Determining the health risks and ecological impacts of particulate matter arising from vehicle brake and tyre wear and road-surface dust

Part 2 – Sensitivity analysis and source-apportionment assessment

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This research was conducted between March and July 2023, and under a previous policy context. For example, the research was developed and undertaken under the 2021–24 Government Policy Statement for Land Transport. Consequently, references contained in the research note may be to policies, legislation and initiatives that have been concluded or repealed. Please consider this in your reading of the note and apply your judgement of the applicability of the findings to the current policy context accordingly.

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The authors would like to thank NZ Transport Agency Waka Kotahi for the opportunity to carry out this work and our reviewer, for their thoughtful and helpful comments on earlier drafts.

Abbreviations and acronyms

HAPINZ 3.0	Health and Air Pollution in New Zealand Study (updated 2016)
NEE	Non-exhaust emission
NIWA	National Institute of Water and Atmospheric Research
NO_x	Nitrogen oxides
NZTA	NZ Transport Agency Waka Kotahi
PM	Particulate matter
PM_{10} and $PM_{2.5}$	The mass of particulate matter with a diameter of 10 or 2.5 µm or less, respectively
P–G	Pasquill–Gifford
SH	State highway
VEPM	Vehicle Emissions Prediction Model

Contents

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Abstract

In 2021, the National Institute of Water and Atmospheric Research (NIWA) completed a comprehensive literature review on the non-exhaust emissions (NEEs) from road transport being discharged to air and water. That review represents part 1 of the current research.

A gap analysis, carried out as part of the literature review, concluded that although there is wide uncertainty around NEEs, there is also uncertainty around the impact that NEEs have in the tools and on the outcomes for which emission factors are commonly used in New Zealand. Thus, the review recommended further work to get an understanding of how the choice of emission factors affects model outputs.

This research note sets out to fulfil the initial recommendations of the literature review in part 1, namely to:

1. survey existing source-apportionment data to determine what detailed analysis is possible that will help to ground-truth current NEE factors, with the proposed analysis to inform stage 2 of the research
2. determine the magnitude of the impact that NEE factors have in air-quality modelling by undertaking a sensitivity analysis
3. determine the need for any further monitoring; and if there is a need, to describe what kind of monitoring would need to be undertaken.

This work shows a discrepancy between the contribution of NEEs to roadside PM_{10} predicted using Vehicle Emissions Prediction Model (VEPM) 6.3 and to that resolved from source-apportionment studies, with the latter predicting an average contribution in the order of 1 μg m^{-3} greater than the former. Although several factors may be contributing to this discrepancy, we find that the most plausible is that the 'missing' PM, which is not predicted by the VEPM, mostly relates to dusts that are resuspended from the road surface by the action of tyre contact or traffic-induced turbulence. This resuspension is not accounted for within the VEPM. In addition, it is unlikely that we can assess the accuracy of the current VEPM brake and tyre emission factors using existing New Zealand data.

Recommendations for further work fall into smaller investment suggestions with relatively immediate impact, and longer-term and larger investments into research, as follows.

Lower-cost immediate impact:

- introduce a crustal matter emission factor into the VEPM for NEE estimation
- change the method used to derive motor-vehicle-derived coarse PM in the Health and Air Pollution in New Zealand Study
- work with councils to continue long-term time series of traffic-related air pollutants.

New higher-cost experimental and observational studies:

- observational study in and around Lyttleton Tunnel on State Highway 74
- national survey of road-dust loading
- transect monitoring of high-dust roads.

1 Introduction

1.1 Background

Road transport is one of the primary sources of airborne particulate matter (PM) in urban environments and a key driver of air pollution's burden of disease worldwide (Health Effects Institute, 2020). PM contains particles over different size ranges and with varying chemical composition, which can in turn undergo atmospheric reactions and transformations influenced by location-specific meteorological factors such as temperature, insolation, humidity, wind speed and rainfall (Baldwin et al., 2015). There are key differences in particle size and composition for the various transport-sector emission sources. Tailpipe emissions of particles from fuel combustion are primarily less than 2.5 µm, with most in the ultra-fine size range (<0.1 µm), and with emissions from diesel vehicles dominating over other sources. Non-exhaust emissions (NEEs) are primarily composed of particles over a range of larger sizes, being the dominant transport contribution of particles in the 2.5 µm to 10 µm range ($PM_{10–2.5}$), and are mainly generated by mechanical abrasion processes.

While transformational technological improvements, coupled with emission-control regulations and policies, have led to a substantial reduction in exhaust emissions from road traffic (Henneman et al., 2021), currently on-road NEEs, such as those generated from brake, tyre, clutch and road-surface wear or road-dust resuspension due to traffic movement, are unabated (Beddows & Harrison, 2021). Most recent data from a number of European cities indicates that the contribution of NEEs to PM concentrations was comparable or even larger than the contribution from exhaust emissions (Saraga et al., 2021). Similar results have been observed for Auckland receptor modelling data, where it appears that NEEs became the dominant motor-vehicle source of PM from 2015 (Davy & Trompetter, 2021).

In recent years, NEEs have gathered international attention due to their:

- contribution to ambient PM loadings
- composition and potential health impacts
- contribution to emissions, independent of vehicle type and power (ie, fossil fuel vs electric)
- concentrations increasing relative to the total motor-vehicle PM emissions burden, as engine technology improves and consumer uptake of hybrid and all-electric vehicles increases.

Additionally, NEEs are largely unregulated compared to tailpipe emissions, which are subject to emission control regulations in many jurisdictions. For example, the Air Quality Expert Group (2019, p. 9) in the UK states that there is an 'immediate priority that NEE are recognised as a source of ambient concentrations of airborne PM, even for vehicles with zero exhaust emissions of particles'. However, the Committee on the Medical Effects of Air Pollutants (2020, p. 1) reported that 'the current body of published work is small and does not provide a compelling narrative of adverse health effects of exposure to non-exhaust particles from road transport', but suggested that 'as this component of traffic emissions will become proportionately more important in future years, it is recommended that new epidemiological and toxicological research should be undertaken to further understand the potential health risk of this aspect of vehicular pollution and to provide a basis for further policy'.

It is therefore important that we have a clear understanding of the New Zealand context regarding the composition of NEEs and the contributions they make to ambient PM loadings.

1.1.1 Sources and processes for non-exhaust emissions

NEEs result from a combination of mechanical abrasion sources (brake linings, tyre wear, road-surface wear) and as such are largely (but not entirely) in the coarse-particle ($PM_{10-2.5}$) and total-suspended-particle size bands typically used in monitoring and regulation. It is likely that the chemical composition of NEEs have changed over time as brake linings, motor-vehicle componentry, road surfaces and tyre composition have also changed.

In addition, PM from both anthropogenic sources (eg, vehicle tailpipe emissions and wear products, construction dust, tracked-out mud) and natural sources (eg, wind-blown dust and desiccated fluvial silt deposits) can be deposited on roadways and then resuspended by the turbulence generated by the passage of motor vehicles across that road surface. Rainfall is likely to wash such accumulated road-surface dusts into local stormwater systems and therefore may also change the relative mix of NEE components as they reaccumulate on the roadway again.

Altogether, NEEs represent a complex mixture of metallic, microplastic and mineral particles from a wide range of initial sources.

1.2 Part 1 – the literature review

In 2021, NIWA completed a comprehensive literature review on the NEEs being discharged to air and water (Semadeni-Davies et al., 2021). This included a detailed description of the various methods used to measure NEEs in both laboratory and field settings, NEE composition and changes over time (including New Zealand relevant data), the range of values reported for NEEs in the literature and a discussion of the challenges in synthesising such a range of testing into a consistent dataset for use.

This literature review represents part 1 of the current research.

A gap analysis completed as part of the review highlighted that most of the knowledge on NEEs comes from research that was undertaken in the late 1990s and early- to mid-2000s, and that a lack of standard methods to determine emission factors, combined with the range of materials used in manufacturing brakes, tyres and road surfaces, leads to a large variability in estimates of emission factors.

The review also looked at the role of the changing vehicle fleet, in particular the introduction of electric vehicles, and how that is changing the relative contributions of different sources of NEEs, and the relative importance of NEEs as a pollutant source compared with vehicle exhaust emissions.

1.3 Part 1 – recommendations

The gap analysis concluded that although there is wide uncertainty around NEEs, there is also uncertainty around the impact that NEEs have in the tools and on the outcomes for which emission factors are commonly used in New Zealand. Thus, the review recommended further work to get an understanding of how the choice of emission factors affects model outputs.

As the emission factors being used in modelling are all derived from work overseas, the review also recommended a further review and reanalysis of the current datasets held in New Zealand that might be used to calculate emission factors, in particular, the large archive of elemental analyses held by the Institute of Geological and Nuclear Sciences (GNS Science) from various measurement campaigns around the country.

Based on the outcomes of these analyses, the NZTA could decide to either retain the current emission factors or undertake new monitoring and modelling to develop and test a new set of emission factors for New Zealand.

1.4 Purpose and scope of part 2

This report sets out to fulfil the initial recommendations of the literature review in part 1, namely to:

1. survey existing source-apportionment data to determine what detailed analysis is possible that will help to ground-truth current NEE factors, with the proposed analysis to inform the work for stage 2 of the research
2. determine the magnitude of the impact that NEE factors have in air-quality modelling by undertaking a sensitivity analysis
3. determine the need for any further monitoring and, if there is a need, to describe what kind of monitoring would need to be undertaken.

Note that this work considers paved or sealed roads. Unpaved roads are not included.

2 Ground-truthing current non-exhaust emission factors

This section examines whether existing data from roadside air-quality measurements contain sufficient information to improve the accuracy of current NEE estimates and hence of the emission factors currently in use. The intention is not to produce new or improved emission factors but to suggest methods for developing them should there be confidence that there is enough suitable data.

2.1 Current non-exhaust emission factors

The most recent version of the Vehicle Emissions Prediction Model (VEPM) includes a crude emission model to estimate emission factors (Waka Kotahi NZ Transport Agency, n.d.). The factors used in the VEPM are currently based on emissions factors published by the European Environment Agency for PM from brake and tyre wear independently over three classes of vehicle (light, medium and heavy duty) and with some modification by vehicle speed. The user is provided with a single combined emission factor for the fleet mix and speeds specified. Year is not an input variable, so the model predicts no change in emission factors (independent of changes in fleet mix and speed) over time.

The most recent version of the VEPM does not include emission factors for road abrasion and resuspended road dust.

2.2 Source-apportionment data

2.2.1 Methods

2.2.1.1 Nature and coverage of source-apportionment data

NEEs cannot be measured directly. There is no measurement device that will clearly distinguish between PM that derives from NEEs relative to other emission or natural sources. Indirect methods have therefore been developed, where the contributions of different sources or source processes to samples of PM can be calculated based on statistical analysis of the composition of those samples.

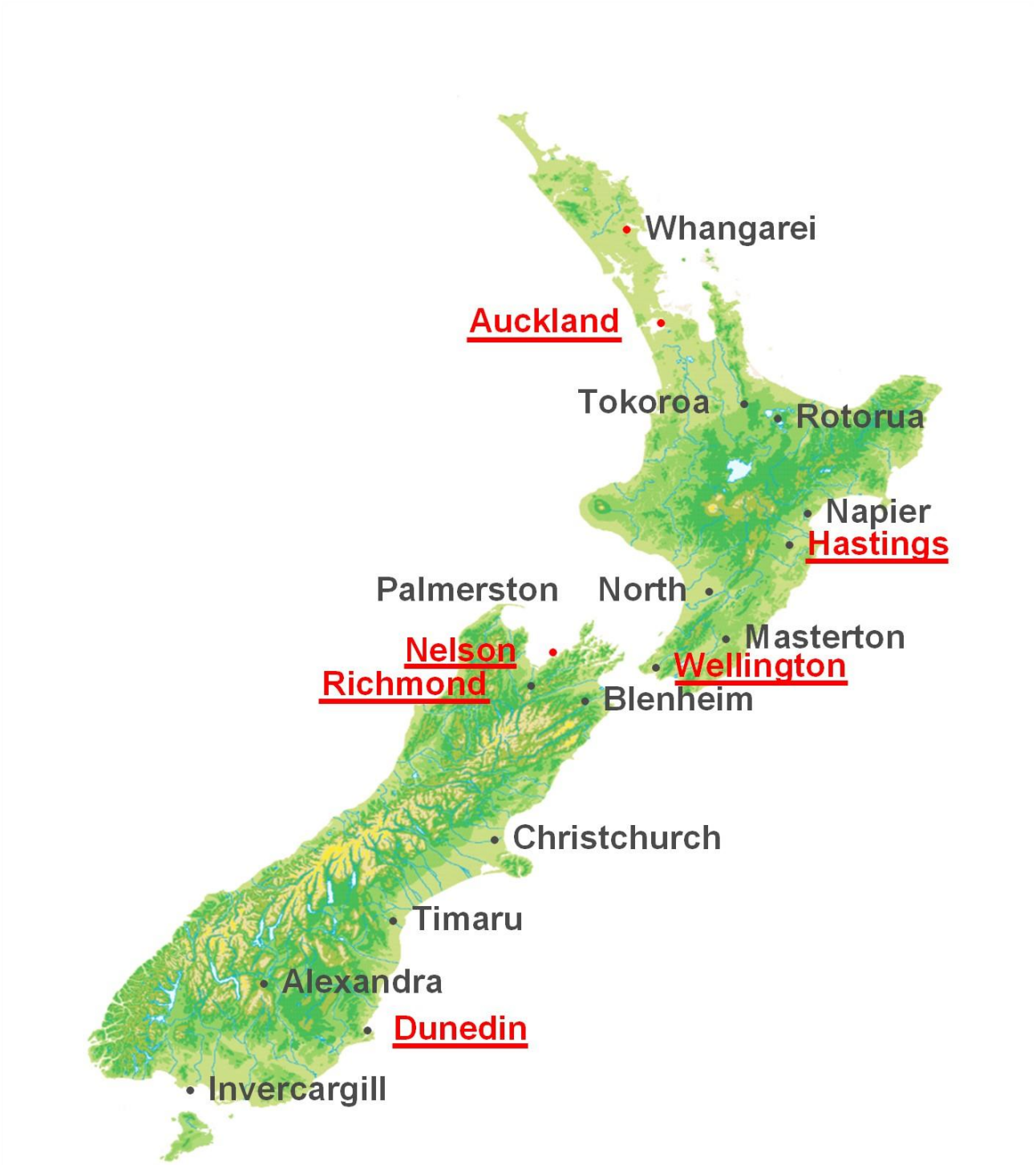
GNS Science has been researching the composition and sources of airborne PM in New Zealand (and overseas) for more than 20 years. Drivers for this research include understanding human health effects (particle size and composition), managing air quality (sources and source contributions to total PM), monitoring changes over time (trends and step changes) to contribute to policy evaluation, measuring the effectiveness of regulation, and understanding the impact of technology.

GNS Science holds an archive of filter-based, time-integrated PM samples that have firstly been analysed gravimetrically to provide ambient PM concentrations, then by appropriate analytical techniques to provide elemental composition data (Trompetter et al., 2005; Barry et al., 2012; Trompetter et al., 2014). The PM compositional data and receptor modelling (described below) datasets are contained in the National Air Particulate Speciation Database (GNS Science, 2020).

