Indoor Combustion in New Zealand Homes: Health Effects and Costs

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Report details

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2	17 Jun 2025			Final draft
3	30 Jun 2025			Final report
4	1 Aug 2025			Corrected typo in asthma prevalence exposure response function (per 4 μg/m^3, not 10 μg/m^3) and reinstated low emission burner incremental exposure discussion to body of report (from Appendix A).
5	24 Sep 2025		Lou Wickham	Minor formatting. Add table with economic & fiscal costs to section 5.

Executive Summary

This study estimates the costs of health impacts associated with emissions of harmful air pollutants from wood burners, gas stoves and unflued gas heaters commonly used in Aotearoa New Zealand homes. The work was commissioned by the Energy Efficiency and Conservation Authority (**EECA**) to enable the health impact costs of these appliances to be included in EECA's costs benefit analyses. These estimates could also be used in future updates of New Zealand Treasury's Cost Benefit Analysis (**CBAx**) model.[1]

The appliances assessed in this study included gas stoves and unflued gas heaters as well as a range of wood burners from ultra-low emission burners and standard, modern burners through to (polluting) open fires. The pollutants assessed were the two most critical air pollutants linked to adverse health effects in New Zealand – fine particulate (**$PM_{2.5}$**) and nitrogen dioxide (**NO_2**).

In the first phase of the study, we undertook a comprehensive literature review to estimate annual indoor concentration increments for $PM_{2.5}$ and NO_2. This found high levels of heterogeneity in the literature due to the complexity of factors contributing to indoor air quality and inconsistencies in the way that data are collected and reported for different research purposes.[2] Preference was given to high quality New Zealand data, benchmarked against the latest evidence in the international literature.

We combined estimated annual increments for each pollutant with Aotearoa-specific exposure-response functions and household composition data to determine estimates of mortality and morbidity health impacts. The costs associated with each impact were then updated to current values (2025 dollars or 2025$). The focus on annual exposure reflects World Health Organisation guidance on the significance of chronic exposure relative to acute exposure. Our approach was consistent with other large studies on the impacts of gas stoves and wood burners in Europe and the United States.

Table ES-1 presents the annual indoor air pollution costs (2025$) estimated for each appliance assessed, based on the central (best estimate) values. Gas appliances (both gas stoves and unflued gas heaters) and older wood burners, specifically open fires, result in the highest indoor air pollution costs per household.

For comparison, Table ES-1 includes estimates for the annual *outdoor* air pollution impacts resulting from the typical use of each appliance. This shows that gas stoves generally have more impact at the household level compared with wood burners, which impact more on the community. The exception is open fires, which impact both individual households and the community.

Sensitivity testing showed the indoor air pollution costs are particularly sensitive to the annual exposure increments as the low/high bounds are very wide, being based on the confidence intervals from the various studies reviewed. The uncertainty is particularly high for gas stoves but

[1] Treasury (2024). *CBAx Spreadsheet Model*. Excel spreadsheet model, The Treasury, New Zealand, November 2024. https://www.treasury.govt.nz/publications/guide/cbax-spreadsheet-model
[2] Understanding heterogeneity (variability within or between studies, participants, or data) is crucial for literature reviews, as it can significantly impact the interpretation of findings and the validity of conclusions.

even for the other indoor combustion appliances, the costs range from around 1/3 to twice the central estimates. This reflects the complexity and variability of indoor air pollution. However, regardless of the assumptions and uncertainty, the indoor air pollution costs per household are still appreciable for all indoor combustion appliances assessed – even at the lower bound.

Table ES-1: Comparison of estimated annual indoor air pollution costs with annual outdoor air pollution costs per appliance (2025$)

Appliance	Total annual air pollution ($PM_{2.5}$ + NO_X) costs (2025$)[1]	
	Indoor	**Outdoor**
Electric stove	*Reference case*	*Reference case*
Gas stove[2]	$9,188	$0
Wood burner type 1a (open fire)	$53,401	$26,751
Wood burner type 1b (non-NES)[3]	$7,679	$16,968
Wood burner type 2 (NES compliant)[3]	$1,823	$12,341
Gas unflued heater[2]	$20,436	$0

Notes:

1. Estimates assume wood burner or unflued gas heater is main heating source and includes health outcomes specified in HAPINZ 3.0 (only, refer Table ES-2). For further details of underlying assumptions and calculations refer body of report.
2. Outdoor air pollution scenario assumes zero emissions to (outdoor) air from gas stoves and unflued gas heaters.
3. NES refers to the wood burner emission and efficiency limits in the Resource Management (National Environmental Standards for Air Quality) Regulations 2004.

Putting this in context

Table ES-2 sums the annual impacts of indoor air pollution to a national level for each appliance type and Table ES-3 presents estimated costs for these impacts. The national estimates of health impacts and costs from indoor air pollution are significant and serious. This is because, not surprisingly, the more people exposed to the indoor combustion emissions the higher the indoor air pollution costs.

Table ES-3 shows further that the impacts and costs of both indoor and outdoor air pollution are significant and, in many cases, comparable. This finding is consistent with the WHO Global Burden of Disease study which estimated similar annual impacts from household air pollution (3.2 million deaths globally) and outdoor air pollution (3.5 million deaths globally, WHO 2024).

It is important to note that this study considers only the health outcomes specified in the Health and Air Pollution in New Zealand 3.0 study (Kuschel *et al* 2022). This means that these identified impacts are an underestimate of the actual impacts of air pollution more broadly on society.

Table ES-2: Total estimated annual indoor air pollution impacts for Aotearoa New Zealand

Cases of modelled health outcome	**Gas stoves**	**Wood burners**[1]	**Unflued gas heaters**
(Number of appliances)	*(360,099)*	*(523,686)*	*(44,645)*
Premature mortality (>30 years)	208	101	57
Cardiovascular hospitalisations (all ages)	236	255	66
Respiratory hospitalisations (all ages)	775	191	211
Restricted activity days (all ages)	-	167,204	-
Asthma prevalence (<18 years)	3,230	-	882

Notes:

1. All wood burner types (open fire + non-NES + NES compliant + ultralow emission + pellet burners). For further details of underlying assumptions and calculations refer body of report.

Table ES-3: Total estimated annual indoor and outdoor air pollution costs for Aotearoa New Zealand

Appliance	**Number**	**Total annual air pollution ($PM_{2.5}$ + NO_X) costs in 2025$M**[1]	
		Indoor	Outdoor
Electric stove[2]	1,485,234	Reference case	Reference case
Gas stove	360,099	$3,308	$0
Wood burner type 1a (open fire)	1,973	$105	$53
Wood burner type 1b (non-NES)	101,236	$777	$1,718
Wood burner type 2 (NES compliant)	369,512	$673	$4,560
Gas unflued heater	44,645	$912	$0
National Total[3] (2025$M)		$5,851	$6,484

Notes:

1. Estimates assume wood burner or unflued gas heater is main heating source and includes health outcomes specified in HAPINZ 3.0 (only, refer Table ES-2). For further details of underlying assumptions and calculations refer body of report.
2. Extrapolated from number total occupied dwellings in 2025 (estimated from Census 2018 and 2023 data) less gas stoves
3. NZ Total includes ultra-low emission wood burners (35,433) + pellet burners (15,532) which are not shown in this table. These are provided in the tables in Section 5 and the associated indoor combustion health impacts model.

One of the issues for this study is whether the impacts of indoor air pollution are a subset of the total effects of air pollution or if they are additional. Our literature review found no correlation between indoor and outdoor air pollution for NO_2 and $PM_{2.5}$. Consequently, we are confident there is no double counting in the method we have used to assess the effects of indoor air pollution by appliance.

The results of this study will give EECA the ability to better estimate the impacts of gas stoves, wood burners and unflued gas heaters, and to inform future policy settings for wood burners in residential settings.

Confidence in the modelling could be improved in future by undertaking Aotearoa measurements of indoor $PM_{2.5}$ concentrations in houses with different types of burners (for example, new NES-compliant wood burners, ultra-low emission wood burners and pellet burners). If considered, this research should adopt robust measurement techniques to ensure repeatability of results.

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Glossary

ALA	American Lung Association
AQEG	Air Quality Expert Group, UK
autumn	the months of March, April and May
cardiovascular	of, pertaining to, or affecting the heart and blood vessels
case	a single instance of an adverse health outcome
CBAx	cost-benefit analysis tool, provided by NZ Treasury
CI	confidence interval – a measure of the certainty that a value falls within a given range, e.g. a 95% CI is the range within which there is a 95% probability that the value we are interested in actually sits
ECan	Canterbury Regional Council
economic costs	costs that consider the broader impact on the entire economy, including things like employment and wage effects
EECA	Energy Efficiency and Conservation Authority Te Tari Tiaki Pūngao
ERF	exposure-response function or relative risk function, the increase in risk for every increment in pollution
fiscal costs	costs that relate to the financial impact on public entities (like government)
GAPF	Global Atmospheric Pollution Forum
gas stove	a gas appliance used for cooking, also known as a gas hob or gas cooktop but not enclosed like a gas oven
GDP	Gross Domestic Product
HAPINZ	Health and Air Pollution in New Zealand study
HAPINZ 3.0	the current HAPINZ update for 2016, undertaken by Kuschel *et al* (2022)
harmful emissions	emissions of air pollutants such as particulate matter and nitrogen dioxide which impact human health locally
health outcome	an adverse effect associated with exposure, e.g. a respiratory hospital admission
heterogeneity	the presence of diverse characteristics, differences, or variability within or between studies, participants, or data. Understanding heterogeneity is crucial in literature reviews and meta-analyses, as it can significantly impact the interpretation of findings and the validity of conclusions.
HH	household
incidence	the proportion or rate of persons who *develop* a condition during a particular time period
MBCM	Monetised Benefits and Costs Manual, produced by NZ Transport Agency Waka Kotahi

MfE	Ministry for the Environment Manatū Mō Te Taiao
morbidity	ill health or suffering
mortality	death
NES	The Resource Management (National Environmental Standards for Air Quality) Regulations 2004 which include a suite of ambient air quality standards and a wood burner emission limit
NES-compliant wood burner	a wood burner that emits less than 1.5 grams of PM per kilogram of dry wood burnt and has a thermal efficiency of no less than 65%
NO_2	nitrogen dioxide, a harmful pollutant
non-NES wood burner	a wood burner (older than 2005 or installed on a property larger than 2 hectares) that emits more than 1.5 grams of PM per kilogram of dry wood burnt
NZTA	NZ Transport Agency Waka Kotahi
open fire	a non-enclosed (open) appliance that burns wood or coal to heat a home, especially in winter
PAF	population attributable fraction, the estimated percentage of total health cases that are attributable to air pollution exposure
pellet burner	an appliance that burns uniformly-sized wood pellets with controlled feed rate and combustion conditions. Pellet burners emit less PM than NES-compliant wood burners and some models meet the ULEB requirements
PM	particulate matter
$PM_{2.5}$	particulate matter less than 2.5μm, a harmful pollutant
prevalence	the proportion of a population who *have* a specific characteristic in a given time period
PRISMA	preferred reporting items for systematic reviews and meta-analyses
RAD	restricted activity day, a day on which people cannot do the things they might otherwise have done if air pollution was not present
respiratory	of, pertaining to, or affecting the lungs and airways
RR	risk ratio
social cost	the total costs to society of health effects, which are more than just the costs incurred by the health system, e.g. the costs of lost productivity and suffering
spring	the months of September, October and November
StatsNZ	StatisticsNZ Tatauranga Aotearoa, the public service department charged with the collection of statistics related to the economy, population and society of NZ

summer	the months of December, January and February when average ambient temperatures are typically at their highest
Treasury	The Treasury Te Tai Ōhanga
$\mu g/m^3$	microgram per cubic metre, a unit of concentration
UK	United Kingdom
ULEB	ultra-low emissions burner
ultra-low emissions burner	a wood burner that meets an emissions and efficiency standard of 38 milligrams per megajoule of useful energy and have a thermal efficiency of 65% or greater. ULEBs typically emit less than 1/3 of the PM emitted by a NES-compliant burner.
µm	micrometre, one millionth of a metre
unflued gas heater	a home heating appliance burning gas but with all combustion products emitted inside (i.e. not vented/flued outside)
US	United States of America
VoLY	value of a life year
VoSL	value of statistical life
Waka Kotahi	NZ Transport Agency Waka Kotahi
WHO	World Health Organization
winter	the months of June, July and August when average ambient temperatures are typically at their lowest
wood burner	an enclosed appliance that burns wood to heat a home, especially in winter
YLL	year of life lost

1. Introduction

1.1 Study scope

The role of the Energy Efficiency and Conservation Authority (**EECA**) is to promote, support, and encourage the use of energy efficient and renewable energy technologies in Aotearoa New Zealand. Cost-benefit analyses are a useful tool in the design and evaluation of EECA programmes as they permit a broader assessment of the full benefit of energy efficient and renewable technologies.

While New Zealanders regularly use a range of appliances for cooking and heating, the only currently published costs for assessing health impacts from combustion emissions in Aotearoa relate to outdoor pollution. Recognising the gap in understanding, EECA funded this study to develop corresponding indoor health impact costs – suitable for being included in the next update of the Treasury's Cost Benefit Analysis (**CBAx**) model.[3] Consequently, it was important that the methodology for estimating indoor pollution and its health costs be clear and able to stand up to scrutiny.

EECA specifications included that the research:

- analysed and synthesised existing international and domestic evidence, into a cohesive set of impacts and related costs relevant to the New Zealand domestic sphere
- focused on gas stoves and wood burners, with a secondary focus on unflued gas heaters
- utilised a desktop approach.

Problem definition

While the outdoor health impacts of wood burner emissions in Aotearoa are well understood (e.g. Kuschel *et al* 2022), the indoor health impacts of gas stove emissions are poorly understood.

Similarly, unflued gas heaters operated indoors are known to cause harm and existing advice warns against using them, however the economic costs of the harm is unquantified.

As a result, indoor air pollution is not currently considered in government cost-benefit analyses.

1.2 Approach

We undertook a review of the New Zealand and international literature and synthesised the findings to estimate:

- in-home exposure to particulate matter less than 2.5 micrometres in diameter (**$PM_{2.5}$**) and nitrogen dioxide (**NO_2**) from wood burners, gas stoves and unflued gas heaters
- associated health effects

[3] Treasury (2024). *CBAx Spreadsheet Model*. Excel spreadsheet model, The Treasury, New Zealand, November 2024. https://www.treasury.govt.nz/publications/guide/cbax-spreadsheet-model

- associated economic costs and fiscal costs for New Zealanders.

Our focus was on *annual* exposure because the effects of long-term exposure are an order of magnitude more significant than short-term exposure for $PM_{2.5}$ (WHO 2021).

We adopted an incremental approach to the counterfactual. In simple terms, health effects arising from infiltration of outdoor air into indoor spaces and other internal discharges to air (including, for example, from cooking food) may be put to one side because all residents experience these. Our focus was on determining the *direct* indoor exposure to emissions from wood burners, gas stoves and unflued gas heaters.

We then modelled selected health outcomes consistent with those assessed in the Health and Air Pollution in New Zealand 2016 (**HAPINZ 3.0**) study (Kuschel *et al* 2022) for exposure to outdoor air pollution - premature mortality, years of life lost, hospitalisations, restricted activity days and asthma prevalence.

These associated costs of the health outcomes were updated to 2025 dollars before being disaggregated to a per pollutant and per appliance basis to develop estimates for the fiscal (health system costs) and economic costs (wider costs of pollution on society). These are presented by pollutant and by appliance in tabular form to permit cost benefit analyses and to provide useful alignment with a potential future update of the Treasury's CBAx model (The Treasury 2024).

Each step in the methodology is discussed in more detail in the relevant chapters that follow.

