads is not contributing to prediction error at the kerbside. In that case, the overprediction in PM₁₀ can be attributed to either:
 - overprediction of PM emission rates by VEPM, or
 - invalidity of the assumption that the setback Luke Street site adequately represented the background, ie all sources affecting the kerbside site except the motorway.

A14.2.3 Site-optimised semi-empirical (SOSE) modelling

- SOSE is a modelling tool which parameterises the relationship between meteorology, traffic and concentrations at a given point. It takes observational data as an input and predicts the time-series of concentrations at the same point as an output.
- **Basic SOSE modelling:** Each of the pollutants (CO, NO₂, NO_x, PM₁₀) were modelled for each of the three main observational sites. Predictions were made based on optimisation using the entire dataset (not independent estimates) as well as independent data (each week systematically removed from the training dataset in order to predict for the given week independently). In general, the model performed well for all pollutants and all sites. However, it tended to underpredict peak concentrations and slightly overestimate for low concentrations (as is the norm for regression models). Weekday predictions were better than weekend predictions, partly a reflection of there being fewer weekend days for optimising the model. The model performed better for the gaseous pollutants than for PM₁₀, probably due to the road dominating the source for the gaseous pollutants more than for PM₁₀.
- **Using SOSE 'background parameter' to estimate background concentration:** If there is a real physical basis to the SOSE model, then it would be expected that the average daily distribution of the background concentration estimated as the intercept in the regression parameter for downwind conditions would match the observed concentrations for 'upwind days' at the site on the opposite side of the road. There was very good agreement between predicted and observed background concentrations, especially for NO₂ (figures A10.17 and A10.18).

- Figures A10.19 and A10.20 show the contribution of the motorway and the background to the total concentration for the westerly and easterly wind sectors, respectively. The results suggest that the motorway contributes a large fraction of the CO and NO_x at night. The NO₂ is mainly not from the motorway, except during the middle of the day. Approximately 20% of the PM₁₀ originates from the motorway.
- Note that most of the NO₂ and NO_x originates from the motorway whereas this contributes only a small fraction to the PM₁₀. The background dominates during the evening hours for PM₁₀ whereas for NO_x, the motorway dominates consistently throughout the day.
- **Choice of meteorological monitoring site:** Each of the pollutants (CO, NO₂, NO_x, PM₁₀) were modelled based on meteorological data from each of the three sites (four pollutants x three AQ sites x three met sites) giving 36 modelled air pollution traces. The model performed well for all of the pollutants at all of the sites when using wind data from the setback sites (Luke Street and 25 Deas Place). The model performed poorly for all three air quality sites when wind data from the kerbside site (Deas Place Reserve) was used. The reason for this is unclear but may be a reflection of the unique wind environment in the immediate vicinity of the motorway. Meteorological measurements from the taller Luke Street mast appeared to be the most 'representative' of the region from the point of view of pollution dispersion. Model predictions for 25 Deas Place were better when using wind data from Luke Street rather the co-located data. This result is consistent with recommendations in the literature with regard to the siting of meteorological towers (more open sites are preferable to ones in built up areas). Model performance was evaluated using data from several other sites in the region for comparison (Onehunga, Pakuranga, airport). This gave a better range of separations between air quality monitors and meteorological stations, and also heights and local topography. Model performance seems to be best when meteorological data is used from:
 - (i) a high meteorological tower (10m better than 6m)
 - (ii) a meteorological tower is in an open area (more representative of the regional flow)
 - (iii) a meteorological observation site that is as close as possible to the air quality monitoring site, as long as (ii) is not compromised.
- **Box height estimates:** When VEPM was included as one of the input variables in the model, the 'box height' (the theoretical depth of air into which vehicle plumes were being mixed – a major control on dispersion) could be estimated. While some 'model instability' was apparent, a box height of less than about 10m at night and in the morning increasing to >100m by the late afternoon was apparent. The fact that box height estimates between pollutants for a given site were consistent suggests that the emissions estimates from VEPM between pollutants were in proportion (the ratio of NO_x/CO, for example, is close to correct).

