Masterton air quality insights

Greater Wellington
Te Pane Matua Taiao

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Technical summary 2004 to 2024

Tamsin Mitchell

For more information, contact the Greater Wellington Regional Council:

Wellington
PO Box 11646

T 04 384 5708
F 04 385 6960
www.gw.govt.nz

Masterton
PO Box 41

T 06 378 2484
F 06 378 2146
www.gw.govt.nz

Upper Hutt
PO Box 40847

T 04 526 4133
F 04 526 4171
www.gw.govt.nz

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www.gw.govt.nz
info@gw.govt.nz

Report prepared by:	T Mitchell	Senior Air Quality Scientist, Knowledge, Environment Group	T. Mitchell
Report reviewed by:	Dr M Oliver	Principal Advisor Knowledge, Environment Group	Megan Oliver.
Report approved for release by:	Dr E Harrison	Manager, Knowledge, Environment Group	E Harrison **Date: 28/04/25**

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Executive summary

This report brings together 20 years of air quality monitoring, science investigations and other sources of information about air quality and pollutant emissions in Masterton (Whakaoriori).

Key insights:

- Air quality is degraded by emissions of fine particulates (PM2.5 and PM10) from household wood burners and nitrogen dioxide (NO_2) from motor vehicles. Masterton's weather and topography mean air pollution at times is less well dispersed than in other more exposed urban areas within the region.

- The estimated health impacts and costs of air pollution are not distributed equally across the region. Of all the region's airsheds, Masterton experiences the highest social costs per person which is correlated with a relatively high population-weighted deprivation index which increases vulnerability to air pollution health impacts. About 40% of the health impacts are estimated to be due to NO_2 from motor vehicles, and the remainder from PM2.5.

- Masterton is close to meeting the daily PM10 national environmental standard based on current monitoring. Since this standard was introduced in 2004, there is new evidence to show adverse health impacts at levels much lower than currently allowed. The Ministry for the Environment is considering the new health evidence and achievability of the 2021 World Health Organization guidelines with a view to regulating PM2.5.

- Wintertime PM10 and PM2.5 and annual NO_2 levels have reduced over their respective monitoring periods. The downward trend was due to a combination of changing weather conditions leading to better dispersion and most likely reducing emissions from home heating and motor vehicles.

- Most Masterton households use wood burners for home heating, but the proportion is trending down as households switch to heat pumps, reducing PM2.5 emissions in the airshed.

- Over 80% of the PM2.5 measured in air comes from home heating. Successful strategies for reducing PM2.5 from home heating are incentives and regulation to remove wood burners or switch to appliances that produce less smoke (pellet or ultra-low emission burners). Education and behaviour change to encourage use of dry wood and lighting fires using 'best practice methods' are unlikely to be effective on their own. Finances are a key factor in people's choice of home heating method.

- Aiming for long term continual improvement in Masterton's air quality will reduce the health burden and align with CO_2 emission reductions for climate change mitigation.

Contents

1. Introduction

Clean air is one of the fundamental resources for healthy living. Greater Wellington is responsible for monitoring and managing air quality to protect the health of our communities and environment. This report focuses on Masterton (Whakaoriori), situated on the upper part of the river plain of the Wairarapa valley. Masterton is the largest town in the Wairarapa with a population of approximately 23,200 (June 2024)[1].

Masterton experiences poor winter air quality that fails to meet National Environmental Standards. This is due to high dependence on wood burners for home heating combined with cold temperatures and low wind speeds which lead to a build-up of wood smoke containing fine particles which are harmful to health.

Purpose and scope

This technical report summarises Masterton air quality monitoring and science-led investigations from 2004 to 2024. It includes work carried out by Greater Wellington and other agencies. The purpose is to present Greater Wellington's current science knowledge base to inform air quality management decision making and identify knowledge gaps.

Report outline

- Section 2 outlines central, regional and local government responsibilities for managing and monitoring air quality and emissions.
- Section 3 provides an overview of air quality indicators, guidelines and standards.
- Section 4 outlines Greater Wellington's monitoring programme and monitoring sites.
- Section 5 describes the emissions pressures, weather and topography that combine to determine air quality.
- Section 6 summarises air quality monitoring data trends.
- Section 7 summarises monitoring studies characterising how air quality impacts from wood burner emissions vary across the Masterton urban area.
- Section 8 reports population level health impacts and their social costs.

[1] https://explore.data.stats.govt.nz/

- Section 9 focuses on social factors, such as what is known about people's burning behaviour and reasons for using wood for home heating.
- Section 10 covers healthy homes, energy efficiency and indoor air quality.

Terms and abbreviations used in this report

Term	**Definition**
Black carbon	Fine soot particles produced through the incomplete combustion of fuels containing carbon, eg fossil fuels and biomass (wood material).
CCA	Copper-Chrome-Arsenate a chemical formulation used to treat timber so it resists decay and can be used outside, eg decking and fences.
EECA	Energy Efficiency & Conservation Authority
GHG	Greenhouse gas
GNS	GNS Science (a crown research institute)
HAPINZ	Health and Air Pollution in New Zealand study published in 2022
MfE	Ministry for the Environment
Multi-fuel burner	An enclosed burner which burns wood as well as coal, this includes incinerators, pot belly stoces, McKay space heaters etc, but does not include open fires.
NEMS	National Environmental Monitoring Standards
NESAQ	Resource Management (National Environmental Standards for Air Quality) Regulations 2024
NIWA	National Institute of Water and Atmosphere (a crown research institute)
NO_2	Nitrogen dioxide
NOx	Oxides of nitrogen
NRP	Natural Resources Plan for the Wellington Region 2023
Open fire	This is enclosed on three sides but open on the front. Includes a visor fireplace.
Pellet fires	Using processed wood pellets.
PM	Particulate matter
PM10	Particles with a diameter of 10 micrometres or less
PM2.5	Particles with a diameter of 2.5 micrometres or less

ULEB	Ultra-low emission burner
WHO	World Health Organization
Wood burner	This is a fully enclosed burner but does not include multi fuel burner like those that burn coal, or a pellet fire.

2. Regulation and policies for air quality and emissions

Burning of fossil fuels and wood produce air pollutants which harm health and greenhouse gases which affect the climate.

Air quality in our region is affected by:

- Particulate matter (PM) from wood burnt in domestic fires;
- Discharges to air from industrial and trade premises; and
- Nitrogen oxides (NOx and NO_2) from motor vehicle exhaust.

Particle air pollution also affects global warming, with some aerosols having a cooling effect (by reflecting sunlight) and others such as black carbon having a short-term heating effect (by absorbing sunlight). The impacts of aerosols (solid or liquid particles suspended in air) on climate warming are the largest source of uncertainty in climate models (Figure 7.6 in IPCC, 2021).

Although air pollutants and GHG emissions often come from the same source, particularly combustion of fossil fuels and wood burning, they are largely managed separately.

2.1 Air quality and air pollutant emissions

Central Government

- Air quality is primarily managed under the Resource Management Act 1991 (RMA) which provides the legislation and planning framework for regional and local government to control discharges of contaminants to air and manage air quality.
- The National Environmental Standards for Air Quality (NESAQ, 2004) are regulations made under the RMA that set nationally consistent minimum limits for some air pollutants, including a daily limit for PM10. Section 44A of the RMA requires local and consent authorities to enforce observance of national environmental standards to the extent to which their powers enable them to do so.
- The NESAQ set emissions and thermal efficiency standards for wood burners newly installed in properties less than two hectares in size.
- Ministry for the Environment (MfE) publicly consulted on updating the NESAQ in 2020 (MfE, 2020).
- The national Ambient Air Quality Guidelines (AAQG, 2002) set non-mandatory guidelines for a wide range of air pollutants.
- Motor vehicle exhaust emissions standards are regulated by Land Transport Rules made under the Land Transport Act 1988. Air quality

impacts from these emissions are not managed but can be influenced by Regional Land Transport Plans made under the Land Transport Management Act 2003.

Regional and local government

The primary responsibility for managing air quality under the RMA lies with regional councils (including unitary authorities). Regional councils have responsibilities for the control of discharges of contaminants to air.

The NESAQ require regional councils to ensure the air quality standards are met in their regions. Councils must identify and monitor areas where air quality is likely, or known, to exceed the NESAQ, using prescribed monitoring methods.

The Natural Resources Plan (NRP) for the Greater Wellington region (Greater Wellington, 2023) covers resource management for the coast, land, water and air. It sets environmental management objectives, policies to implement the objectives, and rules and methods to implement the polices.

For air quality the NRP includes:

- an objective to maintain or improve air quality to the 'acceptable' category (NRP Schedule L1).
- rules for activities that discharge contaminants to air, including whether a resource consent is required. Discharges from domestic fires and outdoor burning are permitted, but burning of 'specified materials' is prohibited. Specified materials include, treated and painted timber, plastics, rubber and other waste products.
- a method requiring airshed action plans to be developed and implemented for polluted airsheds to meet the NESAQ PM10 standard.
- A ban on installing new open fires in polluted airsheds as required by the NESAQ.

Territorial authorities (city and district councils) do not have a specific air quality management function under the RMA but have the primary responsibility for land use (for example, the location of activities that may discharge contaminants to air). However, territorial authorities can make bylaws under the Local Government Act 2002.

Territorial authorities also issue consents under the Building Act 1991 for domestic wood burners.

2.2 Greenhouse gas emissions

Both greenhouse gases (CO_2) and harmful pollutants are emitted during combustion of fossil fuels and wood burning. Therefore, actions to reduce CO_2 emissions through decarbonisation will also improve air quality.

Central government

New Zealand has emissions reduction targets for greenhouse gases to be met by 2050 as per the Climate Change Response (Zero Carbon) Amendment Act 2019. Successive emission budgets are published that include emission reductions by source sector. The Climate Change Commission's review of the fourth emission budget (2036-2040) quantified the health benefits of reductions in the transport sector, ie, improved air quality from a faster transition to electric vehicles and reducing private vehicle use as around $2.1 billion per year (Climate Change Commission, 2024).

NIWA advised the Ministry for the Environment that air quality standards and management should be better integrated with climate change policy (Coulson et al., 2021). The rationale is that CO_2 emission targets for 2050 are effectively a 'zero air pollution' goal, particularly for transport emissions from fossil-fuel combustion.

Regional and local government

- Regional and District Councils produce their own policies and plans for reducing GHG emissions and becoming more resilient to climate change impacts.

- Greater Wellington's mid-term review of the 2021 Regional Land Transport Plan (Greater Wellington 2024a) sets targets for reducing transport-generated carbon emissions and increasing active travel and public transport mode share. The Wellington Regional Land Transport Plan - Annual monitoring report (Greater Wellington 2024b) includes an environmental sustainability indicator that tracks trends in roadside nitrogen dioxide.

- Wellington regional transport emissions reduction pathway[2] (2024) sets out pathway of interventions to reduce transport emissions, such as reducing vehicle km travelled and increasing zero-emission vehicles. The plan notes that there are co-benefits for air quality if there are fewer fossil-fuel vehicles on our roads.

- There are no regional policies for GHG emissions from domestic wood burning. Wood burning is often viewed as a carbon neutral or low-carbon fuel because growing trees re-absorb the CO_2 produced by wood combustion. This assumption depends on how the wood is sourced and where and when replacement trees are grown (van Vugt&

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[2] https://www.gw.govt.nz/assets/Documents/2024/06/WTERP-2024.pdf

Webley 2023). Wood burning also produces other compounds which contribute to short-term global warming, for example, ultra fine soot particles commonly known as black carbon.

- The Masterton District Climate Action Plan (2022)[3] includes goals and actions aimed at reducing transport emissions and improving residential energy efficiency and renewable energy generation. Reducing transport emissions benefits air quality and improved home energy efficiency could improve air quality if it reduces demand for wood burning.

[3] https://www.mstn.govt.nz/community/environment/climate-change

3. Air quality indicators, guidelines and standards

3.1 Air quality indicators

Particulate matter (PM)

Particulate matter refers to tiny airborne particles which can be solid or liquid. These particles are produced by human activities such as wood burning, fuel combustion (particularly diesel), road dust and quarries. Natural sources include fine sea salt and wind-blown soil. The following air quality particulate matter indicators are monitored:

- PM10 particles smaller than 10 micrometres (μm) in diameter
- PM2.5 particles smaller than 2.5 micrometres (μm) in diameter

The size of the particle is important for health impacts as fine particles (PM2.5 and below) can penetrate deep into the lungs. The larger-sized coarse particles (above 2.5 μm and below 10 μm) can be breathed in and affect the upper airways. Figure 3.1 shows the size classification of particles. Health impacts of PM are commonly estimated based on PM2.5 levels in air where people live (Kuschel et al., 2022, Hales et al., 2021).

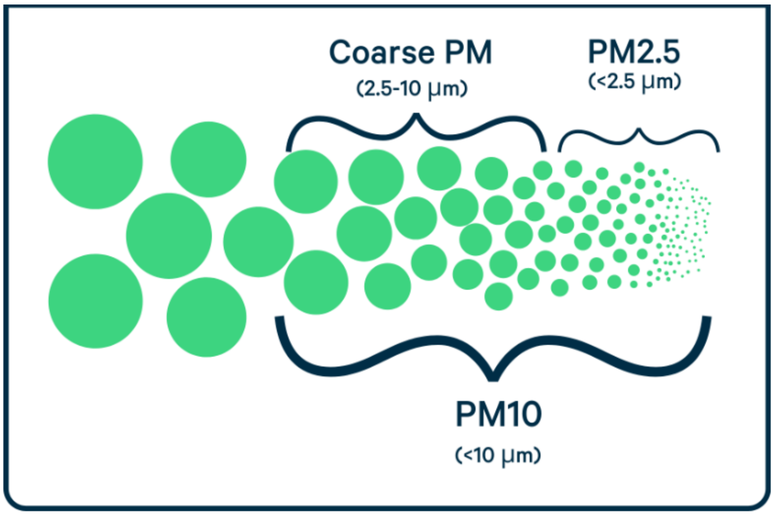

Figure 3.1: PM size fractions based on their diameter in micrometres (μm).
Source: https://learn.kaiterra.com/en/air-academy/particulate-matter-pm

Smoke from wood burning contains a mixture of fine particles, water vapour, gases and other harmful chemicals. In areas where solid fuels are widely used for home heating, winter concentrations of PM2.5 is the primary indicator of wood smoke emissions. In airsheds where home heating dominates emissions, most of the monitored PM10 will be in the PM2.5 size fraction.

Motor vehicle exhaust also contains fine particles, but due to advances in emissions filter technologies, tail pipe PM2.5 emissions per petrol vehicle are now very low compared to wood burners. Diesel vehicles emit more PM than petrol vehicles, although PM emissions from modern diesel vehicles fitted with diesel particulate filters have reduced.

Non-exhaust particle emissions arising from motor vehicle brake, tyre and road wear are becoming a more important source as tail pipe emissions decrease (Davy & Trompetter, 2021).

The relative source contribution of motor vehicles, home heating and other sources to levels of airborne PM2.5 and PM10 varies across New Zealand by monitoring location.

Black carbon (BC)

Black carbon also known as soot, is a particle in the $PM_{2.5}$ or ultrafine size range (smaller than 0.1 μm) produced during combustion. In New Zealand most black carbon is emitted from vehicles (especially diesels), burning wood and coal for home heating, and outdoor burning. Both long- and short-term exposure to BC is linked to serious health effects, such as cardiovascular disease and premature death (Janssen et al., 2012). BC is also a short-term climate warming pollutant as it absorbs heat from the sun (Patel et al., 2024).

Since 2017 we have progressively installed black carbon monitoring at three of our permanent monitoring stations, including Masterton West in 2023. We are evaluating how we can best use this measurement for tracking trends in combustion particles from wood burning and traffic sources.

Nitrogen dioxide (NO_2)

Nitrogen dioxide is a harmful air pollutant gas formed from fuel combustion primarily through the reaction of nitrogen and oxygen in air at high temperatures within combustion engines. This reaction produces oxides of nitrogen (NOx) which is a mix of nitric oxide (NO) and nitrogen dioxide (NO_2). Diesel powered vehicles emit substantially greater amount of NOx per mass of fuel burnt due to high temperatures needed for diesel combustion (Gentner & Xiong, 2017).

Nitrogen dioxide is a strong marker for motor vehicle emissions and the health impacts of this pollutant are estimated based on NO_2 levels in air where people reside (Kuschel et al., 2022, Hales et al., 2021).

3.2 Guidelines and standards for air quality indicators

There are a range of standards and guidelines for assessing air quality covering different pollutants, averaging times and monitoring methods. Standards and guidelines for air quality are periodically reviewed and revised as the evidence base for health impacts evolves. In general, air quality guidelines based on the annual average are more protective of health than the short-term daily guidelines (ESR, 2022). Table 3.1 compares the relevant standards and guidelines for air quality for PM and NO_2.

National Environmental Standard (NESAQ)

National Environmental Standards (NESAQ) set mandatory limits for daily and hourly concentrations of pollutants in air to ensure a consistent level of health protection across the country. The NESAQ standard for PM10 set in 2004 is for no more than one day per year to be above 50 μg/m^3.

National guidelines

Ambient Air Quality Guidelines (AAQG) were set in 2002 (Ministry for the Environment, 2002). These guideline values were, at the time, minimum requirements that all outdoor air quality should meet to protect people and ecosystems from significant adverse effects. The guidelines promote both ecosystem and human health, whereas the NESAQ are based on human health only.

Regional targets

Greater Wellington's Natural Resource Plan (Schedule L)[4] sets regional targets for PM10, PM2.5 and NO_2 based on the 'acceptable' air quality category. The acceptable category is arbitrarily defined as between 33.1% and 66% of a standard or guideline threshold value by Ministry for the Environment (2009). These regional targets were publicly notified in 2015 and became operative in 2023.

International guidelines

Since 1987, the World Health Organization (WHO) has periodically issued global health-based air quality guidelines to assist countries to reduce human exposure to the adverse impacts of air pollution. The 2005 WHO guidelines (published in 2006) were updated in 2021 reflecting the latest health evidence linking air pollution to many aspects of health at lower concentrations than previously understood (WHO, 2021).

WHO acknowledge that achieving the 2021 guidelines might be difficult for many countries and therefore gradual improvement in air quality to meet interim targets (four levels) is a critical indicator of improving population health. Interim targets guide reduction efforts towards ultimately meeting the guideline level.

Review of NESAQ to include PM2.5

The Parliamentary Commissioner for the Environment's air domain review found there was a strong case for monitoring and regulating PM2.5 with a policy goal for progressive improvement (PCE, 2015).

In 2020, MfE publicly consulted on proposed amendments to the NESAQ to introduce a daily and annual standard for PM2.5 (MfE, 2020) based on the 2005 WHO air quality guidelines (WHO, 2006). Subsequently, the 2021

[4] https://www.gw.govt.nz/assets/annual-monitoring-reports/air-quality/air-quality/components/resources/NRP_scheduleL1.PNG

WHO guidelines were released and the MfE proposed PM2.5 standard is equivalent to Interim Target (level four).

The Health and Air Pollution in NZ (HAPINZ) health impacts model estimated significant health impacts and social costs from exposure to NZ's relatively low levels of air pollution, even for people who live in places where the WHO 2021 guidelines are met.

Table 3.1: Standards, guidelines and regional targets for PM and NO_2 with the number of annual exceedances allowed in brackets

Pollutant (ug/m3)	Averaging time	NESAQ 2004	WHO 2005 Guideline	Regional targets (NRP)	NESAQ proposed (2020)	WHO 2021 Guideline
PM2.5	Annual	-	10	<=13	10	5
	24-hour	-	25 (3)	<=7	25 (3)	15 (3)
PM10	Annual	-	20	<=13		15
	24-hour	50 (1)	50 (3)	<=33	50 (1)	45 (3)
NO_2	Annual	-	40	-		10
	24-hour	-	-	<=66		25 (3)
	1-hour	200	200	-		200 (9)

3.3 Conclusion

- The three key indicators used to monitor air quality in NZ are particulate matter, black carbon and nitrogen dioxide, and are derived primarily from wood and coal burning, and motor vehicle exhaust and non-exhaust (brake and tyre wear) emissions.

- There are a range of national and international standards and guidelines for air quality to protect public health. NZ's proposed national standards (2020) are less stringent than the 2021 air quality guidelines set by WHO.

- Standards and guidelines are not limits to pollute up to as health impacts occur at relatively low levels of air pollutants, especially for vulnerable populations such as children, the elderly and those with underlying health conditions.

- Long term continual improvement in air quality across the whole region will reduce the health burden as well as aligning with emission reductions for climate change mitigation (Coulson et al., 2021).

- The NRP regional air quality targets could be reviewed using health risk assessment criteria in the 2021 WHO guidelines and the New Zealand cohort study (Hales et al., 2021).

4. Greater Wellington air monitoring programme

Objectives

Greater Wellington's air monitoring programme aims to:

- Determine compliance with the NESAQ in areas likely to breach the standards;
- Track trends in air quality and urban emissions, in particular impacts of domestic heating and transport throughout the region;
- Evaluate impact of policies and emissions changes on air quality, for example, impacts of public bus fleet decarbonisation on inner city air quality; and
- Carry out studies to improve our understanding of air quality processes and impacts on people's health and the environment.

Airsheds and air quality monitoring sites

Airsheds are analogous to catchments used to manage freshwater. Our region has eight airsheds, delineated by valleys in between steep hills or mountains, with unique microclimates, meteorological conditions, and combination of source pressures. Eight airsheds: Wellington City, Karori, Lower Hutt Valley, Upper Hutt Valley, Wairarapa Valley, Porirua, Kapiti Coast and Wainuiomata were gazetted in 2005 for PM10 NESAQ monitoring and air quality management (Davy, 2005a).

The Masterton Urban airshed was established in 2014 after an investigation found that the original Wairarapa Valley airshed gazetted boundary was too large and included rural areas which did not need to be managed for PM10 (Golder Associates, 2014). The airshed modelling used to inform the establishment of Masterton Urban airshed is discussed in section 7.2. The current airshed boundaries and locations of permanent air quality monitoring stations are shown in Figure 4.1.

Air quality monitoring is required where an airshed is likely to breach the NESAQ in the part of the airshed where people are exposed and where the standard is breached by the greatest margin or most frequently. Masterton Urban airshed is the only sub-regional airshed known to breach the PM10 standard.

Greater Wellington monitors air quality for a range of purposes, including tracking trends in traffic-related air pollution and impacts of interventions. This requires a monitoring programme that includes measuring non-NESAQ pollutants using 'indicative' methods. For example, a large monitoring network of low-cost passive diffusion samplers has been installed across the region to assess the impact of motor vehicle gas emissions on air quality (Mitchell, 2017).

