Determining the ecological and air quality impacts of particulate matter from brake and tyre wear and road surface dust

Stage 1 – Literature review and recommendations for developing new emission factors for New Zealand

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A Semadeni-Davies, NIWA, Auckland
G Coulson, NIWA, Auckland
J Gadd, NIWA, Auckland
E Somervell, NIWA, Auckland
I Longley, NIWA, Auckland
G Olivares, NIWA, Auckland

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

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Finally, our colleague Daniel Morrish (NIWA) prepared the glossary of acronyms and the accompanying spreadsheet containing copies of tables from this report.

Abbreviations and acronyms

AADT	average annual daily traffic
AQEG	Air Quality Expert Group (UK)
C-CALM	Catchment Contaminant Annual Load Model
CLM	Contaminant Loads Model
DA14	14 mm dense asphalt concrete
EF	emission factor
EMEP/EEA	European Monitoring and Evaluation Programme/European Environment Agency
EV	electric vehicle
FWMT	Auckland Council Fresh Water Management Tool
HAPiNZ	Health and Air Pollution in New Zealand
HAWRAT	Highway Agency Water Risk Assessment Tool
ICEV	internal combustion engine vehicle
LM	low metallic
LSPC	Loading Simulation Program C++
MEDUSA	Modelled Estimates of Discharges for Urban Stormwater Assessments
MUSIC	Model for Urban Stormwater Improvement Conceptualisation
NAO	non-asbestos organic
NEEs	non-exhaust emissions
NIWA	National Institute of Water and Atmospheric Research
NPSFM	National Policy Statement for Freshwater Management
OECD	Organisation for Economic Co-operation and Development
PAHs	polycyclic aromatic hydrocarbons
PM	particulate matter
PM_{10} and $PM_{2.5}$	particulate matter with an aerodynamic diameter of 10 or 2.5 µm or less, respectively
PSD	particle-size distribution
RSS	Road Stormwater Screening [model]
SELDM	Stochastic Empirical Loading and Dilution Model
SM	semi-metallic
SUSTAIN	System for Urban Stormwater Treatment and Analysis IntegratioN
TRAKER	Testing Re-entrained Aerosol Kinetic Emissions from Roads
TSP	total suspended particulates
TSS	total suspended solids
VEPM	Vehicle Emissions Prediction Model
VFEM-W	Vehicle Fleet Emission Model – Water
VKT	vehicle kilometres travelled
VTA10	10 mm very thin asphalt concrete
VTA6	6 mm very thin asphalt concrete

Contents

Executive summary

Waka Kotahi NZ Transport Agency commissioned the National Institute of Water and Atmospheric Research (NIWA) to undertake a desktop study to provide updated information on non-exhaust emissions (NEEs) from vehicles in New Zealand. Air- and water-borne particulate emissions from road transport are important contributors to urban pollution. The major sources of particles from road transport are from combustion in the engine emitted as exhaust, and mechanically generated particles from abrasion of brakes, tyres and the road, along with particles re-entrained from the road into the air by wind and the turbulence caused by passing traffic (ie, NEEs). Waka Kotahi estimates the environmental impact of traffic on air and water quality using estimates of emission factors (EFs) from vehicle activities. EFs are defined as the mass of a contaminant emitted by a vehicle per kilometre travelled. The priority contaminants considered are PM_{10} and $PM_{2.5}$ (ie, particulate matter (PM) with a diameter of 10 or 2.5 µm or less, respectively) for air quality, and zinc, copper (associated with tyre and brake wear, respectively) and sediment for water quality.

EFs are used as input to both air and, to a lesser extent, water quality models. The main concern is that the EFs currently used by Waka Kotahi are out of date and do not reflect ongoing modernisation of the vehicle fleet. To address this concern, Waka Kotahi has proposed a two-stage programme whereby the first stage, presented in this report, is a desktop study to identify and collate existing NEE data, determine gaps in that data, and recommend a methodology to address those gaps. The second stage will be to implement the recommended methodology in order to derive robust current and future estimates of EFs and contaminant loadings required for managing air and water quality.

Non-exhaust EFs can be estimated using two approaches. The first, called ‘emissions from vehicles’ in this report, measures the amount of each contaminant that is emitted by vehicles due to wear from roads, brakes and tyres directly under controlled conditions to estimate contaminant EFs for different types of vehicles and driving conditions. The mass of the contaminants emitted for a section of road is then calculated from the number of each type of vehicle travelling along the road over a set period of time. The second approach, here referred to as ‘emissions to air and water’, measures the mass of the contaminants in air, road dust and road runoff and then apportions this mass to brake, tyre and road wear based on the chemical and physical characteristics of the contaminant. Per-vehicle emissions can be estimated by dividing the contaminant mass by the traffic volume.

Current and future fleet

The composition of the current fleet and projected future trends in fleet composition are summarised in section 2 and Appendix A of this report. Of most interest here is that while electric vehicles (EVs) currently make up only 0.3% of the current fleet, EVs are projected to make up around half of passenger vehicles by the 2040s.

Emissions from vehicles

The first method of deriving EFs is presented in section 3 of this report. The main part of this section discusses the mechanisms for the generation of PM due to the wear or abrasion of brakes, tyres, and road surfaces and presents information on the composition of these sources and the chemical and physical characteristics of PM from each source. It was found that the wear rates reported for brakes, tyres and road wear vary greatly between different studies, largely due to differences in the composition of brakes and tyres between vehicle types and brands, and differences in the driving conditions tested and the metrics reported. That is, the amount and composition of NEEs from a vehicle is a function of the type and composition of the vehicle’s brakes and tyres and the type and chemical and physical characteristics of the road surface. The

relative importance of each source to the total contaminant mass emitted by a vehicle varies by vehicle type. NEEs are generally greater from heavy vehicles than light vehicles. Other factors influencing NEEs include driving conditions (eg, road slope, traffic congestion), driver behaviour (eg, speed, braking habits) and climate. For example, freely flowing high-speed traffic is likely to result in lower brake emissions than slow, congested traffic but will increase the amount of road and tyre wear. For this reason, urban roads generally have higher emissions than rural roads due to the stop–start nature of urban driving, and emissions tend to be greater at bends and intersections.

Section 3 also discusses the possible effects of changing technologies on NEEs, such as the development and growth of EVs, and regulation, such as restrictions on copper used in brakes. While we were not able to find examples of measured emission from EVs, there have been several studies that have modelled PM emissions based on vehicle weight (EVs are generally heavier than equivalent internal combustion engine vehicles) and the use of regenerative braking. These studies suggested that emissions from tyre and road wear are likely to increase due to increased overburden, while emissions from brakes are likely to decrease. For this reason, the overall change in PM will be minimal, but the ratio of fine (from brake wear) to coarse particles (from tyre and road wear) is also likely to change. We found no information on zinc and copper emissions from EVs – however, we can speculate that zinc emissions are likely to increase and copper emissions decrease.

Emissions to air and water

The second method of determining EFs is discussed in sections 4, 5 and 6. Section 4 gives a general overview of the delivery of PMs emitted by NEE sources to air and water. Sections 5 and 6 discuss the apportionment to NEE sources of PM found in air and water, respectively.

NEEs from vehicles can be released to air, fall to the road surface as road dust, or be retained by the vehicle. Airborne PM can be held aloft, transported to other locations, or deposited onto the road surface. Road dust accumulated on the road surface can be resuspended by wind and traffic or washed off and transported in runoff during rainfall. The fate of PM from NEE sources depends on the size range, shape and weight of the particles, driving conditions, driver behaviour, road type and condition, and climate.

Section 5 discusses methods of deriving NEEs from air quality monitoring and the statistical methods used to apportion these emissions to specific sources. There are a number of challenges to apportionment. First and foremost, choosing an appropriate tracer to use is challenging given the wide variety of materials and their relative proportions found in the composition of brakes, tyres and roads and the presence of other non-traffic-related sources. Another issue is resuspension, which can be a significant source of PM to air, since it is extremely difficult to separate resuspended road dust and newly emitted tyre and road wear. Finally, apportionment studies are hampered by the fact that road types, driving conditions and the surrounding land use can change over short distances so that the relationships found for one section of road may not be valid elsewhere. Like EFs derived from wear tests, there is a wide range EFs determined from air quality monitoring published in the literature.

NEEs to road dust and road runoff are discussed in section 6. Although NEEs are known sources of zinc, copper and sediment to stormwater, there is very little information on EFs to water published. This in part reflects the fact that, unlike air quality modelling, stormwater quality models generally do not use measured EFs as input. Instead, the parameters that describe the build-up and wash-off of contaminants in water quality models are usually calibrated against the concentrations of the contaminants in either road dust or runoff, or both.

Gap analysis

We identified a considerable number of gaps in the understanding of NEEs that make it difficult to determine the most appropriate EFs for use in New Zealand. These are discussed in section 7 and summarised below.

Paucity of information and understanding

1. A lack of up-to-date information on NEEs. Most of the studies cited were published before 2005. Although there have been a number of reviews on the impacts of NEEs on air quality in recent years, most of the research used to derive PM EFs is at least 15 years old and does not reflect new innovations and fleet modernisation. The most recent estimates of EFs for zinc and copper that we have found were published in 2010.
2. Non-reporting of metadata required to derive EFs from air and water quality data. That is, while there have been numerous studies that have characterised air quality and the quality of road runoff and road dust with respect to the concentrations of the priority contaminants over recent years, information on traffic flows and runoff volumes required to calculate contaminant loads data have largely not been reported.
3. Limited understanding and reporting of the spatial and temporal variability in NEEs despite the fact that estimates of EFs are highly variable in time and space. This means that the representativity of overseas-derived EFs to New Zealand is uncertain. Moreover, the few studies that have been undertaken in New Zealand tend to be Auckland-centric and may not be transferable to other regions.
4. Lack of information on the possible impacts of new technologies and changes in fleet composition on NEEs due to the limited number of studies into these impacts.

Variability

1. A lack of standard methods for determining EFs and uncertainties inherent in the sampling and analytical methods used and variability in the metrics reported.
2. Variability in the wear rates determined for brakes, tyres and roads due to variability in the composition and physical properties of these sources. Taken together, there is therefore variability in the chemical and physical properties of particles emitted by these sources.

Uncertainty

1. Limitations in the ability to apportion NEEs to specific sources due to inadequate tracers and multiple sources of contaminants. Other sources include deposition of aerosols from industry and burning of chromated copper arsenate treated wood in domestic heaters, weathering of galvanised (zinc) and copper roofing and galvanised street furniture, copper in fungicides, and sediment from roadside soils, litter and organic detritus such as twigs and leaves.
2. Uncertainties in the amount of resuspension of road dust and the inability to separate resuspended PM from newly emitted PM, leading to the possibility of double counting PM in the determination of EFs.
3. Inconsistencies in and lack of comparison of EFs estimated from wear rates and those estimated from air and water quality.

Recommendations for Stage 2

The objective of Stage 2 of the research programme proposed by Waka Kotahi is to fill the gaps identified in this report. To do this, we propose that Stage 2 should have two tranches. Tranche 1 will undertake (a) a re-analysis of existing air quality data held by GNS Science to apportion airborne PM from traffic to vehicles, and (b) a sensitivity analysis of selected air and water quality models that use EFs to get an understanding of

how the choice of EF affects model outputs. Based on the outcomes of these analyses, Waka Kotahi could decide to either retain the current EFs or undertake new monitoring and modelling to develop and test a new set of EFs for New Zealand in Tranche 2.

Abstract

Waka Kotahi NZ Transport Agency commissioned the National Institute of Water and Atmospheric Research (NIWA) to undertake a desktop study to provide updated information on non-exhaust emissions (NEEs) of contaminants from vehicles in New Zealand. The key concern is that the emission factors (EFs) previously determined are not representative of the current vehicle fleet, which has increased in size and has been modernising. EFs are used to help manage the effects of roads on roadside air quality and road runoff quality. The priority contaminants considered here are PM_{10} and $PM_{2.5}$ (ie, particulate matter with an aerodynamic diameter of 10 or 2.5 µm or less, respectively) for air quality, and zinc, copper and sediment for water quality.

This report is the outcome of the first stage of a two-part programme initiated by Waka Kotahi to provide new EFs for New Zealand. The report consists of a review of international and local literature to both present the current state of knowledge about NEEs and to identify gaps in this knowledge. The second stage will be to follow the recommendations made in this report to fill the gaps.

The gap analysis revealed that most of the knowledge on NEEs comes from research that was undertaken in the late 1990s and early to mid-2000s. The EFs reported are highly variable due to a combination of factors, including the high variability of the composition of materials used in the manufacture of brakes and tyres; spatial and temporal variability in road conditions, climate and driving styles; and the lack of standard methods to determine EFs. The flurry of reviews on NEEs published over recent years indicates that there has been renewed interest in NEEs and calls for new research to be undertaken.

On the basis of the gap analysis, we propose that Stage 2 of the research programme should have two tranches. Tranche 1 will undertake (a) a re-analysis of existing air quality data held by GNS Science to apportion airborne particulate matter from traffic to vehicles, and (b) a sensitivity analysis of selected air and water quality models that use EFs to get an understanding of how the choice of EF affects model outputs. Based on the outcomes of these analyses, Waka Kotahi could decide to either retain the current EFs or undertake new monitoring and modelling to develop and test a new set of EFs for New Zealand in Tranche 2.

1 Introduction

Waka Kotahi NZ Transport Agency commissioned the National Institute of Water and Atmospheric Research (NIWA) to undertake a desktop study to provide updated information on non-exhaust emissions (NEEs) of contaminants from vehicles in Aotearoa New Zealand. NEEs arise from all vehicles, regardless of type, and refer to particulate matter (PM) released by a vehicle due to the interaction between the vehicle's tyres and the road surface and from brakes due to braking-induced abrasion. NEEs can contribute to ambient air PM concentrations in cities as much as, or more than, tailpipe emissions (Amato et al., 2014; Grigoratos & Martini, 2015; Kukutschová & Filip, 2018) and, as combustion engines become more efficient and vehicles with alternative fuel sources (eg, electric, hydrogen) become more numerous, the proportion of NEEs compared to tailpipe emissions will likely increase. In a recent review, Piscitello et al. (2021) found that the reported ratios of NEEs to tailpipe emissions range between 10% and 40% for PM_{10} and between 15% and 65% for $PM_{2.5}$.[1] NEEs are also a major source of contaminants in road runoff (Müller et al., 2020; Sartor & Boyd, 1972). These contaminants include heavy metals, sediment, microplastics and organic compounds. Indeed, the UK Air Quality Expert Group (AQEG, 2019) estimates brake and tyre wear respectively contribute around 47% and 21% of the total copper and zinc emissions from all sources in the UK. Since NEEs affect both air and water quality, knowing the amount, physical characteristics and chemical composition of NEEs is essential for assessing the potential risks.

Emissions from vehicles are generally expressed as emission factors (EFs) that represent the mass of a contaminant emitted per vehicle kilometre travelled. Non-exhaust EFs can be estimated using two approaches. The first, called 'emissions from vehicles' in this report, measures wear from roads, brakes and tyres directly under controlled conditions to estimate EFs for different types of vehicles and driving conditions. The mass of PM emitted for a section of road is then calculated from the number of vehicles travelling along the road over a set period of time. The second approach, here referred to as 'emissions to air and water', measures PM in air, road dust and road runoff and then apportions the total mass of PM to brake, tyre and road wear based on the chemical and physical characteristics of the PM. Per vehicle emissions can be estimated by dividing the mass by the traffic volume. Theoretically, if traffic is the only source of PM and there are no losses, the PM from NEEs estimated for a section of road using either set of EFs will be the same under the assumption of conservation of mass.

Non-exhaust EFs are key inputs to models used by Waka Kotahi, such as the Vehicle Emissions Prediction Model (VEPM; Metcalfe & Peeters, 2020) developed by Waka Kotahi and Auckland Council, and the Road Stormwater Screening (RSS) model developed for Waka Kotahi (Gardiner et al., 2016; Semadeni-Davies & Moores, 2020; Semadeni-Davies et al., 2017). The VEPM estimates fleet-wide vehicle emissions, including both NEEs and tailpipe emissions. The model uses the method recommended by the European Environment Agency (2019) to estimate PM_{10} and $PM_{2.5}$ emissions from brakes and tyres (see section 3.5.1). The RSS model is a screening tool that assigns a level of risk to receiving waterbodies as a result of road and stormwater runoff. Among other factors, such as the sensitivity of the receiving environment and instream dilution, the risk level is based on the copper and zinc contaminant load from road traffic and non-road (urban) sources. Zinc and copper were chosen as water quality indicators in the model because these are priority metals for stormwater management. Metals from urban areas are estimated using a similar method to that used in the Auckland Council Contaminant Load Model (Auckland Regional Council, 2010b). There are three road EFs each for zinc and copper from roads; these represent free-flowing, interrupted, and

[1] 'PM_{10}' and '$PM_{2.5}$' refer to particulate matter with an aerodynamic diameter of 10 or 2.5 µm or less, respectively.

congested traffic flows, respectively, and are the same as those determined for traffic by Moores et al. (2010).

This study has been commissioned by Waka Kotahi in recognition that the national road vehicle fleet is increasing in size, changing in composition and technology, and these trends are likely to continue into the future. For example, while electric vehicles (EVs) make up less than 1% of the national fleet, there has been an exponential rise in the number of EVs since 2014 from around 1,000 EVs to 12,000 (Ministry of Transport, 2018), and it is anticipated that they will account for 40% or more of light vehicles by 2039/40 (Ministry of Transport, 2017). At the same time, it is expected that there will be changes in the component materials of tyres and brakes, partially due to regulatory changes in other jurisdictions (eg, Grigoratos, 2018). These trends mean that non-exhaust vehicle EFs reported in previous studies (eg, Kennedy et al., 2002; Moores et al., 2010) may no longer be valid for the current or future fleet. Indeed, a recent inventory of New Zealand stormwater research in relation to transport prepared by NIWA for Waka Kotahi (Moores, 2020) shows that much of the work on NEEs and contaminant load factors in New Zealand was undertaken in the late 1990s and 2000s. A report by AQEG (2019) came to a similar conclusion in a review of international literature on NEEs to air.

1.1 Report scope

To answer the need for updated NEE factors that reflect the current and future fleet, Waka Kotahi has proposed a two-stage programme whereby the first stage is a desktop study to identify and collate existing NEE data, determine gaps in that data, and recommend a methodology to address those gaps. The second stage will be to implement the recommended methodology in order to derive robust current and future estimates of EFs and contaminant loadings required for managing air and water quality.

This report documents Stage 1 of the programme. Specific tasks for Stage 1 are to:

1. review local and international literature and data sources to identify available information on EFs from non-exhaust sources and to evaluate this information to the New Zealand context
2. undertake a gap analysis of the information compiled in the literature review to determine what information is needed to update the EFs used in New Zealand
3. make recommendations on how to fill the identified knowledge gaps; these methodologies will form the basis of Stage 2 to be undertaken in a subsequent project.

Note that this project only considers emissions from sealed roads.

1.1.1 Priority contaminants

The priority contaminants considered in this report are airborne total suspended particulates (TSP) PM_{10} and $PM_{2.5}$ for air quality, and total and particulate zinc and copper and total suspended solids (TSS) for water quality.

PM_{10} and $PM_{2.5}$ are the most commonly reported air quality indicators cited in the NEE literature, largely because these are the main regulated species around the world. Particles in these size ranges can be breathed in and contribute to several major health conditions, including stroke, cardiovascular disease, and respiratory illness (Stafoggia & Faustini, 2018; World Health Organization, 2014). Exhaust emissions are in the fine fraction (< 2.5 µm) – in fact, mostly in the ultrafine fraction (< 100 nm) – and contribute very little to mass-based measurements such as PM_x (particles with an aerodynamic diameter of less than or equal to *x* microns) despite being believed to be responsible for many of the health impacts attributed to particulate pollution (Knibbs et al., 2011). NEEs tend to be in the coarse fraction (between 2.5 and 10 µm). Some recent research has investigated the emissions of ultrafine particles from NEE sources (eg, Kwak et al., 2014, and

references therein) but since ultrafine particles are generally measured as a particle number concentration, it is very difficult to relate them to mass-based measurements such as PM_x. Since ultrafine particles contribute very little to PM_x, we have not considered them further here. While some research has been done on the toxicity of components of road dust, such as metals, there is little evidence for health impacts and no suitable guidelines to compare against. For this reason, the toxicity of airborne NEEs is also not covered.

Sediment (usually reported as TSS) copper and zinc are commonly used to characterise stormwater and road runoff water quality in New Zealand. Sediment is a compulsory indicator of freshwater ecosystem health in the National Policy Statement for Freshwater Management (NPSFM; New Zealand Government, 2020). A key concern for stormwater management is the particle size distribution of sediments found in stormwater (Roesner et al., 2007; Semadeni-Davies, 2013) because finer particles generally have higher contaminant concentrations than coarse particles due to their greater surface area per unit of mass. We also note that finer particles are more likely to become resuspended to air, contributing to airborne PM. Copper and zinc are the most commonly reported metals associated with stormwater and road runoff and used as environmental tracers of NEEs (ie, copper is associated with brake wear and zinc with tyre wear). These metals can also indicate the presence of other metals in stormwater. While lead has been used to characterise road runoff in the past (eg, Williamson, 1993), lead from NEEs is very low, and sources from traffic are now largely historical (Huber et al., 2016; Kayhanian, 2012) due to its removal from petroleum products and as a vulcanising agent for tyre rubber. Other contaminants associated with NEEs include hydrocarbons, polycyclic aromatic hydrocarbons (PAHs) and other organic compounds. Microplastics from tyres is an emerging issue in freshwater management, as is noted in section 6.

1.2 Report structure

This rest of this report is broken into the following sections.

- Section 2 summarises the composition of the current fleet and Ministry of Transport predicted changes to the fleet over coming decades.
- Section 3 discusses the mechanisms for the generation of PM due to the wear or abrasion of brakes, tyres and road surfaces. It also presents information on the composition of these sources and the chemical and physical characteristics of PM from each source. The possible effects of changing technologies and regulation are also discussed.
- Sections 4, 5 and 6 discuss the derivation of EFs to air and water. Section 4 gives a general overview of the delivery of PMs emitted by NEE sources to air and water, and sections 5 and 6 discuss the apportionment to NEE sources of PM found in air and water, respectively.
- Section 7 is a synthesis of the gaps found in the previous sections, while section 8 gives recommendation for Stage 2 to fill the gaps.

In addition to these sections, there are three appendices:

- Appendix A gives detailed information on current and projected future fleet composition that is found in section 2.
- Appendix B gives information on the classification of road surfaces found in New Zealand.
- Appendix C lists publications compiled in the ‘Transport Knowledge Hub for Water – Research Stocktake’ prepared by Moores (2020) for Waka Kotahi that are relevant to the derivation of EFs for New Zealand.

2 New Zealand fleet current and future state

Changes to the size and composition of the New Zealand road fleet will have an impact on the amount and characteristics of NEEs. As was noted above, most of the research on vehicle emissions in New Zealand was undertaken in the early to mid-2000s when there were fewer vehicles on the road and EVs were not generally available. This section summarises statistics from the Ministry of Transport on the composition of the current New Zealand road fleet (Ministry of Transport, 2018) and the projected future state of the fleet over the next two decades (Ministry of Transport, 2017). More fleet information is provided in Appendix A.

For the purposes of this report, the road fleet has been split into three classes: motorcycles; light vehicles with a gross vehicle mass less than 3,500 kg (ie, cars, vans); and heavy vehicles with a gross vehicle mass more than 3,500 kg (ie, buses and trucks). No distinction is made between commercial and passenger vehicles under the assumption that the characteristics of these vehicles with respect to NEE will be the same. The report does not consider off-road vehicles such as tractors.

2.1 Current state

The number of registered vehicles per year between 2000 and 2018 is shown in Figure 2.1. While fleet composition remained fairly stable over this time period, the fleet size increased by 60%. In 2018, the road fleet consisted of approximately 4.29 million vehicles, of which 91.4% were light vehicles, 3.7% were heavy vehicles, and the rest were motorcycles (4.1%) and other miscellaneous vehicles, including off-road vehicles (0.8%). The distance travelled by the fleet in 2018 was 44,874 million vehicle kilometres travelled (VKT) for light vehicles and 3,074 million VKT for heavy vehicles.

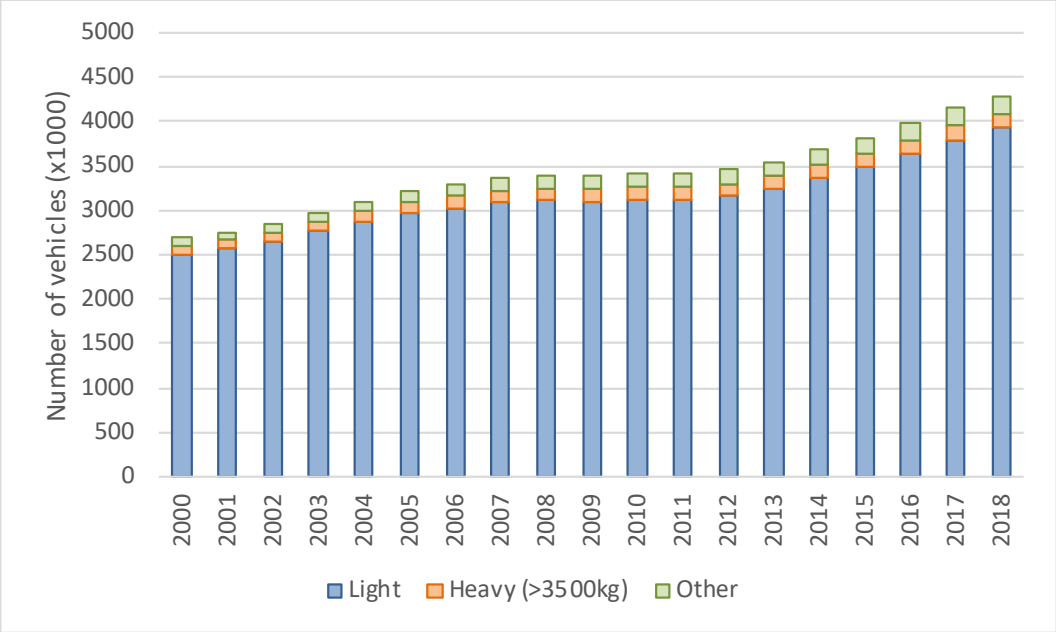
The average age of light vehicles in 2018 was 14.1 years, which the Ministry of Transport reports is several years older than light vehicles in other comparable countries and is largely due to the prevalence of imported used vehicles. The average ages for light vehicles cited by Ministry of Transport (2018) for the USA and Canada are 11.8 and 9.7 years, and the average age for light vehicles in Europe is 10.8 years.[2] Around 76% of light vehicles in New Zealand are less than 20 years old, and 31% were manufactured after 2010. For heavy vehicles, 55% are less than 20 years old and 28% were manufactured after 2010. This has implications for the direct relevance of NEEs derived from overseas fleets, which will overall be made up of newer vehicles, made of newer materials with less wear.

Comparing the Ministry of Transport online statistics for EVs[3] against the total fleet numbers from the Ministry of Transport (2018) shows that there are comparatively few EVs (including plug-in hybrids). Around 0.3% of light vehicles and 0.1% of heavy vehicles in 2018 were either pure electric or hybrid vehicles. The number of light EVs rose from just over 1,000 registered vehicles in 2014 to 12,300 in 2018. However, the increase in heavy EVs over the same period was minimal (from 74 to 80 vehicles).

[2] https://www.acea.be/statistics/tag/category/average-vehicle-age (date of access 30 November 2020)

[3] https://www.transport.govt.nz/statistics-and-insights/fleet-statistics/sheet/monthly-ev-statistics

Figure 2.1 Change in fleet numbers and composition since 2000



Note: Plot created using data from Ministry of Transport (2018).

2.2 Future fleet

According to the Ministry of Transport (2017), the key changes with respect to road transport over the coming decades are likely to be:

- increasing demand for road transport as a result of population growth, which is partially met by public transport in inner city areas and a shift in car ownership as ride- and vehicle-share schemes become more popular
- increasing use of EVs and adoption of new, more efficient vehicle technologies
- growth in road freight tonnage and new freight delivery technologies, including drones, robots and driverless vehicles.

The Ministry of Transport has projected future fleet numbers for five scenarios, which are listed and compared in Appendix A. Under the 'Base Case' projection scenario, which represents a business-as-usual continuation of current demographic and economic trends, the VKT is expected to increase by 51% for light vehicles (from 37.1 to 56.1 billion km) and 41.6% for heavy vehicles (from 2.9 to 4.1 billion km) by 2042 compared to 2012. The '@Home in Town and Country' (69.8 billion km) and 'Golden Triangle' (76.2 billion km) scenarios have the greatest projected increases in VKT. These scenarios assume greater economic and population growth than the Base Case scenario coupled with the adoption of new transport technologies.

Under the Base Case scenario, EVs are projected to make up around 40% of the national fleet by 2039/40 and account for 48% of the total VKT. The increase in the percentage of EVs is projected to be greatest for the @Home in Town and Country (45%) and Golden Triangle (49%) scenarios, and the increase for the other scenarios is very similar to the Base Case scenario.

3 Non-exhaust emissions from vehicles

This section overviews the chemical and physical characteristics of PM emitted by the wear of brakes, tyres and the road surface and presents examples of different approaches that have been used in New Zealand to determine EFs from these non-exhaust sources. While there are other non-exhaust sources of PM from vehicles, such as engine wear (nickel and chromium), paints (chromium, lead, iron, zinc) and wheel studs (tungsten), there is little information on these sources (Müller et al., 2020), and given the lack of reporting, it can be expected that their relative contributions to NEEs are low compared to brake, tyre and road wear. This section also discusses the possible effects of electrification of vehicles and regulation of materials used in the manufacture of brakes and tyres on the composition of NEEs. Finally, EFs derived from wear rates[4] and the composition of brakes and tyres that have been use in New Zealand are presented. Sections 5 and 6 discuss the apportionment of PM from NEE sources in the environment and EFs determined from environmental sampling of this PM.

The main mechanism for the emission of non-exhaust PM is abrasion due to friction and shear forces. A secondary mechanism is the evaporation of surface materials at high temperatures generated during contact releasing gaseous substances to the air. The amount and composition of NEEs from a vehicle is a function of the type and composition of the vehicle's brakes and tyres and the type and chemical and physical characteristics of the road surface (Figure 3.1). The relative importance of each source to the total PM emitted by a vehicle varies by vehicle type. NEEs are generally greater from heavy vehicles than light vehicles due to the higher weight overburden of heavy vehicles. Garg et al. (2000) estimate that the brakes of large cars emit 55% more PM than small cars, and the brake emissions from pick-up trucks are more than double those of small cars. Other factors influencing NEEs from a vehicle include driving conditions (eg, road slope, traffic congestion), driver behaviour (speed, braking habits) and climate. For example, freely flowing high-speed traffic is likely to result in lower brake emissions than slow, congested traffic, but will increase the amount of road and tyre wear (AQEG, 2019). For this reason, urban roads generally have higher emissions per VKT than rural roads due to the stop–start nature of urban driving, and emissions tend to be greater at bends and intersections.

3.1 Brakes

There are two main classes of vehicle brake: disc brakes and drum brakes. A third brake type, regenerative brakes, are used in EVs. These are discussed, along with other innovations, in section 3.3. Disc brakes, which are fitted to most modern passenger vehicles, consist of flat brake pads that are forced against a rotating disc. Drum brakes consist of a curved shoe that is forced against an outer rotating cylinder and when installed are placed on the rear axle of the vehicle. Both disc and drum brakes decelerate a vehicle by converting the kinetic energy of the vehicle to heat generated by the friction produced between the brake pad or shoe (ie, brake lining) and the disc or drum (ie, the brake rotor). This process results in abrasion of both the brake rotor and lining releasing PM that can fall to the road surface or become airborne.

[4] Here, 'wear rate' refers to the rate of material loss from NEE sources, whereas 'EF' refers to the mass of PM that is emitted to the environment. EFs can be reported as total emissions, which are analogous to wear rates assuming all PM generated is emitted, or apportioned into emissions to air or road dust.