A14.2.4 Passive/mobile monitoring and land-use regression modelling

- Passive monitoring of NO₂ was conducted every four weeks at 30 sites across the Otahuhu study area for a year. Supplementary passive monitoring of NO₂ was also conducted to a very similar schedule at 28 sites across Mangere.
- Annual average NO₂ in the study area ranged from 12µg m⁻³ to 26µg m⁻³ (or 22µg m⁻³ when one outlier influenced by a busy intersection was removed). This represented at least a two-fold variation over a relatively small study area.
- Three spatial features were apparent in the data:

- i. strong localised gradients within ~200m of the motorway
 - ii. localised gradients in close proximity to the other two significant roads in the study area: Atkinson Ave (on the far west of the study area) and Princes Street (running east-west through the centre of the study area)
 - iii. a larger scale gradient with slightly higher concentrations in the west and south compared with the north and east.
- Analysis suggests that the larger-scale gradient may be related to general traffic density, or to differences in dispersion characteristics, perhaps due to the influence of the creeks bordering the north and east side of the study area. LUR modelling uses the variable 'distance from coast' to help predict this gradient, but it is unclear if this variable has a physical meaning, or is confounded by local gradients in traffic.
 - Mobile monitoring of CO, PM, BC, UFP and CO₂ covering every street in the study area, lasting approximately an hour, was conducted 20 times by bike and six times by car, covering morning, noon, evening and night periods in May, June, July and August 2010. This data is particularly challenging to interpret due to the close proximity of plumes from other vehicles. Ongoing research is planned to further process, analyse and fully exploit this dataset.
 - An additional passive monitoring campaign was conducted in the nearby community of Mangere, around the south-western motorway (SH20), for the purposes of exploring the generality and transferability of LUR models.
 - LUR models were generated to describe the spatial pattern in long-term average nitrogen dioxide across the study area, and the Mangere area (independently), at 20m resolution. Although both models exhibited a bias towards the null (slight over-estimation of minima and under-estimation of peaks) common in regression models, performance of the Mangere model was good by international standards (R² of 0.77). Performance of the Otahuhu model was weaker (R² of 0.49). This may be due to the very fine spatial scale of the study area within which there is limited geographical variability in land use with which to predict spatial variation in air pollution. Nevertheless, model performance was still acceptable with the main features of observed spatial variation adequately described.

A14.2.5 Roadside corridor model

- Predictions of the contribution of the motorway to annual average concentrations of NO_x and PM₁₀ on both sides of the motorway were made using the roadside corridor model (RCM) recently developed by NIWA (Longley and Gadd 2011).
- Direct evaluation of the RCM predictions was not possible as motorway-only concentrations could not be measured. However, the shape of the predicted concentration gradient perpendicular to the motorway was very similar to that observed using the NO₂ passive monitoring with a residual unmodelled component to the west of the motorway, consistent with the influence there of unmodelled traffic sources (ie local roads and Otahuhu town centre).
- A partial model evaluation was attempted by constructing a pseudo-observed time series of motorway-only concentrations at the project's fixed continuous monitoring sites. This exploited the presence of an 'upwind' site which we assumed represented non-motorway contributions. The evaluation showed good predictive power for NO_x at the kerbside in the am peak, and an overprediction when a full 24-hour period was modelled. The model underpredicted the observed dispersion between the kerbside and setback sites, consistent with the similar underprediction by Ausroads, upon which the RCM is currently based.

- Evaluation of the RCM's predictions for PM_{10} was hampered by the low sensitivity of the PM_{10} instruments, and probably confounded by near-field domestic woodsmoke sources breaking the assumption that upwind sites represented all non-motorway sources.

A14.2.6 Synthesis of findings in the study area

- In lieu of analysis of the mobile data, the combined pieces of evidence suggested that, consistent with international evidence, our fixed sites were outside the corridor of motorway influence, or at least on the outer edges, and thus represented 'setback' rather than 'roadside' locations in terms of passive pollutants (NO_x , particulate mass concentrations, carbon monoxide).
- The combination of passive and mobile monitoring with LUR modelling revealed that traffic on the local road network was significant. Our findings regarding the reduced impact of the motorway relative to model-based expectations made local roads relatively more important than might previously have been thought. This was revealed by consistently higher concentrations to the west of SH1 (a grid of streets connected to Otahuhu town centre and the rest of Auckland) compared with the east (short, low-traffic streets serving the local area only) within our study area, despite the west side being predominantly upwind of the motorway. We also observed significant influence of traffic accessing the motorway via the Princes Street interchange, so that the presence of the interchange probably increased concentrations in this neighbourhood. (Quantifying this increment is the subject of planned ongoing research.) This indicates that the local road network contributes to an elevated baseline to which the motorway adds a highly localised additional burden. For assessment purposes we recommend that equal attention is paid to quantification of that baseline as emissions from the subject road itself. This project has indicated ways in which this may be done.