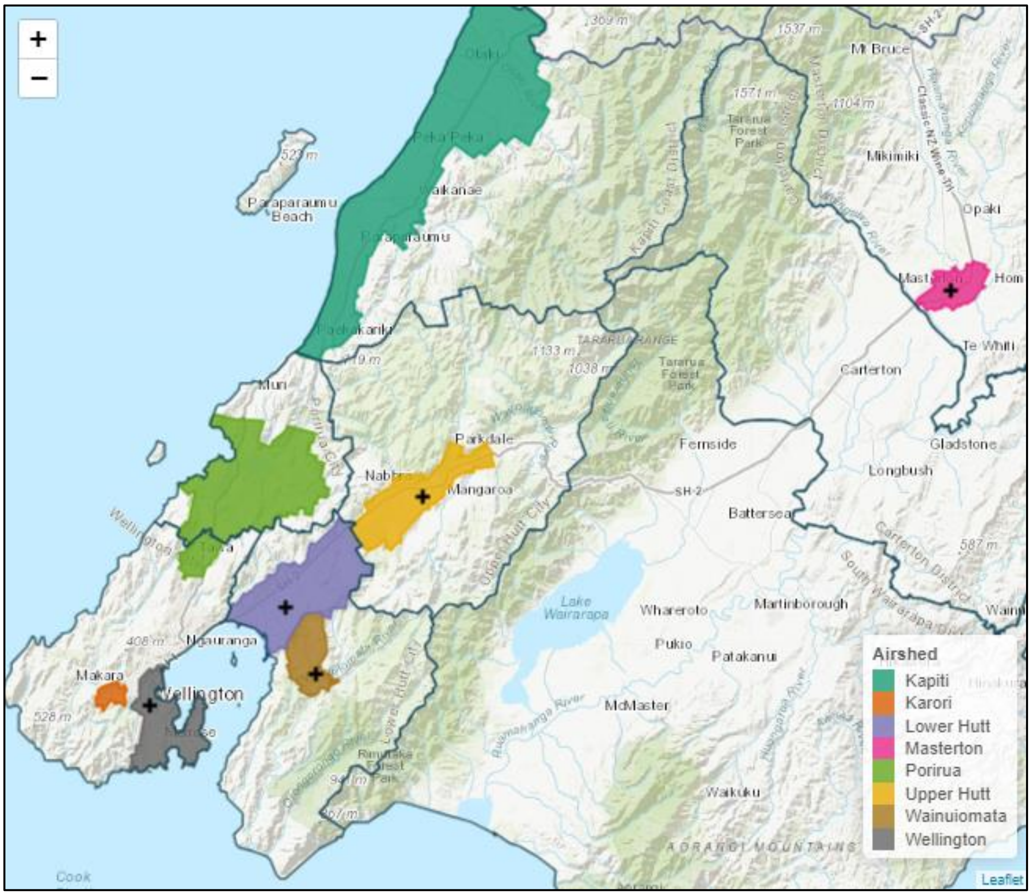

Figure 4.1: Greater Wellington sub-regional airsheds and territorial authority boundaries with locations of permanent air quality monitoring stations shown as +

4.1 Masterton air quality monitoring sites

Until the end of 2022, GW operated two monitoring stations – one located at Wairarapa College (Masterton West) and the other at Chanel College (Masterton East) (Figure 4.2).

PM10 has been monitored at Masterton West since 2002, NO_2 since 2003 and PM2.5 from 2011. Meteorological parameters were also measured at Masterton West using a 10 m mast. PM10 and PM2.5 were measured at Masterton East from 2013 to 2022.

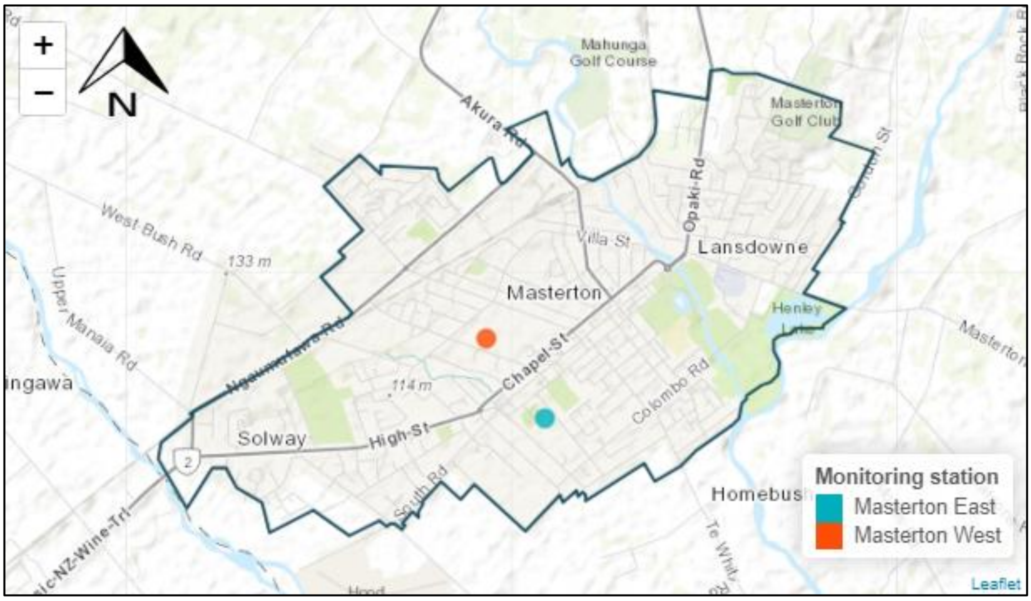

Figure 4.2: Location of the current Masterton West monitoring site and the closed Masterton East monitoring site

The Masterton East station was disestablished because it did not meet the new National Environmental Monitoring Standards (NEMS, 2022) requirements for air quality station siting and failed to meet minimum data capture requirements due to ongoing technical issues that could not be practically resolved.

NO_2 was also monitored at three road corridor sites using passive diffusion tubes on SH2, High Steet (since August 2016), on a local road, Masters Crescent (since August 2017) and in the CBD on Queens Street (2018 to 2022) (Figure 4.3). Passive diffusion tubes are a low-cost indicative monitoring method which provides an estimate of annual average NO_2 that can be compared to other sites across the region and nationally (NZ Transport Agency Waka Kotahi, 2024). This method cannot be used to assess compliance with air quality guidelines or standards, but over the long term is a reliable indicator of trends in traffic-related air pollution.

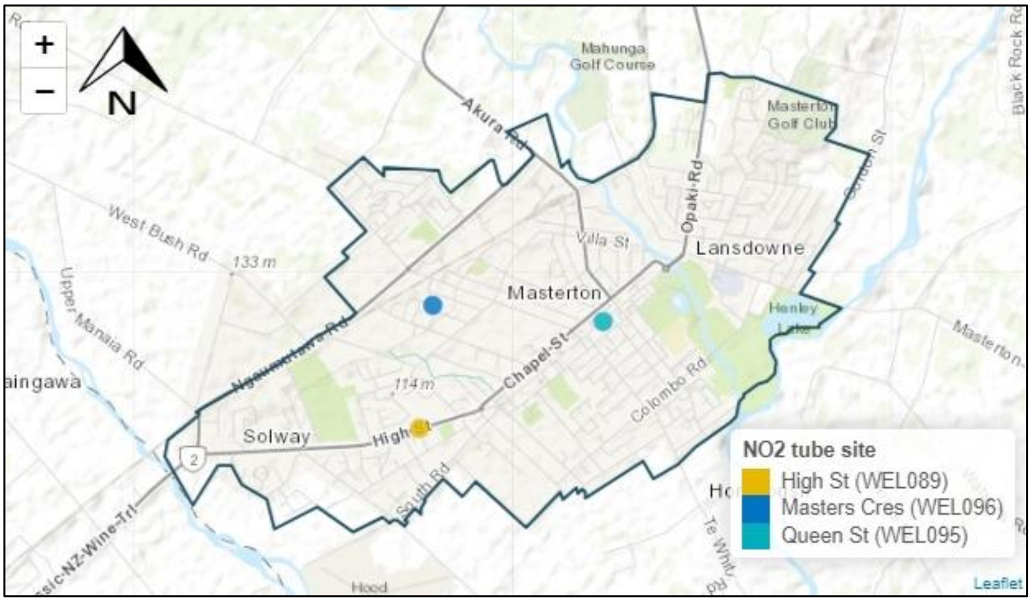

Figure 4.3: NO_2 passive diffusion tube monitoring sites. Note Queen Street site was disestablished in 2023.

5. Pressures on air quality

Air quality is determined by the following pressures:

- Quantity of emissions generated by human activities and natural sources;
- Weather conditions that affect the dispersion and dilution of air pollutant emissions; and
- Topography, which could for example cause emissions to accumulate in low lying valley areas.

Due to the impact of weather and topography, the relationship between emissions and air quality is non-linear, eg a 50% reduction in emissions does not necessarily result in a 50% reduction in air pollution.

5.1 PM emission inventories and Census data

Quantifying emissions

Emission inventories estimate the quantity of human-generated air pollutants emitted from different source sectors, such as motor vehicles, industry, and solid fuel home heating appliances[5]. They are a key tool for identifying sources of emissions and management options.

Activity data for local domestic heating emission inventories are typically based on a telephone survey of a sample of households. These surveys collect information on the type and age of solid fuel heating appliances used, frequency of use, and fuel consumption. Scaling up the survey responses to the number of households in an area of interest means the quantity of PM produced in an airshed can be estimated as follows:

Emissions = activity data (eg, number of households) x emission factor (g/kg) x fuel use factor (kg/day) (Wilton et al, 2015).

Emission factors used in the calculation vary by age and type of burner. The NESAQ required all wood burners installed (on properties less than 2 hectares) after 1 September 2005 to achieve an emissions rate of 1.5g PM per kg of dry wood burnt under laboratory test conditions[6]. Real-world (in-home) testing shows emission factors for NESAQ compliant wood burners are frequently much higher in in-home operation than their laboratory test (Wilton, 2012). Therefore, currently accepted emission factors for New Zealand inventories are 4.5 g/kg for NESAQ compliant burners and 10 g/kg for older burners (Wilton et al., 2015).

[5] Includes all appliances that use solid fuels (ie, wood, coal, pellets) such as wood burners, multi-fuel burners and open fires.
[6] NZS 4013 (2004)

Masterton emissions

Greater Wellington commissioned an emission inventory in 2008 to determine emissions of PM, carbon monoxide, oxides of nitrogen (NOx), sulphur oxides and CO_2 from domestic heating, motor vehicles, industrial activities and outdoor burning (Wilton & Baynes, 2008a). These emission inventory results were used to identify management options for reducing PM10 concentration to meet the NESAQ (Wilton & Baynes, 2008b). The most effective options for meeting the NESAQ were identified as regulation requiring non-NESAQ compliant wood burners be replaced when they reached 15 years, plus an incentives programme to achieve 50% conversion of solid fuel burners to zero-emission heating at their 15-year replacement date (Wilton & Baynes, 2008b).

In 2013, an updated PM domestic heating inventory was commissioned (Sridhar & Wickham, 2013) to provide emission inputs to air dispersion modelling required for changing the Wairarapa airshed boundary. Comparing the inventories, it's estimated that solid fuel emissions of PM decreased by 8% between 2008 and 2013.

A national study estimated that PM10 emissions in Masterton from home heating reduced by 14% between 2006 and 2013 (1,140 kg/winter day to 979 kg/winter day) (Wilton et al., 2015) which is a rate of change of approximately 2% per year. The national trend (kg/day winter) over the same period was a reduction of 22% attributed to improved wood burner technology, reduction in proportion of households using wood and decrease in coal use (Wilton et al., 2015).

Motor vehicles, outdoor burning and industry are minor sources of PM emissions compared to home heating (Figure 5.1).

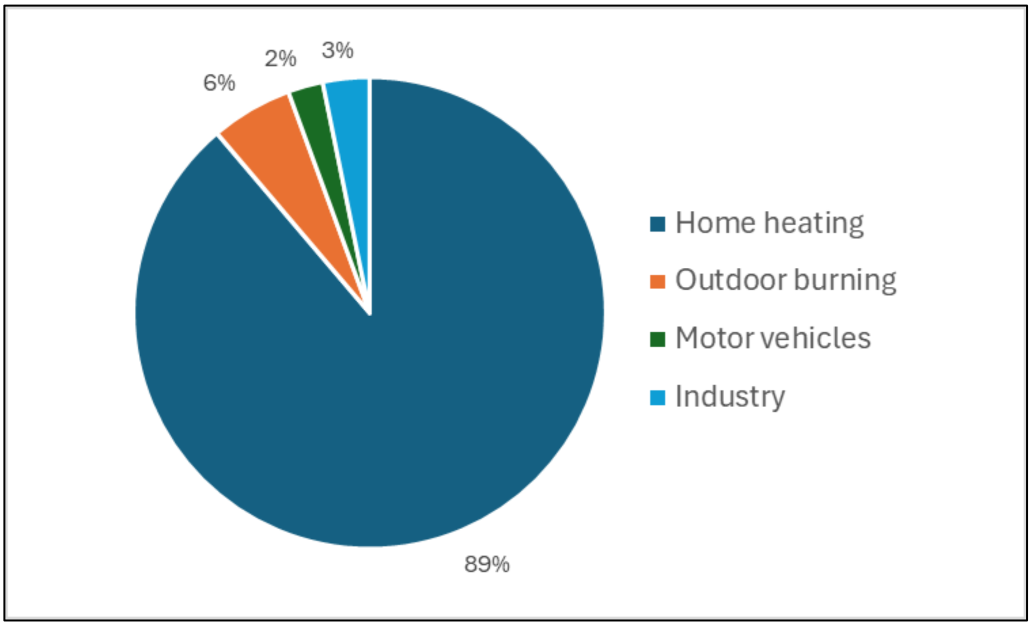

Figure 5.1: Masterton urban airshed PM10 emission sources percentage contribution to total annual PM. Source: Wilton et al., 2015.

The Waingawa industrial area southwest of the Masterton urban airshed has multiple PM industrial emission sources, including a sawmill and laminated timber products plant, concrete batching plant, asphalt plant, and a sawmill. Cumulatively these industries emit significant quantities of PM which are regulated separately through Greater Wellington air discharge resource consents. Air dispersion modelling carried out in 2013 included PM10 emissions from the industrial activities in Waingawa but conclude that these emissions were unlikely to contribute to breaches of the NESAQ inside the airshed but could cause exceedances within the industrial site boundary (Golder Associates, 2014).

Number of households using wood burners for home heating

A broad indicator of emissions pressure from home heating is the number of households using solid fuels for home heating obtained from the 5-yearly New Zealand Census of Population and Dwellings undertaken by Stats NZ. Summary data on home heating methods from the census for the Masterton urban area (SA3) show that between 2018 and 2023 the number of households using wood burners decreased by 231 and the number of households using heat pumps increased by 1,950 (Table 5.1). In 2023, the number of coal burners and pellet fires was extremely low. It is not possible to directly compare the change in proportion of household methods of home heating with previous census years as the census question in 2013 was based on fuel type not heating method[7]. The Greater Wellington 2013 home heating survey estimated 68.2% of households used wood burners, equivalent to 5059 appliances (Sridhar & Wickham, 2013). It's therefore plausible that there are now almost 600 fewer wood burners being used compared to 2013.

Figures from Masterton District Council Building Team show there were 3,263 consents issued for wood burner installations in the Masterton District from 2006 to 2024. Numbers per year have decreased since 2022. Data on whether these are replacement wood burners or new installations was not available. The Residential Tenancies (Healthy Homes Standards) Regulations 2019 requiring heating in main living rooms may have contributed to the increase in the number of heat pumps in Masterton.

[7] https://www.stats.govt.nz/assets/Methods/2018-census-changes-and-how-they-might-affect-the-data/2018-census-changes-and-how-they-might-affect-data.pdf

Table 5.1: Household heating methods for Masterton urban area (SA3) reported by the NZ Census years 2018 and 2023. A household may use multiple sources of heating, so totals do not add to 100%. Source: Infometrics[8]

	2023		2018	
Main types of heating	**Count**	**% of Total stated***	**Count**	**% of Total stated***
No heating used	42	0.5	54	0.7
Heat pump	5,583	71.3	3,633	50.4
Electric heater	2,313	29.5	2,409	33.4
Fixed gas heater	339	4.3	348	4.8
Portable gas heater	162	2.1	480	6.7
Wood burner	4,473	57.1	4,686	65
Pellet fire	42	0.5	51	0.7
Coal burner	21	0.3	39	0.5
Other types of heating	102	1.3	129	1.8
Not elsewhere included	771		663	
Total stated	**7,830**	**100**	**7,206**	**100**
Total occupied dwellings	8,598		7,824	

*Percentages in all census data are calculated using 'Total stated' as the denominator. So that households that did not respond to this question are not included.

5.2 Sources of PM estimated from receptor modelling

GNS Science has carried out receptor modelling on behalf of Regional Councils across New Zealand to assess the relative PM source contributions to monitored air quality. Receptor modelling uses the elemental fingerprint of particulate collected on air filters and multivariate statistical methods to identify PM source profiles and their relative contributions to PM levels in air. Unlike emission inventories, receptor modelling provides information on natural sources of PM and represents the exposure of people living near the monitoring site to different sources of air pollution.

A receptor modelling study was carried out using air particulate filters collected at Masterton West from 2002 to 2004 (Davy, 2007). This was the first New Zealand study of PM sources in a residential area based on monitoring data.

[8] https://rep.infometrics.co.nz/masterton-district/census/indicator/main-types-of-heating

The study identified sources, common to most urban locations in New Zealand, as follows:

- Motor vehicles, from tailpipe and non-exhaust emissions (e.g., road dust, tyre and brake wear);
- Residential heating emissions (mainly wood burning) for winter heating;
- Secondary sulphate from gas-to-particle atmospheric reactions, mainly marine plankton and volcanic activity (Whakaari / White Island);
- Marine aerosol (sea salt) generated in the oceans; and
- Crustal matter (soil) from local dust-generating activities.

The study found that 51% of annual PM10 and 75% of PM2.5 originated from home heating emissions (Figure 5.2). Natural sources, such as marine aerosol, and secondary sulphate were also an important component, making up about 26% of PM10 and 15% PM2.5. The soil source was much lower on the weekend compared to weekdays indicating that it was associated with people's workday activities.

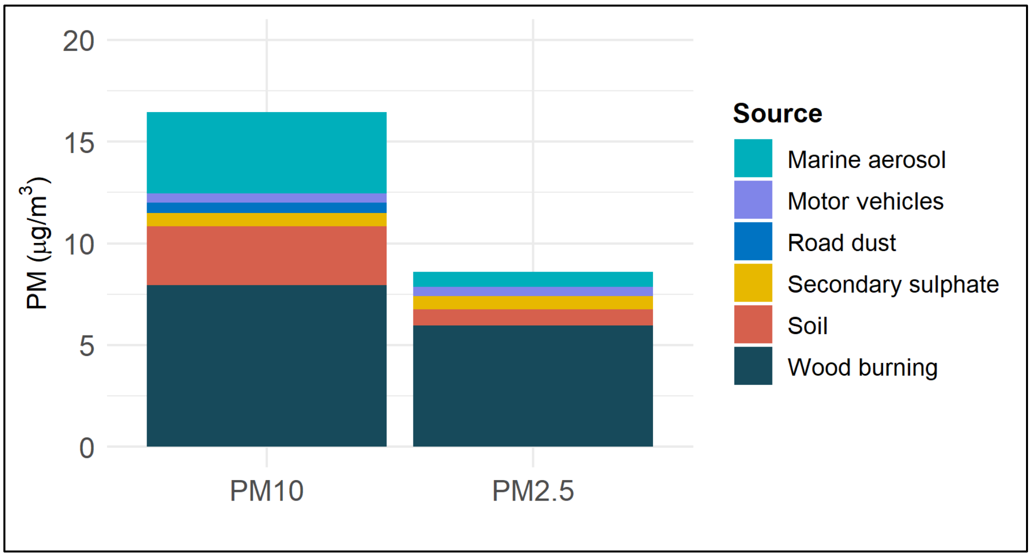

Figure 5.2: Relative source contributions to annual PM concentrations measured at Masterton West from 2003 to 2004 (Davy, 2007)

In 2018, the relative contribution of PM2.5 sources measured at Masterton East was assessed by GNS Science for Greater Wellington (Figure 5.3). The study found wood burning was the dominant source of PM2.5 during the winter months, comprising 84% of the annual average PM2.5.

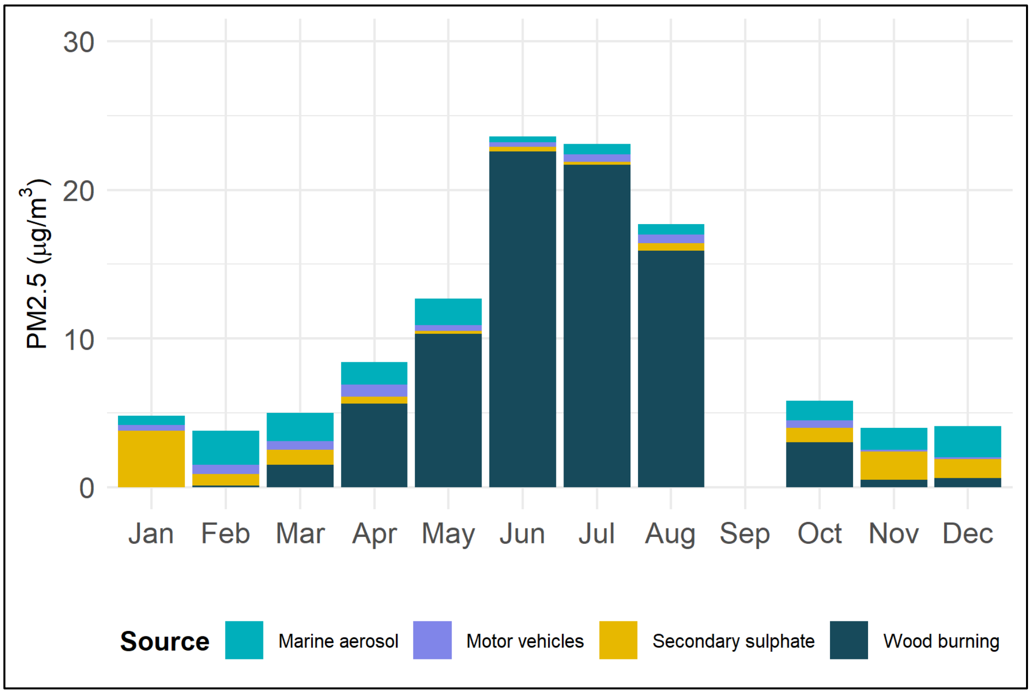

Figure 5.3: Average monthly PM2.5 measured at Masterton East in 2018 by emissions source. Note data for September 2018 was not captured due to instrument malfunction. Data from GNS Science.