Figure 3.1 Factors affecting the amount and composition of NEEs from vehicles

3.1.1 Brake types and composition

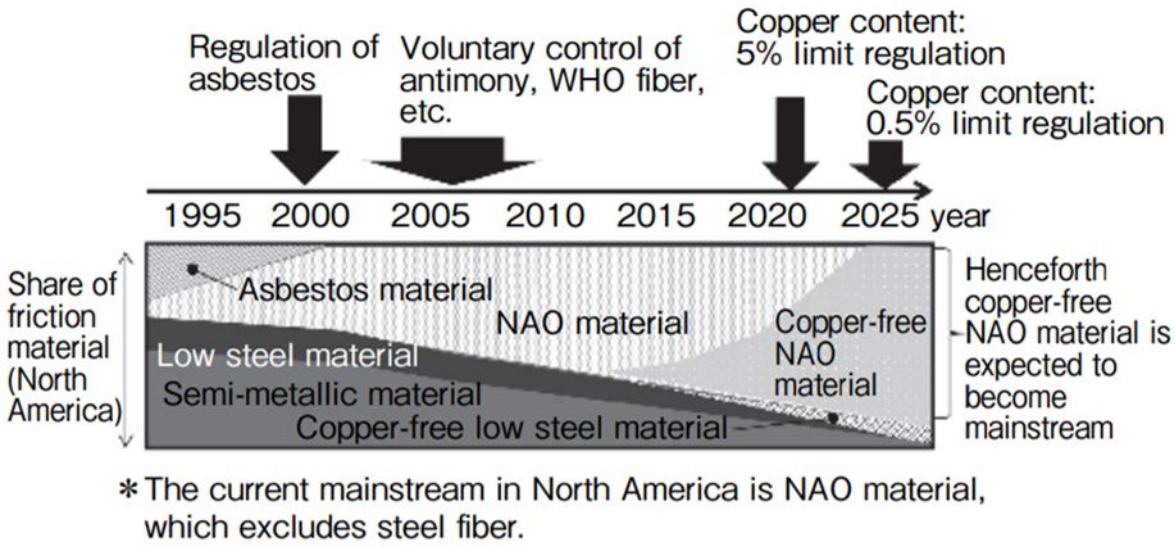
Brake rotors are usually made of grey cast iron or less often composites of carbon, ceramic or aluminium. Brake linings are generally made of four components: binders, fibres, fillers and friction modifiers (lubricants or abrasives). The lining components are listed, along with their purpose and common constituent materials, in Table 3.1. The proportion of the components and materials used vary between the type of lining and manufacturer. There are three main types of brake lining, which vary in their chemical composition and physical properties: non-asbestos organic (NAO), low metallic (LM) and semi-metallic (SM) brakes (Kukutschová & Filip, 2018).

The wide variation in the elemental composition of brake linings was demonstrated by Kennedy and Gadd (2000, revised 2003). They analysed the elemental metal contents of 12 brake pads that were available at that time in New Zealand. Copper, which is a key contaminant associated with brake wear and tear, for instance had a mass concentration ranging between 11 mg/kg and 112,000 mg/kg with a median of 35.5 mg/kg. Zinc had a similarly wide variation ranging from 25 to 34,500 mg/kg with a median concentration of 695 mg/kg.

More recently, Hulskotte et al. (2014) analysed the elemental composition of 65 brake pads and 15 brake discs from eight car brands common in Europe (Volkswagen, Opel, Ford, Citroen, Peugeot, Fiat, Volvo and Mazda) to develop a representative profile of brake composition for the Netherlands' car fleet. They found that iron, copper, zinc and tin make up around 80–90% of metals found in brakes, but the ratios of these metals vary between brands and models. Based on their analysis, they surmise that a typical brake pad contains about 50% non-metal components, 20% iron, 10% copper, 4% zinc and 3% tin. Brake discs, in contrast, consist almost entirely of metal – mostly iron (> 95%) with traces (< 1%) of other metals. There was no evidence of any relationship between the composition of the brakes tested and the car brands. Hulskotte et al. (2014) also analysed the carbon and sulphur content of brake pads and found that these made up around 50% of the non-metal composition.

Of interest here is how brake composition has been changing due to the impacts of new innovation and regulation. For example, Hagino (2020), citing Ito (2013), illustrated how brake pad materials are changing in North America due to regulation (Figure 3.2). These changes are discussed further in sections 3.3.1 and 3.4.1.

Figure 3.2 Changes in brake composition due to the regulation of chemical substances for brake pads



Source: Reprinted from Ito (2013, p. 47)

Table 3.1 Brake lining components

Component	Proportion of lining (%)	Purpose	Example materials
Binders	20–40	Bind the lining components together	Modified phenol-formaldehyde resins
Fibres	6–35	Reinforcement to provide strength	Copper, steel, brass, potassium titanate, glass, polymers and pulp (eg, Kevlar)
Fillers	15–70	Heat and noise regulation	Inorganic compounds (barium and antimony sulfate, magnesium and chromium oxides), silicates, ground slag, stone and metal powders
Lubricants (friction modifier)	5–29	Regulate the wear characteristics of the lining	Graphite is most common, but other materials include ground rubber, metallic particles, carbon black, and antimony
Abrasives (friction modifier)	0–10	Increase friction, maintain cleanliness between contact surfaces and limit the build-up of transfer films	Aluminium oxide, iron oxides, quartz and zircon

Source: Summarised from review by Grigoratos and Martini (2015)

Gadd and Kennedy (2000) assessed organic compounds in brake pads and found phenols to be one of the dominant groups of compounds, measuring ~ 5–1,300 mg/kg. Three phenolic compounds (phenol, hydroxyphenyl methylphenol and methylene bisphenol) were present in all six brake pads investigated. Two other antioxidant compounds (hydroxybenzaldehyde and benzothiazole) were also present in all samples but at lower concentrations (1.5–16 mg/kg) than the phenols.

3.1.2 Characteristics and emission factors of particulate matter from brakes

The EFs of PM from brakes can be assessed in the field or by abrasion testing in a laboratory. Abrasion testing is undertaken using either pin-on-disc tribometers or brake dynamometers. Pin-on-disc tribometers consist of a point or sphere fitted to an armature such that it is in contact with a sample placed on a rotating

disc. The pin scores into the sample as it spins, releasing PM. Different driving conditions (ie, driving programmes) can be simulated by varying the downward pressure and the speed of the disc rotation, and by stopping and starting the rotation at a specific rate of deceleration. Brake dynamometers consist of a housed rig fitted with a full-sized brake that is connected to a motor. These are generally used by the automotive industry to test the performance of brakes by measuring how much force is required to slow and stop the motor. PM studies measure and sample the particles generated by operating the brake according to a set driver programme. Field testing can range from collecting dust particles that have been retained by brake assemblies to instrumenting the brake on an operational vehicle.

Most of the studies we found for this review come from air quality literature and have therefore focused on the mass, size distribution and number of particles released to air from brake wear. Brake wear releases mostly micron-sized particles, which are generally reported in PM_x size bands; however, the number of bands and their size range varies between studies, with studies reporting unimodal, bimodal or multimodal distributions (Grigoratos & Martini, 2014, 2015).

Brake emissions from heavy duty vehicles are generally around an order of magnitude higher than those from light duty vehicles (see Tables 3.2 to 3.4). For example, the European Environment Agency (2019) found that brake EFs reported for cars (between 8 and 20 mg/veh/km) is less than those of heavy vehicles (between 29 and 110 mg/veh/km). Kukutschová and Filip (2018) further state that the amount of PM generated and the size range are dependent on the type of brake, the temperature generated and the driving conditions. NAO brakes contain higher proportions of organic compounds, mineral fibres, graphite and titanates and are relatively soft, with lower friction performance and higher sensitivity to temperature. These brakes produce more PM than other brake types. LM brakes contain a mixture of organic compounds and metallic materials and have higher friction performance and durability and good performance at high temperatures. SM brakes are predominately metallic, which makes them more durable; however, they have higher wear on the rotors than other brake types. The wear rates shown in the tables are highly variable. Most of the EFs for brakes cited in the literature are from the same set of studies undertaken in the early 2000s (eg, Garg et al., 2000; Sanders et al., 2003; Westerlund, 2001) with some more recent exceptions (eg, Hagino et al., 2015, 2016; Nosko & Olofsson, 2017; Perricone et al., 2016). What is interesting is that the tables attribute different emission ranges to some of the same studies, which is due to different metrics (eg, ranges, averages) being grouped by, for example, vehicle class, particle-size class or as total wear rates (ie, TSP) making it difficult to compare emissions reported in different studies directly.

The emission rates reported above have come from laboratory abrasion tests. In contrast, Wahlström et al. (2009) compared the number and volume distributions of airborne PM obtained from lab and field tests. The lab tests were undertaken using both a pin-on-disc tribometer and a brake dynamometer under clean conditions to avoid contamination of the PM. The field test was undertaken by instrumenting the right front brake of a passenger car to measure the temperature at the brake pad/rotor contact, car speed, the pressure of the pad on the rotor, and the number and mass concentrations of PM released by the brake. Samples of the airborne PM were collected by mounting test-tubes to the front of the car. To avoid contamination of the samples due to PM from other sources, the tests were undertaken at a driving circuit away from other traffic and following rainfall. All the tests were undertaken using disc brakes with grey cast-iron rotors and LM brake pads. The particles collected in each test were split into fine (< 1 µm) and coarse (1–32 µm) categories. They found good agreement between the concentration-normalised average particle sizes of PM collected from the tests. The number-weighted mean diameter of PM was around 0.4 µm for each test, and the volume-weighted mean diameter was 1.7 µm for the field test, 2 µm for the pin-on-disc, and 3 µm for the dynamometer.

In a later test, Wahlström et al. (2010) compared the size range of particles released by four each of NAO and LM brake pads using a pin-on-disc tribometer. In keeping with Table 3.4, the NAO brake pads had lower

wear than three of the LM brakes; the fourth LM brake behaved similarly to the NAO materials. They found that while the number of particles differed between the brakes tested, the size distributions had maxima at around 100, 280, 350 and 550 nm.

Alemani et al. (2016) investigated the effect of temperature on five LM discs and one NAO disc. They found that below a critical temperature (ranging from 165 to 190 °C for the different brake pad materials) fine particles outnumber coarse and ultrafine particles, although coarse particles make up the bulk of the PM mass. Analysis of the particle size distributions revealed peaks at 0.19–0.29, 0.9 and 1.7 µm. Above the critical temperature, ultrafine particles constitute almost 100% of the total particle number, and their relative mass contribution can exceed 50%. A further peak in the size distribution appears in the ultrafine particle range at 0.011–0.034 µm. Nosko and Olofsson (2017) similarly reported that particles < 10 µm are not present in brake wear PM below 200 °C while these ultrafine particles make up a substantial proportion of PM_{10} with brake temperatures over 200 °C.

Table 3.2 Summary of average brake wear rates for motor vehicles (mg/veh/km)

Vehicle type		Wear rates (mg/veh/km)	Reference
Passenger car	Small to large cars	11–17	Garg et al. (2000)
	Cars	16	Westerlund (2001)
	Cars	20	Legret and Pagotto (1999)
	Cars	21	US Environmental Protection Agency (1995b)
	Cars	25	Cha et al. (1983)
	Large cars	29	Garg et al. (2000)
Light duty vehicle	Light duty vehicle	29	Legret and Pagotto (1999)
	4-wheeled goods vehicle	84	Westerlund (2001)
Heavy duty vehicle	Heavy duty vehicle	47	Legret and Pagotto (1999)
Buses		110	Westerlund (2001)

Source: Adapted from Kennedy et al. (2002, p. 13)

Table 3.3 Brake PM wear rates used to derive EFs for the US Environmental Protection Agency's Motor Vehicle Emission Simulator (MOVES) emission modelling system (mg/veh/km)

Reference	Vehicle type	$PM_{2.5}$	PM_{10}
Sanders et al. (2003)	Light duty		1.5–7.0
Abu-Allaban et al. (2003)	Light duty	0–5.0	0–80.0
	Heavy duty	0–15.0	0–610.0
Westerlund (2001)	Light duty		6.9
	Heavy duty		41.2
Garg et al. (2000)	Light duty	2.1	2.9
	Large pick-up trucks	5.5	7.5
Rauterberg-Wulff (1999)	Light duty		1.0
	Heavy duty		24.5
Carbotech (1999)	Light duty		1.8–4.9
	Heavy duty		3.5
Cha et al. (1983)	Cars and trucks		7.8

Source: Adapted from US Environmental Protection Agency (2020a, p. 7; 2014, p. 4). These sources were in turn largely based on the review of brake wear by Luhana et al. (2004).

Table 3.4 Overview of studies quantifying the airborne fraction of direct brake wear particle emissions

Reference	Pad type	Type of test	Test procedure	Particle quantification	EFs
Garg et al. (2000)	SM, NAO	dynamometer	Wear test – General Motors BSL-035 50 to 0 km/h, deceleration rate 2.94 m/s^2, temp 100, 200, 300 and 400 °C	Mass of filters (TSP sampling resolved by MOUDI)	2.9–7.5 mg of PM_{10}/veh/km* 2.1–5.5 mg of $PM_{2.5}$/veh/km* 1.2–3.1 mg of $PM_{0.1}$/veh/km*
Sanders et al. (2003)	LM, SM, NAO	dynamometer	Urban driving programme max speed 90 km/h, deceleration rate < 1.6 m/s^2	Mass of filters (TSP sampling resolved by MOUDI and ELPI)	~8 mg of TSP/stop/brake (LM) ~2 mg of TSP/stop/brake (SM) ~2 mg of TSP/stop/brake (NOA)
Hagino et al. (2015)	NAO	dynamometer	Own urban driving programme max speed 60 km/h, deceleration rate < 3.0 m/s^2	Mass of filters (DustTrak and impactor)	0.006–0.016 mg of PM_{10}/braking/wheel
Hagino et al. (2016)	NAO	dynamometer	Wear test – JASO C427 max speed 50 km/h, deceleration rate < 2.94 m/s^2 Japanese exhaust emission/fuel economy tests (JC08 / JE05), max speed 90 km/h	Mass of filters (DustTrak and impactor)	0.04–1.4 mg of PM_{10}/veh/km 0.04–1.2 mg of $PM_{2.5}$/veh/km
Perricone et al. (2016)	LM, NAO	dynamometer	Modified wear test (SAE J 2707) max speed 100 km/h, deceleration rate 3.92 m/s^2, initial rotor temperature 100 to 200 °C	Mass of filters Number concentration (ELPI and cascade impactor)	14.5–46.4 mg of PM_{10}/stop/brake (LM) 8.5–9.2 mg of PM_{10}/stop/brake (NAO) 8–91 × 10^{10} number PM_{10}/stop/brake (LM) 153 × 10^{10} number of PM_{10}/stop/brake (NAO)
Nosko and Olofsson (2017)	LM	pin-on-disc	Own driving programme contact pressure 0.5–1.5 MPa, deceleration 0.8–1.6 m/s	Calculation of volume concentration based on size distribution (FMPS, OPS)	~ 200 µg of PM_{10}/m^3

* originally reported as mg/miles
Source: Adapted from Kukutschová and Filip (2018)
Note: SM = semi-metallic; NAO = non-asbestos organic; LM = low metallic; MOUDI = micro-orifice uniform deposit impactor; ELPI = electrical low pressure impactor; FMPS = fast mobility particle sizer; OPS = optical particle sizer

While most studies on brake wear, including those cited above, have investigated the wear on brake pads, the brake rotor material also has an impact on the characteristics of PM from brakes. Seo et al. (2021) used a brake dynamometer equipped with a particle counter and thermocouple to investigate the effects of four different disc materials (FC170 and FC250 grey iron discs, oxynitride-coated grey iron discs, and ceramic discs) on wear from an LM brake pad. They found that while the mass concentration of the PM was greatest for the FC170 and FC250 disc materials, the number of particles was greatest for the ceramic and oxynitride-coated grey iron discs. They also found that the mass concentration released by the brakes increased with the thermal conductivity of the disc. From this information, they conclude that large particles are produced at lower temperatures, and nano-sized particles are produced at higher temperatures. This finding is in keeping with those of Alemani et al. (2016) and Nosko and Olofsson (2017) cited above.

Relatively few studies have investigated the chemical composition of PM from brakes beyond choosing a suitable tracer for receptor modelling. While the most common elemental constituents of brake wear PM are iron, copper, barium and lead and organic carbon, there is little information on the organic compounds from brake wear (Grigoratos & Martini, 2015). Kukutschová and Filip (2018) note that the variety of brake compositions, the high temperatures generated by braking, and the complex mechanical and chemical interactions on the friction surface during braking make it difficult to predict the chemistry and particle size of PM from brakes.

In New Zealand, Kennedy and Gadd (2000) assessed metal composition of dust samples collected from the brake housings of six vehicles that had been brought to a repair shop. Elemental analysis of these samples showed a wide variation in metal concentrations that is in keeping with the variation in the metal composition described above. The copper content of the brake dust ranged from 75 to 1,980 mg/kg with a median value of 219 mg/kg and the zinc content ranges from 346 to 4,180 mg/kg with a median of 1,600 mg/kg. Kennedy et al. (2002) provided factors to calculate emission rates from the review of brake wear rates and chemical composition. For a passenger vehicle, this would equate to 74 µg/veh/km for copper and 24 µg/veh/km for zinc.

As noted above, Hulskotte et al. (2014) estimated EFs for metals from brakes through a combination of assessing the elemental composition of brakes and analysing road dust for copper and iron as indicator metals. From their own work, and citing other European studies, Hulskotte et al. (2014) state that the ratio of iron to copper in kerbside road dust is fairly stable, and they hypothesise that 70% of PM from brake wear comes from brake discs and 30% from brake pads. The metal EFs range from 5,750 to 10,781 µg/veh/km for iron, 147–546 µg/veh/km for copper, and 115–260 µg/veh/km for zinc.

The studies cited above show that the chemical composition, physical characteristics and EFs of PM from brakes are highly variable. This is largely due to the great variety of brake component materials available and the complex interaction between factors affecting brake wear. It has also been noted that adoption of low copper emission brakes and regenerative braking can change the composition and emission rates from brakes in the future (eg, Gramstat, 2018). In their reviews, Grigoratos and Martini (2015) and Kukutschová and Filip (2018) note that there is no international standard method for estimating brake EFs, and the variety of methods used and output indicators chosen means that it is not possible to make meaningful comparisons between EFs reported by different studies. However, there have been recent moves to standardise test procedures (eg, Mathissen et al., 2018).

3.2 Tyre wear and road wear

Particles from tyre and road wear are emitted as a result of abrasion of the tyre tread on the road surface and the fragmentation of road surface by vehicles. Particles from tyres can also be emitted as a result of fatigue due to oxidation and repeated deformation and thermal decomposition that occurs under extreme

conditions during braking and rapid acceleration (Kennedy et al., 2002). Particles from road and tyre wear cover the entire size range of airborne particles (European Environment Agency, 2019) and typically have an elongated sausage shape (Kreider et al., 2010). They consist of an agglomerate of rubber from the tyre tread, bitumen and minerals from the road and road dust (Adachi & Tainosho, 2004; Dall'Osto et al., 2014). For this reason, emissions from tyre wear and roads are discussed together in this section.

The factors that affect the rate of tyre and road wear include (Panko et al., 2018):

- tyre characteristics: size (radius, width), tread shape and depth, tyre pressure, contact patch area, temperature, chemical composition, tyre age (ie, tyre wear is slightly greater for new tyres as the angular edges of the tread become worn and rounded), new or re-tread, and alignment
- road surface type, camber angle, slope, condition, chip size (ie, roughness) and, where used, gritting materials for skid control during icy conditions
- driver behaviour, including speed, acceleration, frequency and extent of braking and cornering – Boulter (2005) and Councell et al. (2004), for instance, cite multiple studies that show increasingly aggressive driving will result in more rapid and uneven tyre wear than restrained driving
- vehicle characteristics such as weight, location of drive wheels, engine power, power or unassisted steering and suspension type and condition.

3.2.1 Tyre composition

Modern tyres are composed of an inner steel casing reinforced by steel belts covered by the outer rubber tread. Panko et al. (2018) state that while the proportion of materials in tyres change between makes and brands, the components remain consistent. The typical composition of a tyre is natural rubber (14%) and synthetic rubber such as styrene-butadiene (27%); fabric fillers, accelerators and antiozonants (16%) used in rubber manufacture; carbon steel wire (14%); and carbon black (28%) (3R Group Ltd, 2020). Zinc and PAHs found in tyre rubber are of particular concern due to their potential impacts on human health and the environment (Organisation for Economic Co-operation and Development [OECD], 2020). Zinc is added as zinc oxide during rubber manufacture as an accelerator in the vulcanisation process, and zinc makes up around 1% by weight of tyre rubber (Councell et al., 2004). The high zinc content makes tyres a key source of zinc in stormwater (Adachi & Tainosho, 2004; Councell et al., 2004; Legret & Pagotto, 1999; Müller et al., 2020).

The majority of materials used in the tyre tread are organic hydrocarbons; however, there can also be metallic components as well as zinc, including cadmium, copper and lead. Kennedy and Gadd (2000) determined the concentrations of trace elements in 12 tyres available in New Zealand at the time representing different vehicle types. They found that the zinc concentration ranged from 1,190 to 18,300 mg/kg (median 8,310 mg/kg) and was several orders of magnitude greater than for other metals. The priority pollutant metal with the next highest concentration was lead (median 2.72 mg/kg).

Panko et al. (2018) list a number of PAHs that have been found in tyre treads; however, the use of PAHs as extender oils has been banned in Europe since 2009. In their analysis of organic materials found in tyres, Gadd and Kennedy (2000) found one or more low molecular weight PAHs (acenaphthylene, phenanthrene, fluoranthene and pyrene) were present in five out of six of the tyres investigated at that time and benzo[g,h,i]perylene was present in one. Other organic compounds included nitrosamines (N-Nitrosodiphenyl and diphenylamine), anilines and benzidines, aromatic amines and benzothiazole and related compounds. The most abundant of these was N-(1,3-dimethylbutyl)-N'-phenyl-p-phenylenediamine (6PPD), which is a common antioxidant used in rubber manufacture and was present in all tyres at 1,050–1,800 mg/kg. Benzothiazole, used in vulcanisation of rubber, was also present in all tyres, at relatively

consistent concentrations (24–70 mg/kg). These concentrations are supported by a more recent study of benzothiazole in 17 tyres available internationally, measuring 21 to 180 mg/kg (Zhang, Zhang, et al., 2018)

3.2.2 Road composition

Sealed roads[5] consist of ballast or aggregate (ie, stone chips) bonded using bitumen or, less commonly, concrete. Roads can also contain furnace slag and recycled tyre rubber granules (3R Group Ltd, 2020; O'Donnell & Thomas, 2018; OECD, 2020). The combination of aggregate and bitumen means that PM from road wear is dominated by hydrocarbons and crustal elements, such as silicon, calcium, potassium, iron and aluminium (Lindgren, 1996).

According to the Waka Kotahi guidance on road surface noise control (NZ Transport Agency, 2014), most sealed roads in New Zealand have either chipseal (also known as spray-seal or surface dressing) or asphalt mix surfaces. Illustrations and definitions of the surface tyres are reproduced from NZ Transport Agency (2014) in Appendix B. Concrete roads are not common in New Zealand and cover less than 1% of the sealed road length in the country. Chipseal roads account for 89% of the sealed road length nationally and consist of a surface layer of aggregate embedded into a bitumen binder. Asphalt mix roads are blends of aggregate and bitumen that are combined and laid together while hot. Common types include asphaltic concrete, open graded porous asphalt, stone mastic asphalt and macadam. The different road surfaces have different void sizes (from no voids in chipseal roads up to 30% for porous asphalt) and different ranges of aggregate size (from 3 to 25 mm).

The Tyrewise Advisory Group (3R Group Ltd, 2020) report that around 5–8% of asphalt roads in New Zealand have been modified by the addition of rubber crumb to bitumen to give added surface life to cracked road surfaces. They suggest recycled rubber from spent tyres as a possible alternative to imported rubber crumb. They also report that tyre rubber and fibres are routinely added to road aggregates overseas and suggest the inclusion of rubber from tyres as a possible means of dealing with tyre waste in New Zealand as well as lessening road noise and improving road resistance to freeze–thaw weathering. The inclusion of rubber crumb to road surfaces has implications for determining the source of contaminants associated with tyre wear in NEEs.

In New Zealand, coal-tar binders, which are very high in PAHs, were also widely used for road construction until the 1970s; however, these are generally found in the subsurface layers (Depree & Fröbel, 2009) and are therefore less likely to contribute to road wear dust. Depree et al. (2006) investigated PAHs from roads in Christchurch, where roads originally sealed with coal-tar had been resealed with bitumen-based seal layers, to determine the source of PAHs associated with coal-tar that was persistently present in road runoff. They collected road and soil samples from five roads in the city and found that roadside soils that had historically been contaminated by the PAHs are the likely source, negating road wear as a contemporary source.

Gadd and Kennedy (2000; Kennedy & Gadd, 2000) analysed the organic and trace element composition from samples of raw (ie, prior to application to roads) and road bitumen. The organic compounds detected in raw bitumen are summarised in Table 3.5. They found that raw bitumen had relatively low trace element concentrations; however, higher concentrations were found in the road bitumen, which suggests that metals from other sources had been incorporated into the road surface.

[5] Emissions from un-sealed roads are outside the scope of this project and are not reported here.

Table 3.5 Summary of organic composition of a selection of raw bitumen containing no polymers in New Zealand (mg/kg)

Compound	Concentration
Acenaphthene	< 0.2
Acenaphthylene	< 0.2
Anthracene	0.2
Benzo(a)anthracene	1.0
Benzo(a)pyrene	< 0.2
Benzo(b)fluoranthene	< 0.2
Benzo(g,h,l)perylene	2.2
Benzo(k)fluoranthene	< 0.2
Chrysene	1.2
Dibenzo(a,h)anthracene	< 0.2
Fluoranthene	0.4
Fluorene	< 0.2
Indeno(1,2,3-c,d)pyrene	< 0.2
Naphthalene	0.3
Phenanthrene	0.3
Pyrene	2.1

Source: Adapted from Gadd and Kennedy (2000, p. 9)

3.2.3 Characteristics and emission factors of particulate matter from tyre and road wear

According to the recent review by Panko et al. (2018), most of the studies into the chemical composition of particles from tyres were carried out before 2005 (eg, Adachi & Tainosho, 2004; Cadle & Williams, 1978; Rauterberg-Wulff, 1999; Rogge et al., 1993). Generally, these studies used lab-based assessments of tyre wear, such as rolling resistance machines, to simulate different driving conditions to determine wear rates. Wear rates for tyres can also be calculated by determining changes in tyre weights or tread depth over time. The main components of tyre wear particles determined from these studies were plasticisers and oils, polymers, carbon blacks and minerals, which reflects the composition of tyres. However, amongst others, Adachi and Tainosho (2004), Dall'Osto et al. (2014), Kreider et al. (2010), and Williams and Cadle (1978) report that there is a difference in the composition of tyre particles emitted under lab settings and on roads due to both the effects of heat and friction and the incorporation of road dust and wear particles from roads.

Boulter (2005) suggests an average tyre wear factor of 100 mg/veh/km based on the service lifetime of tyres and rubber losses over that lifetime. Councell et al. (2004) compiled tyre wear rates reported in literature published between 1942 and 1996 and found they range from 1 mg/veh/km for gentle 'free-rolling' driving to over 1,000 mg/veh/km for severe driving; normal driving conditions have EFs in the range of 10 to 100 mg/veh/km. Kennedy et al. (2002) and the US Environmental Protection Agency (2020a) similarly reviewed literature on tyre wear rates based on direct studies and assessments of tyre lifetimes citing many of the same studies as Councell et al. (2004) and all dating from before 2003. Kennedy et al. (2002) report wear rates from 24 to 360 mg/veh/km for light vehicles with large differences for different route and driving conditions. The US Environmental Protection Agency (2020a) review reports wear rates ranging from 53 to 110 mg/veh/km for light vehicles and 136 to 1,403 mg/veh/km for heavy vehicles; average tyre wear rates for

non-specified vehicles range from 40 to 360 mg/veh/km. In their review of emissions from tyres, Panko et al. (2018)[6] caution that most of the estimates of wear rates originate from studies undertaken over 20 years ago, as is evidenced in the earlier reviews cited, and therefore are not representative of changes in technology designed to improve fuel efficiency and increase the longevity of tyres.

The wear rates reported above relate to particles of all sizes released by vehicles; however, many of the studies we found report only PM_x emissions to air. Councell et al. (2004) report that the range of tyre EFs for PM_{10} under normal driving conditions is between 10% and 30% of the total emissions from tyres. In their investigation of road wear rates (described below), Gehrig et al. (2010) found that tyres have low emission rates to air for fine particles; the PM_{10} emission rates to air for tyres were 17 mg/veh/km for heavy duty vehicles (assuming 10 wheels) and 0.80 mg/veh/km for light vehicles. Panko et al. (2018) set the range of reported PM_{10} emissions to air at between 2.4 and 13 mg/veh/km with an average of 6.3 mg/veh/km. They note that since measurements are sparse, $PM_{2.5}$ emissions from tyres are generally estimated as a ratio of PM_{10} emissions, making their reliability uncertain. Panko et al. (2018) also state that the differences in reported EFs for PM_x are due to a combination of different methods used to measure and model wear rates and inconsistencies in the metric reported. For example, some studies cited by Panko et al. (2018) report TSP, others suspended particulate matter[7] or PM_{10}. These differences make it difficult to directly compare results from different studies.

In his review, Gustafsson (2018) states that there have been few studies into particulate emissions from road wear outside the Nordic countries, where the main concern is the effect of studded winter tyres on road surfaces.[8] Since studded tyres are not used as winter tyres in New Zealand, the findings of these studies are not applicable to our driving conditions and are therefore not reported here. Most of the studies of road wear due to non-studded tyres have based on apportionment of airborne particles and road dust to road wear (see section 3.5). These studies have been hampered by the difficulty of separating resuspended (and deposited) particles from road wear particles leading to the possibility of double counting PM emissions from roads.

Gehrig et al. (2010) used mobile load simulators to determine PM_{10} EFs from road wear to air. This was done by analysing airborne particles extracted from the load simulators. As cited above, they also established tyre emissions to air using zinc as a tracer (assuming a 1% zinc content of tyres) in order to subtract PM_{10} from tyres from the total PM_{10} collected. The study did not investigate the chemical composition of the wear particles or EFs for larger particles or particles that were not airborne. Mobile load simulators are used by roading engineers to test the durability of road surfaces and consist of a large enclosed trailer within which wheels are pulled in a closed loop over a defined distance of road surface at a defined speed and with a defined axel weight. Two simulators were used, one each to simulate light and heavy vehicles. They conducted tests for two types of road surface with different surface conditions: asphalt concrete (new, good and poor condition) and porous asphalt (new condition). They found that PM_{10} emissions to air from asphalt in good condition are low, at around 3 mg/veh/km for passenger cars. However, the PM_{10} EF for a heavy vehicle on poor condition asphalt was 80 mg/veh/km.

Do et al. (2003) and Gothie and Do (2003) used a carousel to assess tyre and road wear for several different road surfaces under controlled conditions representing cornering. The carousel consisted of a circular road

[6] Only four of the 26 publications cited by Panko et al. (2018) have been published since 2010, and most date from before 2000.

[7] Defined by the OECD as finely divided solids or liquids that may be dispersed through the air from combustion processes, industrial activities or natural sources: https://stats.oecd.org/glossary/detail.asp?ID=2623#:~:text=Definition%3A,Nations%2C%20New%20York%2C%201997

[8] It should also be noted that sand and grit used as friction materials during icy conditions can increase the amount of PM generated over the winter period in cold regions.

track (19 m radius) with four central rotating arms each holding wheels fitted with truck tyres. The rotation speed, number of wheels, the yaw angle of the wheels and the applied load could be varied to represent different driving conditions. Gothie and Do (2003) assessed road polishing for three types of road surface typical of roads in France: 14 mm dense asphalt concrete (DA14), 10 mm very thin asphalt concrete (VTA10) and 6 mm very thin asphalt concrete (VTA6), where the measurements refer to the maximum aggregate size. Road wear was assessed by evaluating the texture and skid resistance of the road surface; the mass of emissions was not determined. They found that skid resistance is less marked for thin asphalt than dense asphalt and that the dense asphalt became more compacted and showed increased roughness due to the removal of aggregate by tyres. Do et al. (2003) assessed tyre wear for the same roading materials with different loads and yaw angles. Tyre wear was assessed by measuring tread groove depth at four points across the width of the tyres. Since neither the tyre composition nor physical properties (eg, weight, density, ratio of tread width to tyre width) were reported, tyre wear rates in mg/veh/km cannot be calculated from the data provided. Do et al. (2003) undertook two experiments, each starting with a set of eight identical new tyres. In the first experiment, the track was sealed with a 7 cm thick layer of DA14 asphalt, and in the second, half of the track was sealed with VTA10 and the other half with VTA6, both to a thickness of 2.5 cm. The change in tyre tread depth after 180,000 rotations (equivalent to 21,500 km) of the carousel for the two experiments is shown in Table 3.6. It was found that tyre wear is greater on the inner side of the tyre, and on average, tyre wear was greatest for the very thin asphalt, which has the ‘sharpest’ surface. It was also found that the change in tread depth is linearly related to the number of rotations, which suggests that emissions are constant with distance.

In contrast, the OECD (2020) generalises that coarse road aggregates result in lower emissions to air of PM from tyres and road wear. This has implications for New Zealand since around 90% of roads have chipseal. They cite several studies showing an inverse power relationship between aggregate size and emission rates to air and note that emissions are higher in drier conditions, leading to spatial and seasonal variations in PM emissions. For example, Padoan et al. (2018) developed an empirical relationship for the size of PM from tyre wear based on season, pavement macro-texture, traffic intensity, and proximity to braking zones based on data collected from Barcelona and Turin. They found that there is an inverse relationship between particle size and pavement macrotexture, traffic intensity, and proximity to braking zones. China and James (2012), however, state that lower emissions to air from coarse aggregates may be partially explained by aggregate chips shielding road dust, resulting in reduced resuspension of road dust. Taken together with the findings of Do et al. (2003) and Gothie and Do (2003), this explanation implies that the total emissions from tyre and road wear may not be reflected in the emissions to air, which has implications for determining emissions from road and tyre wear to water from roads with different roughness.

Similarly, porous paving has been shown to reduce emissions from vehicles to both water (eg, Moores et al., 2013; Zhang, Yong, et al., 2018) and air (OECD, 2020). However, this is likely due to trapping of road dust rather than reduced road or tyre wear per se, which is evidenced by the fact that porous paving requires regular maintenance to prevent clogging and loss of function over time (Chen et al., 2020; Sansalone et al., 2012; Yang et al., 2019).