PM samples have been collected from approximately 40 regulatory authority monitoring sites across New Zealand (Figure 2.1) using National Environmental Standard-compliant methodologies or alongside National Environmental Standard-compliant PM monitoring systems, with some urban areas including multiple sites (most notably Auckland) (Davy & Trompetter, 2019). In addition to the urban monitoring locations, several studies have targeted source-specific PM composition (including in motor-vehicle tunnels) in order to better understand emission-source characteristics and composition (Ancelet et al., 2011; Davy et al., 2011).

Appendix A provides a summary table of monitoring sites, size fraction collected and the time-period of monitoring.

Figure 2.1 Air-quality monitoring sites for source apportionment



Note: Underlined in red = currently active sites.

Two multi-elemental analysis techniques have been used routinely to provide the PM composition data – accelerator-based ion beam analysis and X-ray fluorescence analysis – while light reflectance has been used to determine black-carbon concentrations in all samples. These are well established and internationally accepted methods for determining PM elemental composition (Hyslop et al., 2019; Yatkin et al., 2020).

The multivariate analysis of PM sample composition (also known as receptor modelling or source apportionment) provides groupings (or factors) of elements that vary together over time. This technique effectively 'fingerprints' the sources that are contributing to airborne PM concentrations and the mass of each element (including black carbon) attributed to that source (Hopke, 1999). A direct result of using receptor modelling techniques is that the sources of PM and the component elements are derived, along with the mass contribution that each emission source makes to atmospheric particle concentrations. The advantage of receptor models is that they allow the total (measured) PM concentrations to be apportioned.

A key result of the compositional analysis of airborne PM and source-apportionment process is that the chemical profile for each source identified is extracted from the data. The source chemical profile, therefore, is a reflection of the analytical process that produced the data in the first place.

The chemical profiles derived from receptor modelling are the average contribution for each species over the time-period of the dataset. The following sections discuss the elemental components that have been associated with motor-vehicle sources from receptor modelling and that can be attributed to NEEs.

For this work, the PM analytical and receptor modelling datasets were data-mined in the context of the composition and mass apportionment of NEEs associated with motor vehicles to summarise the current state of New Zealand specific information. The same data has been used previously to quantify source contributions to ambient concentrations of PM for exposure assessment in the Health and Air Pollution in New Zealand Study (HAPINZ 3.0) (Kuschel et al., 2022).

2.2.1.2 Chemical profiles associated with motor-vehicle sources

Each air-quality monitoring site demonstrates varying contributions from motor-vehicle sources to total measured concentrations, depending on: the proximity of the sampling site to the nearest roadway (coarse PM deposits more rapidly due to gravitational settling); the local traffic volume; vehicle characteristics (size, weight, speed, number of wheels); road-surface type; and any local dust generating activities that also deposit material onto the roadway (eg, road works or construction activities).

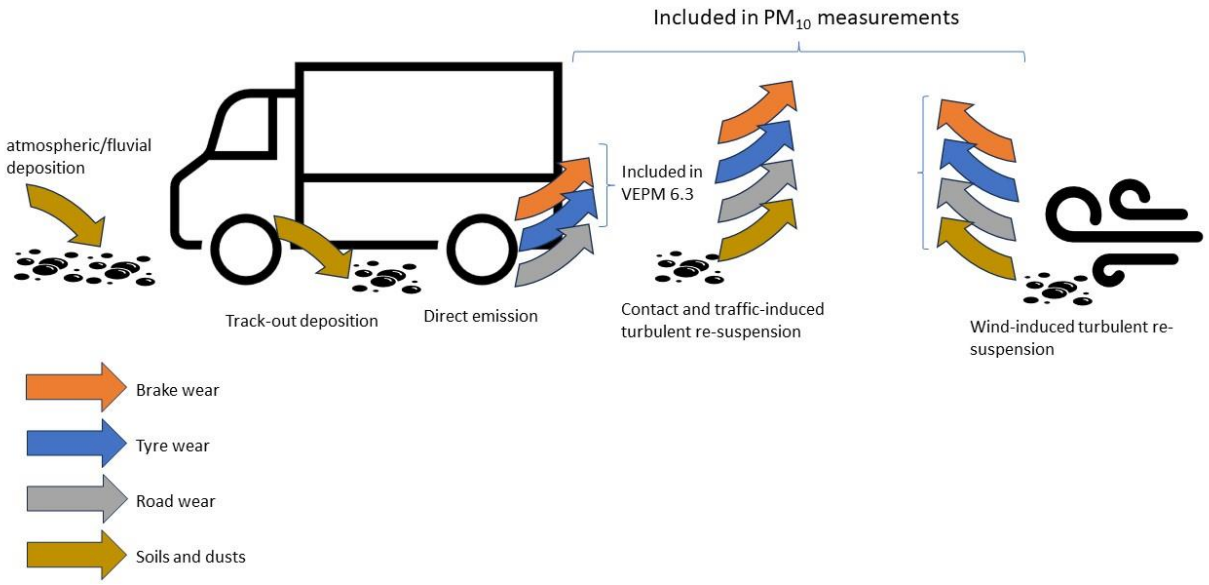
Analysis of the source-apportionment database shows that all urban monitoring sites have an associated motor-vehicle contribution to total PM. The disaggregation of the motor-vehicle contribution to fuel-type (petrol or diesel) exhaust emissions and the non-exhaust component is a function of the PM size fractions monitored, frequency of sampling, length of monitoring and the factors described above. It is clear that all motor-vehicle chemical profiles from PM_{10} and $PM_{2.5}$ compositional data include a tailpipe and NEE component, whereas a coarse fraction ($PM_{10–2.5}$) motor-vehicle contribution only includes the NEEs. Apart from a ubiquitous crustal matter component (Al, Si, Ca, Fe) that is resuspended to atmosphere with the passage of every vehicle along a roadway, copper and zinc are always associated with NEEs, with copper being the primary marker for brake wear. Zinc is present in tyres, but is also a component of tailpipe emissions as it is used in engine lubricants that can get combusted along with fuel.

Table 2.1 shows 'motor-vehicle' source-apportionment mass contributions to PM_{10}, $PM_{2.5}$ and coarse fraction ($PM_{10–2.5}$) at measured locations around the country. For simplicity, it is usually assumed that all coarse motor-vehicle-related PM is NEEs. However, NEEs also contribute a measurable amount to the $PM_{2.5}$ fraction, and while significantly lower than for PM_{10} or the coarse fraction, should ideally be factored into any assessment of the impact of NEE-related PM or in producing emission factors.

From places where there is sufficient data (ie, Auckland and Wellington), it is clear that NEEs are now the dominant source of motor-vehicle-related PM near busy roads, and that this will be the case at more and more locations as technological advances improve engine combustion emissions and electrification of the vehicle fleet occurs.

An important factor to note with the receptor modelling process is that being an analysis of covariance, NEE contributions are primarily included in a single 'motor-vehicle' source, along with tailpipe emissions contributions, as the two sources are well mixed by the turbulent passage of motor vehicles across the road surface (see Figure 2.2). Separation of the two motor-vehicle source types is possible when coarse and fine fractions are collected at a site or by calculating the difference between contributions to $PM_{2.5}$ and PM_{10} if they happen to be collected simultaneously at a monitoring site. However, the covariance problem still exists for the NEE increment associated with $PM_{2.5}$. It is evident from the motor-vehicle source chemical-profile data (see Figure 2.1) that the Al:Si ratio is typical for a crustal matter component (Lide, 1992) and both elements are present in the $PM_{2.5}$ fraction.

Figure 2.2 Schematic showing the physical processes contributing to non-exhaust emissions



2.2.2 Results

2.2.2.1 Average contribution of coarse particles associated with motor vehicles to PM from source-apportionment studies

A simple estimate of the maximum contribution of NEEs to PM_{10} can be calculated if it is assumed that all coarse PM associated with motor vehicles is NEEs (although, as noted in section 2.2.1.2, this does not include the NEE in the $PM_{2.5}$ fraction). This can be calculated from the GNS Science data where a motor-vehicle component for PM_{10} and $PM_{2.5}$ has been resolved at the same monitoring site over the same time period, by subtracting the $PM_{2.5}$ component from the PM_{10} component. This yields estimates, as presented in Table 2.1, in the range of 0.1–2.3 $\mu g\ m^{-3}$, with two outliers: 3.3 $\mu g\ m^{-3}$ at Richmond and 7.4 mg m-3 at Dunedin (see Appendix A for further details). These estimates are based on averages over time periods

varying from months to over a decade, during which time the values may have changed. Therefore, they should be taken as indicative of the order of magnitudes rather than being considered precise. The high value for Dunedin is believed to relate to the nearby construction of the Forsyth-Barr Stadium at the time of the monitoring. This suggests additional resuspension of mineral dust, but potentially additional brake, tyre and road wear from construction vehicles too.

Table 2.1 Source-apportionment mass contributions from PM_{10}, $PM_{2.5}$ and coarse fraction ($PM_{10–2.5}$)

GNS Science site name	Source-apportionment mass contributions		
	PM_{10} motor vehicles[a]	$PM_{2.5}$ motor vehicles	$PM_{10-2.5}$ motor vehicles
Masterton	0.98	0.40	0.59
Upper Hutt	1.48	1.36	0.12
Wainuiomata	1.63	0.79	0.84
Seaview	2.22	0.57	1.65
Kingsland	3.17	1.92	1.25
Takapuna	3.59	1.82	1.77
Queen Street	6.41	4.23	2.19
Penrose	4.11	2.34	1.77
Khyber Pass Road	6.18	4.05	2.14
Nelson City	2.02	0.78	1.25
Dunedin	10.2	2.84	7.4[b]
Woolston	4.89	2.59	2.30
Christchurch (Coles Place)	2.19	1.01	1.18
Awatoto	0.87	0.20	0.67
Marewa Park	1.92	0.15	1.77
Tokoroa	1.51	0.67	0.83
Richmond	3.64	0.38	3.26
Willis Street	1.5	0.9	0.6

Note: Values are derived from time series of variable length.
[a] The coarse component of motor-vehicle PM_{10} gives an indication of NEEs based on the assumption that all coarse PM is NEEs.
[b] The construction of the Forsyth-Bar Stadium near the monitoring site was occurring during the monitoring period. Such activities and the associated truck movements can have a significant effect on the NEE component.

2.2.2.2 Average contribution of non-exhaust emissions to particulate matter from source-apportionment studies

A contribution to PM_{10}, which was explicitly attributed to NEEs, was resolved by GNS Science for one tunnel and five ambient sites, yielding estimates from 0.8–2.8 μg m^{-3} for the latter (see Table 2.2).

Table 2.2 Non-exhaust emissions contribution to PM_{10} and coarse PM

Site name	NEE contribution to PM_{10} (µg m^{-3})	NEE contribution to $PM_{10\text{-}2.5}$ (µg m^{-3})
Johnstone's Hill Tunnel	3.5	
Queen Street	2.8	2.0
Willis Street		1.8
Penrose	1.7	
Khyber Pass Road	1.4	
Wainuiomata	0.8	0.6

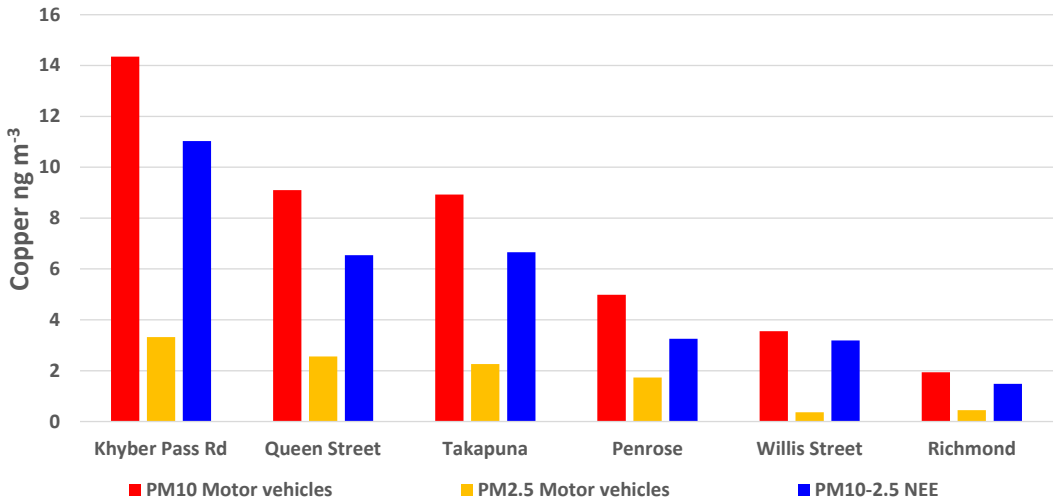
2.2.2.3 Source apportionment: Road dust, brake and tyre wear

No previous source-apportionment analysis based on New Zealand data has yet been able to resolve a source contribution from brake and/or tyre wear independently of other NEE sources. This is due to the high degree of temporal correlation (covariance) between these two source contributions and their relatively low concentrations.

In this section, the PM_{10} motor-vehicle source chemical profiles from source-apportionment sites were reanalysed and combined to examine the relationships between known NEE components (as described in Section 2.2.1.2), namely crustal matter components representing road-surface wear, copper from brake wear and zinc as representing tyre wear. Note that for this analysis, only monitoring sites where all NEE indicator chemical species (Al, Si, Cu, Zn) were available from the receptor modelling have been included.

Figure 2.3 presents the copper (Cu) concentrations in PM_{10}, $PM_{2.5}$ and $PM_{10\text{-}2.5}$ motor-vehicle source chemical profiles at various monitoring locations, all of which are adjacent to roadways (Khyber Pass Road, Queen Street, Takapuna and Penrose are all in Auckland, Willis Street is in the Wellington CBD, and Richmond is in Tasman District). Figure 2.4 shows that the highest Cu concentrations are associated with higher local-traffic density. As the traffic density decreases (or a monitoring site is further away from busy roads), then the associated Cu concentration also decreases, to the point that it may be near or below the analytical detection limit and not included in the receptor-modelling analysis.