5.3 Motor vehicle emissions

The total pollutant emissions from vehicles depend on how far they travel (Vehicle Kilometres Travelled or VKT), engine type and emissions-control technology and what fuel they use (petrol or diesel).

Vehicles must meet emissions standards when they are imported, and these standards have become stricter over time. Nationally, emissions per km (g/km) have reduced for many pollutants (particularly CO and PM2.5) due to the progressive tightening of emissions standards and improvements in engine technology and fuel quality (Kuschel, 2022). However, growth in VKT can offset some of the benefits of lowered per vehicle emissions (NZ Transport Agency Waka Kotahi, 2021).

Annual Average Daily Traffic (AADT) counts measured by NZ Transport Agency Waka Kotahi on High Street, Solway (SH2) have gradually increased, with a fall off during 2020 due to COVID-19 traffic disruption and a reduction in 2023 (Figure 5.4). The annual average traffic data for 2024 was not available at the time of writing but it unlikely to have increased from 2023 as a fall in traffic volumes is consistent with weaker economic conditions.

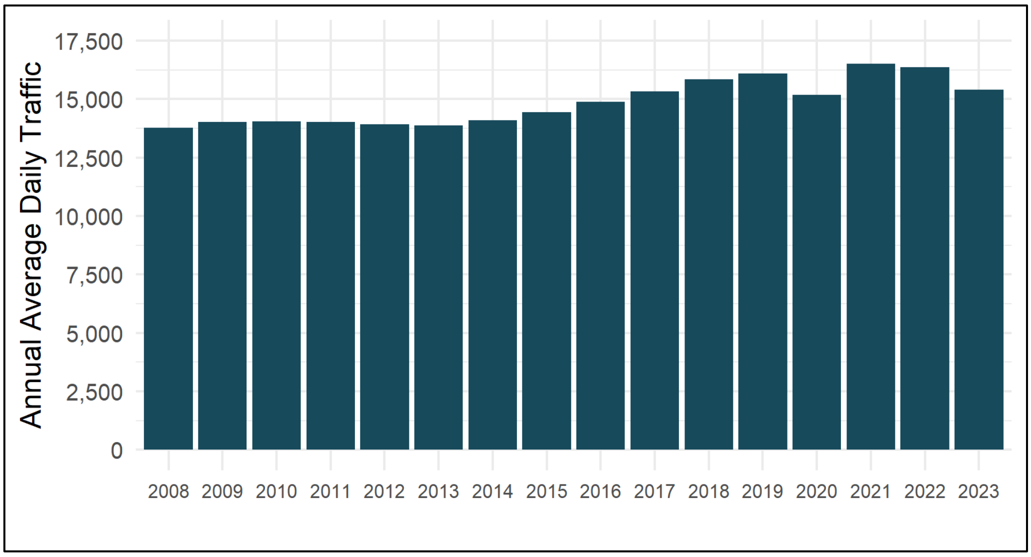

Figure 5.4: Annual Average Daily Traffic monitored on High Street, Masterton. Source: NZ Transport Agency Waka Kotahi Regional Continuous Site ID:00200886.

5.4 Impact of weather, climate and topography on air quality

During wintertime Masterton is prone to conditions that restrict the dispersion of air pollutants. Although the prevailing winds over New Zealand are westerly, the local winds are highly influenced by the sheltering effects of the ranges to the west. This results in a high frequency of calm or very light winds in the inland northern plains area surrounding Masterton; this effect is usually more pronounced at nighttime (Chappell, 2014).

Elevated wintertime PM concentrations in New Zealand towns where solid fuels are used for home heating, including Masterton, are more strongly correlated with ‘environmental confinement’ of emissions due to meteorology and topography than population or number of wood-burning households (Trompetter et al., 2010). Environmental confinement occurs when vertical dispersion of particulate matter from wood burners is restricted to a limited volume of air caused by a low-level temperature inversion.

During winter, when the night sky is clear (cloudless) and wind conditions are calm, a low-level temperature inversion can form as the ground cools and heat is lost to space. A temperature inversion is characterised by temperature increasing, instead of decreasing with altitude. A layer of warmer air over the top of the cold air underneath prevents the cold air from moving upwards and dispersing the wood burner emissions which have been emitted close to ground level. Topography can further strengthen the temperature inversion as cold air drains downwards from nearby hills leading to cooler air pooling in lower-lying areas, cooling the ground even further (Griffiths 2011). Masterton is affected by cold air drainage from the surrounding hills (Figure 5.5).

The temperature inversion usually breaks up at sunrise when the ground is warmed, and the air becomes more buoyant and moves upwards, dispersing any air pollutants that have accumulated.

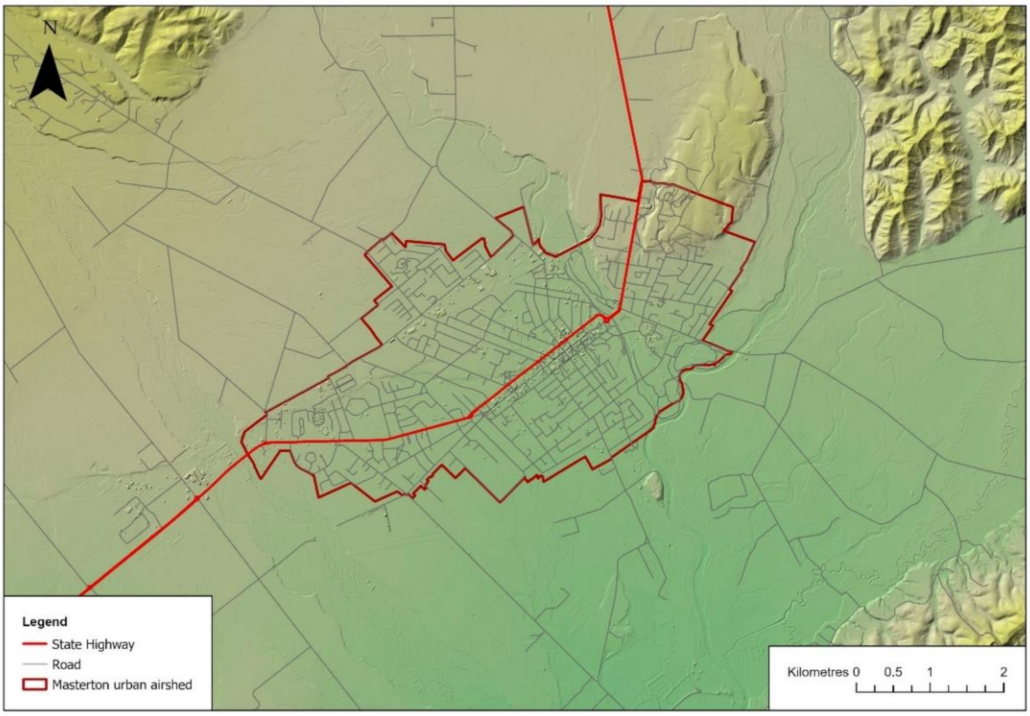

Figure 5.5: Digital elevation map (1m resolution) centred on the Masterton urban area. Darker green areas show lower elevation. (Source: Greater Wellington GIS Team).

Local meteorological conditions that restrict the dispersal of air pollutant such as temperature inversions are associated with anticyclonic systems or highs. An analysis of the relationship between the 12 Kidson weather types[9] (Kidson 2000) and high PM10 concentrations in Masterton found there were two main weather types most often linked to poor air quality: the H type and the HNW type (Figure 5.6) (Griffiths, 2011).

The H weather type is associated with large, slow-moving anticyclones, which produce clear skies and light winds leading to temperature inversions. The HNW can also have clear skies and low wind speeds in the short 'turnaround' period between southerly and imminent northerly flow. Further analysis found that winter days where PM10 was higher than 33 μg/m^3 (between 2002 and 2015) were preceded in the three days before by a strong anticyclone moving across the Tasman Sea from Australia (Fiddes et al., 2016).

[9] The Kidson weather types developed for the New Zealand Region by Kidson (2000) are based on a cluster analysis of midnight and midday 1000 hPa height fields from the NCEP/NCAR Reanalysis Version 1.

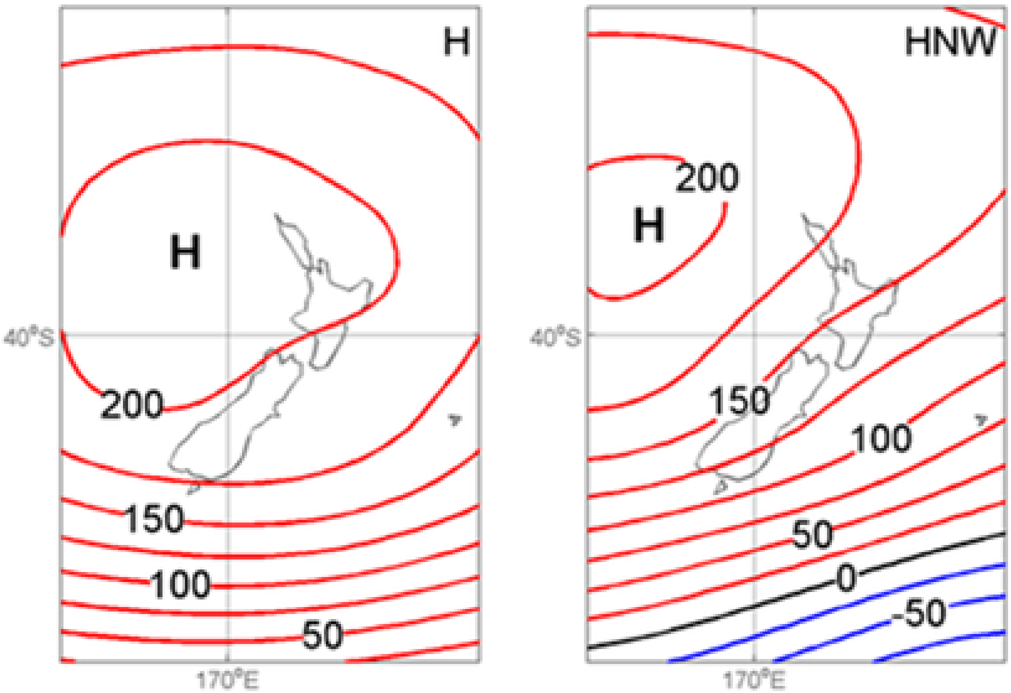

Figure 5.6: Kidson weather types (H and HNW) associated with poor air quality days in Masterton (Griffiths 2011). Image modified from Kidson (2000).

5.5 Climate change

Masterton's mean temperature has increased at a rate of approximately 0.1°C per decade from 1912-2020, with a corresponding positive trend in the number of warm nights, and negative trend in the number of cold days, cold nights and frost days[10] (Macara et al., 2021).

NIWA's climate change projections for the Wairarapa include considerable increases to the annual average number of growing degree days and drought potential due to higher temperatures, increases in the number of hot and extreme hot days, and reductions in the number of frost days (Macara et al., 2021).

Over the longer term, climate change will impact air quality. Although not possible to be precise, it is likely that reduced numbers of frost days (associated with clear skies and light winds) will lead to fewer temperature inversions during winter and therefore fewer high pollution nights, where dispersion of emissions are restricted. Warmer winter nights may also reduce the need for home heating and therefore reduce emissions into the airshed.

A warmer and drier climate is expected to increase wildfire risk (depending on vegetation type and proximity to urban areas) and therefore potential episodes of particulate matter and black carbon air pollution. In addition, Australian wildfires and desert dust storms may also impact local air quality, but this is difficult to predict (Davy et al, 2024). Drier soils and

[10] A frost day is defined as when daily minimum temperature is equal or lower than 0oC.

riverbeds may also lead to more windblown dust (Bolton, 2018). Less rainfall may also result in decreased particulate matter wet deposition, ie, particles being washed out of the atmosphere (Davy et al, 2024). Levels of marine aerosol are projected to increase over New Zealand due to increases in wind speeds and potentially sea-surface temperatures (Davy et al., 2024).

Although, not a regulated contaminant, pollen allergen production is likely to increase due to increasing atmospheric CO_2 levels and longer growing seasons (Newnham, 2021).

5.6 Conclusion

- Residential wood burning is the main source of winter PM2.5.
- Emissions of PM from solid fuel heating are trending downwards as heat pumps are replacing wood burners as the main method of heating and as people replace their old wood burners at the end of their working life with NESAQ compliant models. Updated information is needed on the number of older burners (those installed prior to 2006) that are still in operation, as these are likely to be the highest emitters.
- The climate and topography of Masterton make it prone to air pollution episodes from home heating emissions, especially during winter anticyclonic conditions.
- Masterton is getting warmer and is predicted to be drier under climate change. While no predictions have been made for the impacts of climate change on air quality it is possible that home heating emissions will decline, and the frequency of high pollution days will reduce as dispersion conditions improve. There is potential for additional sources of PM, such from local windblown soils, wildfires and marine aerosol.

6. Air quality monitoring trends

Two key indicators of air quality, PM and NO_2 (nitrogen dioxide) are monitored in the Masterton urban airshed. PM2.5 is primarily associated with home heating emissions from solid fuel burners and NO_2 is an indicator for motor vehicle exhaust emissions.

Air quality monitoring results are compared to the NESAQ and World Health Organization (WHO) guidelines for consistency with Statistics NZ, Ministry for the Environment environmental domain air quality indicator reporting and LAWA[11].

Elevated winter levels of other harmful pollutants associated with residential wood burning, such as benzo(a)pyrene (BaP), arsenic and lead, have been detected in Masterton, but are not routinely monitored.

6.1 Particulate matter trends

Annual average PM10 and PM2.5

Like most other national monitoring sites, PM2.5 in Masterton does not meet the 2021 WHO annual average guideline (MfE & Stats NZ, 2024) (Figure 6.1). From 2008 onwards annual average PM2.5 and PM10 have trended downwards with some variability between years driven by weather effects (Figure 6.2). Note PM10 data prior to 2008 not used due to a change in monitoring method.

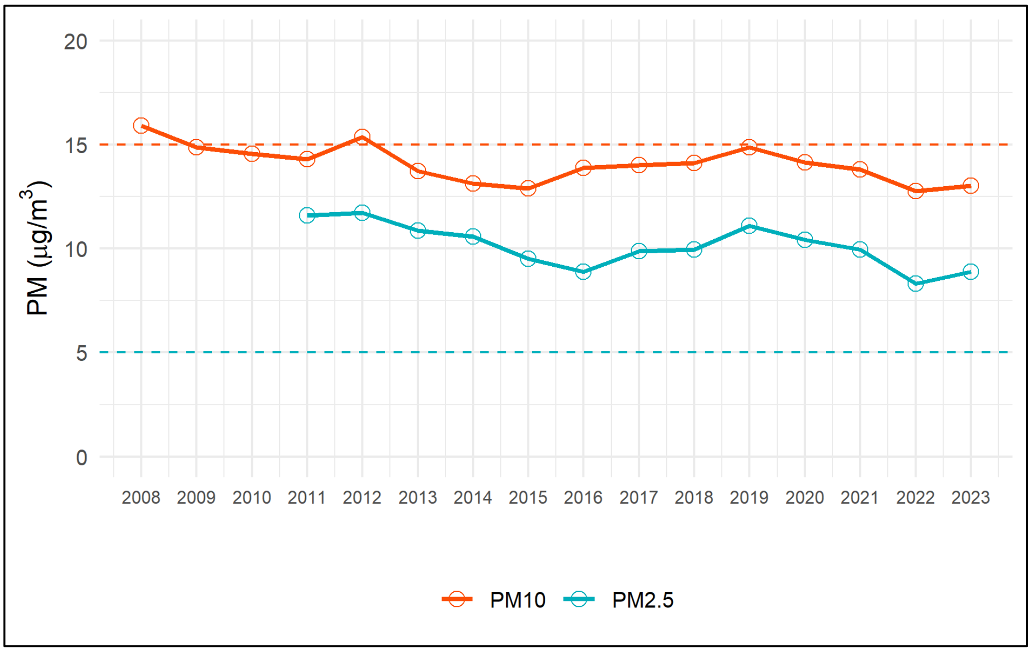

Figure 6.1: Annual average PM measured at Masterton West with the 2021 WHO guideline for each size fraction as a dashed line

[11] https://www.lawa.org.nz/explore-data/air-quality

Daily PM10 exceedances

The PM10 NESAQ allows only one day per year to exceed 50 µg/m³. Figure 6.2 shows the number of exceedance days per year recorded at each monitoring site in operation. The number of exceedances per year largely depends on frequency and persistency of cold calm nights under temperature inversions which restrict the dispersion of home heating emissions (Trompetter et al., 2010).

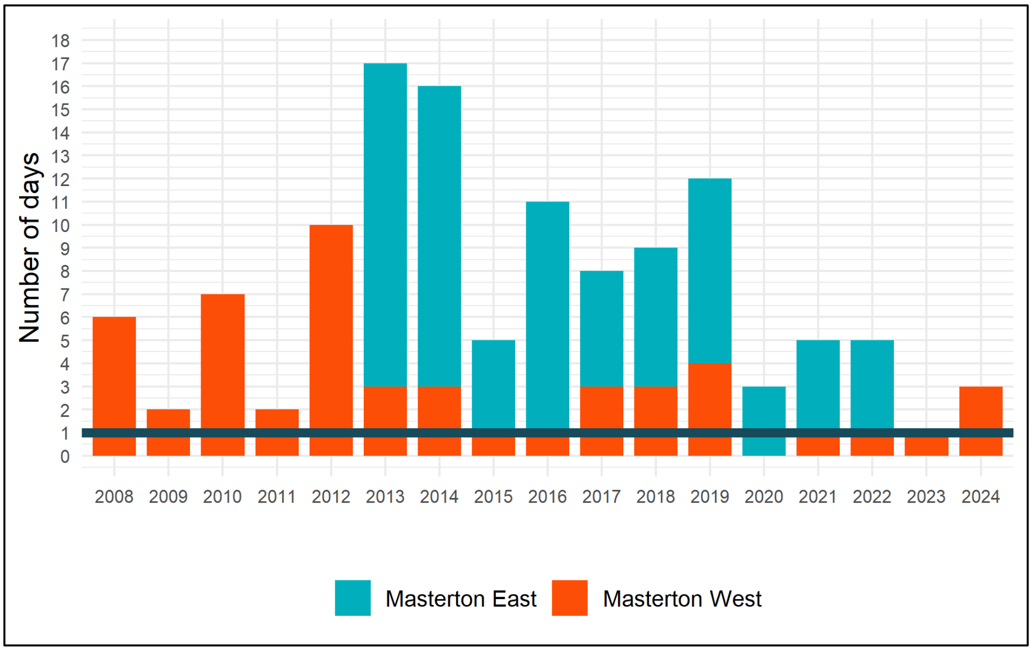

Figure 6.2: Number of days per year recorded at Masterton West and Masterton East where PM10 exceeded the NESAQ threshold of 50 µg/m³

Daily PM2.5 exceedances

The 2021 PM2.5 WHO guideline allows for no more than three days per year where PM2.5 exceeds 15 µg/m³. This replaces the less stringent 2005 WHO guideline of 25 µg/m³. In 2020, MfE proposed adopting the 2005 WHO guideline through an amendment to the NESAQ. Figure 6.3 shows the number of days per year recorded at Masterton West that did not meet the 2021 and 2005 WHO guidelines for PM2.5.

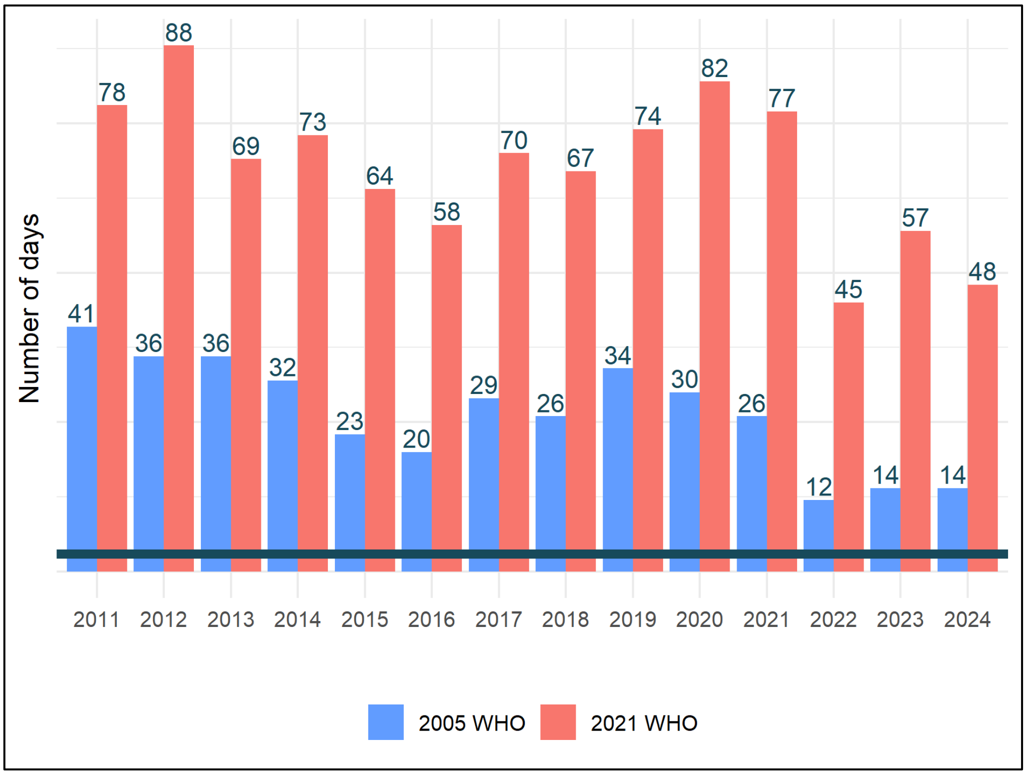

Figure 6.3: Number of days per year recorded at Masterton West where PM2.5 did not meet the 2021 and 2005 WHO guidelines. The horizontal line shows the number of allowed exceedances per year.

All PM2.5 exceedance days occurred during the home heating period (April to September) except for summer 2019/2020 when there were two PM2.5 exceedances attributed to PM from Australian bush fires and dust storms reaching New Zealand (Davy & Trompeter, 2022).

The number of days per winter that exceeded the guideline fluctuated depending on frequency and duration of weather patterns known to restrict the dispersion of wood smoke emissions. Modelling the impact of weather conditions on number of exceedances is technically challenging, but mathematically as the winter PM2.5 average decreases so does the number of exceedances.