Table 3.6 Change in tyre tread groove depth from new measured after 180,000 carousel rotations for two types of asphalt

Tyre	Overburden load (kN)	Yaw angle (°)	Change in tread depth (mm)			
			W (inner)	X	Y	Z (outer)
Experiment 1, DA14*						
A1*	30	−0.3	–	–	–	–
B1*	30	0.2	–	–	–	–
C1	35	0.4	1.826	0.853	0.598	1.126
D1*	35	−0.4	–	–	–	–
E1	35	0.6	2.453	1.4	1.25	1.942
F1	35	0.7	3.992	2.679	2.336	2.966
G1*	37.5	0.4	–	–	–	–
H1	37.5	0.4	2.031	1.091	0.791	1.429
Experiment 2, VTA10 and VTA6						
A2	30	0.7	2.81	1.83	1.623	1.937
B2*	30	±0.4**	–	–	–	–
C2	35	±0.4**	1.945	1.373	1.282	1.956
D2*	35	0	–	–	–	–
E2	35	0.7	4.473	2.715	2.403	2.916
F2	35	0.7	2.726	1.82	1.721	2.068
G2*	37.5	±0.4**	–	–	–	–
H2	37.5	0.5	1.97	1.49	1.422	1.933

* Missing results were due to technical problems with tread measurement.
** The yaw angle was changed alternatively between −0.4° and +0.4° after every tread depth assessment.
Source: Adapted from Do et al. (2003, p. 23)

There have been few studies into the chemical composition of road and tyre wear. As was noted above, particles from the two sources generally become amalgamated with road dust as they are emitted, making source apportionment difficult. Adachi and Tainosho (2004) used cluster analysis to apportion the sources of heavy metals found in PM from tyre and road wear. Four clusters were identified and ascribed to a potential source:

1. iron rich particles with traces of antimony, copper and barium from brake wear
2. chromium and lead from yellow paint used to mark roads
3. multiple metals, including antimony, copper, iron, zinc, tin, lead and barium from brake wear and heavy minerals
4. zinc oxide from tyre tread wear.

Kreider et al. (2010) collected on-road emitted particles as well as particles obtained from a simulated lab driving course and particles cryogenically ground from pieces of unused tread. They examined the morphology of the particles using a scanning electron microscope and found that the shape of the on-road particles and the lab driving course particles had a characteristic elongated ‘sausage’ shape while the cryogenically ground particles were irregular and angular, confirming earlier reports (Adachi & Tainosho, 2004; Cadle & Williams, 1978; Williams & Cadle, 1978). While the particle shape was similar, the

incorporation of road wear and road dust into the on-road particles can clearly be seen in the images. Furthermore, chemical analysis of the particles showed that the on-road particles have higher concentrations of trace elements, particularly silicon from soils and road ballast. They estimate that up to 50% of matter in tyre wear particles comes from other sources and has become integrated into the particle agglomerate.

The incorporation of road dust from sources other than vehicle emissions (eg, soil particles from atmospheric deposition or dislodged from vehicles) into particles generated by road and tyre wear has implications for the determination of EFs from road and tyre wear, as it could result in double counting EFs determined using receptor modelling (see section 5.1.2). It could also result in an overestimation of emissions because particles from other sources – which can have similar chemical composition, such as crustal minerals from roadside soil – are also present in agglomerated wear particles. While we found no information on the effects of adding tyre rubber crumbs to road aggregates on road wear, it can be speculated that this could result in the overestimation of emissions of hydrocarbons and zinc that are seemingly from tyres but actually originate from road wear.

3.3 New technologies

One of the main concerns of Waka Kotahi is that current EFs may not be relevant for the current or future fleet because of ongoing fleet modernisation and adoption of new technologies that were not available when the EFs were determined. The possible impacts of these changes on EFs are discussed in this section.

3.3.1 Reduced emission brakes

Gramstat (2018) and the OECD (2020) have evaluated advances in brake technologies aimed at reducing emissions from brakes. These are summarised in this section. Gramstat (2018) groups these advances into three categories:

- changing the composition and design of brake pads and discs
- development of collection systems to retain emitted PM so that it is not released by the vehicle
- avoidance of wear and particle emissions.

In the most general case, the first category replaces certain materials used in the manufacture of brakes with alternative materials, as is the case of low-copper brake pads (see section 3.4.1), or it adds additional materials to reduce brake wear, such as the addition of titanium or molybdenum to cast iron brake discs. The category also includes ceramic coatings on brake discs and ferritic nitrocarburising treatment to bond nitrogen to brake discs, harden the surface and reduce wear. Ferritic nitrocarburising treatment has wide market penetration for passenger cars in North America. There have been promising early results in using aluminium, ceramic or titanium brake discs; however, these alternatives are more expensive than grey cast iron, which remains the most common material for brake discs. Innovations for brake pads include the addition of a backing plate to improve brake function, the use of resins as binders, and a new generation of fibres to improve wear resistance and temperature stability as well as to reduce the impacts of PM on environmental and human health. Gramstat (2018) states that the costs involved and technological difficulties mean that most of these innovations are still in their infancy and need further development before they are marketable.

The second category is to capture emitted PM before it is released to the environment. While both Gramstat (2018) and the OECD (2020) have reported on efforts to design brake particle collection systems, there are no current commercial systems available, and their efficacy is uncertain, particularly for ultrafine particles. A key challenge for developers is the limited space in wheel-housings within which to place such as system.

Other challenges are developing a system that can operate at high temperatures and that is not overly noisy given the potential for vibration.

The final category includes regenerative brakes and driver-assistance systems. Regenerative brakes are used in EVs and slow the vehicle by converting and storing the kinetic energy that would be lost as heat in other braking systems. Regenerative brakes are insufficient on their own to stop a vehicle and are therefore used in conjunction with disc brakes. Gramstat (2018) states that regenerative brakes lower the friction required for braking and therefore the temperature profile of the brake, which could reduce emissions. While no wear rates or EFs from regenerative brakes are provided by either Gramstat or the OECD, emission models for EVs (see section 3.3.3) have been undertaken based on vehicle weight and assumptions about braking.

The effect of autonomous or driver-assistance vehicles on NEEs is also uncertain (AQEG, 2019; Kukutschová & Filip, 2018). On one hand, these vehicles are likely to be electric and will require additional weight due to sensors and computers needed to optimise driving to driving conditions, leading to increased EFs. On the other hand, the effects of regenerative braking and more regulated and controlled driving with less acceleration and deceleration could reduce emissions.

3.3.2 Low rolling resistance tyres

Low rolling resistance tyres are designed to improve fuel efficiency by reducing the energy loss as the tyre rolls. Low rolling resistance tyres are typically ‘blocky’ tread patterns with more contiguous and connected designs and can be narrower than other tyres, thus reducing emissions. The OECD (2020) states that there is no evidence that these tyres have lower wear rates than other tyres and that further research is required.

3.3.3 Emissions from electric vehicles

Given that driver behaviour and resuspension of road dust are likely to be similar for EVs and other vehicles, the key factors affecting NEEs are the weight of the vehicles and the use of regenerative braking. The effect of EVs on NEEs is largely unknown. There have been few direct comparisons of emissions from EVs and internal combustion engine vehicles (ICEVs) (AQEG, 2019; OECD, 2020), and these comparisons have been modelled based on assumptions on brake and tyre wear. We were unable to find information on zinc or copper emissions from EVs.

Smit (2020) states that there is little agreement in how the weight of EVs should be compared to other vehicles. It is generally accepted that EVs can be 20–25% heavier than similar-sized ICEVs (AQEG, 2019), which would increase tyre and road wear. Timmers and Achten (2018) compared the weights of popular models of EVs against the average for similar-sized vehicles and found that the EVs were between 7% (Smart Electric Drive) and 46% (Chevrolet Spark EV) heavier than ICEVs in the same size class. The exception is the Tesla Model S, which, depending on customisation, varies in weight between 2,036 and 2,314 kg, so some models weigh less than the ICEV size average (−1.8%) while others weigh more (11.6%). Timmers and Achten (2018) also compared EV and ICEV equivalent models from the same manufacturers and found that the EV versions are between 15% (Ford Focus) and 29% (Volkswagen Up, Chevrolet Spark and Smart Drive Coupe) heavier. However, Smit’s analysis of the Australian fleet found that EVs bought in Australia tend to be smaller models than other vehicles. By contrast, he notes that diesel-fuelled passenger vehicles, such as SUVs, are 25–30% heavier than smaller ICEV passenger vehicles. The implication is that while individual vehicles may be heavier than similar-sized vehicles, the shift to EVs could result in a higher proportion of small vehicles fleet-wide, reducing total emissions.

Timmers and Achten (2018) reviewed four studies that compared modelled NEEs from EVs and ICEVs. Of these, three (Soret et al., 2014; Timmers & Achten, 2016; van Zeebroeck & de Ceuster, 2013) report little or

no difference in PM_{10} emissions and only modest increases in $PM_{2.5}$ of 1–3%. However, they note that the source of the PM differs between EVs and ICEVs, with increased emissions from tyre wear due to increased vehicle weight balanced by reduced emissions from brakes due to regenerative braking, which will have an effect on the chemical composition of NEEs from EVs. The fourth study (Hooftman et al., 2016) modelled lower NEEs from EVs than ICEVs; however, Timmers and Achten (2018) dispute their model assumptions and state that the authors did not elaborate on their choice of model parameters or how the modelling was undertaken. In a side comment, Timmers and Achten (2018) suggest that drivers of EVs are more eco-minded and have better driving habits than ICEV drivers, which could further reduce emissions from EVs.

Using fleet data available in Australia, Smit (2020) developed two simple linear models to estimate NEEs from EVs and ICEVs. A Monte Carlo simulation was run that varied the parameters of these models to assess the range of $PM_{2.5}$ and PM_{10} that could be expected from EVs and ICEVs. The model outputs suggest that the EF for $PM_{2.5}$ from EVs could be between 1% and 34% (average 11%) of the EF for ICEVs. The modelled reduction for PM_{10} is between 1% and 46% (average 13%). However, Smit (2020) states that there is considerable uncertainty in the models.

The OECD (2020) review states that while EVs are estimated to emit 5–19% less PM_{10} than ICEVs, they do not necessarily emit less $PM_{2.5}$, and that the effect fleet-wide depends largely on context. For example, Ke et al. (2017) modelled several fleet composition scenarios with different proportions of EVs, estimating that electrification of 20% of light passenger vehicles and 80% of commercial passenger vehicles would result in a reduction of $PM_{2.5}$ emitted to air of between 0.4 to 1.1 $\mu g/m^3$. Beddows and Harrison (2021) modelled emissions by vehicle type (cars, light goods vehicles, rigid and articulated heavy goods vehicles and buses) – assuming that EVs are on average 300 kg heavier than equivalent ICEVs – and by road type (rural, urban, motorway), to compare fleet-wide emissions from EVs and petrol- or diesel-driven ICEVs for several scenarios, including the full electrification of the car fleet. They state that for any decreases in PM emissions from EVs (including tailpipe emissions – PM from NEEs is not reported separately) to occur, regenerative braking systems must be installed in EVs. This is because without regenerative braking, the reduction in tailpipe emissions is balanced by increased NEEs due to increased vehicle weight. With regenerative braking reducing brake wear, PM_{10} could be reduced by up to 26% for urban driving and 12% for rural driving where cars can drive at greater speeds. There were no reductions in PM_{10} emissions modelled for motorways. Furthermore, the modelling indicates that $PM_{2.5}$ could be reduced for all road types by between 1.9% and 27% by fitting regenerative braking to at least 90% of EVs. Beddows and Harrison (2021) caution that the methods used to derive the EFs for the different combinations of vehicle and road types, which were based on the European Environment Agency (2019) guidelines that have also been embedded in the VEPM (see section 3.5.1), are highly uncertain, largely due to the high variability in the values of model parameters reported in the literature.

3.4 Policy and regulation

3.4.1 Regulation of the composition of brakes and tyres

As noted above, there have been changes in the composition of brakes and tyres over recent years due to changes in technology and regulation (Hagino, 2020; Ito, 2013). Grigoratos (2018) has reviewed restrictions on the use of harmful material components in brakes and tyres through regulation. Substances that have been restricted include asbestos, trace elements and heavy metals in brake linings (see Figure 3.2 above) and PAHs in tyres.

Asbestos, which is carcinogenic and associated with respiratory disorders, was a common friction material in brakes due to its high temperature resistance and low wear properties. Asbestos has largely been eliminated

from brakes under national and international directives to reduce occupational exposure of workers involved in the manufacture and distribution of brake pads.

There are moves in some countries to ban or restrict the use of copper in brakes. In the USA, California and Washington State have both introduced legislation to restrict the use of asbestos and some metals from brakes to improve water quality (California Legislative Information, n.d.; Washington State Department of Ecology, n.d.). In California, the amount of chromium, lead, mercury and asbestos in brakes sold in the state cannot exceed 0.1% of the brake friction materials by weight, and cadmium is limited to 0.01% by weight. The Better Brakes Law in Washington State states that brakes manufactured or sold in the state after 2015 cannot contain asbestos, hexavalent chromium, mercury, cadmium or lead. Copper in brakes is being phased out in both states, and brakes with more than 5% by weight of copper will not be allowed to be sold in either state from January 2021. This limit will be reduced in both states to 0.5% after January 2025. The two states worked closely together and with environmental groups, manufacturers and retailers to draft their respective legislation. The Washington State Better Brakes website states that the US Environmental Protection Agency and the Environmental Council of the States have adopted standards for manufacture based on the California and Washington laws, which means that all brakes manufactured in the USA will meet Washington's restrictions on copper and heavy metals. While nickel, zinc and antimony are not covered by the legislation, they are being monitored by the states and may be restricted in the future. We have found no information on emissions from low-copper brakes at either the vehicle or fleet level.

We have been unable to find any studies into the efficacy of restricting copper in brakes with respect to water quality, which probably reflects the novelty of the restrictions. It is uncertain what the legacy of copper accumulation in roadside soils will be given that soils have been shown to accumulate metals, including copper (Hjortenkrans, 2008; Pagotto et al., 2001; Rodríguez-Flores & Rodríguez-Castellón, 1982). The persistence of copper in soils could mean that copper continues to be present in road dust.

While low-copper brake pads are available in New Zealand, and there have been calls from local and regional councils for restrictions of copper used in brake pads, a weekly ministerial report released by the Ministry for the Environment (2018) noted that such restrictions would be a significant undertaking and would require identifying and setting in place appropriate measures such as import controls and economic incentives. This is not to say that regulation is not possible, but that no decisions have yet been made. Correspondence between the Minister for Transport (Phil Twyford) and Environment Canterbury in 2020 suggests that restricting copper in brakes may be on option pending further investigation by the Ministry for the Environment (Helen Shaw, Environment Canterbury, pers. comm., June 2021).

The composition of tyres is largely unregulated with respect to concerns over PM toxicity, although Grigoratos (2018) and the Tyrewise Advisory Group (3R Group Ltd, 2020) note that there are regulations in Japan, Europe and the USA designed to minimise the risks to the environment and to human health as a result of end-of-life management (eg, in the production of tyre granules for use in rubber turf and asphalt). In the EU, there are restrictions in the use of some PAHs as extender oils in the production of tyres.

Without legislation, we can anticipate that low-toxicity brakes and tyres could become more common in New Zealand as demand for them grows internationally. However, it is also possible that remaining stock from countries with restrictions could be imported into New Zealand.

3.4.2 Transport demand control

As previously noted, the Ministry of Transport (2017) developed a set of five future transport scenarios, some of which include transport demand control to reduce fleet size and increase use of public transport. The OECD (2020) and Querol et al. (2018) list a number of regulatory measures designed to reduce traffic congestion and reduce greenhouse gas emissions that could also affect NEEs. These measures include:

- incentivising alternatives to the use of private vehicles by:
  - improving the comfort of and access to public transport
  - subsidising fares on public transport
  - construction of dedicated footpaths and cycling lanes
- introducing congestion charges, road tolls, fuel taxes, parking fees and distance-based road-user charges.

Changes to urban planning towards high-density housing close to workplaces and transport hubs could also have an effect on NEEs by reducing the need for commuting. Adoption of new technologies to allow people to work at home could also reduce the demand. Finally, legislation and incentivisation of EVs over ICEVs could further affect NEEs; however, as was noted in the previous section, there is little information available on NEEs from EVs.

3.5 Emission factors from vehicles used in New Zealand

This section overviews two methods used in New Zealand to estimate EFs for NEEs emitted by vehicles: the European Environment Agency (2019) method of estimating PM_{10} and $PM_{2.5}$ emissions to air, embedded in the VEPM, and the method developed by Kennedy et al. (2002) of estimating contaminant emissions from brake, tyre and road wear.

3.5.1 Emission inventories: European Environment Agency method

The European Environment Agency method estimates PM_{10} and $PM_{2.5}$ emissions from a section of road over a set time period based EFs determined for each NEE source, traffic numbers and fleet composition. The EFs used in the method are an example of an emissions inventory. An emissions inventory is a systematic assessment of expected emissions, broken down by industrial sector, emitting activity or emission source. Other examples include the National Atmospheric Emissions Inventory (2018) in the UK and the US Environmental Protection Agency (1995a) AP-42 Air Emissions Factors and Quantification Inventory. However, there has only been a small amount of New Zealand research, almost all undertaken in Auckland, to establish an emission inventory for New Zealand, which is why the VEPM uses the European Environment Agency method.

The European Environment Agency notes that the factors used in the method have been estimated from a small set of datapoints dating to the early 2000s. PM from road wear is not included in the calculations. Several reports cited in this review (eg, AQEG, 2019; Beddows & Harrison, 2021) have cautioned that the method is uncertain and needs to be evaluated and updated.

The TSP EFs and fractions used to split TSP into particle-size classes for cars and light trucks are given in Tables 3.7 to 3.9 below. The TSP EFs for heavy vehicles are calculated by adjusting the EFs for cars using the following statistical relationships determined by the European Environment Agency (2019, pp. 14–16).

Brakes:

$$EF_{TSP,B,HDV} = 3.13 LCF_B EF_{TSP,B,PC} \quad \text{Equation 3.1}$$

Where $EF_{TSP,B,HDV}$ is the TSP EF (g/km) for brake wear from heavy vehicles, LCF_B is the load correction factor for brake wear, and $EF_{TSP,B,PC}$ is the TSP EF for brake wear from passenger cars. The LCF_B is calculated as:

$$LCF_B = 1 + 0.79 LF_B \quad \text{Equation 3.2}$$

Where LF_B is the load factor and ranges from 0 for an empty load to 1 for a fully laden load.

The calculated EFs are corrected for speed by multiplying by a speed correction factor SCF_B to account for different wear rates from tyres under different speeds:

$$speed < 40\,km/h \;\; SCF_B = 1.67 \quad \text{Equation 3.3}$$

$$40\,km/h \leq speed \leq 95\,km/h \;\; SCF_B = -0.00270.speed + 2.75 \quad \text{Equation 3.4}$$

$$speed > 95\,km/h \;\; SCF_B = 0.185 \quad \text{Equation 3.5}$$

Here speed refers to the mean trip speed. The lower SCF_B at higher speeds reflects motorway and highway driving where braking is less frequent than for slower urban driving. The SCF_B is normalised to a speed of 65 km/h.

Tyres:

$$EF_{TSP,T,HDV} = \frac{N_{axle}}{2} LCF_T EF_{TSP,T,PC} \quad \text{Equation 3.6}$$

Where $EF_{TSP,T,HDV}$ is the TSP EF (g/km) for tyre wear from heavy vehicles, N_{axle} is the number of axles, LCF_T is the load correction factor for tyre wear and $EF_{TSP,T,PC}$ is the TSP EF for tyre wear from passenger cars. The LCF_T is calculated as:

$$LCF_T = 1.41 + 1.38 LF_T \quad \text{Equation 3.7}$$

Where LF_T is the load factor and ranges from 0 for an empty load to 1 for a fully laden load.

The SCF_T for tyres is calculated as:

$$speed < 40\,km/h \;\; SCF_T = 1.39 \quad \text{Equation 3.8}$$

$$40\,km/h \leq speed \leq 90\,km/h \;\; SCF_T = -0.00974.speed + 1.78 \quad \text{Equation 3.9}$$

$$speed > 90\,km/h \;\; SCF_T = 0.902 \quad \text{Equation 3.10}$$

The SCF_T for tyres is one at a mean trip speed of 80 km/h. It is noted that tyre wear decreases as the mean trip speed increases, which is probably due to more frequent braking and cornering on urban roads compared to highways.

Table 3.7 Brake TSP EFs for vehicle classes represented in the VEPM

Vehicle class	TSP EF (g/km)	Range (g/km)
Passenger cars	0.0075	0.0044–0.0100
Light-duty trucks	0.0117	0.0088–0.0145
Heavy vehicles	calculated	0.0235–0.0420

Source: Adapted from European Environment Agency (2019)

Table 3.8 Tyre TSP EFs for vehicle classes represented in the VEPM

Vehicle class	TSP EF (g/km)	Range (g/km)
Passenger cars	0.0107	0.0067–0.0162
Light-duty trucks	0.0169	0.0088–0.0217
Heavy vehicles	calculated	0.0227–0.0898

Source: Adapted from European Environment Agency (2019)

Table 3.9 Size fractions for PM from brakes and tyre wear in the VEPM

Particle-size class	Fraction of TSP	
	Brake wear	Tyre wear
PM_{10}	0.980	0.600
$PM_{2.5}$	0.390	0.420
PM_{1}	0.100	0.060
$PM_{0.1}$	0.080	0.048

Source: Adapted from European Environment Agency (2019)

3.5.2 Particulate contaminant emissions

Kennedy et al. (2002) used the findings of their analyses of the composition of brakes and tyres (Gadd & Kennedy, 2000; Kennedy & Gadd, 2000) to develop a methodology to calculate particulate inorganic contaminant emissions from traffic data for use in stormwater quality models. They state that more work was needed to refine a method for organic contaminants and that they were unable to find suitable wear rates for roads to provide input data into the EF calculations. The method was incorporated into the Ministry of Transport Vehicle Fleet Emission Model – Water (VFEM-W), which was subsequently used by Gardiner and Armstrong (2007) to identify catchments at risk from road runoff. The model was evaluated against EFs determined from road dust (Kennedy & Gadd, 2003), as described later in section 6.3. As far as we are aware, the VFEM-W is not currently available and has not been maintained.

The method calculates emissions separately for brakes and tyres and for each vehicle type using the following governing equation:

$$EF_C = \frac{VKT}{t} EF_P C_C \qquad \text{Equation 3.11}$$

Where EF_C is the contaminant EF for the vehicle type for a given section of road over a unit of time, t; VKT is the vehicle kilometres travelled; EF_P is the particle EF; and C_C is the contaminant concentration of the emitted particles.

The particle EFs for use in Equation 3.11 were developed from literature values (eg, Cadle & Williams, 1978; Garg et al., 2000; Legret & Pagotto, 1999; Rogge et al., 1993; Westerlund, 2001). The per vehicle particle EFs estimated for brakes are reproduced in Table 3.10 for different levels of braking. Table 3.11 gives the per vehicle particle emissions estimated for tyres for free-flowing, interrupted and congested traffic. Typical concentrations of trace elements in brake linings and tyre treads are given in Table 3.12 and were derived from the concentrations reported in Kennedy and Gadd (2000).

Table 3.10 Estimated particle emissions from brake wear (mg/km)

Vehicle class	Average wear rate	Low brake use	Moderate brake use	Intense brake use
Passenger care	21	10.5	31.5	42
Light duty vehicle	30	15	45	60
Heavy duty vehicle	80	40	120	160

Source: Adapted from Kennedy et al. (2002, p. 13)

Table 3.11 Estimated particle emissions from tyre wear (mg/km)

Vehicle class	Traffic flow	Number of axles				
		2	4	6	8	12
Two wheelers	Free flow	30				
	Interrupted	60				
	Congested	120				
Passenger cars	Free flow		60			
	Interrupted		120			
	Congested		240			
Light commercial	Free flow		60	90		
	Interrupted		120	180		
	Congested		240	360		
Medium commercial	Free flow		102	153	204	
	Interrupted		204	306	408	
	Congested		408	712	816	
Heavy duty	Free flow			630	840	1,260
	Interrupted			1,260	1,680	2,520
	Congested			2,520	3,360	5,040
Buses	Free flow			630	840	1,260
	Interrupted			1,260	1,680	2,520
	Congested			2,520	3,360	5,040

Source: Adapted from Kennedy et al. (2002, p. 37)

Table 3.12 Estimated concentrations of selected elements in New Zealand brake pads and tyres (mg/kg)

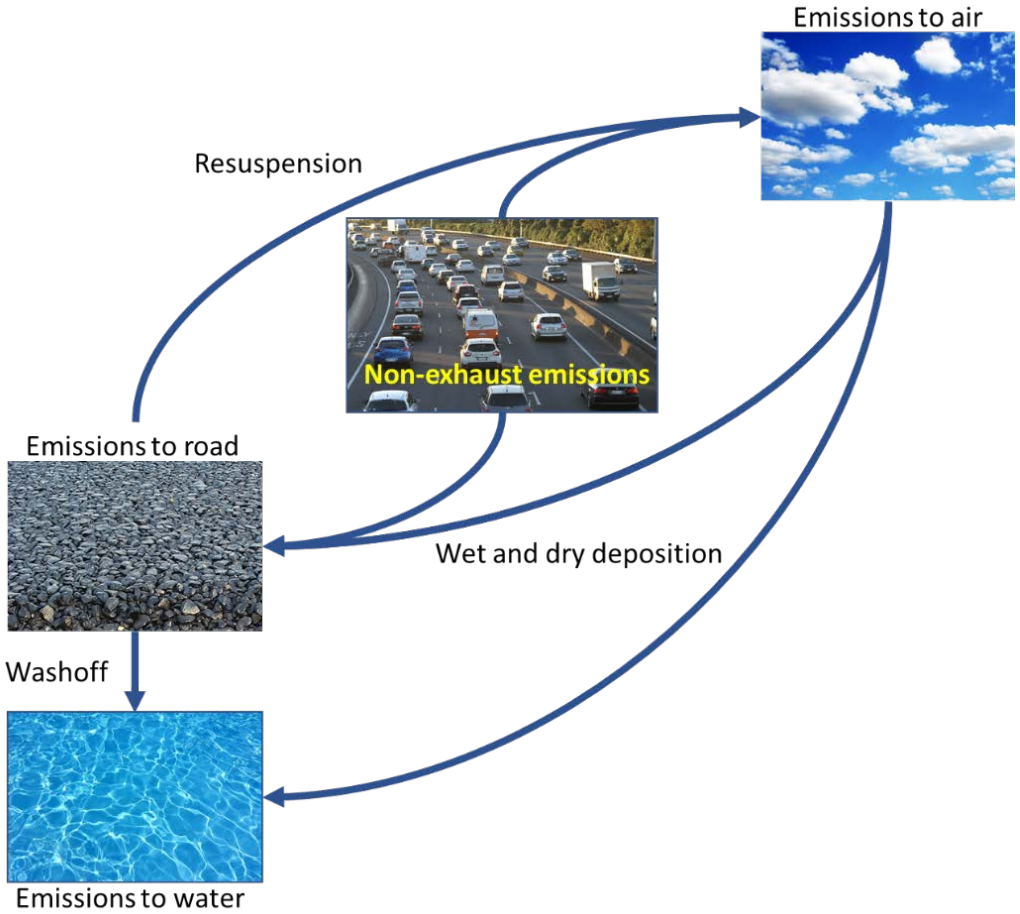
Element	Brakes			Tyres
	10th percentile	Median	90th percentile	Median
Priority pollutant elements				
Ag – silver	0.2	0.2	1.19	< 0.2
As – arsenic	1	1	17.2	–
Cd – cadmium	0.1	0.6	4.6	0.19
Co – cobalt	11.8	24.8	40.2	–
Cr – chromium	41.1	233	435.8	< 1.0
Cu – copper	29	280*	116,000	1
Hg – mercury	0.1	0.2	0.3	< 0.01
Mo – molybdenum	0.7	3.6	136.9	1
Ni – nickel	44.2	342	652.2	1
Pb – lead	5.6	50	949.2	2.7
Sb – antimony	0.7	6.1	158.8	< 0.2
Sn – tin	0.5	1.5	5.6	< 0.5
Zn – zinc	127.2	1,630	37,400	8,310
Other elements				
Al – aluminium	812.4	3,100	17,140	–
Fe – iron	1.2	1.8	19	105
Mn – manganese	143.2	315.5	1,087.6	–
Ti – titanium	32	110	988	–
B – boron	12	47.5	869	–
Ba – barium	558.2	3,195	6,144	26

* Kennedy et al. (2002) report that this value may be an underestimate of the actual concentration.
Source: Adapted from Kennedy et al. (2002, pp. 21, 38)

4 Non-exhaust emissions found in air and water

NEEs from vehicles can be released to air, fall to the road surface, or be retained by the vehicle (Figure 4.1). Airborne PM can be held aloft, transported to other locations or deposited onto the road surface. Dry deposition or settling is a function of the particle size, shape and density and the microclimate of the air parcel (eg, temperature, wind speed, convection currents). Wet deposition refers to the washout of particles in the air during rainfall. PM accumulated on the road surface can be resuspended by wind and traffic[9] or washed off during rainfall. The apportionment of NEEs reported in the literature varies greatly, as discussed in sections 5 and 6. For example, it is estimated that between 30% and 90% of particulates from brakes are emitted as airborne PM_{10} (European Environment Agency, 2019; Grigoratos & Martini, 2015). The fate of PM from NEE sources depends on the size range, shape and weight of the particles, driving conditions, driver behaviour, road type and condition, and microclimate. Under the assumption of conservation of mass, the total emissions determined from PM sampled in air, road runoff and road dust from a specific section of road should be the same as those emitted by vehicles travelling over the section of road as determined in section 3.

Figure 4.1 Schematic overview of the apportionment of NEEs to air and water



[9] In addition to physically lifting particles, traffic can cause road dust to fragment into smaller particles that can become airborne.

The general method of determining EFs from measured air or water quality is to establish the total mass (or load) of the target contaminant sampled from a stretch of road over a set time period and then divide the load by the number of vehicles travelling over the road during the time period. For air quality, the mass of PM is either the mass captured on the filter of the PM sampler or the total mass sampled by a continuous air quality sampler. For water quality, the total load available for wash-off during a rainfall event is equivalent to the accumulated mass of road dust on the road surface, and the load transported in runoff is the product of the contaminant concentration in the runoff and the runoff flow rate.

Apportioning the contaminant to an NEE source requires an extra step that relates the contaminant to the source based on the physical and chemical characteristics of the contaminant. For example, copper, barium, and antimony can be traced back to brakes and zinc to tyres. In either case, the method used to determine EFs assumes that NEEs are the only source of the contaminant, which is not generally the case in urban areas where other sources include deposition of aerosols from industry and burning of chromated copper arsenate treated wood in domestic heaters (copper, chrome, arsenic); weathering of galvanised (zinc) and copper roofing and galvanised street furniture; copper in fungicides; and sediment from roadside soils, litter and organic detritus such as twigs and leaves (eg, Müller et al., 2020; Murphy et al., 2015; Murphy et al., 2014; Wicke et al., 2012). For this reason, the choice of site for air or water quality sampling needs to be carefully considered to isolate the site from these sources. The methods for determining EFs specific to air and water quality are presented in the following two sections respectively.

5 Emissions to air

Determining EFs to air is essential for air quality management because they are used as input into air quality models. EFs can be derived from wear tests (see section 3); emission inventories (as illustrated in section 3.5.1); or by air quality monitoring, as discussed in section 5.1 generally and in section 5.2 with respect to resuspension. The can also be calculated from emission inventories (as illustrated in section 3.5.1), which can contain data from both wear tests and air quality monitoring. EFs derived from all three methods internationally are presented in section 5.3. The New Zealand context is discussed in section 5.4.

5.1 Methods for deriving non-exhaust emission factors from air monitoring

Generally, deriving EFs from measured air quality relies on two stages of work: firstly, a field campaign and/or laboratory work to collect air quality samples and analyse them for composition; secondly, a statistical analysis to apportion the derived concentrations of pollutant to various sources. To calculate EFs from vehicles, another analysis is required that determines the relationship between these emissions and the type and number of vehicles that generated the emissions. In this section, we outline the methods used to derive non-exhaust EFs and give literature examples of these.

5.1.1 Field measurement techniques

Numerous experimental techniques have been used to measure vehicle-generated air pollution (see review in Luhana et al., 2004), some examples of which are presented below.

Claiborn et al. (1995) used tracer techniques to estimate PM_{10} resuspension from paved and unpaved roads. In this study, a known amount of an inert tracer (SF6) was released and concentrations of PM_{10} and SF6 downwind of the road were measured along with meteorological parameters and traffic volume. The results of the tracer experiments showed that within experimental uncertainties, the PM_{10} and the tracer gas disperse in the same manner, suggesting that the use of a tracer in a line source to simulate roadway PM_{10} emissions can provide a tool for improving the existing emission inventories from roads.

