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Arsenic in urban air: sources, health risk and mitigation

A thesis presented in partial fulfilment of the requirements for the degree of

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Abstract

Over recent years, several studies have suggested that high concentrations of arsenic may occur in outdoor air in urban areas of New Zealand on some winter nights. These spikes in arsenic concentrations are presumed to be caused by some householders burning copper-chrome-arsenate (CCA)-treated wood as a fuel for domestic home heating, but detailed examination of the issue has been lacking. The aims of this work are to examine the concentrations and source(s) of arsenic in ambient air in a representative New Zealand wood-burning community, identify and quantify potential health risks linked to both arsenic in air and the activity of burning CCA-treated wood, and undertake an initial exploration of social factors that may contribute to the issue. The town of Wainuiomata in the Wellington region was selected as the representative community.

Concentrations of total arsenic in Wainuiomata outdoor air were measured over two years, along with a number of other relevant variables useful for source characterisation, including two size fractions of particulate matter (PM₁₀ and PM_{2.5}), black carbon and other trace elements. Over both years, concentrations of arsenic in Wainuiomata air were indistinguishable from the national ambient air quality guideline of 5.5 ng/m³ expressed as an annual average. Arsenic levels were strongly seasonal and peaked during the winter months, but with significant variability from night to night. The highest 24-hour concentration recorded during winter was 79 ng/m³. Results of correlation analysis and source attribution provide strong support for the idea that the principal source of elevated arsenic in outdoor air is the domestic burning of wood treated with CCA preservative.

A detailed exposure model was developed and applied to estimate and contextualise potential arsenic exposures that may be experienced by adults and children living in the community, and quantify relative health risks. Potential community health impacts are estimated not to be significant where exposure is limited to outdoor arsenic, including that which infiltrates into the indoor environment, where “not significant” is defined as an additional lifetime cancer risk of less than 1 in 100,000 and a hazard quotient less than 1. Annual average arsenic in outdoor air would need to be around

15 times higher than the guideline value to increase an individual's attributable lifetime cancer risk to 1 in 10,000.

Of more potential concern are health risks arising from indoor exposure for residents who use CCA-treated timber as supplementary firewood where this may lead to fugitive emissions of arsenic from the firebox into indoor air. Not only does the predicted excess lifetime cancer risk approach 7 in 100,000, but there are also non-cancer health risks to children due to short-term exposure to the relatively higher levels of arsenic during the winter months. Hazard quotients above 1 were found to potentially exist for a small number of children (4%) based on the likelihood of living in a home where CCA-treated timber might be burnt combined with the presence of at least one adult smoker. However, overall greatest potential for acute health risk for children was found to be posed by accidental or incidental ingestion of CCA-wood ash, which contains very high concentrations of arsenic.

Results of focus group sessions and community surveying provided useful contextual information about the source activity and identification of some potentially modifiable social factors, along with some understanding about why prohibition of the activity of burning CCA-treated wood may be ineffective. Findings included an upper estimate of the proportion of households that may burn CCA-treated timber (approximately 16%), and identification of the problem that most residents are not able to distinguish treated from untreated wood.

A number of recommendations are made. Despite the preliminary nature of the findings due to uncertainties in the modelling and toxicity reference values, it is recommended that efforts should be made to discourage the practice of CCA-wood burning as a precautionary measure to protect against inhalation exposure to indoor sources of arsenic and ingestion of contaminated ash by children. Community education initiatives would need to be developed from the perspective of local residents, most of whom cannot identify CCA-treated wood. It would be ideal if this were complemented with a high-level review of the policy and regulatory framework which permits the manufacture, use and disposal of CCA-treated wood in New Zealand, to determine where risks might be best managed.

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1 Introduction, aims and structure

1.1 Introduction

This thesis quantifies levels of arsenic in ambient air in a representative New Zealand wood-burning community, determines relative health risks, and explores social elements that contribute to this issue, with a view to determining mitigation options. This Introduction gives a broad contextual overview, including research needs, leading to specific aims and objectives of the work in Section 1.2. A more detailed background is given in Chapter 2 to provide an overview of the nature of pre-existing evidence and introduce some concepts that guide or constrain data interpretation.

Greater Wellington Regional Council (GWRC) has been monitoring PM₁₀ and meteorological variables by continuous monitoring methods in Wainuiomata since mid-2006. In parallel, GNS Science has been collecting fine and coarse particulate matter for elemental speciation at GWRC's monitoring site, under the umbrella of the Australasian Region's International Atomic Energy Agency using nuclear analysis techniques for air pollution monitoring (Markwitz, 2005).

Many areas in New Zealand experience poor winter air quality due to particulate matter (PM₁₀)¹ emissions from solid fuel burners used for home heating (MfE, 2014). Elemental speciation of particulate matter undertaken for source apportionment studies shows that elevated elemental arsenic levels also occur during the home heating season. Although the elemental speciation studies were not designed to quantify arsenic and are indicative only due to the underlying method's high analytical detection limits, they suggest that arsenic concentrations could fail to meet the national ambient air quality guidelines (MfE, 2002) in many wood burning areas throughout New Zealand (Cavanagh, Davy, Ancelet, & Wilton, 2012). Arsenic measurements obtained by a standard reference method are needed to confirm compliance or non-compliance with the national ambient air quality guideline.

In some of the source apportionment studies, including one based in Wainuiomata, arsenic in air was positively correlated with the source factor "biomass burning",

¹ Particles with an aerodynamic diameter of 10 microns or less

suggesting that wood burning was the source of arsenic (Davy, Ancelet, Trompetter, Markwitz, & Weatherburn, 2012). The specific source of arsenic in air was strongly suspected to be the intermittent and opportunistic domestic burning of timber that has been treated with copper-chrome-arsenate (CCA) (Davy, Petersen, Trompetter, & Markwitz, 2011). To test this hypothesis, source attribution needs to be determined for intensively sampled arsenic and other elements using a method with a level of analytical detection less than 1 ng/m³.

Arsenic is classified as a non-threshold, high potency carcinogen (IARC, 1998, 2012). The national ambient air quality guideline limit aims to ensure no more than 1 in 100,000 excess lifetime cancer risk (MfE, 2002). Arsenic in New Zealand's air appears to have an apparently uniquely seasonal profile, consisting of shorter term elevated winter-only concentrations in residential areas, in contrast to the international situation where less variable long term concentrations arise mainly from industrial sources. In addition to the quantification of arsenic exposure in the New Zealand context, it is also necessary to assess whether meeting the national ambient air quality guidelines for arsenic is sufficiently protective of public health.

If it is confirmed through this investigation that the principal source of arsenic in Wainuiomata air is from the burning of CCA-timber (and by extension, other areas where winter arsenic has also been detected), then this would strongly suggest ineffectiveness of the current regulatory approach in which Regional Councils prohibit the burning of CCA-treated timber through rules in Regional Plans enacted under section 30 of the Resource Management Act 1991. Reducing levels of arsenic in air requires people to consciously avoid burning of CCA-treated timber in their domestic fires. Information about the social context surrounding people's use of CCA-wood for burning is largely anecdotal. Insight into the social aspects that support or discourage the practice of CCA-wood burning in a community where arsenic levels fail to meet the health guideline could inform the development of education or behaviour change campaigns to be undertaken by local authorities.

1.2 Aims and objectives

1.2.1 Overview of approach

The work described in this thesis falls into two main areas: a) physical and chemical measurements to quantify the health risk; and b) examination of the sociological context and mitigation. Aims of the first part are to determine levels, sources and potential health risks of arsenic in air in a small community (Wainuiomata) that may derive from the domestic burning of timber that has been treated with copper-chrome-arsenate (CCA). Aims of the second part are to understand relevant social and behavioural factors to identify viable approaches that could be used to discourage the burning of CCA-timber in the community.

Meeting these aims uses a cross-disciplinary research approach. Determination of levels and sources of arsenic in air is primarily in the field of environmental analytical chemistry and relies on targeted monitoring data. Assessment of human health risk entails developing a realistic exposure model to compare plausible scenarios against relevant toxicological dose-response relationships.

Identifying effective intervention measures to reduce human exposure, should it be shown to present a deleterious health risk, sits in the field of social science. Here the focus shifts to the assessment of understanding and awareness and the social context surrounding the use of CCA-treated timber in a local community where arsenic levels in air have already been determined.

At the outset it is recognised that burning of CCA-timber has been suspected as the key source of anthropogenic arsenic in several New Zealand towns on high-arsenic days, but critical appraisal of this idea through use of targeted monitoring data has been limited. Part of the aim is to test the evidence for this idea.

The two overall aims of this work are further broken down into five subsidiary aims with supporting objectives. These are outlined in the next section.

1.2.2 Aims and objectives

As far as the author is aware, the work described in this thesis represents the first time in New Zealand that physical and chemical approaches to air quality monitoring and source characterisation have been combined with a sociological approach to achieve overall aims relating to population exposure to a toxic substance in air. Aims 1, 2 and 3 (below) sit within the domain of the science of air quality monitoring, statistics and toxicology. Aim 4 relies on the research perspective of sociology. Aim 5 draws on the findings of both approaches.

Aim 1

To establish whether arsenic concentration in the Wainuiomata airshed met the national ambient air quality guideline during the monitoring period and to predict compliance for periods when arsenic is not monitored.

Objectives

- i) Obtain 24-hour average time series concentrations of arsenic in outdoor air using a standard reference method and obtain collocated measurements of PM₁₀, PM_{2.5}, black carbon and meteorological parameters over a two year period in Wainuiomata.
- ii) Develop a simple statistical model that could be used to estimate arsenic concentrations from other measured variables in order to predict concentrations in future years when arsenic concentrations are not measured.

Aim 2

To assess whether monitoring results for arsenic and other variables support the hypothesis that the principal source of arsenic in air is the domestic burning of CCA-treated timber.

Objectives

- iii) Establish the relationship between arsenic other measured variables, including a third party multi-element dataset, for the purposes of source attribution.

Aim 3

To quantify the potential human health risks from exposure to arsenic from all known sources (including ambient air monitoring data) through inhalation and ingestion exposure pathways.

Objectives

- iv) Identify plausible source-pathway-receptor linkages to estimate dose received for arsenic from contact with air and other environmental media likely to contain arsenic.
- v) Determine the nature and severity of adverse effects that may occur under various exposure scenarios (both acute and chronic durations) using relevant dose-response relationships for cancer and non-cancer health endpoints.
- vi) Assess the potential impact of exposure to arsenic from exposure routes and pathways associated with burning of CCA-treated timber relative to background sources (diet) and lifestyle factors such as cigarette smoking.

Aim 4

To identify salient features of the social environment surrounding the use of solid fuel for home heating that support or discourage the practice of domestic burning of CCA-treated timber.

Objectives

- vii) Determine the level of awareness of the burning of CCA-treated timber in the community and what knowledge is held about the environmental and health consequences of the practice.
- viii) Identify from residents' perspective, based on their everyday experiences and knowledge, what approaches would be most effective in discouraging the burning of CCA-treated timber in their community.

Aim 5

Determine whether a community education campaign to discourage the burning of CCA-treated timber is warranted, and if so, what approach would be most effective.

Objectives

- ix) Establish the severity and extent of health impacts arising from exposure to arsenic arising from the burning of CCA-treated timber.
- x) Identify viable approaches for reducing exposure to arsenic in air based on insight from the community wood burning practice study.

1.3 Outline of thesis

The next chapter (Chapter 2) sets the context for the thesis by reviewing what is known from the literature and other monitoring studies about arsenic in air and in other environmental media. The wider regulatory context surrounding CCA-treated timber and the social context of air quality management are also examined. The main body of the thesis is split into three chapters (3, 4 and 5). Due to the cross-disciplinary nature of the research, each chapter has its own methodological, results and discussion sections.

Chapter 3 includes all the components of the research that relate directly to the ambient air monitoring programme that was designed to determine compliance with the national ambient air quality guidelines as well as source attribution. Chapter 4 investigates the potential risk of adverse health outcomes for residents of

Wainuiomata from exposure to inorganic arsenic arising from the domestic burning of CCA-wood within the airshed. Chapter 5 encompasses all the components of the research that directly relate to the investigation of the social context surrounding wood burning for home heating with particular focus on aspects concerning the practice of using CCA-wood in the community. The final chapter (Chapter 6) synthesises the findings of Chapters 3, 4 and 5 in order to quantify the issue of arsenic in air, recommend the appropriate level of response, and to identify measures likely to be most effective in discouraging the use of CCA-treated timber based on insights from the social investigation. The outline of the structure of the thesis and how it relates to the aims is shown in Figure 1.1.

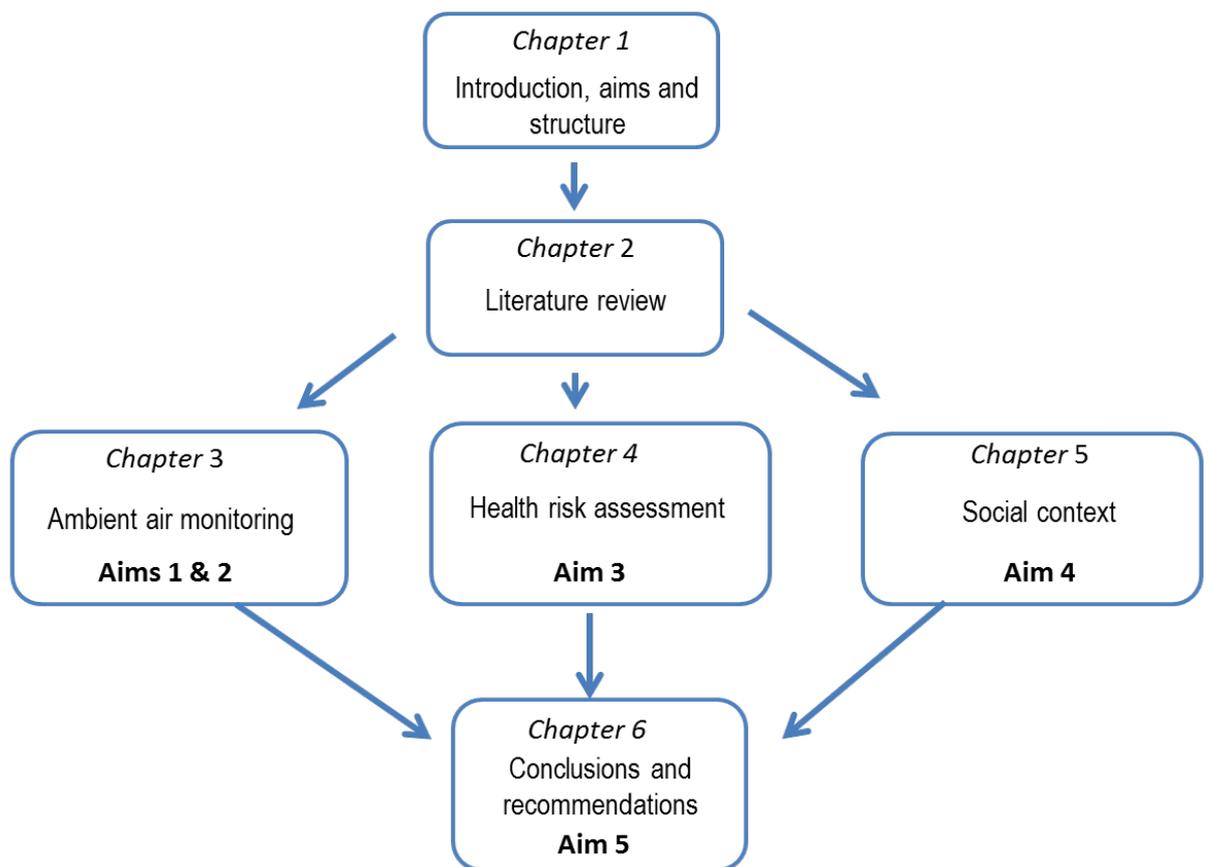


Figure 1.1: Outline of thesis structure

2 Literature review and context

2.1 Introduction

The purpose of this chapter is to provide context for the work described in this thesis. Topics include literature concerning arsenic in the environment, specifically levels in environmental media relevant for human contact and their respective health-based guidelines, the history of CCA-treated wood in New Zealand and regulations surrounding its use, and the contribution of arsenic derived from CCA-treated timber to environmental media. Further aspects include an overview of the fate of arsenic and other elements during combustion of CCA-treated timber, measurement methods and existing monitoring data for levels of arsenic in outdoor air, and results of previous air quality studies which indicate that arsenic levels in New Zealand's air are elevated at times during due to the domestic burning of CCA-treated timber. Finally, the regulatory environment in which the domestic combustion of CCA-wood is controlled is outlined, and the social understanding and level of public awareness regarding the dangers of using CCA-wood for domestic burning is discussed.