Compared to 30 other sites in New Zealand, Masterton had the fifth highest number of PM2.5 exceedances for the period 2020 to 2023 (MfE & Stats NZ, 2024).

Winter average PM2.5

Trends in PM are affected by year-to-year variation in local weather patterns (eg, windiness and temperature). Over the 13-year monitoring period, average winter PM2.5 was highest in 2012, when both temperature and wind speed were the lowest. Conversely, PM2.5 concentration was lowest in 2022, when both temperature and wind speed were relatively high (Figure 6.4).

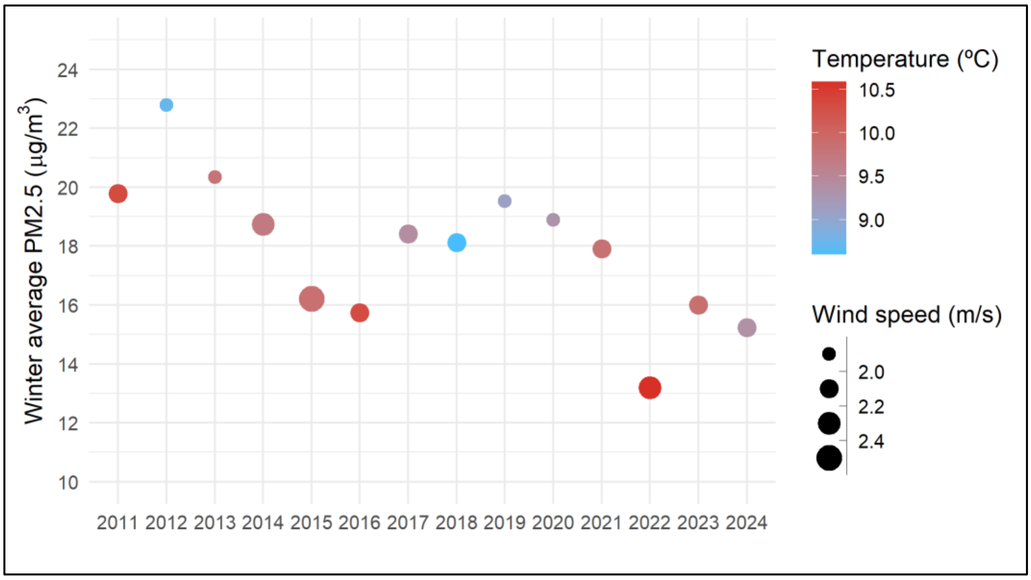

Figure 6.4: Winter average PM2.5 (May to August) coloured by temperature (cooler temperatures in blue and warmer in red) and sized by wind speed (larger diameter points show higher wind speeds)

Figure 6.5 shows average PM10 and PM2.5 levels measured at Masterton West during the heating season (from April to September) improving over time, although concentrations fluctuated year-to-year due to meteorological factors.

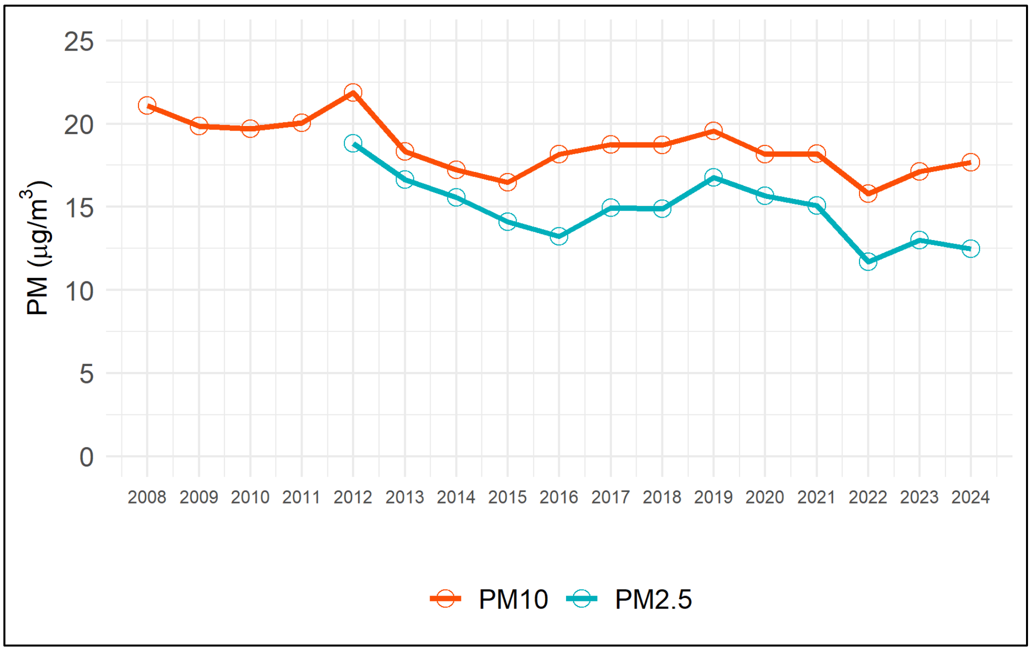

Figure 6.5: Particulate matter averages for the heating period (April to September) measured at Masterton West

De-weathered winter average PM2.5

Statistical modelling described in Appendix A was used to remove, as much as possible, the impact of weather conditions (eg, wind speed and temperature) on average winter quality over the home heating season. This technique, called meteorological normalisation or de-weathering, helps show any underlying trend in emissions once the impact of weather on air quality is removed (Grange & Carslaw, 2019). Figure 6.6 shows a downward trend in de-weathered annual PM winter concentrations, consistent with reducing emissions from home heating.

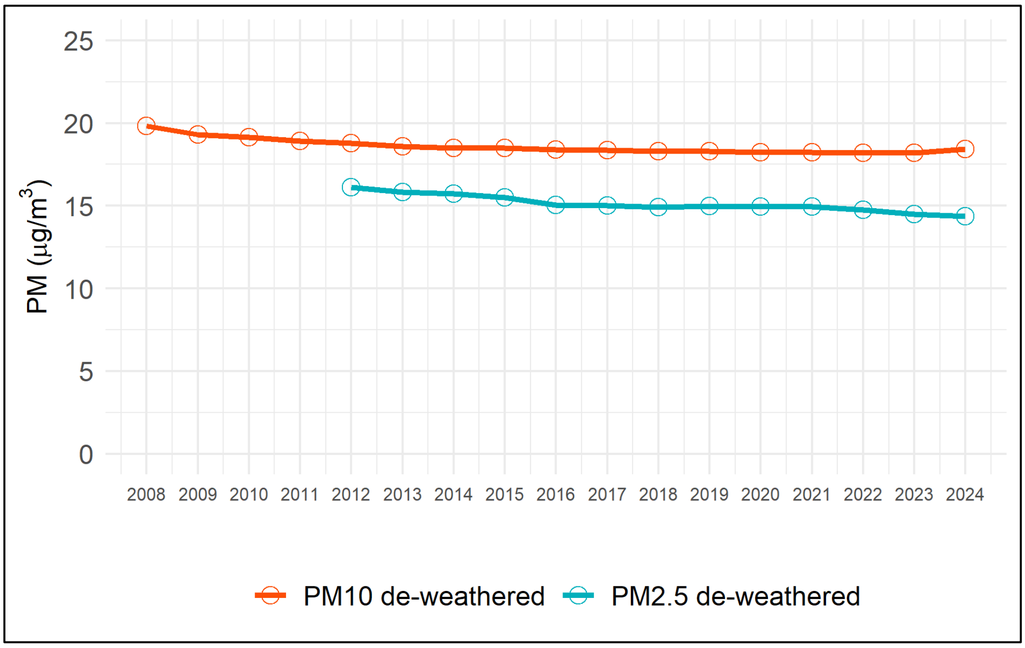

Figure 6.6: De-weathered particulate matter averages for the heating period (April to September) measured at Masterton West

The direction and statistical significance of the trend in de-weathered winter averages was assessed using the Theil-Sen estimator, a non-parametric method commonly used for analysing air quality data (Carslaw & Ropkins, 2012).

Table 6.1: PM2.5 Theil-Sen trends in observed winter averages and de-weathered winter averages, expressed as percentage change per year for the period shown. 95% confidence interval in square brackets and statistical significance denoted by *.

Metric	Period	Observed	De-weathered
PM10	2008-2024	-0.9 [-1.54, -0.57]**	-0.3% [-0.6, -0.2]***
PM2.5	2012-2024	-2.1 [-3.2, -0.4]*	-0.8% [-1.0, -0.5]***

$p < 0.001$ = *** $p < 0.01$ = ** $p < 0.05$ = *

This de-weathered PM winter trend suggests that wood burner emissions are slowly reducing over time. This finding is consistent with the national emission inventory trend of reducing emissions as older higher polluting burners are replaced at end of their working life (Wilton et al, 2015).

6.2 Hazardous air pollutants

Wood smoke from home heating also contains other harmful pollutants, termed ‘hazardous air pollutants’ as they are known or suspected to cause cancer or have other serious health effects. These include:

- Benzo(a)pyrene (BaP) a carcinogen and indicator of polyaromatic hydrocarbons (PAHs) released during incomplete wood combustion, especially from softwoods, such as pine.
- Arsenic a carcinogen produced when wood treated with copper-chrome-arsenate (CCA), eg decking offcuts, is burnt.
- Lead a neurotoxin, especially for young children, emitted when wood coated with legacy lead-based paint is burnt.

These pollutants are not routinely monitored in air as they are resource intensive, requiring particulate matter to be collected on filters and analysed in a laboratory by spectrometry or nuclear techniques.

Benzo(a)pyrene (BaP)

Elevated wintertime BaP occurs across New Zealand, primarily associated with winter wood burning for residential heating (Cavanagh, 2024). A GNS study in 2010 found that PAHs made up 0.3% of winter PM2.5 in Masterton, with average BaP of 4.1 ng/m^3 (Ancelet et al., 2013).

Arsenic and lead

Elevated levels of arsenic and lead in PM from residential winter wood burning have been found in many parts of New Zealand (Cavanagh, 2024). The sources were attributed to the intermittent burning of CCA treated timber and waste wood painted with old lead-based paint (Davy & Trompeter, 2018).

Arsenic and lead were found in Masterton PM air filter samples collected between 2002-2004 (Davy, 2005b, Davy, 2007), in winter 2010 (Ancelet et al., 2012) and more recently in 2018 (Figure 6.7).

A home heating survey conducted in Masterton found that 16 % of wood burners always, often or sometimes burn decking or fence post offcuts on their fires (Sridhar & Wickham, 2013). It is highly likely that decking or fence post offcuts reportedly used as firewood were treated with CCA.

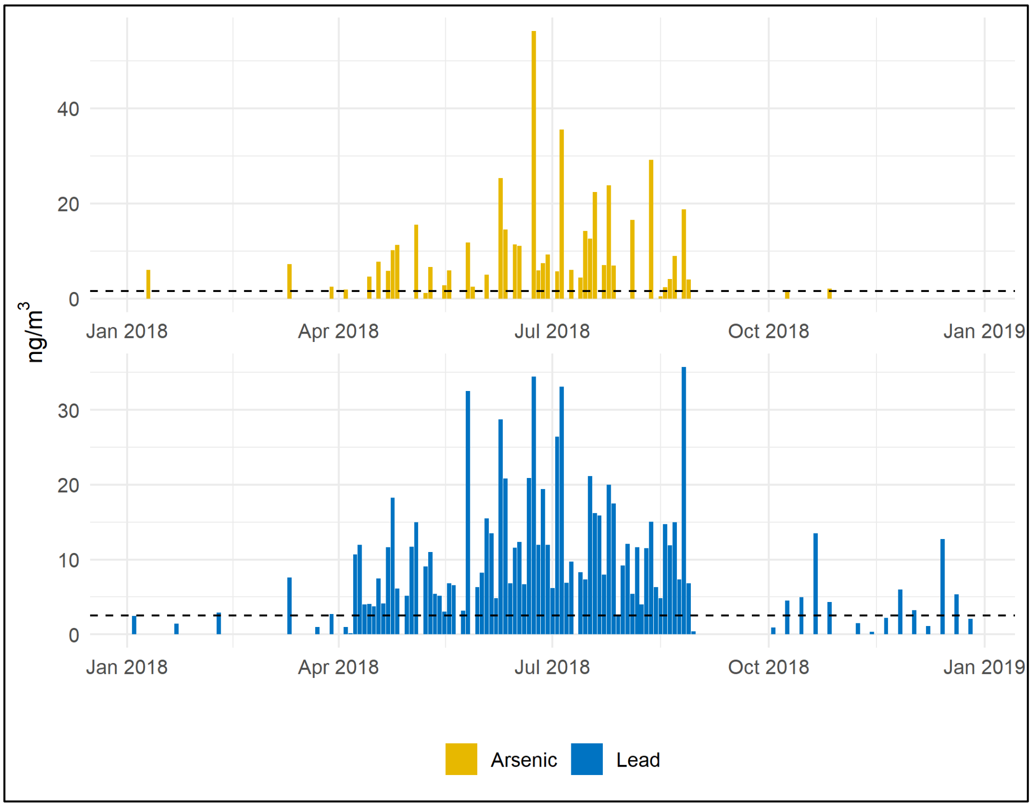

Figure 6.7: Levels of arsenic and lead in air found in PM2.5 filters at Masterton East (2018). Concentrations determined by GNS Science using X-Ray Fluorescence (XRF). The dashed line shows the level of detection for each element.

People are also exposed to lead and arsenic from other routes, such as their diet, soils or from their occupation. When CCA timber is burnt, arsenic accumulates in the ashes and can be inhaled when cleaning the firebox and become incorporated into household dust. An international study found New Zealand indoor household dust samples had the greatest enrichment of arsenic compared to background soil concentrations for all countries studied (Isley et al., 2022).

The health impacts of exposure to BaP, arsenic and lead in air have not been assessed in New Zealand. Although it is expected that managing wood burner emissions to reduce PM2.5 would lead to a reduction in BaP, but this assumption needs to be verified (Cavanagh, 2024).

There is some evidence showing that emissions of PAHs are higher from softwoods (eg, pine) than hardwoods (eg, blue gum) and higher under low-burn inefficient combustion conditions compared to high-burn conditions (Ancelet et al., 2011). Therefore, the fuel type and optimising wood burner operation are key factors for reducing levels of harmful pollutants in air and in indoor dust.

6.3 Trends in nitrogen dioxide from transport

Nitrogen dioxide is a harmful pollutant and is a strong marker for vehicle exhaust emissions. Roadside sites on the footpath close to busy roads are useful for tracking local trends in motor vehicle emissions and represent people's exposure to traffic-related air pollution as they travel on the road network, including inside vehicles, public transport, walking and cycling. NO_2 levels drop off with distance from the roadside and are generally much lower in residential areas away from busy roads (Coulson et al., 2024). Despite the relatively low levels of NO_2 where most people live, the social costs of the health impacts in New Zealand are significant (Kuschel et al., 2022).

Masterton West monitoring site is set well back from the influence of a nearby road and therefore NO_2 measured at this site represents the cumulative impact of dispersed vehicle emissions from many roads forming the 'urban background' concentration. Whereas NO_2 measurements at the roadside tube site capture the local influence of traffic on High Street (SH2), known as the roadside increment. The roadside increment is the difference between the urban background concentrations (which are fairly uniform) and the concentration measured at the roadside.

Annual average NO_2 concentrations measured at both Masterton West and the roadside tube site trended down over their respective measurement periods (Figure 6.8).

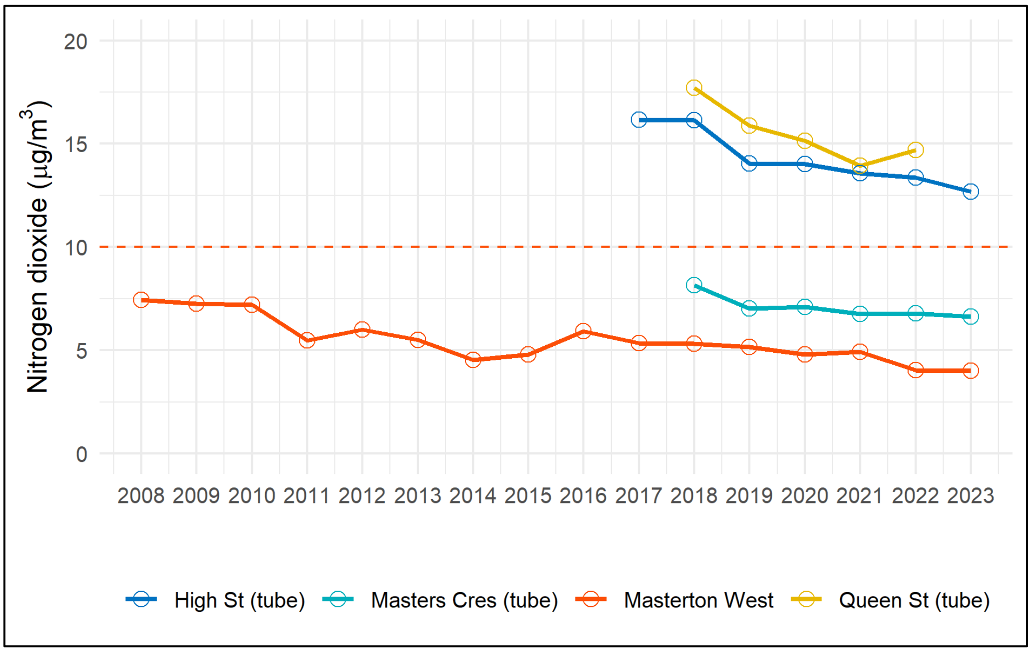

Figure 6.8: Annual average nitrogen dioxide measured at Masterton West and at the passive diffusion tube monitoring sites. The dashed line shows the WHO 2021 guideline which can be compared to Masterton West.

NO_2 trends are affected by year-to-year variation in local weather patterns (eg, windiness and temperature). Statistical modelling described in Appendix A was used to remove as much as possible the impact of weather conditions on annual NO_2 concentrations.

Figure 6.9 shows the downward trend in de-weathered annual average concentrations, suggesting that underlying NOx emissions from motor vehicles are reducing at the roadside site and in the urban background (represented by Masterton West). Although subtle, there was a small downward step in the de-weathered roadside tube NO_2 in 2020 which matches the fall in vehicle numbers on the road adjacent to the tube site due to COVID-19 travel disruptions (see Figure 5.4).

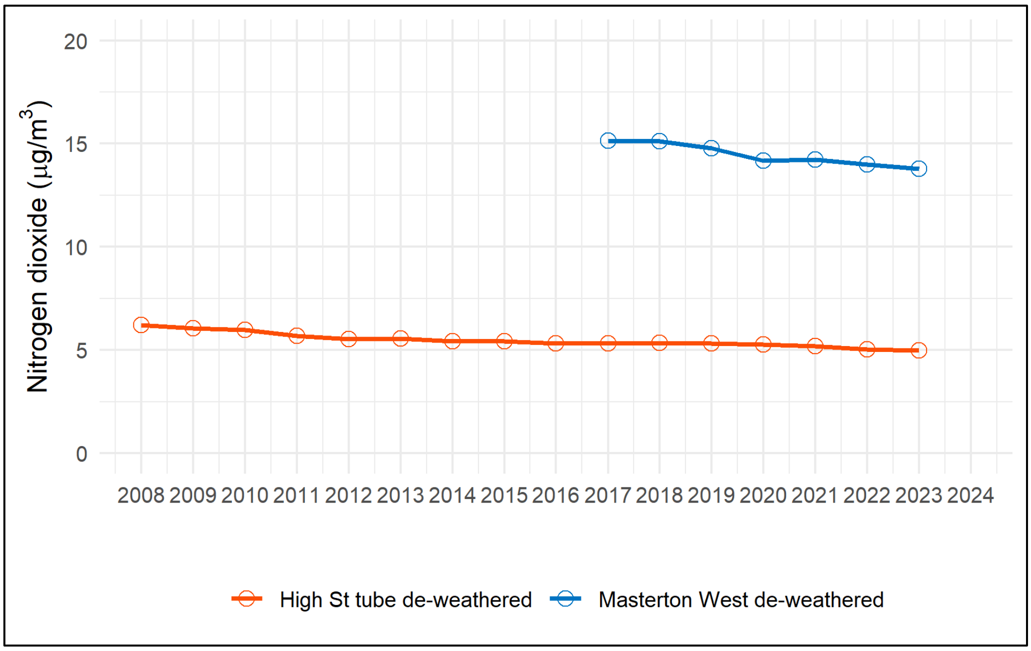

Figure 6.9: De-weathered annual average nitrogen dioxide measured at Masterton West and the roadside tube monitoring sites

Table 6.2: NO_2 Theil-Sen trends in observed annual averages and de-weathered annual averages, expressed as percentage change per year for the period shown. 95% confidence interval in square brackets and statistical significance denoted by *.

Site	Period	Observed	De-weathered
Masterton West	2008-2023	2.9% [-3.3, -1.7]***	1.1% [-1.4, -0.8]***
High Street	2017-2023	3.6 % [-5.4, -1.6]***	1.6% [-2.0, -0.9]***

p < 0.001 = *** p < 0.01 = ** p < 0.05 = *

The downward trend in urban background NO_2 is consistent with national trends for most monitoring sites between 2014 and 2023 (MfE & Stats NZ, 2024; NZ Transport Agency Waka Kotahi, 2024). The gradual decline in roadside NO_2, despite increasing vehicle travel was attributed to improvements in vehicle emissions, noting there were some impacts from COVID-19 travel restrictions.

6.4 Conclusion

- PM10 and PM2.5 annual averages are trending downwards with some year-to-year variability due to meteorology. PM10 annual average meets the 2021 WHO guideline but the PM2.5 annual average does not.

- Once the impact of meteorology on PM2.5 average concentrations is reduced, there appears to be a slow downward trend during the home heating period, April to September. This is consistent with the trend in home heating emissions discussed in Section 5.1.

- PM10 measured at Masterton West is very close to meeting the NESAQ and losing its *polluted airshed* status, as the average number of exceedances in the previous 5 years (2020 to 2024) was 1.2 which is close to the target of 1. However, when Masterton East data is included, the average number of exceedances was 3 per year.

- The number of PM2.5 exceedances compared to the 2021 and 2005 WHO guidelines appear to be trending down overall, but the number varies from year-to-year in response to meteorological conditions. Significant home heating emissions reductions would be needed to meet the 2021 WHO guidelines.

- Throughout New Zealand and in Masterton hazardous air pollutants are co-emitted with PM from domestic fires, including arsenic from burning treated timber and benzo(a)pyrene when naturally occurring organic compounds in wood are incompletely combusted.