Attempts to directly measure resuspension from road surfaces have involved the use of instrumented vehicles. Etyemezian et al. (2006) and Etyemezian et al. (2003) used a system known as TRAKER (Testing Re-entrained Aerosol Kinetic Emissions from Roads). This system uses three inlets: one behind each front wheel to measure PM_{10} emitted by each wheel, and one out the front bumper to derive the 'background' PM_{10} count, which was subtracted from the wheel arch measurements. Since the device only measures the contribution of the front wheels to resuspension, the outputs need to be converted to resuspension rates for the vehicle as a whole.

Measuring air quality on both sides of the road in order to undertake upwind–downwind comparisons has been reported by Lohmeyer et al. (2004) and Gehrig et al. (2004). Lohmeyer et al. (2004) derived non-exhaust EFs by measuring PM at both sides of a road in Karlsruhe, Germany, while Gehrig et al. (2004) measured at both sides of the road in several locations in Switzerland. Gehrig et al. (2004) used a roadside–background comparison and simply interpreted PM_1 as direct exhaust emissions and PM_{10}–PM_1 as mechanically produced emissions from abrasion and resuspension processes. In New Zealand, Longley et al. (2013) and Longley et al. (2015) used roadside PM measurements, along with measurements from upwind and downwind sites to derive estimates of the road contribution to ambient PM concentrations.

Road tunnels can be used as in-situ, 'live' laboratories for isolating vehicle emissions from other contaminant sources. In the UK, Lawrence et al. (2016) and Lawrence et al. (2013) took 12-hour filter samples from the

entrance and exit of a single-direction motorway tunnel to derive an incremental change in concentrations of both organic and non-organic species. Earlier studies include Sternbeck et al. (2002) in Sweden, Lough et al. (2005) in the UK and Fabretti et al. (2009) in France. In New Zealand, Davy et al. (2011) took single filter samples every three hours in and near a motorway tunnel and used elemental analysis to derive concentrations.

5.1.2 Statistical techniques for apportionment

The techniques for apportioning PM from vehicles to air generally fall into three categories[10]:

- elemental tracing
- chemical mass balance
- receptor modelling using multivariate regression techniques (eg, Boulter, 2005).

All three methods involve some form of initial chemical speciation, such as proton-induced X-ray emission, X-ray fluorescence) or atomic absorption spectroscopy, or molecular/ionic extraction, such as ion chromatography, gas chromatography–mass spectroscopy or Soxhlet, to determine the chemical characteristics of the PM.

Elemental tracing relates PM to its source based on a single chemical tracer associated with the source such as zinc for tyres and copper from brakes. For example, if it is assumed that the zinc content of tyre rubber is 1% (Councell et al., 2004), assuming that there are no other sources of zinc, the mass of total emissions from tyres will be 100 times the mass of zinc sampled. The method requires that the source composition is known for all the sources and assumes that the sources specified are responsible for all the PM within that trace; however, removing the background concentrations of the tracer can mitigate the signal from other sources. Harrison et al. (2012) used an elemental tracing approach to determine the sources of PM sampled from air and kerb-side road dust along Marylebone Road, London, based on the chemical composition and particle size range of the particles. They estimate that 38.1 ± 9.7% of the airborne PM was resuspended road dust, 55.3 ± 7.0% was from brakes and 10.7 ± 2.3% was tyres. More recently at the same location, Hicks et al. (2021) used an elemental tracing approach to estimate changes in brake and tyre wear and resuspension, indicated by barium, zinc and silicon respectively. Monitoring was initiated in September 2019 and continued to August 2020 allowing the authors to assess the effect of the Covid-19 lockdown period from March 2020. Lockdown saw a 32% decrease in traffic volumes and a 15% increase in average vehicle speed resulting in lower PM_{10} and $PM_{2.5}$ emissions. While brake emissions were also decreased, brake wear remained the single highest NEE source. The EFs for road and tyre wear were similar to pre-lockdown levels. It was noted that there may be some seasonal differences in emissions between the pre-lockdown (autumn and winter) and lockdown periods (spring and summer), which could also have affected EFs. The authors concluded that the results demonstrate the complexity of determining EFs.

Chemical mass balance is similar to elemental tracing but assigns PM to different sources by assuming that each source emits a characteristic set of multiple chemical species in a fixed ratio. Again, it is assumed that the source composition is known and that each source is responsible for all the PM with the trace characteristics. While chemical mass balance can be applied to NEEs, it has been predominantly applied to tailpipe emissions based on organic molecular tracers. For example, Schauer et al. (1996) used organic compounds as molecular markers to identify a range of sources contributing to local aerosols in the Los Angeles area. Sources identified included diesel exhaust, tyre wear debris, paved road dust, organic detritus, natural gas combustion aerosol, cigarette smoke, meat grilling and frying, catalyst and non-catalyst gasoline-

[10] We were unable to find examples of receptor models for determining the contribution of contaminants from NEEs to water.

powered vehicle exhaust, and wood smoke. Abu-Allaban et al. (2003) used chemical mass balance based on the methods developed by Rogge et al. (1993) and Hildemann et al. (1993) to identify brake and tyre wear in Nevada and North Carolina. They found that resuspended road dust and exhaust emissions were the dominant contributions to total PM_{10} and $PM_{2.5}$. Small contributions from brake wear were observed at places where strong braking takes place (eg, off-ramps) but they could not find evidence of tyre wear at any location.

Multivariate receptor modelling seeks to determine the relationship between the PM found at a specific location and factors that affect PM such as traffic numbers and fleet composition, climate, road type and presence of corners and junctions. Almeida et al. (2006) carried out receptor modelling using principal component analysis and multilinear regression analysis. By comparing weekdays to weekends, they were able to identify a component consisting of crustal material that they attributed to resuspended road dust caused by changes in traffic volumes.

Furusjö et al. (2007) used positive matrix factorisation for receptor modelling (see, for example, US Environmental Protection Agency, 2020b) to identify five components of roadside PM in Stockholm for an urban canyon and a highway. The components were resuspension; vehicle derived (brake and tyre); road salt; regional combustion; and long-range transport of PM from sources outside Scandinavia. Vehicle exhaust was not detected as a separate source and was not identified as a major source for PM_{10}. They found that during the spring months for the urban canyon, as much as 74% of PM_{10} is due to resuspension, whereas at an open highway site, long range transport was the dominant source. The vehicle source was only of major importance at the urban roadside, where it frequently contributed between 10 and 20 $\mu g/m^3$. The difference in the importance of vehicles to emissions between the sites was speculated to be due to the urban canyon restricting air movement and trapping PM, whereas PM at the highway site was regularly transported away by wind. This movement of air masses is also the reason given for the high importance of long range transport at the highway site. They also found that brake wear often contributes over 20% of the PM_{10} at the street canyon site but typically less than 5% at the highway location. In fact, although the traffic signal was significant, Furusjö et al. could not identify vehicle exhaust as a separate source.

Jeong et al. (2019) used positive matrix factorisation and traffic-related metals as tracers to investigate local-scale spatial variation of emissions in Toronto. They found that traffic-related $PM_{2.5}$ sources were mainly from exhaust emissions (9–19% of $PM_{2.5}$) and NEEs including brake wear (2–6% of $PM_{2.5}$) and resuspension of road dust (3–4% of $PM_{2.5}$). The traffic-related sources exhibited strong diurnal and spatial variabilities, while other $PM_{2.5}$ contributors (eg, oxidised organic aerosol and secondary sulphate) did not. They also found considerable spatial variation and that nearly one-third of the traffic-related source contributions were associated with NEEs from brake wear and road dust resuspension in urban areas. Elevated levels of non-exhaust sources were correlated with the number of heavy duty vehicles, rather than total traffic volume, although they still conclude that more research is needed to estimate non-exhaust EFs based on fleet composition.

There are a number of challenges to apportionment. First and foremost, choosing an appropriate tracer to use is challenging given the wide variety of materials and their relative proportions found in the composition of brakes, tyres and roads and the presence of other non-traffic-related sources. For this reason, other metals, notably barium and antimony, have been used as alternatives to copper for tracing wear from brakes (eg, Grigoratos & Martini, 2014; Iijima et al., 2008; Iijima et al., 2007) since there are few other environmental sources of these metals. Another issue is resuspension (discussed below), which can lead to double counting. Finally, apportionment studies are hampered by the fact that road types, driving conditions and the surrounding land use can change over short distances (Thorpe & Harrison, 2008) so that the relationships found for one section of road may not be valid elsewhere. Moreover, even at the same location, seasonal

changes in weather and changes in driving habits and congestion, such as was demonstrated by Hicks et al. (2021) for London under Covid-19 lockdown, can affect EFs.

5.2 Resuspension

Resuspended road dust can be a significant source of PM to air (Boulter, 2005; Gehrig et al., 2010) and is generally considered a secondary vehicle-derived pollutant. Generally (eg, Gehrig et al., 2004), at sites with relatively undisturbed traffic flow they are in the same range as the tailpipe emissions, whereas at sites with disturbed traffic, such as corners and intersections, emissions from abrasion and resuspension can be higher than those from the exhaust pipes.

Although the mechanism that drives the emissions are vehicles interacting with the road surface, the composition of the pollutant is a mixture of primary sources that can have similar chemical and physical characteristics. This presents unique challenges in identifying and quantifying resuspension (Amato et al., 2014; Gustafsson, 2018; OECD, 2020; Panko et al., 2018; Thorpe & Harrison, 2008). For example, it is extremely difficult to separate resuspended road dust and newly emitted tyre and road wear, since road dust, which itself contains previously deposited NEEs as well as roadside soil particles that have been transported to the road surface, can be aggregated into the tyre and road wear particles. For this reason, resuspended particles are often considered together with tyre and road wear in apportionment studies.

Another problem with deriving EFs for resuspension is that the road dust reservoir is not constant. Generally over time, an equilibrium level of resuspension is reached where the mean rate of removal by resuspension will be equal to the deposition rate of airborne particles onto the road surface. Furthermore, Gehrig et al. (2004) suggest that a small number of passing cars may keep road dust in suspension so that resuspension is not increased by a higher number of cars. This means that, unless the input of material onto the road surface is directly related to traffic density, the amount resuspended is not proportional to the number of vehicles passing a given point. Therefore, it is very difficult to establish resuspension as a function of the vehicle distances travelled (Nicholson, 2001).

A third challenge is that resuspension rates vary between wet and dry conditions leading to both spatial and temporal variation depending on local climate and season. Padoan et al. (2018), for example, found that resuspension and emission rates from road and tyre wear are higher in summer. Amato et al. (2013) reported lower emissions to air following rain and found that emission rates recovered more quickly in hot, dry climates (Barcelona) than cool, temperate climates (Utrecht). The lower PM concentrations in air associated with cooler, wetter conditions are due to a combination of wash-off of accumulated road dust depleting the pool of road dust for resuspension, washout of PM in air and caking of remaining road dust on the road surface so that the road dust cannot be resuspended.

5.2.1 Use of AP-42 for estimating resuspension

The US Environmental Protection Agency (1995a) AP-42 method for estimating resuspension is widely used for apportionment studies (Piscitello et al., 2021). The method calculates the resuspension EF for paved roads[11] using an empirical relationship between (a) the emissions of $PM_{2.5}$ and PM_{10}, respectively; (b) the mass of silt in road dust per unit area of road surface (silt is defined in the method as PM less than 75 µm in diameter); and (c) the average weight of the vehicles on the road.

There has been disagreement over the validity of the method (Piscitello et al., 2021), and field results can vary wildly from AP-42 estimates (Claiborn et al., 1995; Kantamaneni et al., 1996; Zimmer et al., 1992). For

[11] A different method is used to estimate resuspension from un-paved roads.

example, in their study cited earlier, Claiborn et al. (1995) compared resuspension EFs derived using tracer techniques with EFs derived from the AP-42 emissions inventory. They found that the EFs determined from two unpaved road experiments (136 g/veh/km and 336 g/veh/km) compared well with the estimates obtained from AP-42, but the results from paved roads varied considerably. Two-lane paved roads yielded results up to 80% higher than predicted (6.7 ± 3.7 g/veh/km compared to 3.7 g/veh/km) while major highways (4+ lanes and traffic in excess of 10,000 vehicles per day) gave EFs, on average, 44% lower than those predicted using standard formulae (1.0 ± 0.5 g/veh/km compared to 1.8 g/veh/km). Using the TRAKER system noted above, Etyemezian et al. (2006) and Etyemezian et al. (2003) estimated values from 60% to 180% higher than those obtained using the AP-42 method.

Doubts have also been raised on the scientific basis of the importance of silt loading without inclusion of the processes contributing to it, such as vehicle speed (Hussein et al., 2008; Venkatram, 2000, 2001). The applicability of the AP-42 method has not been assessed for New Zealand.

5.2.2 Other studies of resuspension

Thorpe et al. (2007) used a combination of methods to estimate resuspension EFs in a busy multi-lane highway in a street canyon in London. They estimated the total source strength of coarse particles ($PM_{2.5–10}$) arising from the road by calculating the roadside incremental concentration of coarse particles above the urban background. This was converted to a source strength by its ratio to nitrogen oxides, the source strength of which is estimated from the traffic mix and mean speed. This coarse particle source strength is assumed to represent the sum of resuspension emissions and the coarse particle component of abrasion emissions. Abrasion emissions were calculated from the EMEP/CORINAIR[12] EF database, the result subtracted from the total coarse particle emissions in order to yield resuspension emissions, and combined with traffic count data to derive fleet-average EFs. They estimated separate average EFs for light- and heavy-duty vehicles. Amato et al. (2016) used vertical profiles to estimate resuspension EFs in Paris.

As noted earlier for studies on road wear (see section 3.2), much of the work on quantifying resuspension has been carried out in the Nordic countries and relates to the use of studded winter tyres. Since these tyres are not used in New Zealand, the findings of these studies are not relevant to New Zealand and have not been reported here.

5.3 Emission factors to air reported in the literature

As is the case for emissions determined from direct vehicle or component testing (see section 3), it is difficult to compare EFs reported in different studies because the methods used and the metrics determined vary widely. For example, resuspension can be reported as a separate emission or be grouped with tyre and road wear (or vice versa), and it is not always stated that the two sources have been grouped. Moreover, the amount of metadata provided (eg, road type and condition, vehicle types, fleet composition, traffic congestion and driving conditions) varies considerably. Because different studies report results in different ways (eg, concentrations, EFs, proportion of total PM and/or size segregated), comparisons are difficult. In order to maintain some coherence in the presentation of results in this section, we will focus on and reference review papers from the past 16 years, rather than a myriad of individual research papers and sources. The reviews we found include:

- the review of measuring techniques in Lahuna et al. (2004)

[12] The EMEP/CORINAIR emission inventory (https://www.eea.europa.eu/publications/EMEPCORINAIR/page005.html) is now known as the EMEP/EEA emission inventory (European Environment Agency, 2019).

- the review by Boulter (2005) for the UK's Transport Research Laboratory
- AQEG's 2005 review on PM sources in the UK (Chapter 4), which was updated in 2019
- the 2014 review by Grigoratos and Martini for the European Commission's Joint Research Centre
- a newly published 2021 review in *Science of the Total Environment* by Piscitello et al.

Most of these reviews cite the same references, with the latest reviews including relatively few studies published after 2005. A feature of all the reviews is the wide range of EFs reported for the different NEE sources. In some cases, the reviews include EFs derived from wear tests and emission inventories (see section 3) as well as from air quality measurements.

Boulter (2005) summarised the EFs reported in 24 studies, all measuring different metrics using a variety of sampling and analytical methods. This summary is further summarised in Table 5.1 as the range of EFs for each NEE source and for all vehicle emissions. Since the EFs have been derived from different studies, the combination columns are not the sum totals of the component parts listed in the table, but come from studies where the NEE sources have been grouped and reported together. It can be seen that the range of results is very wide, but it is also clear that NEEs make up a significant fraction of all vehicle emissions. In some situations, especially in Scandinavia in winter and spring, they can be as much as 70–80% of the total. In other situations, such as free-flowing highways, brake and tyre wear can be negligible but can contribute as much as 20% of the total at intersections, traffic lights and motorway exits.

Table 5.1 Range of PM_{10} EFs (mg/veh/km) from NEEs summarised from Boulter (2005) (the number of studies cited for each emission class is given in parentheses)

Vehicle type	Tyre wear	Brake wear	Combined tyre and brake wear	Road surface wear	Combined tyre, brake and road surface	Re-suspension	All NEE sources	Exhaust
Light duty vehicles	0.5–13 (5)	1–7.8 (5)	6.9 (1)	3.1–10.5 (2)	7–10 (2)	0.8–780 (2)	4–70 (7)	7–16 (4)
Heavy duty vehicles	0–200 (4)	0–610 (3)	49.7 (1)	29 (1)	38–78 (2)	14.4–7,800 (2)	70–1,270 (7)	57–570 (4)

The results are very location-dependent; for example, some of the highest reported values come from Abu-Allaban et al. (2003), who measured PM alongside dusty desert roads in Reno, Nevada. In the same study, lower factors were reported from more temperate North Carolina. Similarly, and more recently, Amato et al. (2016) used vertical profiles to estimate road dust EFs in Paris. They calculated EFs ranging from 5.4 to 17.0 (average 10.4) mg/veh/km. They note that these values were consistent with, but at the lower end of, published values for European studies and comment that this was likely due to the climate and location of Paris compared to studies in Northern Europe, where studded tyres and road gritting are common, and to Southern Europe, which has a warmer and dustier climate.

The European Commission review of non-exhaust PM_{10} EFs to air (Grigoratos & Martini, 2014) is summarised in Table 5.2 for brake and tyre wear for light duty vehicles.[13] The EFs reported for brakes came from three brake dynamometer studies, four emission inventory studies and two receptor modelling studies. The EFs for tyres came from two direct measurement studies, six emission inventory studies and five receptor modelling studies. They conclude that most studies have found PM_{10} EFs to air of around 6–7 mg/veh/km for both brake and tyre wear for light duty vehicles and that the emissions from heavy duty

[13] PM from road wear and resuspension were not considered.

vehicles are approximately one order of magnitude higher. $PM_{2.5}$ EFs in the range of 2.1–5.5 mg/veh/km were also reported by Grigoratos and Martini (2014). Finally, they estimate that around 50% of brake wear and 0.1–10% of tyre wear particles become airborne; this implies that the total emissions from tyre wear are greater than for brake wear.

Table 5.2 Range of PM_{10} EFs (mg/veh/km) from brake and tyre wear summarised from Grigoratos and Martini (2014) for passenger cars and light duty vehicles

Metric	Tyre wear	Brake wear
Minimum	2.4	1.8
Maximum	13	8.8
Average	6.3	6.7
Number of studies cited	13	9

The EFs to air reported in both reviews by AQEG (2005, 2019) were derived from the UK National Atmospheric Emissions Inventory EFs from NEE sources using the European Monitoring and Evaluation Programme/European Environment Agency (EMEP/EEA) guidebook method (see section 3.5.1). Table 5.3 has been adapted from the EFs reported in the 2019 review. The values listed represent the average emissions of PM_{10} from different vehicle classes based on typical speeds for urban and rural roads and highways. AQEG (2019) notes that there are considerable uncertainties in the wear rates upon which these PM_{10} EFs are based, largely due to age, variability and paucity of data used to derive EFs from vehicles in the National Atmospheric Emissions Inventory.

Table 5.3 PM_{10} EFs (mg/veh/km) derived from the National Atmospheric Emissions Inventory for the UK

Vehicle class	Environment	Tyre wear	Brake wear	Road wear
Cars	Urban	8.7	11.7	7.5
	Rural	6.8	5.5	
	Motorway	5.8	1.4	
Light duty	Urban	13.8	18.2	7.5
	Rural	10.7	8.6	
	Motorway	9.2	2.1	
Heavy duty (rigid)	Urban	20.7	51.0	38
	Rural	17.4	27.1	
	Motorway	14.0	8.4	
Heavy duty (articulated)	Urban	47.4	51.0	38
	Rural	38.2	27.1	
	Motorway	31.5	8.4	
Buses	Urban	21.2	53.6	38
	Rural	17.4	27.1	
	Motorway	14.0	8.4	

Source: Adapted from AQEG (2005, 2019)

The most recent review of EFs to air that we have found comes from Piscitello et al. (2021). With the exception of road wear, it is not clear whether the EFs reported by Piscitello et al. (2021) pertain to a specific vehicle class. The range of PM_{10} and $PM_{2.5}$ EFs for tyres and brakes are summarised in Table 5.4. The EFs for tyre wear were determined from road simulator or roadside studies tests (six studies), receptor modelling (one study) and emission inventory studies (three studies). There are no clear differences in the EFs derived from these different methods. For brakes, four of the studies cited measured brake wear using a brake dynamometer, three estimated EFs from emission inventories, and six used receptor modelling. The lowest brake PM_{10} EF came from a brake dynamometer test (Hagino et al., 2015) and was considered an outlier by Piscitello et al. (2021). The range they suggest for PM_{10} is between 2.9 and 8.1 mg/veh/km. The EFs determined using receptor modelling had the greatest variation (2.2–15 mg/veh/km for PM_{10}). The brake EFs derived from inventories (7–18.5 mg/veh/km) for PM_{10} were higher than those reported for the dynamometer tests or the receptor modelling, which had similar ranges.

Table 5.4 Summary of EFs (mg/veh/km) for tyres and brakes reported by Piscitello et al. (2021)

Metric	Tyres		Brakes	
	PM_{10}	$PM_{2.5}$	PM_{10}	$PM_{2.5}$
Minimum	1.9	0.3	0.04*	0.04
Maximum	7.4	5.0	18.5	3.0
Median	6.3	2	7.4	2.3
Number of studies cited	10	6	12	5

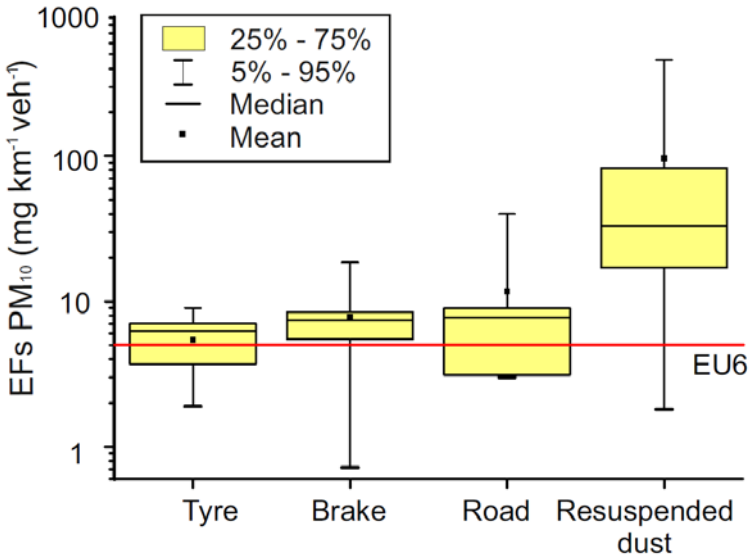
* considered an outlier

The range of PM_{10} and $PM_{2.5}$ EFs for road wear are summarised for non-studded tyres by vehicle class in Table 5.5. Given the low number of studies cited, it is not possible to generalise between the EFs obtained using different methods. What is evident is that the EFs increase with vehicle weight. Only one EF for $PM_{2.5}$ heavy duty vehicles was reported by Piscitello et al. (2021): 21 mg/veh/km, which comes from the UK National Atmospheric Emissions Inventory (2018). The resuspension EFs reported by Piscitello et al. (2021) were all estimated for PM_{10} using the US Environmental Protection Agency AP-42 method (see section 5.2.1). They cite six studies: five from Europe (Portugal, Italy, France and Spain) and one from the USA. The EFs ranged from 5.4 mg/veh/km for an inner-city road in Paris to 330 mg/veh/km for a cobbled road in Portugal. The PM_{10} EFs reported by Piscitello et al. (2021) for all NEE sources are compared in Figure 5.1. The red line denotes the exhaust exceedance limits for PM_{10} set by the EURO 6 air quality standards (5 mg/veh/km), and it is noted that the mean and median EFs determined for all the sources exceed this limit. From the figure, the highest variability in EFs are for resuspension followed by road wear.

Table 5.5 Summary of EFs (mg/veh/km) for roads by vehicle class reported by Piscitello et al. (2021)

Metric	Light duty		Heavy duty
	PM_{10}	$PM_{2.5}$	PM_{10}
Minimum	3.0	2	7.0
Maximum	9.0	4.1	38.0
Median	7.8	4.1	33.5
Number of studies cited	5	3	4

Figure 5.1 Box and whisker plots comparing the EFs for PM_{10} derived for each NEE source reported in the literature cited by Piscitello et al. (2021) (red line denotes the EURO 6 limit for exhaust emissions)



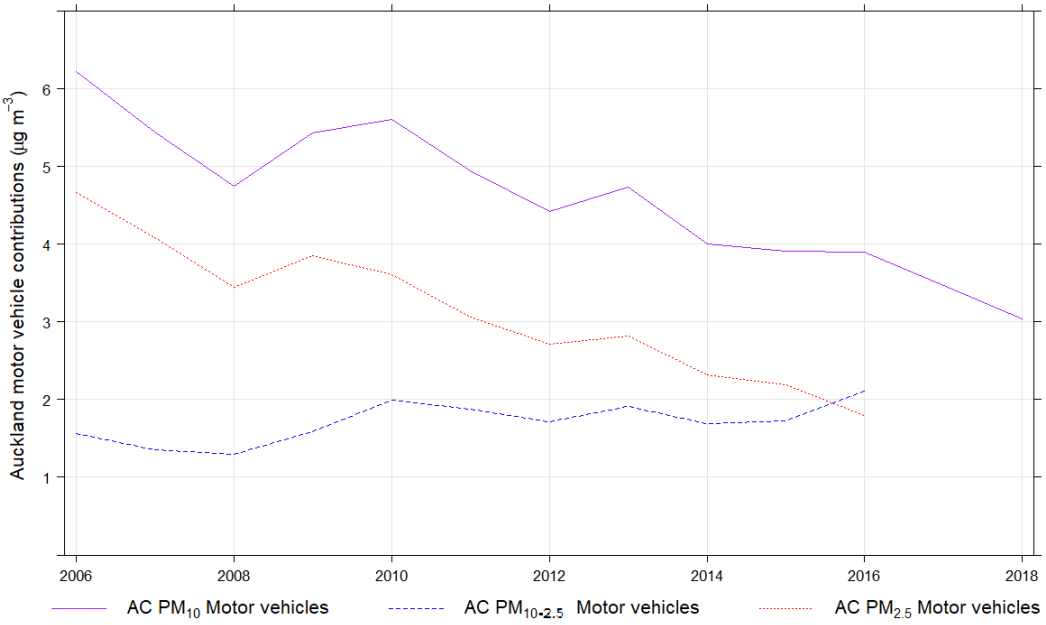
Source: Reprinted from Piscitello et al. (2021, p. 11)

5.4 Non-exhaust particulate in the New Zealand context

Few studies have been carried out in New Zealand explicitly to isolate the non-exhaust traffic signal from the bulk PM. A road dust resuspension study carried out in Auckland by NIWA in 2008 was unable to conclusively quantify resuspended road dust but was able to conclude that it was consistent with then current literature (NIWA unpublished data).

Source apportionment studies have been conducted in New Zealand by Davy et al. (2007), Davy (2007), Scott (2005), and Wilton et al. (2007). All of these studies have been able to identify a traffic component in PM_{10}, but most do not break this down further into exhaust and non-exhaust fractions. Where separate PM_{10} and $PM_{2.5}$ measurements were available for analysis, Davy and Trompetter (2020) attributed the coarse fraction of the 'road' component ($PM_{10–2.5}$) to NEEs and reported that the coarse fraction, primarily from resuspension of road dusts, is increasing at some sites in line with traffic volume increases, while the fine fraction associated with tailpipe emissions is going down. Figure 5.2 shows the trend in PM_{10} and $PM_{2.5}$ for Auckland since 2006. While there has been an overall decrease in PM_{10}, the ratio of fine to course particles has changed. It can be speculated that the intersection between the $PM_{2.5}$ and $PM_{10–2.5}$ shows the increase in importance of NEEs to tailpipe emissions (Perry Davy, GNS Science, pers. comm., March 2021).

Figure 5.2 The contribution of fine and coarse fractions to road-derived PM_{10} in Auckland



Note: Chart kindly provided by Perry Davy (GNS Science) for this report.

Further source apportionment work conducted by GNS on measurements taken in the Johnstone's Hill Tunnel on SH1 north of Auckland (Davy et al., 2011) derived six primary sources that contributed to PM_{10} concentrations: light duty vehicles, heavy commercial vehicles, smoky vehicles, road dust, biomass (wood) burning, and marine aerosol (sea salt). The light and heavy vehicle emissions included both exhaust and brake dust, while tyre wear was included in the road dust component. It is unclear whether or not it was possible to statistically distinguish brake and tyre wear as individual sources from the measurements taken.

None of the work by GNS has connected the concentrations measured to vehicle numbers, and so there are no EFs based on this work.

Emissions inventories in New Zealand have generally not included NEEs. For example, until recently the Auckland Council's Emissions Inventory for Auckland (eg, for 2006, Xie et al., 2014) specifically excluded secondary particulate, natural sources of particles and road dust from its estimates of PM. The latest Auckland Air Emissions Inventory (Xie et al., 2019) now includes estimates of road dust emissions but only reports total annual emissions for the entire Auckland region.

One of the few road emissions-dispersion model validation studies carried out in New Zealand (Longley et al., 2013) found that at sites close to SH1 in South Auckland, a combination of the VEPM and the AUSROADS model (a roadside dispersion model developed by the Victorian Environmental Protection Agency) over-predicted PM_{10} at the kerbside. This was attributed to possible over-prediction of emissions by the VEPM – although the study was not intended to validate the VEPM and the inference was tentative. It was also found that PM_{10} modelling was sensitive to a range of inputs not related to EFs, particularly meteorology and choice of background PM_{10} concentrations.

5.4.1 Challenges in comparing emissions research

The disparity across the literature in reported EFs and apportionment of PM to different sources is intrinsically linked to the operational details of the monitoring campaigns that collect the measurement data. Moreover, these are often constrained or determined by practical matters such as the timing and conditions

of access to monitoring sites and what instruments are available to be used. This has knock-on impacts in the laboratory; it can determine whether only elemental analysis is possible or organic compounds can be analysed as well, and it can determine which species are likely to be detectable.

As an example, the New Zealand Johnstone's Hill Tunnel (Davy et al., 2011) and the UK Hatfield Tunnel (Lawrence et al., 2013) studies appear to be similar; however, there are important differences, which affect their reported results. For the New Zealand study, there was one monitoring site in the tunnel near the exit. Filter samples spanned three hours and so were exposed to only light to moderate pollutant loads, depending on the time of day. The average annual daily traffic (AADT) for the tunnel at the time was 12,742 vehicles per day. In the UK study there were two sites: one near the entrance and one near the exit. This allowed an increment to be calculated that represented the emissions actually emitted within the tunnel. The UK tunnel was more heavily trafficked, with 48,500 vehicles during weekdays to 34,400 at weekends. In addition, the filters were exposed for 12 hours, meaning there were far heavier pollutant loads on the filters.

Furthermore, the UK study collected filters for both elemental and organic analysis (using inductively coupled plasma mass spectrometry and gas chromatography–mass spectrometry) while the New Zealand study only conducted elemental analysis using ion beam analysis. Similar numbers of elements were analysed for, but the UK study could also identify PAHs. Similar statistical methods were used: principal component analysis followed by either positive matrix factorisation or multilinear regression analysis. Both studies were conducted for four to five weeks over winter; however, there are differences in the winter conditions of each country. Both handled missing data or data below limits of detection slightly differently. The results of the two studies are compared in Table 5.6. Note that the source classifications reported are different so that it is not possible to make a direct comparison of the results.

Table 5.6 Summarised results of PM tunnel source apportionment studies from New Zealand (Davy et al., 2011) and the United Kingdom (Lawrence et al., 2013). The source classifications are different in the two studies and are listed for each in order of importance.

New Zealand		United Kingdom	
Source	**% of total PM**	**Source**	**% of total PM**
Marine aerosols	29	Resuspension	27
Road dust	28	Diesel exhaust	21
Heavy commercial vehicles	25	Petrol exhaust	12
Biomass burning	4	Brake wear	11
Smoky vehicles	7	Road surface wear	11
Light duty vehicles	7	Unexplained	18

The use of two monitoring sites, the heavier pollutant loads, and the measurement of organic compounds allows the UK study to focus more directly on vehicle emissions and NEEs in particular. In addition, the UK study went on to calculate EFs, presented in Lawrence et al. (2016); however, this has not been done for the New Zealand study. Indeed, many studies in the literature apportion PM to different sources but never take the final step of relating this apportionment to the traffic activity during the study, and thus never calculate EFs.

5.4.2 Use of emission factors in air quality models

The key use of EFs is to provide input to air quality models, which are in turn used to inform air quality management. To go from EFs to assess the impact of traffic emissions to air, it is necessary to bring several other modelling tools. First, models like the VEPM (Metcalfe & Peeters, 2020) output emission rates per level

of activity, so information about the activity levels (ie, VKT and fleet composition) is required. In New Zealand, models like Emme[14], VISUM[15] and CUBE[16] are used to estimate the traffic volumes through the roading network. Once the emission estimates have been scaled by the activity levels, they can be used in atmospheric dispersion models such as AERMOD (US Environmental Protection Agency, 2019) or TAPM (Hurley, 2008; Hurley et al., 2005). All these models also require meteorological information to translate the emission rates into ambient concentrations. These meteorological data are generated by models like NZLAM (Webster et al., 2008) or by analysing historical observations. Finally, in order to estimate the health burden of the concentrations estimated by these models, it is necessary to use an exposure and health effects model such as Health and Air Pollution in New Zealand (HAPiNZ) (Fisher et al., 2007; Kuschel et al., 2012). Each model in the suite of models chosen to assess air quality has its own set of assumptions and parameters, each with their own associated uncertainties. For example, dispersion models often represent topography as roughness coefficients to simplify model calculations and data requirements. The propagation and interaction of model error and uncertainties, including those associated with EFs, through the model suite is not known.