2.2 Arsenic in the environment

2.2.1 Physical, chemical and toxicological properties

Arsenic (atomic symbol As) appears in Group 15 (V) of the periodic table and is classified chemically as a metalloid, having both metal and non-metal properties. In the environment arsenic occurs in inorganic and organic forms and in different valence or oxidation states. Arsenic occurs most frequently in the environment in the inorganic form combined with the elements oxygen, chlorine or sulphur. The less toxic organic form occurs when arsenic is associated with carbon and hydrogen.

In compounds, arsenic typically exists in one of three valence states: arsine (-III), arsenite (+III) and arsenate (+V). Elemental arsenic (valence (0)) is mainly used for strengthening alloys of copper and lead and as a doping agent for semiconductors. The valence states of environmental interest are As(+III) as in arsenic trioxide (As_2O_3) and As(+V) as in arsenic pentoxide (As_2O_5), due to their toxicity to humans. These two oxidation states are often jointly referred to as "inorganic arsenic". Of these, the most

toxicologically potent arsenic compounds occur in the trivalent oxidation state due to their reactivity with sulphur containing compounds and generation of reactive oxygen species (Hughes, Beck, Chen, Lewis, & Thomas, 2011). Because arsenate (AsO_4^{3-}) has similar physiochemical properties to phosphate (PO_4^{3-}), arsenate can substitute for phosphate in several biochemical reactions and one consequence of this is diminished formation of ATP (adenosine triphosphate) which is essential for cellular metabolism in human and animals (Hughes et al., 2011).

Inorganic arsenic has been classified as a known human carcinogen by the International Agency for Research on Cancer (IARC) (Group 1) (IARC, 1998, 2012) and by the US EPA (Group A)².

2.2.2 Arsenic sources, environmental fate and partitioning

Arsenic is released to the environment from natural processes such as weathering of crustal matter, volcanic activity, sea spray and forest fires. Arsenic released from anthropogenic sources far exceeds natural sources (ATSDR, 2007). Anthropogenic sources include non-ferrous metal mining and smelting, pesticide application, coal and wood combustion and waste incineration. Worldwide, most anthropogenic releases of arsenic are to land or soil in the form of pesticides or solid waste. However substantial amounts are also released to air and water (ATSDR, 2007). Because arsenic is a natural component of the Earth's crust, trace levels of arsenic are found in all environmental media, i.e., soil, water, air and biota (animals and plants).

Arsenic in air is present mainly as inorganic arsenic adsorbed onto the surface of fine particles (mostly less than 2 μm) as a variable mixture of the trivalent and pentavalent forms (Maggs, 2000; WHO, 2000). The residence time of arsenic in the atmosphere depends on particle size and meteorological conditions. During dispersal by wind and air currents arsenic particles are returned to ground by wet or dry deposition where they enrich soils and water bodies. Terrestrial plants may accumulate arsenic by root uptake from soil or by absorption of airborne arsenic particulate deposited on leaves (ATSDR, 2007). A Danish modelling study of plant uptake of arsenic emitted from incineration of waste CCA-treated timber found atmospheric fall out was the dominant

² <http://www.epa.gov/IRIS/subst/0278.htm#woe>

pathway for uptake by leafy vegetables (kale) and that uptake by root crops (potatoes and carrots) was as a result of both soil uptake and atmospheric deposition (Larsen, Moseholm, & Nielsen, 1992).

In Europe concentrations of arsenic in outdoor air range from 0 to 1 ng/m³ in rural areas, 0.5 to 3 ng/m³ in urban areas and up to 50 ng/m³ in the vicinity of some industrial plants (EC, 2000). A review of international arsenic concentrations in PM₁₀ found 0.41 ng/m³ in southern California with the maximum of 272 ng/m³ found in Serbia (Fang, Chang, Huang, & Huang, 2012).

Trace elements in indoor dust represent an important potential exposure site for incidental ingestion, especially by children, and for inhalation of re-suspended particles. Elements such as arsenic enter from the outdoor environment not only via airborne particles, but also with tracked in soils and dust. Levels of arsenic in house dust in a mining area were an order of magnitude higher than in houses in an unaffected area although were not correlated with levels of arsenic in garden soils (Rieuwertz, Searle, & Buck, 2006).

In a rural Wisconsin households where scraps of CCA-treated plywood were burnt in an indoor stove, samples of dust and ash collected from around the stove area contained arsenic at 100 to 600 mg/kg (Peters, Croft, Woolson, Darcey, & Olson, 1984). Dislodgeable arsenic residue from outdoor CCA-timber decks can be tracked inside onto carpets via foot traffic (Patch, Ullman, Maas, & Jetter, 2009). Further research is needed to determine if this exposure route for arsenic is significant compared to exposures from other sources.

A potential indoor source of arsenic is environmental tobacco smoke, emitted from the burning tip of the cigarette and exhaled by the smoker, which may contain trace levels of arsenic. These very fine aerosols may be deposited onto household surfaces and contribute to enrichment of household dust. Carpets tend to accumulate more contaminants than hard surfaces and are more likely to represent long-term reservoirs of tobacco smoke constituents (Apelberg et al., 2013). Levels of arsenic in indoor environments where smoking occurred (e.g., coffee shops) were slightly higher than in other places without smoking (such as libraries) (Landsberger & Wu, 1995).

Arsenic concentrations in environmental media vary greatly across the world depending on natural background levels and degree of enrichment due to past and present anthropogenic activities that release arsenic into the environment. In terms of global human exposure to arsenic food and drinking water are the principal routes of exposure, with the exception of some industrial workers. The following values for percentage absorbed daily dose of inorganic arsenic are provided (EC, 2000):

Air	<1%
Cigarette smoke	0-16%
Drinking water	0-33%
Food	50-98%

Whilst potential absorbed arsenic dose from exposure to air is much less than the dose obtained from the other media, it may still be toxicologically significant as the lung is a principal site for carcinogenicity (EC, 2000).

Section 2.2.3 provides a brief review of the levels of arsenic found in New Zealand's drinking water, soils, indoor dust, including where known, the contribution of arsenic arising from CCA-treated timber to these media. National health-risk guidelines for arsenic specific to each media are also reported.

2.2.3 Arsenic in the New Zealand environment

2.2.3.1 Drinking water

New Zealand's public drinking water is typically sourced from surface waters and groundwater that are naturally low in arsenic, with the exception of the Taupo Volcanic Zone (TVZ) where naturally occurring arsenic in volcanic rock is dissolved as super-heated geothermal water rises to the surface (Robinson et al., 2004).

Furthermore, discharge of geothermal fluids from the Wairakei thermal power station contributed significantly to the influx of arsenic into the Waikato River (Webster-Brown & Lane, 2005). Consequently Hamilton's drinking water supplied from the Waikato River that runs through the TVZ, can contain elevated levels of arsenic (McLaren & Kim, 1995). The New Zealand numerical standard for arsenic in drinking water is 10 µg/L designed to ensure an acceptable lifetime excess risk of skin cancer

(MoH, 2008). Levels in the Waikato River exceed this but receive municipal treatment to remove the arsenic before public consumption. A small number of people obtain their drinking water from rainfall roof run off. Arsenic contamination of roof-supplied drinking water was found where exposure CCA-timber formed part of the rainwater collection system (Simmons, Hope, Lewis, Whitmore, & Gao, 2001).

2.2.3.2 Food

Arsenic is ubiquitous in the environment and therefore diet is the largest source of both organic and inorganic arsenic for typical individuals (Hughes et al., 2011). Fish and seafood can accumulate arsenic from their environment, which is mostly converted to less toxic organic forms of arsenic. The arsenic content of plants, such as rice, grain and vegetables is usually determined by the arsenic content of the soil, water, air, and agricultural residues, and is predominantly inorganic arsenic (Vannoort & Thomson, 2011). Arsenic (as arsenate) is an analogue of phosphate and so may be taken up by plant roots and accumulate in plant stems and leaves. Elevated levels of arsenic have been found in the traditional food source watercress (*Lepidium sativum*) and several other plants growing in the Waikato River (Robinson, Duwigg, Bolanc, Kannathasan, & Saravanan, 2003), but this is primarily geothermal in origin and thought to deposit from the water to the exterior of the plants in iron oxide or biofilm coatings (Robinson et al., 2006)

The Australia New Zealand Food Standards Code –Standard 1.4.1 – Contaminants and Natural Toxicants³ specify a maximum limit (ML) of 1 mg/kg for total arsenic in cereals and a ML of 2 mg/kg for inorganic arsenic in crustacea and fish and 1 mg/kg for seaweed and molluscs.

2.2.3.3 Soils and indoor dust

The national natural background for average arsenic concentration in topsoil is 4.5 mg/kg (MfE, 2011d). The historical use of arsenic based pesticides in sheep-dips has resulted in an estimated 50,000 sites contaminated with arsenic and persistent organic pollutants in New Zealand (Robinson et al., 2004). Elevated levels of arsenic were also found in orchard soils due to the historical use of arsenic-based pesticides to control

³ <http://www.comlaw.gov.au/Details/F2015C00052>

chewing insects (Gaw, Wilkins, Kim, Palmer, & Robinson, 2006). Reclaimed land built using old gold mine tailings in Thames was also found to have elevated arsenic (Foote & Rumsby, 2011).

Copper, chrome and arsenic leach from CCA-structures above and below ground into surrounding soils (Ko et al., 2007). Plants grown in soils close to CCA-timber, e.g., in garden borders, may absorb arsenic leached into the soil adjacent to the wood. CCA-wood sawdust applied to the surface of the soil (increased surface to volume ratio) arsenic may leach into soil (APVMA, 2005). Leaching of arsenic and copper into soils from in situ CCA-treated posts in vineyards has been observed but arsenic did not appear to be taken into the grapevine plants (Ko et al., 2007).

The New Zealand soil contaminant standard for soils of new residential subdivisions is 20 mg/kg (MfE, 2011d).

There is little or no information on the levels of arsenic in indoor dust in New Zealand. Elemental speciation of dust samples in 29 homes in the Hutt Valley in 2006 found that soil contributed to about half the measured dust (Trompetter, Boulic, Ancelet, Davy, & Phipps, 2013). A closer examination of the results for arsenic shows that seven samples (of 59) contained arsenic above the level of detection for IBA. The mean for arsenic using all samples was 28 ppm, with ranging from 0 to 170 ppm. These arsenic concentrations in indoor dust are similar to that found in a Canadian study designed to obtain representative urban house dust metal concentrations (Rasmussen et al., 2013). Earlier, Fergusson and Kim (1991) had reported a median of 16 mg/kg for arsenic in house-dust based on four studies.

2.3 Arsenic in New Zealand's air

2.3.1 Outdoor air

To date there have only been limited monitoring campaigns undertaken with the objective of determining arsenic in air in New Zealand. Short-term measurements of arsenic and lead were made the vicinity of a lead-acid battery recycling plant in Lower Hutt in 2008 and in 2009 to evaluate appropriateness of environmental controls on fugitive emissions (Mitchell, 2008, 2009). This study determined average

concentrations of arsenic in TSP (determined by ICP-MS) of 4.4 and 3.9 ng/m³ respectively for two community monitoring sites close to the plant (Mitchell, 2009). Monitoring of arsenic deposition rates, required as a condition of the battery-recycling plant's discharge to air permit issued by GWRC, found the median deposition rate for arsenic was 135 µg/m²/month between 2006 and 2011 at the two community monitoring sites. Contamination of a local stream that received storm water containing lead and arsenic arising from deposition of airborne emissions from the recycling plant onto paved surfaces has also been documented (Markland, Strange, & Van Erp, 2005). The battery recycling plant closed in 2012 and there are now no other known significant industrial emitters of arsenic in particulate matter in New Zealand.

Coal contains trace levels of arsenic, especially coals containing pyrite. The potential for release of arsenic into atmosphere as a result of industrial or domestic burning has not been investigated in New Zealand. New Zealand's largest coal-fired power station in Waikato does not have any requirements to monitor arsenic in the flue gas or in ambient air suggesting that this is not of concern (Genesis Energy, 2011). Domestic coal burning in part of China was shown to cause arsenic poisoning where coal with usually high arsenic content due to local geologic factors was burnt indoors in unventilated stoves and used to preserve food (Liu et al., 2002).

Two other studies have specifically looked at arsenic levels in the urban environment. In the first study, Waikato Regional Council also found an average arsenic concentration of 6.1 ng/m³ in PM₁₀ (determined by ICP-MS) using filters exposed for a consecutive four day period in urban Hamilton from November 2010 to November 2012⁴. The second study involved a retrospective determination of multi-element concentrations by XRF in particulate matter collected on PM₁₀ glass fibre filters in urban Whangarei from 2004 to 2012. In this study the average concentration of arsenic determined by XRF was 3 ng/m³ (Davy & Ancelet, 2014).

As part of a wider study evaluating the *in vitro* toxicity of particulate matter, a winter average of 13.0 ng/m³ for water-soluble fraction of arsenic was found in PM₁₀ collected on glass fibre filters in Timaru (Cavanagh, Trought, & Sheenan, 2010). None

⁴ Data provided courtesy of Environment Waikato.

of these monitoring campaigns were designed explicitly to follow the standard sampling frequency and/or arsenic analysis techniques to assess compliance with the national ambient air quality guideline.

The potential for combustion of CCA-wood as a source of arsenic emissions was signalled in by a technical review of the 1994 ambient air quality guidelines published in 2000 (Chido & Rolfe, 2000). However, national level interest/concern about arsenic in ambient air that might arise from domestic burning of CCA-treated timber has only relatively recently been raised following the emergence of a common pattern of winter elevated arsenic concentrations largely from studies involving elemental speciation of particulate matter (Cavanagh et al., 2012; Davy, Ancelet, Trompetter, & Markwitz, 2014). In addition, hourly elemental speciation measurements in four urban areas of New Zealand found arsenic concentrations displayed the same diurnal pattern as particulate matter concentrations in wood smoke environments with a large peak in the evening and a smaller peak in the morning (Ancelet, Davy, & Trompetter, 2014). These studies are discussed in more detail in Section 2.3.4.

Environment Canterbury is currently undertaking a two year monitoring programme to measure arsenic in air motivated by concern about the large amount of demolition timber available in the community following building remediation required following the damage caused by the Christchurch earthquake in 2011. Monitoring results were unavailable at the time of writing.

2.3.2 Air quality guidelines

The New Zealand national ambient air quality guideline for inorganic arsenic is 5.5 ng/m³ in PM₁₀ measured as an annual average concentration (MfE, 2002). This guideline was set to restrict the population risk of lung cancer associated with long term exposure to arsenic to an acceptable level of 1 in 100,000. There are no ambient guidelines for short-term exposure to arsenic. Workplace exposure standards for arsenic and arsenic compounds are 10,000 times higher (50 µg/m³) than the ambient guideline, perhaps reflecting the shorter 8-hour exposure period and historical tolerances. There are no national or international health-risk guidelines for arsenic deposition. The health risk associated with arsenic deposition onto soil and plants and

into water and subsequent uptake by biota and potential human exposure via consumption can be dealt with through a multi-pathway risk assessment process (US EPA, 2004).

The national ambient air quality guideline for arsenic is based on total arsenic and does not differentiate between arsenic species in particular the two common valence states, As (III) and As (V). It is generally accepted that As (III) is more toxic than As (V) in both acute and chronic exposure scenarios (ATSDR, 2007). A review of arsenic speciation studies for particulate matter in air found As (V) to be more prevalent than As (III) but it was unclear how this ratio might vary with particle size and source (Lewis, Reid, Pollock, & Campleman, 2012). Exposure dose relationships underpinning the US EPA ambient air guideline for arsenic are based of occupational exposure to As (III) and so in evaluating health risks based on total ambient arsenic, which may be mostly composed As (V), may overestimate the health risk in environmental exposure situations (Lewis et al., 2012).

2.3.3 Measurement techniques

The monitoring method recommended for determining compliance with the national guideline for inorganic arsenic is PM₁₀ gravimetric sampling in accordance with 40 CFR Part 50, Appendix J, followed by analysis of particulate aerosol collected on filter media using atomic absorption spectroscopy (AAS) or an equivalent method (MfE, 2002). There are a range of analysis methods which differ in their detection limits, sample preparation, and cost.

Inductively Coupled Plasma – Mass Spectrometry (ICP-MS) has largely replaced AAS as the preferred commercially-available equivalent analytical method to AAS as it is a less labour-intensive method, has lower detection limits and can measure multiple elements (up to 60) simultaneously. However, ICP-MS is relatively costly and, like AAS, requires physical destruction of the sample as the filter is digested in an acid solution prior to analysis.