Figure 2.3 Copper concentrations in motor-vehicle-related particulate matter size fractions at various locations



Based on geochemical principles that account for the primary oxidation state of elemental species, the crustal matter mass associated with NEEs can be estimated from the following equation (Cahill et al., 1989; Cohen, 1999; Malm et al., 1994):

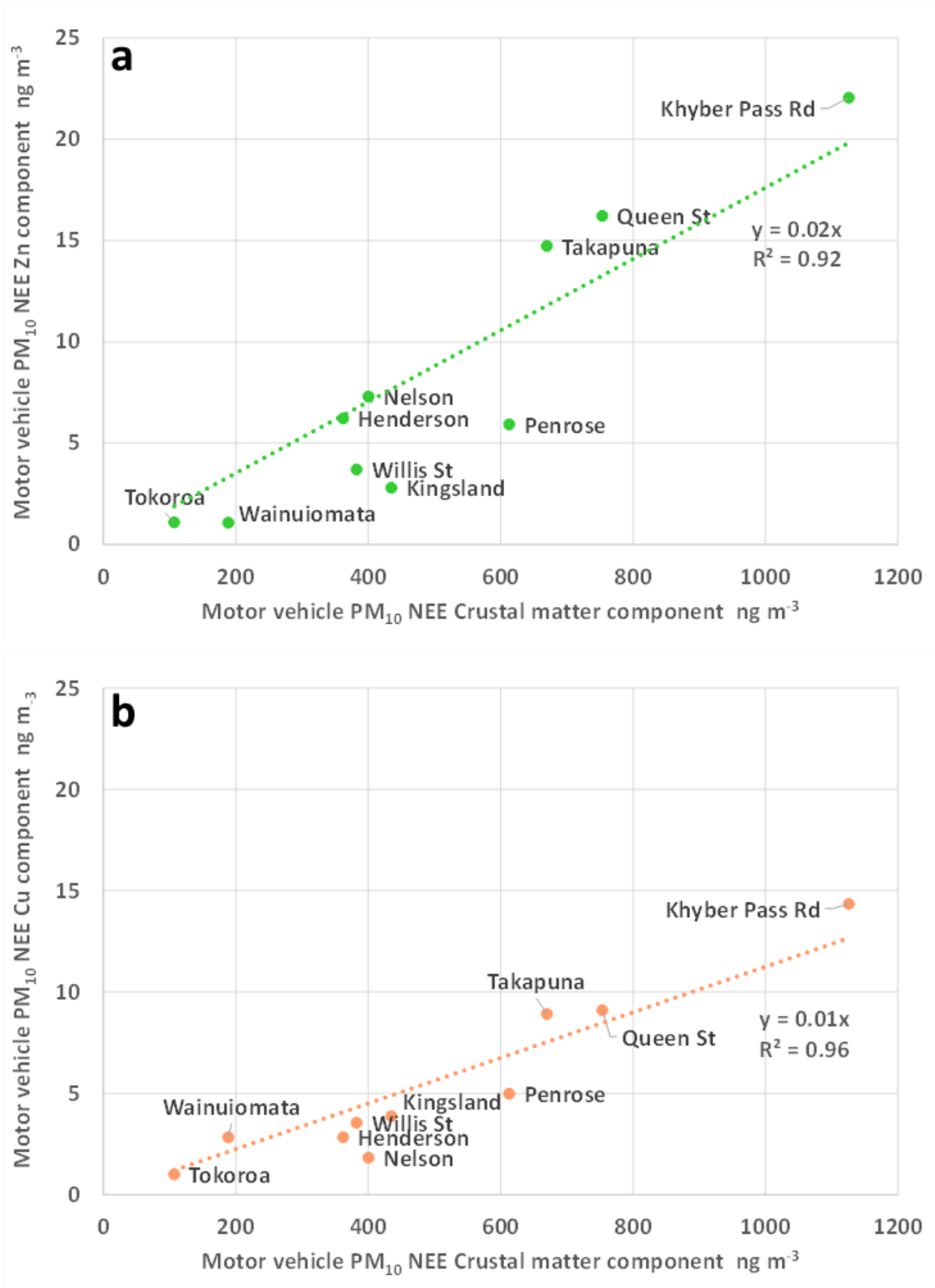
$$[Crustal\ matter] = 2.20[Al] + 2.49[Si] + 1.63[Ca] + 2.42[Fe] + 1.94[Ti] \qquad \text{(Equation 2.1)}$$

Equation 2.1 produces a reasonable estimation of the crustal matter contribution to NEEs, due to the largely inorganic nature of the material and that all of the major elemental components are measured in the samples.

Copper is assumed to be a clear marker of brake wear and is likely to be present in trace quantities (ie, as measured) along with other brake lining components, and zinc concentrations are likely to be proportional to the total mass of tyre wear particles.

Figure 2.4 presents the relationship between these three different NEE components, and shows that they are well correlated, demonstrating their covariant nature. The figure indicates that generally at any roadside location, NEE will be composed of the various sources in a well-mixed state. Location-specific deviations may occur depending on surrounding activities, the road-surface state and road-surface dust loading.

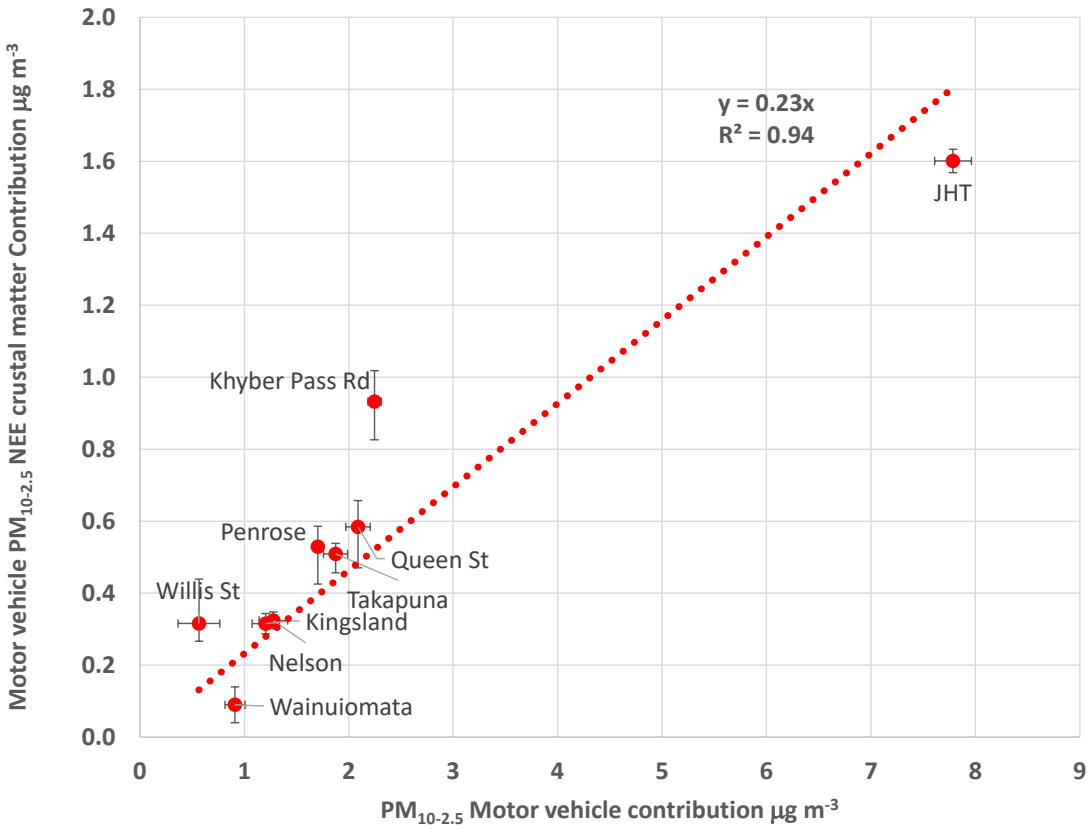
Figure 2.4 Relationship between motor-vehicle-related non-exhaust emissions particulate matter components at various locations: a (top) zinc vs crustal matter; and b (bottom) copper vs crustal matter



When the crustal matter component of NEEs was compared to the total motor $PM_{10\text{-}2.5}$ contribution (calculated as per Table 2.2), it was also well correlated, as shown in Figure 2.5. The data indicates that the

NEE crustal matter component is consistently around 20–25% of the $PM_{10-2.5}$ contribution across multiple sites.

Figure 2.5 Relationship between motor-vehicle-related $PM_{10-2.5}$ and the non-exhaust emissions crustal matter components at various locations



Note: Number of samples included in the analysis = 11,500.

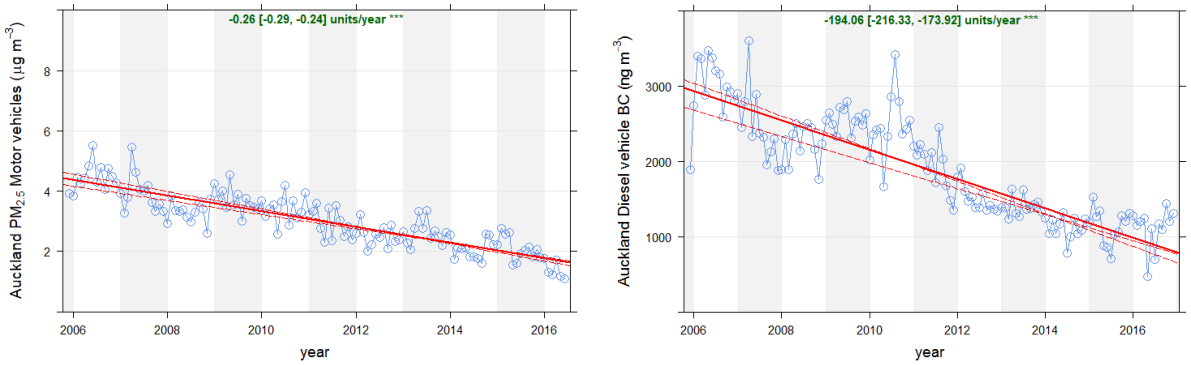
As described in Section 2.2.1.1, the PM elemental composition data from ion beam analysis or X-ray fluorescence analysis used for source apportionment primarily covers the inorganic components. It is clear from the data that the NEE components are covariant and correlate with the total motor-vehicle contribution. However, one of the components not measured is the organic carbonaceous material that would also be associated with NEEs. This includes road-surface wear (tar or bituminous material), organic binders and resins used in brake linings, along with the rubber and plastics that are the dominant components of tyres, all of which are described in detail in Semadeni-Davies et al. (2021). If it is assumed that crustal matter, brake wear and tyre wear are the main contributors to total NEEs, with crustal matter being 20–25% as estimated above, then that suggests that the organic carbonaceous component may be as much as 75% of the total $PM_{10-2.5}$ NEE contribution.

2.2.2.4 Long-term trends in non-exhaust emissions from source-apportionment data

The Auckland Council PM-speciation dataset is the longest continuous dataset available for trend analysis, with PM_{10} samples collected at three sites (Henderson, Takapuna and Queen Street) since 2006 (Davy & Trompetter, 2022). The long-term trend for motor-vehicle-related PM shows that there has been a decrease over time, largely driven by a fall in motor-vehicle-related $PM_{2.5}$ (tailpipe) concentrations, with a significant

drop in diesel-vehicle black-carbon emissions, as shown in Figure 2.6 (Davy & Trompetter, 2020; Davy & Trompetter, 2021).

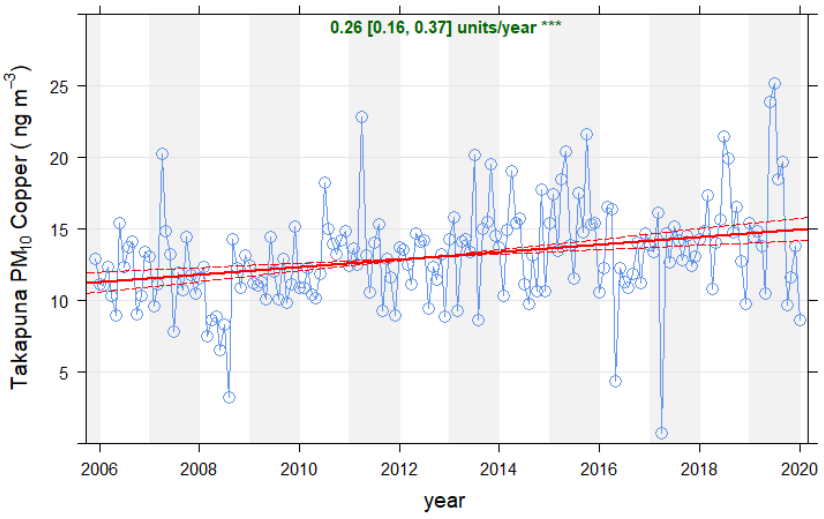
Figure 2.6 Long-term trend (de-seasonalised) in $PM_{2.5}$ motor-vehicle source contributions across all Auckland monitoring sites, showing that contributions from diesel vehicles have decreased



Note: Data is statistically significant at the 99.9% confidence interval.

The chemical data shows that copper concentrations (as an indicator of NEE contributions) have been increasing year-on-year at the Takapuna site (Figure 2.7), in line with increasing traffic volumes on roads (including State Highway (SH) 1) near the site (Davy & Trompetter, 2021).

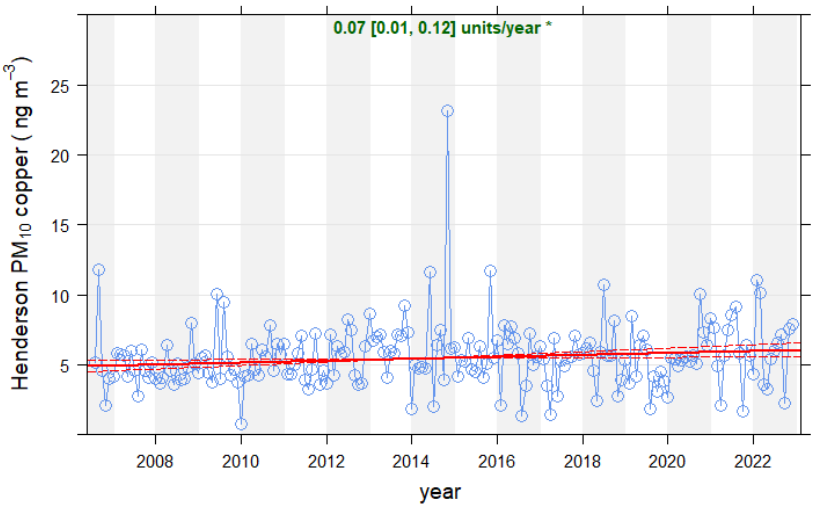
Figure 2.7 Long-term trend (de-seasonalised) showing the increase of elemental copper in PM_{10} at the Takapuna site



Note: Data is statistically significant at the 99.9% confidence interval.

Copper concentrations at the Henderson site (adjacent to Lincoln Road) are lower than at Takapuna due to lower traffic volumes, but also show a gradual increase since 2006, as presented in Figure 2.8.