Therefore, in order to evaluate the quality of the emission estimates from the VEPM and assess the scale of the impact of their uncertainties, it is necessary to also address the uncertainties and assumptions used by all the modelling layers that go into the process.

[14] INRO | Emme – Multimodal Transport Planning Software: https://www.inrosoftware.com/en/products/emme/

[15] PTV group Visum – Traffic Planning: https://www.ptvgroup.com/en/solutions/products/ptv-visum/

[16] Bentley Systems Transportation & Land-Use Modeling; CUBE by Citilabs: https://www.bentley.com/en/products/brands/cube

6 Emissions to road dust and water

The main source of particles and contaminants from vehicles in road runoff is wash-off of road dust that has accumulated on the road surface. This means that the characteristics of PM and contamination in road dust and road runoff are similar, and both have been used to derive EFs alone and in combination. Like receptor modelling for air quality, deriving EFs from contaminant loads assumes that traffic is the only source of contamination to road runoff. However, particularly in urban areas, other sources of contaminants in road runoff can include:

- deposition of aerosols (including PM from NEEs)
- weathering of galvanised (zinc) and copper roofs, gutters and downpipes
- fungicides
- soil
- gross pollutants/litter
- organic debris (eg, animal waste, leaves and twigs).

The presence of other sources means that sites used to determine EFs need to be chosen to minimise contamination from other sources.

Methods of sampling and characterising PM in road dust and road runoff have been discussed in detail by, amongst others, Burton and Pitt (2002), Clark et al. (2009) and Roesner et al. (2007). These publications note that the sampling method chosen can have a profound effect on the contaminant concentration and particle size distribution of sampled solids, and care must be taken in site selection, instrumentation and the timing of samples (Edwards & Glysson, 1999). Common sampling methods for road dust include vacuuming (Kayhanian et al., 2012; Vaze & Chiew, 2002; Zanders, 2005), collection of road dust on boards placed on the road surface (Wicke et al., 2012), washing and collection of wash water, and street sweeping (German & Svensson, 2002). Runoff water can be sampled manually (ie, grab-sampling) or with passive and automatic samplers. PM in runoff can also be collected by sampling solids that have settled in catchpits (Moores, Hunt, et al., 2009) or that have been trapped in, for example, catchpit inserts and filters (Moores et al., 2012).

Contaminants in road dust are usually measured as a mass concentration (eg, mg/kg) whereas contaminants in runoff are expressed as a volume concentration (eg, mg/L). Since PM can be diluted or dissolved in runoff, it is not possible to compare concentrations in road dust and runoff directly. Comparisons are therefore done using estimates of the load of the contaminants, which includes both the particulate and dissolved fractions in runoff. The load of road dust is estimated by sampling and weighing PM from a stretch of road while the load from runoff is estimated as the product of the contaminant concentration and the runoff volume over the duration of a flow event. Since the amount of accumulated road dust and the quantity and quality of runoff measured at a monitoring site will change between events due to differences in rainfall dynamics (intensity, duration, frequency, antecedent dry days), wind, road surface roughness etc, the loads need to be determined over a sufficient length of time to capture the full range of climate conditions at the site. If it is assumed that a catchment consists of a section of road with no sources other than traffic and that all road dust accumulated between storm events is washed off during an event, then the catchment loads determined from either road dust or road runoff will be the same and will be indicative of the contaminant and particle emissions from vehicles emitted over the accumulation period. However, it reality, it is unlikely that all accumulated street dust is washed off due to, for example, depression storage of rain water, particularly for rough road surfaces, as was discussed in section 3.2.3 and demonstrated for road dust by Zhao et al. (2016).

6.1 Particle-size distribution

The particle-size distribution (PSD) of sediments is important for stormwater management since many contaminants are commonly bound to particles in different size ranges (Characklis & Wiesner, 1997; Sansalone et al., 1998), which has implications for the treatment of stormwater and road runoff. PM in road dust and road runoff cover a size range of more than six orders of magnitude, from nanometre-sized colloidal organic material to millimetre-sized sand and gravel (Driscoll et al., 1986; Kayhanian et al., 2012; Roesner et al., 2007). Clark et al. (2009) report median grain sizes of total solids in stormwater from 8 to 1,200 µm, although the upper value was associated with the application of grit to snow for friction control. Semadeni-Davies (2013) reviewed the PSD of particles found in road runoff, urban stormwater and road dust reported in New Zealand and internationally. Those reported for road runoff or road dust are reproduced in Table 6.1 and are plotted for the New Zealand studies in Figure 6.1. More recently, Charters et al. (2015) looked at the PSD of particles in stormwater in Christchurch to elucidate how PSD can be used to select stormwater treatment options. They found that the PSD varies between rainfall, which is partially explained by wash-off of finer particles during low-intensity rainfall events. Road contributed the highest loads of sediment compared to other urban surfaces, and the PSD of particles in road runoff had the highest correlation to rainfall intensity – which is in keeping with the notion that coarser particles are trapped by road surfaces during low-intensity rainfall events. Their results for road runoff have been added to Table 6.1.

Of interest here is that most particles found in road runoff or road dust have a diameter greater than 10 µm. This means that they cannot be resuspended to air as PM_{10} without further fragmentation; however, particles greater than 10 µm can become airborne and contribute to TSP.

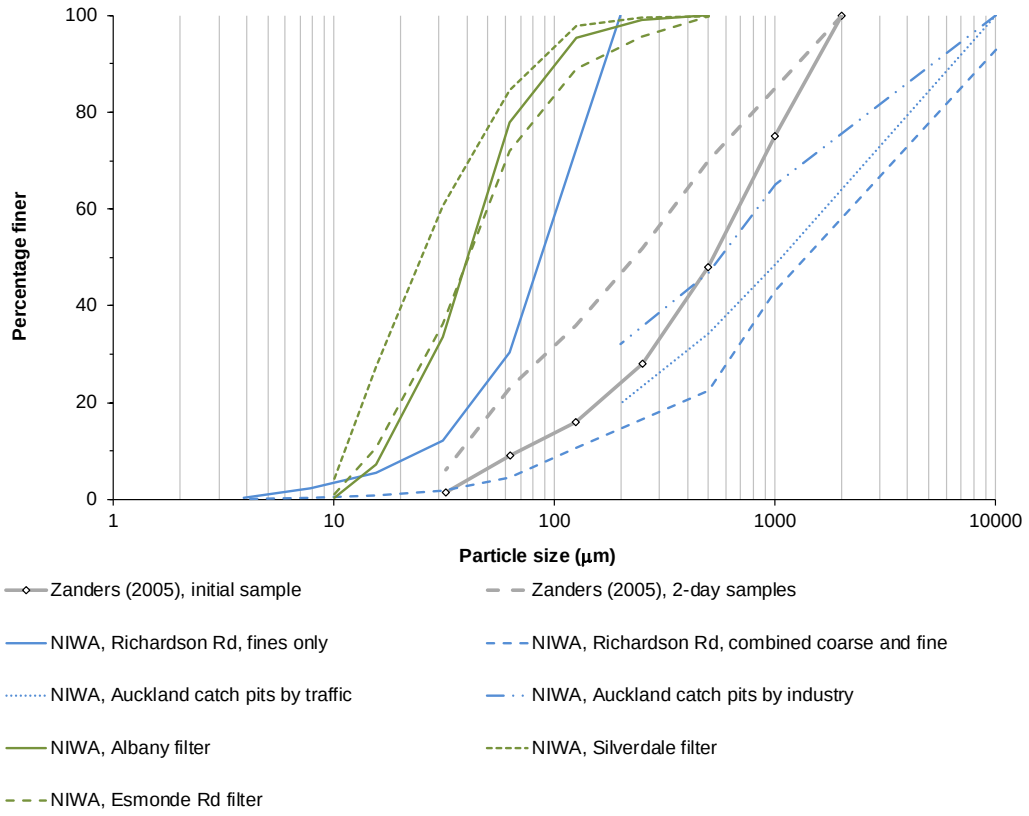
There can be some differences in the PSD of suspended solids in runoff compared to road dust because larger particles are either not entrained by runoff, especially during low-intensity rainfalls, or settle out during transport. The PSD of particles in runoff varies by land use and can vary between events at the same location due to seasonal differences in the length of the accumulation period between rainfalls (ie, due to particle fragmentation) and intra-event changes in rainfall dynamics (eg, Vaze & Chiew, 2002).

Table 6.1 Particle size ranges for particles found in road runoff and road dust reported internationally and in New Zealand

Study	Analytical method	Particle size range
Driscoll et al. (1986)	Settling columns	2–82 µm
Sansalone et al. (1998)	• Sieving (mesh between 9.25 mm and 20 µm) • Light obscuration counter (particles < 20 µm)	555 µm mean median (d_{50}) grain-size over 13 events
Cristina et al. (2002)	• Sieving (mesh sizes not reported) • PSD reported as percentage of the particle total surface area by particle size band	98% of total surface area associated with particles > 75 µm
Furumai et al. (2002)	Sieving (mesh for stormwater particles – 20–250 µm, mesh for road dust up to 800 µm)	• Variable across events with median grain size between < 20 and > 106 µm • Coarse sediments are associated with high TSS loads
German and Svensson (2002)	Sieving (mesh between 250 µm and 75 µm) of road dust collected pre- and post-street-sweeping	Fraction finer than 250 µm: • pre-sweeping 26% • post-sweeping 40% • swept particles 20%
Kayhanian et al. (2012)	• Samples taken from highway surface sediments, highway runoff and settled sediments from dry detention ponds • Settling and laser diffraction counter, 38 µm used as cut-off between coarse and fine sediments	Proportion of sediments < 38 µm varied, averages are: • surface sediments 0.7% • runoff sediments 67.4% • pond sediments 32.3%
Zanders (2005)	Sieving of road dust sampled in Hamilton (mesh between 2,000 µm and 32 µm)	• Initial sample had 28% particles < 250 µm • Subsequent samples had 52% of particles < 250 µm
Moores, Hunt, et al. (2009)	• NIWA Richardson Rd catchpit sediments • Sieving for coarse sediments (1 cm to 200 µm mesh) and laser obscuration counter for fines (< 200 µm)	Around half of solids are in the 1 mm to 1 cm size class
Moores, Pattinson, et al. (2009)	• NIWA catchpit sediments from traffic • Sieving for coarse sediments (1 cm to 200 µm mesh)	Proportion of particles in the 1 mm to 1 cm size range varies between 27% and 85%
Gadd et al. (2010)	• NIWA catchpit sediments near industry • Sieving for coarse sediments (1 cm to 200 µm mesh)	Proportion of particles in the 1 mm to 1 cm size range varies between 10% and 61%
Moores et al. (2012)	• NIWA filter study, pre-filtration of road runoff • Sieving for coarse sediments (1 cm to 250 µm mesh) and laser obscuration counter for fines (< 200 µm)	• Median grain size in the 62.5–125 µm size band for Albany and Esmonde Rd • Median grain size for Silverdale site is in the 31.2–62.5 µm size band
Charters et al. (2015)	• PSD of sediment from different urban surface types determined with particle analyser • PSD determined for 28 samples of road runoff from an asphalt road with 11,000 AADT	• 10th percentile size ranged from 4 to 57 µm, average of 23.2 µm • Median size ranged from 11.7 to 102.9 µm, average 71.6 µm • 90th percentile size ranged from 42.4 to 784.5 µm, average 177.2 µm

Source: Adapted from Semadeni-Davies (2013)

Figure 6.1 Particle size distributions of PM found in road runoff or road dust reported in New Zealand



Note: Grey = road dust; blue = catchpit sediments; green = pre-filter sediments.
Source: After Semadeni-Davies (2013)

6.2 Particulate contamination

Contaminants derived from the wear of brakes, tyres and road that are commonly found in road dust and road runoff are listed in Table 6.2 along with key references. The concentrations of these contaminants in road dust and runoff vary widely both spatially and temporally. NEEs have long been recognised as a major source of sediment, heavy metals (especially zinc and copper) and PAHs in road dust and road runoff internationally (Legret & Pagotto, 1999; Loganathan et al., 2013; Lundy et al., 2012; Müller et al., 2020; Sansalone & Buchberger, 1997; Sartor & Boyd, 1972) and in New Zealand (Kennedy, 2003b; Kennedy & Gadd, 2000; Timperley et al., 2005). Müller et al. (2020) note that much of the earlier literature on stormwater and road runoff quality is outdated due to changes in manufacturing processes and materials and regulations such as the global removal of lead from petrol and adoption of low-copper brake pads in parts of the USA. Moreover, there has been increased focus in recent years on emerging contaminants that has broadened the range of contaminants analysed and reported in the literature, such as microplastics (see discussion below). Indeed, a compound commonly found in tyre wear particles (N-(1,3-dimethylbutyl)-N'-phenyl-p-phenylenediamine, known as 6PPD) has recently been implicated in mass salmon fish death in urbanised streams along the Pacific Northwest coast of the USA (Tian et al., 2021).

Table 6.2 Non-exhaust sources of contaminants in stormwater

Source	Contaminants	Key references
Tyres	sediment, cadmium, copper, zinc, microplastics, PAHs	• Councell et al. (2004) • Horton, Walton, et al. (2017) • Kose et al. (2008) • Legret and Pagotto (1999) • McKenzie et al. (2009) • Muschack (1990)
Brakes	sediment, cadmium, copper, nickel, lead, antimony, zinc, PAHs	• Hjortenkrans et al. (2007) • Markiewicz et al. (2017) • McKenzie et al. (2009)
Roads abrasion	sediment, PAHs, microplastics	• Horton, Svendsen, et al. (2017) • Hvitved-Jacobson and Yenisei (1991) • Lindgren (1996) • Markiewicz et al. (2017)

Source: Adapted from Müller et al. (2020)

Several studies have investigated factors other than traffic characteristics than can influence contaminant loadings on road dust and runoff water quality (De Silva et al., 2016; Hjortenkrans, 2008; Loganathan et al., 2013). The most comprehensive review we found is presented by Huber et al. (2016), who undertook a meta-analysis of published road runoff water quality data collected from 294 sites around the world to determine how traffic volumes and other factors affect heavy metal concentrations. Most of the studies cited were published in the mid-1990s to early 2000s and were undertaken in Europe and the USA; however, data from Asia, Africa and Australia are also included, as are Auckland data reported by Timperley et al. (2005) and Moores et al. (2010). The data were grouped into eight traffic classes and were analysed using available metadata. The traffic classes were parking lots; bridges; roads with traffic volumes of < 5,000 AADT, 5,000–15,000 AADT, and > 15,000 AADT; urban highways with > 30,000 AADT; non-urban highways with > 30,000 AADT; and highways with < 30,000 AADT. Where traffic volumes were not available, assumptions were made on the basis of the type of road – for example, feeder roads were assumed to have < 5,000 AADT. The site-specific factors investigated were:

- land use characteristics, including both rural and urban land-use classes
- traffic area site data, such as road design (eg, surface type, speed limits, traffic signals, speed and noise barriers, exit lanes, intersections), topography, and vegetation cover
- traffic characteristics and operation, such as level of congestion, types of vehicles (eg, proportion of heavy vs light vehicles), road maintenance (eg, mowing grass verges, street sweeping, application of de-icing salts and grit material) and institutional regulations (eg, removal of lead from petrol)
- climate, including rainfall dynamics, antecedent dry period, season and wind turbulence (which affects deposition and resuspension).

They also noted the number of samples, sampling methods (eg, type of sampler and whether the sampling was for the event first-flush or covered the entire event using flow or time proportional sampling) and analytical methods used to determine water quality.

Zinc and copper concentrations collated by Huber et al. (2016) are reproduced in Table 6.3 and show wide variation in concentrations between and within road classes. It is not possible to calculate EFs from these data as no loadings or road lengths are provided. They concluded that the pollutant concentrations are highly site specific and depend not only on traffic characteristics but also on the other factors investigated, including

sampling and analytical methods. A comparison of metal concentrations categorised by continent found that the median copper concentration for North America (30 µg/L) was almost half those for Asia, Australia and Europe. For zinc, the overall median concentrations were approximately 170 µg/L for North America, 250 µg/L for Europe, 300 µg/L for Australia, and 620 µg/L for Asia. Bridges tended to have higher zinc loadings than highways with similar traffic counts due to emissions of zinc from safety fences along the bridge; however, this finding differs from that of Bakr et al. (2020), who report significantly lower zinc concentrations in bridge runoff (between 66 and 760 µg/L). Metal concentration in runoff from parking lots varied widely with employee parking lots (single use per day) having lower concentrations than supermarkets (multiple use by light vehicles) or truck stops (multiple use by heavy vehicles). Runoff from roads with more than 5,000 vehicles per day tended to have higher metal concentrations than highway runoff with greater traffic volumes. This is due to intersections, speed limits and traffic signals on urban roads leading to increased brake and tyre wear. Interestingly, they report that metal concentrations in highway runoff are not significantly influenced by either land use or traffic volumes worldwide. They conclude that any study that monitors road runoff should provide site metadata that can be used to characterise the monitoring sites in order to develop meaningful relationships between traffic and water quality.

Table 6.3 Summary of zinc and copper total concentrations (µg/L) in road runoff for different road categories published in international literature

Metal	Road class	Mean average		Percentiles				Range		SD
		Geometric	Arithmetic	Median 50th	75th	90th	95th	Min	Max	
Zinc	Parking lot	155	201	178	295	397	586	39.0	620	147
	Road AADT < 5,000	141	212	149	272	525	a	25.0	940	219
	Road AADT 5,000–15,000	203	285	274	384	533	886	23.0	1,000	217
	Road AADT > 15,000	379	474	351	556	982	1,805	120	1,940	397
	Bridge	611	2,231	498	700	15,572	a	77.0	19,100	5,607
	Highway AADT < 30,000	207	306	217	371	642	1,540	32.3	1,760	359
	Non-urban highway AADT > 30,000	248	385	216	388	1,202	1,875	52.5	2,210	477
	Urban highway AADT > 30,000	237	338	213	414	682	986	21.0	2,234	358
Copper	Parking lot	23.4	40.7	19.6	56.0	116	165	5.0	220	49.2
	Road AADT < 5,000	37.5	53.7	48.0	77.4	104	a	6.0	180	43.9
	Road AADT 5,000–15,000	38.6	64.6	30.5	100	175	264	7.0	280	71.3
	Road AADT > 15,000	86.9	105	86.9	149	197	275	26.0	288	66.6
	Bridge	54.9	63.9	54.0	93.2	133	a	20.0	136	36.1
	Highway AADT < 30,000	49.4	60.7	48.0	91.7	124	138	13.3	140	38.8
	Non-urban highway AADT > 30,000	61.8	84.4	53.0	97.0	205	394	23.0	430	90.4
	Urban highway AADT > 30,000	51.4	63.5	56.4	74.7	118	145	13.0	274	45.5

SD = standard deviation; AADT = average annual daily traffic; a = too few samples available
Source: Adapted from Huber et al. (2016, p. 116)

In New Zealand, Sampson (2017) carried out a statistical analysis to elucidate the factors affecting the metal (including zinc and copper) concentrations of road dust collected in Christchurch. Road dust was collected over seven months from 30 sites across the city representing different traffic volumes and land use types. With the exception of antimony and chromium, there were no significant relationships found between traffic numbers and most of the metals. This finding is surprising for zinc and copper; the lack of a relationship was ascribed to the contributions of other metal sources, including atmospheric deposition from industry for zinc and the prevalence of home-heating wood burners for copper and arsenic. Indeed, Sampson (2017) did find high zinc concentrations in an industrial area containing a tyre repair shop, a petrol station and a car-yard. Several studies in Christchurch identified atmospheric deposition (Murphy et al., 2015; Murphy et al., 2014; Wicke et al., 2012) as a source of copper, lead, zinc and sediment. The effect of atmospheric deposition was to dampen the effect of land use adjacent to the monitoring site on water quality. Finally, also in Christchurch, Charters et al. (2021) found that zinc concentration from galvanised roofs and guttering were higher than the concentrations from other surfaces, and the greatest concentrations were from older roofs. Similarly, typical concentrations of copper in runoff from copper roofs are an order of magnitude higher than copper concentrations in road and carpark runoff. However, roads were associated with the highest sediment concentrations. While this work ranked contaminant concentrations from common urban surfaces, it is important to remember that the loadings of contaminants to stormwater also depend on the source area, so the total contribution of these sources may differ. For example, while copper roofs are associated with high copper concentrations, copper roofs are relatively uncommon in New Zealand. However, galvanised roofs and guttering, which are very common in New Zealand, especially in older suburbs, are a major source of zinc to stormwater.

Tyres as a source of microplastics in waterways is an emerging issue, and there have been numerous studies into the abundance and environmental effects of microplastics from tyres in recent years (eg, Horton, Svendsen, et al., 2017; Horton, Walton, et al., 2017; Järlskog et al., 2020; Leads & Weinstein, 2019). Järlskog et al. (2020) define microplastics as articles in the range of 1–5,000 μm, composed of synthetic polymers with thermoplastic or thermo-set properties, elastomers (eg, styrene-butadiene rubber), and polymer modified bitumen. They identified tyre wear as the source of microplastics found in stormwater and road dust on the basis of shape and the presence of bitumen and mineral incrustations from road wear. Similarly, Kovochich et al. (2021) used the shape and chemical composition of tyre particles to characterise these particles to support further research into the dispersion of microplastics from road and tyre wear in the environment. However, AQEG (2019) questions whether particles from tyre wear can be considered plastic both due to the same presence of road fragments and because some rubbers found in tyres do not meet the ISO 472: 2013[17] standard definition of plastic. If included in the definition of plastic, then rubber fragments from tyre wear can be a considerable source of microplastics – indeed, Kole et al. (2017) suggest that tyre wear could be contributing 5–10% of microplastics entering oceans each year.

6.3 Emission factors to water

While there have been numerous studies that have characterised the quality of road runoff and road dust over recent years (eg, see reviews in Bakr et al., 2020; Huber et al., 2016; Müller et al., 2020), relatively few have provided EFs or contaminant loadings and traffic information with which to calculate EFs. This observation was previously noted by Moores et al. (2010) in their review of EFs reported in New Zealand and international literature. We found no EFs reported after 2010 either in New Zealand or internationally. The international EFs for zinc and copper as indicators of NEEs identified by Moores et al. (2010) are presented in Table 6.4. The New Zealand examples are presented and discussed in more detail below. Note that most

[17] https://www.iso.org/obp/ui/#iso:std:iso:472:ed-4:v1:en

of the EFs in Table 6.4 were calculated by Moores et al. (2010) using loads and traffic volumes published in the respective references.

Table 6.4 Zinc and copper vehicle EFs (mg/veh/km) derived from international studies

Reference	Location	AADT	Copper	Zinc
Hoffman et al. (1985)	Rhode Island, USA	101,500	1.0	22.0
Harrison et al. (1985)	M6, UK	30,000	0.33*	–
Hewitt and Rashed (1992)	M6, UK	37,600	0.19*	–
Legret and Pagotto (1999)	Nantes, France	11,800	0.067*	0.56*

* Calculated by Moores et al. (2010)
Source: Adapted from Moores et al. (2010, p. 21)

The recently compiled 'Transport Knowledge Hub for Water – Research Stocktake' prepared by Moores (2020) for Waka Kotahi lists 39 papers, theses and reports from New Zealand that present data characterising the physical and chemical properties of road dust or road runoff or that include vehicle EFs to water. These are listed in chronological order along with short descriptions of their contents in Appendix C. Most of the documents were published before 2008; only 11 have been published since then. This means that the fleet composition and traffic volumes at the time of sampling will not be representative of the current fleet composition or traffic volume. EFs are provided or discussed in nine of the documents; these are listed in Table 6.5.

Table 6.5 Transport Knowledge Hub for Water – Research Stocktake entries that present characterisations of road dust and/or road runoff

Year	Citation	Description	Commissioning Agency	Research Agency	Link
2000	Gadd and Kennedy (2000). *Preliminary examination of organic compounds present in tyres, brake pads and road bitumen in New Zealand*. Wellington: Ministry of Transport. 17pp.	Analysis of organic compounds in samples of tyres, brake pads and road bitumen.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwater-organiccompounds2.pdf
2000	Kennedy and Gadd (2000). *Preliminary examination of trace elements in tyres, brake pads and road bitumen in New Zealand of metal concentrations in samples of tyres, brake pads and road bitumen.* Wellington: Ministry of Transport. 19pp.	Analysis of metal concentrations in samples of tyres, brake pads and road bitumen.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwaterinorganiccompounds.pdf
2002	Kennedy et al. (2002). *EFs for contaminants released by motor vehicles in New Zealand*. Wellington: Ministry of Transport. 104pp.	Estimates of vehicle contaminant emissions based on a review of the vehicle fleet, brake pad composition, tyre composition, emissions of lubricants and greases, emissions of coolants, emissions from exhausts and wear of bitumen road surfaces.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwateremissionfactors.pdf
2002	Moncrieff and Kennedy (2002). *Road transport impacts on aquatic ecosystems – Issues and context for policy development* (updated 2004). Wellington: Ministry of Transport. 88pp.	Development and application of framework for assessing effects of road-derived contaminants on aquatic ecosystems.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwater-road-effects.pdf
2003	Kennedy (2003a). *The effects of road transport on freshwater and marine ecosystems*. Wellington: Ministry of Transport. 142pp.	Summary report bringing together series of Ministry of Transport reports produced between 2000 and 2002. Covers: • sources of transport-derived contaminants • measured contaminant concentrations in road dust and stormwater • contaminant fate and effects in receiving freshwater and marine environments.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwaterroadtransporteffects.pdf
2003	Kennedy (2003b). *Metals in particulate matter material on road surfaces*. Wellington: Ministry of Transport. 99pp.	Analysis of metal concentrations in road dust samples from Wellington and Waitakere City, and review of other data sources.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwaterparticulatematerial.pdf

Year	Citation	Description	Commissioning Agency	Research Agency	Link
2003	Kennedy and Gadd (2003). *Evaluation of road surface contaminant loadings in Waitakere City for the development of the Vehicle Fleet Emission Model – Water*. Wellington: Ministry of Transport. 34pp.	Analysis of contaminant concentrations in road dust samples collected in Waitakere City and development of EFs to support modelling of road-derived contaminant loads.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwatercontaminantloadingswaitakere-city.pdf
2005	Timperley et al. (2005). *Source and loads of metals in urban stormwater.* NIWA Client Report: AKL2004-070 prepared for Auckland Regional Council.	Estimates of loads of copper, zinc and lead in Auckland stormwater from roads and other sources. Based on extensive sampling of road, commercial, industrial and residential stormwater.	Auckland Regional Council	NIWA	http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.623.6620&rep=rep1&type=pdf
2008	Moores et al. (2008). *Mitigation strategies for controlling the dispersion of particulate metals emitted from vehicles*. NIWA report AKL-2008-048 prepared under FRST contract C01X0405. 78pp.	Estimates of performance of stormwater treatment (ponds, swales and roadside channels) for removal of particulate metals based on sampling at locations north of Auckland. Reviews dispersion of airborne emissions of particulate metals form roadside.	Foundation of Research, Science and Technology	NIWA and Landcare Research	–
2010	Moores et al. (2010). *Enhancing the control of contaminants from New Zealand's roads: Results of a road runoff sampling programme*. New Zealand Transport Agency research report 395. 161pp.	Stormwater sampling study on state highways in the Auckland region generating estimates of EFs for loads of copper and zinc discharged in road runoff, and the performance of stormwater treatment devices (ponds and swales) for removing these metals and TSS.	Waka Kotahi NZ Transport Agency	NIWA	https://www.nzta.govt.nz/assets/resources/research/reports/395/docs/395.pdf

Source: Adapted from Moores (2020)

Most of the documents listed in Table 6.5 were prepared as a series by Paul Kennedy and colleagues at Kingett Mitchell Ltd for the Ministry of Transport between 2000 and 2003. The earliest reports in the series (Gadd & Kennedy, 2000; Kennedy & Gadd, 2000; Kennedy et al., 2002) determined the organic and inorganic composition of brakes, tyres and road wear and developed a methodology to calculate EFs from vehicles. These reports are cited in section 3 of this report. Moncrieff and Kennedy (2002) and Kennedy (2003a) used the EFs from vehicles to develop a framework for evaluating the effects of vehicle emissions on marine and freshwater-receiving environments with the VFEM-W.

The remaining reports in the series (Kennedy, 2003b; Kennedy & Gadd, 2003) were undertaken to support the development of that model. Kennedy and Gadd (2003) analysed metal and PAH concentrations and PSD of road dust collected from Waitakere City (West Auckland) and Wellington. The Wellington road dust was collected from sites in Wellington and Lower Hutt in 1980. The Waitakere sampling and analysis was carried out in 2002 for 20 sites and is described in detail in Kennedy and Gadd (2003). The Waitakere samples were typically taken three days after rainfall by vacuuming the road surface. The proportion of each metal originating from vehicles was estimated, where possible, on the basis of apportionment from international studies.

The concentrations of metals and PAHs determined for the Waitakere study were used to validate the EFs of zinc, copper, lead and PAHs calculated for the sites using the methods presented in Kennedy et al. (2002) that were incorporated into the VFEM-W (see section 3.5.2). Measured EFs were derived from the road dust concentrations by dividing the measured load of each contaminant by the length of road sampled and the number of vehicles using the roads over the antecedent dry period before sampling – this method did not take into account other contaminant sources or the fraction of PM lost to air or resuspension. The modelled and measured emissions are summarised in Table 6.6. It was found that the measured concentrations were generally higher than those predicted for copper and lead but were lower for zinc and PAHs. The discrepancies between the modelled and measured EFs were ascribed to contaminants from other sources (eg, the high lead concentration in road dust was ascribed to lead bound to soil particles from historical tailpipe emissions of lead from petrol) and high uncertainty in the wear rates and metal concentrations determined by Gadd and Kennedy (2000) and Kennedy and Gadd (2000).

Table 6.6 Waitakere City estimated EFs (mg/veh/km)

Estimate	Copper	Lead	Zinc	PAHs
VFEM-W normal driving conditions*	0.086	0.0035	1.2	0.079
VFEM-W congested driving conditions*	0.12	0.0043	2.3	0.098
Measured 25th percentile	0.017	0.024	0.049	0.004
Measured median	0.16	0.11	0.18	0.014
Measured 75th percentile	0.50	0.62	1.0	0.059

* based on the average New Zealand fleet at the time of modelling
Source: Adapted from Kennedy and Gadd (2003, p. 25)

The three remaining studies listed in Table 6.5 were undertaken by NIWA. Timperley et al. (2005) calculated yields of copper, lead and zinc by undertaking a mass balance of sediments sampled in runoff and captured in catchpits from a 500 m long section of Richardson Road in Auckland. The total sediment load to water from the road section was estimated by fitting a build-up/wash-off model developed by NIWA (Stormqual) to the measured sediment data. Richardson Road is a residential single carriage-way with an estimated traffic volume at the time of about 17,000 AADT. The site was selected to avoid contamination from other sources;

that is, the catchment area is readily defined, has a single drainage point suitable for flow monitoring and sampling with an automatic sampler, and has no inflow from other drainage sources (eg, roof runoff from downpipes). EFs were determined from the contaminant loads based on total traffic numbers over the monitoring period; these are shown in Table 6.7. A TSS EF was also determined for Richardson Road (65.2 g/veh/km), which was used to develop the sediment yields for roads in the Auckland Council Contaminant Load Model (Auckland Regional Council, 2010b), as discussed in the following section.

Table 6.7 Richardson Road EFs for zinc, lead and copper (mg/veh/km)

Metal	Total	Particulate
Zinc	0.45	0.26
Copper	0.078	0.061
Lead	0.068	0.067

Source: Timperley et al. (2005)

The study reported in Moores et al. (2008) was undertaken to determine the fate of these metals in receiving environments, to establish the relative importance of airborne and water-borne dispersion pathways, and to evaluate the effectiveness of mitigation strategies to control metal dispersion. Part of this investigation was to estimate EFs to water (and air, as discussed above) from along a section of the Northern Motorway and at the Silverdale interchange in the John Creek catchment (catchment area approx. 4.5 km^2) about 25 km north of Auckland. The EFs of metals to water were estimated from metal loads in runoff discharged to two stormwater ponds (one collecting runoff from the motorway section and one from the Silverdale interchange).

The estimated copper and zinc loads and EFs to water and air are given in Table 6.8. The estimated EFs of copper and zinc conveyed in runoff from the motorway are approximately half those determined for the interchange, which is likely due to heavy tyre wear and frequent braking at the interchange. The EFs of metals transported in the highway runoff is about three times greater than for the air-deposited PM. The authors note that interception of airborne particles may have reduced the load. Finally, they recommend that there needs to be greater investigation of the relationships between traffic behaviour and the metal concentrations in PM and suspended sediments.