Alternative methods for multi-elemental analysis of air particulate include Ion Beam Analysis (IBA) and X-Ray Fluorescence (XRF). IBA describes a range of elemental

analysis techniques based on the use of ion beams generated by particle accelerators, including Particle Induced X-Ray Emission (PIXE) and Particle Induced Gamma-ray Emission (PIGE) that can be used to determine the concentrations of elements from sodium to uranium (atomic weights ranging from 11 to 92). These techniques are non-destructive and require minimal sample preparation (Trompeter & Markwitz, 2005). IBA techniques and have been used by GNS Science in New Zealand for elemental speciation of PM for statistical modelling to assist regional councils to better understand the relative contribution of different emission sources to air pollution in their regions. IBA has an analytical level of detection (LOD) for arsenic of between 3 and 30 ng/m³ depending on filter media (Cavanagh et al., 2012). Teflon filters typically used for collecting particulate matter for receptor modelling studies had detection limits for arsenic ranging from 20-30 ng/m³ depending on the sample composition and the presence of any spectral interference from other elements. X-Ray Fluorescence (XRF) is another non-sample destructive method that uses X-ray techniques to simultaneously determine concentrations of elements with atomic weights from aluminium and above based of the specific fluorescence of each element. For both of these surface analysis techniques, accurate calibration can be complicated by differences between the sample matrix and those of the reference standards. This problem is less of an issue in ICP-MS, where both samples and standards are presented as dilute acid extracts.

2.3.4 Source apportionment based on trace elements in air

Source apportionment using receptor modelling to determine the relative contributions of emission sources (e.g., vehicles, domestic burning, industry and natural sources, such as soils and sea-salt) to ambient particulate matter concentrations is available for many areas of New Zealand. Receptor modelling utilises multi-element data sets for PM₁₀, PM_{2.5}, PM_{2.5-10} (Trompeter & Markwitz, 2005). A common factor to receptor modelling in urban environments dominated by wood smoke was elevated levels of arsenic during the winter months, for example, in Hastings (Wilton, Davy, & Smith, 2007), Masterton (Ancelet et al., 2012), and Auckland, Wainuiomata and Tahunanui (Davy et al., 2011). At some of these locations, including Wainuiomata, Nelson, and Hastings, arsenic was associated with the “biomass

burning” source factor profile (Ancelet, Davy, & Trompetter, 2013; Davy et al., 2012; Wilton et al., 2007). The authors of these studies suggested that the reason arsenic was associated with the “biomass burning” source was due to the domestic burning of CCA-treated timber.

A review of arsenic concentrations determined largely by IBA in New Zealand receptor modelling studies concluded that it was likely that arsenic levels would exceed national and international guidelines at a number of locations and therefore further monitoring using a method capable of detecting arsenic at low concentrations, i.e., < 0.1 ng/m³ is needed (Cavanagh et al., 2012). The pseudo-annual averages for arsenic calculated by Cavanagh using all IBA values for arsenic including those below LOD, as well as for winter periods only. Therefore arsenic annual averages calculated from IBA measurements may under or over-estimate concentrations relative to the standard method. Furthermore, the finding that New Zealand’s levels of arsenic in air are characterised by shorter-term elevated winter concentrations requires investigation as to whether an annual average guideline based on life-time exposure is the appropriate metric for assessing health impact.

2.4 Arsenic in wood preservatives

2.4.1 History and background

Timber is extensively used in New Zealand as the main structural component of many buildings, particularly residential dwellings. Wood, because it contains carbohydrates and nutrients, is an attractive food source for a variety of insects and fungi⁵. Wood, depending on the species, will deteriorate, discolour and decay in the outdoor environment, especially when in the ground or wet. Many chemical treatments are available to extend the durability of wood depending on its intended use, for example, light organic solvents (LOSP) for resistance to insect attack, boron salts to prevent borer attack, and ACQ⁶ and copper azole. Copper-chrome-arsenate (CCA) is a broad spectrum biocide, with copper used to control fungi and marine borers, arsenic to control insects and some copper-resistant fungi, and chromium to fix the copper and arsenic in wood (Read, 2003). CCA treatment trademarked as “Tanalised” was

⁵ <http://www.nzic.org.nz/ChemProcesses/forestry/4B.pdf>

⁶ Alkaline or ammoniacal copper quaternary

developed in the 1930s and introduced into New Zealand in 1955, mainly for the treatment of fencing and poles (Marston & Singh, 2013). CCA-treated timber's use in building expanded in the late 1960s where it was used mainly for subflooring and then began to be used extensively for high-decay areas in New Zealand housing in the 1970s. In New Zealand the predominant wood type treated is the softwood *Pinus radiata*. The most recently available production estimates for CCA-treated timber are 574,750 m³ in 2006 which comprised 69% of total production for common treated timber types (Love, 2007). It is estimated that CCA-treated timber accounts for nearly one percent (by weight) of all construction industry-generated material wastes in New Zealand (Jaques, 2003). CCA-treated timber is widespread in the New Zealand building and residential environment and therefore construction and demolition waste is likely to contain a significant proportion of CCA-wood.

2.4.2 Building industry regulation

Timber treatment in New Zealand is governed under NZ 3640:2003, *Chemical Preservation of Round and Sawn timber*. This framework specifies what chemicals can be used to treat timber in New Zealand, and minimum retention required for a given hazard class. Hazard classes describe an environment or condition of use where timber is at particular risk of biodegradation by one or more biological agents (e.g., fungi, insects, bacteria or marine organisms). NZS 3640:2003 states that CCA can be used as a treatment method for all hazard classes as it has broad spectrum effectiveness against insects, wood-destroying fungi, and marine borers. CCA preservative is specified as comprising copper, chromium and arsenic salts or oxides dissolved in water to give a solution containing active elements in the following proportions: 23-25% copper, 38-45% chromium, and 30-37% arsenic. The prescribed formulation representative of CCA oxide formulations currently sold in New Zealand is shown in Table 2.1 (McCallum, 2010).

Table 2.1: Formulation of CCA in New Zealand

Formulation	Active element composition
CuO	23 %
H ₂ CrO ₄	41%
H ₃ AsO ₄	35%

There are minimum requirements for the penetration and retention of CCA components in treated wood expressed as a percentage based on mass per unit mass in the specified retention analysis zone (Table 2.2). Therefore final arsenic levels in CCA-treated wood differ according to the treatment level required to meet the retention levels for a particular hazard class.

Table 2.2: Total active element minimum retention as specified in NZS 3640:2003

Hazard class	Example	CCA	% m/m oven-dry weight of wood
H.1.1	Interior finishing timber	Arsenic	0.04
H.1.2	Structural framing timber	N/A	N/A
H3.1	Fascias, weatherboards	Arsenic	0.57
H3.2	Decking and outdoor structures	Arsenic	0.37
H4	Fence posts	Arsenic	0.72
H5	House piles, retaining wall poles	Arsenic + Copper + Chromium	0.95
H6	Marine piles	Copper	0.40

Thus, for example, the treated zone of H4 timber is required to contain 7200 mg/kg arsenic⁷. There are best practice guidelines for ensuring the health and safety of workers during the production of CCA-treated timber, e.g., to minimise oral, inhalation and dermal exposure through the use of protective clothing and housekeeping practices (NZTPC, 2005). These guidelines also advise against burning of CCA-timber for waste disposal due to the release of arsenic to air. Other health and safety precautions are aimed towards people working with the final product, builders and such like, and are available from the website of the Building Research Association of New Zealand (BRANZ)⁸.

2.4.3 Identification of CCA-timber

Timber freshly treated with CCA has a distinctive greenish tinge arising from the oxidation of copper in the treatment formulation. Other timber preservatives containing copper, but not arsenic, are also provided for by NZS 3640:2003, e.g., copper azole, alkaline copper quaternary, micronized copper quaternary and

⁷ A factor of 10,000 times converts between percentage (%) and parts per million or mg/kg.

⁸ <http://www.renovate.org.nz/1940-60s/common-problems-and-remedies/health-risks-treated-timber/>

micronized copper azole. Therefore these other copper-containing preservatives will also impart a greenish tinge to wood. The green colour of CCA-treated wood also varies in shade and intensity depending on the degree of exposure to sunlight immediately after treatment. Over time the greenish tinge fades to grey following weathering in the outdoor environment, depending on the treatment retention level. Therefore, colour cannot be used to reliably identify whether wood has been treated with CCA.

Timber containing arsenic can be identified by preservative class branded on to the wood, 01 for CCA oxide and 02 for CCA salts. However, not all wood product types that are allowed to be treated with CCA need to be branded. For example, fence battens and droppers are required to be treated to H3.2 but have no branding requirements.

There are various chemical tests that can be applied to wood to test for the presence of arsenic-containing preservatives. These are discussed in detail in an overview of identification methods for waste wood separation in the context of disposal options for demolished residential housing post the 2011 Christchurch earthquake (Marston & Singh, 2013) and include use of portable XRF. Of these tests, ascorbic acid colour indicator, could be used for small-scale identification process, but does not appear to be commercially available for domestic users. A product called "Burnsafe" was identified in an article on timber preservation that could be used for determining whether timber was treated with CCA prior to use in the home fire or BBQ (Croucher & Parker, 1998). This product was not found using an on-line search.

Lack of public education or the difficult identification of CCA-wood after weathering and painting, the domestic burning of CCA-wood is inevitable (Tame, Dlugogorski, & Kennedy, 2003).

2.4.4 Environmental fate of arsenic during CCA-wood combustion

The fate of arsenic during the combustion of CCA-wood has attracted research attention directed at finding environmentally benign end of life management options for large quantities of waste CCA-wood. Studies show in the presence of excess air, between 10 and 80 per cent of arsenic is volatilised and emitted as aerosol or

particulate, with little or no loss of copper or chromium which is preferentially retained in the ash (APVMA, 2005; Hirata, Inoue, & Fukui, 1993). During combustion it is hypothesised that pentavalent arsenic (As_2O_5) is reduced to arsenic trioxide (As_2O_3) (the more toxic form) and that the quantity of arsenic evolved depends on temperature and residence time (Helsen, Van den Bulck, Van Bael, & Mullens, 2003).

Limited experiments have been carried out on the fate of arsenic during burning of CCA-wood in a domestic wood burner operated under New Zealand conditions. A New Zealand trial found that when CCA-wood was burnt in a domestic wood burner, between 15.9% and 18.9% of arsenic present in the wood was evolved in the flue gas, while 99% of the copper and chromium was retained in the firebox ash (Abbott & Rogers, 1989). A study that determined the concentrations of metals and PAHs in particulate emitted during the in-situ use of pre-1994 domestic wood burners in Tokoroa, found high levels of water-soluble arsenic in particulate samples (1860 ng/mg) from one burner where “old-decking” had been burnt which equates to an emission factor for arsenic of 19.55 mg/kg (Cavanagh, Trought, Brown, & Duggan, unpublished manuscript). In contrast emissions from a modern burning pine, macrocarpar and blue gum when analysed by IBA failed to detect arsenic (Ancelet, Davy, Trompetter, Markwitz, & Weatherburn, 2011b).

The concentration of arsenic in CCA-wood ash depends on the level of timber treatment, degree of weathering and combustion conditions, in particular residence time and temperature of the ash bed (McMahon, Bush, & Woolson, 1986; Solo-Gabriele, Townsend, Messick, & Calitu, 2002). It has also been found that copper present in CCA-treated wood can act as a catalyst for the formation of polychlorinated dibenzo-*p*-dioxins and dibenzofurans (PCDD/Fs) which accumulate in the residual ash especially under low oxygen smouldering conditions (Tame et al., 2003).

Therefore the residual ash from the burning of CCA-treated timber must be appropriately disposed of, due to risk of metals leaching from the ash into other media and the potential risk to human health during handling and disposal of contaminated ash – if it is ingested or inhaled.

2.4.5 Health risks of CCA-wood

There is no legislation in New Zealand that restricts the use of CCA-treated wood to particular end-uses. General recommendations include not using treated timber for food receptacles, toys, barbecues, smoking meat or fish, or domestic fires (Read, 2003). Treated timber sawdust should not be used for areas under playground equipment.

In terms of public health issues from exposure to CCA-timber the main focus in New Zealand has been on the potential for children to ingest dislodgeable arsenic residues through their contact with CCA-wood structures, particularly in playgrounds. The new use of CCA-treated timber in residential and recreational settings in USA and EU was banned from 2004. A New Zealand review of health risk to children as result of their contact with CCA-structures in playgrounds found the research to be inconclusive as to whether children's exposure to arsenic posed a significant risk and there was insufficient evidence for banning its use or replacing existing structures in New Zealand (Read, 2003). Furthermore, arsenic exposure due to burning of CCA-treated timber was excluded from the public health review as this exposure pathway was thought to be less likely and of a short-term nature (Read, 2003).

2.5 Social and regulatory context of domestic wood burning

2.5.1 Air quality regulation

The primary regulatory instrument for managing outdoor air quality in New Zealand is the Resource Management Act 1991 (RMA). Under section 15 of the RMA, discharge of contaminants to air are subject to provisions of the regional plan which activities are permitted and which require resource consent. These rules are laid down in Regional Plans made under section 30 of the RMA. The current Regional Air Quality Management Plan for the Wellington region adopts a non-regulatory approach to discharges of contaminants from fires. At the time of writing the draft Natural Resources Plan for the Wellington region specifically prohibits the burning of CCA-timber in domestic fires.

Pursuant to section 43 of the RMA, a national environmental standard for air quality was introduced in 2004⁹. This standard prohibits certain activities, sets emission and design standards for new wood burners and limits for concentrations of PM₁₀, carbon monoxide, nitrogen dioxide, sulphur dioxide and ozone in outdoor air. The intent of this standard is to provide a guaranteed minimum level of health protection for all New Zealanders in terms of air quality (MfE, 2011a).

Regional Councils, in areas of New Zealand that do not meet the limit for PM₁₀ in outdoor air (i.e., no more than one 24-hour average above 50 µg/m³ per year), are required to implement measures that will lead to reduced emissions of PM₁₀ sufficient to ensure their airsheds comply with the standard by 2016 and 2020 (MfE, 2011b). Regional Council's policy interventions typically include a suite of measures such as regulation, information, education and incentives principally aimed at reducing the number of wood burners through households switching from solid fuel heating to clean heat or lower emission technology burners (Wilton & Bluett, 2012a). In an attempt to achieve further reductions in emissions, some Regional Councils, notably, Environment Canterbury are adopting behaviour change campaigns based on social marketing to encourage "environmentally-responsible" wood burner operators – for example, fire lighting techniques that reduce the amount of smoke emitted from the chimney¹⁰.

Territorial Authorities who permit the installation of wood burners under the Building Act 1991 also have powers to issue local bylaws under the Local Government Act 2002, for example, that control the type of wood burning appliances allowed to be installed.

2.5.2 Social understanding of air quality management

In the context of air quality management the use of wood for home heating is viewed as an environmental "pressure" as this activity results in emissions to air of PM₁₀ that then result in poor air quality leading to adverse health effects at the population level (MfE, 2014). The use of wood for home heating has a different meaning for those involved in the practice. A qualitative study in Christchurch investigated how residents

⁹ Resource Management (National Environmental Standards for Air Quality) Regulations 2004

¹⁰ <http://letscleartheair.co.nz/>

made sense of their home heating practices using wood burners in an environment with high levels of winter air pollution (PM₁₀) and where substantial educational campaigns and financial incentives to switch to “clean heat” have occurred (Cupples, Guyatt, & Pearce, 2006). This study found that although Christchurch residents were very aware of the air pollution problem and its causes, they were reluctant to change their heating methods and indeed felt good about burning wood as it was connected with a masculine pioneering spirit (i.e., tough and practical) and thriftiness (Cupples et al., 2006). In a similar vein, residents in an Australian town (Armidale) with high winter air pollution due to use of wood for home heating justified their use by a variety of strategies, including presenting wood heating as a natural, traditional and social activity strongly linked to a shared rural identity (Reeve, Scott, Hine, & Bhullar, 2013).

Although it has been established that domestic wood burning is the principal source of emissions leading to degraded air quality (Kuschel et al., 2012), the public perception is that poor air quality is caused by vehicles and transport, and industries (Hughey, Kerr, & Cullen, 2013).

The burning of CCA-treated timber is prohibited through most regional plans enacted under the RMA. The dangers of the practice have been passively publicised through web pages of Ministry for the Environment (MfE), New Zealand Home Heating Association (NZHHA) and BRANZ. Despite the prohibition and the warnings, there is evidence that this practice persists in the domestic setting. A home heating survey in Auckland found that 17% of wood burning households burnt decking or fence posts offcuts on their fires in the previous week (Stones-Havas, 2014). A home heating survey conducted in Masterton found that 16% of wood burners always, often or sometimes decking or fence posts offcuts on their fires (Sridhar & Wickham, 2013). It is highly likely that some component of decking or fence post offcuts reportedly used as firewood was treated with CCA.

No research articles investigating the level of public awareness of the dangers of burning treated timber in the home were found. However, studies showed that personal knowledge about poor air quality or health effects is not an important consideration for home heating choices (Cupples et al., 2006; Smith, McChesney, &

Butcher, 2005). Anecdotal evidence suggests that many people are unaware (or possibly unconvinced) that burning treated timber releases contaminants to air that can be harmful if inhaled. Informal discussions with air quality practitioners and people in the community indicate that there is a widely held perception that treated timber is used as a wood source by “poor” people or those classed as lower-socioeconomic because they can get waste wood for free and they cannot afford to buy clean burning fuel. The basis for this view has not been formally investigated.