A14.3 Further implications for road project assessment

- RCM and Ausroads appear to be conservative at setback sites. This may make them suitable for screening applications.
- RCM has good potential for describing roadside NO_2 gradients and being used as a screening tool. Ongoing research is currently investigating this.
- The probability of NES exceedance at roadside locations is dominated by the baseline, ie the concentrations due to all other sources, including surrounding roads, at any given site. For example, the relatively large margin of compliance for NO_2 (WHO guideline and NES) in our study area was due in large part to the relatively low levels of non-motorway traffic. Our results show that elevated levels of local traffic would substantially raise the baseline.
- However, our results imply that exceedance of the NES for NO_2 is unlikely in other Auckland locations with similar traffic volumes and relatively open topography leading to good dispersion. Our modelling suggests that a doubling of traffic would lead to an approximate doubling in the contribution of the motorway to NO_2 levels, although this could not be confirmed in our study. This is, however, consistent with NO_2 levels observed near SH1 in Newmarket and the Central Motorway Junction.
- We have shown that the prediction performance of cumulative assessments for pollutants for which road traffic sources dominate, and hence have a high sensitivity to proximity to traffic sources (eg NO_x), is very dependent on the source of background data. This sensitivity leads to a directional bias in data from any site that is more affected by traffic emissions sources in some wind directions than in others. If the directional bias of the background site and the assessment receptor align (ie they are both downwind of the key source in the same wind direction) then double counting is likely to occur

as the 'background' site is over-influenced by a road source. On the other hand, if the biases are misaligned the proxy site being used may seriously underestimate the background in the subject area. Ideally efforts should be made to choose sites with minimal or acceptable bias, or for the bias to be analytically compensated for. The effect is less pronounced for PM_{10} due to its relatively poor sensitivity to traffic sources. This leads to relatively poor prediction performance even with locally observed background data, and hence relatively smaller impact from using biased background data..

A14.4 Further implications for health risk assessment and epidemiological applications

- The RCM was intended for this kind of application. However, this project has revealed some limitations. It is currently unclear whether the RCM in its present form represents a better means of assessing population exposure to traffic emissions on major roads than alternative approaches (such as interpolation of passive monitoring, inverse-distance-weighting, LUR, or simple proximity measures). Ongoing research will conduct a wider evaluation of RCM (including any future development which may arise as a result of the current project) specifically for the public health application.
- Roadside corridor impacts in Auckland appear to be highly localised, but not trivial. Assessments in urban areas should not rely on considering a single dominant road only, but the local network of roads also.
- PM_{10} is a poor (insensitive) measure of the potential health impact of vehicle emissions on air quality.
- NO_2 was found to exhibit distinct spatial gradients which could be related to spatial patterns in traffic volumes. Ongoing research will determine how these gradients are related to gradients in other traffic-related pollutants.
- LUR modelling has been shown to be an effective, low-cost tool for making assessments of spatial variation in traffic-related air pollution from fine (km) to super-fine (10s of m) scales.

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Annex A: Mobile observations

AA1 Overview

Mobile measurements of particle mass and number concentrations, BC, CO and CO₂ were conducted on a fixed route which covered every road in the study area. These approximately hour-long surveys took place on 25 occasions. The purpose was to gain insight into local spatial variations in concentrations across the study area at different times of day. Analysis of this dataset is particularly challenging and will require further research effort to interpret.

AA2 Aims

Mobile monitoring was conducted across the study area in order to:

- 1 Provide far greater spatial coverage than could practically be achieved using continuous methods
- 2 Obtain detailed spatial information at specific times of the day (morning traffic peak, evening traffic peak, inter-peak, evening during heating season)
- 3 Help understand the relative significance of non-motorway sources
- 4 Provide an initialisation/validation dataset for future modelling assessments.

It was not possible to meet aim 3 within the scope of this project.

AA3 Methods

AA3.1 Platforms

Two mobile platforms were used for mobile monitoring: a bicycle and a car.

In the case of the bicycle, miniature battery-powered instruments were carried in a bag secured onto a bracket fastened to the front of the cycle. For the car, instruments were placed on the rear passenger seat and sampled outdoor air via a stainless steel inlet through a false window.

AA3.2 Instruments

Tables AA.1 and AA.2 list the instruments used for mobile monitoring. For the cycle-based monitoring each instrument was self-logging (internal clocks were synchronised at the start of each survey). For the car-based monitoring, data was logged centrally using a custom-built Labview software application on a central computer.