1.3 Report structure

The report is structured as follows:

- Chapter 2 outlines the approach we followed for the literature review, the resources identified and their key findings
- Chapter 3 presents the best estimates for the annual indoor pollution increments and their modelled health impacts
- Chapter 4 presents the updated values for the costs of the mortality and morbidity impacts
- Chapter 5 reports on the results and the findings of the sensitivity analyses.

A glossary of terms/abbreviations is included at the front of the report, with all references listed at the end.

2. Literature review

2.1 Methodology followed

The critical input to the health modelling was the estimate of incremental exposure. This relied on the findings of a literature review that we undertook in line with good practice (Page *et al* 2021) as follows:

Step 1. Previous Studies

- We undertook a broad literature scan to identify recently published, relevant, systematic reviews (searching PubMed and EBSCO databases) and health impact studies. This identified three recent studies and 10 important reports on the health impacts of indoor combustion.
- Together, these 13 "previous studies" comprised health impact assessments, systematic literature reviews and meta-analyses of more than 2,277 studies – and all in relation to indoor combustion.

Step 2. New Identified Studies

- We next undertook a systematic review of the New Zealand and Australian literature with two specific searches for studies to 10 February 2025, [4] based on the following terms (EBSCO database):
  - (wood burner OR wood stove OR wood fire) AND (indoor air quality OR indoor air pollution) AND (Australia OR New Zealand)
  - (gas cooking OR gas stove OR gas hob) AND (indoor air quality OR indoor air pollution) AND (Australia OR New Zealand)
- These were screened to remove double ups, non-relevant research and studies that were not based in Australia or New Zealand. These are the "new identified studies".

Step 3. Follow-up Studies

- We further searched relevant citations from the reviews and studies identified in step 1 and step 2 to identify relevant exposure data. These are the "follow-up studies" and comprised an additional seven reports and 52 papers.

Figure 1 (overleaf) presents the features of our review using the preferred reporting items for systematic reviews and meta-analysis (Page et al 2021) guideline flowchart. It should be noted that a systematic review was only undertaken for new identified studies (items in orange boxes in Figure 1). The literature review, with notes on key studies, is provided as **supplementary information** (*Indoor Combustion Lit Review 1 May.xlsx*).

Note to EECA: An additional six papers to those reported in our memo dated 25 February 2025 were subsequently identified and incorporated into our health modelling. (Sun *et al* 2025 to Kornartit *et al* 2010) as noted in the supplementary information.

[4] The search was rerun on 17 March 2025 with the inclusion of "cooktop". This identified one additional (non-relevant) study.

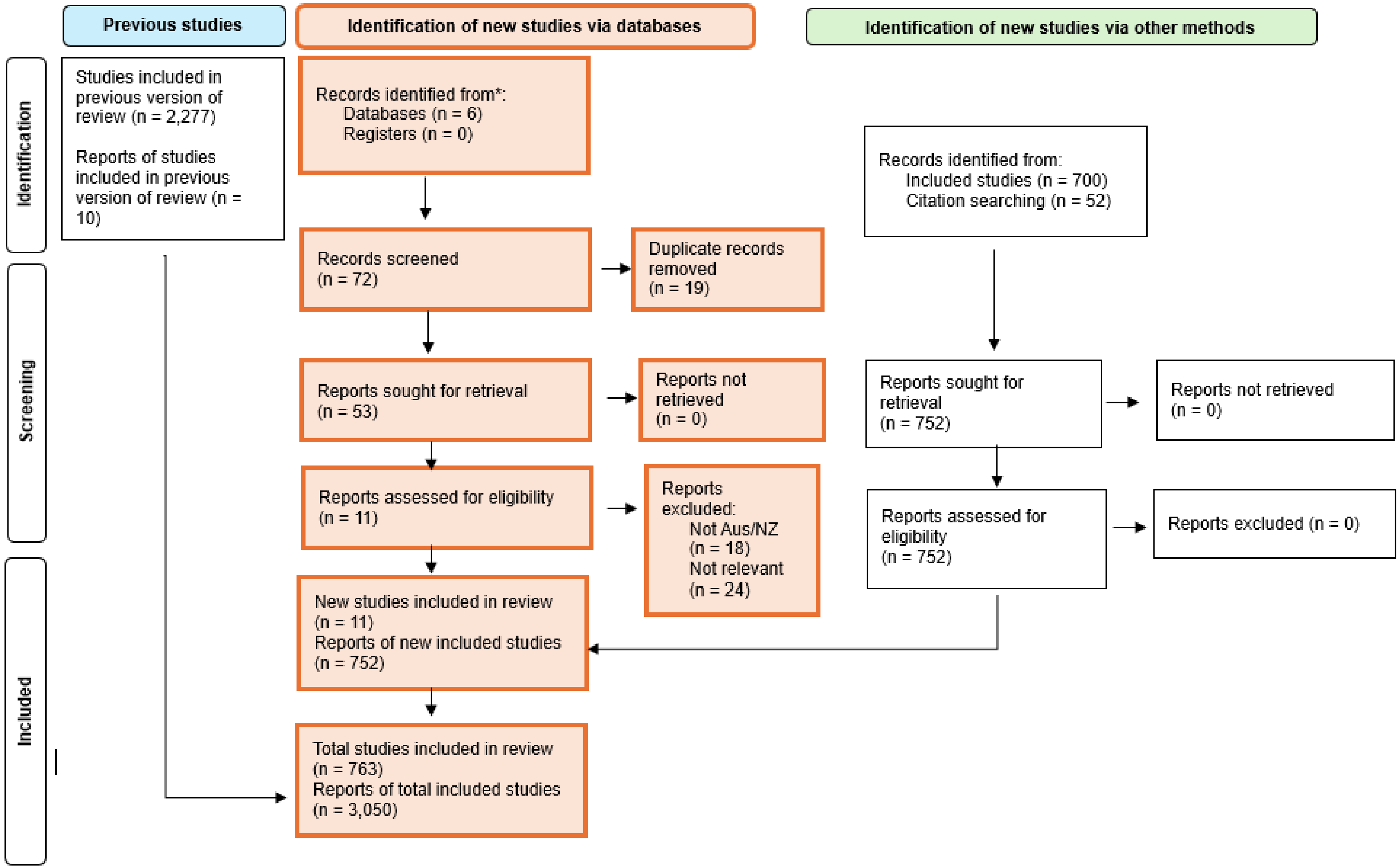

Figure 1: PRISMA 2020 flow diagram for indoor combustion study literature review

Note: Only orange boxes meet PRISMA 2020

2.2 Indoor air quality

Three studies provide extensive discussion of current scientific evidence and evidence gaps relating to indoor air quality overall.

2.2.1 Key resources identified

1. Air Quality Expert Group (2022). *Indoor Air Quality in the UK*.
2. Vardoulakis *et al* (2020). *Indoor Exposure to Selected Air Pollutants in the Home Environment: A Systematic Review*.
3. Wei & Semple (2023). *Exposure to fine particulate matter ($PM_{2.5}$) from non-tobacco sources in homes within high-income countries: a systematic review*.

2.2.2 Key findings

- Indoor air quality is important because people spend a substantial amount of time indoors.
- Indoor air quality is complex. It is influenced by pollution levels in outside air, the rate of air exchange between indoors and outdoors, processes such as deposition to surfaces, and complex chemical reactions as well as indoor sources of pollution (including gas and solid fuel combustion).
- Compared with outdoor air quality, evidence about exposure to air pollution indoors is sparse. The only quantitative evidence on indoor air quality comes from individual research studies in specific indoor micro-environments (e.g. homes, schools, transport, rail stations, shops etc.) with fragmented and inconsistent pollutant speciation. Most research studies report information for only a small number of pollutants over a short period of sampling (providing only a snapshot of concentrations) and with limited information on occupant activities.
- There is wide variation in reported indoor air pollutant concentrations. This is due to the complexity of factors contributing to indoor air quality and inconsistencies in the way that data are collected and reported.

The following sections outline the key resources identified for each appliance types and important findings that guided our approach to estimating incremental exposure.

2.3 Gas stoves

2.3.1 Key resources identified

(i) Delgado-Saborit *et al* (2024). *Assessment of the health impacts and costs associated with indoor nitrogen dioxide exposure related to gas cooking in the European Union and the United Kingdom*.

(ii) Kashtan *et al* (2024). *Nitrogen dioxide exposure, health outcomes, and associated demographic disparities due to gas and propane combustion by U.S. stoves*.

(iii) Puzzolo *et al* (2024). *Estimated health effects from domestic use of gaseous fuels for cooking and heating in high-income, middle-income, and low-income countries: a systematic review and meta-analyses*.

2.3.2 Key findings

Health effects indoors

- It is widely acknowledged that gas stoves emit NO_2, and that exposure to NO_2 is associated with health impacts.
- A systematic review for WHO (Puzzolo *et al* 2024) of the health effects from the use of gas for cooking found increased risk of pneumonia, chronic obstructive pulmonary disease and asthma in children (although asthma in children was not statistically significant).
- Recent studies in Europe and the US have estimated paediatric asthma and premature mortality associated with exposure to NO_2 from gas stoves (Delgado Saborit *et al* 2024, Kashtan *et al* 2024). These studies apply the exposure response function (**ERF**) for asthma ratios from Puzzolo *et al* (2024), as well as the ERF for premature mortality from outdoor exposure to NO_2. These studies assume that ERFs for effects due to long-term exposure to outdoor NO_2 from these meta-analyses can be equally applied to long-term exposure to indoor NO_2.
- We have Aotearoa-specific ERFs for effects relating to long-term exposure to NO_2, which are based on a large cohort study undertaken by Hales *et al* (2021). However, NO_2 exposure in New Zealand is largely attributed to traffic (Kuschel *et al* 2022).
- It is important to note that the health impacts of traffic-related NO_2 may be different to those from gas stove related NO_2. Kashtan *et al* (2024) acknowledge that applying ERFs for outdoor exposure to NO_2 will likely overestimate the NO_2 attributable health burden because of additional pollutants found in traffic-related air pollution. However, paradoxically, they may also underestimate health burdens because their estimates account for long-term only NO_2 exposures and not short-term exposures to high concentrations, which routinely exceeded 100 ppbv in their measurements.

Exposure

- To estimate health effects, we need to estimate indoor exposure to NO_2 from gas stoves in Aotearoa.
- New Zealand studies are the most relevant to this study. However, results are limited. To help bolster our exposure assessment, we reviewed international studies together with promising papers identified in these studies.

2.3.3 Impact of gas stoves on indoor NO_2 exposure

New Zealand studies

- High quality Aotearoa exposure data are available from Gillespie-Bennett *et al* (2008). The study measured NO_2 concentrations in almost 350 New Zealand homes over 16 weeks during winter in 2005 and 2006. The study was focused on the impact of unflued gas heaters, but data on use of gas stoves for cooking were also collected. The study found NO_2 levels were approximately 1.5 times higher in living rooms in households that had gas cooktops compared with those households which did not have gas cooktops.

  Despite being nearly 20 years old, this study sets a high bar with a large, Aotearoa cohort and good quality data from measurements of key exposure parameters.

- To estimate annual exposure to NO_2 from gas stoves, we need to account for seasonal variation. NO_2 indoors is expected to be lower during summer compared to winter (because windows tend to be open etc).

International studies

- Sun *et al* (2025) measured NO_2 concentration in 344 Canadian homes for 5 to 7 days in winter and summer. The study found no correlation between measured NO_2 and presence of a range hood. 71% of homes had a range hood, 39% of these vented outside, but only 17% of respondents used them during gas cooking
- Kashtan *et al* (2024) developed a model to estimate exposure in US homes based on emission rates (from tests of 82 burners), residence size and room layout, time spent with windows open, frequency of use, range hood efficiency and time spent in kitchen and different rooms. The study estimates that gas combustion in US stoves increases long term NO_2 exposure by 4 ppbv on average.
- Delgado-Saborit *et al* (2024) estimated the indoor/outdoor ratio of NO_2 for homes with gas stoves in Europe. The study was based on the results of monitoring undertaken in about 180 households across seven countries for a period of 13 days in winter (Jacobs *et al* 2023). The study found no significant association between measured NO_2 and extraction hoods, kitchen volume or ventilation. Monitoring data are available in the Jacobs *et al* (2023) report. The authors note that exposure to NO_2 from gas stoves indoors is expected to be lower during summer compared to winter.
- Some data on seasonal variation of indoor NO_2 in households with gas stoves are available (Sun *et al* 2025, Kornartit *et al* 2010).

2.3.4 Mitigation measures

The AQEG (2022) report states:

> *A major source of NO_X and PM indoors is cooking using gas, either natural gas or LPG mixtures. **A simple mitigation measure is to replace gas cooking appliances with electric equivalents whenever possible**, a change that is likely to be needed as part of wider home decarbonisation and transition to net zero.*

2.4 Wood burners

2.4.1 Key resources identified

(i) McCarthy (2024). *Review on Wood Burning Stoves and Indoor Air Pollution*.

2.4.2 Key findings

Health effects generally

- Systematic reviews of evidence about the health impacts of exposure to air pollution from solid fuel combustion are generally inconclusive (Guercio *et al* 2021, Guercio *et al* 2022). However, this is due to limited evidence about specific exposure and impacts of solid fuel combustion emissions.

- WHO (2015) includes useful discussion of the epidemiological evidence for domestic solid fuel burning as follows:

  An important consideration is to what extent results from epidemiological studies on urban PM can be generalized to PM from residential wood combustion. In the WHO air quality guidelines (WHO 2006) it was concluded that ***there was little evidence that the toxicity of particles from biomass combustion would differ from the toxicity of more widely studied urban PM****. This same approach was followed in the analysis presented in section 4 and in the recent GBD Study (Lim et al 2012), in which all combustion particles, regardless of source, were considered hazardous depending on the exposure level.*

- We have Aotearoa-specific ERFs for $PM_{2.5}$, which are based on a large cohort study undertaken by Hales *et al* (2021). $PM_{2.5}$ exposure in New Zealand is largely due to domestic fires used for home heating (Kuschel *et al* 2022).
- Health impacts and social costs associated with exposure to **outdoor** air pollution from individual domestic fires of different appliance types can be estimated as follows:
  - Exposure to $PM_{2.5}$ from domestic fires used for home heating was associated with 962 premature deaths and 3,375 hospitalisations in New Zealand in 2016 (Kuschel *et al* 2022). The health impacts of both $PM_{2.5}$ and NO_2 from all sources in Aotearoa were valued and then linked to national air emissions inventory results to arrive at damage costs (expressed as social costs in $per tonne of pollutant) for exposures ranging from urban (worst case) to rural, with New Zealand average values also estimated.
  - Damage costs are a simple method to evaluate the impact of policy on air pollution health impacts. They are a proxy only, but are widely used in policy analysis, such as the evaluation of vehicle emission regulations for Ministry of Transport (Metcalfe & Kuschel 2022).
  - The Treasury publish Aotearoa average damage costs annually (Treasury 2024) and urban and rural damage costs are available in NZTA (2024). However, current values are in 2024 dollars or earlier and need to be updated to 2025 dollars (2025$).
  - Damage costs can be applied to estimate the social cost of <u>outdoor</u> air pollution <u>per domestic fire appliance</u> based on published real world emission factors (e.g. Environment Canterbury 2024).

Health effects indoors

- It is widely acknowledged that domestic fires can cause elevated concentrations of $PM_{2.5}$ **indoors.** However, other sources (e.g. cooking, smoking, candles, incense) can be more significant (AQEG 2022, Lea-Langton 2022, Wei & Semple 2022, Vardoulakis *et al* 2020, WHO 2015).
- The health effects of **indoor** air pollution from domestic fires can be estimated based on health effects of exposure to $PM_{2.5}$, assuming the relationships between exposure and health impacts **outdoors are applicable indoors**. This approach has been implemented by WHO to assess the health impacts of $PM_{2.5}$ from solid fuel burning for cooking indoors (Bennitt *et al* 2021) and to assess the health impacts of NO_2 emissions from gas stoves indoors in the US and Europe (Kashtan *et al* 2024, Delgado-Saborit *et al* 2024).