Table 6.8 Annual loads and EFs determined for copper and zinc by Moores et al. (2008)

Location and pathway	Annual load (kg)		Vehicle EF (mg/veh/km)	
	Copper	Zinc	Copper	Zinc
Air deposited, Northern Motorway	1.3	6.5	0.028	0.14
Runoff, Northern Motorway	3.9	20.9	0.08	0.45
Runoff, Silverdale interchange	0.5	3.2	0.13	0.87

The second NIWA report (Moores et al., 2010) was undertaken for Waka Kotahi and the Auckland Regional Council with the objectives of determining representative contaminant loadings for different types of roads and evaluating the effectiveness of existing contaminant control methods. EFs for copper and zinc were determined for four sites in Auckland representing roads with normal and congested traffic flows. The EFs were then compared to those reported nationally and internationally. The EFs were determined from zinc and copper concentrations in samples using an event-based sampling method similar to that used by Moores et al. (2008) and by fitting the Stormqual model to the measured concentrations similar to the method used by Timperley et al. (2005). EF estimates produced by the event-based method were generally higher than those

resulting from the modelling of contaminant loads. This indicates that the events sampled were relatively effective at removing contaminants from the road. Less-effective events, such as those characterised by low rainfall depths and intensities, were under-represented in the sampling programme. The modelled EF estimates (Table 6.9) were less sensitive to the characteristics of individual storm events, and therefore deemed likely to be a better estimate of the long-term contaminant load discharged from the road at each site. Moores et al. (2010) also calculated EFs for three further Auckland sites using existing road runoff data collected between 2006 and 2007 and estimated traffic numbers; these are given in Table 6.10.

Table 6.9 Event-based and modelled vehicle EFs (mg/veh/km) for zinc and copper from Auckland highways

Site	Congestion*	Copper			Zinc		
		Event based		Modelled**	Event based		Modelled**
		Aggregate (all events)	Range		Aggregate (all events)	Range	
SH18 @ Westgate	1.38	0.099	0.021–.045	0.08	0.56	0.15–2.31	0.5
SH1 @ Northcote	0.81	0.028	0.012–0.066	0.011	0.08	0.019–0.16	0.04
SH16 @ Huapai	0.52	0.052	0.005–0.15	0.054	0.23	0.011–0.66	0.28
SH1 @ Redvale	0.40	0.078	0.030–1.16	0.039	0.42	0.11–8.5	0.28

* Ratio of AADT to capacity (Gardiner & Armstrong, 2007)
** Adjusted for catchpits
Source: Adapted from Moores et al. (2010, p. 85)

Table 6.10 Vehicle EFs (mg/veh/km) for zinc and copper estimated from unpublished water quality data for three Auckland sites

Location	Traffic volume (vehicles per day)	Copper	Zinc
SH17 (nr Horseshoe Bush Rd), Dairy Flat	6,387	0.06	0.33
SH17 (nr Green Rd), Dairy Flat	6,387	0.29	1.64
East Coast Rd, North Shore	20,040	0.02	0.15

Source: Moores et al. (2010, p. 26)

Finally, Moores et al. (2010) compared their EFs to those reported internationally and in New Zealand (including Kennedy et al., 2002, and Timperley et al., 2005). The EFs determined for New Zealand were split into two groups: relatively low to mid-range estimates that are less than 0.086 mg/veh/km for copper and 0.45 mg/veh/km for zinc, and relatively high estimates that are greater than or equal to 0.12 mg/veh/km for copper and 0.87 mg/veh/km for zinc. They concluded that the estimated EFs fall into two groups, defined on the basis of traffic characteristics rather than traffic volumes. Relatively high EF estimates largely coincide with roads on which brake and tyre wear are likely to be greater than elsewhere whereas the low to mid-range estimates largely coincide with roads on which traffic may be generally expected to move freely. These groupings are consistent with the findings of, amongst others, De Silva et al. (2016), Huber et al. (2016), and Müller et al. (2020) cited above. On the basis of these groupings, Moores et al. (2010) recommended two sets of zinc and copper EFs for New Zealand; these relate to normal/free-flowing and congested/intersection

traffic respectively and are given in Table 6.11. The recommended EFs are used in the RSS model to determine zinc and copper loads from roads (Gardiner et al., 2016).

Table 6.11 Recommended zinc and copper vehicle EFs (mg/veh/km) for New Zealand estimates for zinc and copper

Traffic characteristics	Copper	Zinc
Normal (free flow)	0.047	0.28
Congested and intersection	0.095	0.62

Source Moores et al. (2010, p. 10)

6.4 Alternatives to vehicle emission factors for water quality modelling

The uncertainty surrounding the development and representativity of EFs means that alternative methods of characterising road runoff are usually used in stormwater and road runoff water quality modelling.

The relationship between the contaminant loads in road dust and runoff is at the heart of build-up/wash-off models (Sartor & Boyd, 1972; Sutherland & Jelen, 2003), which are used to predict water quality in most of the proprietary stormwater management models such as MIKE URBAN[18], SWMM[19] and Infoworks[20]. While EFs can be used to derive build-up rates for roads (eg, Auckland Regional Council, 2010b), most build-up/wash-off models calibrate model parameters against water quality data for land use classes where roads are included implicitly as part of the land use signal. In some cases, highways may be included as a specific land-use source. The amount of wash-off of accumulated road dust during a rainfall event will depend on the amount of accumulated material available and the dynamics of the rainfall event. The amount of build-up is a function of the accumulation rate and antecedent time period between rainfalls. Accumulation rates can be determined separately for sediment and contaminants, or by associating the contaminant load to the sediment load based on contaminant concentration ratios and the partitioning and fractionation of contaminants. Both the exponential and Michaelis–Menton build-up equations that are commonly used in stormwater quality modelling (eg, USEPA SWMM; Huber & Dickinson, 1988) have a calibrated parameter that sets the maximum accumulation value above which it is assumed that the amount of deposition on the surface is in equilibrium with the losses due to redistribution and resuspension. The amount of wash-off during a rainfall event depends on the intensity, duration and depth of the rainfall, the pH of the rain, the roughness and porosity of the road surface, and the physical properties (size, weight, shape) of the particles (Egodawatta et al., 2007; Egodawatta et al., 2013).

There have been several recent developments in build-up/wash-off models to improve model parametrisation and to incorporate site-specific factors and help explain the variation in contaminant concentrations spatially and temporally. Gunawardena et al. (2014) carried out statistical analyses to estimate metal loadings on sediment released by vehicles to provide road build-up parameters. Two analyses were undertaken representing different metal groupings: one for nickel and chromium that were associated with tailpipe emissions, and one for lead, copper and zinc that were associated with brake and tyre wear. Principal component analysis revealed traffic volumes and land use as the key determinants of the metal loadings.

[18] https://www.mikepoweredbydhi.com/products/mike-urban

[19] https://www.epa.gov/water-research/storm-water-management-model-swmm
https://www.innovyze.com/en-us/products/xpswmm

[20] https://www.innovyze.com/en-us/products/infoworks-icm

Regression analysis was then undertaken to determine a relationship between the grouped metal loadings and traffic (traffic volume, congestion) and land use parameters (percentage of industrial and residential land uses). Hong et al. (2017) used an artificial neural network to model sediment and particulate metal build-up on urban road surfaces based on land use (commercial, industrial and residential fractions) noting that each land use has its own traffic and road characteristics.

In New Zealand, the Modelled Estimates of Discharges for Urban Stormwater Assessments (MEDUSA) model (Charters, 2016; Charters et al., 2017) developed by the University of Canterbury uses a spatially distributed build-up/wash-off model to estimate loads discharged from urban surfaces, including roads, during rainfall. The model parameters for roads were derived from water quality monitoring of runoff from an asphalted road. Concentrations of runoff from urban surfaces, including roads and carparks, have recently been investigated with an eye to updating MEDUSA (Charters et al., 2021). In the model, the total zinc and total copper parameters are proportional to those determined from monitoring for TSS based on a strong correlation between the metals and TSS, and the parameters for dissolved zinc and copper are proportional to those for total zinc and copper respectively. The model was applied to several small urban catchments with low traffic-counts, but has not been applied to catchments with highways, although Lincoln Road in the Addington catchment is considered a major arterial road. MEDUSA was developed as a toolbox for ArcGIS, but has since been integrated into MIKE URBAN to provide an online tool for Environment Canterbury.[21] To date, the model is only available for Christchurch.

Also in New Zealand, the Auckland Council Contaminant Loads Model (CLM; Auckland Regional Council, 2010a, 2010b) is a steady-state simplification of a build-up/wash-off model that estimates annual loadings for sediment, metals and hydrocarbons on the assumption that over the course of a year, the amount of build-up and wash-off will be at equilibrium. The advantage of this method is that it does not require dynamic hydrological modelling or contaminant transport modelling. CLM was adapted by NIWA as a GIS tool for spatially distributed modelling called the Catchment Contaminant Annual Load Model (C-CALM; Semadeni-Davies et al., 2009; Semadeni-Davies et al., 2010); C-CALM uses the same yields as CLM. The yields from urban non-road surfaces from CLM were used to derive the urban yields used in the RSS model. CLM has six road classes; these are listed with their associated annual yields in Table 6.12. The annual loads from each road class are estimated by multiplying the yield by the total road area which is calculated as the product of the road length and width. The road width is estimated as 3.5 m/lane plus a 5 m verge on either side of the road. The yields for TSS were estimated by statistical analysis of road runoff quality in Auckland to derive EFs for sediment. The yields for zinc were estimated using a method similar to that used by Kennedy et al. (2002) based on manufacturer estimates of wear rates, the composition of tyres and traffic and fleet data. The copper yields were estimated using a ratio of 1:3 copper to zinc, based on the results of analyses of runoff samples. Similarly, the total hydrocarbons yield is estimated using a ratio of 22.7:1 of hydrocarbons to copper. Neither CLM nor C-CALM has been validated against monitored water quality data.

CLM and C-CALM have not been maintained for several years, and work on a new regional model for Auckland called the Freshwater Management Tool (FWMT) is ongoing (Bambic & Riverson, 2016, 2017; Grant et al., 2018). This model couples the LSPC (Loading Simulation Program C++; Tetra Tech Inc., 2009)[22] catchment hydrology and water quality model and the SUSTAIN (System for Urban Stormwater Treatment and Analysis IntegratioN; Shoemaker et al., 2009)[23] stormwater management model developed in the USA in a customised application. The FWMT splits the region into hydrological response units that

[21] https://www.dhigroup.com/global/references/apac/overview/effective-online-stormwater-quality-management-for-cities

[22] https://cfpub.epa.gov/si/si_public_record_Report.cfm?Lab=NERL&dirEntryId=75860&CFID=22884508&CFTOKEN=98267566

[23] https://www.epa.gov/water-research/system-urban-stormwater-treatment-and-analysis-integration-sustain

represent unique combinations of land use and catchment characteristics. Roads have been separated into a number of hydrological response units based on AADT (from < 1,000 to > 100,000 vehicles per day) and slope, and the parameters describing runoff quality from roads have been calibrated against existing water quality data (John Riverson, Paradigm Environmental, pers. comm., March 2021).

The Australian Model for Urban Stormwater Improvement Conceptualisation (MUSIC; eWater, 2014)[24] uses a stochastic modelling approach rather than build-up/wash-off equations whereby each land cover in the catchment is assigned a distribution of contaminant concentrations from which a concentration is drawn stochastically for each rainfall event under the assumption that over a long time period, the concentration distribution seen in receiving waters will approximate the actual distribution of contaminant concentrations. The contaminant concentration distributions are assumed to be log-normal and are defined by the mean and standard deviation of event mean concentrations for rainfall events specific to the contaminant and land cover type. Runoff from roads can be modelled implicitly within a land cover class, or explicitly as a separate user-defined source provided that contaminant concentration distributions are available. MUSIC has default concentration distributions for TSS, phosphorus and nitrogen derived from Australian water quality data, but users can provide their own local concentration distributions for these contaminants or swap TSS for other contaminants such as metals. Another model developed by eWater that has been used for catchment planning in mixed urban and rural catchments is SOURCE. SOURCE was used by the Greater Wellington Regional Council to route contaminant loads from various sources to Porirua Harbour (Moores et al., 2017). The yields used for roads and urban surfaces in the model were adapted from those used in the CLM.

Table 6.12 Traffic contaminant annual yields (g/m^2/year) used in the Auckland Council CLM

Road class (vehicles per day)	Number of lanes (width)	TSS	Zinc	Copper	Total hydrocarbons
< 1,000	2 (17 m)	28	0.0266	0.0089	0.2013
1,000–5,000	2 (17 m)	53	0.1108	0.0369	0.8387
5,000–20,000	2 (17 m)	96	0.2574	0.0858	1.9474
20,000–50,000	3 (21 m)	158	0.4711	0.1570	3.5645
50,000–100,000	4 (24 m)	234	0.7294	0.2431	5.5192
> 100,000	6 (31 m)	28	0.0266	0.0089	0.2013

Source: Adapted from Auckland Regional Council (2010b, p. 33)

The Stochastic Empirical Loading and Dilution Model (SELDM)[25] is an example of a stochastic modelling approach specifically developed to predict water quality from highways. SELDM was developed jointly by the United States Geological Survey and the Federal Highway Administration to help develop planning-level estimates of event mean concentrations, flows, and loads in stormwater from a site of interest and from the catchment area upstream of the site (Granato, 2013, 2014). SELDM uses information about a highway site, the associated receiving-water catchment precipitation events, stormflow, water quality, and the performance of mitigation measures to produce a stochastic population of runoff-quality variables. SELDM uses Monte Carlo methods to produce the random combinations of input variable values needed to generate the stochastic population of values for each component variable. The model has been used to predict phosphorus loadings to streams in North Carolina (Granato & Jones, 2015), and a national water quality database that provides input data to the model has been set up for the USA (Granato & Jones, 2019). This

[24] https://ewater.org.au/products/music/

[25] https://pubs.usgs.gov/tm/04/c03/

database includes water quality properties (eg, oxygen demand, conductivity, pH) and concentrations of sediment, nutrients, metals and organic compounds. SELDM does not include other contaminant sources and can only be applied to rural roads.

Another model specifically developed for highways is the Highway Agency Water Risk Assessment Tool (HAWRAT; Crabtree et al., 2008).[26] Like SELDM, HAWRAT does not include urban contaminant sources. HAWRAT is a spreadsheet tool that provides for an assessment of the impacts of highway runoff quality on receiving water quality, groundwater quality and flooding. Similar to the RSS model, this tool was designed to provide improved guidance on where, and to what level, treatment of runoff is required for UK highway designers to manage the risk of ecological impact from highway runoff. The receiving water quality assessment involves comparing estimates of receiving water concentrations of contaminants with toxicity-based threshold values, providing for an absolute assessment of risk. HAWRAT allows assessment of both acute effects, based on dissolved concentrations of copper and zinc, and chronic effects, based on sediment-bound concentrations of a range of metal and hydrocarbon compounds. HAWRAT estimates statistical distributions of contaminant concentrations in road runoff for use in the model based on a previous programme of highway runoff research (Crabtree et al., 2008). These concentrations are related to traffic volumes and climate information in the model.

[26] Now maintained by Highways England, the latest version, HAWRAT 2.0, was released in 2015.

7 Gap analysis

Air- and water-borne particulate emissions from road transport are important contributors to urban pollution. The major sources of particles from road transport are from combustion in the engine and emitted as exhaust, and mechanically generated particles from abrasion of brakes, tyres and the road, along with particles re-entrained from the road into the air by wind and the turbulence caused by passing traffic. Waka Kotahi estimates the environmental impact of the traffic on its roads using estimates of EFs from vehicle activities, both exhaust and non-exhaust. As emissions from vehicle exhausts fall, the non-exhaust fraction will become relatively more important, but it is the fraction that we know least about.

In previous sections we have examined the current understanding of emissions of NEEs from brake, tyre and road wear and resuspension of road dust. We found that there is a large range of estimated EFs from all NEE sources and many different approaches to deriving non-exhaust EFs, which can make even the most basic comparison of the results of different studies difficult. This means that deciding on the most representative EFs from those that have been determined overseas to the New Zealand context is not possible. The generalised gaps in knowledge and limitations of the determination of EFs from non-exhaust sources are summarised below and expanded on in the rest of this section.

Paucity of information and understanding

- There is a lack of up-to-date information on NEEs. Most of the studies cited were published before 2005.
- The metadata required to derive EFs from air and water quality data are often not reported. That is, while there have been numerous studies that have characterised air quality and the quality of road runoff and road dust with respect to the concentrations of the priority contaminants over recent years, information on traffic flows and runoff volumes required to calculate contaminant loads data have largely not been reported.
- There is limited understanding and reporting of the spatial and temporal variability in NEEs and therefore uncertainty in the representativity of overseas-derived EFs cited in the literature to New Zealand and of EFs largely determined for Auckland to other regions of New Zealand.
- There is a lack of information on the possible impacts of new technologies and changes in fleet composition on NEEs due to the limited number of studies into these impacts.

Variability in methods

- There is a lack of standard methods for determining EFs and uncertainties inherent in the sampling and analytical methods used.
- The metrics reported vary.
- The wear rates determined for brakes, tyres and roads vary because the composition and physical properties of these sources vary. Taken together, there is therefore variability in the chemical and physical properties of particles emitted by these sources.

Uncertainty

- Limitations in the ability to apportion NEEs to specific sources exist due to inadequate tracers and multiple sources of PM and contaminants.
- Uncertainties in the amount of resuspension of road dust, and the inability to separate resuspended PM from newly emitted PM, lead to the possibility of double counting PM in the determination of EFs.
- There are inconsistencies in and lack of comparison of EFs estimated from wear rates and those estimated from air and water quality.

7.1 Data sources

One of the objectives of this study was to identify and review new information that could be used to update the NEE EFs used by Waka Kotahi to reflect the current fleet composition and expected modernisation of the fleet, including improved brake and tyre technologies and increased use of EVs. However, we found that most of the studies used to develop non-exhaust EFs both in New Zealand and internationally were published before 2005.

For air quality, the most recent reviews of EFs to air referred to in this report (eg, AQEG, 2019; Amato, 2018; OECD, 2020; Piscitello et al., 2021) have cited much the same information as reported in previous reviews (eg, Boulter, 2005; Grigoratos & Martini, 2015; Lahuna et al., 2004; Thorpe & Harrison, 2008). The handful of more recent publications reflect the lack of research into NEEs over the last decade. Indeed, in the latest review by Piscitello et al. (2021), the most recent studies cited were published in 2010 for brake and tyre wear and 2013 for tyre wear.

The lack of recent data means that the input data to air and water quality models requiring EFs from non-exhaust sources have not been updated. The most recent versions of widely used emission inventories for air quality have not updated EFs from NEE sources (European Environment Agency, 2019; National Atmospheric Emissions Inventory, 2018; US Environmental Protection Agency, 1995a, 2020a) since the early to mid-2000s. For example, AQEG (2019) notes that the method set out by the European Environment Agency (2019) to estimate PM_{10} and $PM_{2.5}$ EFs from tyres and brakes to air is based on data from the 1990s, and the guidance from the agency has not been updated in 15 years. This method is embedded in the VEPM v.6 (Metcalfe & Peeters, 2020) used in New Zealand (see section 3.5.1). The latest version of the equivalent MOVES3 model released by the US Environmental Protection Agency (2020a) also relies on data that are at least 20 years old to derive EFs from brake and tyre wear.

However, the recent flurry of reviews into the effects of NEEs on air quality, including this report, indicates that NEEs are emerging as a topic of interest internationally and may presage renewed research activity in this area. Indeed, according to Roy Harrison (peer reviewer, pers. comm., April 2010), research is either underway or is being planned in the USA (California Air Resources Board), the UK (Department for Transport) and Germany (Federal Government). Harrison also stated that in preparation for new NEE standards, the EU is funding new research on low-emission materials for brakes and is planning for further research into tyre wear.

Most of the road runoff water quality and road dust data reported in the literature also date from the 1990s and early to mid-2000s (eg, Huber et al., 2016). The most comprehensive study on EFs to water undertaken in New Zealand dates back to the early 2000s (Kennedy, 2003a, 2003b; Kennedy & Gadd, 2000, 2003; Kennedy et al., 2002). The study determined EFs (overviewed in section 3.5.2) based on wear rates and the composition of brakes, tyres and roads, which were then compared to EFs determined from road dust collected in West Auckland. These were compared to EFs derived from sediment, zinc and copper loadings measured in road dust (see section 6.3). The most recent estimates of EFs to water in New Zealand were made by Moores et al. (2010) based on the analysis of water quality data collected from Auckland motorways in the mid to late 2000s. The EFs derived for zinc and copper have been adopted in the RSS model developed for Waka Kotahi (Gardiner et al., 2016). It should be noted that the general lack of any information on EFs to water is partly due to the types of models used for stormwater quality management where contaminants from roads tend to be calibrated rather than measured.

7.2 Estimation of emissions from brake, tyre and road wear

Section 3 reported that there is wide variation in the wear rates reported for brakes and tyres and the chemical and physical characteristics of PM emitted by brakes, tyres and road wear. This variation is summarised in Table 7.1 along with key references.

Table 7.1 Chemical composition of non-exhaust PM (most common traces in bold)

Source	PM_{10} (% of TSP)	$PM_{2.5}$ (% of TSP)	Main materials (> 1% in mass)	Wear rates (mg/veh/km)
Brakes	63–98%[a–d]	39–63%[a–d]	**iron, copper, barium, antimony**, zinc, aluminium, chromium, potassium, titanium, magnesium[g]	11–30 (passenger cars)[j]
Tyres	60%[d] 1% total tread wear[e,f]	42%[d] 0.4% total tread wear[e,f]	**zinc,** silicon, sulphur[h]	1–1,000 depending on vehicle size and driving conditions and driver behaviour[k]
Roads	50%[d]	27%[d]	**silicon, calcium,** potassium, iron[i]	3–80 mg/km PM_{10} to air depending on road condition and vehicle size[l]

[a] Garg et al. (2000); [b] Sanders et al. (2003); [c] Iijima et al. (2008); [d] European Environment Agency (2019); [e] Kreider et al. (2010); [f] McAtee et al. (2009); [g] Grigoratos and Martini (2015); [h] Panko et al. (2018); [i] Gustafsson (2018); [j] Kennedy et al. (2002); [k] Councell et al. (2004); [l] Gehrig et al. (2010)
Source: Adapted from OECD (2020, p. 25)

The high degree of variability in EFs and the ratios of PM_{10} and $PM_{2.5}$ from brakes and tyres seen in Table 7.1 is due to a range of possible reasons, including the following.

- Different material compositions and their associated properties are used in the manufacture of brakes and tyres.
- There are inconsistencies in the equipment and methods used to measure wear rates for brakes and tyres, which can include lab-based controlled abrasion tests as well as full-scale field tests. Equipment used to assess brake wear includes pin-on-disc tribometers, brake dynamometers and collection of dust from brake housing. Equipment used for assessing tyre wear includes rolling resistance machines, mobile load simulation machines and tread gauges. Each of these methods has their own associated assumptions and uncertainties.
- There are inconsistencies in the vehicle type, driving conditions and driver behaviour used in wear tests, such as acceleration and deceleration rates, cornering, and braking frequencies. The classification of vehicles varies, with some studies referring to vehicles by weight (eg, light and heavy vehicles) and others to vehicle function (eg, passenger cars, light commercial, trucks and buses). Few studies have defined their classification of vehicle types. Most of the studies cited have focused on emissions from light vehicles, which means that there are less data available for heavy vehicles.
- The spatial and temporal variation in the road, climate and driving conditions assessed means that the representativity of EFs developed for a specific site and time (eg, diurnal and seasonal changes) may not be transferrable to other locations and times.

Another concern is that there is relatively little information published on some aspects that can affect NEEs that may not be related to traffic. For example, there have been comparatively few studies that have assessed road surface wear under controlled conditions, and the effect of different types of road surfaces on tyre wear is largely unknown. Most studies of this type have been undertaken in cold regions and relate to the use of studded winter tyres that are not used in New Zealand. Moreover, the metrics and units used to

describe wear rates and EFs can vary between studies, making it difficult to compare the results directly. For example, EFs for brake and tyre wear can be provided as total emissions or EFs to air only, and it is not always clear which metric has been provided. For emissions to air, PM emissions can be reported as TSP with PM_{10} and $PM_{2.5}$ calculated as a proportion of TSP, or directly by particle-size class.

The complexity of interactions between the factors that affect emissions from NEE sources can lead to differences in emission rates and the physical (eg, particle size distribution) and chemical properties (eg, compounds emitted) from the same source. The inability of simulating all combinations of these factors make the outcomes of emission studies somewhat stochastic in nature.

The priority contaminants considered in this report were PM_{10} and $PM_{2.5}$ for air quality, and zinc and copper for water quality. These are the most commonly reported NEE indicators in the literature we reviewed; however, other contaminants present in NEEs can have detrimental effects on human and ecosystem health. We note that most of the studies into the chemical composition of emitted particles reported only elemental composition and have not provided information on compounds emitted or fractionation of contaminants into particle-size classes, making it difficult to assess their possible impacts on human or environmental health.

7.3 New technologies and fleet modernisation

There is little information available on the possible effects of new technologies on NEEs at either the vehicle or fleet level, which has ramifications for the development of future EFs. Modelled estimates of the impact of new technologies arrive at a large range of forecast EFs due to different initial assumptions of expected changes.

For brakes, innovations include changes in the materials and design to reduce wear and the development of PM collector systems to reduce emissions from brakes. While the OECD (2020) and Gramstat (2018) state that these technologies look promising, neither provide quantitative information on emissions or wear rates. They also state that the costs and technical difficulties mean that these innovations are not yet available or have limited the market penetration. While it is noted that low-copper brakes are being mandated in some jurisdictions (eg, California and Washington in the USA), we found no studies into their efficacy for improving water quality in road runoff or receiving waters – which likely reflects the novelty of these regulations. It is uncertain whether New Zealand will ban or restrict copper in brakes, although low-emission brake pads are available and there has been an emerging interest in this in New Zealand's stormwater management sector. If a ban is put in place, it is uncertain what the legacy of copper accumulation in roadside soils will be, but studies have shown that roadside soils are a potential source of historical contaminants, such as PAHs from coal-tar sealants (Depree et al., 2006; Depree & Fröbel, 2009) and lead from petroleum.

There is also little information on wear rates from low rolling resistance tyres. The OECD (2020) states that there is no evidence to date that low rolling resistance tyres have reduced emissions, but more research is required.

The Ministry of Transport projects that the proportion of light EVs in the national fleet could increase by between 40% and 50% over the coming decades, which could affect NEEs by increasing tyre wear, due to increased vehicle weight, while reducing emissions from brakes due to the use of regenerative braking. It is generally accepted that EVs are heavier than equivalent-sized ICEVs; however, there is evidence from Australia that people who buy EVs tend to buy smaller cars (Smit, 2020), which could lead to an overall reduction in car sizes as EVs become more prevalent and replace larger vehicles such as SUVs. While we were able to find qualitative modelled emission rates for EVs, we were not able to find comparisons of measured EFs. The modelling that has been undertaken suggests that there will be little or no change in PM_{10} emissions and only modest changes in $PM_{2.5}$ as increased emissions from tyres balance out emission

reductions from regenerative brakes. We were unable to find EFs for particulate contaminants from EVs, but we can speculate that the ratio of zinc and copper found in NEEs is likely to change.

The effect of self-drive and driver-assist cars may also result in reduced emissions as driver behaviour is evened out, resulting in consistent and reduced speeds and therefore less braking, acceleration and deceleration. However, we found no information on EFs for these vehicles.

7.4 Estimation of emissions to air and water

Like the estimation of EFs from vehicles discussed above, there are no standard methods for determining EFs to air and water from measured air and water quality. Moreover, methods for selecting, sampling and analysing the quality of air, road runoff and road dust are many and varied, and each has its own set of inherent uncertainties. There are also inconsistencies in the metrics reported and, for water, a lack of information on contaminants other than zinc and copper for water.

A major challenge in determining EFs from NEE sources to the air and water is that vehicles may not be the only contaminant, particularly in urban areas. This means that finding suitable tracers that are specific to an NEE source that can be used to apportion NEEs is problematic. Although alternative tracers, such as barium or antimony can be used for measurements of airborne NEE from brakes, source apportionment studies in New Zealand have so far found them to be below the limits of detection.

A related issue for air quality is the near impossibility of separating PM freshly released by vehicles from resuspended PM, especially with respect to emissions from tyre and road wear, since the PM has the same sources and therefore similar chemical and physical characteristics (Panko et al., 2018). Resuspension in particular is highly dependent on location because the road dust reservoir is not constant and is dependent on road type, construction and maintenance, vehicle numbers, local climate and geology. Most studies into resuspension assume an equilibrium state where each passing car resuspends as much as it deposits, which is only true under unchanging conditions. For water quality, build-up/wash-off models represent this equilibrium state by calibrating a maximum build-up mass for road dust.

Where reported, EFs to water are generally provided as EFs that do not separate tailpipe emissions and NEEs; instead, it is assumed that the priority contaminant metals, copper and zinc, are associated with brake and tyre wear respectively. The statistical analyses by Huber et al. (2016) and Sampson (2017), which showed that non-traffic-related factors also affect the quality of road runoff and road dust respectively, have implications for the determination of EFs to water and reiterate the importance of careful selection of monitoring. To this end, we note that the studies undertaken in New Zealand to develop EFs have sought to isolate road runoff from stormwater by selecting small, well-defined sections of road with no connection to stormwater drainage from other surfaces (Kennedy & Gadd, 2003; Timperley et al., 2005) or by sampling road runoff from highways outside urban areas (Moores et al., 2010; Moores et al., 2008). However, zinc from galvanised street furniture and atmospheric deposition of metals from other sources are possible. Similarly, studies to determine airborne PM from roads (tailpipe and NEEs) seek to use a section of road with as little interference from surrounding roads as possible, such as in a tunnel, although disentangling local road emissions from other sources can be difficult where, for example, road dust may be impossible to separate from crustal material from other sources.

Another challenge to determining EFs to both air and water is the spatial and temporal variations in the factors that affect NEEs, including road and driving conditions and climate. These differences are poorly understood and have complex interactions. For example, emissions are generally higher under congested traffic conditions, which means that the time of day when samples are taken is important (ie, peak vs off-peak traffic volumes). Wet conditions following rainfall can result in lowered emissions to air and resuspension as road dust is washed from road surfaces or is caked onto the road. Moreover, the recovery

of emission rates to air after rain is more rapid in warm climates. This means that emission to air will generally be lower in temperate climates compared to dry climates and during wet seasons. Seasonal differences in water quality are driven by changes in rainfall dynamics; namely, the frequency, intensity and duration of rainfall. Since EFs are generally determined from total loads over a set time period, a longer sampling period can mitigate against short-term changes in emissions. Wind is an obvious factor that affects the resuspension and transport of airborne PM. Less obvious is that road roughness can shelter road dust from wind, resulting in less resuspension. Road roughness can also affect the entrainment and transport of road dust in runoff such that particulates may only be entrained following high-intensity rain falls.

These local and seasonal variations in emissions make it difficult to generalise emission determined for a specific location and time period to other locations and times, which has implications for New Zealand because the country covers a range of climate zones (from hot and dry to temperate rainforest) and has variable topography (from mountains to coastal plains). Most of the monitoring to date in New Zealand to determine EFs to air and water has been largely undertaken in Auckland, over a limited period of time and at a small number of sites. While air quality source apportionment studies have been carried out in many towns and cities in New Zealand, the largest dataset comes from Auckland. Consequently, studies have been able to disaggregate more sources in Auckland than elsewhere (Davy & Trompetter, 2020).

An issue common to both air and water quality is that while there have been numerous studies that have characterised roadside air quality and the quality of road runoff, neither EFs nor the metadata required to determine EFs are often provided. EFs are more commonly reported in air quality studies because they are required for input to dispersion models. However, it is still the case that most studies, particularly source apportionment studies, only report contributions of individual sources to total airborne PM and do not supply enough information, such as traffic counts or wind conditions, with which to calculate EFs. Similarly, for water quality, comparatively few studies have provided contaminant loadings or traffic data required to determine EFs. Indeed, we were only able to find a handful of EFs to water reported internationally and were unable to find any reported after Moores et al. (2010). We note that EFs are seldom used for water quality modelling, and where contaminant accumulation rates for roads are required for modelling, these are usually calibrated against contaminant concentrations in road runoff.

While air quality reviews have compared EFs reported in the literature that have been derived using different methods (Grigoratos & Martini, 2014; Piscitello et al., 2021), such as from emission inventories, direct measurement or receptor modelling, the wide range of EFs calculated by each of the methods and the low number of citations for each method class means that is not possible to draw any conclusions on whether these methods give equivalent EFs.

We found only one comparison between EFs determined from water quality monitoring as from brake, tyre and road wear (Kennedy & Gadd, 2003). This study compared zinc and copper emissions generated by the VFEM-W, which uses EFs determined from wear studied as input, and the contaminant loads of zinc and copper in road dust and runoff sampled in West Auckland. This study showed poor agreement between the modelled and measured contaminant emissions. The discrepancies were ascribed to contaminants from other sources; for example, the high lead concentration in road dust was ascribed to lead bound to soil particles from historical tailpipe emissions of lead from petrol, and high uncertainty in the wear rates and metal concentrations determined by Gadd and Kennedy (2000) and Kennedy and Gadd (2000).