A home heating survey in 2006 found that 61% of people who used wood for home heating in Wainuiomata obtained some or all of their wood for free. Information is not available on the nature of self-collected or free wood and this may include waste wood: for example, commercial premises that make pallets available for members of the public to collect (Figure 2.1).



Figure 2.1: Photo showing typical scenario of pallets available for public collection (Petone, July 2013)

2.6 Synthesis

Many areas in New Zealand experience poor winter air quality due to particulate matter emissions from solid fuel burners used for home heating. Results of elemental speciation of particulate matter, undertaken for source receptor modelling, show that in some areas fine particulate matter originating from wood burners contains inorganic arsenic. Arsenic is not normally found in wood species burnt for home heating and therefore it is likely that some households are also burning CCA-wood. Experimentally arsenic has been shown to be emitted when CCA-wood is burnt in a domestic wood burner, while copper and chromium are preferentially retained in the ash.

The New Zealand health-based guideline for inorganic arsenic in air is based a long term annual average to protect the public against life-time cancer risk. The elemental speciation studies have used methods which have a high detection limit for arsenic and therefore may under or over-estimate the arsenic concentration with respect to the national ambient air quality guideline. Given the seasonal occurrence of arsenic in air and the short term indoor air exposure that could occur when CCA-wood is burnt inside the house, it is unclear whether there are additional health risks that are not adequately mitigated meeting by the long term annual average for arsenic. Other environmental media, such as food and drinking water, also contribute to arsenic exposure and therefore it is pertinent to consider the possible contribution of arsenic in air compared to other environmental exposures.

Offcuts and waste CCA-wood are likely to be widely available due to their prevalence in New Zealand building materials. The traditional air quality management approach of prohibiting burning of certain materials through regional plans promulgated under the RMA does not appear to be effective in preventing the incidental burning of CCA-wood in the domestic context. The effectiveness of the passive dissemination of cautions against burning CCA-wood through official websites, such as the Ministry for the Environment (MfE), New Zealand Home Heating Association (NZHHA) and New Zealand Timber Preservation Council (NZTPC), is not clear. The motivations behind householder's use of treated timber on their fires is, by some, anecdotally ascribed to fuel poverty and ignorance, although this has not been formally investigated. More

insight is required into people's use of CCA-wood on their home fires so that more effective methods for discouraging its use can be developed.

3 Ambient air monitoring

3.1 Introduction

This chapter includes all the components of the research that relate directly to the ambient air monitoring programme that are designed to meet Aims 1, 2 and in part Aim 3 of this thesis. Specifically this involves determining the impact of intra- and inter-annual variation for assessing likely 'compliance' with the national ambient air quality guideline (Aim 1); investigating the relationship between arsenic and other measured variables to evaluate evidence for domestic burning of CCA-treated timber as the primary source (Aim 2); estimating annual and winter means and the daily maxima of arsenic concentrations that could be used as inputs for a human health risk assessment (Aim 3); and building a simple explanatory model that could be used to estimate arsenic concentrations for future years or other locations that do not have monitoring data (Aim 1).

Details of the methodology are provided in Section 3.2, and results and their interpretations are discussed in Section 3.3.

3.2 Methodology

3.2.1 Monitoring site

The ambient air monitoring site was located in the town of Wainuiomata, population estimate 16,786 (Statistics New Zealand 2013 Census), in the Wainuiomata valley, Lower Hutt. This is the same location used by the Greater Wellington Regional Council (GWRC) for monitoring ambient PM_{10} and $PM_{2.5}$ using a non-gravimetric method and co-located meteorological parameters. Collection of PM_{10} by gravimetric method was undertaken at the same location so that concentrations of arsenic and other elements could be determined as well as providing comparison of PM_{10} data from two co-located instruments.

The Wainuiomata valley is located east of the Hutt Valley and 20 km northeast of central Wellington. The valley is sheltered by surrounding hills to the west (300 m), north (600 m) and east (800 m). To the south, the valley narrows to a constricted valley that runs approximately 20 km to the ocean.

These topographical features mean that the Wainuiomata valley is essentially an enclosed valley system with a wider observed temperature range and lower average wind speeds than other more coastal locations, such as Wellington city and the Kapiti Coast (Griffiths, 2011). During winter, light wind and clear sky conditions lead to cold air drainage and ponding into the valley, resulting in the formation of low-level temperature inversions (Griffiths, 2011).

Within the town of Wainuiomata, the monitoring site was located at the Wainuiomata Bowling Club greens on Moohan Street, at an elevation of 87 m, NZTM E 1763651 N 5429683 (Figure 3.1) which is approximately 1 km southeast of the main shopping centre. The predominant land use around the site is residential, with some adjacent recreational activities such as a swimming pool complex and rugby fields.

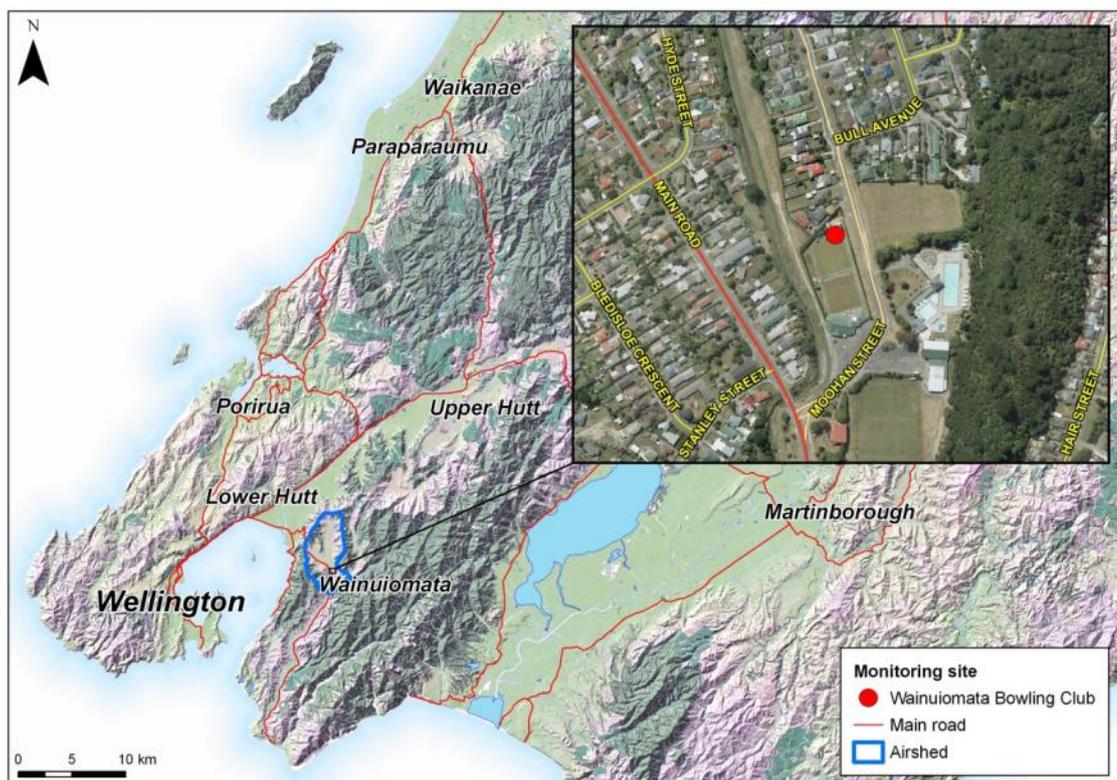


Figure 3.1: Monitoring site located at the Wainuiomata Bowling Club

3.2.2 Data collection and sampling methods

3.2.2.1 Overview

The preferred reference method for measurement of arsenic in air, outlined further below, involves chemical analysis of filters collected using the high volume method for gravimetric determination of PM₁₀, following United States Environmental Protection Agency (US EPA) protocols (MfE, 2002). Applied to arsenic, this method assumes that a) almost all arsenic in air is likely to be in particulate (rather than vapour) form, and b) this arsenic will be present in the PM₁₀ fraction (Maggs, 2000). As the fraction retained by the lungs, PM₁₀ should also represent the best estimate of inhalation exposure to arsenic in air.

3.2.2.2 Sampling frequency and rationale

Typically, reference methods used for monitoring concentrations of metals in particulate matter by the gravimetric method rely on a non-automatic manual 'batch' sampling approach following US EPA protocols. Regulatory monitoring requirements usually adopt a consecutive 1-in-3 day or 1-in-6 day sampling regime which is deemed sufficient to calculate a representative annual average from a long term sequence of low-time resolution data (Bortnick & Stetzer, 2002). The non-consecutive sampling frequency also reflects the resource-intensive nature of non-automatic sampling that requires manual filter changes and off-site laboratory preparation and analysis. This sampling strategy may not be appropriate in situations where data variance is high (e.g., due to strong seasonality) or where health effect assessment from short term exposure periods (e.g., winter only or 24-hour) is required.

Previous elemental speciation of particulate matter at both Raumati South (Kapiti Coast) and Wainuiomata, which was undertaken for receptor modelling (source apportionment) indicates that arsenic concentrations are very low or non-detectable during the non-winter period and are elevated and highly variable during the winter months when people use fires for home heating. Therefore, in order to account for the strong seasonal variability in arsenic concentration, sampling frequency was increased from 1-in-3 day to 1-in-2-day during the winter months (May to August inclusive) over the two year monitoring period. As a result of this sampling strategy, the population

estimate of the mean will be more precise as the more variable strata (winter) was sampled more intensively than the less variable strata (non-winter). Section 3.3.4 discusses the method for taking into account the difference in sampling frequency when calculating the annual means.

Sampling was undertaken from 22 October 2011 until 29 October 2013, providing a 24-month period that captures two winters and enables inter-annual variation to be assessed.

3.2.2.3 PM₁₀ gravimetric sampling

PM₁₀ was sampled using a high volume sampler (Lear Siegler, serial no. A107) fitted with a PM₁₀ size selective inlet (Graseby Andersen, serial no. 4491). This instrument was operated in accordance with AS/NZS 3580.9.6:2003 *Method 9.6: Determination of suspended particulate matter – PM₁₀ high volume sampler with size selective method – gravimetric method*¹¹. The instrument was calibrated monthly using a critical orifice calibrator (Lear Siegler, serial no. 026) to ensure flow rate was maintained between 65 and 75 m³ per hour as required by AS/NZS 3580.9.6:2003.

Sequentially numbered 8x10 inch Whatman grade QM-A quartz filters were conditioned and weighed at constant humidity by GNS Science in accordance with AS/NZS 3580.9.6:2003. These filters were exposed for 24-hours (midnight to midnight) on a one-in-three (1:3) day sampling regime which was increased to a one-in-two (1:2) day frequency between May and August. After exposure, filters were re-conditioned and re-weighed and then stored at -18°C to minimise loss of the volatile fraction of particulate matter prior to further analysis.

During the monitoring campaign, 14 field blanks were deployed for quality assurance purposes to ensure that filter handling and transport did not bias results. Field blanks are filters that were subject to the same handling procedures as sample filters except that the high volume sampler was not turned on and the filter was mounted in the sampler and then removed immediately. Likewise, laboratory blanks are filters that were prepared for use but were not removed from the laboratory environment to

¹¹ This standard is based on US EPA 40 CFR Part 50, Appendix J – Reference Method for the Determination of Particulate Matter as PM₁₀ in the Atmosphere, which is the monitoring method recommended by the ambient air quality guidelines (Ministry for the Environment, 2002).

ensure that laboratory conditions were not biasing results. In total, 10 laboratory blanks were processed. According to AS/NZS 3580.9.:2003, field and laboratory blanks should be used at a frequency of not less than 1 in 20 (5%), which in this case would equal 14 filters.

PM₁₀ concentrations per filter were calculated using Equation 3.1 corrected to standard temperature and pressure (STP), i.e., 273.15 K (0°C) and 101.3 kPa, using barometric pressure (hkPa) and temperature (°C) measurements from 2 m above ground level at the GWRC weather station at Wainuiomata Bowling Club. Volumetric flow rate was maintained at 70 m³ per hour based on the instrument's flow rate set point.

Equation 3.1:

$$C_{PM10} = \frac{(m_f - m_i) \times 10^3}{V}$$

Where:

- C_{PM10} = concentration of PM₁₀ (µg/m³)
- m_f = final mass of filter (mg)
- m_i = initial mass of filter (mg)
- V = volume corrected to STP, calculated as:

$$V = Q \times t \times \frac{273}{T_1} \times \frac{P_1}{101.3}$$

Where:

- V = volume of air sampled (m³) corrected to reference conditions of 0°C and 101.3 kPa
- Q = volumetric flow rate (m³/minute)
- t = sampling time (minutes)
- T_1 = mean ambient temperature over sampling period (K)
- P_1 = mean ambient barometric pressure over sampling period (kPa)

3.2.2.4 PM₁₀ and PM_{2.5} continuous sampling

In addition to gravimetric 24-hour PM₁₀ measurements, PM₁₀ concentration was also measured by GWRC using a Thermo Scientific series FH62 C14 Beta Attenuation Monitor (BAM). This method is an US EPA Automated Equivalent Method for PM₁₀ monitoring (EQPM-1102-150). The sample inlet heater was set to 40°C, in accordance with recommended best practice (MfE, 2009). PM_{2.5} concentrations were also measured with a BAM fitted with a PM_{2.5} Very Sharp Cut Cyclone (VSCC) after the PM₁₀ head. This method is not US EPA equivalent because the BAM is an older model without a dynamic heating system. The filter tape was set to advance at midnight for both instruments, which were also both set to correct PM mass concentrations to 0°C and standard pressure.

3.2.2.5 Determination of arsenic concentration in PM₁₀

Four subsamples consisting of a 47 mm diameter punch was taken by the author from each exposed high volume PM₁₀ filter at the GNS Science laboratory facilities (Figure 3.2). Punches were taken using a stainless steel machine-milled cylinder and the cutting edge of the punch was cleaned with laboratory grade ethyl alcohol between each filter. A total of 303 filter punches (including 14 field blanks and 10 lab blanks) were couriered to Hill Laboratories (Hamilton) in a chilly bin in two batches. The first batch was sent for analysis 6 December 2012 (144 filters) and the second batch was sent on 16 August 2014 (159 filters).

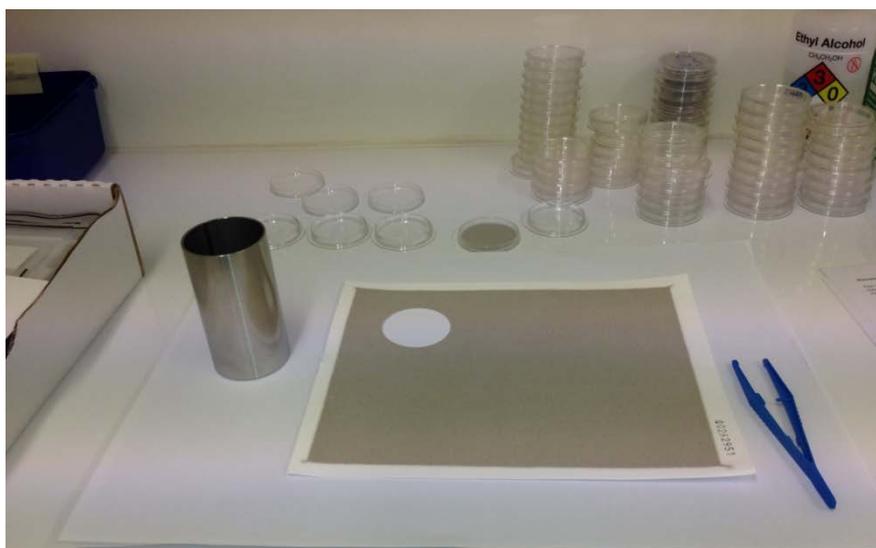


Figure 3.2: Laboratory working space and equipment used for filter punching

The reference method (US EPA) for determining arsenic concentrations in PM₁₀ is acid digestion followed by Atomic Absorption Spectrometry (AAS) or an equivalent method. The equivalent method used by Hill Laboratories was ICP-MS (Inductively Coupled Plasma Mass Spectrometry) which is faster, more precise and more sensitive than AAS. Hill Laboratories used a modified *aqua regia* digestion NIOSH method 7303, issue 1.

The concentration of arsenic in air was calculated as per Equation 3.2.