Exposure

- To estimate health effects, we need to estimate indoor exposure to $PM_{2.5}$ from wood burners.

- New Zealand studies are the most relevant to this study. However, results are limited. To help bolster our exposure assessment, we reviewed international studies together with promising papers identified in these studies.

2.4.3 Impact of wood burners on indoor $PM_{2.5}$ exposure

Aotearoa studies

- Trompetter & Davy (2019) measured indoor and outdoor air quality for seven homes in Wellington. Each home was intensively monitored for 15-20 days sequentially between May and September 2017 (wintertime). The concentration and composition of particulate matter were determined with a two-hour resolution inside and outside. The composition data enabled source apportionment to be used to confirm the sources of particulate. The major sources identified were:
    - Cooking emissions resulting in large evening particulate matter concentration peaks
    - Leakage of particulate matter to indoor spaces from wood burners in those houses where such appliances were used for space heating[5]
    - Re-entrained soil/dust generated indoors when homes are occupied
    - Infiltration of outdoor particulate matter sources, including wintertime wood burning
    - Marine aerosol (sea salt).
- The Trompetter & Davy (2019) study provides high quality, quantitative data.
- Longley (2020) investigated indoor air quality in Arrowtown using low-cost air quality monitors. A modelling approach was developed to estimate the contribution of outdoor sources to indoor air pollution. The study estimated that outdoor sources contribute around 40-80% (mean of 57%) in July to October, and 50-100% (mean of 82%) of indoor $PM_{2.5}$ in June (midwinter). Longley suggests that key sources of indoor air pollution may include cooking and smoking. This is broadly consistent with findings of Trompetter & Davy (2019) as well as international literature. Additional monitoring has been undertaken but has not been published (refer memo from Ian Longley to EECA dated February 2025). [6]

International studies

- Several studies have shown that indoor $PM_{2.5}$ concentrations are somewhat higher in homes with a wood burner compared to homes without:
    - Wyss *et al* (2016) measured $PM_{2.5}$ in 36 households in Norway for one week and found mean indoor concentrations were higher in homes with older wood burners (20.2 $\mu g/m^3$ compared to homes without a wood burner (12.6 $\mu g/m^3$). However, homes with new woodstoves had levels similar to homes without a wood burner.
    - Fleisch *et al* (2019) investigated $PM_{2.5}$ levels in 137 homes in New England, USA for one week and found 21% higher $PM_{2.5}$ levels in homes with a wood burner compared to homes without. The authors considered that $PM_{2.5}$ is not the best indicator of wood

[5] For those houses that had wood burners used for space heating, it was found that the combustion emissions from the appliances leaked into the indoor room space and that indoor concentrations were unrelated to outdoor concentrations due to differing concentration and activity profiles.
[6] Longley (2025). *Data and data products for Indoor Combustion Study*. Memo by I Longley, Air Quality Collective to EECA, 3 February 2025.

smoke pollution due to the impact of other indoor sources (e.g., cleaning, candles and incense). Fleisch *et al* (2019) noted that their results are broadly similar to previous US studies which found PM concentrations to be 4% higher (24 homes, Sexton *et al* 1984), 5% higher and 36% higher (35 homes and 45 homes respectively, Leadearer *et al* 1994) in homes with versus without wood stoves.

- Other studies have found no significant effect of wood burners on the indoor concentration of $PM_{2.5}$:
    - Semple *et al* (2011) measured 24-hour $PM_{2.5}$ inside 100 houses with either solid fuel, gas combustion or tobacco smoke in Ireland and Scotland. The study found that $PM_{2.5}$ levels were similar in homes burning wood, coal or gas to outside levels.
    - Siponen *et al* (2019) measured $PM_{2.5}$ in 37 homes in Finland for 22-hour time periods in the heating season. The study found that cooking was the only activity associated with higher indoor $PM_{2.5}$. However, the study found that wood stove use was associated with higher personal exposure to $PM_{2.5}$ (measured with a device carried in a backpack) and higher levels of indoor $PM_{2.5}$ light absorption (a proxy for indoor black carbon levels).
- Other studies have examined the difference between indoor air quality when wood burners are lit compared to when they are not:
    - A study in the UK compared $PM_{2.5}$ levels when stoves are lit with the period when they are not in use (Chakraborty *et al* 2020). Levels were measured in 20 houses with stoves over four weeks. The study found that average indoor $PM_{2.5}$ is higher (12.21 $\mu g/m^3$) when stoves are lit compared to the period when they are not in use (4.12 $\mu g/m^3$). They also found high peak levels due to 'flooding' events associated with opening the stove door.
    - Vicente *et al* (2020) undertook comprehensive monitoring of particulate and toxic air pollutants in an unoccupied house during operation of an enclosed wood burner. The tests found that PM_{10} concentrations were approximately double initial background and outdoor PM_{10} over the 3 x 8-hour periods when the wood burner was in use.
- Overall, the international evidence suggests that indoor PM concentrations increase during wood burner use. However, the results are highly variable.

2.4.4 Mitigation measures

- Several studies have shown improvements in indoor air quality following the replacement of older sold fuel stove technologies with newer ones that meet more stringent emissions criteria, and other studies have found an assocation between newer wood burners and better indoor air quality (Walker *et al* 2021, Wyss *et al* 2016, Noonan *et al* 2012, Ward et *al* 2011).
- Conversely, several studies have found no association between wood burner age and indoor air quality (Rahman *et al* 2022, Ward *et al* 2017, Allen *et al* 2009). Ward *et al* (2017) suggests that one possible explanation for the highly variable outcomes of stove upgrade studies is that residents are not operating their stoves optimally.
- Recent studies have found that regular flue cleaning has a significant impact on indoor air quality and may be more effective as a mitigation than changing out wood stoves to newer models (Rahman *et al* 2022). Walker *et al* 2021 reported that households that had not cleaned

the chimney within the past year had 65% higher geometric mean indoor $PM_{2.5}$ compared to those with chimney cleaned within 6 months (95% confidence interval [CI]: -1,170]) n = 88.

Overall, the evidence suggests that factors such as combustion efficiency and fuel characteristics can be significant. When wood moisture content is high and combustion efficiency is low, pollutant concentrations are generally shown to increase. Studies also highlight the positive effect of good wood stove maintenance such as regular cleaning and fixing leaks and drafts, both of which can reduce indoor pollutant concentrations (McCarthy 2024).

2.5 Literature review conclusions

The literature review had two stand-out findings:

(i) Indoor air quality is significantly influenced by an extremely wide range of factors and is **not** directly correlated with outdoor air quality; and

(ii) Cooking is a major source of indoor particulate matter.

It was important therefore, that the impacts of cooking and outdoor air quality were not confused or double counted when estimating the impacts of indoor air pollution by appliance. This led us to focus on determining the *direct* indoor exposure to emissions from wood burners, gas stoves and unflued gas heaters using an incremental assessment against the counterfactual.

Our priority was sourcing robust data, ideally from **Aotearoa research**. Key identified studies were:

- **Gillespie-Bennett *et al* (2008)** – for estimating indoor NO_2 from cook stoves and unflued gas heaters. This was cross-checked against Sun *et al* (2025), Kashtan *et al* (2024), Jacobs *et al* (2023) and Kornatit *et al* (2010).

- Vicente et al (2020), Fleisch *et al* (2019), Wyss *et al* (2016) – for estimating indoor $PM_{2.5}$ exposure increments from open fires and old wood burners;

- Chakraborty *et al* (2020) and Salthammer *et al* (2014) – for estimating indoor $PM_{2.5}$ exposure increments from new wood burners;

- **Trompetter & Davy (2019)** for cross-checking the relevance of the above PM studies for a New Zealand context.

Full details are in **Appendix A**.

2.6 Further work /implications for future research

There is sufficient evidence on the adverse effects of NO_2 from natural gas combustion. We therefore do not recommend additional research on indoor NO_2 from gas combustion.

It would potentially be of interest to consider the difference, if any, between indoor concentrations of $PM_{2.5}$ from different types of burners (for example, new NES-compliant wood burners, ultra-low emission wood burners and pellet burners). This should adopt robust measurement techniques to ensure repeatability of results.

3. Health effects modelling

Using the findings from the literature review (Chapter 2), we estimated:

- in-home exposure to particulate matter less than 2.5 micrometres in diameter (**$PM_{2.5}$**) and nitrogen dioxide (**NO_2**) from wood burners, gas stoves and unflued gas heaters
- associated health effects.

This chapter summarises the methodology followed.

All calculations, assumptions and references are documented in an indoor combustion emissions model, provided as **supplementary information** (*Indoor combustion health impacts model.xlsx*).

3.1 Annual indoor exposure increments by appliance

Our focus was on determining:

(i) the *direct* indoor exposure to emissions from wood burners, gas stoves and unflued gas heaters; and

(ii) *annual* exposure because the effects of long-term exposure are an order of magnitude more significant than short-term exposure for $PM_{2.5}$ (WHO 2021).

The counterfactual, or reference case, was assumed to be a house with an electric stove and a heat pump. The counterfactual further assumes:

- The same level of indoor concentrations of $PM_{2.5}$ from cooking and other indoor sources (e.g., candles, vacuuming) as in the houses with gas stoves, unflued gas heaters and wood burners. Accordingly, zero effects are ascribed to electric stoves as a reference case.

- The same level of indoor concentrations of $PM_{2.5}$ from the infiltration of outdoor air pollution as in the houses with gas stoves, unflued gas heaters and wood burners.

- All houses are heated equally with no difference in cost. This ignores the cost-effectiveness of heat pumps and potentially free sources of firewood as well as differences in the effectiveness of different forms of heating but reflects the purpose of the study to assess the health effects of air pollution (only) arising from different appliances.

Accordingly, we estimated incremental, indoor, annual exposure to NO_2 (from gas stoves and unflued gas heaters) and $PM_{2.5}$ (from wood burners) using the best available data to arrive at central, default values.

We also developed high and low values for sensitivity testing (**Chapter 5**), based on the confidence intervals (**CI**s) reported in the literature as well as a range of realistic assumptions (e.g. seasonal fuel use).

A full discussion of the references used to develop the increments is contained in **Appendix 1.**

3.1.1 NO_2 exposure increment for gas stoves

To estimate annual indoor exposure to NO_2 from gas stoves, we relied primarily on Gillespie-Bennett *et al* (2008) because this reports good quality, Aotearoa data which are directly relevant.

The wintertime NO_2 concentration increment for houses with gas stoves measured in Gillespie-Bennett *et al* (2008) is similar to the increment found in a recent European study (Jacobs *et al* 2023) but is lower than increments found in North American (Kashtan *et al* 2024) and Canadian studies (Sun *et al* 2025).

We also needed to account for seasonal[7] variation which is impacted by house ventilation. NO_2 indoors is expected to be lower during summer compared to winter because windows and doors tend to be open with greater air flow and reduced indoor concentrations of pollutants.

A study undertaken in England found that NO_2 in houses with gas stoves is similar to houses without gas stoves during summer (Kornartit *et al* 2010). However, a Canadian study found that while NO_2 in houses with gas stoves was significantly higher in summer and winter compared with houses without gas stoves, this difference was less in summer (Sun *et al* 2025).

For New Zealand houses with **gas stoves**, we estimate an
annual indoor NO_2 exposure increment of 2.88 μg/m^3
(with low to high values 0.30 – 7.43 μg/m^3)

The **central** estimate assumes:

- The winter increment is 4.6 μg/m^3 based on Gillespie-Bennett *et al* (2008)
- The summer increment is 25% of the winter increment based on Kornartit *et al* (2010)
- The autumn and spring increments are 63% of the winter increment (midpoint of 25-100%).

The **low** value assumes:

- The winter increment is 0.6 μg/m^3 based on the lower 95% CI in Gillespie-Bennett *et al* (2008)
- The summer increment is 0 μg/m^3
- The autumn and spring increments are 50% of the winter increment (midpoint of 0-100%)

The **high** value assumes:

- The winter increment is 9.9 μg/m^3 based on the upper 95% CI in Gillespie-Bennett *et al* (2008)
- The summer increment is 50% of the winter increment based on Sun *et al* (2025)
- The autumn and spring increments are 75% of the winter increment (midpoint of 50-100%)

[7] Winter is considered to include the months of June to August, Spring includes September to November, Summer includes December to February and Autumn includes March to May.

3.1.2 NO_2 exposure increment for unflued gas heaters

To estimate annual indoor exposure to NO_2 from unflued gas heaters, we relied primarily on Gillespie-Bennett *et al* (2008) because this reports good quality, Aotearoa data which are directly relevant.

We also needed to account for seasonal variation, due to fuel use. Unflued gas heater fuel use, because it is used for home heating during colder weather, can be reasonably assumed to mirror wood burner fuel use. Robust estimates of monthly operation of wood burners across New Zealand are reported in Wilton *et al* (2015).

For Aotearoa houses with **unflued gas heaters**, we estimate an
annual indoor NO_2 exposure increment of 6.62 μg/m^3
(with low to values 4.44 – 9.92 μg/m^3)

The **central** estimate assumes:

- The winter increment is 18.2 μg/m^3 based on Gillespie-Bennett *et al* (2008)
- The increments for other seasons are the same as the average use of wood burners outside winter across New Zealand based on Wilton *et al* (2015), i.e. spring = 18%, summer = 1% and autumn = 26% of winter values.

The **low** value assumes:

- The winter increment is 15 μg/m^3 based on the lower 95% CI in Gillespie-Bennett *et al* (2008)
- The increments for other seasons are the same as the lowest use of wood burners outside winter across Aotearoa based on Wilton *et al* (2015), i.e. spring = 4%, summer = 0% and autumn = 14% of winter values.

The **high** value assumes:

- The winter increment is 21.8 μg/m^3 based on the upper 95% CI in Gillespie-Bennett *et al* (2008)
- The increments for other seasons are the same as the highest use of wood burners outside winter across New Zealand based on Wilton *et al* (2015), i.e. spring = 33%, summer = 7% and autumn = 42% of winter values.

3.1.3 $PM_{2.5}$ exposure increments for wood burners

From the literature review, we concluded that **indoor air concentrations of key pollutants from wood burners are highly variable** as they are impacted by a wide range of factors independent of indoor (and outdoor) combustion sources. Variables for emissions from the wood burners themselves *include,* but are not limited to:

- house design
- ventilation
- type of burner
- operator behaviour

- fuel quality; and
- hours of operation.

In addition to these, other sources of $PM_{2.5}$ in houses can also make significant contributions. These can include cooking, smoking, vacuuming, candles, incense, etc. The contribution of these other sources to indoor $PM_{2.5}$, particularly cooking, can be more significant than wood burner emissions (Trompetter & Davy 2019).

Complicating matters further, there is no standard methodology used to measure the impact of sources on indoor air quality. Key variables include the location of the monitor, the monitoring period, the type of monitor and the calibration or quality assurance procedures.

To estimate annual indoor exposure to $PM_{2.5}$ from wood burners we relied on the most relevant studies from our literature review based on the following criteria:

- Studies that utilised reasonably robust monitoring methods (gravimetric monitoring methods, or optical methods which have been collocated and calibrated with gravimetric methods).
- Studies that attempted to isolate the incremental contribution from indoor (and outdoor) combustion and attributed indoor $PM_{2.5}$ concentration to wood burner use.