Table AA.1 Air quality instruments designated to bike-based monitoring

Parameter	Instrument
CO	Langan T15n personal monitors
CO ₂	TSI Q-Trak
PM ₁₀	Grimm 1.107 spectrometer
UFP	TSI 3007 portable condensation particle counter
Position	GPS-enabled Nokia N82

Table AA.2 Air quality instruments designated to car-based monitoring

Contaminant	Instrument
UFP	TSI 3007 portable condensation particle counter
BC	Magee AE21 aethalometer
Position	GPS

AA3.3 Routes and rationale

A route was chosen that covered every public road within the study area. The route was as similar as practical between car and bicycle, and was repeated in the same way each survey. The route was also designed to pass all of the passive monitoring sites. For the bicycle the survey circuit took approximately 100 minutes. For the car, the circuit took approximately 40 – 50 minutes.

AA4 Data coverage and quality assurance

The bicycle was used 20 times during four different time periods (five times each): am peak, inter-peak, pm peak and night (10 pm – midnight). The car was used over five evenings (6pm to midnight) to enable the use of an aethalometer (too large to carry on the bicycle) to identify BC from home heating emissions. The car also made multiple circuits of the same route.

Table AA.3 Bicycle-based surveys

Survey #	Date	period
1	3 May 2010	Pm peak
2	5 May 2010	Pm peak
3	5 May 2010	Inter-peak
4	6 May 2010	Pm peak
5	7 May 2010	Inter-peak
6	16 May 2010	Pm peak
7	3 Jun 2010	Night
8	15 Jun 2010	Night
9	16 Jun 2010	Night
10	17 Jun 2010	Am peak
11	17 Jun 2010	Night
12	24 Jun 2010	Night
13	1 Jul 2010	Inter-peak
14	9 Jul 2010	Am peak
15	9 Jul 2010	Inter-peak
16	9 Jul 2010	Pm peak
17	12 Jul 2010	Am peak
18	12 Jul 2010	Inter-peak
19	13 Jul 2010	Am peak
20	14 Jul 2010	Am peak

Table AA.4 Car-based surveys

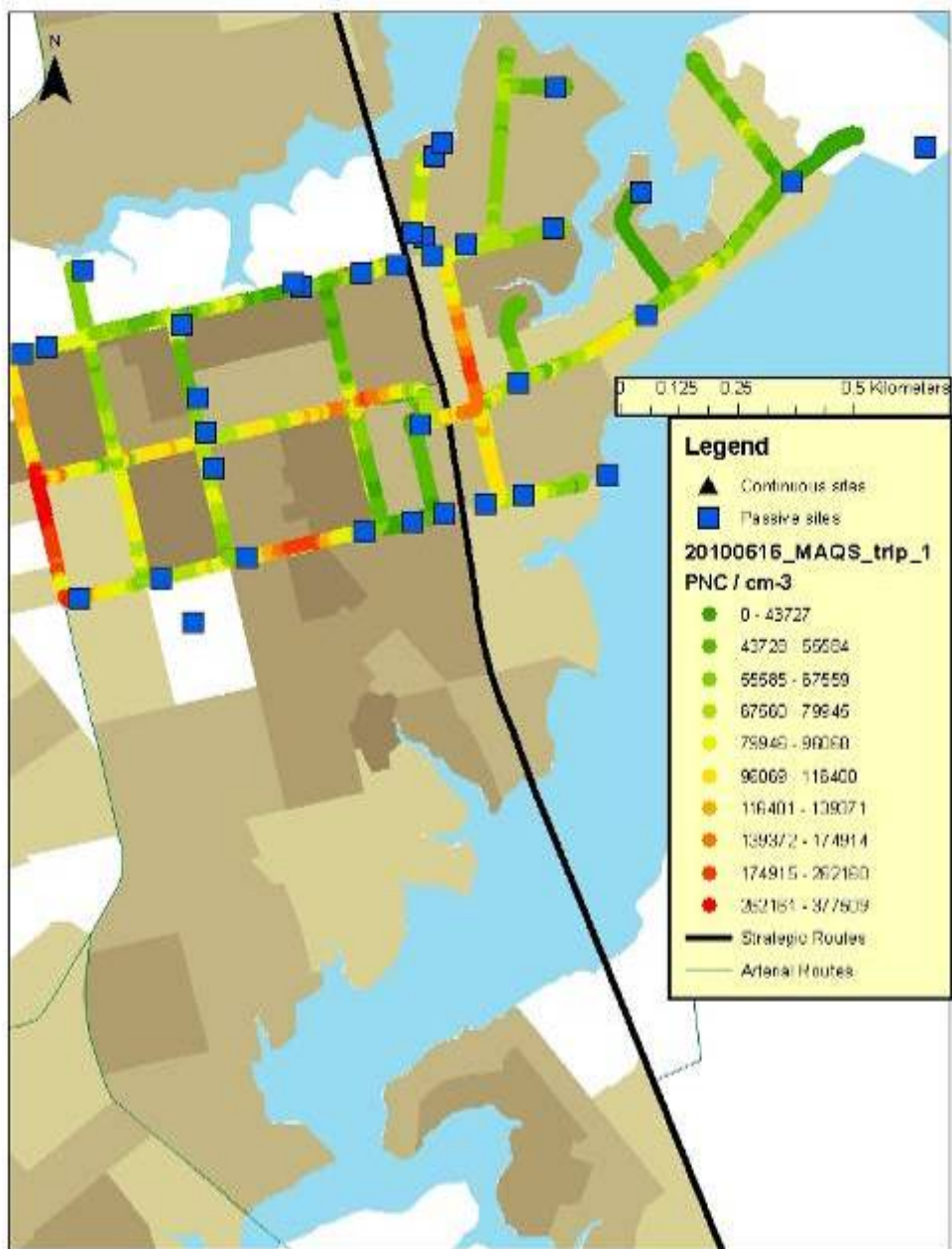
Survey #	Date	#Circuits
1	16 Jun 2010	3
2	17 Jun 2010	5
3	1 Jul 2010	4
4	12 Jul 2010	3
5	10 Jul 2010	4