8 Recommendations for Stage 2

The previous section of this report has identified a large number of uncertainties and limitations in the determination of emissions from non-exhaust sources. These knowledge gaps mean that the EFs currently used by Waka Kotahi in the VEPM and the RSS model may be unrepresentative of NEEs in New Zealand. The age of the research used to develop the EFs means that the EFs are likely to reflect neither the current composition of the national fleet nor modernisation of the fleet. Moreover, there has been very little research on NEEs in New Zealand, and it is uncertain that EFs determined overseas, even from the most recent studies, can be transferred to New Zealand given the high degree of variability in EFs reported generally and the heterogeneity of EFs spatially due to regional and international differences in climate, roading materials and driving conditions.

The objective of Stage 2 of the research programme proposed by Waka Kotahi is to fill the gaps identified in this report. To do this, we propose that Stage 2 should have two tranches. Tranche 1 is to undertake (a) a re-analysis of existing air quality data held by GNS Science to apportion airborne PM from traffic to vehicles and to determine the needs for any further air quality monitoring in Tranche 2, and (b) a sensitivity analysis of selected air and water quality models that use EFs to get an understanding of how the choice of EF affects model outputs. Based on the outcomes of these analyses, Waka Kotahi could decide to either retain the current EFs (status quo) and not continue with Tranche 2 or undertake new monitoring and modelling to develop and test a new set of EFs for New Zealand in Tranche 2.

While most of the recommendations can be applied to EFs for both air and water quality, they are most relevant to air quality modelling. This is because water quality models generally do not use EFs directly and, with the exception of the RSS model, there is no water quality model in New Zealand that is currently used by Waka Kotahi for highway runoff. The RSS model does have a water quality component and calculates mean annual zinc and copper loads and concentrations using the EFs determined by Moores et al. (2014); however, it is not intended to be used to predict absolute estimates of runoff quality.

8.1 Tranche 1

8.1.1 Re-analysis of GNS Science air quality data

The purpose of this task is to re-analyse existing air quality data held by GNS Science to provide 'ground-truthing' of the impact of NEEs on air quality and existing EFs. Two approaches are available: to use observational source apportionment data to evaluate the accuracy of modelled impacts using existing EFs, or to inverse-model real-world EFs from observed air quality by calibrating the EFs. This task would provide interim estimates of PM_{10} and $PM_{2.5}$ EFs that could either confirm or replace exiting EFs.

GNS holds an archive of filter-based, time-integrated PM samples that have been analysed for elemental composition. PM samples have been collected from approximately 40 regulatory authority monitoring sites across New Zealand. In addition to the urban locations, several studies have targeted source-specific PM composition, including the Johnson Hill Tunnel study (Davy et al., 2011).

Multivariate analysis of PM sample composition effectively 'fingerprints' the sources contributing to airborne PM concentrations. If suitable ancillary data are available for the monitoring sites, such as traffic numbers, VKT, and dispersion conditions, then estimates can be made of the emissions per vehicle or stretch of road. Note that these EFs would be for total emissions rather than for NEEs *per se*; however, if the PM fine (ie, $PM_{2.5}$) and coarse fractions (ie, particles between 2.5 and 10 µm) have been analysed separately, in the absence of other sources it can be assumed that the coarse fraction is from NEE sources. While separate

analysis of different size fractions has been undertaken at some sites (eg, Auckland, see Figure 5.2), this has not been undertaken for all sites.

There are two possible options for estimating vehicle EFs from the GNS data (Perry Davy, GNS Science, pers. comm., March 2021):

1. The Johnsons Hill Tunnel studies (Davy et al., 2011) identified six primary sources that contributed to PM_{10} concentrations: light duty vehicles, heavy commercial vehicles, smoky vehicles, road dust, biomass (wood) burning, and marine aerosols (sea salt). The light and heavy vehicle emissions included both exhaust and brake dust, while tyre wear was included in the road dust component. Since the study sampled solely vehicle emissions, then if the number and type of vehicles passing through the tunnel are known, the EFs per vehicle can be estimated.
2. Source apportionment has been carried out in several urban locations around New Zealand. These identify the different sources from all the surrounding roads impacting the monitoring location. If VKT is known by Census Area Unit[27], then EFs could be derived for all the roads impacting a point, thus avoiding the need to isolate the signal from a single road. This has not been done anywhere in the world to our knowledge.

8.1.2 Sensitivity analysis

The purpose of this task is to determine the sensitivity of air and water quality outcomes (as estimated using appropriate models) to EFs to get an understanding of how the uncertainty associated with EFs could affect estimated air and water quality. Where models are coupled, the analysis would also be able to investigate how that uncertainty propagates from one model to the next. The steps would be to first select the model or suite of models to be used to estimate air or water quality and then undertake a sensitivity analysis. The outcome of the sensitivity analysis would be used to guide the decision on whether to continue with Tranche 2 based on criteria developed with Waka Kotahi.

The first step in the analysis is to decide which models should be used for the evaluation of the existing EFs. The choice of model(s) should be guided by, amongst other factors, the purpose of the modelling, data availability and the scale of interest. This choice will dictate the data requirements for both running and calibrating the models in Tranche 2. Ideally, the models should be transferable to different locations around the country, be freely available to allow multiple users to have access to them and be able to provide outputs that fulfil the needs of Waka Kotahi.

For air quality, the VEPM is currently used by Waka Kotahi to estimate vehicle emissions for a particular length of road based on fleet composition and traffic numbers. The VEPM includes emissions from both tailpipes and non-exhaust sources. To determine air quality at a specific location, the VEPM is coupled to a suite of other models as described in section 5.4.2 (eg, weather and air dispersion models). These models would need to be selected and run together to establish the impact of the EFs on air quality predictions for a range of common and critical scenarios.

With the exception of the RSS model addressed above, there is currently no national water quality model used by Waka Kotahi, and we know of no specific highway or road runoff models that have been developed or applied in New Zealand. However, highways and roads have been included as contaminant sources in urban drainage models and catchment water quality models (see section 6.4). Most of the models cited in section 6.4 do not use EFs directly as input, although EFs have been used to derive model parameters such as contaminant accumulation rates. This means that, depending on the water quality model chosen, it may

[27] It is our understanding that VKT by Census Area Unit is available from Waka Kotahi.

not be possible to run a sensitivity analysis for EFs. It should be noted that NIWA has been engaged in another project with Waka Kotahi (WBS 80002569 – RSS model improvements) to evaluate the RSS model and to undertake a sensitivity analysis of the model parameters. Since the output of the model is a relative risk value, the model should be reasonably robust to the EFs chosen. While this work is ongoing and cannot be reported here at this stage, limited analyses conducted at the time of the development of the model did indicate that risk estimates were relatively insensitive to changed EFs (Gardiner et al., 2016)

For the sensitivity analysis, the selected model(s) would be set up and run using default parameters, including the existing EFs, and generic input data. There are a number of different methods that can be used for sensitivity analysis, but the general principle is to change each parameter incrementally over the expected range of the parameter, either alone or in combination with other parameters, to see how the model output changes. Assessing the other parameters in combination with EFs would enable an assessment as to whether the EFs are correlated with other parameters and whether there are possible compensating errors between parameters. The method of sensitivity analysis should be chosen to reflect the type of modelling undertaken and the number and type of parameters to be tested.

A low sensitivity to EFs would give more confidence to retaining the existing EFs. If the model predictions have high sensitivity to EFs, then the choices are to either accept the uncertainty associated with the EFs or to continue with Tranche 2, which should be informed by the sensitivity of outcomes relative to policy objectives (ie, a significance test, such as whether choice of EF influences probability of compliance with standards or other policy objectives). While the latter option would not alter the sensitivity of the models to EFs, evaluating model performance and, if required, developing new EFs could give greater confidence to their use in models.

8.1.3 Status quo

Under the status quo option, Waka Kotahi would continue to use either the current EFs or, for air quality, those derived from the GNS data, in its models while acknowledging the uncertainty and limitations of these EFs. This choice would be the most pragmatic option where the model outcomes have low sensitivity to EFs.

For air quality, the VEPM contains the same method that is recommended for use by the European Environment Agency (2019). While there has been concern expressed over the age and limited amount of data used to develop the EFs and the relationships between emissions and vehicle weight and speed (AQEG, 2019; OECD, 2020), the method and EFs from the European Environment Agency (2019) emission inventory are still in use internationally. For example, the method was used, along with UK National Atmospheric Emissions Inventory (2018) EFs, to determine typical emissions of PM_{10} from highways and rural and urban roads in the UK (AQEG, 2005, 2019). The most recent study we found that uses the method was published earlier this year (Beddows & Harrison, 2021). We also note that alternatives, such as applying the AP-42 or MOVES3 methods for estimating vehicle emissions developed by the US Environmental Protection Agency (1995a, 2020a), are also hampered by the age and paucity of the NEE data. Indeed, a common finding in this report is that most of the EFs for NEEs reported in the literature have been developed using the same set of studies dating from the late 1990s and early 2000s. Continuing to use the method and EFs within the VEPM would imply an uncertainty that is consistent with other applications internationally.

We also note that the flurry of reviews over recent years (AQEG, 2019; Amato, 2018; Amato et al., 2014; OECD, 2020; Piscitello et al., 2021) shows a renewed interest in NEE research that has been at least partially driven by technological innovations such as low-emission brakes and more reliable batteries for EVs, particularly given reductions of tailpipe emissions as EVs become more common, which has increased the relative importance of NEEs with respect to total emissions. These reviews have all called for further research into NEEs. There have also been calls for the development and adoption of standard methods of

sampling and analytical methods and publication of metadata (eg, Mathissen et al., 2018) to enable the determination of EFs that are comparable between studies. As noted above, new research has been either initiated or is being planned in Europe and the USA (Roy Harrison, peer reviewer, pers. comm., April 2010). It is therefore our opinion that the next five to 10 years will see new research into NEEs internationally that could be translated into EFs that are more relevant to New Zealand and could be adopted by Waka Kotahi.

8.2 Tranche 2: Monitoring and modelling

This option is to undertake new monitoring to provide purpose-collected data that can be used to evaluate the performance of the selected models using existing EFs – or for air quality, the EFs derived from the GNS data – and, if required, develop new EFs from the monitored data. The process would be similar for both air and water quality modelling. The process steps are described in the rest of this section. The ideal situation would be to monitor air and water quality at the same times and locations to gain a comprehensive picture of the fate of NEEs in the environment. Whether Tranche 2 is undertaken, and the monitoring campaign and methods used for model evaluation, will largely depend on the outcomes of Tranche 1. For this reason, specifics of the experimental design are not given here.

8.2.1 Site selection

Given the high spatial variability in EFs and the tendency for previous monitoring of roadside air quality and highway runoff to be in Auckland, sites for monitoring should be located in different parts of the country to represent different combinations of climate and driving conditions (eg, traffic numbers, congestion, urban vs rural driving styles). The number of sites required will need to be determined with Waka Kotahi to ensure adequate monitoring for target conditions.

For rural roads, the sites should be as isolated as possible from other contaminant sources on air and water quality. For urban roads, background sources will need to be assessed.

Air quality sites should be located on open ground with no obstructions such as trees or buildings. It should be possible to install monitoring equipment in a transect across the road perpendicular to the prevailing wind and at a variety of distances from the road to get upwind and downwind air quality samples. Another option is to locate air quality sites in tunnels to eliminate the need for dispersion modelling.

Water quality sites should have well-defined catchment boundaries, ideally consisting of a section of road with a single drainage outlet leading directly to the stream with no stormwater treatment (including roadside swales). The sites will need to be located near to easily accessible streams where it is possible to safely install monitoring equipment.

8.2.2 Monitoring campaign

One of the issues with using existing data for modelling is that these data are often not in a form that is usable for modelling and do not include metadata required to derive EFs (eg, see general discussion of modelling needs for river water quality in Davies-Colley et al., 2011). For this reason, we recommend undertaking a monitoring campaign to obtain purpose-collected data suitable for running the selected model(s) and for model calibration and testing.

Recommendations for monitoring are as follows:

1. The monitoring period should continue over a period long enough to provide data from both model set-up and model evaluation and capture seasonal differences in air and water quality. For water quality, the data at each site should cover a wide range of rainfall events so that they are representative of the range

of rainfall intensities, durations and frequencies (and therefore antecedent dry periods) experienced at each site. It should be noted that emissions to air can be expected to peak after extended dry periods.

2. Road dust should be collected and analysed at each site to allow calculation of total contaminant loads and assessment of resuspension.
3. Input data required for modelling will vary for different models and can include, for example, wind speed and direction for air quality and rainfall and stream flow data for water quality as well as EFs.
4. Include any metadata required to characterise the site and to develop new EFs, such as road surface type, fleet composition, traffic counts and surrounding land use and land cover required to characterise the sites.

8.2.3 Evaluation of model fit

The intention of this step is to see whether the selected model(s) can capture air and water quality using the existing EFs. A common method for testing models is to split the data from each site into two sets, one for model set-up and the other for testing. Under this method, the set-up data are used to calibrate the model against the air/water quality data from the same time period. In this case, the calibration would be used to set the value of the parameters other than EFs. The model(s) would then be run and tested, using some evaluation criteria, against the remaining air/water quality data.

Good model fit using existing EFs means would give confidence in their continued use. Poor model fit would trigger the next step in this option – to develop new EFs on the basis of receptor modelling.

8.2.4 Receptor modelling

Receptor modelling would be used to estimate new EFs that are representative of NEEs for each monitoring site by apportioning the contaminant loads determined from monitoring to different NEE sources. Receptor modelling methods and assumptions are discussed in section 5.1.2 for air quality. For water quality, we know of no study that has apportioned contaminant loads to NEE sources; although, in the absence of other sources, zinc is generally attributed to tyre wear and copper to brake wear. Monitoring of runoff from different urban surfaces, such as that presented in Charters et al. (2021), could be used to help apportion urban contaminant sources. Once apportioned, the contaminant loads attributed to each source would be used to calculate EFs that are representative of different regions and conditions according to the number of vehicles travelling on the road during the monitoring period.

8.2.5 Model re-evaluation

This step would be to repeat the model evaluation outlined above using the new representative EFs to assess whether these have improved model fit. If the model(s) still has poor fit, other parameters in the model may need to be investigated.

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Appendix A: New Zealand fleet composition

Vehicle classification

Table A.1 presents the Ministry of Transport classification of vehicles for New Zealand; this classification is consistent with international standards (eg, European Environment Agency, 2019) and is primarily based on vehicle body type and weight. In this report, with exclusion of vehicles classed as either motorcycles or miscellaneous, 'light vehicles' refers to any vehicle that weighs up to 3,500 kg while 'heavy vehicles' refers to those with a weight greater than 3,500 kg. We make no distinction between passenger and commercial light vehicles.

Table A.1 Ministry of Transport vehicle categories

Vehicle category	Motor vehicle register vehicle type	Weight class
Light passenger vehicles	• Passenger car/van	Up to 3,500 kg
Light commercial vehicles	• Goods van/truck/utility • Motor caravan • Bus	Up to 3,500 kg
Buses*	• Bus (including minibus)	Over 3,500 kg
Trucks	• Passenger car/van • Goods van/truck/utility • Motor caravan	Over 3,500 kg
Motorcycles	• Motorcycle • ATV • Moped	NA
Miscellaneous**	• Mobile machine • Special purpose vehicle • Tractor • Agricultural machine	NA

* Vehicles classified in the Motor Vehicle Register as buses weighing 3,500 kg or less are counted as light commercial vehicles because they are usually physically identical to vans.
** A small number of vehicles are classified as 'miscellaneous'. These are generally not included in the Ministry of Transport's statistics. Many of these vehicles are exempt from licensing.

Current fleet composition and trends

The fleet data presented in this section come from Ministry of Transport (2018) fleet statistics for 2018 and data summaries published online.[28] The fleet composition is summarised from the 2018 fleet statistics in Table A.2 and Figure A.1 nationally and in Figure A.2 by region. In 2018, the fleet consisted of approximately 4.29 million vehicles, of which 91.4% were light vehicles, 3.7% were heavy vehicles, and the rest were motorcycles (4.1%) and miscellaneous vehicles (0.8%). Total fleet numbers increased by 60% between 2000 and 2018; there was an increase of 57% for light vehicles and 60% for heavy vehicles.

[28] https://www.transport.govt.nz/statistics-and-insights/fleet-statistics/2018-annual-fleet-statistics/ (date of access 27 November 2020)

Table A.2 Fleet composition (1,000 vehicles) (percentage increase in vehicle numbers relative to the year 2000 in parentheses)

Year	Light Duty	Heavy Duty	Other	Total
2000	2,495.0	100.2	90.4	2,685.7
2001	2,563.6 (3%)	102.7 (3%)	91.2 (1%)	2757.5 (3%)
2002	2,647.9 (6%)	107.0 (7%)	93.3 (3%)	2848.2 (6%)
2003	2,759.4 (11%)	112.5 (12%)	96.7 (7%)	2968.6 (11%)
2004	2,866.9 (15%)	119.8 (20%)	102.3 (13%)	3,089.0 (15%)
2005	2,967.1 (19%)	126.3 (26%)	111.8 (24%)	3,205.1 (19%)
2006	3,029.6 (21%)	131.0 (31%)	122.7 (36%)	3,283.4 (22%)
2007	3,088.6 (24%)	135.9 (36%)	134.7 (49%)	3,359.2 (25%)
2008	3,108.6 (25%)	138.8 (39%)	148.8 (65%)	3,396.3 (26%)
2009	3,099.8 (24%)	138.3 (38%)	153.6 (70%)	3,391.7 (26%)
2010	3,122.3 (25%)	137.1 (37%)	155.2 (72%)	3,414.6 (27%)
2011	3,117.5 (25%)	135.9 (36%)	155.9 (72%)	3,409.3 (27%)
2012	3,165.7 (27%)	136.0 (36%)	159.0 (76%)	3,460.8 (29%)
2013	3,243.5 (30%)	138.3 (38%)	164.6 (82%)	3,546.3 (32%)
2014	3,359.1 (35%)	142.0 (42%)	174.2 (93%)	3,675.3 (37%)
2015	3,482.7 (40%)	145.7 (45%)	182.7 (102%)	3,811.0 (42%)
2016	3,631.8 (46%)	149.6 (49%)	190.5 (111%)	3,971.9 (48%)
2017	3,790.5 (52%)	154.9 (55%)	199.4 (120%)	4,144.8 (54%)
2018	3,920.4 (57%)	160.2 (60%)	209.3 (131%)	4,289.9 (60%)

Figure A.1 Change in fleet composition since 2000

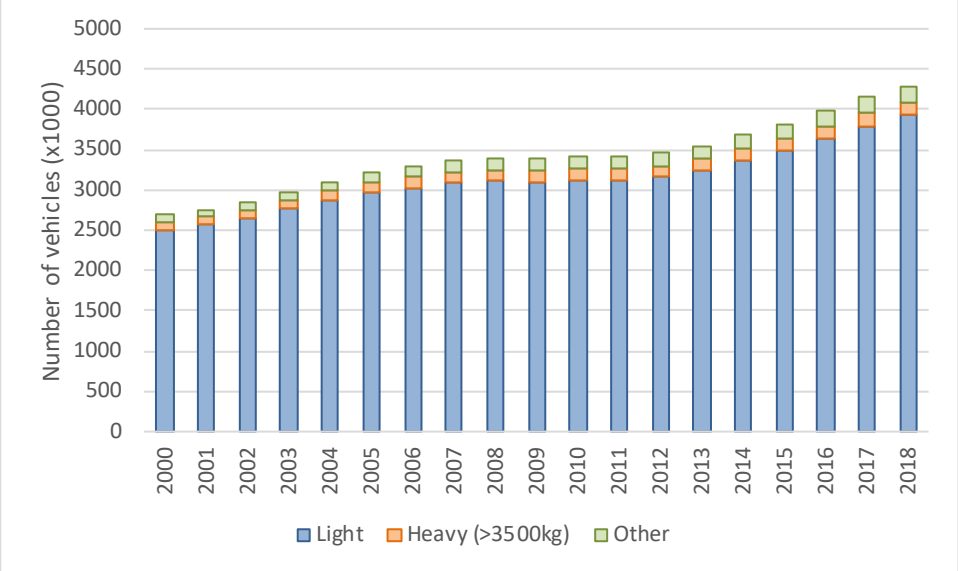
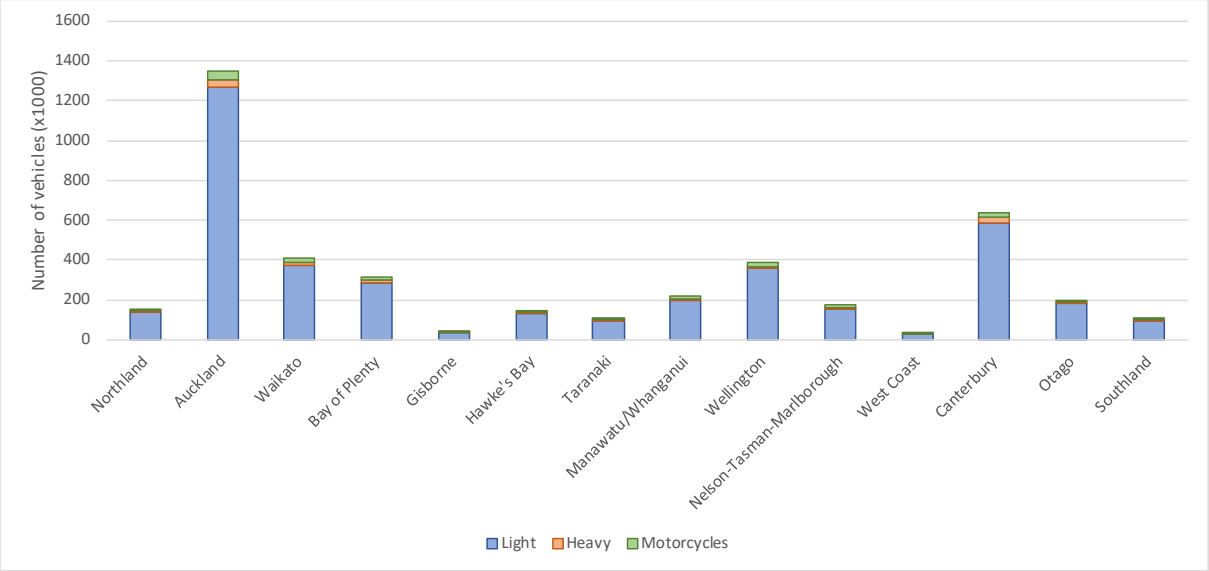
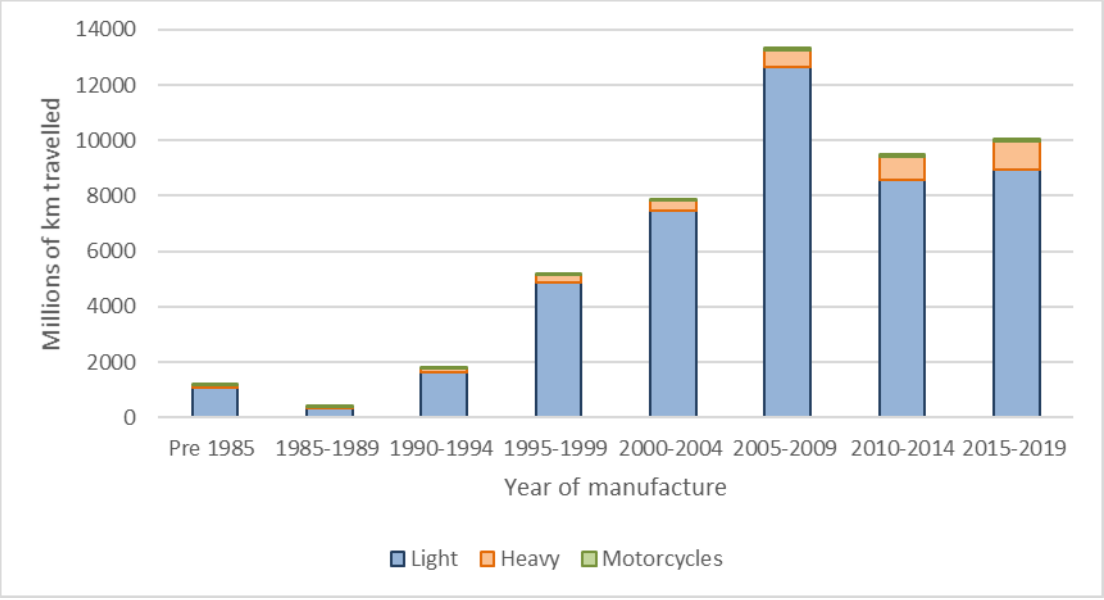


Figure A.2 Fleet size and composition, by region for 2018



The vehicle age and distance travelled (VKT) in 2018 are plotted by vehicle class and year of manufacture in Figure A.3 and Figure A.4, respectively. Vehicle age is also reported in Table A.3. The average age of light vehicles in 2018 was 14.1 years – which the Ministry of Transport reports is several years older than light vehicles in other comparable countries. In Europe, for example, the average age of a light vehicle is 10.8 years.[29] Around 76% of light vehicles are less than 20 years old and 31% were manufactured after 2010. For heavy vehicles, 55% are less than 20 years old and 28% were manufactured after 2010. The heavy vehicle fleet is older than the light vehicle fleet in part due to increased imports of used trucks from Japan following the banning of older vehicles in some Japanese cities in the 2000s. However, there are now more new trucks than used trucks entering the heavy vehicle fleet.

Figure A.3 Kilometres travelled (VKT) in 2018, by vehicle class and year of manufacture



[29] https://www.acea.be/statistics/tag/category/average-vehicle-age (date of access 30 November 2020)

Figure A.4 Fleet vehicle age 2018: year of manufacture, by vehicle class

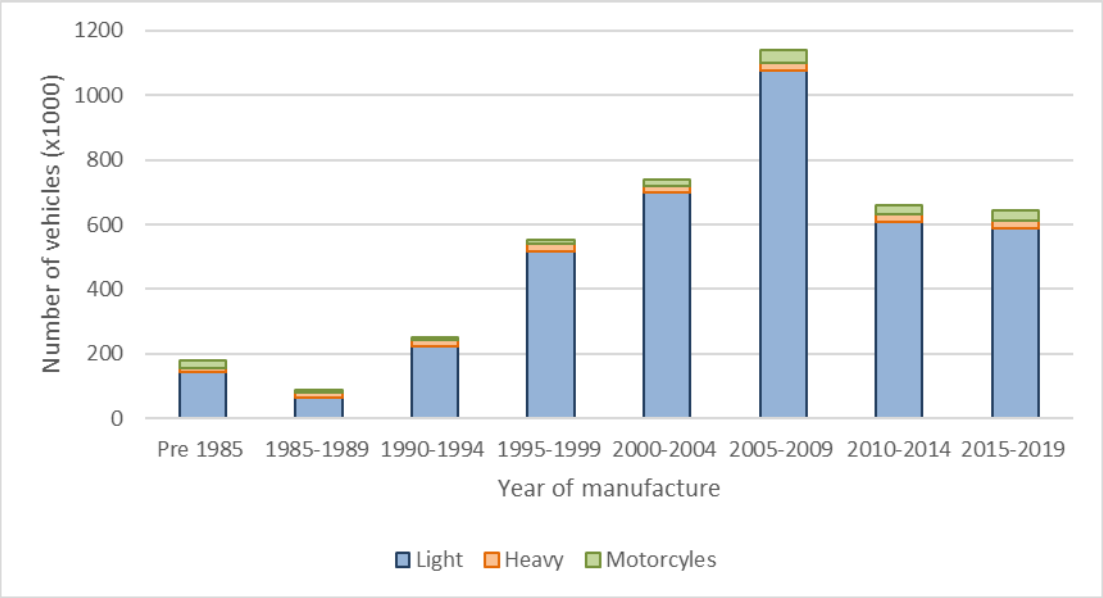


Table A.3 Fleet vehicle age 2018: number of vehicles (×1,000) by year of manufacture (percentage of vehicles within each class in parentheses)

Year of manufacture (5-year blocks)	Light vehicles		Heavy vehicles		Motorcycles	
Pre-1985	142.8	(4%)	11.9	(7%)	24.2	(14%)
1985–1989	65.1	(2%)	14.2	(9%)	11.2	(6%)
1990–1994	221.9	(6%)	22.5	(14%)	7.8	(4%)
1995–1999	516.4	(13%)	24.3	(15%)	11.9	(7%)
2000–2004	701.1	(18%)	18.5	(12%)	19.7	(11%)
2005–2009	1074.6	(27%)	24.1	(15%)	40.8	(23%)
2010–2014	609.3	(16%)	21.3	(13%)	28.9	(16%)
2015–2019	589.2	(15%)	23.5	(15%)	32.2	(18%)

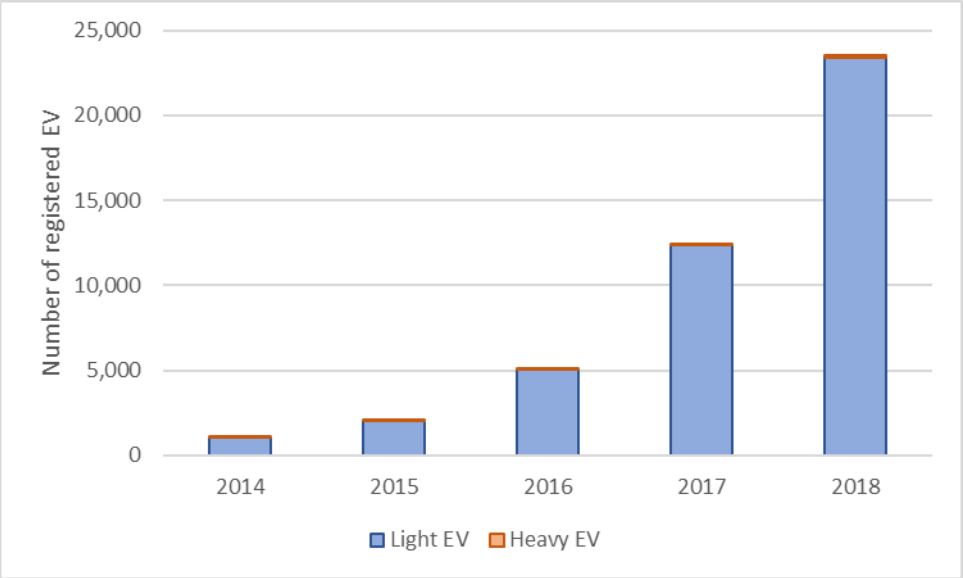
According to the Ministry of Transport,[30] there are comparatively few EVs (including hybrids) in the fleet compared to other fuel types. Around 0.4% of light vehicles and 0.1% of heavy vehicles are currently electric or hybrids. Since 2014, the number of light EVs has risen exponentially (Table A.4 and Figure A.5); however, the increase in heavy EVs has been more modest.

[30] https://www.transport.govt.nz/statistics-and-insights/fleet-statistics/sheet/monthly-ev-statistics (date of access 1 December 2020)

Table A.4 Number of registered EVs by year (percentage of EVs compared to the heavy and light vehicle fleet and total fleet in parentheses)

Year	Heavy EVs	Light pure EVs	Total fleet
2014	74 (0.05%)	1,050 (0.03%)	1,124 (0.03%)
2015	73 (0.05%)	2,055 (0.06%)	2,128 (0.06%)
2016	75 (0.05%)	5,051 (0.14%)	5,126 (0.13%)
2017	80 (0.08%)	12,368 (0.33%)	12,448 (0.3%)
2018	128 (0%)	23,402 (0.6%)	23,530 (0.55%)

Figure A.5 Number of registered light and heavy EVs by year



Future projections

According to the Ministry of Transport (2017), the key changes with respect to road transport over the coming decades are likely to be:

- increasing demand for road transport as a result of population growth, which is partially met by public transport in inner city areas and a shift in car ownership (as much as 60%) as ride- and vehicle-share schemes become more popular
- increasing use of EVs and adoption of new, more efficient vehicle technologies
- growth in road freight tonnage and new freight delivery technologies, including drones, robots and driverless vehicles.

Future fleet projections were made for five scenarios:

1. **Base Case** – this is a conservative business-as-usual scenario that assumes slow, non-disruptive technological changes and a continuation of current demographic and economic trends.
2. **Staying Close to the Action** – this scenario assumes the same demographic and economic trends as the Base Case. The key difference compared to the Base Case is that there is increased urban density in inner cities leading to greater use of public transport as well as walking and cycling. Demand management and road pricing in Auckland and Wellington has almost eliminated traffic congestion under this scenario.