From the findings, we developed exposure increments for five different types of wood burners as follows:

- **open fires**: non-enclosed appliances that burn wood or coal to heat a home
- **older (non-NES compliant)** [8] **wood burners**: appliances (typically older than 2005) that emit more than 1.5 grams of PM per kilogram of dry wood burnt
- **new (NES-compliant) wood burners**: appliances that emit less than 1.5 grams of PM per kilogram of dry wood burnt and have a thermal efficiency of no less than 65%
- **ultra-low emissions (ULEB) wood burners**: appliances that meet stricter emissions and efficiency standards and typically emit less than 1/3 of the PM emitted by a NES-compliant burner.
- **wood pellet burners**: appliances that burn uniformly-sized wood pellets with controlled feed rate and combustion conditions and emit less PM than NES-compliant wood burners.

We also needed to account for seasonal variation, due to fuel use. Robust estimates of monthly operation of wood burners across Aotearoa are reported in Wilton *et al* (2015).

For New Zealand houses with **open fires**, we estimate an
annual indoor $PM_{2.5}$ exposure increment of 17.5 μg/m^3
(with low to high values 5.9 - 35.3 μg/m^3)

[8] The Resource Management (National Environmental Standards for Air Quality) Regulations 2004 (also known as the NES) include a suite of ambient air quality standards and a wood burner emission limit.

The **central** estimate assumes:

- The winter increment is 48.2 μg/m^3 based on Vicente *et al* (2020)
- The increments for other seasons are the same as the average use of wood burners outside winter across Aotearoa based on Wilton *et al* (2015), i.e. spring = 18%, summer = 1% and autumn = 26% of winter values.

The **low** value assumes:

- The winter increment is 19.9 μg/m^3 based on the lower 95% CI in Vicente *et al* (2020)
- The increments for other seasons are the same as the lowest use of wood burners outside winter across New Zealand based on Wilton *et al* (2015), i.e. spring = 4%, summer = 0% and autumn = 14% of winter values.

The **high** value assumes:

- The winter increment is 77.6 μg/m^3 based on the upper 95% CI in Vicente *et al* (2020)
- The increments for other seasons are the same as the highest use of wood burners outside winter across Aotearoa based on Wilton *et al* (2015), i.e. spring = 33%, summer = 7% and autumn = 42% of winter values.

For New Zealand houses with **older (non-NES-compliant) wood burners**, we estimate an **annual indoor $PM_{2.5}$ exposure increment of 2.18 μg/m^3**
(with low to high values 0.74 - 4.78 μg/m^3)

The **central** estimate assumes:

- The winter increment is 6.0 μg/m^3 based on the average of Vicente *et al* (2020), Fleisch *et al* (2019) and Wyss *et al* (2016) (refer **Appendix A**)
- The increments for other seasons are the same as the average use of wood burners outside winter across Aotearoa based on Wilton *et al* (2015), i.e. spring = 18%, summer = 1% and autumn = 26% of winter values.

The **low** value assumes:

- The winter increment is 2.5 μg/m^3 based on the lower 95% CI in Vicente *et al* (2020)
- The increments for other seasons are the same as the lowest use of wood burners outside winter across New Zealand based on Wilton *et al* (2015), i.e. spring = 4%, summer = 0% and autumn = 14% of winter values.

The **high** value assumes:

- The winter increment is 10.5 μg/m^3 based on the upper 95% CI in Vicente *et al* (2020)
- The increments for other seasons are the same as the highest use of wood burners outside winter across Aotearoa based on Wilton *et al* (2015), i.e. spring = 33%, summer = 7% and autumn = 42% of winter values.

For New Zealand houses with **new (NES-compliant) wood burners**, we estimate an **annual indoor $PM_{2.5}$ exposure increment of 0.51 μg/m^3** (with low to high values 0.16 - 0.98 μg/m^3)

The **central** estimate assumes:

- The winter increment is 1.4 μg/m^3 based on the average of Chakraborty *et al* (2020) and Salthammer *et al* (2014)
- The increments for other seasons are the same as the average use of wood burners outside winter across Aotearoa based on Wilton *et al* (2015), i.e. spring = 18%, summer = 1% and autumn = 26% of winter values.

The **low** value assumes:

- The winter increment is 0.5 μg/m^3 based on the lower 95% CI in Chakraborty *et al* (2020)
- The increments for other seasons are the same as the lowest use of wood burners outside winter across New Zealand based on Wilton *et al* (2015), i.e. spring = 4%, summer = 0% and autumn = 14% of winter values.

The **high** value assumes:

- The winter increment is 2.15 μg/m^3 based on the upper 95% CI in Chakraborty *et al* (2020)
- The increments for other seasons are the same as the highest use of wood burners outside winter across Aotearoa based on Wilton *et al* (2015), i.e. spring = 33%, summer = 7% and autumn = 42% of winter values.

Low emission burners

For modern wood burners that are operated correctly, studies show that indoor exposure to PM is dominated by peaks that occur when the wood burner door is opened to load firewood (Chakraborty *et al* 2020, Salthammer *et al* 2014). This suggests that these emissions would be similar whether the burner is an ultra-low emissions burner of a NES compliant burner.

On this basis we would assume that the indoor exposure increment for ultra-low emission wood burners is the same as the increment for NES burners.

By contrast, wood pellet burners are fed automatically via external hoppers and the door is only opened regularly for cleaning/ash removal.

In the absence of any explicit research, it may be assumed that the indoor exposure increment for wood pellet burners is around one third of that for ultra-low emissions wood burners.

3.1.4 Summary of annual increments

Table 1 summarises the increments assumed in the exposure model.

Table 1: Annual incremental exposure attributed to each appliance type (with low high estimates)

Appliance	Indoor air pollution annual increment		
	Value (μg/m^3)	Low	High
$PM_{2.5}$			
Electric stove*	Reference case	Reference case	Reference case
Gas stove*	0	0	0
Wood burner type 1a (open fire)	17.5	5.9	35.3
Wood burner type 1b (non-NES)	2.18	0.74	4.78
Wood burner type 2 (NES-compliant)	0.509	0.160	0.978
Wood burner type 3 (ultra-low emissions)	0.509	0.160	0.978
Wood pellet burner	0.170	0.053	0.326
Gas unflued heater	0	0	0
NO_2			
Electric stove	0	0	0
Gas stove	2.88	0.30	7.43
Wood burner type 1a (open fire)	0	0	0
Wood burner type 1b (non-NES)	0	0	0
Wood burner type 2 (NES-compliant)	0	0	0
Wood burner type 3 (ultra-low emissions)	0	0	0
Wood pellet burner	0	0	0
Gas unflued heater	6.62	4.44	9.92

* Indoor $PM_{2.5}$ emissions from cooking are assumed to be the same in all scenarios

3.2 Modelled health impacts

The estimated annual increments, combined with exposure-response functions and household composition data, enabled us to determine the likely resultant mortality and morbidity (illness) health impacts (in terms of the number of cases) on a per appliance basis.

3.2.1 Health outcomes

We assessed the same health outcomes for indoor exposure to be consistent with those assessed in the HAPINZ 3.0 study for outdoor exposure (Kuschel *et al* 2022).

Kashtan *et al* (2024), Delgado-Saborit *et al* (2024) and Bennitt *et al* (2021) also estimated the health impacts of indoor air quality and assumed the same relationships between exposure and health impacts *outdoors* applied *indoors*. This supports our approach.

Table 2 compares the exposure response functions adopted in this study with those in the literature where available.

Table 2: Exposure response functions used in this study compared with the literature

Health outcome	Relative risk	Unit	Low/High
This study			
Annual $PM_{2.5}$			
Premature mortality for all adults (30+ years)	1.105	per 10 µg/m^3	[1.065 – 1.145]
or Years of life lost for all adults (30+ years)	1.105	per 10 µg/m^3	[1.065 – 1.145]
Cardiovascular hospitalisation for all ages	1.115	per 10 µg/m^3	[1.084 – 1.146]
Respiratory hospitalisation for all ages	1.07	per 10 µg/m^3	[1.021 – 1.112]
Restricted activity days for all ages	0.9	per 10 µg/m^3	[0.5 – 1.7]
Annual NO_2			
Premature mortality for all adults (30+ years)	1.097	per 10 µg/m^3	[1.074 – 1.12]
or Years of life lost for all adults (30+ years)	1.097	per 10 µg/m^3	[1.074 – 1.12]
Cardiovascular hospitalisation for all ages	1.047	per 10 µg/m^3	[1.031 – 1.064]
Respiratory hospitalisation for all ages	1.130	per 10 µg/m^3	[1.102 – 1.159]
Asthma/wheeze hospitalisations (0-18 years)	1.182	per 10 µg/m^3	[1.094 – 1.276]
Incidence of paediatric asthma (0-18 years)	1.050	per 4 µg/m^3	[1.02 – 1.07]
Kashtan *et al* (2024)			
Annual NO_2			
Incidence of paediatric asthma in the US	1.09	per 15 ppb	[0.91 – 1.31]

Health outcome	Relative risk	Unit	Low/High
Delgado-Saborit *et al* (2024)			
Annual NO_2			
Premature mortality for adults in EU (30+ years)	1.008	per 10 μg/m^3	[1.004 – 1.016]
or Years of life lost for adults in EU (30+ years)	1.02	per 10 μg/m^3	[1.01 – 1.04]
Incidence of paediatric asthma (0-19 years)	1.09	per 28 μg/m^3	[0.91 – 1.31]
Bennitt *et al* (2021)			
Annual $PM_{2.5}$			
Ischaemic heart disease, Stroke, Chronic obstructive pulmonary disease, Lung cancer, Type 2 diabetes (> 25 years)	Varying relative risks (RR) across different exposure ranges		
Lower respiratory infections for all ages			
Neonatal disorders			

3.2.2 Health incidence data

We utilised the same health incidence data (relating to total cases for each health outcome) as in HAPINZ 3.0, which are assigned to 2016 based on annual statistics averaged over 2015-2017.

Although this information is dated (nearly 10 years old), it can be divided by the relevant 2016 population to get a factor (rate) that can be applied per household in the later calculations. This approach is consistent with that recommended in the HAPINZ 3.0 Health Effects Model - Users' Guide (Sridhar *et al* 2022).

3.2.3 Household demographics

The per household impact of each appliance required assumptions to be made about typical Aotearoa household composition in terms of the number of adults and the number of children. Some of the health outcomes assessed in the model apply to all people (e.g. hospitalisations and restricted activity days), all adults (e.g. premature mortality), or only children (e.g. asthma prevalence) hence the need to make assumptions for the numbers of both.

For **adults** (people aged 30 or more years), we reviewed census data and assumed **a default of 2.0 per household**, with a low estimate of 1.0 and a high of 4.0. StatisticsNZ report that 57% of all New Zealand households in 2018 contained two adult persons, 17.3% had extended families and the balance had only single adults (StatsNZ 2021).

For **children** (people aged less than 18 years), we assumed **a default of 2.0 per household**, with a low estimate of 0 and a high of 4.0. StatsNZ report an average of 1.9 children per household (where present) and that 67% of households had no children present (StatsNZ 2021).

Although a newer census was undertaken in 2023, we were unable to find updated comparable values. Reviewing previous censuses, however, we found that there was little change in our key metrics between the 2013 and 2018 censuses and assumed the same between 2018 and 2023.

Based on the above assumptions, the number of people exposed to each outcome in the model for the default case varied between 2.0 (either only adults or only children) and 4.0 (for adults and children). Sensitivity analyses were undertaken for low and high estimates (see section 5.2).

3.3 Are the impacts of indoor air pollution additive?

One of the issues for this study is whether the impacts of indoor air pollution are a subset of the total effects of air pollution or if they are additional.

HAPINZ 3.0 estimated the *total* impacts of air pollution in New Zealand (in 2016) based on an epidemiological study (Hales *et al* 2021) that used cross-sectional analysis to understand how the differences in experienced ambient concentrations of air pollutants resulted in differences in health outcomes (after adjustment for confounding factors age, sex, ethnicity, income, education, smoking status and ambient temperature).

The HAPINZ 3.0 model does not explain all deaths etc. - only those that result from ambient (outdoor) air pollution. If indoor air pollutant concentrations are not correlated with outdoor air pollution but affect the number of deaths etc. then the effects of indoor air pollution will be additional.

Our literature review found no correlation between indoor and outdoor air pollution for NO_2 and $PM_{2.5}$. To the contrary, Aotearoa research shows that the primary source of indoor $PM_{2.5}$ was cooking (Trompetter & Davy 2019). Consequently, **we are confident there is no double counting in the method we have used to assess the effects of indoor air pollution by appliance.**

We note our method is further consistent with the approach adopted in other studies that have estimated the health impacts of indoor combustion (Kashtan *et al* 2024, Delgado-Saborit *et al* 2024 and Bennitt *et al* 2021).

3.4 Comparison with outdoor air pollution

For comparison, we also estimated the likely annual *outdoor* air pollution impacts resulting from the typical use of each appliance (where relevant) based on typical fuel use, published emission factors and updated damage costs. These are provided for context only and are subject to broad and mostly conservative assumptions as follows.

3.4.1 Gas stoves

One of the more notable findings of the literature review was the apparent ineffectiveness of extraction in removing oxides of nitrogen from the indoor environment – Delgado-Saborit *et al* (2024) found no significant association between measured NO_2 and extraction hoods, kitchen volume or ventilation. Accordingly, we have assumed that 100% of the emissions from gas stoves are discharged indoors.

As a check, we estimated annual gas use assuming an average gas stove uses 0.2 litres gas/hour (PlusGas 2025) and that households use it every day for cooking for one hour per day for a year.[9] This was converted to energy (TJ) based on the calorific value of New Zealand LPG (Barber & Stenning 2023) to match with emission factors for residential LPG combustion (GAPF 2019).

Annual emissions could then be estimated for nitrogen oxides (NO_X) rather than NO_2 to align with published damage costs for Aotearoa. Damage costs for NO_X were updated to 2025$ based on the updated mortality and morbidity values (refer Chapter 4). For New Zealand houses with gas stoves, this equates to an estimate of annual outdoor NO_X emissions per appliance of 0.0983 kg/year with an associated cost of only $62 per appliance in 2025$. We are comfortable therefore, that the assumption of zero discharges of NOx to outside air is reasonable and does not unduly affect overall estimates.

Outdoor $PM_{2.5}$ emissions from gas stoves were assumed to be zero.

3.4.2 Unflued gas heaters

There is likely negligible impact of unflued gas heater emissions on outdoor air pollution as there is no direct conduit (flue) to the outside. We assumed that 100% of the emissions are discharged indoors.

For Aotearoa houses with unflued gas heaters, we estimate annual outdoor NO_X emissions per appliance to be zero.

3.4.3 Wood burners

While wood burners do release a portion of their $PM_{2.5}$ emissions indoors, we assumed (for the modelling of outdoor impacts) that 100% of the $PM_{2.5}$ emissions were discharged outside.

We reviewed published home heating emissions inventories to estimate annual average fuel use across New Zealand. Fuel use differs by region across Aotearoa but we found the values surprisingly consistent. Typical annual fuel use for different wood burners from the *Canterbury Home Heating Emissions Database* ranges between 2.3 and 2.4 tonnes/year (ECan 2024). These estimates are likely to be similar in other parts of the South Island and some areas in the central North Island. However, they may potentially over-estimate usage in other areas. For example, the annual wood fuel usage for home heating in the Auckland region in 2016 was estimated at 1.7 tonnes/year (Metcalfe *et al* 2018).

We used the data underpinning a comprehensive evaluation of home heating emissions inventory and other sources by Wilton *et al* (2015) to estimate national average annual wood fuel use per household. This adjusted the values for fuel use and seasonality to give an average annual fuel use per wood burner of 2.474 tonnes/year.

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[9] This value is for typical Australia gas stove fuel consumption but considered reasonable for New Zealand gas stove use. Annual household gas consumption figures are available for Aotearoa but are not disaggregated by appliance so cannot be used.