No standard absolute calibration procedure exists for the particle instruments used in the mobile observations. However, these instruments, and the associated CO and CO₂ instruments, are to be used in this project principally to provide relative measures, ie spatial variation across the study area. For this purpose, each mobile instrument was run alongside each other and the fixed instruments at Deas Place Reserve for ~1 week in April or May. Instrument-specific calibration curves were derived from this data.

AA5 Results

Data from mobile monitoring is particularly difficult to interpret. This is because measuring on-road means that measured concentrations do not just represent local air quality, but can also be considerably biased by sampling exhaust plumes from nearby vehicles. This is more significant at times of greater traffic flow. A secondary complication is the distance travelled by the monitoring platform between collecting an air sample at the inlet, drawing that sample into the instrument and the instrument recording a value. This time varies between instruments, while the speed of the vehicle also varies. During that time the vehicle may have moved onto a different road. Thus ascribing an observed concentration to a particular location is analytically challenging. An example of the kind of data generated from a single circuit is shown in figure AA.1, which displays a map of PNC across the study area. Some of the regions of red points are likely to be attributable to local traffic or even individual vehicle plumes.

Figure AA.1 Example of location-tagged concentration data collected from a mobile platform



Note: PNC were recorded from a vehicle on 16 June 2010. High concentrations (red) are seen on the eastern (downwind) side of the motorway, in Otahuhu town centre (far left) and at a signalised intersection west of the motorway interchange. Brown-scale background represents population density. Passive and continuous measurement sites are also shown (see chapters A4 and A5 respectively).

Although trial analysis to remove biases from the observed concentrations was conducted we are unable to present quality controlled results at this time. Analysis will continue outside the scope of this project.

Appendix B: Glossary

AADT	average annual daily traffic
AIC	Akaike information criterion
AQG	air quality guidelines
ARC	Auckland Regional Council
BTEX	benzene, toluene, ethylbenzene and xylenes
CAU	census area unit
CO	carbon monoxide
CO ₂	carbon dioxide
CPC	condensation particle counter
EPA	Environment Protection Authority (Victoria, Australia)
GIS	geographic information system
HCV	heavy commercial vehicles
IOP	intensive observational period
LCV	light commercial vehicles
LUR	land-use regression
metset	meteorological dataset
NES	national environmental standards for air quality
NIWA	National Institute of Water & Atmospheric Research
NO	nitric oxide
NO ₂	nitrogen dioxide
NO _x	oxides of nitrogen (NO + NO ₂)
NTMG	New Zealand Map Grid
NZTM	New Zealand Transverse Mercator
O ₃	ozone
P-G	Pasquill-Gifford
PM	particulate matter
PNC	particle number concentration
QUIC	quick urban and industrial complex (model)
RCM	roadside corridor model
RFEDM	reduced-form emission-dispersion model
RMSE	root-mean-squared error

RSD	relative standard deviation
SH1	State Highway 1
SOSE	site-optimised semi-empirical (model)
TIM	traffic impact model
UFP	ultrafine particles
VEPM	vehicle emission prediction model
WMO	World Meteorological Organisation