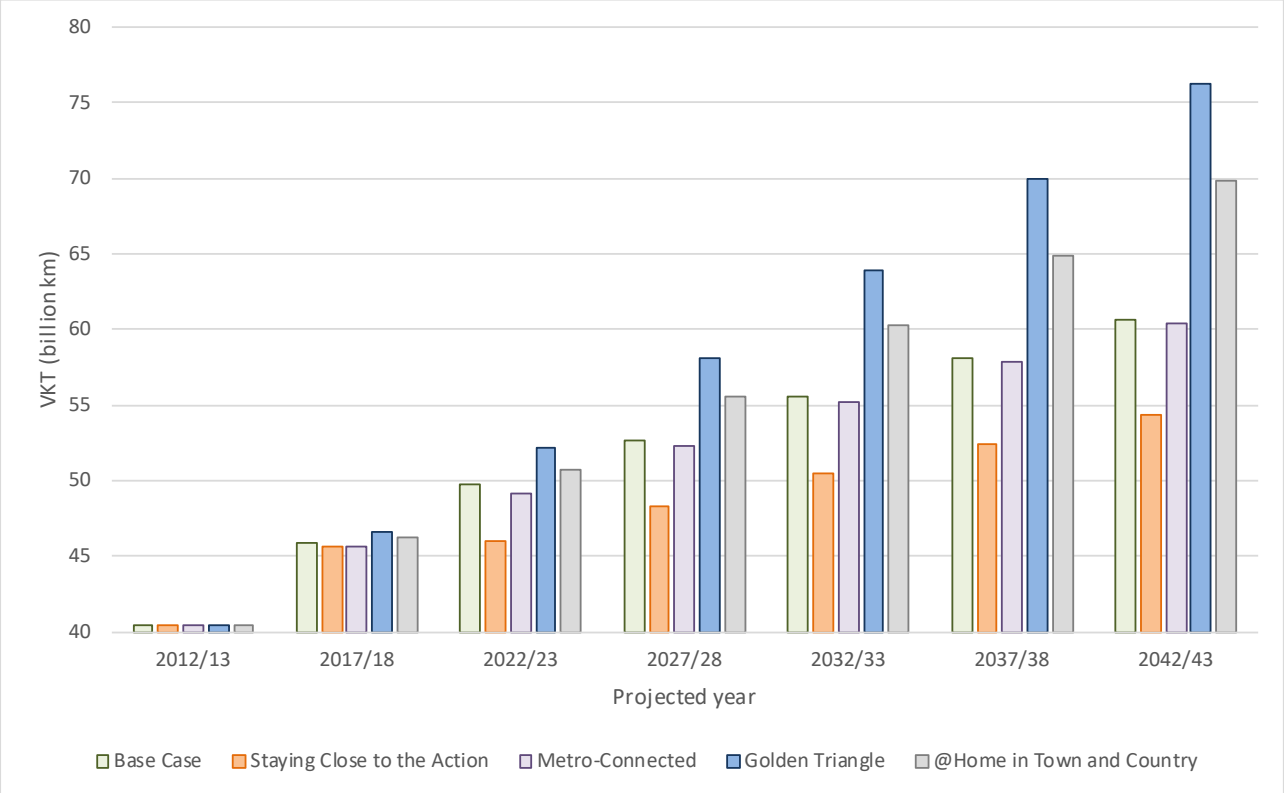
3. **Golden Triangle** – this scenario assumes more rapid technical development and economic growth and population growth, centred in the triangle of Auckland, Waikato and Bay of Plenty. The scenario assumes greater urban sprawl in the triangle resulting in longer commuting distances.
4. **Metro-Connected** – this scenario is similar to the Staying Close to the Action scenario; however, people prefer to connect and work digitally rather than face to face. This means that there is less inner-city intensification and people are able to live and work in regional centres other than Auckland and Wellington. The scenario assumes that there is less demand for urban transport as people tend to work from home.
5. **@Home in Town and Country** – this scenario has the same growth in technology and economy as the Golden Triangle scenario, but with a similar preference for digital connection as the Metro-Connected scenario. As a consequence, the population is more evenly spread across the country, including in smaller towns and rural areas.

Projected increases in VKT by vehicle type are given in Table A.5 to the year 2042/43 for the Base Case scenario. Under this scenario, the VKT is projected to increase by 51% for light vehicles and 41.6% for heavy vehicles over the projected 30-year period, and the total fleet VKT is projected to be 60.7 billion km. The projected change in VKT is compared for the five scenarios in Figure A.6; the @Home in Town and Country (69.8 billion km) and Golden Triangle (76.2 billion km) scenarios have the greatest projected increases in VKT.

Table A.5 Projected VKT (billions km) by vehicle type for the Base Case scenario (percentage increase compared to 2012 in parentheses)

Year	Light vehicles		Heavy vehicles		Motorcycles		Total fleet	
2012	37.1		2.9		0.4		40.4	
2017	42.1	(13.4%)	3.3	(14.7%)	0.4	(12.5%)	45.9	(13.5%)
2022	45.7	(23%)	3.6	(25.9%)	0.5	(20.2%)	49.8	(23.1%)
2027	48.5	(30.5%)	3.8	(31.4%)	0.5	(23.3%)	52.7	(30.5%)
2032	51.2	(37.8%)	3.9	(36.6%)	0.5	(25.1%)	55.6	(37.5%)
2037	53.6	(44.4%)	4.0	(39%)	0.5	(24.7%)	58.1	(43.8%)
2042	56.1	(51.1%)	4.1	(41.6%)	0.5	(23.7%)	60.7	(50.2%)

Figure A.6 Projected VKT (billion km) by road vehicles by scenario and year



Projections of the number of EVs up to 2039/40 are presented in Figure A.7 for the Base Case scenario. Under this scenario, EVs will make up around 40% of the national fleet by 2039/40 and will account for 48% of the total VKT. The projected increases in the percentage of EVs to the total fleet is shown for the different scenarios in Figure A.8. The increase in the percentage of EVs is projected to be greatest for the @Home in Town and Country (45%) and Golden Triangle (49%) scenarios, and the increase for the other scenarios is very similar to the Base Case scenario.

Figure A.7 Projected change in fleet composition for EVs compared to other fuels for the Base Case scenario

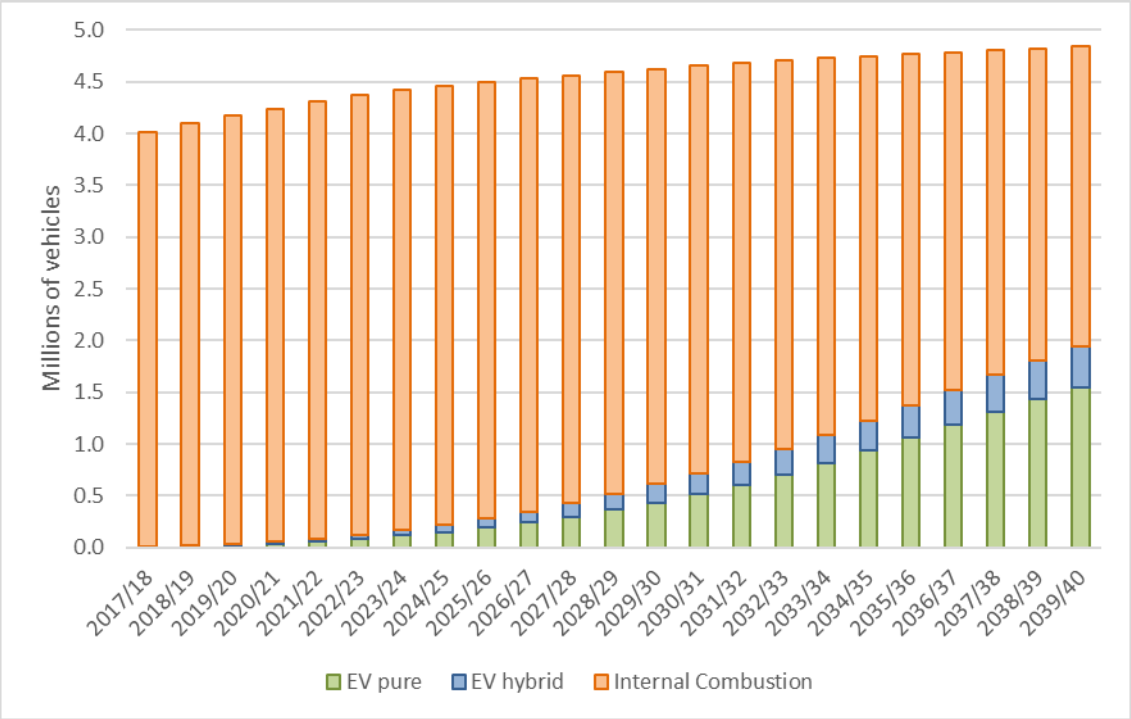
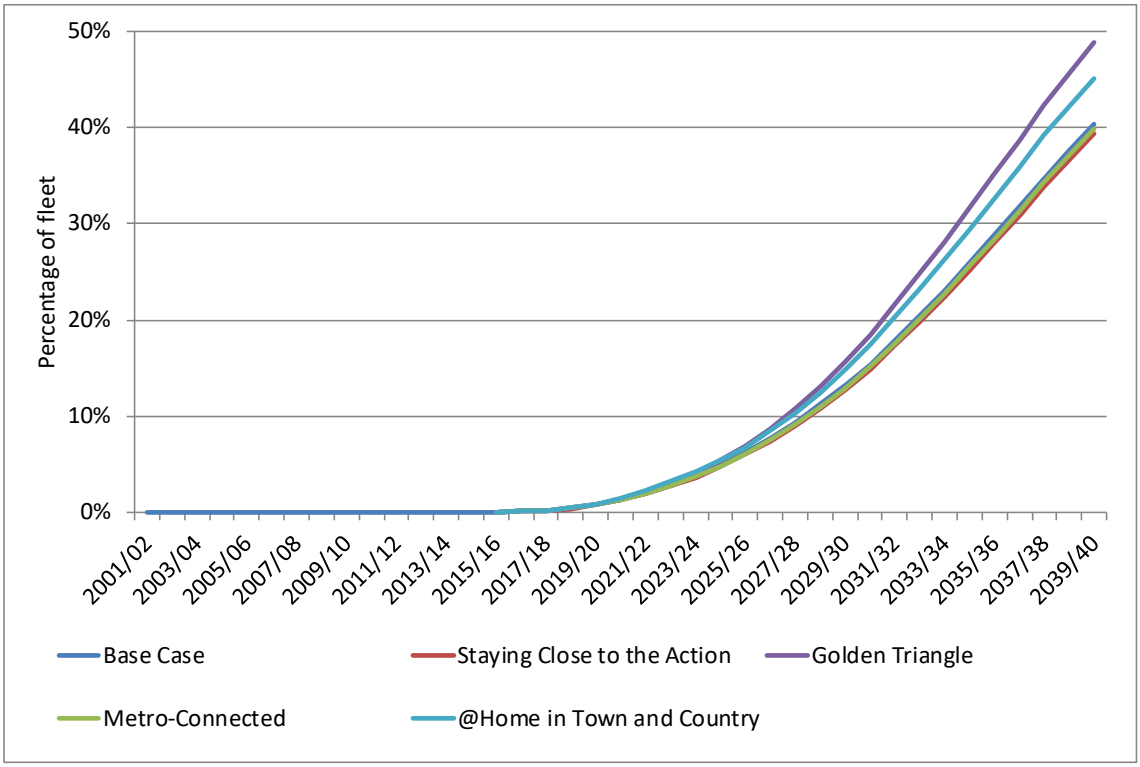


Figure A.8 Projected increases in the percentage of EVs in the national fleet by scenario



Appendix B: Road surface types in New Zealand

The following definitions and illustrations of road surface types in New Zealand have been copied from the Waka Kotahi guidance on road noise (NZ Transport Agency, 2014, pp. 12–13).

Single coat seal

A single coat seal is a sealing binder with a single application of chip. When a road is first made this single coat will be referred to as a 'first coat seal'.

FIGURE 1.9 Single coat seal

Typically roads are resealed every eight to 15 years, so additional single coat seals are often laid over the existing layers.

FIGURE 1.10 Reseal

Two-coat seal

A two-coat chipseal has two applications of binder and two applications of chip, the second smaller in size to the first. The smaller chip of the second coat locks and supports the larger chip of the first coat.

FIGURE 1.11 Two-coat seal

Racked-in seal

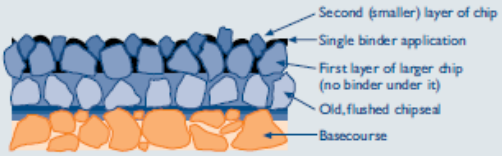
A racked-in chipseal consists of one application of binder and two applications of chip.

FIGURE 1.12 Racked-in seal

Sandwich seal

A sandwich seal is a variation of a two-coat seal and is used as a re-seal where there is excess 'flushed' bitumen on the original surface.

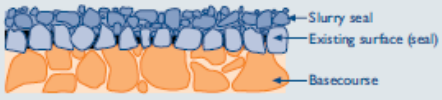
FIGURE 1.13 Sandwich seal



Slurry seal

A specifically designed mix of aggregates plus an emulsified binder.

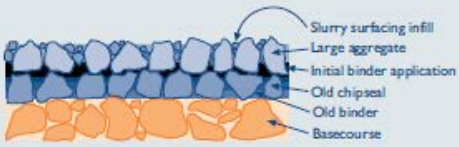
FIGURE 1.14 Slurry seal



Cape seal

A cape seal is a two-coat seal where the first coat is a chipseal and the second coat is a slurry seal, which fills the texture of the chipseal.

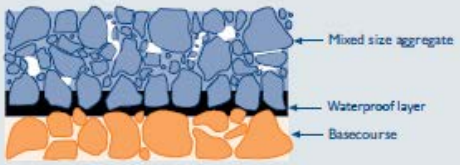
FIGURE 1.15 Cape seal



Asphaltic concrete (AC)

These mixes have a range of aggregate particle sizes and filler (evenly distributed from coarse to fine) and a low design air void content generally of 4%.

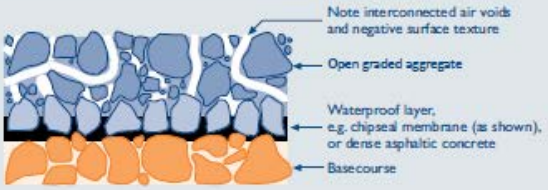
FIGURE 1.16 Asphaltic concrete



Open graded porous asphalt (OGPA)

OGPA has less fine aggregates than AC and more air voids, typically between 15% and 30%.

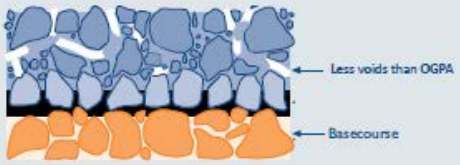
FIGURE 1.17 OGPA



Stone mastic asphalt (SMA)

SMA has a high-course aggregate content and has similarities with OGPA but with fewer air voids (typically 4%).

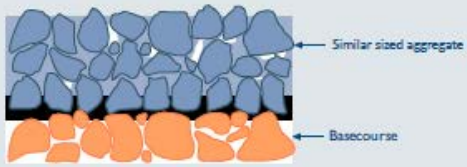
FIGURE 1.18 SMA



Macadam

A surface type currently unspecified in New Zealand. These surfaces are similar to ACs and SMAs.

FIGURE 1.19 Macadam



Appendix C: Knowledge hub entries

The Transport Knowledge Hub for Water – Research Stocktake of research carried out in New Zealand into the effects of transport infrastructure (roads and rail) on stormwater quality was compiled by NIWA for Waka Kotahi (Moores, 2020) and contains 69 entries, each representing a separate data source, report or paper. The entries provide a description of the resource, search keywords, the commissioning and research agencies, the key citation and, where provided, an internet link.

The 39 entries listed in Table C.1 are those that present characterisations of road dust and/or runoff water quality. The keywords searched were:

- bitumen (B)
- metals (M)
- PAHs
- road surface (RS)
- sediments (S)
- stormwater quality (WQ)
- road dust (D)
- vehicle emission factors (EF).

The entries do not include publications on the effectiveness of stormwater treatment.

Table C.1 Transport Knowledge Hub for Water – Research Stocktake entries that present characterisations of road dust and/or road runoff

Year	Reference	Description	Commissioning agency	Research agency	Link	Keywords
1998	Sherriff, J. (1998). *An investigation into water quality effects of transport.* Wellington Regional Council Publication No. WRC/RINV-T-98/25. 15pp.	Field study collecting samples of road runoff from the Wellington motorway (SH1), generating data on concentrations of suspended solids, faecal coliforms, metals and hydrocarbons.	Wellington Regional Council	Wellington Regional Council	–	M, WQ
2000	Gadd, J., & Kennedy, P. (2000). *Preliminary examination of organic compounds present in tyres, brake pads and road bitumen in New Zealand*. Wellington: Ministry of Transport. 17pp.	Analysis of organic compounds in samples of tyres, brake pads and road bitumen.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwater-organiccompounds2.pdf	EF
2000	Kennedy, P., & Gadd, J. (2000). *Preliminary examination of trace elements in tyres, brake pads and road bitumen in New Zealand of metal concentrations in samples of tyres, brake pads and road bitumen.* Wellington: Ministry of Transport. 19pp.	Analysis of metal concentrations in samples of tyres, brake pads and road bitumen.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwaterinorganiccompounds.pdf	M, EF
2002	Kennedy, P., Gadd, J., & Moncrieff, I. (2002). *Emission factors for contaminants released by motor vehicles in New Zealand*. Wellington: Ministry of Transport. 104pp.	Estimates of vehicle contaminant emissions based on a review of the vehicle fleet, brake pad composition, tyre composition, emissions of lubricants and greases, emissions of coolants, emissions from exhausts and wear of bitumen road surfaces.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwateremissionfactors.pdf	M, EF
2002	Moncrieff, I., & Kennedy, P. (2002). *Road transport impacts on aquatic ecosystems – Issues and context for policy development.* Wellington: Ministry of Transport. 88pp.	Development and application of framework for assessing effects of road-derived contaminants on aquatic ecosystems. Should be read in conjunction with the other Ministry of Transport reports from the 2000 to 2002 period.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwater-road-effects.pdf	WQ, EF

Year	Reference	Description	Commissioning agency	Research agency	Link	Keywords
2002	O'Riley, A., Pandey, S., Langdon, A., & Wilkins, A. (2002). *Characterisation of runoff contaminants from New Zealand roads, & effect of rainfall events*. Transfund New Zealand research report 228. 119pp.	Field study collecting samples of road runoff from a roundabout site in Hamilton, generating data on concentrations of metals and hydrocarbons.	Transfund NZ	Landcare Research and University of Waikato	https://www.nzta.govt.nz/resources/research/reports/228/	M, WQ
2003	Kennedy, P. (2003). *Metals in particulate matter material on road surfaces*. Wellington: Ministry of Transport. 99pp.	Analysis of metal concentrations in road dust samples from Wellington and Waitakere City, and review of other data sources.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwaterparticulatematerial.pdf	M, D, EF
2003	Kennedy, P. (2003). *The effects of road transport on freshwater and marine ecosystems.* Wellington: Ministry of Transport. 142pp.	Summary report bringing together series of Ministry of Transport reports produced 2000 to 2002. Covers: • sources of transport-derived contaminants • measured contaminant concentrations in road dust and stormwater • contaminant fate and effects in receiving freshwater and marine environments.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwaterroadtransporteffects.pdf	S, WQ, EF
2003	Kennedy, P., & Gadd, J. (2003). *Evaluation of road surface contaminant loadings in Waitakere City for the development of the Vehicle Fleet Emission Model – Water*. Wellington: Ministry of Transport. 34pp.	Analysis of contaminant concentrations in road dust samples collected in Waitakere City and development of vehicle EFs to support modelling of road-derived contaminant loads.	Ministry of Transport	Kingett Mitchell Ltd	https://rcaforum.org.nz/sites/public_files/documents/stormwatercontaminantloadingswaitakere-city.pdf	M, WQ, EF
2003	Ng, W., Buckeridge, J., & Ockleston, G. (2003). *Distribution of heavy metals in road sediments.* Third South Pacific Stormwater Conference, Auckland, May 2003.	Collection and analysis of road dust samples for concentrations of metals by particle size fractions. Concluded that removal of road sediments > 100 µm would reduce metal concentrations below Australian and New Zealand Environment and Conservation Council guideline levels.	Auckland City Council	AUT	–	M, D

Year	Reference	Description	Commissioning agency	Research agency	Link	Keywords
2003	Timperley, M., Bailey, G., Pattinson, P., & Kuschel, G. (2003). *Zinc, copper and lead in road run-off*. 26th Australasian Transport Research Forum, Wellington, 1–3 October 2003.	Presents results of road runoff sampling at Ash St and Richardson Rd, Auckland (there was also a second period of sampling at the latter location – refer Moores et al., 2009). The sampling results were used to model loads of metals in road runoff as a proportion of the total loads of metals in Auckland stormwater.	Foundation of Research, Science and Technology	NIWA	–	M, WQ
2005	Brown, J. N., & Peake, B.M. (2005). Sources of heavy metals and PAHs in urban stormwater runoff. *Science of the Total Environment, 359* (2006), 145–155.	As part of a study of urban stormwater quality in Dunedin, road dust samples were analysed for PAHs and heavy metals.	N/A	University of Otago	https://www.sciencedirect.com/science/article/pii/S0048969705003876	M, D, WQ
2005	Timperley, M., Williamson, B., Mills, G., Horne, B., & Hasan, M. (2005). *Source and loads of metals in urban stormwater.* NIWA Client Report: AKL2004-070 prepared for Auckland Regional Council.	Estimates of loads of copper, zinc and lead in Auckland stormwater from roads and other sources. Based on extensive sampling of road, commercial, industrial and residential stormwater.	Auckland Regional Council	NIWA	http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.623.6620&rep=rep1&type=pdf	M, WQ, EF
2005	Zanders, J. M. (2005). Road sediment: Characterization and implications for the performance of vegetated strips for treating road run-off. *Science of the Total Environment*, *339*, 41–47.	Sampling and analysis of road dust in Hamilton to characterise particle size distribution and metal concentrations. Discussion of implications for treatment of road runoff by vegetated swales.	Foundation of Research, Science and Technology	Landcare Research	https://www.sciencedirect.com/science/article/pii/S0048969704005704	M, D, S
2006	Ahrens, M., & Depree, C. (2006). *Legacy PAH contamination of aquatic sediments by roading tar in Auckland.* NZWWA Stormwater Conference, Rotorua, 4–5 May 2006.	Field study to investigate sources of high PAHs in Auckland stream and estuarine sediments. Results indicate coal-tar derived materials used in old road seal layers are the major source of elevated PAH levels found in streams.	Foundation of Research, Science and Technology	NIWA	–	PAH

Year	Reference	Description	Commissioning agency	Research agency	Link	Keywords
2006	Depree, C., Ahrens, M., & McNeill, S. (2006). *Legacy contaminants in Christchurch: Are roadside soils and ongoing major source of PAHs in runoff today?* NZWWA Stormwater Conference, Rotorua, 4–5 May 2006.	Field study to investigate sources of high PAHs in Christchurch stormwater runoff. Results indicate coal-tar derived materials used in old road seal layers have accumulated in roadside berms and soils and that these act as a reservoir of PAHs.	Christchurch City Council	NIWA	–	PAH
2006	Herrington, P. R., Ball, G.F.A., & O'Halloran, K. (2006). *Aquatic ecotoxicity of cutback bitumen*. Land Transport New Zealand Research Report 285. 52pp.	Analysis of bitumen samples for ecotoxicity to New Zealand aquatic fauna. Results suggest classification of bitumen is too stringent.	Land Transport New Zealand	Opus Central Laboratories; Landcare Research	www.nzta.govt.nz/resources/research/reports/285	B, RS
2006	Herrington, P., Kvatch, I., & O'Halloran, K. (2006). *Assessing the environmental effects of new and recycled materials in road construction: Proposed guidelines*. Transfund New Zealand Research Report No 306. 70pp.	Development of guidelines for assessing whether new materials for road construction contain leachable hazardous substances.	Land Transport New Zealand	Opus Central Laboratories; Landcare Research	https://www.nzta.govt.nz/assets/resources/research/reports/306/docs/306.pdf	RS
2007	Fassman, E., & Voyde, E. (2007). *Sediment retention efficiencies of in-use catchpits.* 2007 South Pacific Stormwater Conference, Auckland.	Lab testing at University of Auckland to investigate the influence of catchpit sediment accumulation and inflow rates on sediment capture.	Auckland Regional Council	University of Auckland	–	S
2007	Gardiner, L. R., & Armstrong, W. (2007). *Identifying receiving environments at risk from road runoff*. Land Transport NZ research report 315. 68pp.	Development of GIS screening tool for identifying risk to depositional coastal receiving environments from stormwater runoff based on road VKT. Concept developed further under NZ Transport Agency research report 493.	Land Transport New Zealand	MWH New Zealand Ltd	https://www.nzta.govt.nz/assets/resources/research/reports/315/docs/315.pdf	WQ
2007	Moores, J., Hunt, J., & Pattinson, P. (2009b). *Quantification of catchpit sediments and contaminants. Data collection.* Prepared by NIWA Ltd for Auckland Regional Council. Auckland Regional Council Technical Report 2009/123.	Analysis of sediment quality in samples from catchpits collected around Auckland.	Auckland Regional Council	NIWA	–	M, S

Year	Reference	Description	Commissioning agency	Research agency	Link	Keywords
2008	Ball, G. F. A., Herrington, P. R., & Patrick, J. E. (2008). *Environmental effects of emulsions*. Land Transport New Zealand Research Report 343. 46pp.	Results of ecotoxicological testing of bitumen emulsions, indicating classification as 'safe' or 'slightly harmful' to the aquatic environment.	Land Transport New Zealand	Opus Central Laboratories	https://www.nzta.govt.nz/assets/resources/research/reports/343/docs/343.pdf	B, RS
2008	Depree, C. (2008). *Contaminant characterisation and toxicity of road sweepings and catchpit sediments: Towards more sustainable reuse options.* Land Transport NZ research report 345. 114pp.	Analysis of ecotoxicants in catchpit and street sweeping samples in Auckland, Hamilton and Christchurch, for assessment of potential re-use as an alternative to landfill disposal.	Land Transport New Zealand	NIWA	https://www.nzta.govt.nz/assets/resources/research/reports/345/docs/345.pdf	M, S, D
2008	Kennedy, P., & Sutherland, S. (2008). *Urban sources of copper, lead and zinc*. Auckland Regional Council Technical Report 2008/023.	Estimates of loads of copper, zinc and lead in Auckland stormwater from roads and other sources. Updates Timperley et al. (2005).	Auckland Regional Council	Golder Associates	http://www.aucklandcity.govt.nz/council/documents/technicalpublications/TR2008_023%20-%20Urban%20sources%20of%20copper,%20lead%20and%20zinc.pdf	M, WQ
2008	Moores, J., Pattinson, P., Reed, J., McHugh, M., & Cavanagh, J. (2008). *Mitigation strategies for controlling the dispersion of particulate metals emitted from vehicles.* NIWA report AKL-2008-048 prepared under FRST contract C01X0405. 78pp.	Estimates of performance of stormwater treatment (ponds, swales and roadside channels) for removal of particulate metals based on sampling at locations north of Auckland. Reviews dispersion of airborne emissions of particulate metals form roadside.	Foundation of Research, Science and Technology	NIWA and Landcare Research	–	WQ, EF
2008	Moores, J., Hunt, J., & Pattinson, P. (2009b). *Quantification of catchpit sediments and contaminants. Data collection*. Prepared by NIWA Ltd for Auckland Regional Council. Auckland Regional Council Technical Report 2009/123.	Analysis of sediment quality in samples from catchpits collected around Auckland.	Auckland Regional Council	NIWA	–	M

Year	Reference	Description	Commissioning agency	Research agency	Link	Keywords
2008	Reed, J., Depree, C., Trip, E., Chappell, P., & Talbot, N. (2008). *Sampling receiving environments close to state highways*. NIWA Client Report: AKL2008-035 prepared for NZ Transport Agency.	Collection and analysis of sediment samples from five receiving environments downstream of state highways in Auckland. Assessment of contribution of roads to elevated sediment metal and PAH contributions.	NZ Transport Agency	NIWA	https://rcaforum.org.nz/sites/public_files/documents/SRE-2008.pdf	M, S
2008	Skeen, M., & Timperley, M. (2008). *Waitakere City vehicle testing station raingarden – monitoring runoff*. NZWWA Stormwater Conference, Rotorua, 15–16 May 2008.	Field study of performance of raingarden to treat stormwater runoff from vehicle testing station, Waitakere City. Results indicated high removal rates of TSS, copper and zinc.	Auckland Regional Council	Auckland Regional Council and NIWA	–	M, S
2009	Fassman, E., & Blackbourn, S. (2008). *Long-term permeable pavement field monitoring*. NZWWA Stormwater Conference 2008.	Results of a study to assess hydrological and water quality of permeable pavement, North Shore, Auckland. Pavement was found to perform well both as a hydrological and stormwater treatment device.	Auckland Regional Council and North Shore City Council	University of Auckland	–	WQ
2009	Fassman, E., Liao, M., Hellberg, C., & Easton, H. (2009). *Monitoring of the treatment train at the Albany park n ride*. NZWWA Stormwater Conference 2009.	Field study to monitor performance of treatment train for removing TSS and metals from stormwater from park n ride. Treatment train includes swales, rain garden, a storm filter and a constructed wetland. The results indicated the system performed well, other than for removal of TSS.	Auckland Regional Council	University of Auckland	–	M, S
2009	Moores, J., Pattinson, P., & Hyde, C. (2009a). *Richardson Road study measurement and sampling of runoff and catchpit solids*. Prepared by NIWA for Auckland Regional Council. Auckland Regional Council Document Technical Report 2009/119.	Presents results of road runoff sampling and catchpit solids sampling (including measurement of solids loading) from Richardson Rd, Auckland. This was the second period of sampling at this location – refer Timperley et al. (2003) for the earlier results.	Auckland Regional Council	NIWA	–	M, WQ

Year	Reference	Description	Commissioning agency	Research agency	Link	Keywords
2010	Moores, J., Pattinson, P., & Hyde, C. (2010). *Enhancing the control of contaminants from New Zealand's roads: Results of a road runoff sampling programme.* New Zealand Transport Agency research report 395. 161pp.	Stormwater sampling study on state highways in Auckland region generating estimates of vehicle EFs for loads of copper and zinc discharged in road runoff, and the performance of stormwater treatment devices (ponds and swales) for removing these metals and TSS.	NZ Transport Agency	NIWA	https://www.nzta.govt.nz/assets/resources/research/reports/395/docs/395.pdf	M, WQ, EF
2012	Good, J., O'Sullivan, A., Wicke, D., & Cochrane, T. (2012). Contaminant removal and hydraulic conductivity of laboratory rain garden systems for stormwater treatment. *Water Science and Technology, 65*(12): 2154–2161.	Lab testing of bioretention media for hydraulic performance and removal of metals from stormwater.	Christchurch City Council and Environment Canterbury	University of Canterbury	https://iwaponline.com/wst/article/65/12/2154/17150/Contaminant-removal-and-hydraulic-conductivity-of	M, WQ
2012	Moores, J., Gadd, J., Pattinson, P., Hyde, C., & Miselis, P. (2012). *Field evaluation of media filtration stormwater treatment devices*. NZ Transport Agency research report 493. 255pp.	Field study to assess the performance of three commercially available media filtration stormwater treatment devices for removing suspended solids, copper and zinc from road runoff. Field conditions were found to have a marked influence on device performance.	NZ Transport Agency	NIWA and AECOM	https://www.nzta.govt.nz/assets/resources/research/reports/493/docs/493.pdf	M, S, WQ
2013	Moores, J., Pattinson, P., & Hyde, C. (2013). Variations in highway stormwater runoff quality and stormwater treatment performance in relation to the age of porous friction courses. *Water Environment Research*, *85*(9): 772–781.	Field study to assess impact of road surface type and age on road runoff quality. Runoff from 1-year-old open-graded porous asphalt had lower particulate contaminant concentrations than other road surfaces. Stormwater treatment (pond and swale) was effective on runoff discharged from all road surface types monitored.	NZ Transport Agency	NIWA	https://onlinelibrary.wiley.com/doi/epdf/10.2175/106143012X13461650921176	M, RS, S

Year	Reference	Description	Commissioning agency	Research agency	Link	Keywords
2016	Charters, F., Cochrane, T., & O'Sullivan, A. (2016). Untreated runoff quality from roof and road surfaces in a low intensity rainfall climate. *Science of the Total Environment*, *550*, 265–272.	Field study of stormwater quality in Christchurch investigating variations in suspended sediment and metal concentrations across four different types of impermeable surfaces (including roads).	University of Canterbury	University of Canterbury	https://www.sciencedirect.com/science/article/pii/S0048969716300754?via%3Dihub	M, S, WQ
2016	Poudyal, S., Cochrane, T., & Bello-Mendoza, R. (2016). *First flush stormwater pollutants from carparks in different urban settings*. Water New Zealand November/December 2016.	Collection of first flush stormwater samples from three urban car parks in Christchurch and analysis of samples for TSS and metal concentrations.	University of Canterbury	University of Canterbury	https://www.waternz.org.nz/Attachment?Action=Download&Attachment_id=1847	M, S, WQ
2017	Sampson, C. (2017). *Trace elements in Christchurch road dust* [Master's thesis, University of Canterbury].	Collection of road dust from 30 sites. Analysed for PSD and As, Cd, Co, Cr, Cu, Fe, Mn, Ni, Pb, Sb, V and Zn. Found no relationship with traffic, age or weather for most metals, but relationship for Zn with land use.	University of Canterbury	University of Canterbury	https://ir.canterbury.ac.nz/bitstream/handle/10092/13803/Sampson%2C%20Christopher_Master%27s%20Thesis.pdf?sequence=1	M, D
2020	Charters, F., Cochrane, T., & O'Sullivan, A. (2020). Predicting event-based sediment and heavy metal loads in untreated urban runoff from impermeable surfaces. *Water*, *12*(4), 969.	Description of the MEDUSA model developed to evaluate loads of stormwater solids and metals for roads and other impermeable surfaces, Christchurch.	University of Canterbury	University of Canterbury	https://www.mdpi.com/2073-4441/12/4/969/htm	M, WQ

Source: Adapted from Moores (2020)