Equation 3.2:

$$C_{As} = \frac{C_s \times 1000}{V \times p}$$

Where:

- C_{As} = arsenic concentration in air (ng/m³) at STP
- C_s = arsenic concentration in test solution (µg/sample)
- V = volume of air sampled (m³) corrected to reference conditions of 0°C and 101.3 kPa
- p = proportion of filter used (unitless), calculated as:

$$p = \frac{\text{Area filter punch}}{\text{Total area exposed filter}}$$

Where:

- Filter punch* = 17.33 cm²
- Exposed filter* = 419 cm²

The limit of detection (LOD) reported by Hill Laboratories for arsenic obtained by ICP-MS is 0.05 µg per sample. Therefore the LOD for arsenic in ambient air (ng/m³) may differ slightly for each exposed filter depending on the volume of air sampled in that 24-hour period. In this case the LODs for arsenic concentrations ranged from 0.74 to 0.78 ng/m³.

3.2.2.6 Determination of black carbon and multi-element concentrations

Black carbon is the solid fraction of particulate matter that absorbs light strongly. Black carbon measurements (taken from one of the quadruplicate set of PM₁₀ filter punches) were obtained by GNS Science using a light reflectance method with a M43D Digital

Smoke Stain Reflectometer (Ancelet, Davy, Trompetter, Markwitz, & Weatherburn, 2011a).

Multi-element concentrations in PM₁₀ (using one of the quadruplicate set of PM₁₀ filter punches) were determined using X-Ray Fluorescence spectroscopy (XRF) by GNS Science at the New Zealand Ion Beam Analysis Facility. A general description of the XRF method used for ambient air filters is provided in Davy and Ancelet (2014).

3.2.3 Data analysis

3.2.3.1 Handling of measurements below the limit of detection

Thirty-four per cent of the filters analysed by Hill Laboratories reported arsenic concentrations below the laboratory limit of detection (LOD). In this case the concentration of arsenic was known only to be somewhere between zero and the ICP-MS detection limit of 0.05 µg/sample. These measurements are considered too uncertain to report as a single number and so were reported as “<0.05”. There is no regulatory guidance for how to incorporate results below the LOD when calculating an annual average to be compared against national ambient air quality guidelines.

In environmental chemistry, a variety of approaches can be used for incorporating values reported at below the LOD into statistical analyses. Typically, substitution methods are used, for example, all values reported below LOD can be set to zero, set at half the LOD or at the LOD threshold. Alternatively, the actual non-reported laboratory values (if available) can be used. In this thesis the LOD was set to zero for calculating summary statistics for use in the health risk assessment and for comparing against the regulatory guidelines for consistency with previous studies. In the statistical modelling, all values below the LOD were set equal to the LOD and were converted to concentrations based on volume of air sampled in the 24-hour exposure period.

3.2.3.2 Graphical presentation and statistical analyses

Data analysis was carried out using R version 3.1.0 (R Core Team, 2014) to assess data distributions and look for potential correlations and dependencies among variables. Data aggregation, time series, scatterplots, time variation, correlation plots and the wind rose plots were produced using the R package *Openair* (Carslaw & Ropkins,

2012). Box plots, density plots and summary statistics were produced in the default *Base R* package.

A non-parametric bootstrap method was used to calculate means and associated confidence intervals for the metrics required for objectives underpinning Aim 1 (Chapter 1). This method was used as it was not possible to calculate a confidence interval following the *post hoc* weighting of the mean to account for the non-uniform monitoring frequency. A further advantage of the non-parametric bootstrap method is that statistical inference does not need to be based on the assumptions of a particular probability distribution (e.g., normal) as it is based on the assumption that the empirical sample observations represent the population distribution (Efron & Tibshirani, 1986). The bootstrap analysis employed the *sample* function in the default *Base R* package to calculate means for replicate sample observations and two-sided confidence intervals at the 0.05 significance level, these being the 0.025 and 0.975 quantiles of the frequency distribution of 10,000 bootstrapped means for the period of interest, i.e., annual, winter and non-winter. The major assumption for valid inferences based on the bootstrap method and for correlations between time series variables is that the observations in the original sample are independent of one another (Zieffler, Harring, & Long, 2011). In environmental time series data there is potential for autocorrelation, that is, temporal correlation in which an observation on one day (x_t) may be correlated with the previous day's observation (x_{t-1}). Air quality and meteorological time series are prone to positive autocorrelation due to physical processes. For instance, higher than normal temperature on one day is often associated with higher than normal temperature on the next day (Weatherhead et al., 1998). Violation of the independence assumption can result in systematic underestimation of variation in a population which leads to erroneous inference (Zieffler et al., 2011).

Principal Components Analysis (PCA) was used with the multi-element data set and the *prcomp* function in the *Stats R* package was used to identify elements that are associated with each other (i.e., co-vary) and therefore may indicate a common emission source.

Simple multiple linear regression using the least squares estimator was used to model the dependence of arsenic on other measured variables using the *lm* function in the default *Base* R package. Potential candidate predictor variables were identified using recursive partitioning and regression tree analysis using *rpart* function in the *Rpart* R package, using Pearson's correlation coefficients, and as having physical plausibility. Diagnostics for the linear regression used plots of the fitted model residuals (error term) to check for violations of assumptions underpinning model validity, i.e., linear relationship between mean response and predictor, independence, constant variance and approximately normal distribution. Omnibus F tests were carried out using the *anova* function in R to ensure that the most parsimonious model was selected, i.e., best fit for the least number of predictor variables.

3.2.3.3 Regulatory guideline comparison

The New Zealand ambient air quality guideline for inorganic arsenic is expressed as an annual average of arsenic in PM₁₀ of 5.5 ng/m³ (MfE, 2002; Section 2.3.2). Although not stated in the MfE guidelines, it is the norm to express annual average based on a calendar year. In this study the timing and duration of the monitoring campaign did not match calendar years. For comparing results to the MfE guideline value, the annual averages were taken as the period 1 November to 31 October and were weighted to account for non-uniform sampling frequency.

There is no regulatory guidance for how to incorporate results below the LOD when calculating an annual average to be compared against national ambient air quality guidelines. In this case all observations below the LOD were set to equal zero as outlined in Section 3.2.3.1).

3.3 Results and discussion

3.3.1 Numerical summary of arsenic, PM₁₀, PM_{2.5} and black carbon results

In total 279 PM₁₀ filters were collected between 22 November 2011 and 29 November 2013. All field and laboratory blanks for gravimetric PM₁₀ measurements were well within the tolerances for variations in mass in quartz filters of ± 5 mg and ± 8 mg respectively, in accordance with AS/NZ 3580.9.6:2003. The field and laboratory blanks all yielded arsenic concentrations below the analytical level of detection by ICP-MS. Statistical summaries for observations during the monitoring period are shown in Table 3.1. Daily arsenic concentrations ranged from <0.7 to 79.2 ng/m³, with an overall mean (unweighted for sampling frequency) of 7.3 ng/m³. Over the same period, particulate matter ($\mu\text{g}/\text{m}^3$) ranged from 0.1 to 33.8 for PM₁₀ (gravimetric), 1.0 to 32.2 for PM₁₀ (BAM) and 0.1 to 30.8 for PM_{2.5}. The overall mean values of PM₁₀ were 11.4 $\mu\text{g}/\text{m}^3$ (gravimetric) and 10.3 $\mu\text{g}/\text{m}^3$ (BAM). Black carbon ranged from 0.1 to 5.7 $\mu\text{g}/\text{m}^3$.

3.3.2 Inter-method agreement for PM₁₀ and PM₁₀:PM_{2.5} ratio

In general there was good fit between PM₁₀ measurements obtained by the co-located gravimetric and BAM methods ($R^2 = 0.88$). A scatterplot of these results is shown in Figure 3.3. There appeared to be a tendency for a seasonal effect on the magnitude and sign of the difference between the co-located PM₁₀ measurements, with less variation in the difference between the methods and a tendency for the gravimetric method to produce higher concentrations than those obtained by the BAM during non-winter months. This effect is shown in Figure 3.4. The reason for this is not known as previous New Zealand work has found that BAMs tend toward over-measurement relative to gravimetric in all seasons, but less so during winter (Bluett et al., 2007).

Overall, PM_{2.5} contributed just over half of the PM₁₀ measured concentration, with the overall mean and median ratios of PM_{2.5}/PM₁₀ (BAM) being 0.58 and 0.52 , respectively. These ratios are strongly seasonally dependent, with PM_{2.5} comprising the majority of PM₁₀ in the winter months due to the dominance of the wood smoke contribution to ambient particulate matter (Figure 3.5).

Table 3.1: Summary statistics for arsenic, PM₁₀, PM_{2.5} and black carbon (24-hour averages)

Variable	Units	Winter (1:2-day sampling) average	Non-winter (1:3-day sampling) average	All the data (un-weighted for sampling frequency)
Arsenic				
Min	ng/m ³	<0.7	<0.7	<0.7
Mean	ng/m ³	14.6	1.8	7.3
Median	ng/m ³	8.2	0.7 (0)	1.7
Maximum	ng/m ³	79.2	55.3	79.2
SD	ng/m ³	16.1	5.2	12.9
Variance	(µg/m ³) ²	259.5	27.1	166.5
CV	%	110	288	176
n		120	159	279
PM₁₀ (gravimetric)				
Min	µg/m ³	1.0	1.4	0.1
Mean	µg/m ³	13.0	10.1	11.4
Median	µg/m ³	11.1	9.7	10.4
Maximum	µg/m ³	33.8	23.9	33.8
SD	µg/m ³	7.0	4.8	6.0
Variance	(µg/m ³) ²	48.4	23.1	35.9
CV	%	54	47	52
n		120	159	279
PM₁₀ (BAM)				
Min	µg/m ³	2.5	1.4	1.0
Mean	µg/m ³	12.1	8.9	10.3
Median	µg/m ³	11.0	8.3	9.6
Maximum	µg/m ³	32.2	23.4	32.2
SD	µg/m ³	6.1	4.2	5.3
Variance	(µg/m ³) ²	37.4	17.7	28.5
CV	%	50	47	51
n		118	158	276
PM_{2.5} (BAM)				
Min	µg/m ³	1.2	0.1	0.1
Mean	µg/m ³	9.1	4.5	6.9
Median	µg/m ³	6.6	4.1	5.1
Maximum	µg/m ³	30.9	12.7	30.9
SD	µg/m ³	6.7	2.1	5.6
Variance	(µg/m ³) ²	44.7	4.5	31.3
CV	%	74	47	81
n		116	100	216
Black carbon				
Min	µg/m ³	0.2	0.1	0.1
Mean	µg/m ³	2.2	0.6	1.2
Median	µg/m ³	1.7	0.5	0.7
Maximum	µg/m ³	5.7	3.3	5.7
SD	µg/m ³	1.4	0.5	1.2
Variance	(µg/m ³) ²	2.0	0.2	1.5
CV	%	64	83	100
n		120	159	279

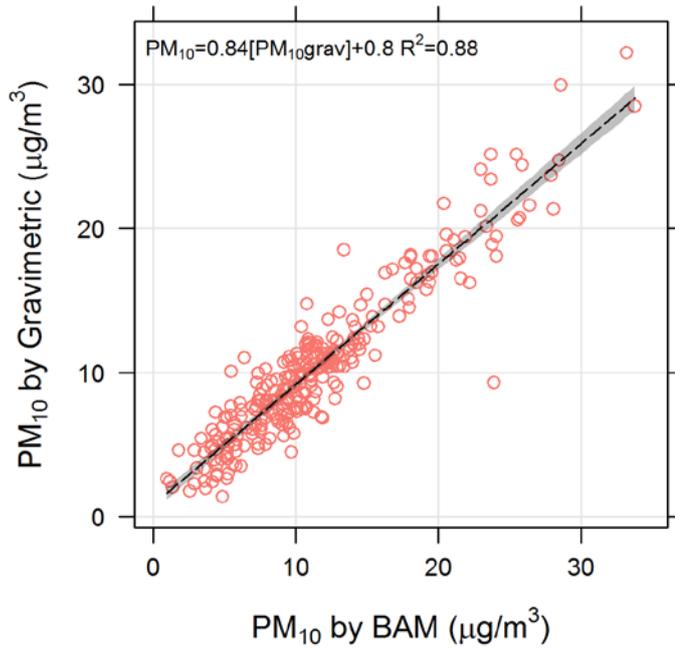


Figure 3.3: Scatterplot showing linear fit of PM₁₀ gravimetric regressed against PM₁₀ BAM measurements

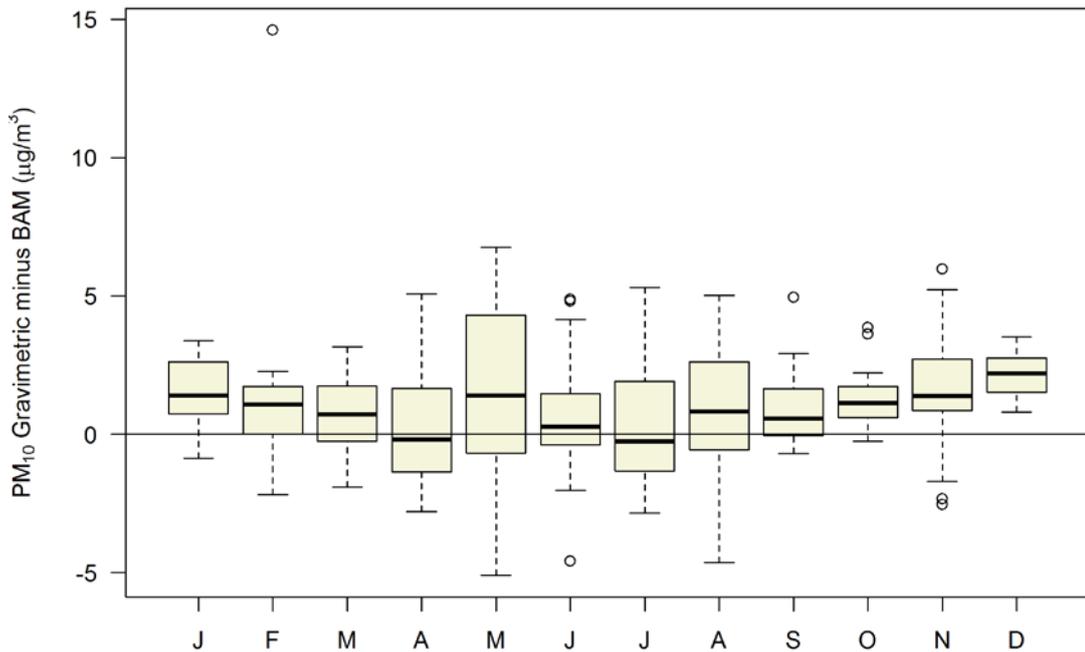


Figure 3.4: Boxplot of differences between PM₁₀ gravimetric and PM₁₀ BAM (24-hour average) measurements aggregated by month

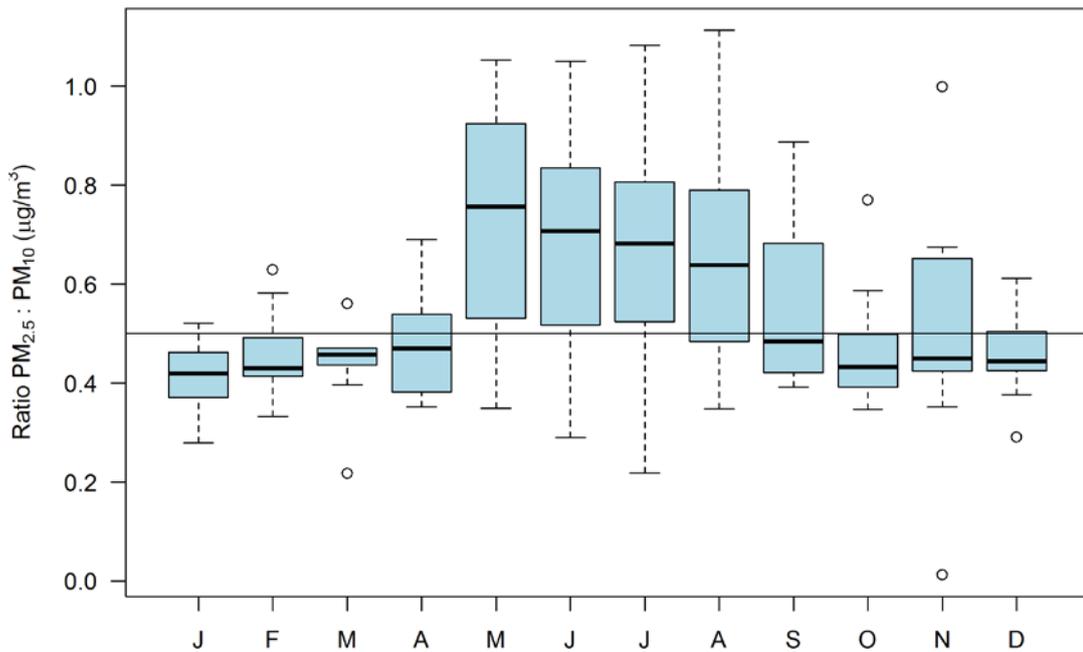


Figure 3.5: Boxplot of $PM_{2.5}:PM_{10}$ ratios for BAM measurements aggregated by month

3.3.3 Local meteorology at monitoring site

One-hour meteorological measurements were obtained from the 10 m meteorological mast co-located at the Wainuiomata air quality monitoring station. Wind roses for the monitoring period by season show the year-round dominance of winds from the northerly quarter and the lower wind speeds during autumn and winter (Figure 3.6). The relative lack of westerly and easterly winds is a consequence of the valley topography being orientated north to south. The distribution of 1-hour temperature averages and minima for each month showed the expected seasonal effects, i.e., lower temperatures and wind speeds during the winter months compared to the rest of the year (Figure 3.7). Note that June and August in 2013 were warmer and windier than the corresponding months in 2012 as shown in Figure 3.7.