- Annual average nitrogen dioxide measured at Masterton West representing the ‘urban background’ met the 2021 WHO guidelines. NO_2 appears to be reducing slowly overtime as the vehicle fleet modernises and in recent years likely reflecting impact of COVID-19 and the economic downturn on vehicle activity. The roadside indicative monitoring site, operating for a shorter period, also shows improving NO_2 levels.

7. Spatial evaluation of particulate matter from home heating

Air quality across an area varies in space and in time depending on emission sources, their strength, topography and atmospheric processes that favour or hinder pollutant dispersion. Therefore, a single monitoring station is unlikely to capture the spatial and temporal variation in air pollutant levels in an airshed.

Spatial studies have been conducted in Masterton to provide information on the extent of the airshed boundary, the representativeness of monitoring sites, to optimise future decisions about where to monitor, and to identify areas for targeted interventions to reduce emissions.

7.1 Air quality differences between Greater Wellington monitoring stations

A GNS study during winter 2010 collected particulate matter on filters at Masterton West and at another location, approximately 1.2 km to the southeast (Masterton East) (Figure 4.2). The study found both sites were affected by the same PM emission sources, namely local wood burning, soils, motor vehicles, road dust and long-range secondary sulphate and marine aerosol (Ancelet et al., 2012). The contribution of wood burning to PM10 measured at Masterton East was approximately 20% higher than found at Masterton West. The study authors concluded that PM from wood burning was transported towards Masterton East by cold air draining south eastward along the downslope contours of the river plain

The NESAQ requires PM10 to be monitored in the area within an airshed where air quality is poorest. Therefore, based on the GNS findings, Greater Wellington established a second monitoring site (Masterton East) in 2013. From 2016 to 2022 PM10 and PM2.5 were measured by the same instrument type (Thermo Scientific 5014i) at Masterton West and Masterton East allowing measurements to be directly compared.

The winter diurnal profile for hourly average PM2.5 at both sites had a small morning peak and larger evening peak (Figure 7.1). The morning PM peak is attributed to people relighting or restoking their fires as found by the 2010 GNS winter study (Ancelet et al., 2012).

Hourly average concentrations measured at the two sites converged from 2019 to 2020. The reason for the substantial reduction in difference between PM2.5 winter averages (April to September) measured at both stations reduced from 2018 to 2020 is not known (Figure 7.2).

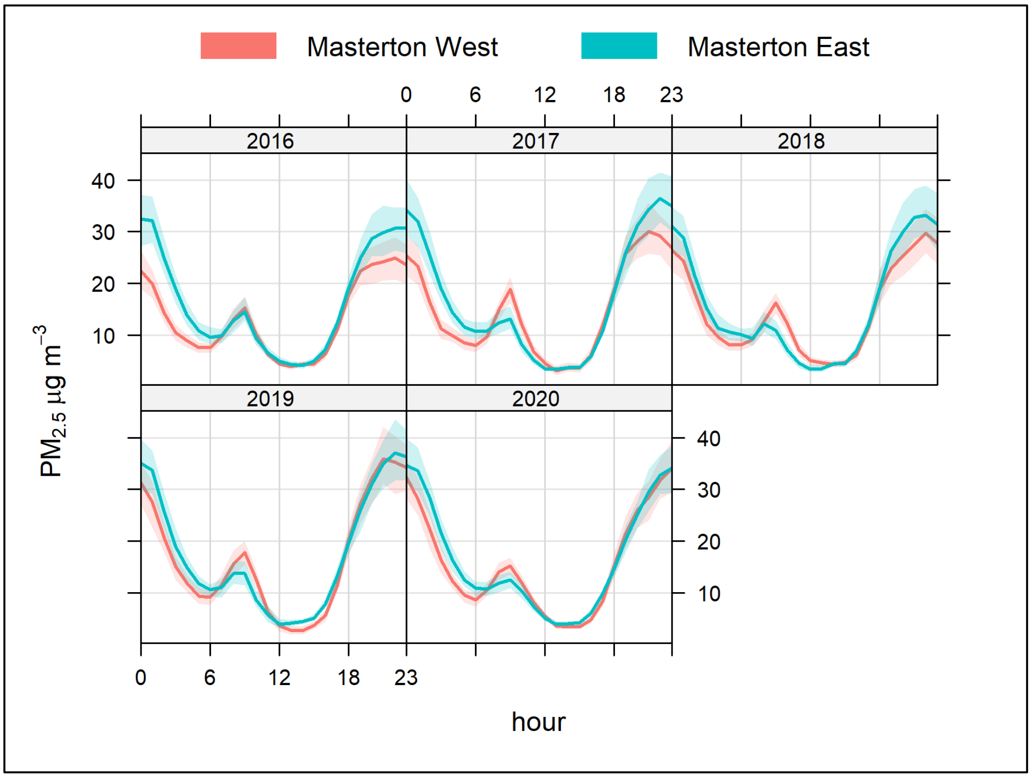

Figure 7.1: PM2.5 diurnal profile 1-hour averages for April to September 2016 to 2020. The shading shows 95% confidence interval in the mean.

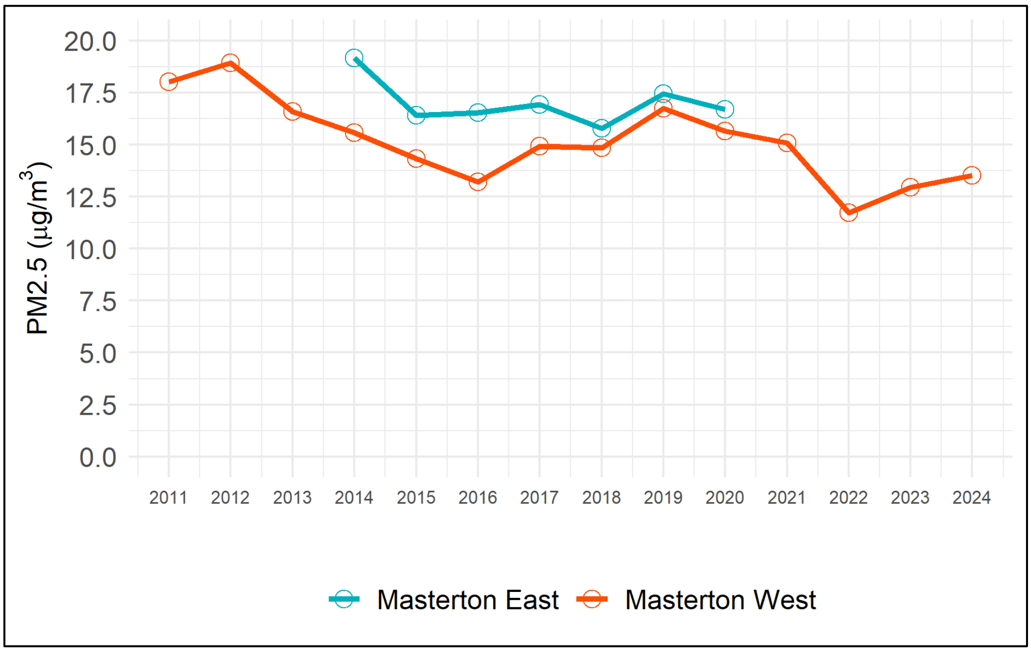

Figure 7.2: PM2.5 winter averages for April to September 2011 to 2024. Note PM2.5 data only available to 2020 due to incomplete data capture in 2021 and 2022.

7.2 Airshed modelling

In 2005 GW gazetted the entire Wairarapa valley as an airshed for monitoring and managing PM10 under the NESAQ (Davy, 2005). This was problematic as the Wairarapa airshed, based on Masterton air quality monitoring, was then deemed to be a 'polluted' airshed under the NESAQ and faced restrictions on the granting of discharge to air consents in areas with complying air quality.

In 2013, GW commissioned Golder Associates to determine which parts of the Wairarapa valley airshed were likely to fail to meet the PM10 NESAQ daily standard so that the airshed boundary could be changed. Golder Associates (2014) carried out air dispersion modelling of PM10 emissions from the Masterton urban, Carterton urban and the Waingawa industrial area. The air dispersion modelling was based on the 2013 Masterton domestic emissions inventory (Sridhar & Wickham, 2013), a five-year meteorological model and estimations of natural source contributions (eg, sea salt), soils and motor vehicles from previous source apportionment studies (Ancelet et al., 2012).

The results of the airshed modelling presented in Figure 7.3 show predicted contours for PM10 concentrations of 50 µg/m^3 as a 24-hour average (2nd highest per year over a five-year period) under worst-case emissions (ie, all wood burners in Masterton being used 24-hours per day. The air dispersion modelled levels of PM10 were closer to reality, ie, they matched the monitoring results at Masterton West, under the worst-case emissions scenario. It was suggested by Golder Associates (2014) that this could be due to additional emissions in the airshed after 10pm from smouldering that can occur when wood burners are 'dampened' down for the night with reduced oxygen flow.

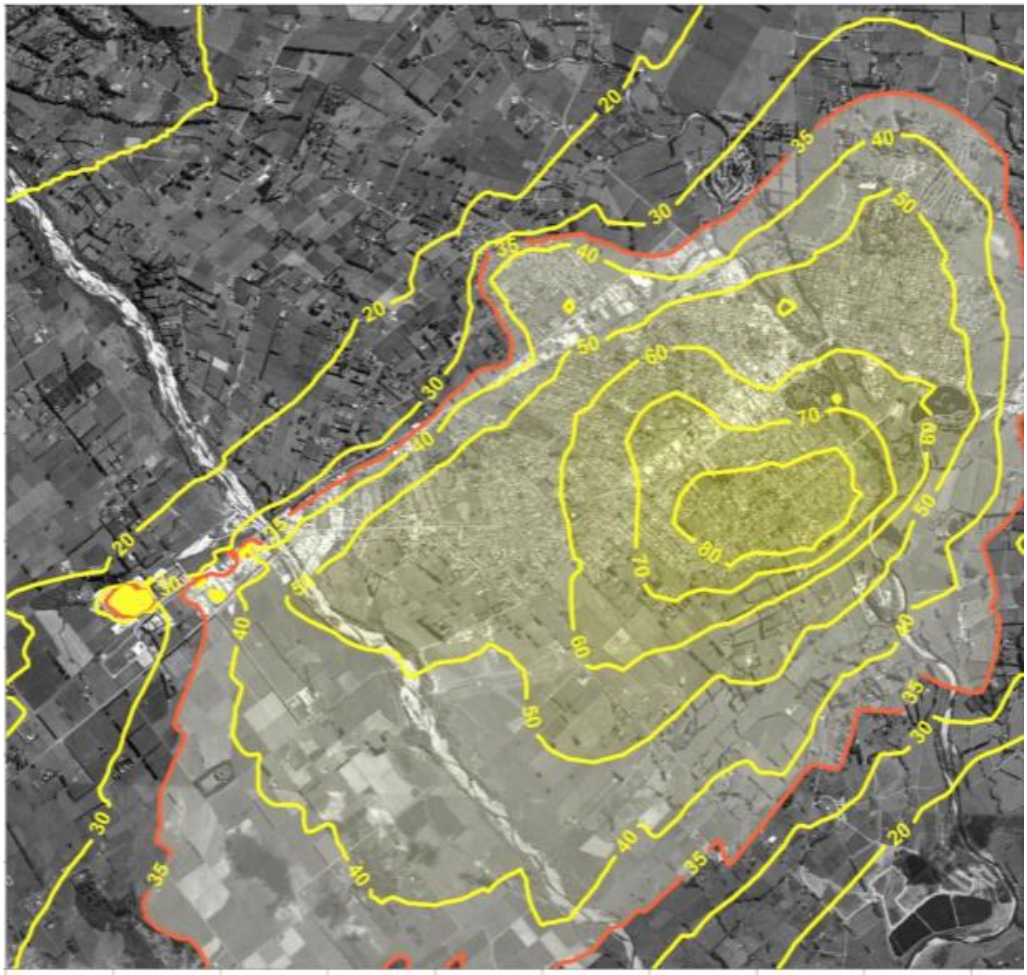

Figure 7.3: Modelled second-highest 24-hour PM10 concentration around Masterton and the Waingawa industrial area from all sources. The 50 µg/m^3 contour represents the predicted area of NESAQ non-compliance. The red line (35 µg/m^3 contour) indicates possible extent of non-compliance with WHO 2006 PM2.5 guideline (Golder Associates 2014).

The industrial area outside of Masterton (Waingawa) was predicted to comply, apart from a small area within one of the industrial site boundaries which is exempt from the provisions of the NESAQ. Masterton domestic emissions contributed to a significant background level of PM10 in the Waingawa area and to a lessor but not insignificant contribution to Carterton. A simpler box modelling technique undertaken for the remaining urban areas: Greytown, Featherston and Martinborough indicated that these areas were likely to comply with the NESAQ.

The boundary defined by the Wairarapa Combined District Plan was used to define the Masterton urban airshed because it covers the residential area of Masterton where domestic fire emissions arise, and it is the area where most people live and work, and will be exposed to air pollution, including at levels that fail to meet the NESAQ.

7.3 Mobile monitoring

In winter 2018, a short mobile air quality monitoring campaign was carried out with support from MDC. A vehicle fitted with a “SmokeTrak” air quality monitoring device linked to a geolocation system was driven through Masterton on eight evenings from approximately 17:00 to 23:00 (Figure 7.4). The purpose of the monitoring was to explore how levels of evening wood smoke varied across Masterton, and to see if there were ‘hot spot’ areas of high PM2.5 that could potentially receive targeted assistance for reducing emissions (Mitchell, 2018). It was hypothesised that high emitting wood burners could have a disproportionate impact on air quality (McGreevy & Barnes, 2016). High emitting burners can lead to very localised degradation of air quality from one or a small number of wood burners (Brain et al., 2022).

Figure 7.4: SmokeTrak (Kenelec Scientific) measuring system using a TSI DustTrak Aerosol Monitor (model 8533) with an in-line PM2.5 filter housed in a travel case together with a GPS sensor, a modem and a tablet for displaying measured PM2.5 concentrations

The mobile monitoring found PM2.5 from wood smoke varied considerably depending on the time of day and location, with the highest concentrations detected late evening (after 10 pm) in the lower-lying areas in Masterton East, Lansdowne and Solway South. The lowest concentrations were found outside the main urban area and at higher elevations in Lansdowne.

Figure 7.5 shows the PM2.5 measurements (5-second resolution). The study was not designed to identify specific high emitting chimneys as this requires on-foot repeated observations of individual chimneys or a much larger sample size of mobile measurements.

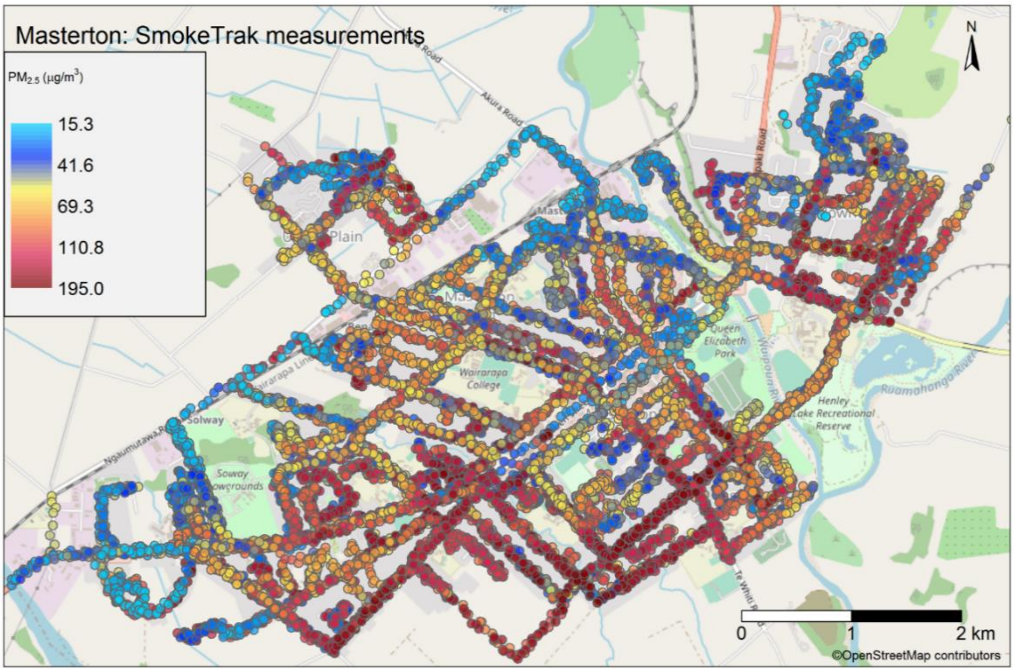

Figure 7.5: SmokeTrak-adjusted PM2.5 µg/m³ (5-second averages) measured in Masterton over eight evenings in June and July 2018 (top 5% of measurements removed), n=17,839. Points are 'jittered' to minimise overlapping data points (Mitchell, 2018).

A limitation of mobile monitoring is that a very large number of repeated observations by location and time of day are required to build a representative picture of how PM2.5 levels vary across an area, especially under different weather conditions. This is because emissions and PM2.5 concentrations vary by time as well as spatially, ie, concentrations may be high because of the time of day (coinciding with peak emission period) not necessarily associated with the location. For this reason, a fixed dense network of sensors operating at the same time is more accurate for characterising spatial variability than mobile surveillance methods.

7.4 Fixed monitoring using dense sensor network

"Community Observation Networks for Air" (CONA) was a NIWA research initiative started in 2015. The research aimed to explore how air quality information from dense networks of low-cost air particle sensors could be used to increase public understanding and engagement with air quality issues and encourage actions to reduce emissions (Longley, 2020a). Other research applications include improved understanding of representativeness of reference monitoring sites, identifying areas with higher health risk from poor air quality, informing options for targeted mitigation, and providing before and after evaluation of air quality interventions[12].

[12] https://niwa.co.nz/atmosphere/community-air-air-quality-issues-nz-towns/air-quality-monitoring-low-cost-sensors

Greater Wellington commissioned NIWA to design a monitoring campaign for winter 2020 using a network of optical PM2.5 sensors (ODINs) fixed to lamp posts (Figure 7.6). The ODIN (Outdoor Dust Information Node) was a low-cost sensor package developed in-house by NIWA to research impacts of domestic heating (and other sources) on particle air pollution.

ODINS were installed at 25 locations across the Masterton urban and peri-urban area to cover a regularly spaced grid as far as possible. Siting constraints (available street pole and clear sky angle for solar power) meant that desired locations could not always be met. Ten-minute data was telemetered from each ODIN to NIWA's IoT cloud server using the 2G mobile phone network. During the campaign there were issues with instrument telemetry due to either poor signal from the mobile phone network or lack of sunlight to power the ODIN's battery. However, 14 sites achieved at least 75% data capture during the monitoring period which is a common benchmark for a representative air quality sample.

ODIN sensors are classed as a low-cost 'indicative' monitoring method and produce data that cannot be directly compared to the regulatory-grade monitors at Greater Wellington monitoring stations. Their main purpose is to show spatiotemporal pattern in solid-fuel combustion particles during the monitoring period (16 June to 5 September 2020).

Figure 7.6: Solar powered NIWA ODIN PM2.5 monitor containing a PMS3003 dust sensor mounted on street pole in Masterton, winter 2020

Figure 7.7 shows the location of ODIN sensors during winter 2020 and the winter average PM2.5 measured at each site. NIWA used the monitoring results to create daily animations of how woodsmoke moves across the airshed which were available from their website during the monitoring period and were used by Masterton District Council and Greater Wellington for public engagement.

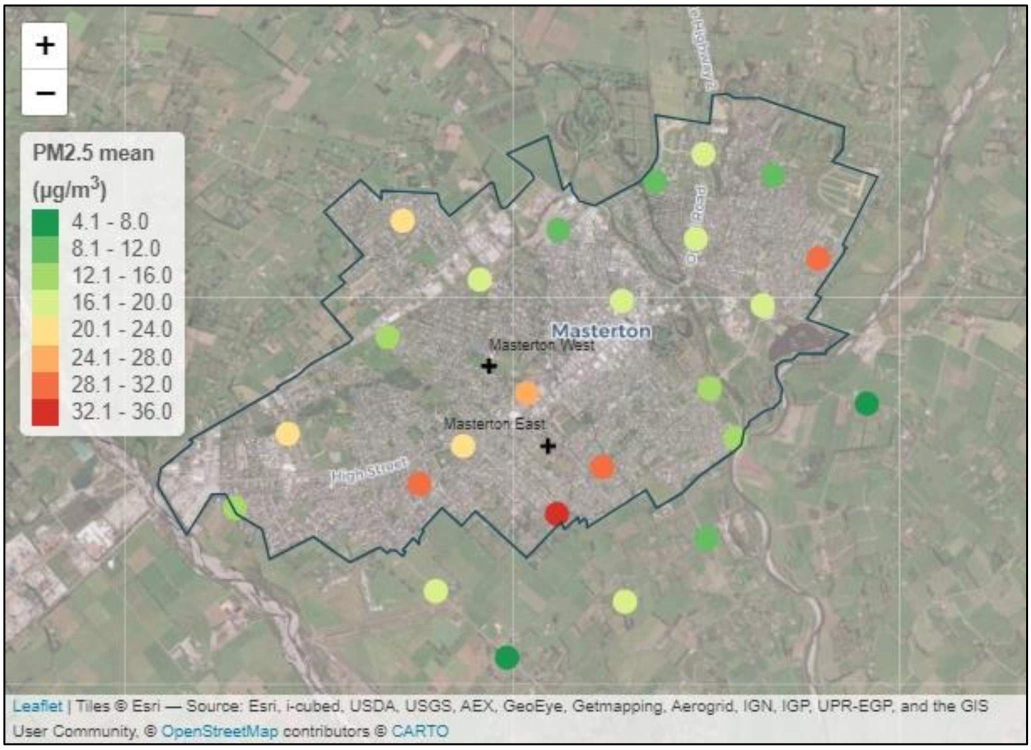

Figure 7.7: Average winter PM2.5 (µg/m³) measured by NIWA ODIN sensors from June to September 2020 (dataset provided by NIWA). GW monitoring sites (+ symbol) and the Masterton Urban airshed boundary (dark blue line).

The PM2.5 winter average for all ODIN sites within the airshed boundary, with at least 75% data capture (n=12) was 19.6 ug/m3. PM2.5 measured at each of these 12 sites ranged from 50% to 170% of the airshed mean. The PM2.5 winter average at the peak ODIN site was just over three times that observed at the site with the lowest concentrations (inside the airshed).

This finding is consistent with NIWA research in other New Zealand airsheds (Longley et al., 2024).