The average fuel use per wood burner (regardless of type) was then multiplied by the relevant $PM_{2.5}$ emission factor per appliance from ECan (2024). This approach permits alignment with published damage costs for New Zealand.

Pellet burners were also assumed (for the purpose of the outdoor impacts modelling) to discharge 100% of their $PM_{2.5}$ emissions outside. However, less information is published for average annual pellet use in pellet burners. We relied on the *Canterbury Home Heating Emissions Database* which estimates annual pellet use at 1.463 tonnes/year (ECan 2024). The average pellet use per burner was then multiplied by the $PM_{2.5}$ emission factor per appliance from the same reference (ECan 2024).

This results in the following estimates of annual *outdoor* $PM_{2.5}$ emissions per appliance:

- Open fires: 20.81 kg/year
- non-NES wood burners: 13.20 kg/year
- NES-compliant wood burners: 9.60 kg/year
- Ultra-low emissions NES-compliant wood burners: 2.474 kg/year
- Pellet burners: 2.048 kg/year

Damage costs for $PM_{2.5}$ were updated to 2025$ based on the updated mortality and morbidity values (refer Chapter 4).

Outdoor NO_X emissions from all wood burners and pellet burners were assumed to be zero.

3.5 National estimates

Total Aotearoa impacts were estimated for both indoor and outdoor air pollution from the use of the assessed appliances by multiplying the per appliance values by the estimated numbers of appliances in New Zealand.

The numbers of wood and gas heating appliances were estimated from the *2023 Census of Population and Dwellings* (StatsNZ 2025a). The 2023 Census only reports wood burners as an overall total so this figure was disaggregated into the different types using the relative number of wood burner types from ECan (2024) with the splits shown in Table 3.

The number of gas stoves were estimated from *The Machine Count* (Rewiring NZ 2025).

Table 3: Estimated numbers of different appliances across Aotearoa New Zealand in 2024

Appliance	No of appliances across NZ	Assumed type split
*Electric stove**	*1,485,234*	
Gas stove	360,099	
Wood burner type 1a (open fire)	1,973	
Wood burner type 1b (non-NES)	101,236	*20%*
Wood burner type 2 (NES-compliant)	369,512	*73%*
Wood burner type 3 (ultra-low emissions)	35,433	*7%*
Wood pellet burner	15,532	
Gas unflued heater	44,645	

* Extrapolated from number total occupied dwellings in 2025 (from Census 2018 and 2023 data) less gas stoves

4. Economic analysis

Our health effects modelling utilised the mortality and morbidity outcomes from HAPINZ 3.0 (Kuschel *et al* 2022). While values for each outcome were published in HAPINZ 3.0, the costs were in 2019 dollars and reflected base data and assumptions made at the time for the purposes of that study.

To estimate the costs of health effects from indoor combustion, we needed to update the HAPINZ 3.0 values to current dollar values, taking account of inflation and any updates to the underlying data. The following sections outline our approach for updating individual values for the purposes of this study.

The associated costs of the health effects in 2025 dollars were then disaggregated to a per pollutant and per appliance basis to develop estimates for the fiscal (health system costs) and economic costs (wider costs of pollution on society).

For comparison, we also estimated the likely annual outdoor air pollution impacts resulting from the typical use of each appliance (where relevant) so also updated published damage costs.

4.1 Mortality impacts

The mortality impacts of air pollution in HAPINZ 3.0 were valued based on either premature deaths or years life lost (**YLL**). The former is matched to a Value of Statistical Life (**VoSL**) and the latter to a Value of Life Year (**VoLY**).

4.1.1 Value of Statistical Life (VoSL)

The air pollution VoSL used in HAPINZ 3.0 was based on the road safety VoSL used in the Ministry of Transport's regular report on the social cost of road crashes (Ministry of Transport 2020). The Ministry of Transport's VoSL was originally calculated in 1991 and inflated each year to $4,527,300 in 2019$.

This national estimate of VoSL was updated to $12,500,000 in 2021$ following a study for Waka Kotahi published in 2023 that employed a new survey and a new methodology (Denne *et al* 2023). This 2021$ value has subsequently been adopted by Waka Kotahi in the *Monetised Benefits and Costs Manual* (**MBCM**) (NZTA 2024), by the Ministry of Transport in the latest *Social cost of road crashes and injuries* report and spreadsheet[10] and also by Treasury in their CBAx excel model (Treasury 2024).[11]

Different agencies employ different approaches to cost inflation from a base year of 2021 and these produce different results as shown in Table 4. Waka Kotahi only produces values in 2021 dollars, the Ministry of Transport inflates values using an index based on average hourly earnings whereas Treasury's CBAx tool uses a Gross Domestic Product (**GDP**) inflator. CBAx includes the

[10] https://www.transport.govt.nz/area-of-interest/safety/social-cost-of-road-crashes-and-injuries

[11] CBAx is a spreadsheet model that contains a database of values to help NZ government agencies monetise impacts and do cost benefit analysis. https://www.treasury.govt.nz/information-and-services/public-sector-leadership/investment-management/investment-planning/treasurys-cbax-tool

most recent estimate of GDP (current as of March 2025), whereas the Ministry of Transport uses an index that is updated only at the end of June each year.

Table 4: Estimates of VoSL using Ministry of Transport and Treasury inflators

Year	NZTA MBCM	MoT	CBAx
2021	$12,500,000	$12,500,000	$12,500,000
2022		$13,294,735	$13,261,431
2023		$14,215,334	$14,416,024
2024		$14,930,955	$15,053,763
2025			$15,691,757[12]

For this analysis we used Treasury's CBAx approach. This is the most appropriate because it produces values suitable for all applications of VoSL rather than solely transport-related ones which are the focus of the Ministry of Transport's estimates.

To provide low and high estimates, we used the range of estimates provided in the Waka Kotahi study to produce the updated VoSL (Denne *et al* 2023): $8.1 to $16.9 million in 2021 dollars. These are used to provide percentage differences from the central estimate as shown in Table 5.

Table 5: VoSL values for sensitivity analyses

	Unit	Central	Low	High
VoSL 2021$	$/premature death	$12,500,000	$8,100,000	$16,900,000
VoSL 2025$	$/premature death	$15,691,757	$10,168,258	$21,215,255
% difference			-35%	+35%

4.1.2 Value of a Life Year (VoLY)

There are several ways to estimate the value of a life year (**VoLY**) including those that measure it directly (e.g. via survey) and those that derive it from VoSL (Chilton *et al* 2020). The latter approach was used in HAPINZ 3.0 in the absence of NZ studies to measure VoLY directly. We adopted the approach from HAPINZ 3.0, which uses the following formula from Aldy & Viscusi (2008).

$$\mathbf{VoLY} = \frac{\mathbf{r\,.\,VoSL}}{\mathbf{1 - (1 + r)^{-\mathit{L}}}} \qquad \text{Equation 1}$$

Where: r = discount rate, L = life expectancy (years)

[12] The value provided here is different from that included in the November 2024 CBAx Tool Spreadsheet Model. It corrects for an error in which the $12,500,000 VoSL was assumed to be in 2022 values rather than 2021. The CBAx model estimates a value of $14,790,784 rather than the corrected value of $15,691,757 using GDP values of $342,899 million in 2021 and $430,568 million for 2025.

HAPINZ 3.0 used the Treasury's then recommended discount rate for policy analysis of 5% and a life expectancy of 40 years, which is the weighted average life expectancy of road transport fatalities for which the VoSL was developed.

For a 2019 VoSL of $4,527,300, Equation 1 provides a VoLY of $263,843. This value can be updated using one of two ways as follows:

1. Recalculating the value using updated estimates of the weighted average life expectancy and the discount rate, if applicable.
2. Using the same updating approach as for VoSL. This would apply the same GDP-based multiplier as used for the VoSL in CBAx.

The second approach is probably the only one that is consistent with the assumptions underlying the HAPINZ 3.0 numbers.

The VoSL has been calculated as a single value for one reduced road fatality of the average age for such a death. It represents the value of saving a statistical life, e.g. the value of reducing the risk of a fatality by 0.1% for 1,000 people.

Converting the VoSL to a VoLY using Equation 1 effectively assumes that the VoSL represents the *present* value of reducing the risk of fatality in each of the future years of expected life. It assumes that people expressing a VoSL are implicitly making the calculation shown in Equation 2.

$$\mathbf{VoSL} = \frac{\mathbf{VoLY} \times (\mathbf{1} - (\mathbf{1} + \mathbf{r})^{-L})}{\mathbf{r}} \qquad \text{Equation 2}$$

The inputs to the VoLY estimate in HAPINZ (5% discount rate and 40 years of life) are assumed to be private estimates reflected in the expressed willingness to pay to reduce mortality risk. These would not change with the update to the Treasury recommended discount rate, for example, so we assume these assumptions still hold. The resulting VoLY is then the same as estimated by using the same multiplier used to update the VoSL (i.e. 3.466), thereby resulting in an updated central value to $914,488 (Table 6).

For sensitivity analysis, we used different approaches for the low and high values. For the low value we used current GDP per capita (StatsNZ 2025b), consistent with the approach in HAPINZ 3.0. This assumes that affordability sets the maximum amount to spend on saving a life as a useful minimum value for the VoLY.

For the high VoLY, the same percentage difference as used for VoSL (+35%) was used.

Table 6: VoLY values for sensitivity analyses

	Unit	Central	Low	High
VoLY 2025$	$/year life lost	$914,488	$80,016	$1,236,387
Basis			GDP per capita	Central value +35%

4.2 Morbidity impacts

Estimates of the costs of morbidity impacts were made in HAPINZ 3.0 using estimates of the number of days in hospital resulting from the identified health effects, hospitalisation costs, lost income (as a proxy for lost economic output) and quality of life effects. The main input assumptions are still regarded as reasonable, such that the only changes to the values are to take account of inflation.

The morbidity impacts from HAPINZ 3.0 were updated from 2019 dollars to 2025 dollars using the CBAx GDP inflator (i.e. a multiplier of 1.388). To provide low and high estimates, we used the low and high estimates published in HAPINZ 3.0 and applied the same CBAx GDP inflator as for the central value. The resulting morbidity values are shown in Table 7.

Table 7: Morbidity values for sensitivity analyses

Health outcome	Unit	Central	Low	High
Cardiovascular hospitalisation	$/admission	$50,902	$15,006	$657,056
Respiratory hospital admission	$/admission	$44,075	$8,178	$642,446
Restricted activity day	$/RAD	$124	$68	$174
Asthma hospitalisation	$/case	$2,529	$1,265	$3,794
Asthma prevalence	$/case	$178	$89	$267

4.3 Economic and fiscal costs per case

The total impacts per case are summarised in Table 8.

Table 8: Updated central values per case in 2025$

Health outcome	Unit	HAPINZ 3.0 value (2019$)	Multiplier[1]	Updated value (2025$)
Mortality – economic costs				
Value of Statistical Life (VoSL)	$/premature death	$4,527,300	3.466	$15,691,757
Value of Life Year lost (VoLY)	$/years of life lost	$263,843	3.466	$914,488
Morbidity - fiscal costs				
Cardiovascular hospitalisation	$/admission	$36,666	1.388	$50,902
Respiratory hospital admission	$/admission	$31,748	1.388	$44,075
Restricted activity day	$/RAD	$89	1.388	$124
Asthma hospitalisation	$/case	$1,822	1.388	$2,529
Asthma prevalence	$/case	$128	1.388	$178

[1] The multipliers are rounded in the table. They are derived from the following values of GDP (as found in CBAx) for 2019, 2021 and 2025 respectively (in $million): $310,149; $342,989; $430,568

The costs are split into economic and fiscal costs, as requested by EECA, using simplifying assumptions.

- The **mortality impacts** were derived from a study that measured individuals' willingness to pay to avoid fatalities. The study did not identify the reasons for the expressed willingness to pay and this is likely to include factors that are readily monetarised, including costs falling on the health sector, and those that are not, such as the costs of grief for those still alive. For simplicity, we have defined all of these costs as economic, in contrast to fiscal costs which are those falling on the government.

- The **morbidity impacts** are assumed to result in costs that fall on the government in the form of the costs of hospitalisations. Although the input assumptions include estimates of costs from lost income to households (which is assumed to equal the costs falling on New Zealand from reduced economic output), this element is not separated out here, partly because of the inclusion of some fiscal costs within the mortality costs estimates.

The multiplier for mortality impacts includes the update to the VoSL (to 2021$) and the GDP inflator to convert to 2025$ values. The multiplier for morbidity includes only the GDP inflator.

4.4 Marginal impacts

HAPINZ 3.0 noted that estimates of the *marginal* impacts of pollutant reduction[13] should include a lag function because the effects of pollutant concentrations on respiratory health are largely cumulative over time and will not repair quickly following cessation of pollution. A lag function can be used to quantify how much of the total benefit of pollution reduction is assumed to occur in each year following cessation (of the pollution).

Cessation lag assumptions were taken from the US Environmental Protection Agency (**USEPA**) because these have been widely adopted internationally (USEPA 2004). These assume:

- 30% of the effect occurs in the first year
- 12.5% in each of years 2-5
- 20% are spread over years 6-20 (i.e. 1.33% per year).

This is consistent with the recommendations of the World Health Organization (**WHO**) on methods to assess the health risks of $PM_{2.5}$ and for NOx (WHO 2013) and recent economic studies on the benefits of reducing pollution (e.g. Walton *et al* 2025, Edwards *et al* 2024).

Following HAPINZ 3.0, the lagged effects would only apply to VoSL and VoLY as morbidity effects are assumed to be acute. For a 2% discount rate (the Treasury's recommended rate for non-commercial proposals),[14] the resulting impact is equal to 93.4% of the unlagged effect when estimated by discounting the percentage of the total benefit over 20 years (Figure 2 shows the effects of discount rate on the total lagged benefit).

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[13] By marginal, we mean the effects of a (small) change in pollutant concentrations rather than the total effects relative to a counterfactual with no pollution.
[14] https://www.treasury.govt.nz/information-and-services/public-sector-leadership/guidance/reporting-financial/discount-rates

This assumes that the removal of a polluting appliance will result in the householder living in a less polluted atmosphere for the next twenty years. In practice a householder may not live at the same address for that long but will move to another house that may or may not have a polluting device. At the same time, another person will move to the house from which the polluting appliance has been removed and will experience the health benefits. This type of dynamic situation is modelled by Edwards *et al* (2024). We do not have the data to do this here and have made the following simplifying assumptions accordingly.