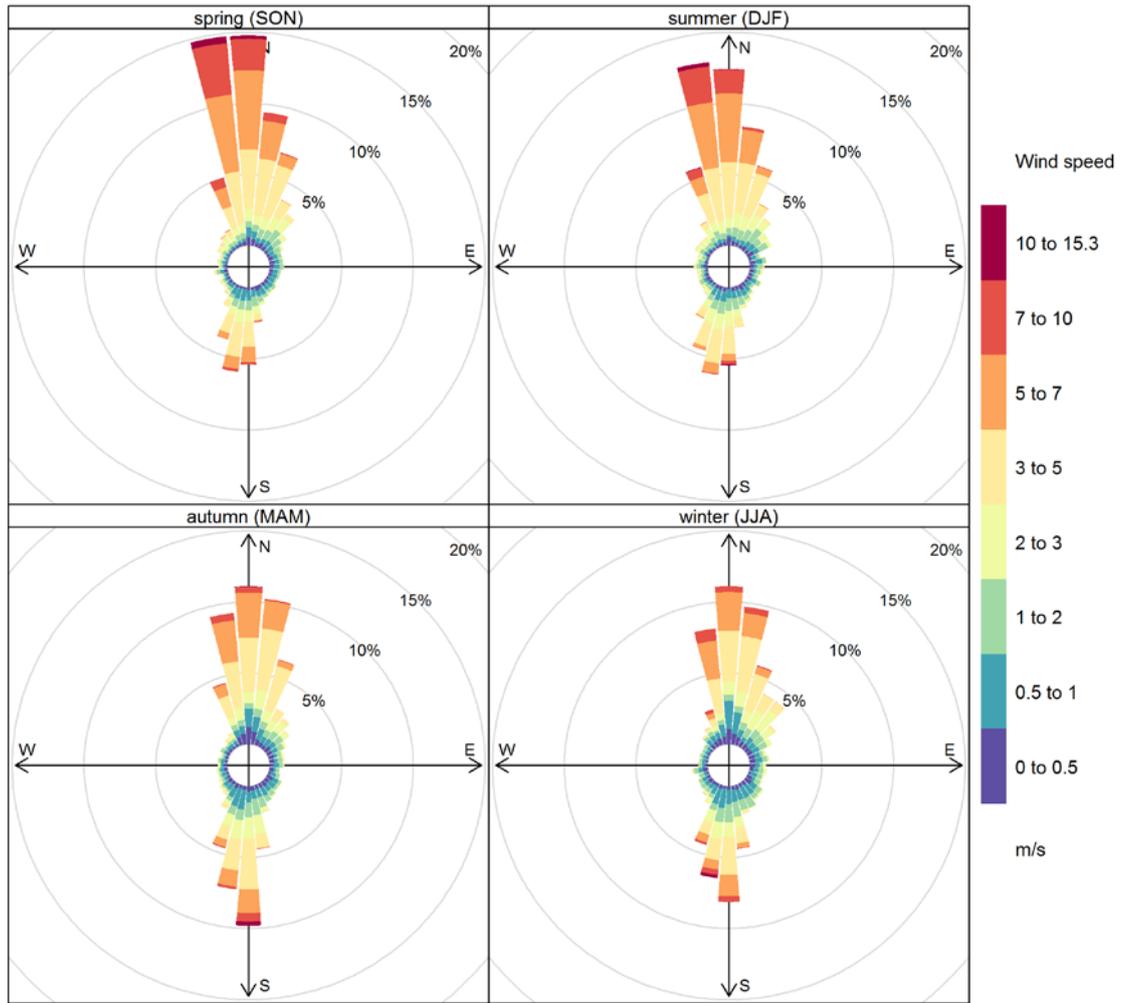


Figure 3.6: Wind roses showing frequencies of hours by wind direction for the monitoring period by season

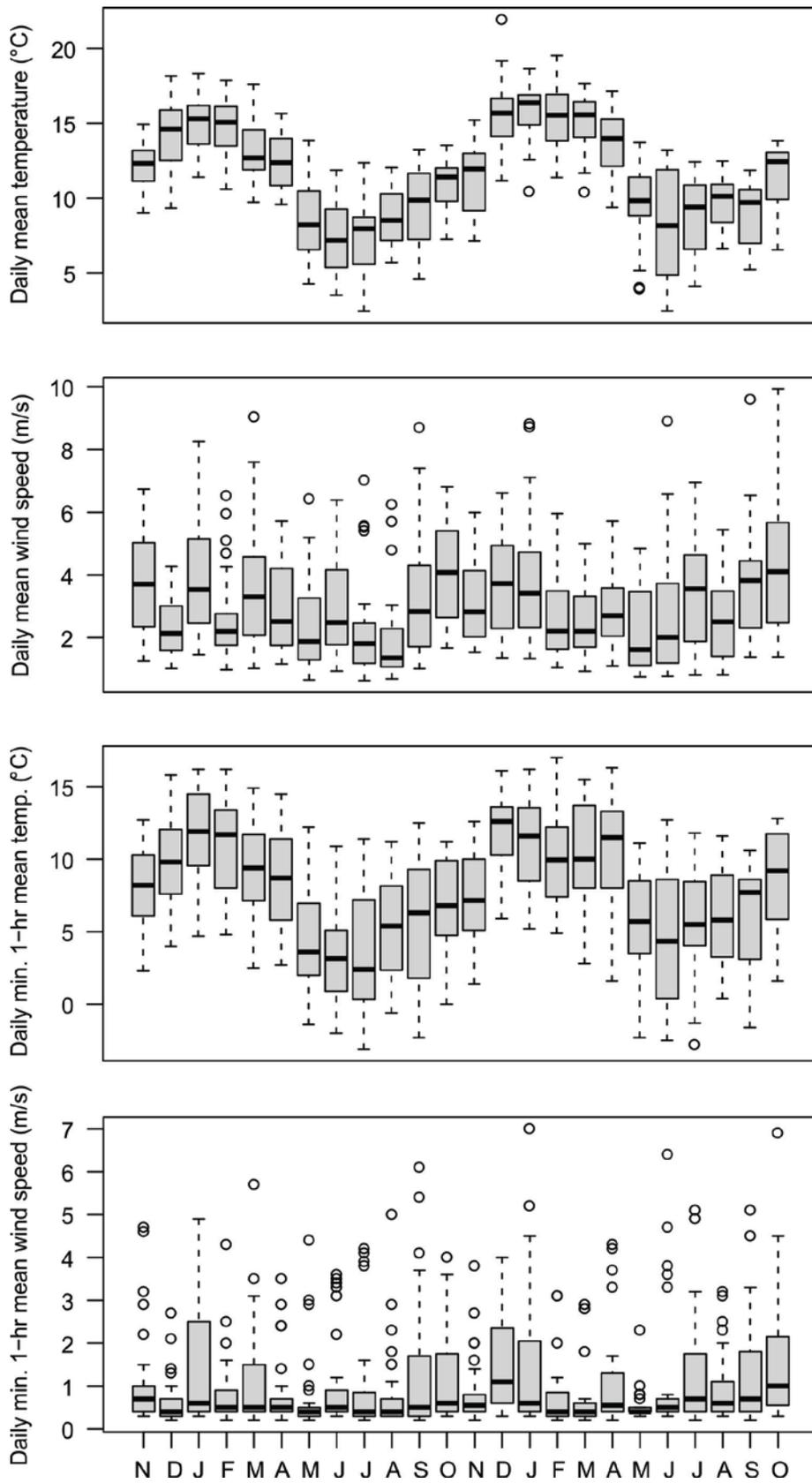


Figure 3.7: Boxplots of temperature and wind speed variables by month

3.3.4 Comparison with regulatory guidelines and values for health risk assessment

The monitoring results were split into two 12-month periods, November to October inclusive, so that two consecutive annual averages could be calculated for comparison with the national ambient air quality guideline value of 5.5 ng/m³ (MfE, 2002). Annual averages used to assess compliance with the guideline were derived using the bootstrap percentile method, to take into account non-uniform monitoring frequency and to produce confidence intervals, and are presented in Table 3.2. The annual averages for the consecutive 12-month monitoring periods were 6.8 ng/m³ and 5.3 ng/m³ respectively. These bootstrapped means were virtually identical to those calculated as an arithmetic mean based on proportion of year of data coverage (Table 3.1). Based on the bootstrapped point estimates, the annual average guideline was not met in the first year and was met in the second year.

Input values for the health risk assessment were taken as 6.1 ng/m³ to represent the long term average outdoor concentration and 14.6 ng/m³ to represent the shorter term winter outdoor concentration.

Table 3.2: Arsenic weighted annual, winter, non-winter means with 95% confidence intervals

12-month or 24-month period	Statistic and units	Non-winter (1:3-day sampling)	Winter (1:2-day sampling)	Weighted annual
1/11/2011 to 31/10/2012	Mean [95% CI], ng/m ³	2.0 [1.1, 3.9]	16.1 [12.0, 20.6]	6.8 [4.8, 8.9]
	Proportion of total days	243/366 (0.6639)	123/366 (0.3361)	366/366 (1)
1/11/2012 to 31/10/2013	Mean [95% CI], ng/m ³	1.3 [0.7, 2.1]	13.1 [9.6, 17.0]	5.3 [3.8, 7.0]
	Proportion of total days	242/365 (0.6630)	123/365 (0.3370)	365/365 (1)
1/11/2011 to 31/10/2013	Mean [95% CI], ng/m ³	1.7 [1.1, 2.6]	14.6 [11.8, 17.6]	6.1 [4.8, 7.4]

The spread of the 95% confidence intervals for the point estimate of the mean for each year indicate it was plausible that either year's mean was above or below the guideline value of 5.5 ng/m³. In addition, the 95% bootstrapped confidence interval for the difference between the two annual means (absolute value 1.5 ng/m³) was -1.0 to 4.0

ng/m³. As this interval includes zero, it follows that there was no statistically significant difference in the true population means between the two sampling years.

Figure 3.8 shows the inter-annual variation in arsenic, PM₁₀ and black carbon concentrations for the two monitoring years. Annual mean PM₁₀ concentrations (based on continuous BAM data) for the two monitoring years were the same, i.e., 10.3 µg/m³, which is well within the national ambient air quality guideline of 20 µg/m³ (MfE, 2002). PM_{2.5} data were only available from May 2012 onwards so an annual average was calculated only for the second monitoring year; at 5.6 µg/m³ this was well within the World Health Organization (WHO) annual guideline of 10 µg/m³ for PM_{2.5} (WHO, 2006). Daily PM_{2.5} concentrations were higher than the WHO guideline threshold of 25 µg/m³ on seven days in the first year and on four days in the second year. Compliance with the WHO 24-hour average guideline for PM_{2.5} is based on no more than three exceedance days per year. There are no regulatory guidelines for black carbon.

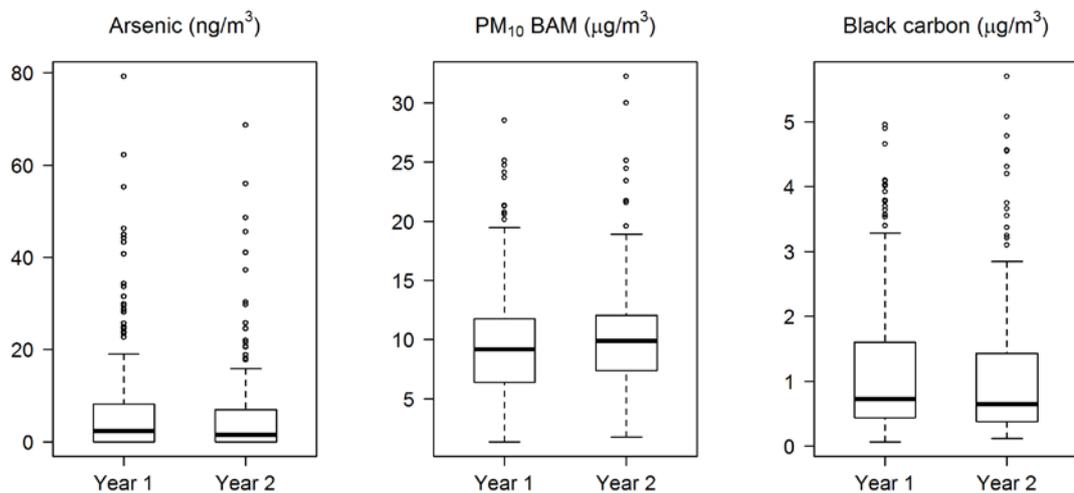


Figure 3.8: Boxplots inter-annual variation in arsenic, PM₁₀ and black carbon

The confidence interval constructed for the point estimates of annual average arsenic concentration quantifies the potential random sampling error associated with not capturing measurements for all the 24-hour periods over the monitoring period. If we had selected different days for sampling then the resulting mean might have been different. The 95% confidence interval represents the range of possible means that

could be obtained 95 times out of 100 times based on that particular sample size. There are two other main areas of uncertainty in our estimation of the mean that have not been taken into account: heterogeneity in arsenic concentrations across the airshed and instrument measurement error. The mean and its associated confidence interval represent only outdoor air at the monitoring site. There is no information on spatial variability of arsenic in other parts of the airshed. However, the monitoring station is thought to be broadly representative of the wider airshed and as the station is located at the valley exit it is likely to be influenced by drainage of polluted air from other parts of the valley.

Measurement error refers to the ability of the monitoring instrument to accurately and precisely measure the contaminant of interest. Accuracy, in terms of whether the instrument is measuring the 'true' quantity of particulate, as defined by the standard reference method, was achieved through appropriate instrument operation and calibration. There is no suitable certified reference approach for determining the concentration of particulate matter in ambient air. Instrument precision, that is, how repeatable the measurements were, was not assessed in this work as this would have required collocation of duplicate instruments which were not available. Analytical uncertainty for ICP-MS measurements made at Hill Laboratories was undertaken as part of normal IANZ Accreditation quality control and quality assurance procedures and therefore not further considered.

3.3.5 Seasonal variation in arsenic, PM₁₀, PM_{2.5} and black carbon

Time series of 24-hour averages for arsenic, PM₁₀, PM_{2.5} and black carbon are presented in Figures 3.9 to 3.13, below, with red dashed lines showing the winter monitoring periods where sampling frequency was increased to 1:2 days.