NIWA concluded that there was substantial spatial variation in PM2.5 with reasonably consistent low and high air pollution spots. They also noted there was small but subtle systematic variation in the spatial pattern in that concentrations along the northern side of town peak and fall earlier than elsewhere. This effect is seen in most airsheds studied elsewhere in New Zealand and is almost certainly due to down-slope advection from the north and accumulation in the south. The transect map (Figure 7.8) shows the downward slope across the Masterton urban area. On calm evenings cold denser air drains downslope transporting emissions, so they accumulate in the lower-lying areas.

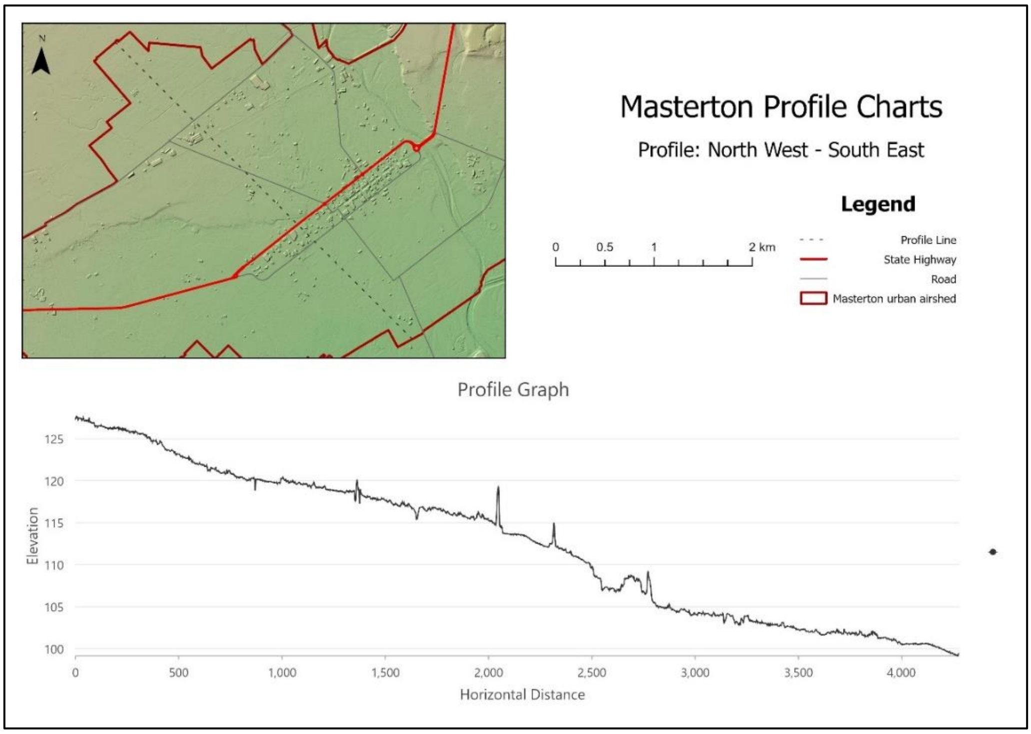

Figure 7.8: Digital elevation map (1m resolution) centred on the Masterton urban area. Darker green areas show lower elevation. The elevation profile Northwest to Southeast shows the downward slope. Source: Greater Wellington GIS Team.

7.5 Conclusion

- Masterton West and Masterton East monitoring locations likely represent the 'airshed average' but there are unmonitored locations which would regularly experience higher winter PM2.5 concentrations, particularly during the late evening period (21:00 to 00) during low wind, clear sky conditions.

- The observed spatial pattern of PM2.5 levels was consistent with wood smoke emissions moving across the airshed from higher elevation to lower elevation areas shown on the digital elevation map. Therefore, emissions reductions are needed across the whole airshed not just in areas observed to have high winter PM2.5 concentrations.

- If the spatial variability observed in the NIWA ODIN study is relatively stable from year to year, then it may be possible to predict winter PM2.5 concentrations in unmonitored locations (eg, at the peak ODIN location on the southeast side of the urban area) based on Masterton West measurements, assuming a constant offset. However, this assumption may not hold given that the difference in PM concentrations between Masterton East and Masterton West has varied over time.

- An option for future monitoring to evaluate any air quality management initiatives in Masterton would be to deploy new generation commercially available indicative optical sensors in targeted areas, including co-location at Masterton West. This would allow any changes in air quality across the airshed to be evaluated.

- Depending on the outcome of the MfE NESAQ review, the airshed boundary for a PM2.5 standard may differ from the current PM10 airshed.

8. Health impacts of long-term exposure to PM2.5 and NO_2

Air pollution is a complex mixture of gases and particles. Both short-term exposure (hours to days) and long-term exposure (years to lifetime) to air pollution are associated with a wide range of adverse health impacts, including reduced life expectancy and a range of cardiovascular and respiratory diseases. Generally, the health impacts from long-term exposure are greater than those arising from short-term exposure (WHO, 2021). The monitored air pollutants most strongly associated with health impacts in New Zealand are PM2.5 and nitrogen dioxide (NO_2).

8.1 National air pollution health impacts model

The health impacts and social costs of air pollution were estimated by the Health and Air Pollution in New Zealand (HAPINZ) model (Kuschel et al., 2022).

HAPINZ estimated health impacts by census area unit for a base year of 2016 for:

- Annual average PM2.5 fine particles from wood burning used for home heating, motor vehicles, industrial and wind-blown dust (from human activities such as construction and road dust); and
- Annual nitrogen dioxide (NO_2) from vehicle exhaust emissions, particularly diesel.

The model inputs included:

- National health datasets for mortality rates and hospital admissions from Ministry of Health and normally resident population data from Stats NZ; and
- Results of a cohort study that established New Zealand specific air pollution exposure-response functions for health endpoints (Hales et al., 2021). These results were adjusted for temperature, age, sex, ethnicity, income, education and smoking.

The health endpoints assessed were number of cases of:

- Premature death (mortality) for adults aged over 30 years;
- Hospital admissions for cardiovascular disease and for respiratory disease (all ages):
- Hospital admissions for asthma for children ages up to 18 years (a subset of hospital admissions for respiratory disease); and
- Restricted activity days (all ages).

The HAPINZ authors concluded that although New Zealand's air quality is relatively good compared to some other countries, air pollution has significant health impacts and costs. This finding is consistent with the revised World Health Organization (WHO) guidelines which recognise that there is no 'safe' level for PM2.5 and NO_2.

8.2 Health impacts by airshed

HAPINZ health impacts were disaggregated by regional council, territorial authority, airshed and health district (formerly known as District Health Board). For this report, health impacts are shown by airshed as this is the spatial area of interest. The results would be different if territorial authority boundaries were used instead.

Estimated health impacts and their associated social costs vary between airsheds, depending on population size, underlying health status and levels of air pollution where people live. Impacts are shown as a rate of cases per 100,000 people so that airsheds with different population sizes can be compared.

Of all the region's airsheds, Masterton has the highest rate (per 100,000 people) of hospitalisations and reduced life expectancy attributable to PM2.5 from home fires (Figure 8.1) as well as the highest rates due to exposure to NO_2 from motor vehicles (Figure 8.2). However, because Masterton has a smaller population than most of the other airsheds (except for Karori and Wainuiomata) this equates to fewer predicted hospitalisation and premature death cases per year than more highly populated airsheds, such as Wellington City.

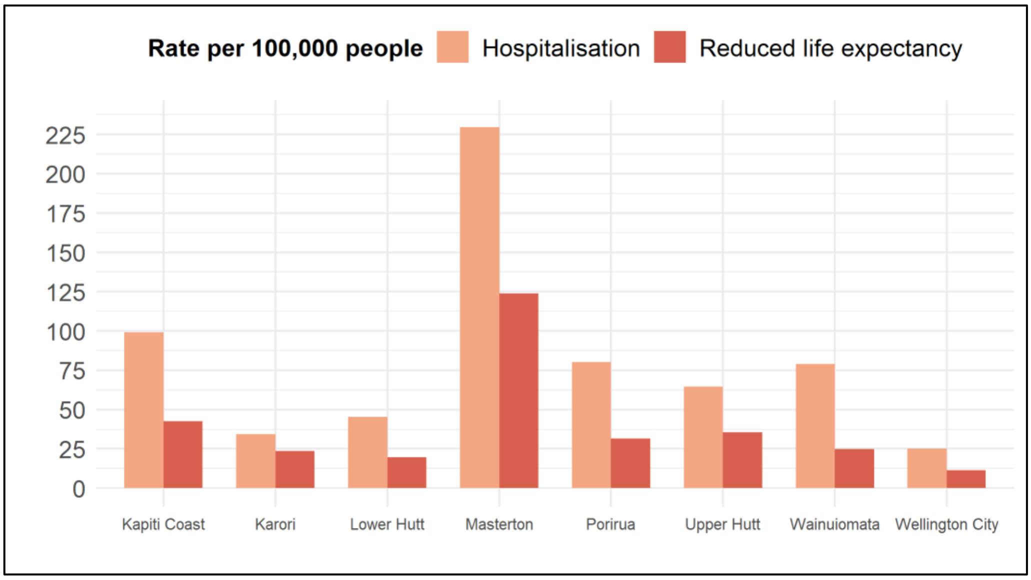

Figure 8.1: Airshed health impacts of PM2.5 air pollution from home fires per 100,000 people. Premature death for adults aged over 30 years and hospitalisations for cardiovascular and respiratory disease. Source: HAPINZ (2022).

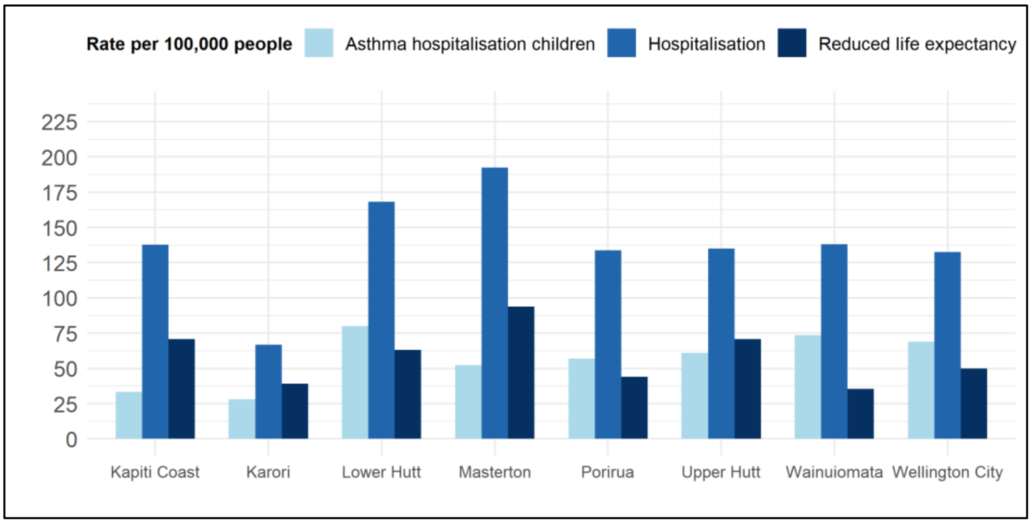

Figure 8.2: Health impacts of NO2 from motor vehicles per 100,000 people by airshed. Reduced life expectancy rate for adults aged over 30 years, hospitalisations for cardiovascular and respiratory disease and asthma hospitalisation rate for children aged 0 to 18 years (subset of respiratory hospitalisations). Source: HAPINZ (2022).

Restricted activity days are days that occur when exposure to air pollution causes symptoms which prevent people being able to work, attend school or undertake their usual activities. Figure 8.3 shows Masterton has a high rate of restricted activity days from exposure to PM2.5 from home heating compared to the other airsheds.

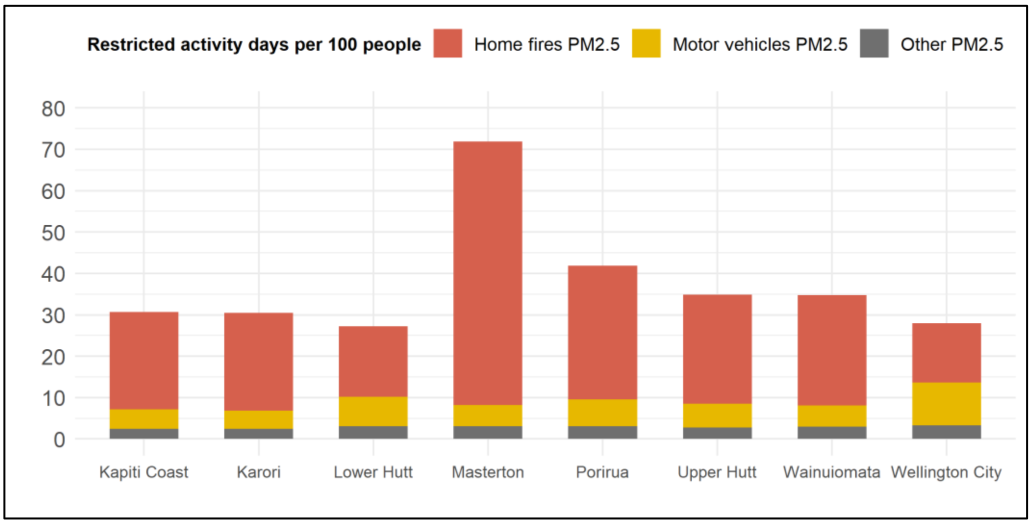

Figure 8.3: Airshed restricted activity days per 100 people due to PM2.5 by source. Source: HAPINZ (2022).

8.3 Social costs by airshed

Social costs are the total costs to society of health effects associated with air pollution, not just the direct medical costs but also the wider costs due to loss of output (income and time off work or school for those who need to care for affected family and friends) and recovery.

Social costs of air pollution were calculated in HAPINZ as follows:

Social costs = Health Effects (cases) x cost per Case

Figure 8.4 shows the total annual social costs attributed to health effects from air pollution sources in different airsheds. Masterton has the highest per capita social costs for all pollutant sources and health endpoints (Table 8.1). In Masterton, 60% of the social costs were due to exposure to PM2.5 and 40% from traffic-related NO_2.

Table 8.1: Airshed social costs ($) per person based on 2016 population data. Source: HAPINZ (2022).

Airshed	All air pollution	PM2.5	NO2
Masterton	6,748	4,065	2,683
Kapiti Coast	4,028	1,792	2,235
Upper Hutt	3,364	1,358	2,006
Lower Hutt	2,652	895	1,757
Porirua	2,290	1,116	1,174
Karori	1,939	861	1,078
Wellington City	1,822	575	1,247
Wainuiomata	1,783	863	920

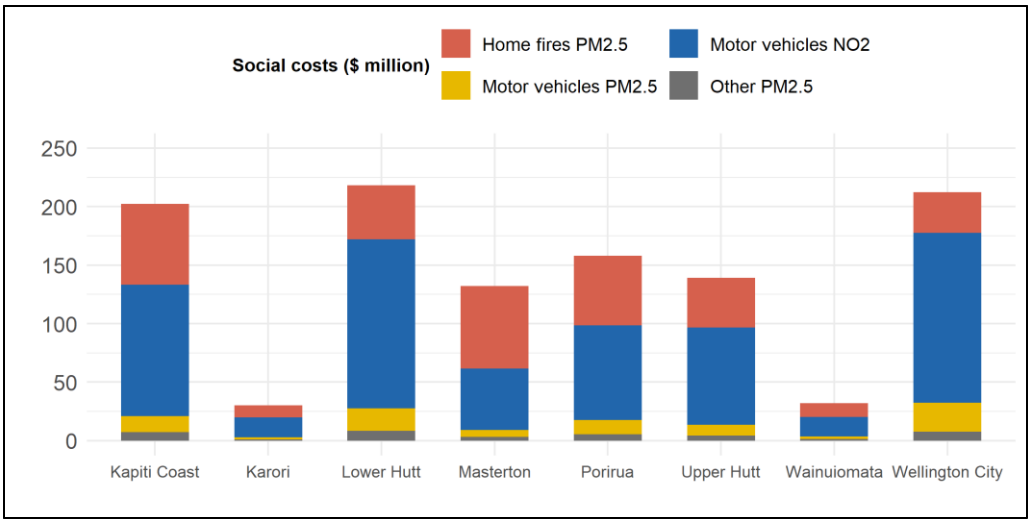

Figure 8.4: Airshed total annual social costs ($ million) of human-made air pollution by pollutant source for all health endpoints. Source: HAPINZ (2022).

8.4 Social inequity in health impacts

The outputs from the HAPINZ model analysed by the New Zealand Deprivation Index (NZDep)[13] show there was social inequity of exposure to air pollution and associated health impacts (Wickham et al., 2023). Across New Zealand health impacts from air pollution were higher on average in more deprived areas (decile 10) compared with the least deprived areas (decile 1).

Wickham et al., 2023 state that increases in health impacts from air pollution in more deprived areas can be explained by two factors:

(i) underlying structural inequities, specifically higher base health incidence rates of all health outcomes studied (mortality, respiratory & cardiovascular hospitalisations, childhood asthma prevalence) in more deprived areas; and

(ii) higher levels of exposure to air pollution in more deprived areas.

This means that air pollution health impacts will be worse in more deprived areas due to a combination of higher pollutant concentration in these areas and the higher base health incidence rates.

NZDep were available by Territorial Authority boundary but not by airsheds. Therefore, health impacts (rate of premature mortality per 100,000 people) attributed to PM2.5 from home fires and to NO_2 from motor vehicles by territorial authority were compared to population weighted deprivation index available from MBIE Regional Economic Activity Web Tool[14].

Table 8.2 shows Masterton District had both the highest population-weighted deprivation index and rate of reduced life expectancy from air pollution exposure. This is broadly consistent with the social inequity findings of Wickham et al., 2023 noting that there are other factors such as differences between Territorial Authorities in the use of solid fuels for heating, for example Lower Hutt has a low proportion of households that use wood for heating, and therefore less exposure to PM2.5 from domestic heating.

[13] The NZDep is an area-based measure of socioeconomic deprivation in Aotearoa New Zealand. It measures the level of deprivation for people in each small area based on nine Census variables. NZDep is displayed as deciles, which each decile containing about 10% of small areas in New Zealand. Decile 1 represents areas with the least deprived scores and Decile 10 represents areas with the most deprived scores. (Atkinson et al, 2024)

[14] Deprivation index indicator based on NZDep2018 meshblock-level available from https://www.mbie.govt.nz/business-and-employment/economic-development/regional-economic-development/activity-tools/web-tool

Table 8.2: Territorial authority population weighted deprivation index and rate of premature mortality (per 100,000 people) due to exposure to NO_2 from motor vehicles and PM2.5 from home fires

Territorial Authority	Population weighted DEP	Reduced life expectancy per 100,000 people	
		NO2 motor vehicles	PM2.5 home fires
Masterton District	6.4	74.7	95.2
Lower Hutt City	5.7	56.5	20.3
Porirua City	5.6	41.2	31.3
Upper Hutt City	5.1	67.7	33.7
Kapiti Coast District	5	68.2	40.4
Carterton District	4.9	45.1	26.0
South Wairarapa District	4.6	42.0	24.7
Wellington City	4	45.4	15.3

Masterton's relatively high health impact rate compared to the other airsheds is due to high PM2.5 levels from home fires combined with higher index of deprivation means that residents on average may be more vulnerable to the effects of air pollution due to poorer underlying health status as base health incidence case rates are higher for lower decile areas compared to higher decile areas.

8.5 Conclusion

- The estimated health impacts and costs of air pollution are not distributed equally across the region. Masterton airshed experiences the highest social costs per person which is correlated with a relatively high population-weighted deprivation index which increases vulnerability to air pollution health impacts. Masterton also experiences the highest levels of monitored PM2.5 in the region.

9. Wood burner behaviour change

9.1 Airshed wood burner emissions

PM emissions from home heating depend on type, age and design of solid fuel appliances (wood, coal, pellet, multi-fuel, open fire), fuel type and quality, as well as user behaviour – how the fire is lit and maintained (MfE, 2020).

Airsheds that have achieved significant improvements in air quality, such as Rotorua (Weir, 2023), Christchurch (Pearce & Scott, 2019), and Nelson (NCC, 2024) used a mix of incentives (loans and grants) and rules to switch residents to zero emissions heating (heat pumps) or solid fuel burners that produce less smoke (pellet or ultra-low emissions burners (ULEBs[15]). One council used a district council bylaw to remove non-complying solid fuel burners at the time the house was sold. Community engagement and awareness was important for successfully implementing incentives, regulations and behaviour change to improve air quality.

The relative effectiveness of interventions aiming to change user behaviour to reduce emissions and improve air quality across an airshed has not been systematically evaluated in New Zealand and would be difficult to do given the range of variables in play.

9.2 User behaviour

In-home testing of wood burners being operated under real-life conditions in New Zealand found that PM emissions from NESAQ compliant burners are on average around three times higher than their regulatory limit of 1.5g/kg wood burnt (Wilton et al., 2015).

Variability in operator behaviour and fuel quality has a profound impact on wood burner emissions and efficiency (Wilton, 2012, Coulson et al, 2015). ULEBs performed much better in real-life testing with average emission rates of 1.6 to 2.1 g/kg (Applied Research Services, 2023).

The following operational factors can increase emissions:

- Fuel quality – high moisture content (wet wood)
- Dampening down the appliance by restricting airflow (oxygen availability for combustion.
- Size of wood logs - unsplit wood generates more emissions than split wood.
- Defective or poorly maintained wood burner.

[15] ULEBs under real-life operating conditions must meet an emissions and efficiency standard of 38 milligrams per megajoule of useful energy and have a thermal efficiency of 65% or greater. This is based on testing being carried out in accordance with the Canterbury Method (CM1) or approved alternative. www.ecan.gov.nz

Given the impact of fuel quality and operator behaviour on wood burner emissions, identifying high emitting wood burners and providing targeted education to these households has been identified as a strategy for reducing emissions (McGreevy & Barnes, 2016). However, the efficacy of this strategy has not been confirmed, e.g., monitoring evaluation of an intensive wood burner education campaign in Tasmania did not find evidence of improvements in air quality (Robinson, 2016). Air quality management options for reducing PM in airsheds, such as that carried out in Richmond, Tasman, include an option for implementing a behaviour change programme that achieves a 10% reduction in emissions (Wilton & Zwar-Reza, 2021).

9.3 Household wood burner behaviour change project

In 2013, Environment Canterbury received funding from the Ministry for the Environment's Community Environment Fund to develop a household wood burner behaviour-change tool kit and intervention. The project aimed to understand household wood burning behaviour and identify and test the behaviour change interventions required for people to reduce smoke particulate emissions from home fires. Part of the funding was to pilot the programme in other regions.