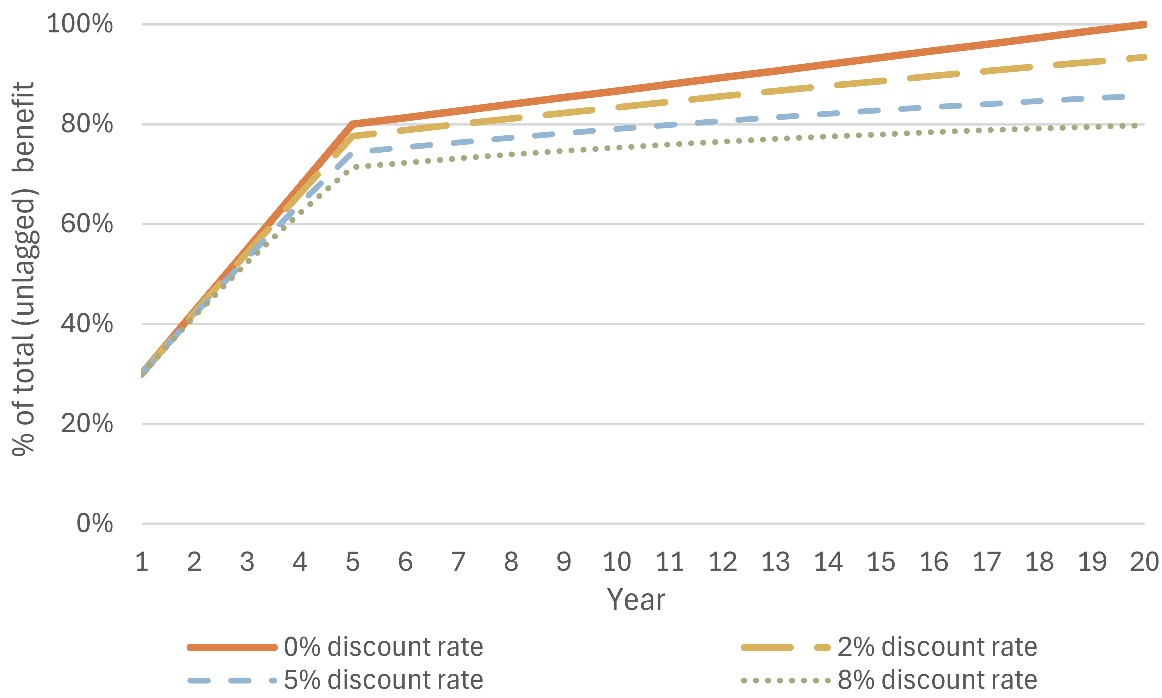

Figure 2: Impacts of discount rate on cumulative lagged benefit at different discount rates

When a new polluting appliance is added to a property an *inception lag* is assumed to be the inverse of the cessation lag, consistent with Edwards *et al* (2024). This results in the same distribution of costs as the distribution of benefits using the cessation lag assumptions. The circumstances under which the lagged or unlagged values are used are as shown in Table 9.

Table 9: Circumstances under which lagged or unlagged values are used

Scenario	**Approach**	**% of unlagged effect at discount rate**		
		@ 2%	**@ 5%**	**@ 8%**
Current total costs of indoor air pollution	Unlagged	100%	100%	100%
Benefits of removing an existing polluting appliance	Lagged	93.4%	85.7%	79.8%
Costs of adding a new polluting device	Lagged	93.4%	85.7%	79.8%

For analysis of a change in emissions (either as an increase or decrease), as opposed to total costs, Table 10 shows the estimates of VoSL and VoLY using the USEPA lag structure and a 2% discount rate.

Table 10: VoSL and VoLY values for policy analysis using the USEPA lag structure and a 2% discount rate

Value	Central unlagged	Central lagged	Low lagged	High lagged
VoSL	$15,691,757	$14,659,907	$9,499,620	$19,820,194
VoLY	$914,488	$854,353	$74,754	$1,155,086

4.5 Damage costs (comparative analysis only)

For comparative purposes we have also estimated the *outdoor* air pollution impacts from the typical use of each appliance (where relevant).

For this, we revised the published damage costs (NZTA 2024) using the updated mortality and morbidity values from section 4.1 and section 4.2, respectively, and using the same assumptions for the split between fiscal and economic costs.

The revised values per pollutant are shown in Table 11.

Table 11: Updated damage costs per pollutant in 2025$

Damage costs	Unit	National average	Rural	Urban
Mortality - economic costs				
$PM_{2.5}$	$/tonne	$1,258,422	$79,235	$2,049,642
NO_X	$/tonne	$625,549	$37,966	$1,679,682
Morbidity - fiscal costs				
$PM_{2.5}$	$/tonne	$27,025	$2,243	$43,653
NO_X	$/tonne	$7,715	$475	$20,704

Note: The national average damage costs are the central estimates for overall exposure to outdoor air pollution in New Zealand. The Rural damage costs represent the low estimates and the Urban damage costs the high estimates (because a given tonne of emission affects more people in more populated areas).

5. Results

The modelled health effects were disaggregated to a per pollutant and per appliance basis to develop estimates for the fiscal (health system costs) and economic costs (wider costs of pollution on society). The overall findings summarise the results for the central (best estimate) values. Also provided are the results of sensitivity testing on key parameters to indicate the potential uncertainty for each result.

5.1 Overall findings

Table 12 presents the annual indoor air pollution costs in 2025$ estimated for each appliance assessed, based on the central (best estimate) values.

Table 12: Estimated annual indoor air pollution costs per appliance

Appliance	**Annual indoor air pollution ($PM_{2.5}$ + NO_2) costs (2025$)**		
	Economic	Fiscal	Total
Electric stove	$0	$0	$0
Gas stove	$9,058	$129	$9,187
Wood burner type 1a (open fire)	$51,911	$1,490	$53,401
Wood burner type 1b (non-NES)	$7,480	$198	$7,679
Wood burner type 2 (NES-compliant)	$1,776	$47	$1,823
Wood burner type 3 (ultra-low emissions)	$1,776	$47	$1,823
Wood pellet burner	$594	$16	$610
Gas unflued heater	$20,148	$286	$20,433

For comparison, Table 13 presents the outdoor air pollution costs in 2025$ estimated for each appliance assessed (where relevant), based on the central (best estimate) values and assuming Aotearoa average outdoor exposure.

Table 13: Estimated annual outdoor air pollution costs per appliance

Appliance	Annual outdoor air pollution ($PM_{2.5}$ + NO_X) costs (2025$)		
	Economic	Fiscal	Total
Electric stove	$0	$0	$0
Gas stove	$0	$0	$0
Wood burner type 1a (open fire)	$26,188	$562	$26,751
Wood burner type 1b (non-NES)	$16,612	$357	$16,968
Wood burner type 2 (NES-compliant)	$12,081	$259	$12,341
Wood burner type 3 (ultra-low emissions)	$3,114	$67	$3,181
Wood pellet burner	$2,578	$55	$2,633
Gas unflued heater	$0	$0	$0

Table 14 presents the estimated annual indoor air pollution costs as a fraction of the total (indoor plus outdoor) costs. This suggests that indoor air pollution is estimated to be responsible for the bulk of the estimated costs of total air pollution (indoor and outdoor) for gas and open fires when considered on a per appliance basis.

Table 14: Indoor air pollution as a fraction of the total (indoor+outdoor) annual costs per appliance

Appliance	Indoor fraction of total (indoor + outdoor) annual air pollution costs		
	Economic	Fiscal	Total
Electric stove	n/a	n/a	n/a
Gas stove	100%	100%	100%
Wood burner type 1a (open fire)	66%	73%	67%
Wood burner type 1b (non-NES)	31%	36%	31%
Wood burner type 2 (NES-compliant)	13%	15%	13%
Wood burner type 3 (ultra-low emissions)	36%	41%	36%
Wood pellet burner	19%	22%	19%
Gas unflued heater	100%	100%	100%

Table 15 sums the annual impacts of indoor air pollution to a national level for each appliance type. Of interest the modelled impacts of indoor air pollution are a similar order of magnitude to the modelled impacts of outdoor air pollution. This is consistent with the WHO Global Burden of Disease study which estimated similar annual impacts from household air pollution (3.2 million deaths globally) and outdoor air pollution (3.5 million deaths globally, WHO 2024).

Table 15: Total estimated annual indoor air pollution impacts for Aotearoa New Zealand

Cases of modelled health outcome	**Gas stoves**	**Wood burners**[1]	**Unflued gas heaters**
(Number of appliances)	*(360,099)*	*(523,686)*	*(44,645)*
Premature mortality (>30 years)	208	101	57
Cardiovascular hospitalisations (all ages)	236	255	66
Respiratory hospitalisations (all ages)	775	191	211
Restricted activity days (all ages)	-	167,204	-
Asthma prevalence (<18 years)	3,230	-	882

Notes:

1. All wood burner types (open fire + non-NES + NES compliant + ultralow emission + pellet burners). For further details of underlying assumptions and calculations refer body of report.

Table 16 presents estimated economic and fiscal costs for these impacts summed to the national level. These are considerable amounts – the values in Table 16 are in millions of dollars.

Table 16: Total estimated annual indoor air pollution costs for Aotearoa New Zealand

Appliance	**Total annual indoor air pollution ($PM_{2.5}$ + NO_X) costs (2025$M)**		
	Economic	Fiscal	Total
Electric stove	$0	$0	$0
Gas stove	$3,262	$47	$3,308
Wood burner type 1a (open fire)	$102	$3	$105
Wood burner type 1b (non-NES)	$757	$20	$777
Wood burner type 2 (NES-compliant)	$656	$17	$673
Wood burner type 3 (ultra-low emissions)	$63	$2	$65
Wood pellet burner	$9.2	$0.2	$9
Gas unflued heater	$900	$13	$912
National Total 2025$M	$5,749	$102	$5,851

Table 17 provides a comparison of the total annual indoor air pollution costs with estimated costs for total outdoor air pollution. It should be noted that the indoor air pollution cost estimates are based on assumed exposure and published response functions, whereas the outdoor air pollution cost estimates are based on assumed appliance use and published damage costs.

Table 17: Total estimated annual indoor and outdoor air pollution costs for Aotearoa New Zealand

Appliance	**Number**	**Total annual air pollution ($PM_{2.5}$ + NO_X) costs in 2025$M**[1]	
		Indoor	Outdoor
Electric stove[2]	*1,485,234*	*Reference case*	*Reference case*
Gas stove	360,099	$3,308	$0
Wood burner type 1a (open fire)	1,973	$105	$53
Wood burner type 1b (non-NES)	101,236	$777	$1,718
Wood burner type 2 (NES-compliant)	369,512	$673	$4,560
Wood burner type 3 (ultra-low emissions)	35,433	$65	$113
Wood pellet burner	15,532	$9	$41
Gas unflued heater	44,645	$912	$0
National Total 2025$M		$5,851	$6,484

Notes:

1. Estimates assume wood burner or unflued gas heater is main heating source and includes health outcomes specified in HAPINZ 3.0 (only, refer Table ES-2). For further details of underlying assumptions and calculations refer body of report.
2. Extrapolated from number total occupied dwellings in 2025 (from Census 2018 and 2023 data) less gas stoves

5.2 Sensitivity testing

5.2.1 Effect of annual exposure increment

Table 18 shows the effect on estimated total annual indoor air pollution costs of the low/high bounds for the annual indoor pollution exposure increments versus the central estimate.

The low/high bounds for the annual exposure increments are largely based on the wide confidence intervals from the various studies reviewed and reflect the heterogeneity in the literature.

As can be seen, the uncertainty is particularly high for gas stoves. However, even for the other indoor combustion appliances, the costs range from around 1/3 to twice the central estimates. Regardless of the assumptions the indoor air pollution costs per household are appreciable.

Table 18: Effect of exposure increments on the total annual indoor air pollution costs per appliance

Appliance	**Total annual indoor air pollution costs (2025$)**			**% Diff**	
	Central	Low	High	*Low*	*High*
Electric stove	$0	$0	$0	*n/a*	*n/a*
Gas stove	$9,187	$983	$22,748	*11%*	*248%*
Wood burner type 1a (open fire)	$53,401	$19,980	$93,106	*37%*	*174%*
Wood burner type 1b (non-NES)	$7,679	$2,643	$16,374	*34%*	*213%*
Wood burner type 2 (NES)	$1,823	$574	$3,483	*32%*	*191%*
Wood burner type 3 (ultra-low)	$1,823	$574	$3,483	*32%*	*191%*
Wood pellet burner	$610	$192	$1,169	*31%*	*192%*
Gas unflued heater	$20,433	$13,987	$29,713	*68%*	*145%*

5.2.2 Effect of household composition

Table 19 shows the effect on estimated total annual indoor air pollution costs of the low/high bounds for the household composition versus the central estimate.

Table 19: Effect of household composition on the total annual indoor air pollution costs per appliance

Appliance	**Total annual indoor air pollution costs (2025$)**			**% Diff**	
	Central	Low	High	*Low*	*High*
Electric stove	$0	$0	$0	*n/a*	*n/a*
Gas stove	$9,187	$4,561	$18,373	*50%*	*200%*
Wood burner type 1a (open fire)	$53,401	$26,328	$106,803	*49%*	*200%*
Wood burner type 1b (non-NES)	$7,679	$3,790	$15,357	*49%*	*200%*
Wood burner type 2 (NES)	$1,823	$900	$3,645	*49%*	*200%*
Wood burner type 3 (ultra-low)	$1,823	$900	$3,645	*49%*	*200%*
Wood pellet burner	$610	$301	$1,219	*49%*	*200%*
Gas unflued heater	$20,433	$10,145	$40,867	*50%*	*200%*

The bounds for the household composition assume:

- low = a single adult, no children household for the low value
- high = a four adult (extended family), four children household
- central = a two adult, two children household.

Not surprisingly, the more people exposed to the indoor combustion emissions the higher the indoor air pollution costs. The costs are essentially (but not quite) proportional to the number of people exposed as some of the health outcomes modelled affect adults exclusively and vice versa.

Care needs to be taken in using these estimates (and/or the associated model) to understand the underlying assumptions. The health impacts are estimated from population-level epidemiological studies. It is important to note that this study considers only the health outcomes specified in the Health and Air Pollution in New Zealand 3.0 study (Kuschel *et al* 2022). It will become rapidly apparent that children do not show up in the economic cost estimates. This is an artefact of the selected health outcomes modelled in the HAPINZ 3.0 study (mortality estimates are calculated for adults aged > 30 years only). This does not mean that these effects do not occur, only that they are not modelled. This means that these identified impacts are an underestimate of the actual impacts of air pollution more broadly on society.

5.3 Conclusions

Gas appliances (both gas stoves and unflued gas heaters) and open fires result in the highest indoor air pollution costs per household.

The indoor air pollution costs are particularly sensitive to the annual exposure increments as the low/high bounds are very wide, being based on the confidence intervals from the various studies reviewed. The uncertainty is particularly high for gas stoves. However, even for the other indoor combustion appliances, the costs range from around 1/3 to twice the central estimates. This reflects the complexity and variability of indoor air pollution.

Regardless of the assumptions and uncertainty, the indoor air pollution costs per household are appreciable for all indoor combustion appliances assessed – even at the lower bounds.

Not surprisingly, the more people exposed to the indoor combustion emissions the higher the indoor air pollution costs. When considered at the national level, the impacts and costs of both indoor and outdoor air pollution by indoor combustion appliance are significant and, in many cases, comparable. This finding is consistent with the WHO Global Burden of Disease study which estimated similar annual impacts from household air pollution (3.2 million deaths globally) and outdoor air pollution (3.5 million deaths globally, WHO 2024).

One of the issues for this study is whether the impacts of indoor air pollution are a subset of the total effects of air pollution or if they are additional. Our literature review found no correlation between indoor and outdoor air pollution for NO_2 and $PM_{2.5}$. Consequently, we are confident there is no double counting in the method we have used to assess the effects of indoor air pollution by appliance.

The results of this study will give EECA the ability to better estimate the impacts of gas stoves, wood burners and unflued gas heaters, and to inform future policy settings for wood burners in residential settings.

Confidence in the modelling could be improved in future by undertaking Aotearoa measurements of indoor $PM_{2.5}$ concentrations in houses with different types of burners (for example, new NES-compliant wood burners, ultra-low emission wood burners and pellet burners). If considered, this research should adopt robust measurement techniques to ensure repeatability of results.

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Appendix A – Studies used to develop exposure increments

A.1 Indoor NO_2 exposure increments for gas stoves

Gillespie-Bennett *et al* (2008)

Method:

- Measured NO_2 concentration in 349 Aotearoa homes over 16 weeks during winter in 2005 and 2006 using passive diffusion tubes.
- Passive diffusion tubes were collocated with a continuous chemiluminescent analyser (reference method) for a period of six days in a subsample of 40 homes. This found that passive diffusion tubes read on average 32% higher than the chemiluminescence analyser.
- The study was focused on the impact of unflued gas heaters, but data on use of gas stoves for cooking were also collected.