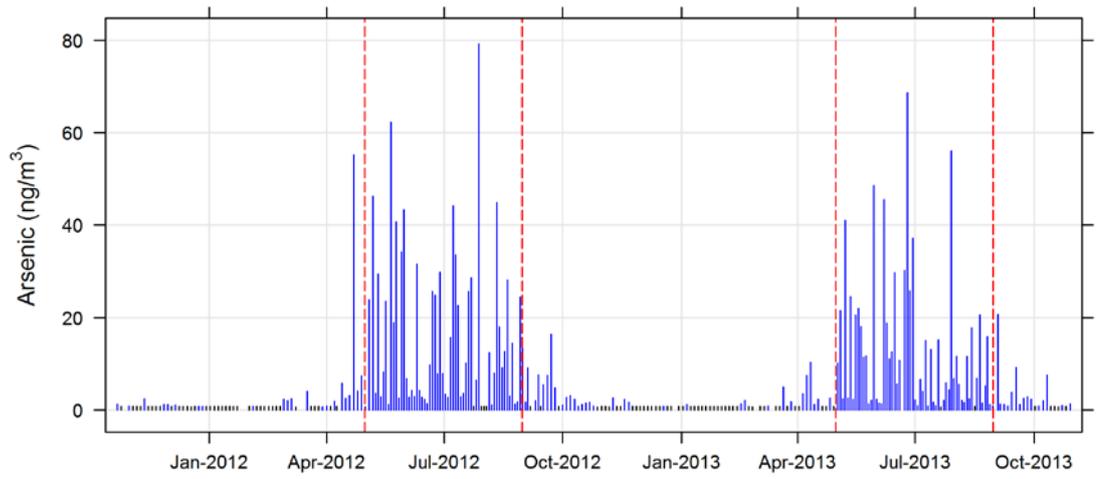


Figure 3.9: Time series of arsenic (24-hr average) with measurements below the LOD in black

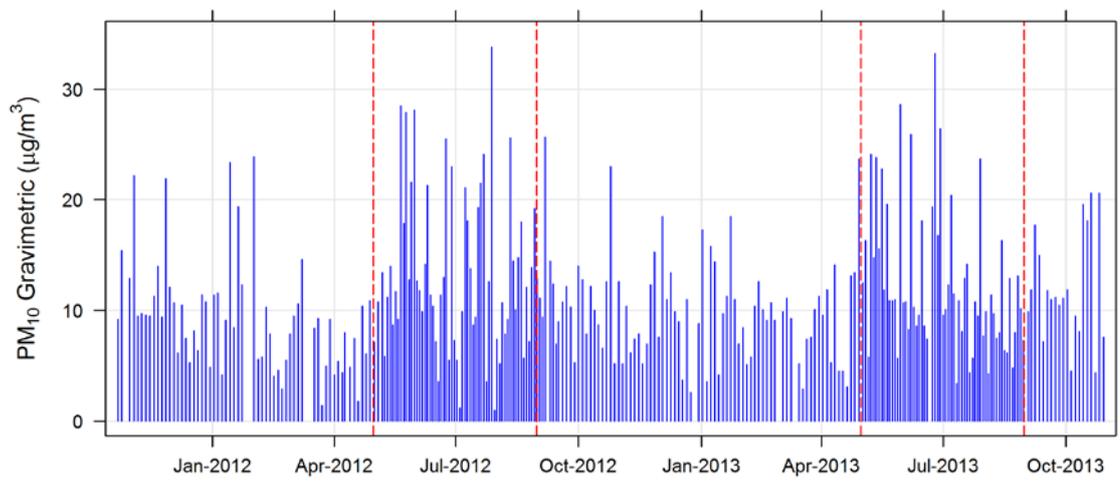


Figure 3.10: Time series of PM_{10} by gravimetric method (24-hr average)

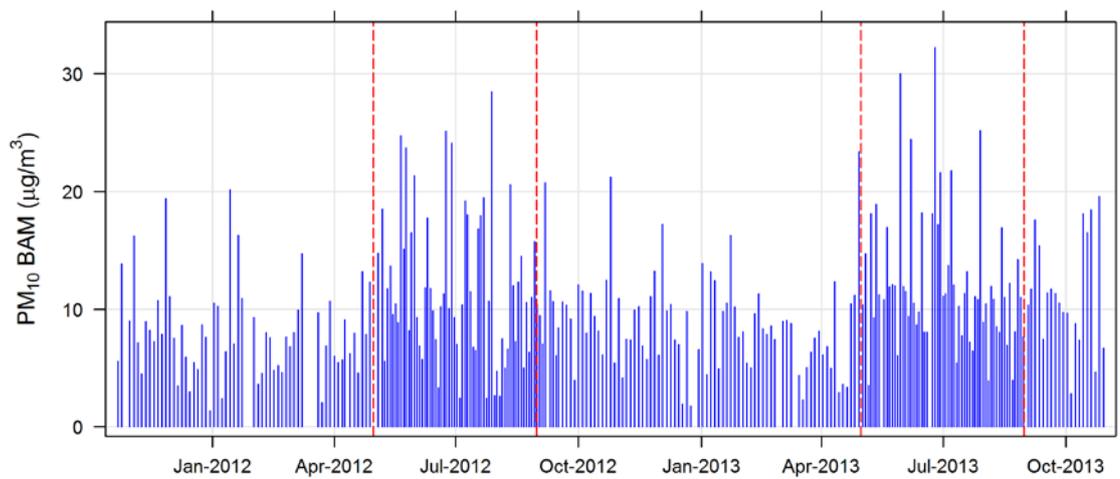


Figure 3.11: Time series of PM_{10} by BAM (24-hr average)

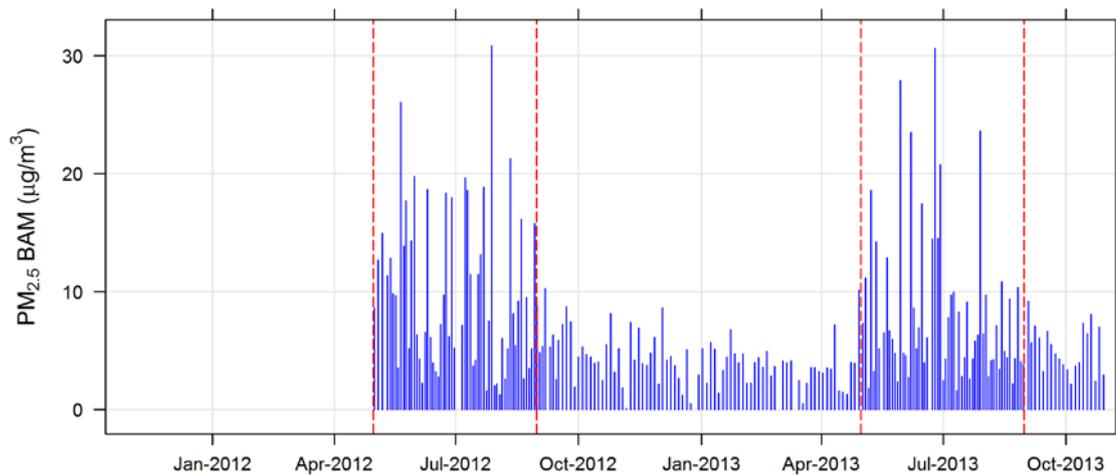


Figure 3.12: Time series of PM_{2.5} by BAM

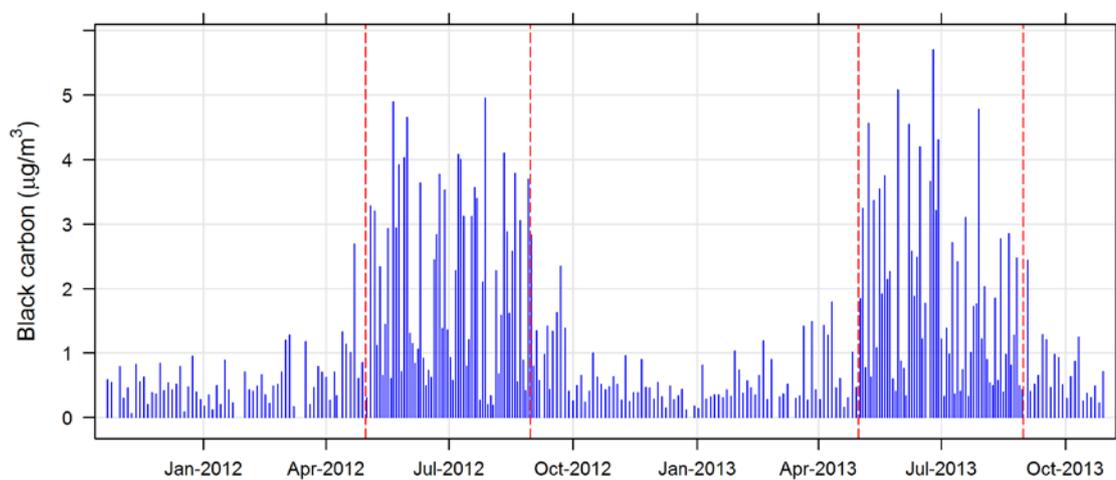


Figure 3.13: Time series of black carbon (24-hr average)

Arsenic measurements were strongly seasonal, with elevated concentrations observed almost exclusively during the winter monitoring period as defined from May to August inclusive. There was not an abrupt seasonal demarcation as arsenic was also present in the ‘shoulder’ months of April and September when home fires are still used, albeit by a lesser proportion of households (Wilton, 2006). Both PM₁₀ gravimetric and BAM measurements showed less seasonal variation than arsenic, PM_{2.5} and black carbon, although the number of 24-hour PM₁₀ averages above 20 µg/m³ was greater during the winter compared to non-winter period (Figure 3.10 and Figure 3.11). The less pronounced seasonality for PM₁₀ is due to the influence of non-combustion sources, e.g., “crustal matter”, “marine aerosol” and “road dust” that are present all year

round, as identified by past receptor modelling (Davy, Trompetter, & Markwitz, 2009). PM_{2.5} and black carbon were strongly seasonal, indicating the inferred impact of domestic fires on concentrations during the winter period. Other receptor modelled sources of PM_{2.5} that have been found to be present throughout the year are “motor vehicles” and “secondary sulphate” (Davy et al., 2009).

In the non-home heating season (January, February, March, October, November, December) average arsenic levels were very low, 1.1 ng/m³ and mostly at levels below the analytical limit of detection. These non-winter levels most likely represent the background levels of arsenic as a naturally occurring trace element in crustal matter. The arsenic, PM_{2.5} and black carbon measurements show enhanced right-skew during the winter months, particularly so for arsenic (Figure 3.14, Figure 3.15 and Figure 3.16). However, arsenic shows a much higher coefficient of variation (CV) compared to the other parameters, indicating a much larger variability compared with the mean (Table 3.1).

For PM₁₀ and PM_{2.5}, these results are as expected and similar patterns are observed elsewhere in New Zealand, with the majority of days (particularly in summer) showing good air quality, a subset of winter days showing degraded air quality, and relatively few days exhibiting the worst air quality. In general the elevated particulate matter concentrations observed throughout New Zealand during the winter period are due to the presence of meteorological conditions that promote “environmental confinement” of biomass burning emissions (Trompetter, Davy, & Markwitz, 2010).

The finding of higher day-to-day variability in winter arsenic levels compared to PM_{2.5} and black carbon is most likely due to greater spatial and temporal variability in the emission sources of arsenic. The number of households using CCA-wood instead of untreated wood will vary from night to night and from place to place within the Wainuiomata airshed.

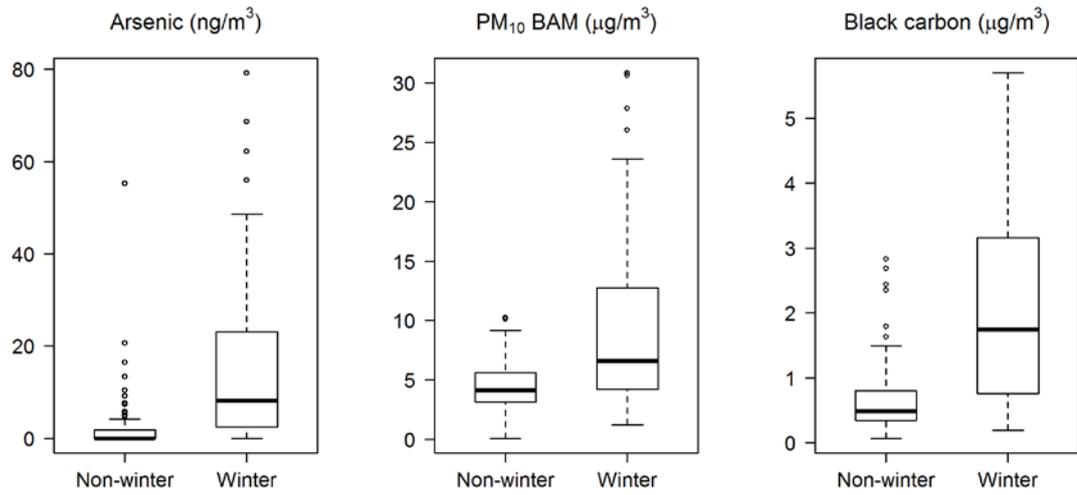


Figure 3.14: Boxplots of arsenic, PM_{2.5} and black carbon by season

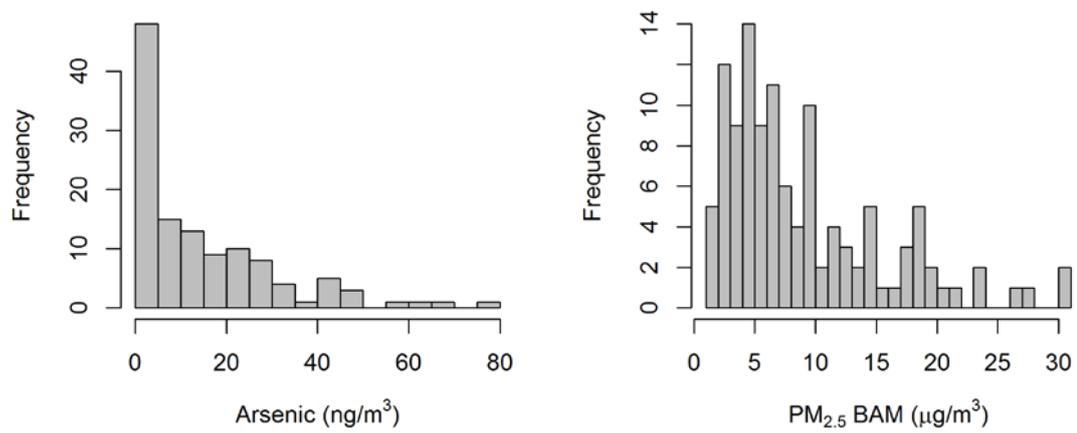


Figure 3.15: Histograms of winter measurements of arsenic (left) and PM_{2.5} (right)

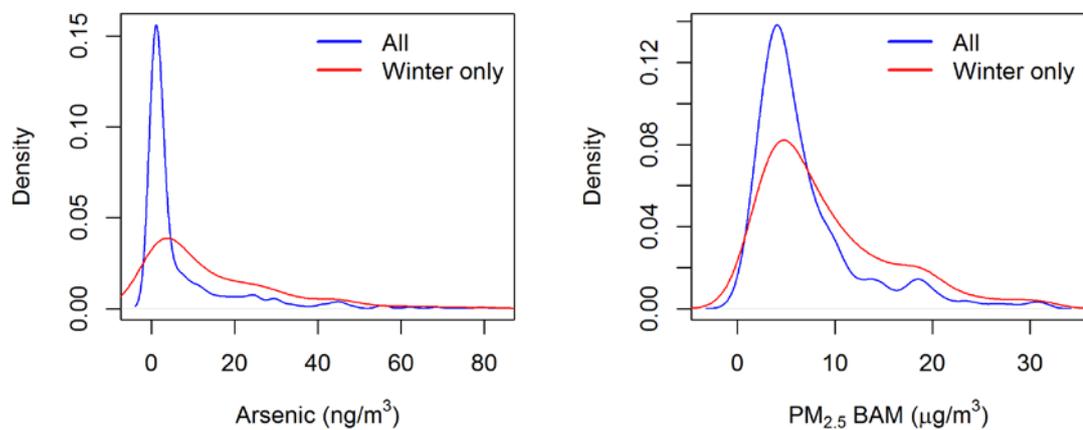


Figure 3.16: Density plots of arsenic (left) and PM_{2.5} (right)

To explore spatial variability in arsenic, a data set of concurrent hourly measurements of arsenic (determined by IBA) and black carbon, obtained by GNS Science at two monitoring stations approximately 1.2 km apart in Masterton during winter 2009 (Ancelet et al., 2012) was examined as part of this work. These data were aggregated into 24-hour averages for this analysis due to the analytical noise associated with the hourly values. At the east Masterton site, both PM₁₀ and black carbon measurements were substantially higher than those observed for the same time period at the west Masterton site. Although there was strong inter-site correlation for black carbon ($r=0.84$) and PM₁₀ ($r=0.78$), the correlation for arsenic was weaker ($r=0.54$). The inter-site difference in PM₁₀ and black carbon concentration has previously been attributed to nocturnal cold air drainage patterns that entrain and transport emissions along a west to east gradient (Ancelet et al., 2012). At the both, sites 24-hour average arsenic was strongly correlated with black carbon: $r=0.75$ (west Masterton) and $r=0.83$ (east Masterton). Interestingly, there was no difference between the winter mean arsenic concentrations measured at each site, even though arsenic levels would be expected to be higher at the east site relative to the west site based on the intra-site correlations between arsenic and black carbon. These findings suggest that local sources, i.e., one, two or more households sited close to a particulate monitoring station burning CCA-wood instead of untreated wood on a particular night, could explain the lack of correlation in day to day arsenic concentrations between the two sites.

3.3.6 Relationships between arsenic, PM₁₀, PM_{2.5}, black carbon and meteorology

As discussed in Section 3.2.3.2, autocorrelation or serial correlation present in air quality time series data can violate the assumption of independence in the data and therefore the results of statistical inference from serially correlated data must be treated with caution. Although autocorrelation was present in the PM_{2.5} time series obtained from consecutive measurement days ($r=0.54$), this was not present when PM_{2.5} observations were matched to the non-consecutive arsenic sample days ($r=0.17$, $p=0.013$). Therefore, the non-consecutive day sampling frequency has removed most of the autocorrelation in the time series.