Research for the project was carried out by ChangeHub (a social marketing company) and formed the basis for Environment Canterbury's "Warmer Cheaper" [16] campaign which aimed to change people's fire lighting practice so they could burn 'smoke-free' with no visible emissions apart from a short period when the fire was first lit. Environment Canterbury's behaviour change programme included a Trusted Good Wood Merchants scheme, public workshops, in-home demonstrations on better burning techniques, free kindling promotions, and providing wood to low income households. Environment Canterbury's wood burning behaviour change programme was backed up by incentives, regulation and enforcement.

Smoke-free fires pilot programme in Masterton – focus group insights

In September 2015, Masterton District Council took part in the Ministry for the Environment funded pilot programme investigating how to best to assist a sample of households identified as having high smoke producing wood burners reduce their emissions by adopting cleaner burning techniques.

The pilot involved using 'spotters' to identify 20 high smoke chimneys which were then sent a letter about smoke-free burning techniques and invited to participate in a focus group run by the consultancy ChangeHub.

Key insights from the focus group research (8 participants) were:

- Masterton is a country town, and we know how to light fires;

[16] https://www.warmercheaper.co.nz/

- Offers of help to improve fire-lighting techniques could be seen as condescending or patronising;
- Smoke is not an issue in Masterton;
- Traffic and industry need to be dealt with first; and
- Positive messaging works best, such as, we want people to enjoy their fires but in the right way.

Proposal for Masterton air quality behaviour change project

In 2017, Greater Wellington and Masterton District Council commissioned ChangeHub to provide an outline and indicative costings for a multi-year behaviour change strategy and action plan for reducing wood burning emissions in Masterton. The strategy was informed by the “Four approaches to change” model (Figure 9.1), insights from the Masterton focus group research, and work done by Environment Canterbury. ChangeHub noted that successful behavioural interventions are informed by and have solutions from across the four approaches.

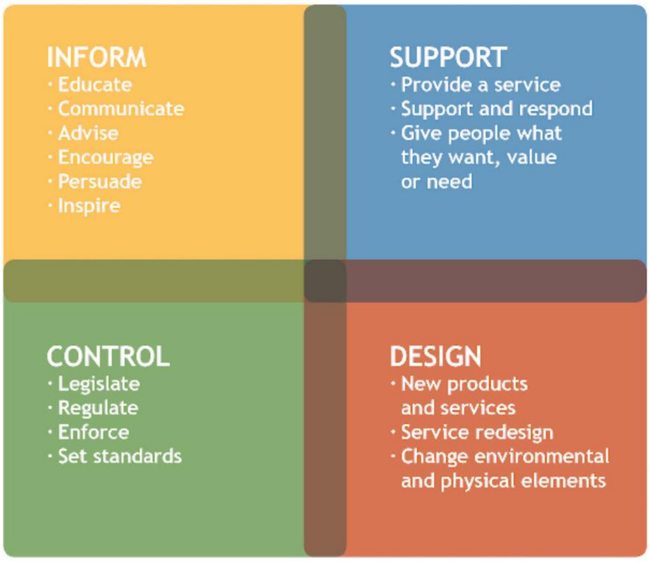

Figure 9.1: Four approaches to change model. Source: ChangeHub

In 2017, Greater Wellington’s environmental science unit with support from Masterton District Council sought Long-Term Plan (LTP) funding for a behaviour change programme to improve Masterton’s air quality as directed by Method 5 of the then Proposed Natural Resources Plan. The funding proposal, informed by ChangeHub’s proposal, focused on the Inform and Support elements and did not include any regulation or enforcement provisions. LTP budget for the programme was approved (starting 2028/29) for the purpose of meeting national and regional air quality requirements.

In 2018 GW implemented a region-wide digital 'better burning campaign[17]' which included information about smoke free burning techniques and how to identify treated timber to avoid burning it.

9.4 Masterton Home heating survey insights

A home heating emissions survey commissioned by Greater Wellington in 2013 also collected information on people's burning behaviour and attitudes and perceptions about air quality (Sridhar & Wickham, 2013). The survey data is now over 10 years old so may not fully represent current behaviour or attitudes.

Reasons for using wood burner

Half of the solid fuel users surveyed cited financial reasons for burning wood, with 27% saying that the fuel supply was cheap and 15% saying that it was free. A further 8% also said that using wood saves on electricity costs. Just under a quarter of respondents said that they used a burner because it was already there when they moved in. Eighteen per cent reported that enjoyment and ambience was a reason for using wood. Figure 9.2 shows respondents reasons for using wood for heating. Respondents were able to choose multiple reasons therefore percentages sum to more than 100%.

A qualitative study in 2003, also found that perceived affordability (electricity vs wood) was the main factor in Masterton for people's heating choices (Dunn, 2003).

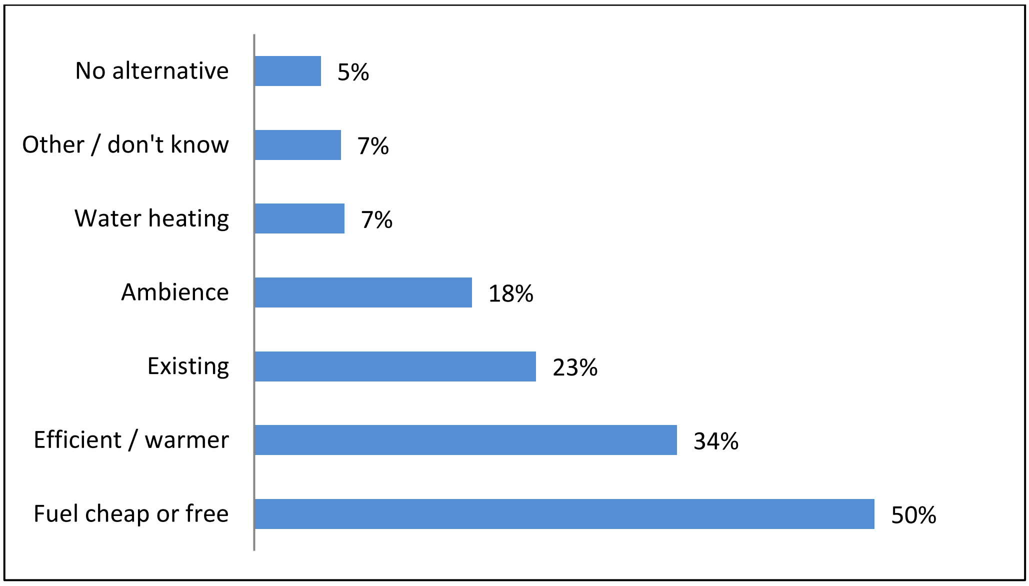

Figure 9.2: Reported reasons for using wood for heating (n = 369 respondents, margin of error 4.8%)

[17] https://www.gw.govt.nz/environment/air-quality/cleaner-home-heating/learn-how-to-burn-smoke-free/

Circumstances for changing heating methods

All survey respondents were asked about what circumstances could lead to them changing to another type of home heating but not using wood or coal. Of particular interest are the responses of those who currently use solid fuels. Figure 9.3 shows the reasons provided when respondents were asked under what circumstances they would consider switching from wood or coal to other forms of heating.

- Forty-seven per cent said they would not change under any circumstances.
- Twenty-seven percent would do so if there was financial benefit or assistance, i.e., if there were cheaper heating alternatives available (18%) or financial subsidy (9%).
- Nine per cent said they would change for rules or regulations.
- Only one per cent said that they would consider changing if their burner needed replacing.
- Other reasons were given such as no longer having access to firewood (2.4%) or if gas was available (0.3%).

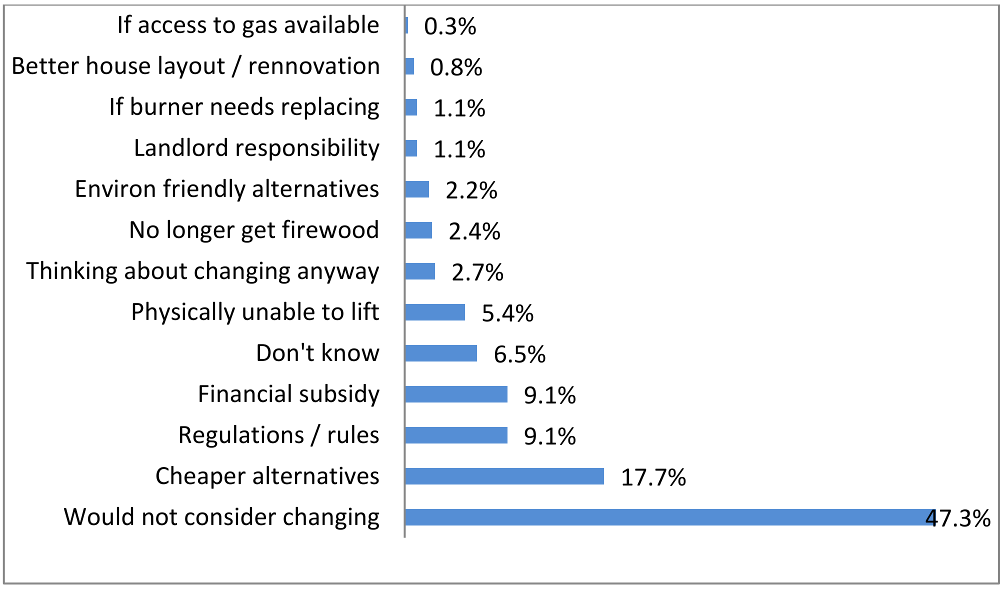

Figure 9.3: Reported circumstances under which existing solid fuel users would consider changing to other forms of heating – not wood or coal (n = 369 respondents, margin of error 4.8%)

Burning practice

- Twenty-seven per cent of households reported using free firewood exclusively, 54% always bought their wood, and 19% of respondents were using a mixture of free and purchased wood.

- With some models of wood burners is it possible to restrict the oxygen flow to effectively 'dampen' down the fire so it burns for longer at a lower temperature. This practice enables the fire to be more easily re-started the following morning by opening the air flow and adding more fuel so the fire can re-kindle. Re-loading a wood burner and then dampening down for an overnight burn result in significantly higher emissions as the fire smoulders and produces more smoke and particulate due to the inefficient combustion conditions. Just under half of all households with a wood burner (44%, margin of error 4.9%) equating to 2221 households (based on 2013 census) reported that they kept their fire burning over night by dampening it down.

- Households also reported burning non-firewood materials (Figure 9.4). Sixteen percent of respondents either *always*, *often* or *sometimes* burnt decking offcuts which release hazardous air pollutants (such as arsenic if CCA-treated).

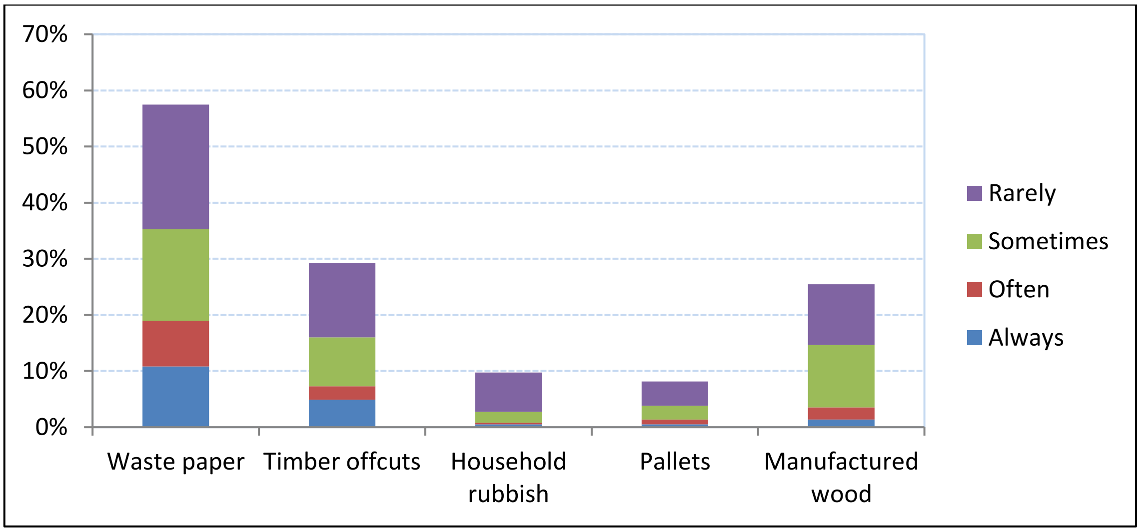

Figure 9.4: Reported frequencies of households burning non-fire wood materials in the home fire by type of material (n = 369 respondents, margin of error 4.8%)

9.5 Conclusion

- Wood burning can be an effective and financially viable method of home heating, especially for those who source their own wood.

- Although operator behaviour and fuel quality have a significant impact on individual wood burner emissions the impact of behaviour change programmes aiming to reduce emissions on airshed air quality has not been systematically evaluated.

- Positive community engagement and involvement is needed to find appropriate ways of reducing wood burner emissions. Experience in New Zealand shows that incentives and regulation have been most effective in reducing emissions and improving air quality.

10. Healthy homes and energy efficiency

New Zealanders on average spend most of their time, almost 70%, indoors (Khajehzadeh & Vale, 2017). People's housing situation is an important determinant of health with cold, damp and poorly ventilated housing associated with poorer health outcomes (Howden-Chapman et al., 2021).

Housing, energy, health and air quality are interconnected as shown by Figure 10.1. Improving households' insulation and converting to clean and efficient heating has co-benefits for both indoor and outdoor air quality and improves health outcomes (Howden-Chapman & Preval, 2014; Fyfe et al., 2022).

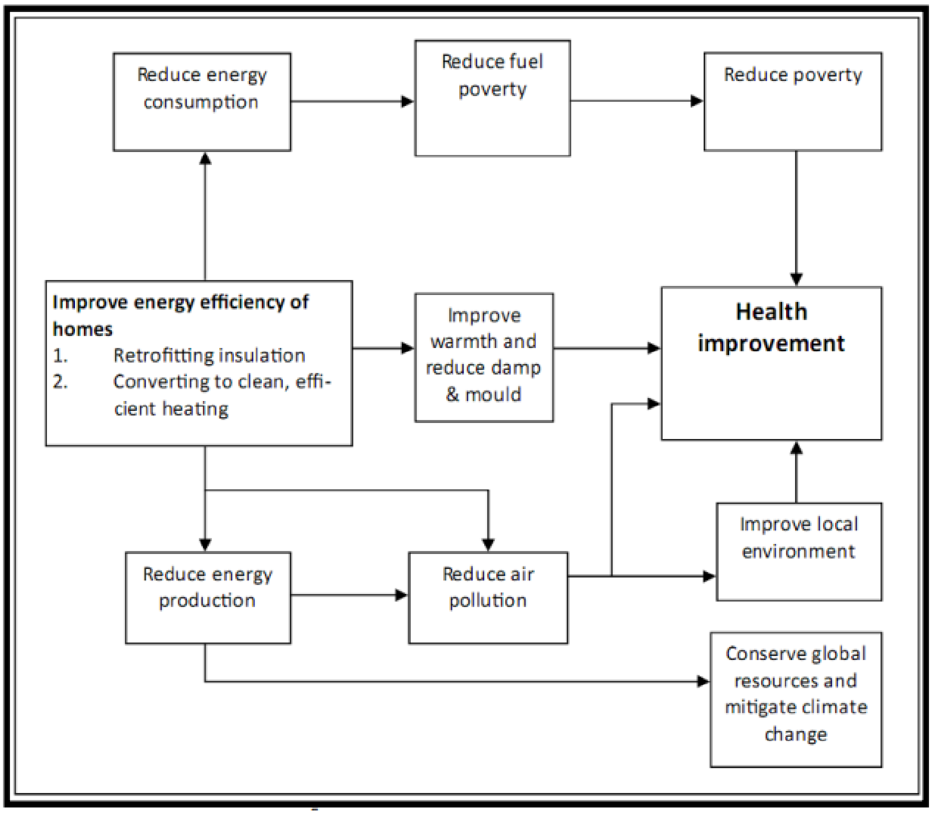

Figure 10.1: Connections between housing, energy, health and air quality. Source: Canterbury District Health Board (2012)

10.1 Insulation and clean heat retrofits

Since 2009, EECA has administered various programmes providing subsidies for retrofitting home insulation and from July 2019, heat pumps, pellet and wood burners.

Figures provided by the Energy Efficiency and Conservation Authority (EECA)[18] show that between 2009 and September 2024, 3,844 houses in Masterton were retrofitted with insulation and for 508 of those households, this also included heat pump installation, and for 13 households solid fuel burners were installed.

In 2010 Greater Wellington started the Warm Greater Wellington (WGW) Programme which allowed ratepayers (with houses built before 2000) to top-up EECA home insulation funding with a loan $3,900 (7% interest) to be paid back over nine years through property rates. In 2015 an option to borrow $5,000 towards upgrading solid fuel burning appliances to a heat pump or NESAQ compliant burner was added to the programme for properties within the Masterton and Wainuiomata airshed boundaries. This provision was to address degraded air quality due to solid fuel burner emissions in these two airsheds. There was no explicit provision for new heat pump installs to replace an existing wood burner, so it is not clear how effective the scheme was in reducing airshed emissions. Data are not available to analyse the breakdown of insulation and clean heat (heats pump and NESAQ compliant wood burner) installs by airshed areas.

The WGW programme was paused in 2021 due to compliance issues with the Consumer Credit Control Finance Act and in 2024 GW decided to discontinue the scheme altogether.

The Wairarapa Healthy Homes (WHH) programme, a community-based initiative, was established in 2004. The programme started as a partnership between EECA and the Wairarapa Community Funders (three Wairarapa Councils, District Health Board, and Trust House Foundation) through the insulation provider Energy Smart (Terra Lana). The scheme focused on providing homeowners in deprivation areas access to insulation and clean heating options. The community funder contributed 10% of the project funds with ECCA funding 90%. Masterton District Council has been contributing to the WHH scheme for the past 16 years. Through the programme, almost 2,800 low-income Wairarapa homes have received 100% subsidised home insulation. In 2021 Greater Wellington contributed $10K towards the Masterton Healthy Homes to support low emission heating options.

Another local initiative is the Home Health Assessment Kit[19] which can be borrowed from the public libraries in Masterton, Carterton and Greytown. This toolkit helps people check their homes for moisture, leaks, draughts and whether their firewood is dry.

[18] Pers comm EECA, via email 22/11/2024

[19] https://www.mstn.govt.nz/community/environment/home-health-assessment-kit

10.2 Indoor air quality

Although people spend most of their time indoors, air quality indoors is an under researched area in New Zealand. Research on a range of residential houses in Wellington region found that combustion particles (eg, from wood smoke) infiltrated readily from outdoors to indoors. Indoor sources of pollutants included wood burners, cooking, gas appliances, and vehicle exhaust from internal garages (BRANZ, 2019a). Ventilation plays a key role in indoor air quality and moisture management in houses (BRANZ, 2019b).

An indoor PM monitoring campaign in Arrowtown found outdoor air pollution had a larger impact on air quality inside the home than from indoor sources and PM2.5 concentration indoors was highly variable (Longley, 2020b). An earlier study demonstrated that wood burner's leak particles inside when they are first lit and then when re-loaded with wood during operation (Longley & Gadd, 2011).

Using a wood burner pollutes indoor as well as outdoor air, so switching to electrical heating improves both indoor and outdoor air quality Replacing un-flued gas heaters reduces indoor exposure to NO_2 (Gillespie-Bennett et al., 2008) and dampness.

10.3 Conclusion

- Outdoor and indoor air quality is affected by housing insulation, energy demand, heating appliances used, ventilation rate and building airtightness.
- Indoor air quality co-benefits could be integrated into healthy housing initiatives where these include switching to low emission heating and tested through indoor as well as outdoor air quality monitoring.

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Appendix A: Statistical modelling

De-weather modelling method

Concentrations of air pollutants are affected by a wide range of factors. Two of the most important factors are emission strength and meteorology. From observational data alone it can be very difficult to know how much of the variation in air quality is due to changes in emissions or the effect of meteorology. Air quality data can be analysed in a way that accounts for changes in meteorology using techniques referred to as meteorological normalisation or de-weathering (Grange & Carslaw 2019). The aim of de-weathering is to remove the impact of meteorology (as much as possible) on the variation in air pollutant concentrations so that underlying trends in emissions can be inferred.

De-weathering was conducted using the *rmweather* R package (version 0.2.6) (R Core Team 2024; Grange et al., 2018; Grange & Carslaw 2019). This method uses predictive random forest models to remove variation in pollutant concentrations due to weather so underlying trends and impacts of any interventions can be explored in a robust way.

The random forest models were used to calculate de-weathered trends to produce a timeseries of air pollutant concentrations under average meteorological conditions. Trends in the de-weathered timeseries were formally tested and evaluated using the Theil–Sen slope estimator and 95% confidence intervals from the R *openair* package (version 2.18-2) (Carslaw & Ropkins 2012).

Modelling and hyperparameters

The *rmweather* tool randomly assigns 20% of observations within the training data as testing data. The other 80% of the training data was used to train the model. The test data was withheld from training procedures and was used to assess model performance.

The number of trees for the random forest models was fixed at 300, the minimal node size was five, and the number of variables split at each node was the default for regression mode: the rounded down square root of the number of independent variables.

PM models

Variables used in the modelling

The explanatory variables (Table A.1) used: air temperature, ventilation coefficient, boundary layer height, relative humidity, wind speed, and rainfall. The temporal predictor variables used were: Unix date (number of seconds since 01-01-1970) as the trend term, Julian day (91 to 273) and month to represent emissions pressure. Training was only conducted on observations which had non-missing wind speed and the pollutant being modelled. Three hundred predictions were used to calculate the meteorologically normalised trend. The normalised trends were aggregated to seasonal resolution (April to September) for presentation in Section 3.

Table A: PM April to September 2008 to 2024

Parameter	24-hour	Source	Range (min, max)	Number observations
PM10 (μg/m^3)	Average	Masterton West	1.7 – 88.8	3069
PM2.5 (μg/m^3)	Average	Masterton West	1.2 – 94.3	2507
Explanatory variables:				
Air temperature 10m (^{o}C)	Average	Masterton West	1.2 – 21.3	3096
Wind speed 10m (m/s)	Average	Masterton West	0.48 – 8.04	3077
Boundary layer height (BLH) m	Average	ERA5 (1-hr)	89.7 – 1,763.0	3111
Ventilation coefficient	Average	Derived from 1-hr BHL x wind speed.	69.6 – 14,019.33	3077
Relative humidity (%)	Average	Masterton West	43.2 – 98.0	3062
Rainfall (mm)	Total	Masterton West	0.0 – 76	3111

Boundary layer height from ERA5 reanalysis atmospheric model was downloaded from the Copernicus Climate Data Store (ERA5 hourly data on single levels from 1940 to present at 30 km x 30 km resolution). Boundary layer height represents the depth (m) of the vertical mixing zone for air next to the Earth's surface. When the boundary layer height is low, higher concentrations of pollutants (emitted from the Earth's surface) can develop than when the boundary layer height is high. The boundary layer height calculation is based on the bulk Richardson number (a measure of the atmospheric conditions).