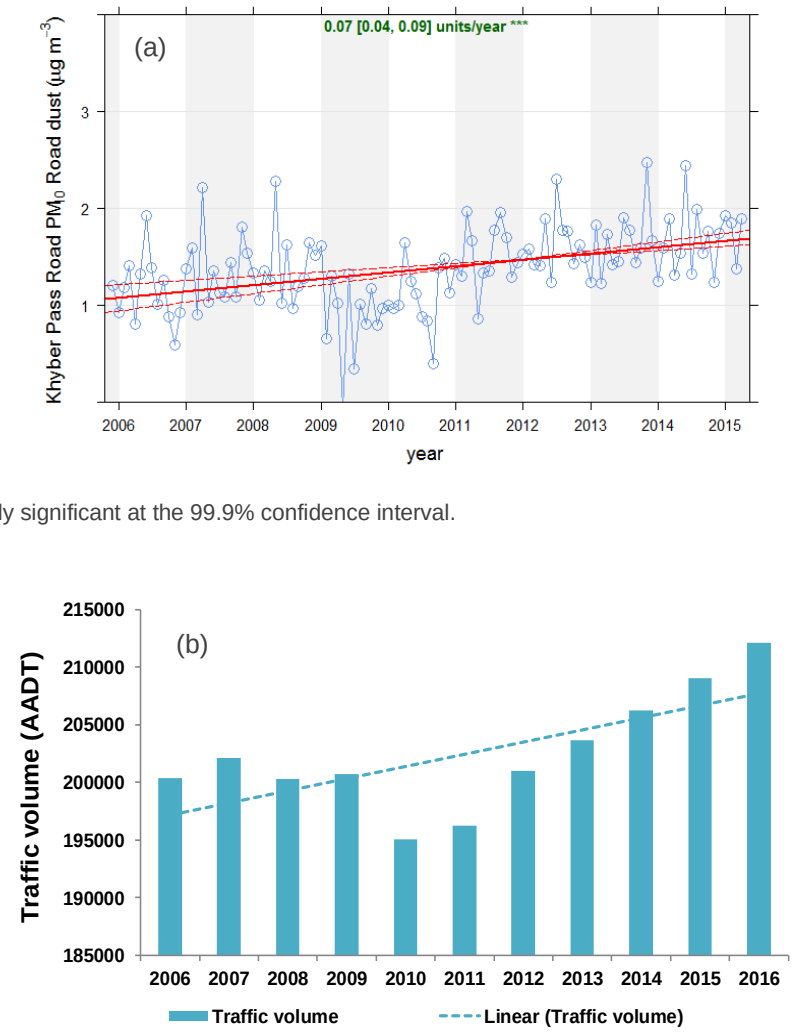
Figure 2.8 Long-term trend (de-seasonalised) showing the increase of elemental copper in PM_{10} at the Henderson site



Note: Data is statistically significant at the 99.9% confidence interval.

Source-apportionment data from the Khyber Pass Road site shows increasing NEE contributions to PM_{10} concentrations commensurate with the increasing traffic volumes on the adjacent motorway, as shown in Figure 2.9 (Davy & Trompetter, 2020). The NEE component tracks with a dip in traffic volumes during 2010 and 2011 (most likely related to traffic constrictions during the Newmarket viaduct reconstruction).

Figure 2.9 **(a) Long-term trend (de-seasonalised) for Khyber Pass Road PM_{10} non-exhaust emissions source contributions, showing that concentrations have increased, as have (b) traffic volumes on the adjacent motorway (Waka Kotahi NZ Transport Agency, 2019)**



Note: Data is statistically significant at the 99.9% confidence interval.

The Queen Street data reflects the changes in traffic regimes and bus routes in the street canyon over the years, and copper concentrations have generally declined at the monitoring site in line with decreased traffic volumes.

2.3 Comparing source-apportionment data with emission factors

2.3.1 Method

Source apportionment and emission factors represent different things (average concentrations and average emissions per vehicle, respectively), which are linked through site-specific relationships driven by dispersion conditions. We have considered two methods to allow for those conditions to enable valid comparisons. The simplest method is to use a third well-characterised pollutantsto confidently link emissions and concentrations at that site. Nitrogen oxides (NO_x) are ideal for this purpose as they are measured to a high degree of accuracy at most monitoring sites and are well-described by the VEPM. The alternative method is to use a dispersion model combined with the VEPM to simulate a time series of concentrations at the site.

Although yielding more detailed data, this method introduces several additional sources of error (background concentrations, traffic-fleet data, meteorological data and the dispersion model itself), as well as being significantly resource intensive. We therefore have relied on the simpler ratio method for this review.

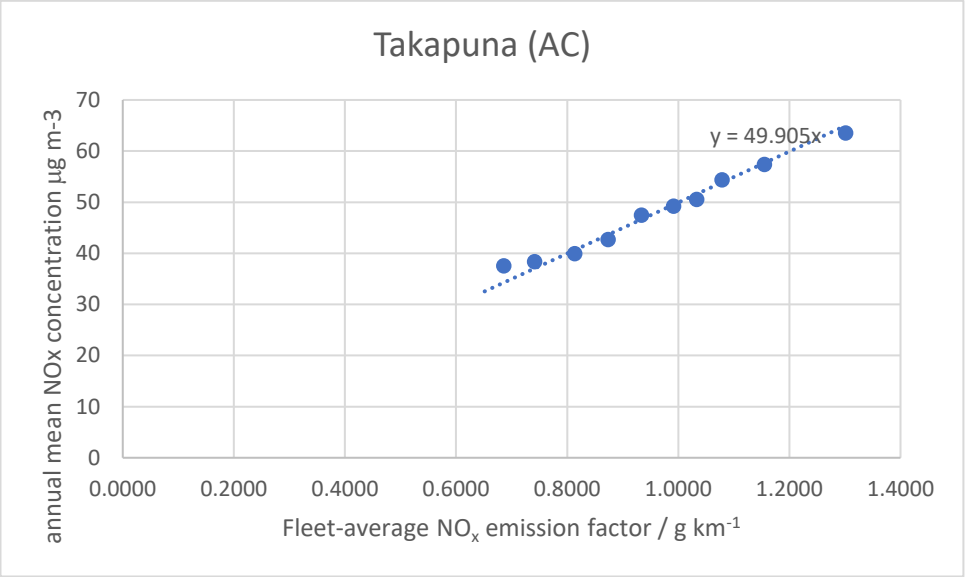
In this analysis we assume that the VEPM provides an accurate description of long-term trends in NO_x emissions. For the period 2006 to 2016, the VEPM predicts that (for a national average fleet composition) NO_x emissions fell from 1.30 g/km to 0.69 g/km, that is a fall of 47%. This assumption is supported by the observation that ambient concentrations of NO_x observed at regulatory monitoring sites at Takapuna and Penrose, both < 100m from busy Auckland motorways, fell by 49% and 47% respectively over the same period.

In each case, a site-specific 'dilution ratio' can be established by comparing the on-road emission factor and the near-road observed concentration. Although variable in the short term, this ratio should converge to a fixed value if averaged over the long term. This dilution ratio will be constant for all conserved pollutants. Brake-and-tyre-wear particles are typically in the size range 3–5 μm in diameter (eg, Oroumiyeh & Zhu, 2022), which have been shown to decay in concentration from a major road at approximately the same rate as tailpipe emissions (Zheng et al., 2022). This method can therefore be used to compare emission factors and observed concentrations for NEEs. Because NEEs cannot be observed directly, but need to be derived from source apportionment and receptor modelling (see above), this method involves some uncertainty.

2.3.2 Results: Ambient roadside sites

Making the simplifying assumption that traffic volumes and speeds have not changed significantly, we derived an approximate long-term average dilution ratio of 50 for both Takapuna and Penrose.[1] Figure 2.10 shows the relationship at Takapuna between the VEPM NO_x emission factor and the annual mean measured NO_x concentrations.

Figure 2.10 Fleet-average emission factor and annual mean measured concentration for NO_x at Takapuna, Auckland, 2006–2016



[1] Takapuna and Penrose were the only sites with long enough NO_2 and traffic time-series to derive dilution ratios.

This approach reveals that, according to the VEPM, brake-and-tyre-wear emissions should make a contribution to PM_{10} at both the Takapuna and Penrose sites, calculated thus (based on an average speed of 80 km/h and default fleet composition for the years 2006 to 2016, PM_{10}–$PM_{2.5}$):

VEPM brake & tyre wear emission factor x dilution ratio (Equation 2.2)

That is, 0.0059 g km^{-1} x 50 = 0.29 μg m^{-3}.

If there is no change in vehicle numbers, the VEPM predicts that this contribution should not have changed significantly over the 2006 to 2016 period.

This estimate cannot be directly compared to observations, as brake and tyre PM is not measured directly, nor has it currently been resolved by receptor modelling. What has been resolved, however, is:

- the total motor-vehicle contribution to PM_{10} and $PM_{2.5}$, from which the total motor-vehicle contribution to coarse PM can be calculated
- the average contribution of crustal matter, which can be calculated.

These values are 1.7 – 1.8 μg m^{-3} and 0.6 – 0.7 μg m^{-3}, respectively. Compared to the VEPM-predicted brake and tyre contribution of 0.29 μg m^{-3}, it appears that 1.4–1.5 μg m^{-3} is unaccounted for by the emission factor. Noting that the VEPM emission factor is only for brake and tyre wear, this indicates that (assuming the brake-and-tyre-wear emission factor is approximately correct) the unaccounted PM is related to other emission processes or sources (mineral or organic road dust or vehicle wear products other than brakes and tyres).

At Coles Place in Christchurch, uncertainty over vehicle speeds and fleet mix (the site is well setback from major roads) also introduces considerable uncertainty, as does the dataset being of only 2 years' duration. However, a very rough estimate indicates a predicted brake-and-tyre-wear contribution in the order of 0.2 μg m^{-3}, compared to the resolved coarse motor-vehicle component of 1.2 μg m^{-3} from source apportionment; again suggesting a missing ~ 1.0 mg m^{-3}, which we may speculate is from the crustal-matter component.

2.3.3 Results: Johnstone's Hill Tunnel

Three-hourly time-integrated particulate filter samples were collected from inside the Johnstone's Hill Tunnel during June 2010.

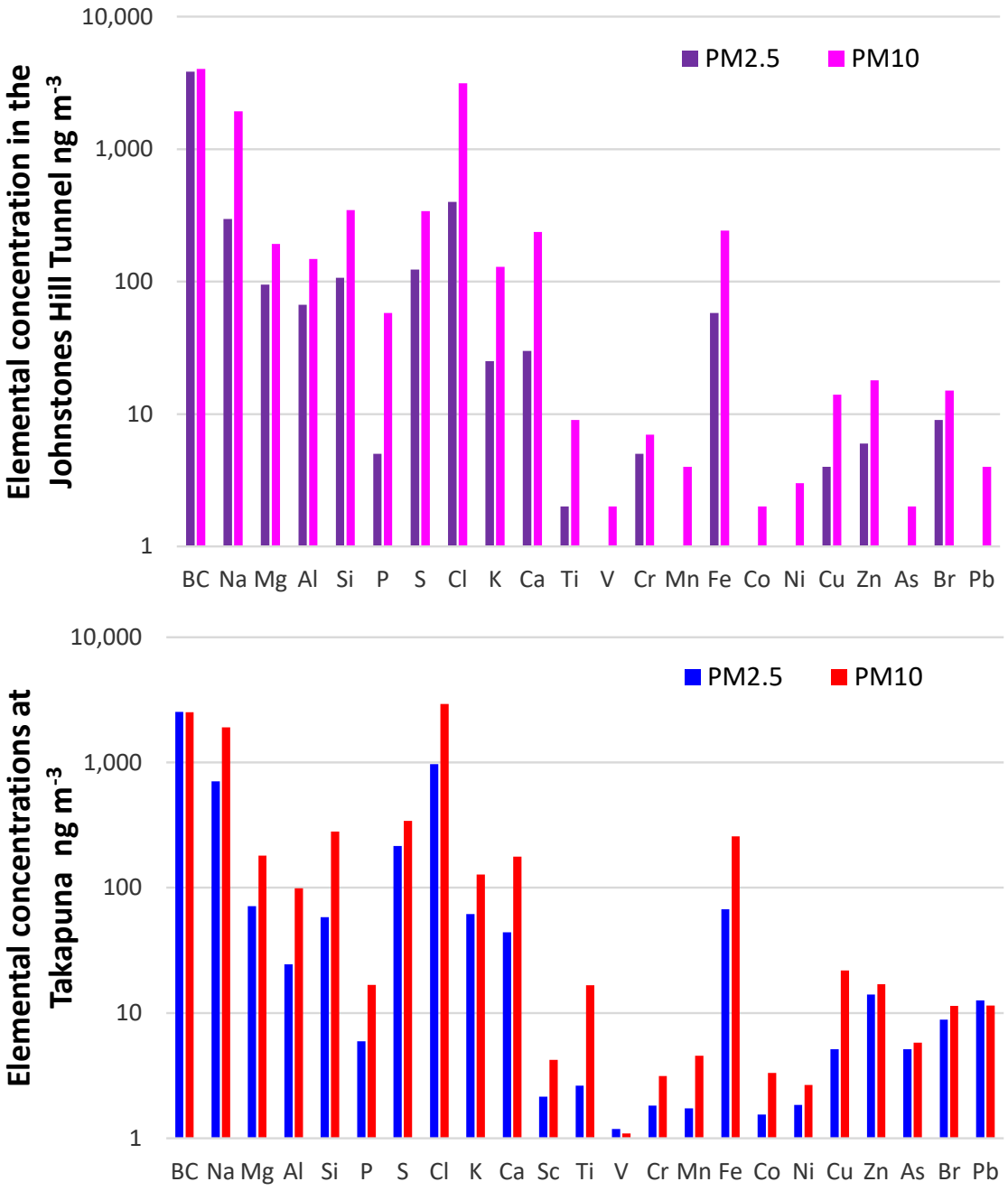
The Johnstone's Hill Tunnel is a twin tunnel system on SH1 near Orewa, north of Auckland City, that was opened in January 2009. The tunnels are two identical semi-circular uni-directional tubes, approximately 12 m wide, 9 m high and 380 m long. They were built to carry two lanes each, plus a shoulder and an emergency pathway. At the time of the study, the southbound tunnel had two lanes open but the northbound tunnel had only one lane open, due to the merging of the traffic into a single lane immediately after the tunnel. The annual average daily traffic for the Johnstone's Hill Tunnel in 2009 was estimated to be approximately 12,750 vehicles per day, with the proportion of heavy commercial vehicles estimated at 10.2%.

PM samples for elemental determination were collected using a Streaker sampling system, which is a rotating filter sampler designed to collect high temporal resolution $PM_{2.5}$ and $PM_{10\text{-}2.5}$ samples (Davy et al., 2011).

As with data collected at the Mt Victoria Tunnel, the coarse fraction PM in Johnstone's Hill Tunnel dominates the elemental concentrations except for black carbon (Figure 2.11). Receptor modelling of the Johnstone's Hill Tunnel data indicated that of the motor-vehicle PM sources identified, a coarse fraction road-dust source was the primary contributor to PM_{10}. However, the data also shows that the chemical species that are associated with NEE make a measurable contribution to the fine-fraction PM associated with motor vehicles.

However, neither a brake nor tyre wear component were separately resolved for the tunnel, but would have been included in the general motor-vehicle source contributions.

Figure 2.11 Elemental concentrations in $PM_{2.5}$ and PM_{10} collected in Johnstone's Hill Tunnel during 2010 (above), compared with analysis from the Takapuna ambient monitoring site, 2006–2016 (below)



Because only a single month of data was collected, it is not possible to use long-term trends in emission factors to derive the tunnel's dilution ratio.

The main source of variability in emissions was diurnal variation in traffic volumes. In this study, seven samples per day were collected covering that diurnal variation. Unfortunately, the tunnel ventilation jet fans were not operating during the observational study, meaning that air flow in the tunnel was variable being

typically higher during the day due to traffic-induced turbulence and the ‘piston effect’, and lower and less stable at night. This prevents us from determining an average dilution ratio without significant uncertainty.