Results:

- During winter 2006 the geometric mean indoor NO_2 concentration across 349 households was 11.4 µg/m^3. The mean outdoor NO_2 concentration was 7.4 µg/m^3.
- NO_2 levels were approximately 1.42 times higher in living rooms in households that had gas stove-tops (n=34) compared with those households which did not have gas stove-tops (n=314). (GMR = 1.42, 95% CI: 1.05-1.93, p=0.02).
- Based on these results, we estimate the concentration in houses without a gas stove was approximately 11.0 µg/m^3 and the concentration in houses with a gas stove was approximately 15.6 µg/m^3.
- This is a **wintertime NO_2 concentration increment** for houses with gas stoves of **4.6 µg/m^3** (95% CI approximately: 0.6 µg/m^3 - 9.9 µg/m^3).

Sun *et al* (2025)

Method:

- Measured NO_2 concentration in 344 Canadian homes 5 to 7 days in winter and summer using passive diffusion tubes.
- Monitoring was undertaken in the living or family room where participants typically spend most of their waking time.

Results:

- During **winter** the geometric mean indoor NO_2 concentration measured:
  - across houses with gas stoves was 29.83 µg/m^3
  - across houses without gas stoves was 10.27 µg/m^3
- During **summer** the geometric mean indoor NO_2 concentration measured:
  - across houses with gas stoves was 17.16 µg/m^3
  - across houses without gas stoves was 8.03 µg/m^3
- This is a **wintertime** NO_2 concentration increment for houses with gas stoves of **19.6 µg/m^3** and a summertime NO_2 concentration increment for houses with gas stoves of **9.6 µg/m^3**.
- Found no correlation with presence of a range hood (consistent with Jacobs *et al* 2023).

- 71% of homes had a range hood with 39% vented outside, but only 17% of respondents use them during gas cooking.

Kashtan *et al* (2024)

Method:

- Developed a model to estimate exposure in US homes based on emission rates (from tests of 82 burners), residence size and room layout, time spent with windows open, frequency of use, range hood efficiency and time spent in kitchen and different rooms.

Results:

- The study estimates that gas combustion in US stoves increases long term NO_2 exposure by 4 ppbv on average
- This is an **annual NO_2 concentration increment for houses with gas stoves of approximately 8 µg/m^3**

Jacobs *et al* (2023)

Method:

- Monitoring undertaken in about 250 households across seven countries in Europe.
- Using passive sampling tubes placed in the kitchen, living room, bedroom and outside. An active micro-sensor (not a reference method) was also installed in the kitchen to estimate short term peak concentrations.
- Monitoring was undertaken for a period of 13 days in winter.

Results:

- The average concentration of NO_2 was significantly higher in the kitchen and living room of houses with a gas stove in all countries except Romania.
- The average concentration of NO_2 in living rooms of all houses with electric cooking was 12.6 µg/m^3 (n=58). The average concentration of NO_2 in living rooms of all houses with gas stoves was 17.7 µg/m^3 (n=178).
- The **wintertime NO_2 concentration increment** for houses with gas stoves is **5.1 µg/m^3.**

Kornartit *et al* (2010)

Method:

- Monitoring undertaken in about 60 households in north London.
- Using passive sampling tubes
- Monitoring was undertaken for a period of 7 days in winter and 7 days in Summer.

Results:

- During **winter** the average indoor NO_2 concentration of NO_2 measured:
  - across houses with gas stoves was 13.7 ppb
  - across houses without gas stoves was 7.9 ppb
- During **summer** the geometric mean indoor NO_2 concentration measured:
  - across houses with gas stoves was 14.7 ppb
  - across houses without gas stoves was 13.1 ppb

- This is a **wintertime NO_2 concentration increment** for houses with gas stoves of about **11 μg/m³** and a **summertime NO_2 concentration increment** for houses with gas stoves of about **3 μg/m³**.

A.2 Indoor NO_2 exposure increments for unflued gas heaters

Gillespie-Bennett *et al* (2008)

Method:

- Measured NO_2 concentration in 349 New Zealand homes over 16 weeks during winter in 2005 and 2006 using passive diffusion tubes.
- Passive diffusion tubes were collocated with a continuous chemiluminescent analyser (reference method) for a period of six days in a subsample of 40 homes. This found that passive diffusion tubes read on average 32% higher than the chemiluminescence analyser.
- The study was focused on the impact of unflued gas heaters.

Results:

- During winter 2006 the geometric mean indoor NO_2 concentration across 349 households was 11.4 μg/m³. The mean outdoor NO_2 concentration was 7.4 μg/m³.
- NO_2 levels were approximately 3.35 times higher in living rooms in households that had unflued gas heaters (n=111) compared with those households which did not have unflued gas heaters. (GMR = 3.35, 95% CI: 2.83-3.96, p<0.001)
- Based on these results, we estimate the concentration in houses without an unflued gas heater was approximately 7.7 μg/m³ and the concentration in houses with an unflued gas heater was approximately 26 μg/m³.
- This is a **wintertime NO_2 concentration increment** for houses with unflued gas heaters of **18.2 μg/m³** (95% CI approximately: 15.0 μg/m³ – 21.8 μg/m³).

A.3 Indoor $PM_{2.5}$ exposure increments for wood burners

To estimate annual indoor exposure to $PM_{2.5}$ from wood burners we relied on the most relevant studies from our literature review based on the following criteria:

- Studies that utilised reasonably robust monitoring methods (gravimetric monitoring methods, or optical methods which have been collocated and calibrated with gravimetric methods).
- Studies that attempted to isolate the incremental contribution from indoor (and outdoor) combustion and attributed indoor $PM_{2.5}$ concentration to wood burner use.

We note there is a body of literature about the impact of wood burners on indoor air quality which has not been specifically included in our exposure assessment. This includes some large studies attempting to quantify the impact of wood burner interventions and mitigation measures on indoor air quality. These studies tend to compare indoor air quality before and after interventions, but don't isolate the impact of wood burners. We briefly discuss the findings and implications of these studies (refer section 2.4.4 on mitigation measures where helpful), but the results are not directly relevant for exposure assessment.

We also note that many indoor air quality studies use optical sensors to estimate particulate concentration. These sensors have advantages compared to standard air quality monitoring methods of being low cost, small, easy to deploy and quiet. They provide good information on relative concentrations, which is valuable for understanding spatial and temporal variation of pollution levels. However, these sensors do not provide a reliable estimate of absolute concentrations (the purpose of this study) unless they are collocated and calibrated with a standard method.

Trompetter & Davy (2019)

Method:

- Measured indoor and outdoor air quality for seven homes in the Wellington Region. Each home was intensively monitored for 15-20 days.
- Outdoor PM_{10} was measured with an E-BAM (a regulatory standard method).
- Indoor PM_{10} concentration was measured with a DustTrak, which was collocated with the E-BAM for a period of 1 week.
- 2 hourly PM_{10} samples were also collected indoors and outdoors. The samples were analysed to determine the composition of the particulate.
- The composition data was analysed to identify the sources of indoor particulate.
- The indoor monitors were located in the lounge area.
- The characteristics and frequency of usage of the wood burners in each house are not reported.

Results:

- The major sources identified were:
    - **Cooking emissions** resulting in large evening particulate matter concentration peaks
    - Leakage of particulate matter to indoor spaces from wood burners in those houses where such appliances were used for space heating
    - Re-entrained soil/dust generated indoors when homes are occupied
    - Infiltration of outdoor particulate matter sources, including wintertime wood burning, and
    - Marine aerosol (sea salt).
- The indoor concentration of PM_{10} was highly variable, ranging from 72 µg/m^3 (house 1) to 9 µg/m^3 (house 4).
- The relative contribution of each source to indoor PM_{10} concentrations is illustrated in Figure 10.5 from Trompetter & Davy (reproduced as follows). This shows that cooking is the dominant source of PM_{10} in all houses except house 6. The occupants of House 6 mostly had their evening meals away from home.
- Houses 3 and 6 did not have a wood burner, so the biomass combustion source is from infiltration of particulate from outdoors. House 6 was the dwelling with the highest natural ventilation rate with indoor concentrations essentially the same as outdoor.
- The concentration of indoor biomass PM_{10} in the lounge of homes with a wood burner ranged from 0.2 µg/m^3 to 2.9 µg/m^3 (average 1.5 µg/m^3).

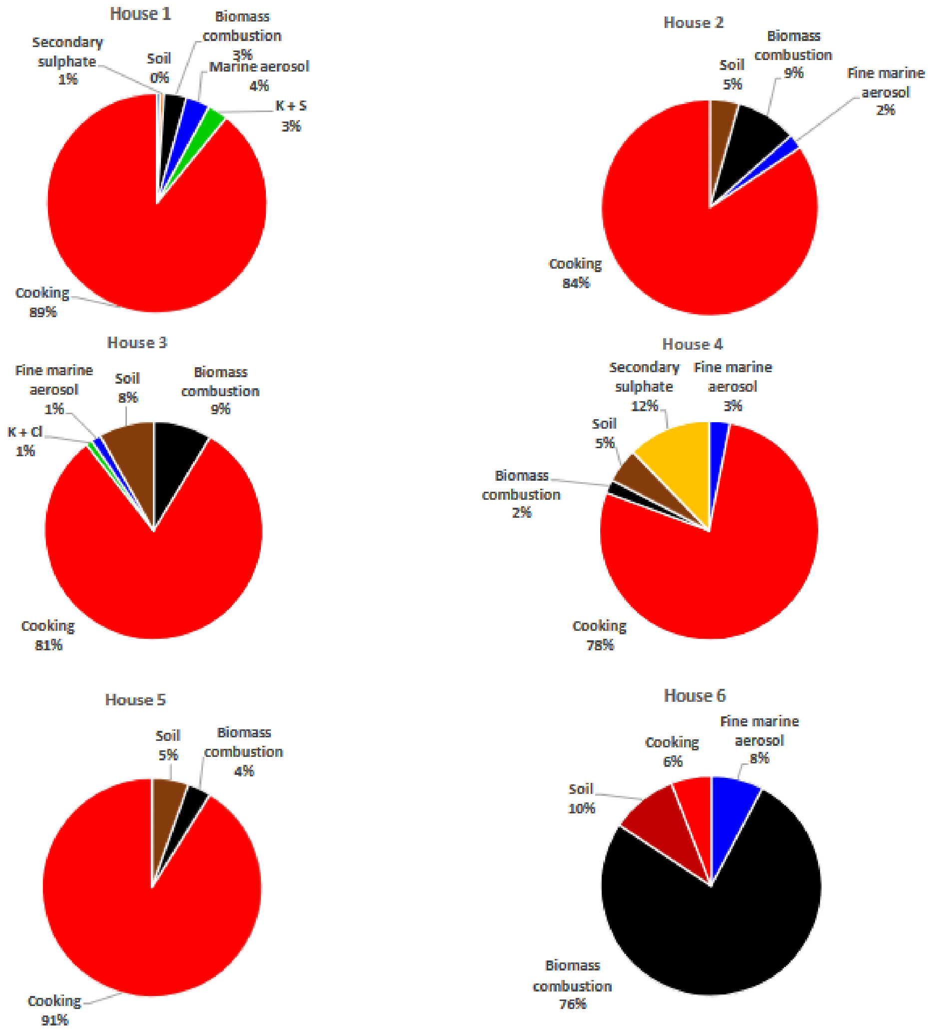

Figure 10.5 Relative source contributions to PM_{10} concentrations inside houses 1 to 6 monitored in this study.Houses 1, 2, 4 and 5 all used wood burners for space heating.

- The relative contribution of indoor and outdoor generated biomass to the indoor concentration is not estimated, however the report states that indoor concentrations tended to be dominated by the use of the wood burner in that house:

> *For those houses (H1, H2, H4, H5) that had wood burners used for space heating, it was found that the combustion emissions from the appliances leaked into the indoor room space and that indoor concentrations were unrelated to outdoor concentrations due to differing concentration and activity profiles. That is,* ***outdoor concentrations were most likely to reflect all wood burning activity in the neighbourhood and the prevailing weather conditions, whereas indoor concentrations tended to be dominated by the use of the wood burner in that house****. Figure 10.4 presents a time series plot for the indoor and outdoor wood burner source contributions showing the differing concentration patterns for indoor and outdoor biomass contributions.*

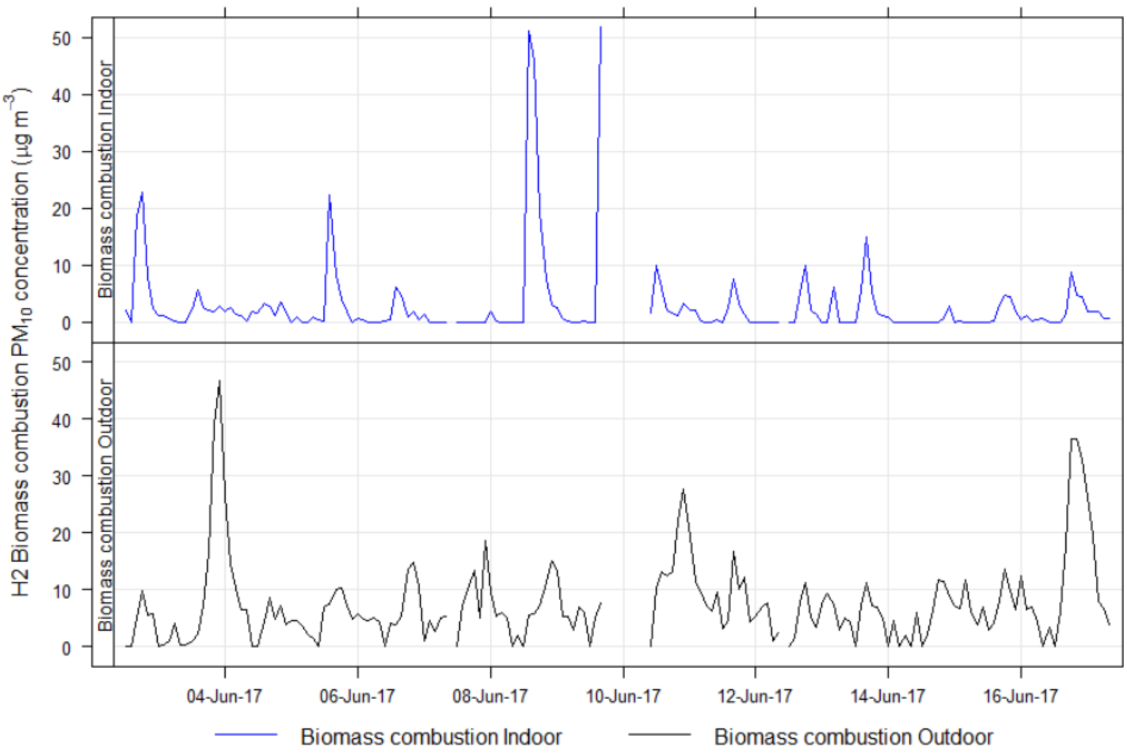

Figure 10.4 PM_{10} concentrations from Biomass combustion emissions inside and outside House 2

Wyss *et al* (2016)

Method:

- Measured $PM_{2.5}$ in 36 households in Norway for one week
- Using MicroPEM light scattering nephelometers, fitted with Teflon filters to calibrate the data based on gravimetric analysis. The air flow rate was set at 0.5 litres/minute.
- The monitors were placed in the main living space of homes.
- Over the 7-day monitoring period, wood burners were used for a median of 21.5 hours across households using burners.
- Wood burners included a mix of older (manufactured before 1997) and newer burners.

Results:

- Adjusting for ambient concentrations, the study found mean indoor $PM_{2.5}$ concentrations were higher in homes with older (manufactured before 1997) wood burners (20.2 µg/m^3, n=6) compared to homes without a wood burner (12.6 µg/m^3, n=22). However, homes with newer wood burners had levels similar to homes without a wood burner (11.9 µg/m^3, n=8).