Ventilation coefficient (VC) is a derived variable VC (m^2/s) = ws (m/s) *BLH (m). It is an indicative variable used to characterise the capacity of atmospheric conditions to disperse air emissions (Sujatha et al., 2016).

PM model performance and results

The random forest models using the explanatory variables in Table A.1 on the training dataset explained 67% of the variance in both 24-hour PM10 and PM2.5.

	No. observations	Prediction error MSE	R^2
PM10	2455	40.5	0.67
PM2.5	2005	38.8	0.67

The model prediction error describes how well the model can predict the 20% of data that was withheld from the training dataset.

	RMSE (root mean square error)	R^2
PM10	6.42	0.66
PM2.5	5.76	0.66

Table A2 shows the relative importance of variables in predicting PM concentrations as outputted by the *rmweather* package. The importance values are defined as the permutation importance differences of prediction error and are unit less. These statistics represent the increase in prediction ability after the inclusion of each explanatory variable and, therefore, give an indication of the importance of each variable for representing the dependent variable (PM in this case) in a random forest model.

Table A2: Relative importance of explanatory variables

Variable importance	**PM10**	**(%)**	**PM2.5**	**(%)**
Wind speed	61.7	33.0%	53.5	33.8%
Air temp	37.3	20.0%	30.7	19.4%
Ventilation	22.7	12.1%	18.9	11.9%
Julian day	20.4	10.9%	23.3	14.7%
Boundary layer height	17.2	9.2%	10.8	6.8%
Month (Apr-Sep)	9.6	5.1%	8.2	5.2%
Rainfall	8.0	4.3%	4.6	2.9%
Relative humidity	7.6	4.0%	5.6	3.5%
Time trend	2.5	1.3%	2.9	1.8%

Partial dependencies

The partial dependence plots from the random forest model for PM2.5 of the four predictor variables that explain most of the variation in 24-hour averages are shown in Figure A1. Wind speed has the largest influence on PM2.5 concentrations, followed by air temperature.

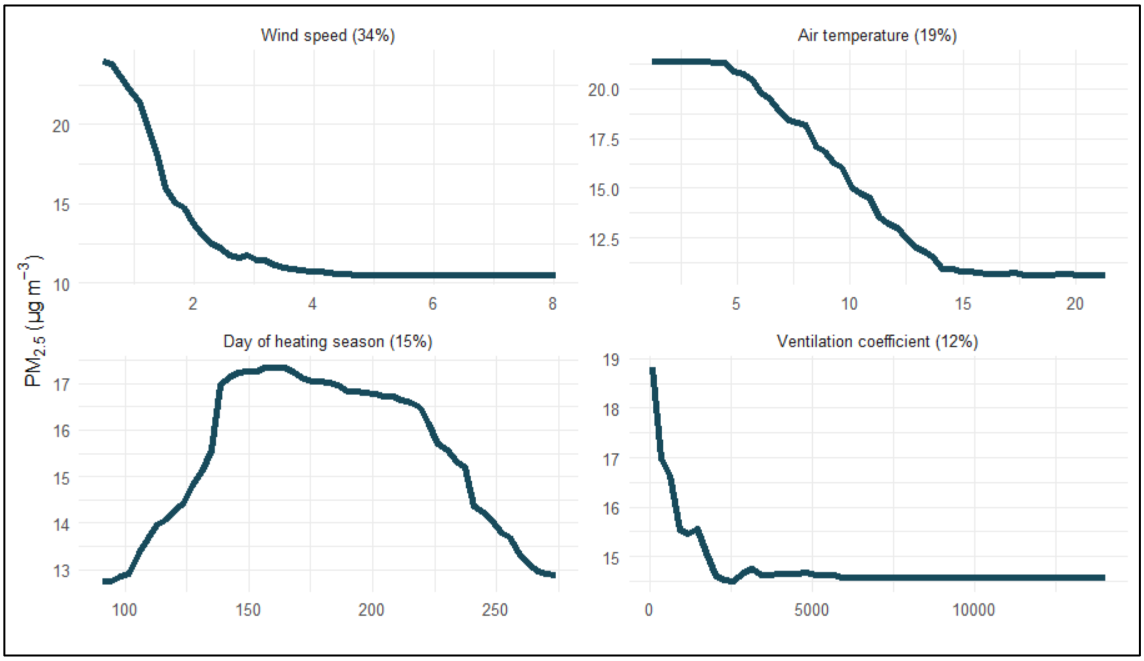
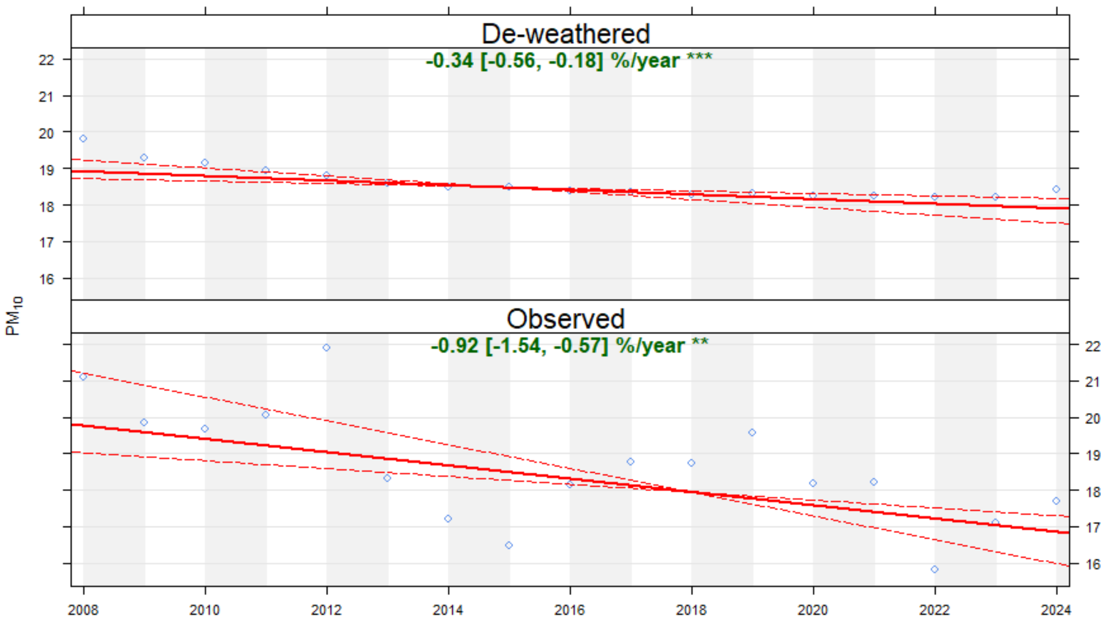


Figure A1: PM2.5 Masterton West partial dependencies of four most influential explanatory variables from the random forest model

PM Theil-Sen trends



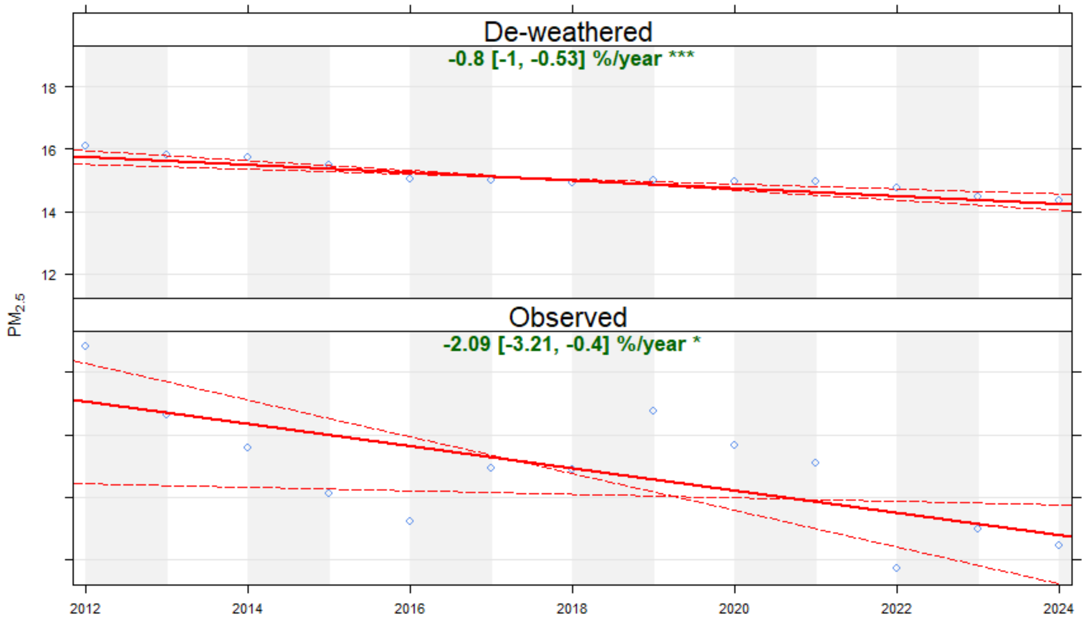


Figure A2: Trends in de-weathered and observed PM10 and PM2.5 at Masterton West calculated as percent change per year. The solid red line shows the trend estimate and the dashed red lines show the 95% confidence intervals for the trend based on re-sampling methods.

NO_2 model

Variables

The same hyperparameters as PM were used for NO_2. The monthly explanatory variables used are shown in Table A.3: air temperature, ventilation coefficient, boundary layer height, relative humidity, wind speed, and rainfall. NO_2 tube data is only available at monthly resolution for modelling. The NO_2 tube data are nominal calendar months as the sample exchange dates are the first Wednesday of the month +/- two days and the second Wednesday of the month in January. The NO_2 monthly data from Masterton West was the calendar month (ie, not matched to the tube exchange dates).

The temporal predictor variables used were Unix date (number of seconds since 01-01-1970) as the trend term, Julian day (1-366) as the seasonal term and month to represent emissions pressure. Three hundred predictions were used to calculate the meteorologically normalised trend. The normalised trends were aggregated to annual resolution (January to February) for presentation in Section 3.

Table A3: NO_2 January to December 2008 to 2023

Parameter	1-month	Source	Range (min, max)	Number observations
NO_2 (µg/m^3)	Average	Masterton West	1.2 – 12.7	191
NO_2 (µg/m^3)	Average	High Street	6.8 – 26.0	82
Explanatory variables:				
Air temperature 10m (°C)	Average	Masterton West	7.1 – 21.0	190
Wind speed 10m (m/s)	Average	Masterton West	1.5 – 3.3	186
Boundary layer height (BLH) m	Average	ERA5 (1-hr)	389 – 944	192
Ventilation coefficient (m^2/s)	Average	Derived from 1-hr BHL x wind speed.	921 – 3,578	186
Relative humidity (%)	Average	Masterton West	56.3 – 85.8	185
Barometric pressure hPa	Average	Masterton West	990.4 – 1008.9	183
Rainfall (mm)	Total	Masterton West	0.0 – 329.6	192

NO_2 model performance and results

The random forest models using the explanatory variables in Table A.2 on the training dataset explained 82% of the variance in monthly NO_2 at Masterton West and 79% of the variance in monthly NO_2 at the High Street passive diffusion tube site.

	n training observations	Prediction error MSE	R^2
Masterton West	152	1.37	0.82
High Street	65	5.01	0.79

The model prediction error describes how well the model can predict the 20% of data that was withheld from the training dataset.

	RMSE (root mean square error)	R^2
Masterton West	1.51	0.69
High Street	1.68	0.76

Table A4 shows the relative importance of variables in predicting PM concentrations as outputted by the *rmweather* package. The partial dependence plots from the random forest model for monthly NO_2 from Masterton West and the High Street roadside site for seven predictor variables that explain most of the variation are shown in Figures A3 and A4. Wind speed had the largest influence on PM2.5 concentrations, followed by air temperature.

Table A4: Relative importance of explanatory variables

Variable importance	Masterton West	(%)	High Street	(%)
Air temp	4.05	44.1%	10.7	44%
Relative humidity	1.59	17.3%	2.98	12.1%
Month (Jan-Dec)	1.41	15.3%	2.18	8.9%
Wind speed	0.887	9.7%	3.23	13.1%
Time trend	0.689	7.5%	1.47	6.0%
Ventilation	0.225	2.4%	2.43	9.9%
Boundary layer height	0.218	2.4%	1.51	6.1%
Rainfall	0.105	1.1%	0.0208	0.1%
Barometric pressure	0.0148	0.2%	0.073	0.3%

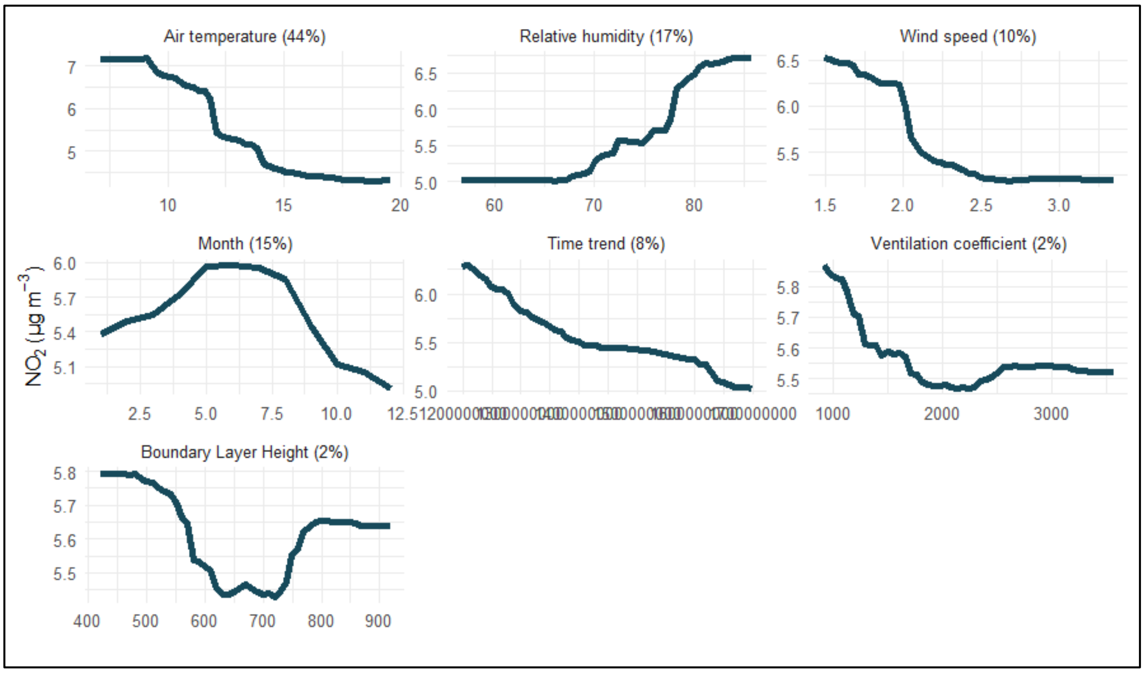


Figure A3: NO_2 Masterton West partial dependencies of seven most influential predictor variables from the random forest model

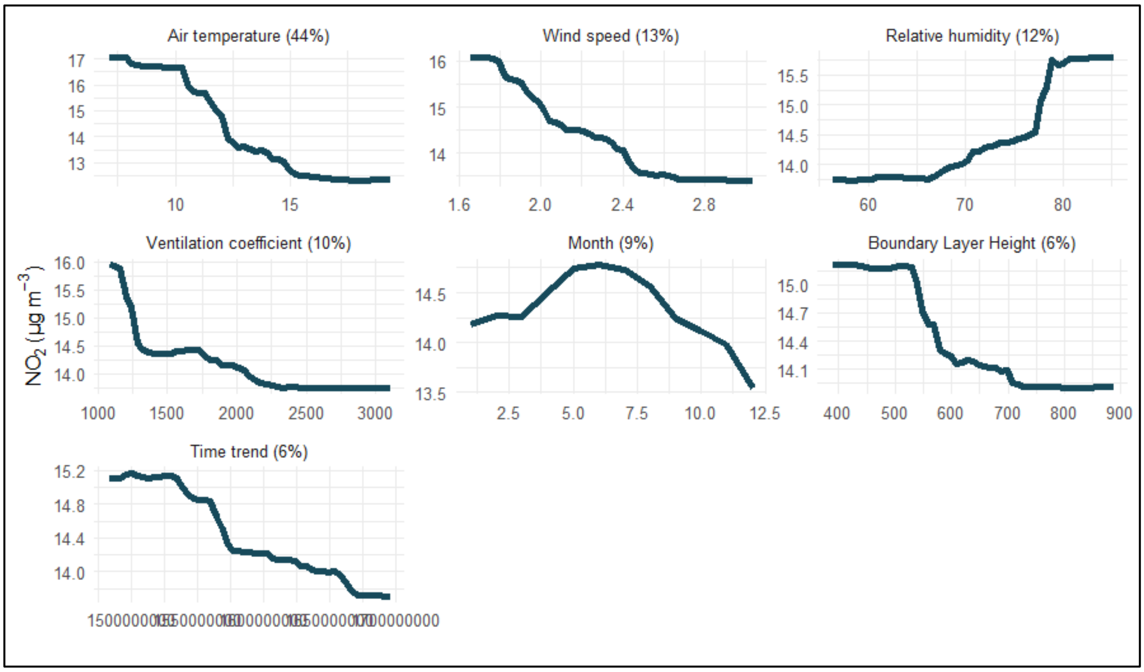
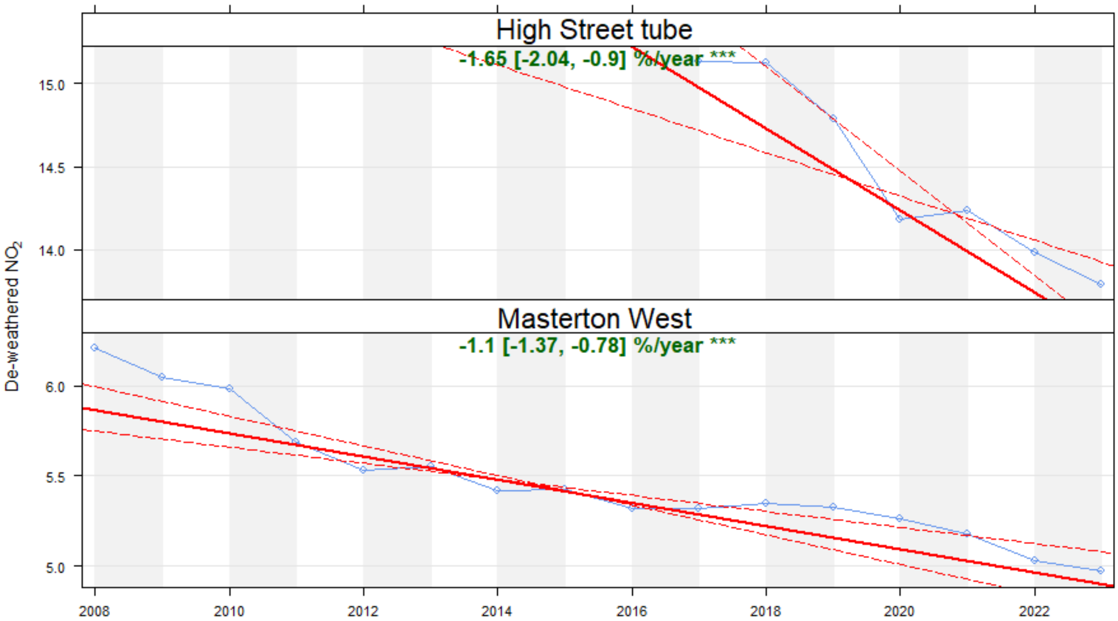


Figure A4: NO_2 tube High Street partial dependencies of seven most influential explanatory variables from the random forest model

NO_2 Theil-Sen trends



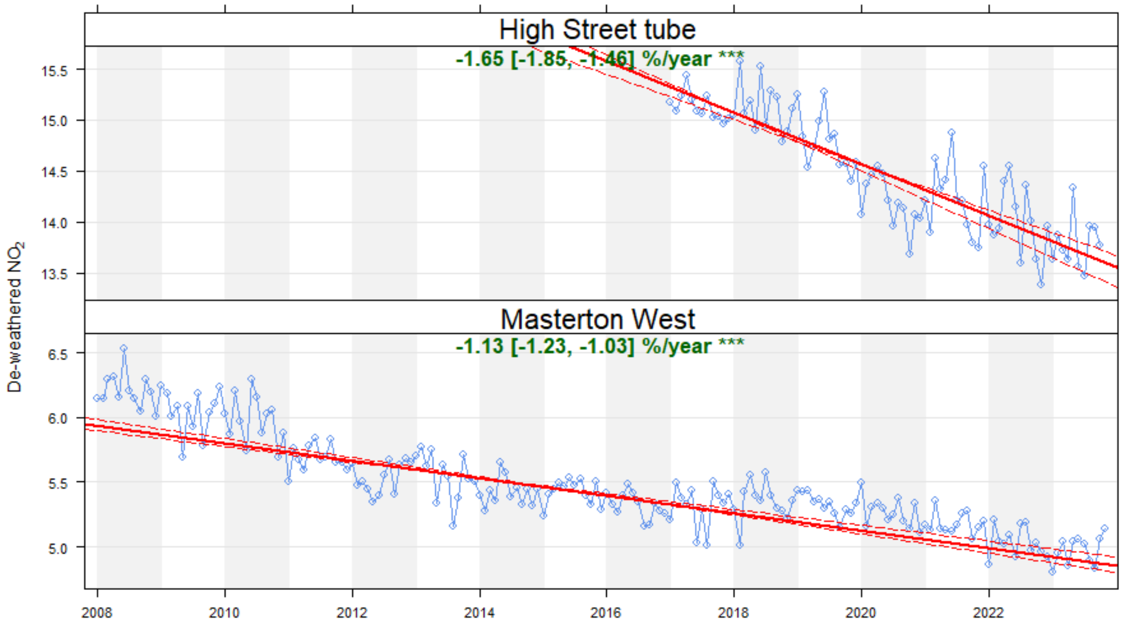


Figure A5: Trends in de-weathered NO_2 at Masterton West and High Street tube site calculated as percent change per year. The solid red line shows the trend estimate and the dashed red lines show the 95% confidence intervals for the trend based on re-sampling methods.

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