From reviewing this dataset we found that:

- the ratio of observed NO_x concentrations to sampled PM concentrations attributed to road dust was similar to the ratio between NO_x and brake-and-tyre-wear emissions derived from VEPM 6.3 (~80)
- the ratio of resolved road dust to observed NO_x was higher overnight when in-tunnel airflow was generally lower
- the ratio of copper concentrations to NO_x (or NO_2) concentrations was much lower than at ambient sites.

Together these observations could imply that mineral and/or resuspended road dust was a much lower contributor to PM levels (relative to traffic volumes) than at ambient roadside sites, with direct emission of brake-and-tyre-wear particles probably being the dominant source of non-exhaust coarse-particle emissions. However, it should also be noted that the Johnstone’s Hill Tunnel had only been open for around 18 months when the sampling was conducted.

Nevertheless, these results indicate how a future tunnel-based study has the potential to distinguish brake and tyre wear from other road-dust emissions, so long as the air flow is constrained and consistent as much as possible, and concentrations are sufficiently high to resolve the desired components.

3 The impact non-exhaust emission factors have in air-quality modelling

This section discusses how much potential errors in NEE emission factors could impact air-quality modelling and health assessments.

3.1 Sensitivity of dispersion modelling to non-exhaust emissions emission factors

We used the AusRoads dispersion model to investigate the sensitivity of model output to changes in a range of input parameters (emission factors, traffic, meteorology and background concentrations). The sensitivity analysis concluded that uncertainty in background concentrations had the largest impact on results and that any single road source is a minor contributor to the total PM measured at a site, even at close distances. Uncertainties in measured or estimated background concentrations can mask the impact of almost any other reasonable changes in input data. Increasing the NEE to account for a contribution from road dust had the most substantial impact on concentrations generated by the road.

Detailed results of the sensitivity analysis are given in Appendix B.

3.2 Representativeness and national applicability

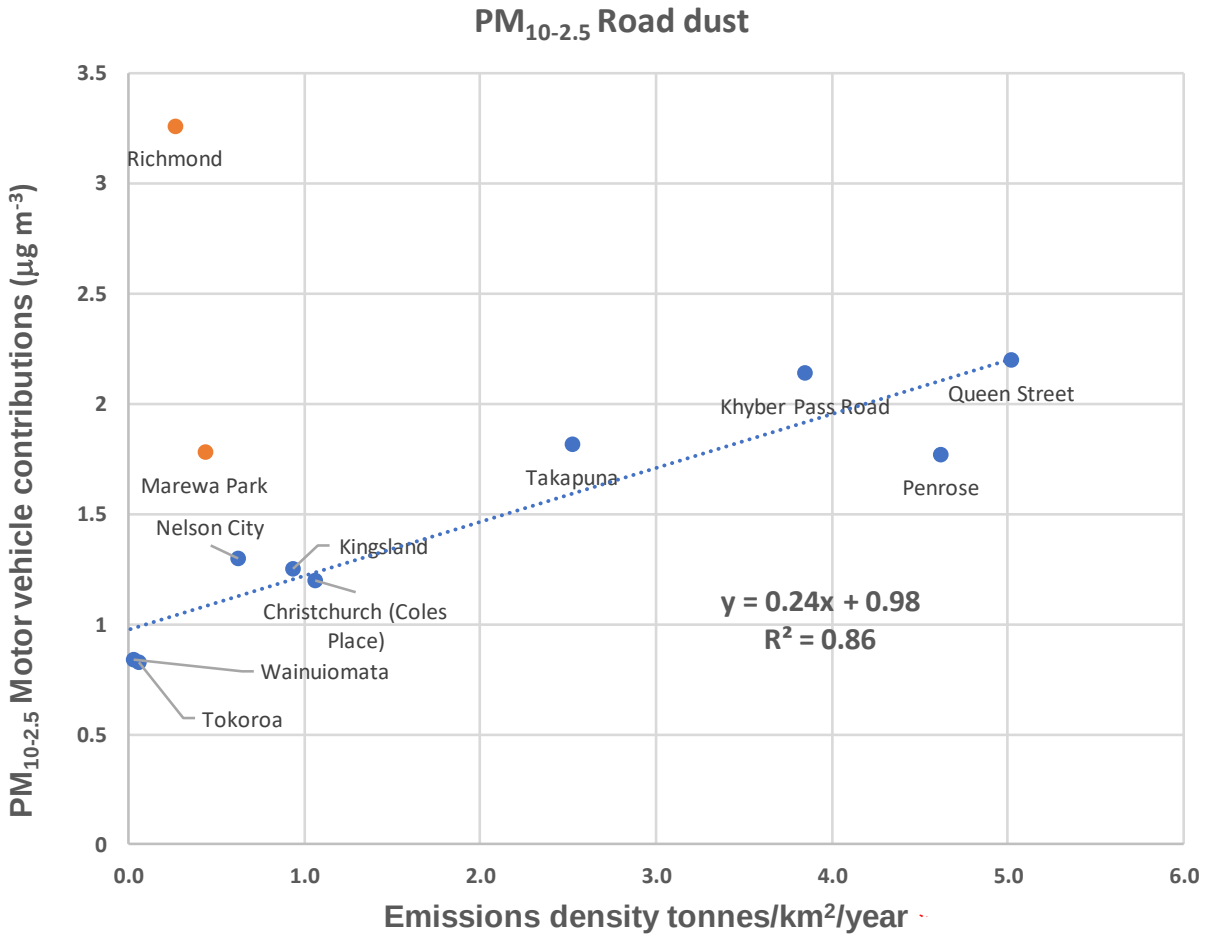
From the previous discussion, we infer that the contribution at most urban receptors is well below 1 $\mu g\ m^{-3}$. We have also shown above how this impact is most likely under-predicted if the VEPM is used, as the model only includes brake-and-tyre-wear factors for NEE and not road abrasion and resuspended road dust.

Rather than using the VEPM and a dispersion model, however, other simpler methods exist for estimating NEE contributions at unmonitored locations. For example, the HAPINZ 3.0 exposure model estimates NEE by applying a linear equation that empirically links NEE PM_{10} in a census area unit to estimated vehicle-emission density within that unit – a relationship calibrated using GNS Science source-apportionment data for the few sites where it is available.

The chemical composition and source-apportionment data indicate that ambient concentrations of PM NEE are closely linked with local traffic density as would be expected. While many factors (eg, road-surface type, vehicle weight, speeds, number of wheels) may affect the amount of NEE generated from a particular road or roading network, metrics that reflect local traffic density such as vehicle-kilometres travelled, vehicle counts or related derivations of motor-vehicle activity are likely to be linearly related to NEE concentrations. Analysis of source-apportionment data for the HAPINZ 3.0 study found that the PM_{10} and $PM_{2.5}$ source contributions data compared favourably with emissions density data from the National Emissions Inventory to the extent that it was used to provide motor-vehicle contributions for health impacts analysis (Kuschel et al., 2022).

Figure 3.1 presents the coarse fraction motor-vehicle contribution for various sites around New Zealand compared to the same emissions density data, showing a good correlation ($R^2 = 0.86$), except for two sites (Richmond in Tasman District and Marewa Park in Napier) that lie significantly outside the linear relationship. It is likely that there may be a mismatch for those sites, since the emissions density data was for 2013, whereas many of the source-apportionment studies were for other time periods or across multiple years. More direct metrics of local vehicle activity at the time of the sample collection may improve the relationship for those sites.

Figure 3.1 $PM_{10–2.5}$ motor-vehicle source contributions compared to traffic emissions density (adapted from National Emissions Inventory data, 2013)



According to HAPINZ 3.0, motor vehicles contribute 0.4–1.2 µg m^{-3} to annual $PM_{2.5}$ in most census area units and 1.5–2.3 µg m^{-3} of annual PM_{10}.

This method is subject to a few error sources, including:

- the monitoring site where the calibration data was collected may not be representative of the census area unit in which it is sited (this is increasingly likely for more road-influenced sites)
- emission density is only one factor influencing concentration and doesn't take into account variability in dispersion, topography etc.

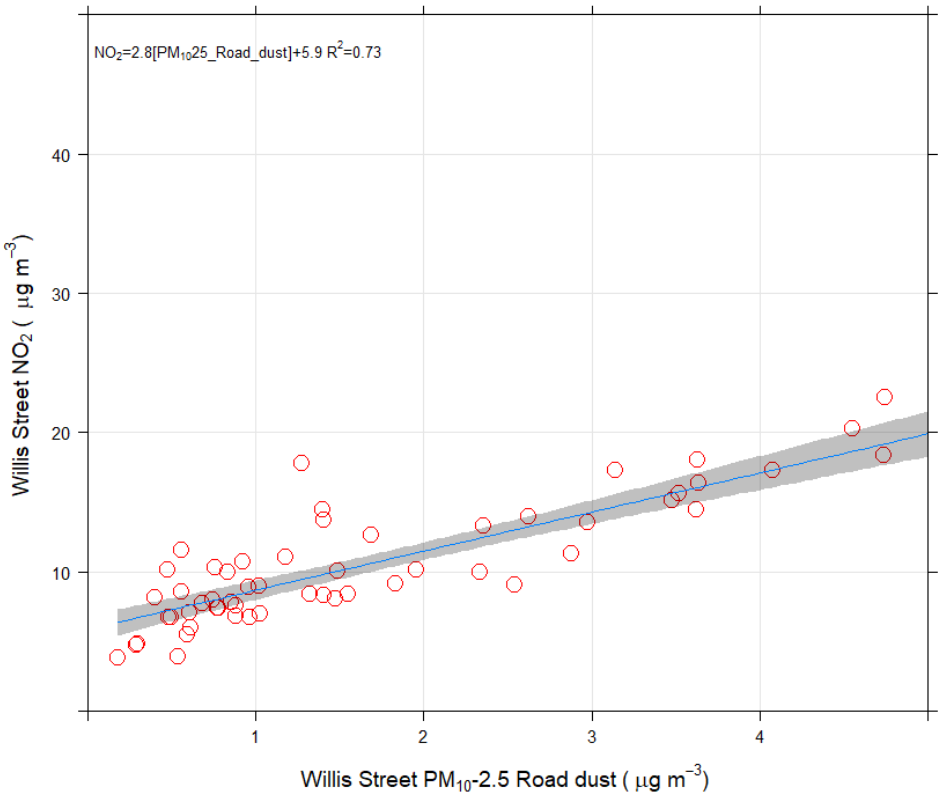
The likely implications of these uncertainties is a 'regression to the null' effect, where the model predicts a narrower range of concentrations than actually exists, especially at the lower end (eg, 'real' NEE contributions can be lower than the 1 mg m^{-3} minimum implied by the best fit line in Figure 3.1).

This is potentially remedied by replacing vehicle (PM) emission density with a more accurate, responsive and easily updated measurement.

NO_2 is a strong predictor of motor-vehicle emissions which is also readily assessable and updatable for any location in the country. For example, NO_2 monitoring data at the Willis Street site in central Wellington shows

a reasonable correlation (Figure 3.2) with the road-dust component, since the ambient NO_2 concentrations reflect local (diesel vehicle) tailpipe emissions that are covariant with NEE.

Figure 3.2 $PM_{10-2.5}$ road-dust (non-exhaust emissions) source contributions compared to NO_2 concentrations at the Greater Wellington Regional Council's Willis Street site (Wellington)



Furthermore, fairly accurate estimates of NO_2 can now be generated across the whole country (using either the NO_2 model within HAPINZ 3.0 or NIWA's Traffic Impact Model).

It is possible that this method could be expanded by correlating concentrations of individual species related to NEEs (such as copper, silicon, zinc and aluminium) with NO_2 (or NO_x).

3.3 Example of the impact of errors in non-exhaust-emissions emission factors on policy evaluation based on HAPINZ 3.0

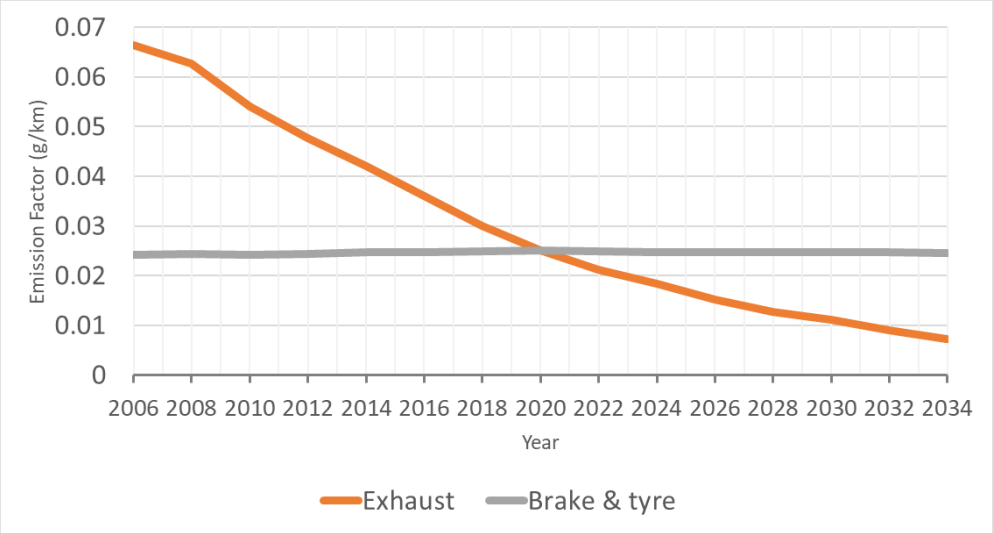
For 2016, HAPINZ 3.0 indicates 'motor vehicle' sources contribute 13% of all $PM_{2.5}$ and 15% of all PM_{10} mortality and morbidity.

HAPINZ 3.0 does not break down 'motor vehicle' any further and states that '…motor vehicles [includes] exhaust, brake/tyre wear and resuspended road dust – based on a motor vehicle fingerprint [taken from Davy and Trompetter (2020)].'

Figure 3.3 shows the change in emission factors over time for both exhaust emissions and emissions from brakes and tyres at 50 km/hr taken from the VEPM. This chart shows how changes in engine and fuel technology have progressively reduced fleet-average tailpipe particulate emissions over the last two decades, with reductions expected to continue for at least another decade. Meanwhile brake and tyre wear emissions have not benefited from comparable technological change leading to no significant change in

emission factors over the same time period. A consequence of this difference in technological trajectories is that brake and tyre wear particulate emissions were estimated to dominate over tailpipe particulate emissions in New Zealand from 2020 onwards.

Figure 3.3 Changing emission factors for exhaust and brake-and-tyre emissions over time



According to VEPM 6.3, (assuming an average speed of 50 km/h):

- over the period 2006 to 2034, fleet-average brake-and-tyre-wear emission factors will be effectively static, while tailpipe emission factors will fall from 54 µg/km to 7 µg/km
- tailpipe emission factors fell below brake and tyre in 2020, and by 2034 will constitute only 23% of motor-vehicle emissions
- PM_{10} from motor vehicles will halve between 2016 and 2034.

According to HAPINZ 3.0, this means the total costs attributed to PM_{10} will fall from $17.3 M/yr to $16.2 M/yr (7%; ie, roughly half of the 13–15% total).