Estimated increment Wyss *et al* (2016):

- We estimate an average wintertime $PM_{2.5}$ increment of **7.6 µg/m^3** for homes with older wood burners based on:
  - Average indoor $PM_{2.5}$ concentrations in homes with older wood burners (manufactured before 1997) = 20.2 µg/m^3
  - Average indoor $PM_{2.5}$ concentrations in homes without wood burners = 12.6 µg/m^3.
  - 7.6 µg/m^3 = 20.2 µg/m^3 - 12.6 µg/m^3.

Fleisch *et al* (2019)

Method:

- Investigated $PM_{2.5}$ levels in 113 homes in New England, USA for periods of one week using micro-environmental gravimetric monitors with a flow rate of 1.8 litres/min.
- Monitors were placed in the room where the participant spends the most time, excluding the kitchen, and as far away from the wood burner as possible.
- Wood burners included a mix of older (more than 10 years old) and newer burners.

Results:

- Median household $PM_{2.5}$ was 6.65 μg/m^3.
- Analysis found 21% higher $PM_{2.5}$ levels in homes with a wood burner (n=41 homes) compared to homes without (n=75). However, 95% confidence intervals included the null (-10.6, 62.6).
- Homes with a newer (<10 years) wood burner (n=22) had similar levels to homes without a wood burner (n=75).
- Homes with older wood burners (n=11 homes) had 62% higher $PM_{2.5}$ levels compared to homes without a wood burner (n=75).
- This study also measured black carbon and other indicators of wood smoke pollution and concludes that $PM_{2.5}$ is not the best indicator of wood smoke pollution and its potential effects.
- Fleisch *et al* (2019) conclude that their results are broadly similar to previous studies in the USA which found PM concentrations to be 4% higher (24 homes, Sexton *et al* 1984), 5% higher and 36% higher (35 homes and 45 homes respectively, Leadearer *et al* 1994).

Estimated increment Fleisch *et al* (2019):

- We estimate a wintertime average $PM_{2.5}$ increment of **3.8 μg/m^3**, based on:
    - Median household $PM_{2.5}$ concentration was 6.65 μg/m^3 across all households in the study. We assume median concentration = average concentration.
    - $PM_{2.5}$ concentration was 62% higher in homes with an older wood burner (n=11) compared to homes without a wood burner (n= 75).
    - We calculate average indoor $PM_{2.5}$ concentrations in homes with older wood burners = 10 μg/m^3 and average indoor $PM_{2.5}$ concentrations in homes without older wood burners = 6.2 μg/m^3
    - 3.8 μg/m^3 = 10 μg/m^3 - 6.2 μg/m^3

Siponen et al (2019)

Method:

- Measured $PM_{2.5}$ in 37 homes in Finland for 22-hour time periods in the heating season using gravimetric monitors with an air flow rate of 4 litres/minute.
- The monitors were placed in the living room or other common space used by all residents of the house.

Results:

- The study found that cooking was the only activity associated with higher indoor $PM_{2.5}$.

- However, the study found that wood stove use was associated with higher levels of indoor $PM_{2.5}$ light absorption (a proxy for indoor black carbon levels) and higher personal exposure to $PM_{2.5}$ (measured with a device carried in a backpack).

Salthammer *et al* (2014)

Method:

- Three days of sampling in seven houses.
- Extensive monitoring, including 24-hour particulate concentration with a gravimetric sampler set at a flow rate of 2.3 m^3/hour (~33 litres/min).
- Compared indoor air quality when wood burners are lit with times when they are not lit.
- Monitors were placed around 2-3 m from the wood burners.
- Wood burners were modern low emission burners.

Results:

- The study found increased **particle number** when wood burners are in use, compared to the background levels for all burners.
- The results for 24-hour $PM_{2.5}$ concentration were more variable. In some cases, the concentration during firing was lower than the background concentration (when burners are not firing). On average, however, the $PM_{2.5}$ concentration during firing was higher (22.1 µg/m^3) when burners were lit compared to the period when they were not in use (13.7 µg/m^3).

Estimated increment Salthammer *et al* (2014):

- We estimate an average wintertime $PM_{2.5}$ increment of **1.4 µg/m^3** based on:
  - Average indoor $PM_{2.5}$ = 22.1 µg/m^3 when wood burners are lit
  - Average indoor $PM_{2.5}$ = 13.7 µg/m^3 when wood burners are not in use
  - We assume 28 hours wood burner use per week (mean of 4 hours per day)
  - This equates to an average $PM_{2.5}$ increment of 1.4 µg/m^3 across a week.

Chakraborty *et al* (2020)

Method:

- A UK study comparing $PM_{2.5}$ levels when stoves are lit with the period when they are not in use.
- Levels were measured in 20 houses with stoves over four weeks using low-cost optical sensors. The sensors were collocated with regulatory monitors for four weeks prior to the study and results were calibrated.
- Monitors were placed at a minimum distance of 3m from the wood burner, but in the same room.
- The mean duration of wood burner use was 4 hours per day
- Wood burners were modern low emission DEFRA approved burners.

Results:

- The study found that average indoor $PM_{2.5}$ is higher (12.21 µg/m^3) when stoves are lit compared to the period when they are not in use (4.12 µg/m^3).
- They also found high peak levels due to 'flooding' events associated with opening the stove door.

Estimated increment Chakraborty *et al* (2020):

- We estimate an average wintertime $PM_{2.5}$ increment of **1.3 ± 0.8 μg/m³** based on:
    - Average indoor $PM_{2.5}$ = 12.21 μg/m³ when wood burners are lit
    - Average indoor $PM_{2.5}$ = 4.12 μg/m³ when wood burners are not in use
    - The mean duration of wood burner use was 4 hours per day
    - This equates to an average $PM_{2.5}$ increment of 1.3 μg/m³ across a week.
- We estimate **high and low values** based on:
    - Calculated confidence interval of average indoor $PM_{2.5}$ when wood burners are lit, based on SD = 10.36, sample size = 20 (as reported in Chakaborty *et al* 2020), t at 19 degrees freedom = 2.093
    - For all scenarios, assume average indoor $PM_{2.5}$ = 4.12 μg/m³ when wood burners are not in use, and mean duration of wood burner use = 4 hours per day.

Vicente et al (2020)

Method:

- Undertook comprehensive monitoring of particulate and toxic air pollutants in two unoccupied houses during operation of an open fire and an enclosed wood burner.
- Monitoring was undertaken over 8 hours, on four separate occasions for the open fire and on three occasions for the wood burner.
- $PM_{2.5}$ was monitored indoors and outdoors with optical DustTrak instruments and high volume gravimetric instruments simultaneously.
- The indoor instruments were placed in the middle of the room.
- The wood burner was a low efficiency cast iron woodstove.

Results:

- The average PM_{10} concentrations (8-hr average) was 319 ± 173 μg/m³ in the room with the open fire and 78.5 ± 24 μg/m³ in the room with the wood burner.
- The tests found that PM_{10} concentration was approximately double initial background and outdoor PM_{10} over the 3 x 8-hour periods when the wood burner was in use (shown in Figure 2).

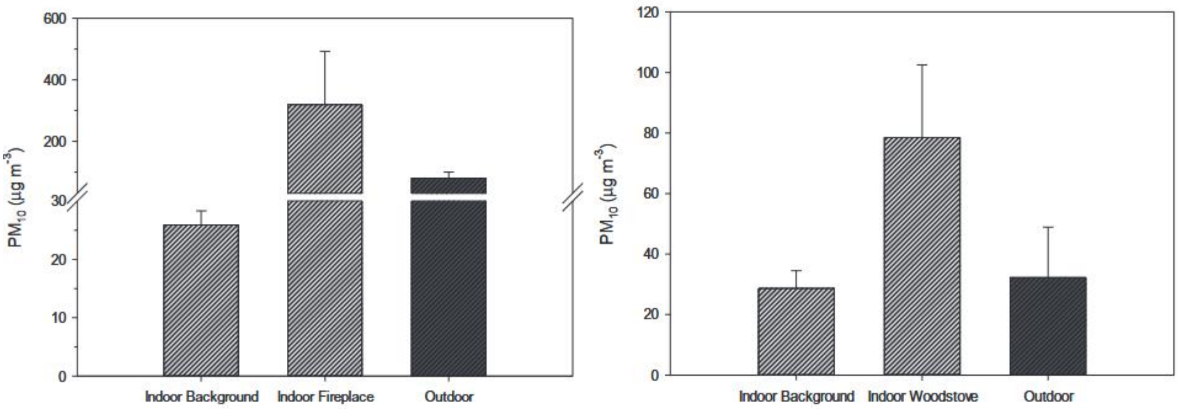

Fig. 2. PM_{10} concentrations indoors (while using combustion appliances and room background air) and outdoors.

Estimated increments Vicente *et al* (2020):

- We estimate an average wintertime $PM_{2.5}$ increment of **48.2 μg/m³ (19 .9– 77.6 μg/m³)** for the open fire based on:
  - Average indoor PM_{10} concentration = 319 ± 173 μg/m³ when open fire was lit.
  - PM_{10} concentration was approximately twelve times initial background and outdoor PM_{10} over the 3 x 8-hour periods when the wood burner was in use. We assume that average indoor PM_{10} concentration = 27 μg/m³ when wood burner is not in use.
  - We assume 28 hours open fire use per week (mean of 4 hours per day).
  - We assume all PM_{10} from the open fire is $PM_{2.5}$.
- We estimate an average wintertime $PM_{2.5}$ increment of **6.5 μg/m³ (2.5 – 10.5 μg/m³)** for the wood burner based on:
  - Average indoor PM_{10} concentration = 78.5 ± 24 μg/m³ when wood burner was lit.
  - PM_{10} concentration was approximately double initial background and outdoor PM_{10} over the 3 x 8-hour periods when the wood burner was in use. We assume that average indoor PM_{10} concentration = 39 μg/m³ when wood burner is not in use.
  - We assume 28 hours wood burner use per week (mean of 4 hours per day).
  - We assume all PM_{10} from wood burners is $PM_{2.5}$.
- We estimate **high** and **low values** based on:
  - The reported confidence interval for the average indoor PM_{10} concentration when wood burner or open fire was lit.
  - Average indoor PM_{10} concentration when wood burner is not in use is the same for all high, central and low scenarios, and duration of wood burner use = 28 hours per week (the same as the central estimate).

Wintertime $PM_{2.5}$ exposure increment for open fires

We estimate the indoor $PM_{2.5}$ exposure increment **during winter** for houses with **open fires** as **48.2 μg/m³ (19.9 – 77.6 μg/m³).**

This is based on results from Vicente *et al* (2020).

Wintertime $PM_{2.5}$ exposure increment for older (non-NES compliant) wood burners

We estimate the indoor $PM_{2.5}$ exposure increment **during winter** for houses with **older wood burners**, or wood burners not operated or maintained correctly as **6.0 μg/m³ (2.5 – 10.5 μg/m³).**

- 6.0 μg/m³ is the average of concentration increments we estimate based on results from Vicente *et al* (2020), Fleisch *et al* (2019) and Wyss *et al* (2016) as follows:
  - **Vicente *et al* (2020):** average wintertime $PM_{2.5}$ increment of **6.5 μg/m³** for a low efficiency cast iron wood stove.
  - **Fleisch *et al* (2019):** average wintertime $PM_{2.5}$ increment of **3.8 μg/m³** for wood burners more than 10 years old.
  - **Wyss *et al* (2016):** average wintertime $PM_{2.5}$ increment of **7.6 μg/m³** for wood burners manufactured before 1997.
- The estimated high and low values are based on the high and low values estimated based on results from **Vicente *et al* (2020).**

Wintertime $PM_{2.5}$ exposure increment for new (NES compliant) wood burners

We estimate the indoor $PM_{2.5}$ exposure increment **during winter** for houses with **new wood burners**, operated in accordance with manufacturer instructions as **1.4 µg/m³ (0.5 – 2.2 µg/m³)**

- **1.4 µg/m³** is the average of concentration increments we estimate based on results from Chakraborty *et al* (2020) and Salthammer *et al* (2014) as follows:
    - **Chakraborty *et al* (2020):** average wintertime $PM_{2.5}$ increment of **1.3 µg/m³** for low emission DEFRA approved wood burners.
    - **Salthammer *et al* (2014):** average wintertime $PM_{2.5}$ increment of **1.4 µg/m³** for wood burners of modern design operated with mature wood.
- The estimated high and low values are based on the high and low values estimated based on results from **Vicente *et al* (2020)**.

The estimated increment of **1.4 µg/m³** is consistent with findings from **Trompetter & Davy (2019)** who found that:

- the average concentration of indoor biomass PM_{10} in four Aotearoa homes with a wood burner was **1.5 µg/m³ during winter**, and
- the indoor concentration was dominated by the use of the wood burner in that house.

We note that **Wyss *et al* (2016)** and **Fleisch *et al* (2019)** found no significant difference between indoor $PM_{2.5}$ in homes with or without a modern wood burner. This is not unexpected given the high variability in indoor $PM_{2.5}$ concentrations and the relatively small difference in $PM_{2.5}$ concentrations (in the order of **1 µg/m³)** when the concentration increment is averaged across longer time periods.

A.4 Seasonality adjustments to annualise indoor exposure and impacts

The review of literature resulted in estimates of wintertime exposure increments for the different appliances. These in turn needed to be adjusted for seasonal fuel use/operation to annualise the *indoor* exposure and impacts.

Wood, pellet and gas heating appliances

Wilton *et al* (2015) presents monthly home heating emissions for different airsheds across New Zealand as a proportion of daily winter (July) emissions.

From this information, we developed central, high and low annual increments for wood burners and pellet burners (for exposure to $PM_{2.5}$) and unflued gas heaters (for NO_2) as follows:

- Central: annualising the *central* estimated wintertime increment for each appliance from the relevant literature using the *average* seasonal splits of fuel use for wood burners across Aotearoa (Spring = 18%, Summer = 1% and Autumn = 26% of wintertime values based on Wilton *et al* 2015).

- High: annualising the *high* estimated wintertime increment for each appliance from the relevant literature using the *highest* seasonal splits for fuel use for wood burners across New Zealand (Spring = 33%, Summer = 7% and Autumn = 42% of wintertime values based on Wilton *et al* 2015).

- Low: annualising the *low* estimated wintertime increment for each appliance from the relevant literature using the *lowest* seasonal splits for fuel use for wood burners across Aotearoa (Spring = 4%, Summer = 0% and Autumn = 14% of wintertime values based on Wilton *et al* 2015).

Gas stoves

Annualisation of wintertime increments for gas stoves was undertaken based on the findings of Kornatit *et al* (2010) and Sun *et al* (2025).

From this information, we developed central, high and low annual increments for gas stoves (for NO_2) as follows:

- Central: annualising the *central* estimated wintertime increment for a gas stove from Gillespie-Bennett *et al* (2008), assuming Summer = 25% of wintertime values (incremental exposure to NO_2 in winter), with Spring and Autumn midway between at 63% based on Kornartit *et al* (2010).
- High: annualising the *high* estimated (upper 95% CI limit) wintertime increment for a gas stove from Gillespie-Bennett *et al* (2008), assuming Summer = 50% of wintertime values (incremental exposure to NO_2 in winter), with Spring and Autumn midway between at 75% based on Sun *et al* (2025).
- Low: annualising the *low* estimated (lower 95% CI limit) wintertime increment for a gas stove from Gillespie-Bennett *et al* (2008), assuming Summer = 0% of wintertime values (incremental exposure to NO_2 in winter), with Spring and Autumn midway between at 50% based on the possibility of being able to open windows/ventilate the space during a typical New Zealand summer.