If the VEPM NEE emission factors are always half of the true value (ie, there is a consistent under-estimation error) then:

- non-tailpipe emissions actually became dominant in 2011
- tailpipe emissions will be only 13% by 2034
- PM_{10} from motor vehicles will only have fallen 33%, not 50%, by 2034
- therefore total costs will have only fallen 4%, not 7%.

If the VEPM was correct in 2016, but is under-predicting the rise in NEE emission factors since then (doubled by 2034) then:

- tailpipe emissions will be only 13% by 2034
- PM_{10} from motor vehicles will only have fallen 7% not 50% by 2034
- therefore total costs will have only fallen 1%, not 7%.

So, how can we tell which of these scenarios is playing out? The brief analysis above implies the second scenario is playing out (ie, VEPM NEE emission factors are consistently ‘low’ in the sense that they do not include resuspended road dusts). This means we risk over-estimating how much of PM is responsive to

actions to reduce tailpipe emissions. Source-apportionment trend analysis shows tailpipe components are decreasing and NEE components are increasing (Section 2).

3.4 Other limitations of existing modelling

Both the VEPM and the HAPINZ 3.0 exposure model are designed to describe average emissions and hence concentrations in typical situations. To derive the relationship between NEE contributions and vehicle density, the HAPINZ 3.0 model needed to omit calibration data from two sites (Richmond and Napier) that did not seem to fit the general pattern. This is quite likely due to atypical conditions (either vehicle fleet, prevalence of road dust or specific activities that impacted the location during the monitoring period) at those sites. Monitoring sites are also differentially impacted by wind-driven resuspension and dispersion conditions that may increase or decrease particle deposition.

This means that our ability to correctly estimate NEE impacts (and impacts of policy changes) is weakest in areas where they potentially pose the greater risk; that is, in areas where there are atypically high brake-and-tyre-wear emissions (eg, where there are high levels of heavy-duty vehicles, road inclines, curves, congestion or signalised intersections) and where road-dust deposits are likely to be atypically high (eg, near quarries, demolition and construction sites, on and near unpaved roads, where mud track-out is likely, near desiccated land exposed to high winds, and in post-flood conditions).

4 Conclusions and recommendations for future stages

This report sets out to fulfil the initial recommendations of the literature review of part 1, namely to:

1. survey existing source-apportionment data to determine what detailed analysis is possible that will help to ground-truth current NEE factors, with the proposed analysis becoming the work for stage 2 of the research
2. determine the magnitude of the impact that NEE factors have in air-quality modelling, by undertaking a sensitivity analysis
3. determine the need for any further monitoring, assuming stage 2 provides the anticipated results; and if there is a need, to describe what kind of monitoring would need to be undertaken.

4.1 Conclusions

This work shows a discrepancy between the contribution of NEEs to roadside PM_{10} predicted using VEPM 6.3 and that resolved from source-apportionment studies, with the latter predicting an average contribution of the order of 1 μg m^{-3} greater. Although there may be several factors contributing to this discrepancy, we find that the most plausible is that the 'missing' PM, which is not predicted by the VEPM, mostly relates to dusts that are resuspended from the road surface by the action of tyre contact or traffic-induced turbulence. This resuspension is not accounted for within the VEPM. The VEPM does include brake-and-tyre-wear emission factors, but these relate to the direct emission in response to abrasion. Although based on highly uncertain analysis, this explanation appears to be corroborated by data from the Johnstone's Hill Tunnel showing a much smaller discrepancy in an environment where road dusts other than brake-and-tyre-wear products are likely to be much less prevalent. This means that the use of the VEPM in its current form will systematically under-estimate the concentration of PM at roadside receptors. It also means that projections of future concentrations are likely to be under-predicted if based on the VEPM.

In this review we found no strong evidence of rapid change in NEE emission factors over time. This suggests that trends in vehicle weights, including those related to the increasing penetration of electric and hybrid vehicles carrying large batteries, are currently too slow or insignificant to impact roadside PM concentrations to a detectable degree. Therefore, an assumption of very low or zero growth in NEE emission factors over time is unlikely to introduce additional large errors into future projections. However, over a decade or more this may cease to be true.

Using existing data, it is unlikely that we can assess the accuracy of the current VEPM brake-and-tyre emission factors. It has not yet been possible to resolve an observational brake-and-tyre-wear contribution to PM because it has not been possible to analytically separate brake-and-tyre-wear particles from resuspended road dust, due to their strong correlation on the timescales at which sufficient masses of constituents can be sampled above level of detection. Given their basis in peer-reviewed laboratory studies, we find it reasonable to assume the emission factors are sufficiently accurate. We can provide no strong argument as to why New Zealand emission factors would deviate from those elsewhere in the world. It may be possible to experimentally validate current VEPM brake-and-tyre emission factors in a tunnel-based study where other road-dust sources are, or can be, minimised.

Although capturing more observational data of the same nature in the same places will increase the power of the source-apportionment techniques described in this report, there are diminishing returns and physical limits associated with the methods. Statistical methods are only able to resolve components that are not

strongly corelated in time. The analysis of over 10 years of data from some Auckland sites, for instance, has still not allowed road dust and brake and tyre wear to be separately resolved, due to their strong correlation. We conclude that using more of the same data and the same technique is unlikely to yield any new breakthroughs.

4.2 Implications

Typically, we found that total road dust (including brake and tyre wear) contributes in the order of 10% of total measured PM_{10} at roadside sites, whereas brake and tyre wear (as predicted by VEPM 6.3) is around 3% of total measured PM_{10}. Therefore, any error in either contribution represents an error in assessed PM_{10} equal to a fraction of those values. For example, using the VEPM values for brake and tyre wear but omitting road dust will lead to an under-estimation of total PM_{10} (or its change in response to a policy or project), or its impacts (eg, associated mortality or social costs) in the order of 7% (assuming the brake-and-tyre-wear emission factors are accurate). This may have implications for estimating trends, due to overestimating the overall reduction in motor-vehicle PM that will come from the reduction in exhaust emissions.

The implications of the uncertainty regarding the emission factors for brake and tyre wear is significant only in the relatively limited context of policies or technologies that might substantially change these emissions independently of road dust (ie, major changes to brake and tyre composition) or large-scale adoption of alternative technologies (eg, regenerative braking). Fleet changes that have an impact on vehicle weight may be less significant, as they are likely to impact on the resuspension of road dust too.

4.3 Recommendations for stage 2 and beyond

Below we list a range of recommendations, from low-cost immediate-impact solutions to longer-term higher-cost ones. Before embarking on any but the immediate low-cost options, we recommend an exercise to define the scope, priority and probable cost of each major option. Such an exercise will enable decisions on further resourcing to be made with confidence.

4.3.1 Lower cost, immediate impact

Specific recommendation #1: Introduce a road-dust emission factor into the VEPM

The use of the VEPM to model the impact of road vehicles on roadside air quality appears to carry with it a significant error. We recommend that this error is reduced by introducing a dust emission factor model into the VEPM. Although ideally informed by detailed experimental and/or observational research overseas and in New Zealand (see below), on an interim basis, a model based on existing overseas research and New Zealand-based data and analysis (like that presented in this work) could be implemented. Such an emission factor could include a term for road abrasion emissions based on European Monitoring and Evaluation Programme and European Environment Agency factors (European Environment Agency, 2023), which could also contain more vehicle categories (including battery electric and hybrids).

Specific recommendation #2: Change the method to derive motor-vehicle derived coarse PM in HAPINZ 3.0

To reduce some errors inherent in any health impact assessments that use the HAPINZ 3.0 model, we recommend a minor change in the HAPINZ 3.0 exposure model. Specifically, we recommend that the use of traffic-emission density (used to derive estimates of motor-vehicle derived coarse PM) is replaced by NO_2, given that NO_2 presents a stronger relationship with non-exhaust vehicle PM and data on NO_2 is more readily available and easily updated.

Specific recommendation #3: Work with councils to continue long-term time series of pollutant concentrations

At present, long-term trends in NEE emission factors appear to provide small changes, but this could change in the future, and the degree and timing of any change is uncertain. Given the difficulty and cost in accurately modelling emission factors and in using laboratory studies to track trends, it is important to maintain an observational monitoring programme. In order to track emerging trends in real-world NEE emission factors, we recommend that either continuous or periodic filter sampling (currently conducted by Regional Councils as budgets and priorities allow) is maintained, and supported by a programme of filter analysis and receptor modelling.

Currently, filter sample records from Auckland are, in our opinion, largely representative of many locations, and a 10-year dataset from Nelson is expected to be available in late 2024, which will be representative of smaller towns. Data-sharing arrangements with councils that are carrying out filter sampling should be sought by the NZTA to maintain access for ongoing monitoring of NEE. The NZTA should work with councils and the Ministry for the Environment to raise this need, including considering whether there are a minimum number of sites required and the duration and frequency of sampling – for example, if continuous monitoring cannot be achieved, then would, for example, 1 year in 5 be adequate, with locations being rotated on a 5-year cycle? Encouraging regulators to include NO_2 monitoring at source-apportionment sites would also be helpful.

Working to ensure traffic data (eg, hourly time-series traffic counts, including vehicle size and number of wheels) is readily available for these sites will also greatly extend the analysis possible.

4.3.2 Higher cost: New experimental and observational studies

The contribution of NEEs to ambient PM concentrations in most of urban New Zealand is low by international standards. Given that no device exists to measure such contribution directly, indirect methods, which were not designed for this purpose, have previously been used. These methods are reaching their limits of detection. We face diminishing returns from continued investment in these methods.

At present, we find that the total contribution of NEE emissions to ambient PM concentrations is relatively small and the impact of errors in their estimation is therefore even smaller. It is, however, important to monitor such emissions given the risk of their rising into the future.

We therefore recommend that new methods are developed that are purposely designed to track non-exhaust vehicle emissions. This predominantly means raising the sensitivity of methods. In principle this is done in one of three ways:

- increasing measurable concentrations (by moving sampling sites closer to roads, or sampling within road tunnels, on moving vehicles or in the laboratory)
- selecting sites with or inducing higher emissions
- investigating improved monitoring or speciation technologies and methods.

Although we find that the road-dust component (not included in the VEPM) may contribute around 1 μg m^{-3} to PM_{10} at a typical roadside site, this value is likely to vary significantly with respect to vehicle speed and wind conditions. However, this value could also be a substantial under-estimate in certain situations, such as in:

- the presence of demolition or construction dust, quarry or agricultural dust
- high winds
- cases of post-flood silt deposition on roads
- situations where there is damage to roads.

Increasing storm and extreme weather events are expected, making high-dust scenarios more common. There could be a growing need for a post-disaster risk-assessment tool for disaster management and scenario planning. We recommend that the scope of any future research takes into account such situations, as the physical processes involved are the same as those impacting low-dust roads in normal conditions.

We therefore recommend that a research programme is scoped to develop a process-driven approach to modelling road-dust resuspension, including brake and tyre wear, based on data collected in New Zealand.

Such a programme should ideally consider the following elements:

- environmental data collection in at least one road tunnel, and alongside the roads leading to that tunnel
- surveys (and monitoring) of road-dust loading at representative sites
- opportunistic studies using naturally deposited road dust
- instrumented vehicles or trailers being driven over test tracks and roads of varying dust conditions
- laboratory studies (especially focussing on brake and tyre wear).

Here, we make specific recommendations for the first three elements. The last two would require considerable resources and are the current focus of research programmes in Germany and the US. We therefore recommend a wait-and-see approach until the results are available from these countries.

Specific recommendation #1: Observational study in and around the Lyttelton Tunnel on SH74

Tunnel studies are well-established as a means of observing vehicle emission factors.[2] However, an important requirement is stable, uni-directional monitored airflow.

Although recent public data is not available, we expect concentrations of air pollutants to be higher in the SH74 Lyttleton Tunnel than the Waterview, Johnstone's Hill and Victoria Park tunnels. Lyttelton Tunnel is also used by a wider range of vehicle types, with a high proportion of heavy-duty vehicles. Although the tunnel is bi-directional, the mechanical (transverse) ventilation system fulfils our requirement of stable, uni-directional monitored airflow, with fresh air taken in and exhaust air expelled both at the northern portal. The Terrace and Mt Victoria tunnels in Wellington suffer from variable airflow.

We recommend a study around the Lyttelton Tunnel (although the Waterview and Mt Victoria tunnels may be investigated as alternatives). A pilot study may be advantageous, to establish the feasibility, duration and intensity of measurements. The study should include the following features.

- Four monitoring sites, representing different expected contributions of brake and tyre wear and other road dusts:
  - Lyttelton Tunnel fresh air intake
  - Lyttelton Tunnel exhaust air (for brake and tyre wear, as substantial braking occurs at the south end of the southbound tunnel)
  - ambient roadside on the main tunnel approach road (ie, being used by the same vehicles as the tunnel), for example, the SH74 roadside north of the tunnel portal
  - a nearby intersection on the main tunnel approach road (ie, with curves requiring acceleration and deceleration, thus increasing brake and tyre emissions), for example, the SH74/SH76 intersection.

[2] Note: Tunnel studies can be used for multiple objectives (see, for example, *NZTA research report 687 Improving our understanding of New Zealand's vehicle fleet greenhouse gas and harmful emissions using measured emissions data – Stage 1* (Smit et al., 2022)). We therefore suggest coordinating with any other projects to ensure that studies are achieving best outcomes.

- High-resolution monitoring of NO_x, $PM_{2.5}$, PM_{10} and black carbon at all sites:
  - daily and hourly PM filter sampling at all sites
  - detailed monitoring of traffic-fleet composition, volumes and speeds in the tunnel and at both ambient sites
  - monitoring of airflow through the tunnel (allowing dilution ratios linking emissions and concentrations to be constrained)
  - monitoring of road-surface wetness (which influences road-dust loading and hence, emissions)
  - local ambient meteorology (must include wind speed and direction to aid with source apportionment)
- An ideal study would also include the deliberate disturbance of natural conditions by occasionally:
  - adding salt, sand or silt to the study road
  - washing the tunnel
  - sampling from the road surface (to establish road-dust loading and composition).
- Further additional features that would enhance the value of the study would include:
  - video surveillance of vehicles to detect load shedding and the potential for vehicles to introduce dust and precursors into the tunnel.

From this data, the following analyses could be carried out:

- source apportionment of PM samples
- NO_x emission factors, with some breakdown for different components of the vehicle fleet using the tunnel, thus providing validation of VEPM emission factors (and a correction factor if required)
- validation of the use of NO_x emission factors when combined with a dispersion model using data from ambient sites
- traffic-related road-dust emission factors, possibly with some breakdown for different components of the vehicle fleet
- improved estimation of brake-and-tyre-wear emission factors, possibly with some breakdown for different components of the vehicle fleet
- indications of the impact of:
  - acceleration on brake-and-tyre-wear emission factors
  - road-surface wetness on road-dust emission factors
  - wind speed on road-dust emission factors
  - road-dust loading on road-dust emission factors.

Specific recommendation #2: National survey of road-dust loading

Road-dust emissions are known to be strongly related to the amount of road dust deposited on the road. The amount of road dust deposit on roads across the country, and how it varies over time, is effectively unknown. Day-to-day dust loading can vary enormously at any location as it depends on many variables. Early research indicated that it was almost impossible to quantify any potential dust reservoirs with any certainty (see, for example, Semadeni-Davis et al. (2021) and references therein) and hence predict dust loading. Therefore any approach to dust loading would have to be empirical, and rely on measurements taken at as many locations and under as many conditions as possible.

We recommend an initial survey at sites across the country that are expected to cover the range of likely dust loadings. Road-dust loading is strongly modified by the pattern of recent weather, with rain causing both

run-on and run-off. We further recommend an investigation into the logistics of turning this into a regular monitoring programme.

Specific recommendation #3: Transect monitoring of high-dust roads

In order to translate the dust loading into concentrations, we recommend a transect monitoring programme. As a minimum we recommend establishing monitoring for $PM_{2.5}$ and PM_{10}, using optical instruments (calibrated for road dust) on either side of a road with a known high road-dust loading. Ideally, we recommend that this is extended to multiple sites extending from either side of the road to form 'transects'. When paired with local wind direction and speed data, this will allow the contribution of the road to be determined by subtracting the upwind concentration from the downwind concentration.

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Appendix A: Source apportionment

The following tables show the long-term average PM_{10} (Table A-1) and $PM_{2.5}$ (Table A-2) values measured during source-apportionment studies around the country, and the proportion of these attributed to motor vehicles.

Table A-1 Long-term average PM_{10} values measured during source-apportionment studies around the country and the proportion attributed to motor vehicles

Region	Site	Time period	Source-apportionment averages			
			PM_{10} ($\mu g\ m^{-3}$)	PM_{10} motor vehicles ($\mu g\ m^{-3}$)	Uncertainty (stderr) ($\mu g\ m^{-3}$)	Percentage uncertainty
Wellington	Masterton	2002–2004	17.51	0.98	0.06	6
Wellington	Upper Hutt	2000–2002	9.90	1.48	0.08	5
Wellington	Wainuiomata	2006–2008, 2011–2014	13.44	1.63	0.05	3
Wellington	Seaview	2002–2004, 2005–2007	16.60	2.22	0.13	6
Auckland	Kingsland	2004–2007	16.11	3.17	0.12	4
Auckland	Takapuna	2006–onwards	15.56	3.59	0.12	3
Auckland	Queen Street	2006–onwards	17.45	6.41	0.03	0.5
Auckland	Penrose	2006–2016	16.50	4.11	0.06	2
Auckland	Khyber Pass Road	2006–2015	17.83	6.18	0.09	1
Auckland	Henderson	2006–onwards	13.79	1.94	0.05	3
Auckland	Patumahoe	2010	10.54	NA	NA	NA
Nelson	Tahunanui	2008–2009	19.95	2.30	0.20	9
Nelson	Nelson City	2006–2012	20.41	2.02	0.14	7
Otago	Dunedin	2010	26.87	2.84	0.26	9
Canterbury	Christchurch (Coles Pl)	2013–2015	19.48	2.19	0.11	5
Hawkes Bay	Awatoto	2016–2017	13.59	0.87	0.15	17
Hawkes Bay	Marewa Park	2017–2018	13.08	1.92	0.24	12
Waikato	Tokoroa	October 2015–October 2016	13.05	1.51	0.07	5
Tasman	Richmond	2013–2016	17.08	3.64	0.19	5
Marlborough	Blenheim	2007	10.75	1.14	0.11	10

Table A-2 Long-term average $PM_{2.5}$ values measured during source-apportionment studies around the country and the proportion attributed to motor vehicles

Region	Site	Time period	Source-apportionment averages			
			$PM_{2.5}$ ($\mu g\ m^{-3}$)	$PM_{2.5}$ motor vehicles ($\mu g\ m^{-3}$)	Uncertainty (stderr) ($\mu g\ m^{-3}$)	Percentage uncertainty
Wellington	Masterton	2002–2004	9.28	0.40	0.02	6
Wellington	Upper Hutt	2000–2002	5.40	1.36	0.07	5
Wellington	Wainuiomata	2006–2008, 2011–2014	6.64	0.79	0.03	4
Wellington	Seaview	2002–2004, 2005–2007	5.14	0.57	0.03	6
Wellington	Masterton East	2018	10.11	0.44	0.06	14
Auckland	Kingsland	2004–2007	7.12	1.92	0.06	3
Auckland	Takapuna	2007–2016	6.50	1.82	0.03	2
Auckland	Queen Street	2006–2016	8.76	4.23	0.06	1
Auckland	Penrose	2006–2016	7.05	2.34	0.05	2
Auckland	Khyber Pass Road	2006–2015	8.11	4.05	0.05	1
Auckland	Patumahoe	2010	3.34	NA	NA	NA
Nelson	Nelson City	2006–2012	16.03	0.78	0.05	7
Otago	Dunedin	2010	10.29	2.84	0.26	9
Canterbury	Timaru	2006–2007	16.12	0.88	0.06	7
Canterbury	Woolston	2013–2014	10.26	2.59	0.08	3
Canterbury	Christchurch (Coles Place)	2013–2015	8.73	1.01	0.04	4
Hawkes Bay	Hastings	2006–2007	11.75	1.11	0.08	8
Hawkes Bay	Awatoto	2016–2017	3.75	0.20	0.04	18
Hawkes Bay	Marewa Park	2017–2018	6.80	0.15	0.02	11
Waikato	Tokoroa	October 2015–October 2016	10.12	0.67	0.03	5
Tasman	Richmond	2015–2016	10.57	0.38	0.02	5

Appendix B: Sensitivity analysis

This section shows how sensitive dispersion model results are to the choice of emission factors, and examines whether the errors introduced by such choice cause any significant changes to results compared to other input choices.

We have run a scenario using a sequence of changes to inputs with the simple Gaussian dispersion model AusRoads. This model was developed in the 1990s by the Victoria Environmental Protection Agency for the purpose of carrying out roading assessments in situations where the terrain was sufficiently uncomplicated that a steady-state model was appropriate. It was used in the joint NZTA and NIWA roadside project (Longley et al., 2013), which investigated the impact of a major Auckland motorway on local air quality.

B.1 Baseline scenario

The baseline scenario is the one to which all model runs are compared. In each subsequent model run, only one model input or parameter is changed at a time in order to see the effect of that input on the final prediction.

The baseline scenario was run on a 2,000 m stretch of four-lane road running directly north to south. Receptors were sited along a transect running away from the midpoint of the road perpendicular to it to a distance of 1,000 m. The wind direction was constantly 270^{o}, meaning the receptors were directly downwind of the road. The wind speed was 5 m s^{-1} and atmospheric stability was neutral (category D in the Pasquill–Gifford (P-G) stability categories). The annual average daily traffic for the road was 12,000 vehicles, or 500 vehicles per hour. The emission factors used were the default values in the VEPM for the year 2022 (Metcalfe & Peeters, 2022) for a speed of 50 km hr^{-1}. The emissions factors for exhaust $PM_{2.5}$ emissions and the brake-and-tyre-wear PM_{10} emissions were added together.

B.2 Using emissions calculated in the Vehicle Emissions Prediction Model

The VEPM is a New Zealand developed and maintained model generating fleet-weighted emission factors for the main air pollutants of concern, CO_2 and fuel consumption. It can calculate emission factors for a range of years to supply input data for past, present and future scenarios, and for a range of driving conditions and vehicle fleets. The default inputs include a best estimate of New Zealand's on-road fleet composition. The model also calculates an estimate for brake and tyre wear, but does not estimate emission factors for road wear or resuspended dust.

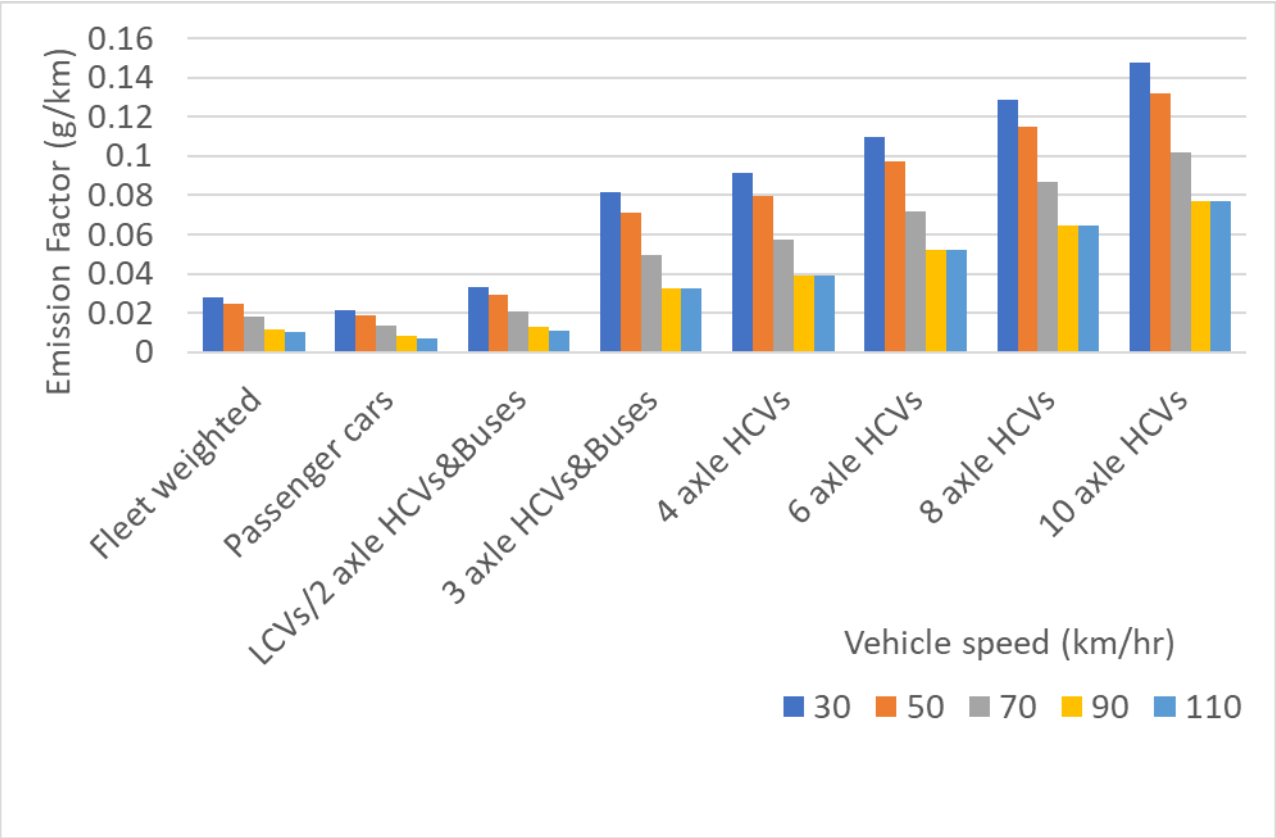
The VEPM is the single source of estimates of emissions from vehicles and, as such, is fundamental to almost every assessment of environmental and health impacts from vehicle-related air pollution that is undertaken for regulatory purposes.

The emission factors for brake and tyre wear in the VEPM are speed dependent; tyre wear decreases with speed while brake wear increases. Emissions from road wear and resuspended dust are not calculated. The fleet-weighted emission factors are dependent upon the breakdown of the fleet by vehicle type and age.

For 50 km hr^{-1}, the emission factors are 18.6 mg km^{-1} for passenger vehicles, 29.2 mg/km for light commercial vehicles, and range from 29.2 to 132.1 mg km^{-1} for buses and heavy commercial vehicles.

Figure B.1 shows the range of emission factors at various speeds by vehicle type assigned by the VEPM for the year 2022.

Figure B.1 Emission factors for various vehicle classes at various speeds used in the Vehicle Emissions Prediction Model 6.3, 2022



Note: LCVS = light commercial vehicles; HCVs = heavy commercial vehicles.

B.2.1 Use of findings from stage 1A

The literature review in stage 1A of this work reported on emission factors that had been derived through a variety of methods and with a variety of vehicles under a variety of laboratory, testing and environmental conditions. It is little wonder that the range of values in the literature was wide. Table 7-1 in Semadeni-Davies et al. (2021, p. 76) summarises the findings. For brake wear, the range reported for passenger vehicles was 11–30 mg/vehicle/km; for tyre wear the range was 1–1000 mg/veh/km, dependent on the size of vehicle and driving conditions and behaviour; and for road wear, the range was 3–80 mg/vehicle/km, again dependent on vehicle size and road condition.

B.3 Other model inputs considered

Baseline values are in **bold** text.

B.3.1 Meteorology

Windspeed: 4.6, 4.8, **5.0**, 5.2, 5.4 m s^{-1}.

Atmospheric stability: categories C, **D**, E.

B.3.2 Traffic

Annual average daily traffic: 11,500, **12,000**, 12,500 vehicles.

B.3.3 Emission factors

As the VEPM does not include road dust in its emission factors, we added this by doubling the VEPM NEE emission factors estimate.

This increased the emission factors from the baseline 0.046 g km^{-1} to 0.071 g km^{-1}.

B.3.4 Background concentrations

As an indicative background concentration, relatively unaffected by immediate sources, we used the annual average concentration of $PM_{2.5}$, as measured at the rural Auckland site at Patumahoe (4.7 µg m^{-3}). We chose to use $PM_{2.5}$, rather than PM_{10}, so as to exclude any agricultural dust that may be impacting the measurement. We varied this input by reducing or increasing the concentration by 25%, to give a lower value of 3.5 µg m^{-3} and a higher value of 5.9 µg m^{-3}.

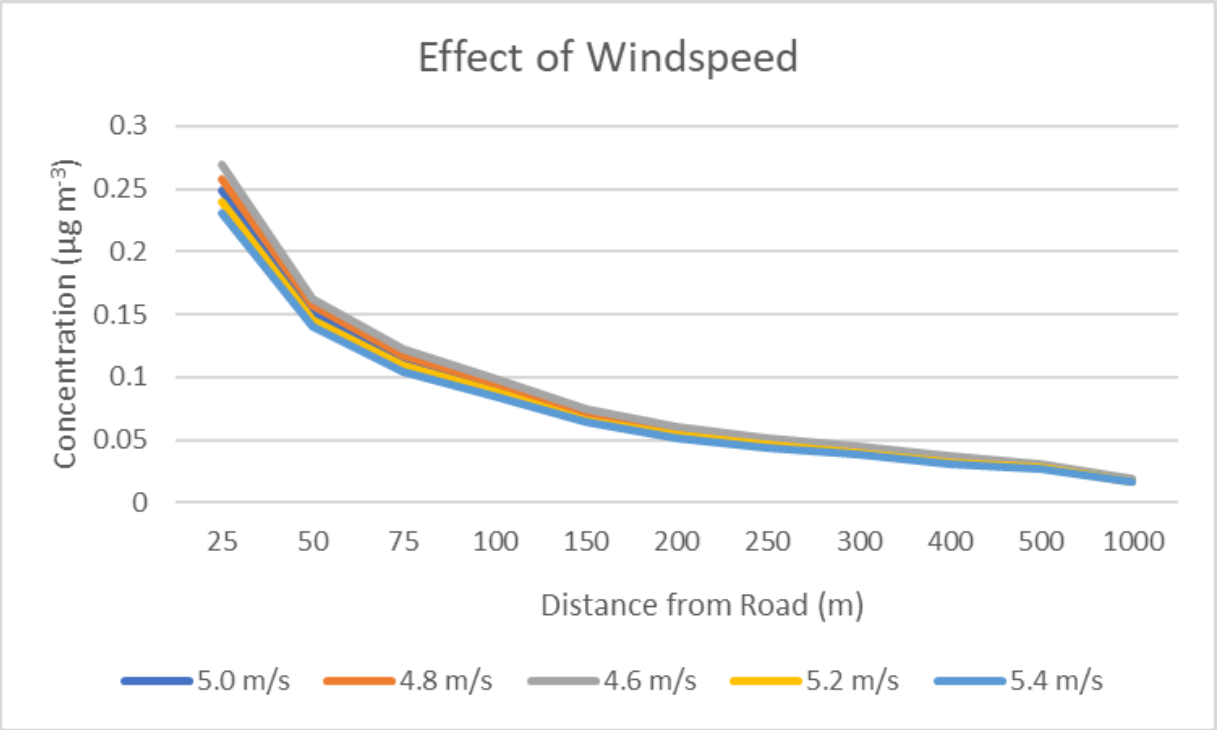
B.4 Results

All changes in values noted are from the results at 25 m from the road.

B.4.1 Changes in meteorological inputs

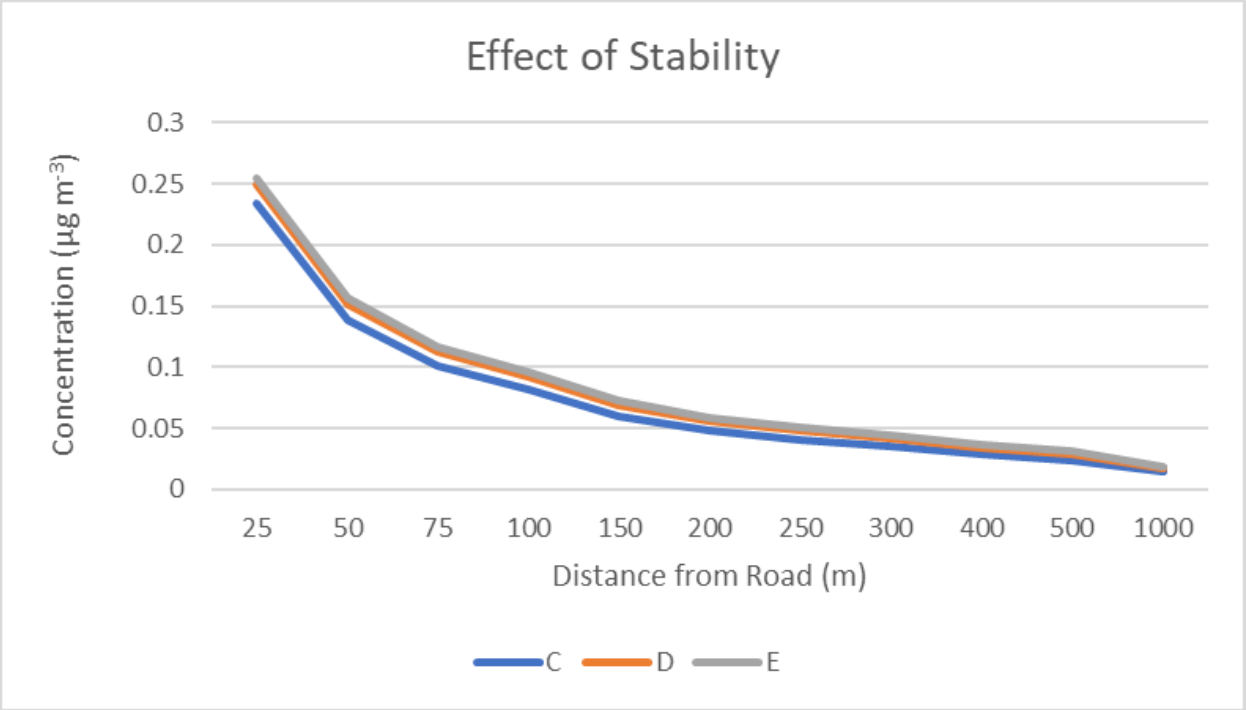
Figures B.2 and B.3 show the impact of changes in windspeed and atmospheric stability on the predicted PM_{10} concentrations. In order to show the small variation better, these were run without background concentrations; only the emissions from the road were modelled. The results changed by only 7% (for a 0.4 m/s change) and 8% (for a one category change) respectively, from the baseline scenario.

Figure B.2 Variation in predicted PM_{10} concentrations (µg m^{-3}) with differing wind speeds



Note: The scale of distance from road is not linear.

Figure B.3 Variation in predicted PM_{10} concentrations (µg m^{-3}) with differing atmospheric stability, parametrised as P–G stability categories

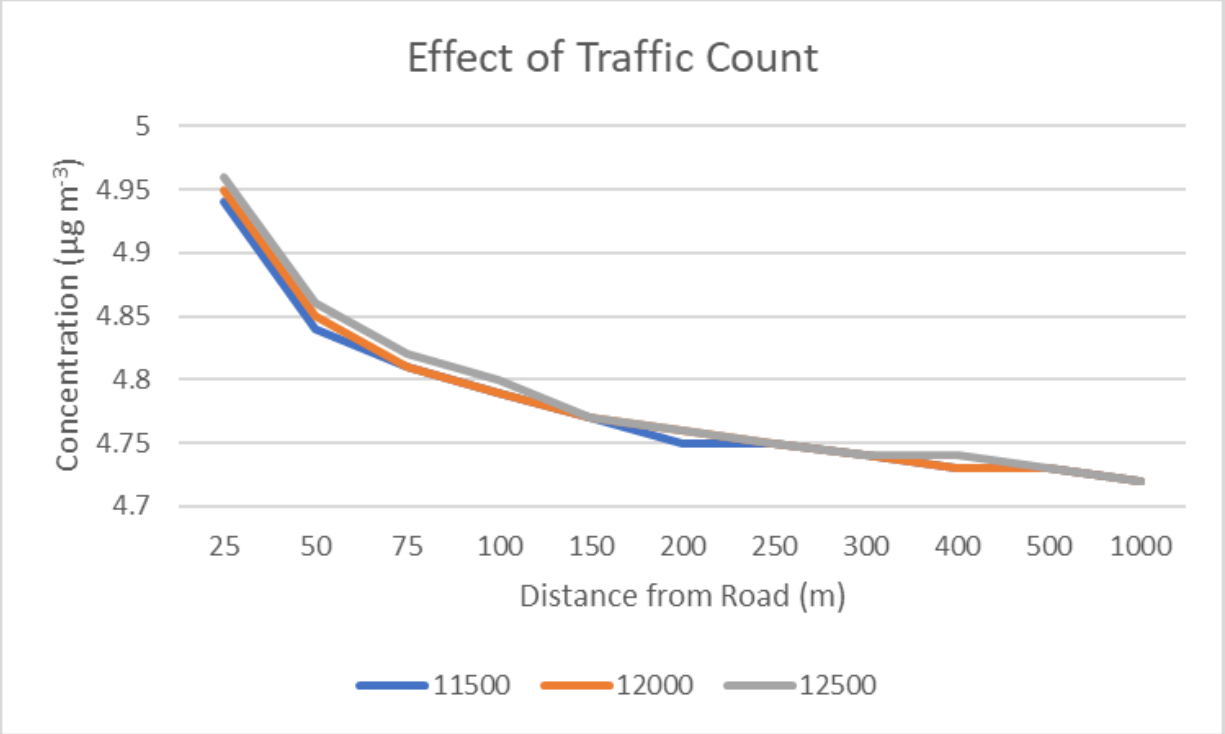


Note: The scale of distance from road is not linear.

B.4.2 Changes in traffic inputs

Changes in annual average daily traffic also made little difference, as shown in Figure B.4. A 4% difference in vehicle numbers resulted in a 0.2% difference in concentrations resulting from the road and background PM_{10}.

Figure B.4 Variation in predicted PM_{10} concentrations (µg m-3) with differing vehicle counts

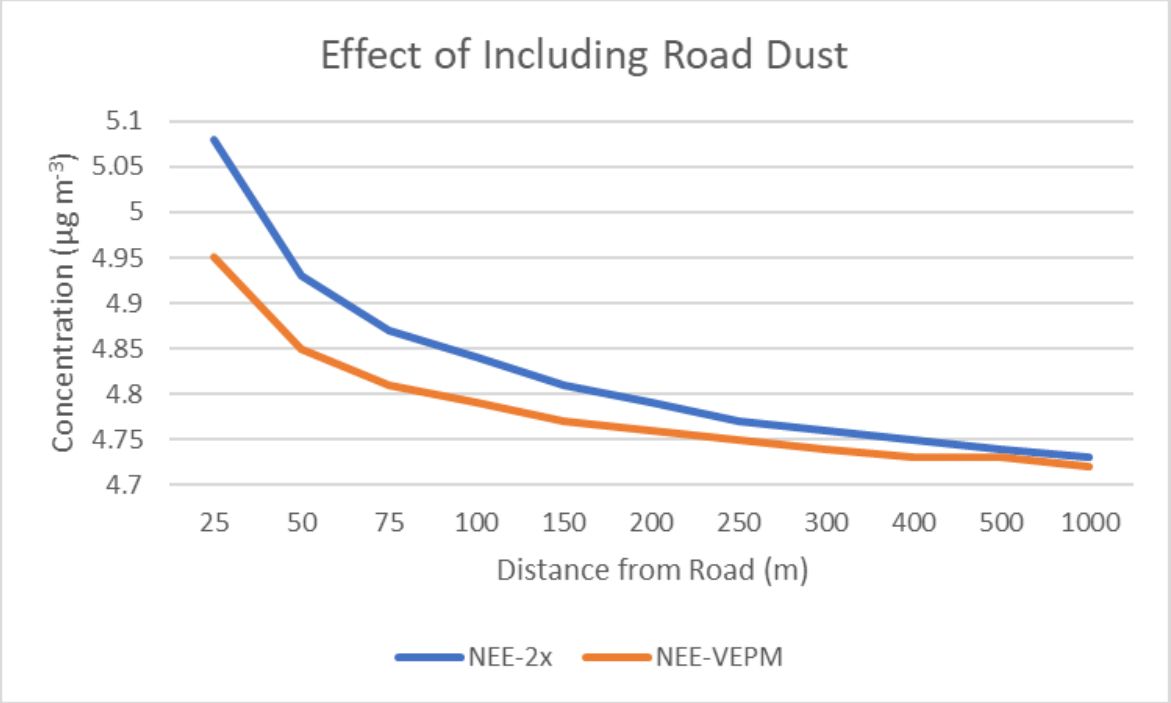


Note: The scale of distance from road is not linear.

B.4.3 Changes in non-exhaust-emissions emission factors

Adding a 'placeholder' for dust emissions from the road to the emission factors resulted in larger change as shown in Figure B.5. The increase in concentrations from the road was 13% above the baseline scenario, from a 54% increase in the fleet-average emission factors used.

Figure B.5 Variation in predicted PM_{10} concentrations (µg m-3) by adding a road-dust component

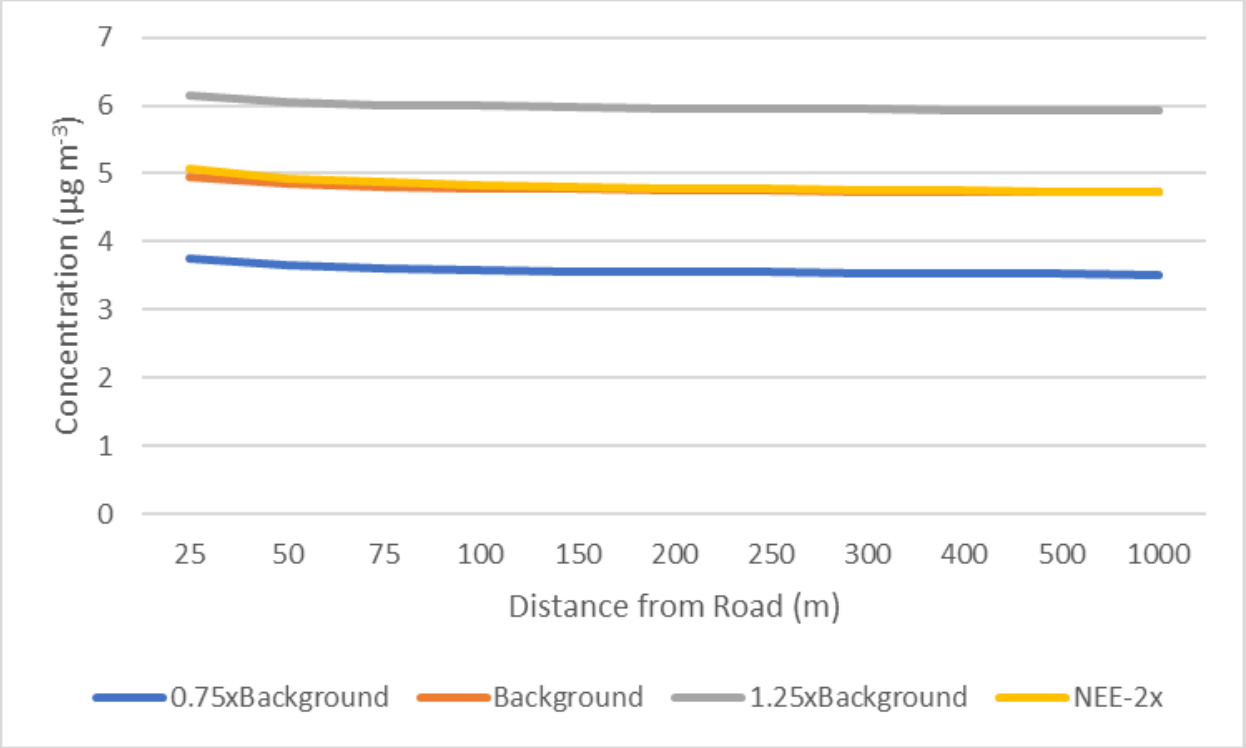


Note: The scale of distance from road is not linear.

B.4.4 Changes in background concentrations

The impact of changing the background concentrations, however, is nearly an order of magnitude greater than any of the changes to the other inputs. Figure B.6 shows the predicted concentrations using differing background concentrations, and includes the results from the model run with an emission factor for road dust incorporated, to show the minimal difference from the baseline scenario compared to varying the background concentration.

Figure B.6 Variation in predicted PM_{10} concentrations (µg m-3) from varying background PM



Note: The scale of distance from road is not linear. The impact of adding a road-dust component is also included.

B.5 Conclusions based on model runs

The sensitivity analysis emphasises how any single road source is a minor contributor to the total PM measured at a site, even at close distances. Uncertainties in measured or estimated background concentrations can mask the impact of almost any other reasonable changes in input data. Increasing the NEE to account for a contribution from road dust had the most substantial impact on concentrations generated by the road.