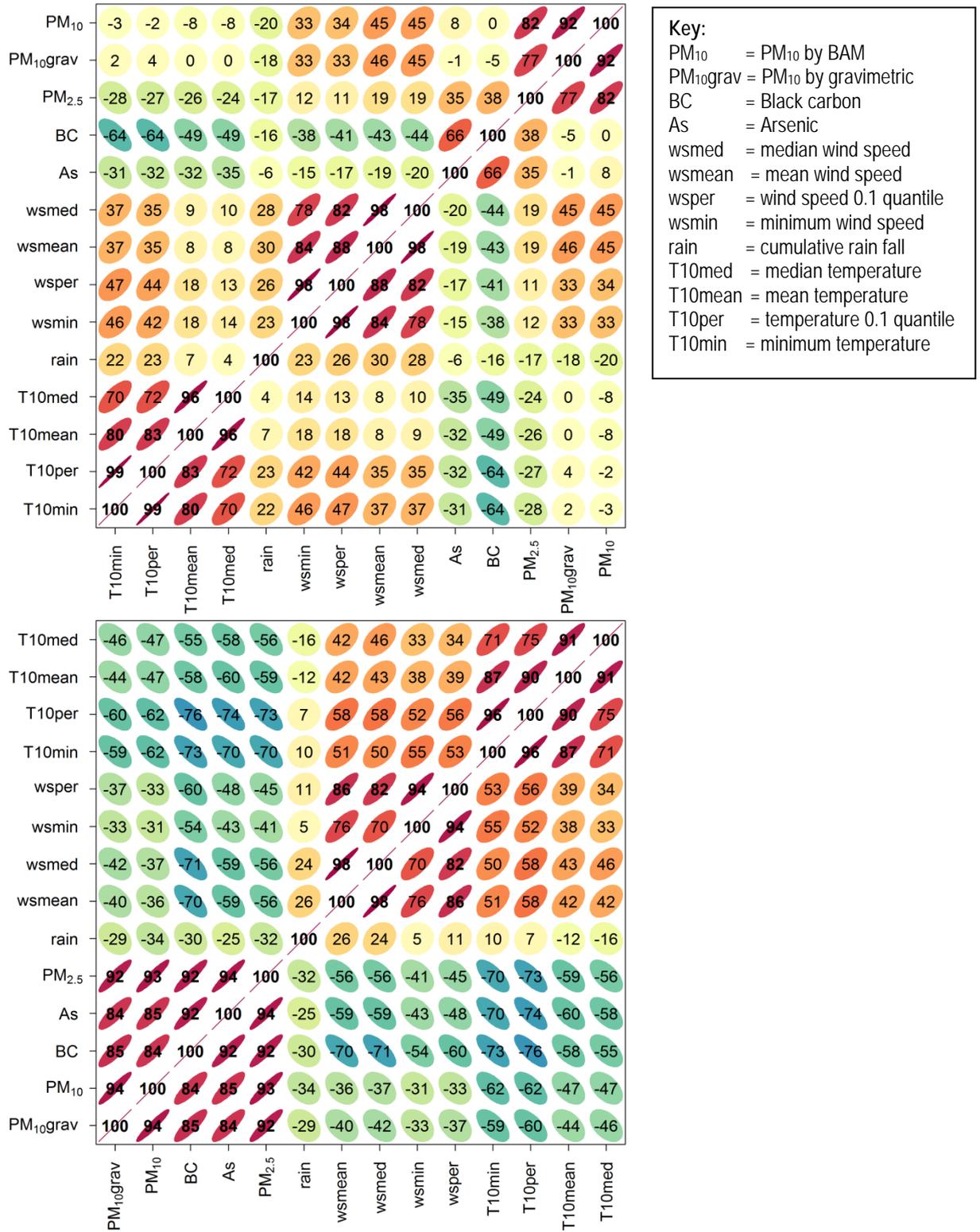


Figure 3.17: Correlation matrices showing Pearson's correlation coefficients (r -values multiplied by 100) between variables for the non-winter (top) and winter (bottom) periods

A correlation matrix showing the strength of the relationship between all pairs of variables in the dataset split by the non-winter ($n=158$) and winter periods ($n=120$) is presented in Figure 3.17. The correlation matrix ellipse shapes and colours indicate strength of correlation, with cool colours indicating negative correlation, and warm colours indicating positive correlation. As expected, arsenic, $PM_{2.5}$ and black carbon measurements are all strongly correlated with each other ($r > 0.90$), especially during the main winter wood burning season (May to August inclusive). These relationships are not unexpected as both $PM_{2.5}$ and black carbon are emitted as incomplete products of combustion from biomass burning. During the winter season, arsenic was most strongly correlated with $PM_{2.5}$ ($r = 0.94$) and most strongly negatively correlated with the 0.1 quantile of 1-hour average temperatures for a 24-hour period ($r = -0.74$), followed by the minimum 1-hour average temperature for a 24-hour period ($r = -0.70$). There was also positive correlation between temperature and wind speed ($r = 0.58$) indicating that colder days also tend to be less windy. During winter, both arsenic and $PM_{2.5}$ concentrations tend to increase together as temperature decreases and wind speeds become lower, as shown in the scatterplot of arsenic versus $PM_{2.5}$ conditioned by levels of the 0.1 quantile of 1-hour temperatures on a given day (Figure 3.18).

A smooth curve was fitted to each of the bivariate plots (in red) using a locally weighted regression procedure (LOWESS) implemented in R (Cleveland, 1979) in order to help visualise any non-linear relationships between variables (Figure 3.19). The relationships between arsenic and $PM_{2.5}$ and black carbon, whilst fairly linear, do show some scatter around the fitted line, indicating that arsenic is not always proportional to $PM_{2.5}$. The curvature in the smoothed fit between black carbon and other variables at peak black carbon concentrations has been observed elsewhere and has been attributed to self-absorption within the sample during the light reflectance measurements, as the thickness of the black carbon loading on the filter increases (Ancelet et al., 2013). The non-linear relationships between arsenic, $PM_{2.5}$, black carbon and wind speed shows that low wind speeds are not always associated with high contaminant concentrations, but that high concentrations always occur at low wind speeds.

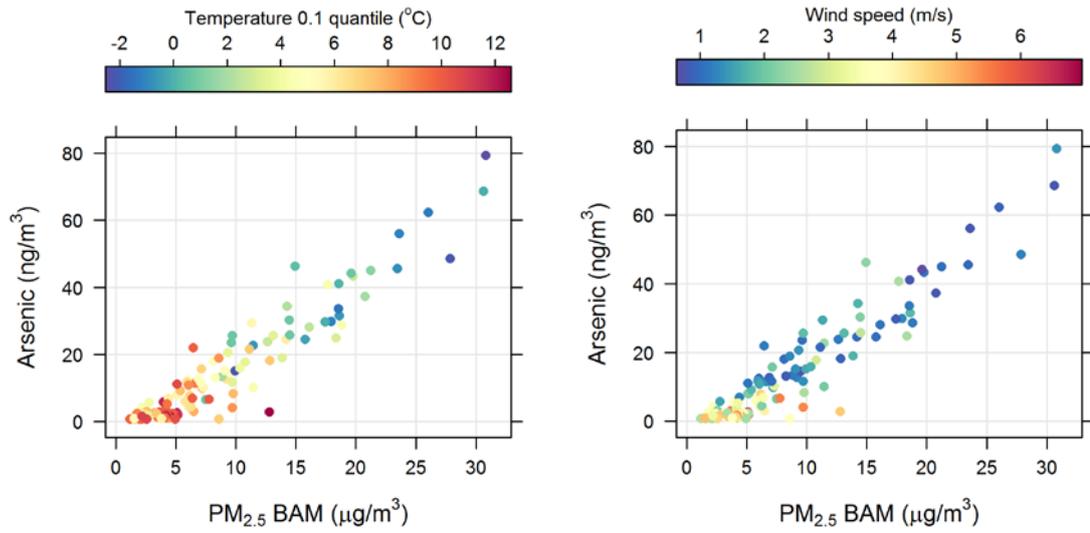


Figure 3.18: Scatterplots of winter arsenic versus $PM_{2.5}$ conditioned by temperature (left) and mean wind speed (right)

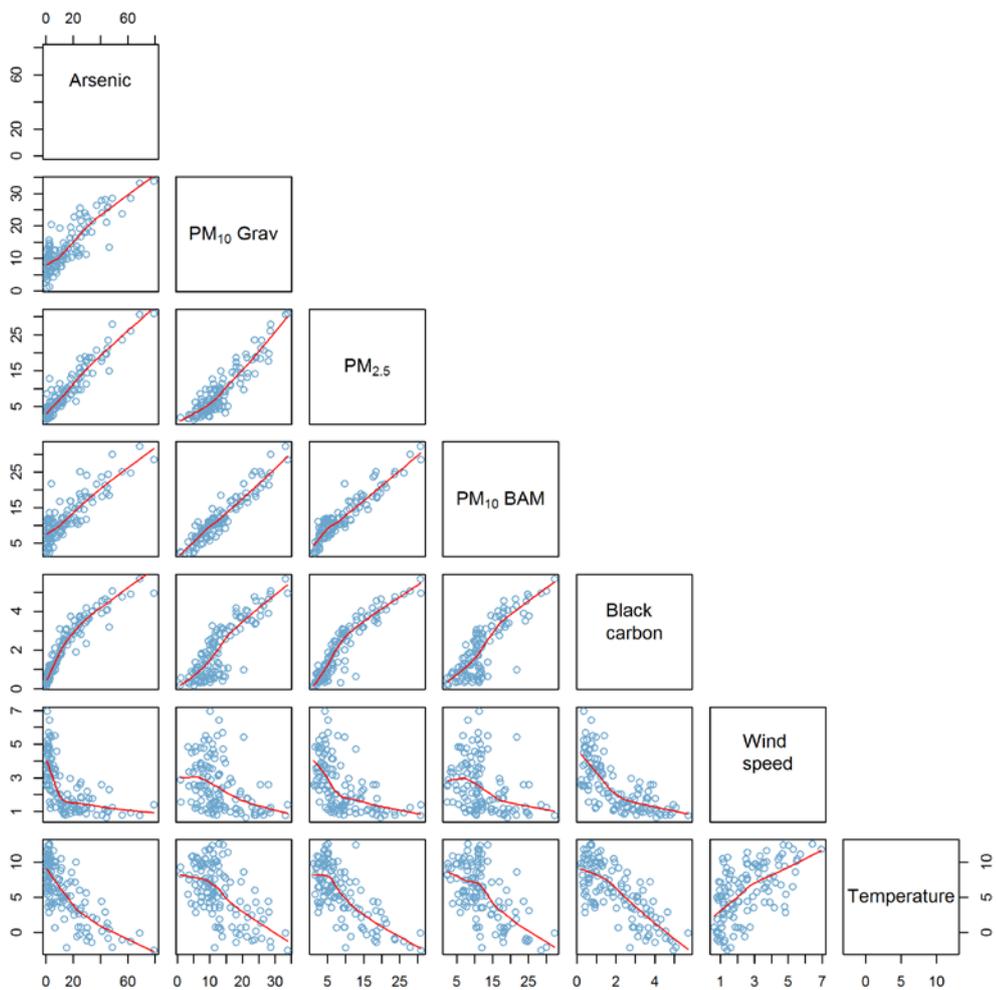


Figure 3.19: Scatterplot matrix of key variables (winter only) with fitted LOWESS curve shown in red

3.3.7 Relationship between arsenic and other trace elements for source attribution

As outlined in Section 3.2.2.6, concentrations of 23 elements from one of the quadruplicate sets of PM₁₀ filter punches (n=279) were determined using XRF by GNS Science at the New Zealand Ion Beam Analysis facility. This elemental speciation data set was used here to explore associations between elements, including arsenic, that may indicate a common emission source of particulate matter. Two techniques were used: a correlation matrix with hierarchical clustering, and PCA. Summary statistics for concentrations of these elements determined by XRF are provided in Table 3.3 and the relative spread of their values are shown in Figure 3.20. Of note is that mean arsenic determined by XRF (4.8 ng/m³) is lower than that determined by ICP-MS (7.3 ng/m³) (Table 3.1). This finding is explored further in Section 3.3.7.1.

Table 3.3: Summary of elemental concentrations (ng/m³) determined by XRF for variables used in PCA

	Na	Mg	S	Cl	K	Ca	Ti	Cr	Fe	Cu	Zn	As	Pb
Mean	348.2	29.1	70.2	652.2	51.9	44.3	1.8	0.7	18.6	1.5	6.3	4.8	18.6
Median	327.9	23.9	57.1	507.3	44.9	37.6	0.8	0.5	6.7	1.0	3.5	0.8	6.7
Max	1045.5	136.3	314.0	3046.9	281.6	324.4	24.4	3.0	152.5	40.4	705	59.8	123.3
SD	198.3	26.2	53.0	544.1	32.3	35.8	2.9	0.7	23.6	3.3	7.7	9.1	26.0
% not detected	6.9	15.1	6.5	0	1.8	6.8	44.4	30.8	2.9	25.1	1.1	42.2	15.4

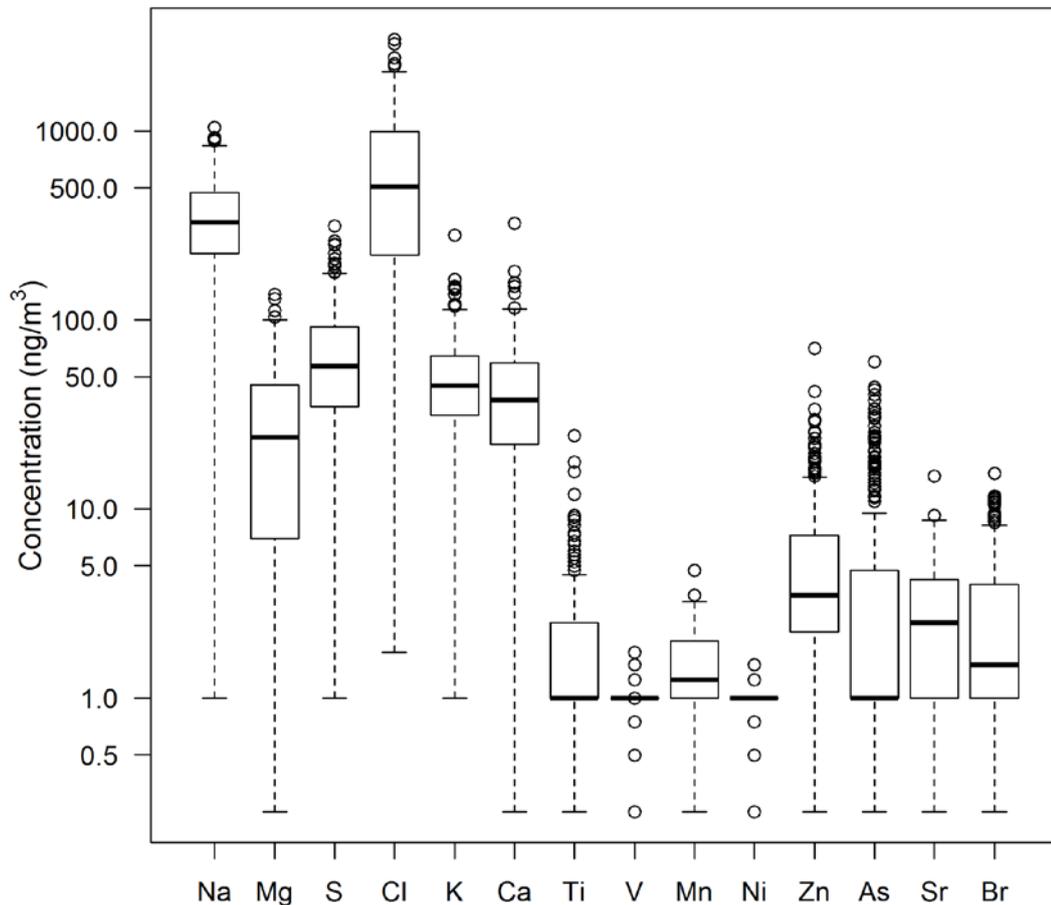


Figure 3.20: Boxplot of elemental concentrations (ng/m³) determined by XRF for variables used in PCA

3.3.7.1 Comparison between arsenic concentrations determined by XRF and ICP-MS

The measurements of arsenic determined by ICP-MS were strongly correlated with the measurements of arsenic determined by XRF from the same filter media ($r=0.98$, $n=279$). Although the relationship was linear, XRF underestimated arsenic concentrations relative to those determined by ICP-MS, as shown in Figure 3.21. A similar magnitude of underestimation for XRF relative to ICP-MS was also found in the Whangarei PM₁₀ receptor modelling study where the inter-methods correlation was $r=0.74$. Arsenic determined by XRF was, on average, 64% of the arsenic value obtained by ICP-MS (for glass fibre filters) (Davy & Ancelet, 2014). The reasons for the differences in concentrations measured by the two analytical methods are not known.

Despite the difference in arsenic concentrations found XRF and ICP-MS, the existence of a strong linear correlation between the two sets of arsenic data obtained by the different analytical methods indicates that most of the variability in arsenic concentration determined by ICP-MS was appropriately reflected in the XRF results. Therefore, the XRF results (including those derived for arsenic) were suitable for use in PCA, because this technique is based on correlations between elements in multivariate space rather than their absolute values.

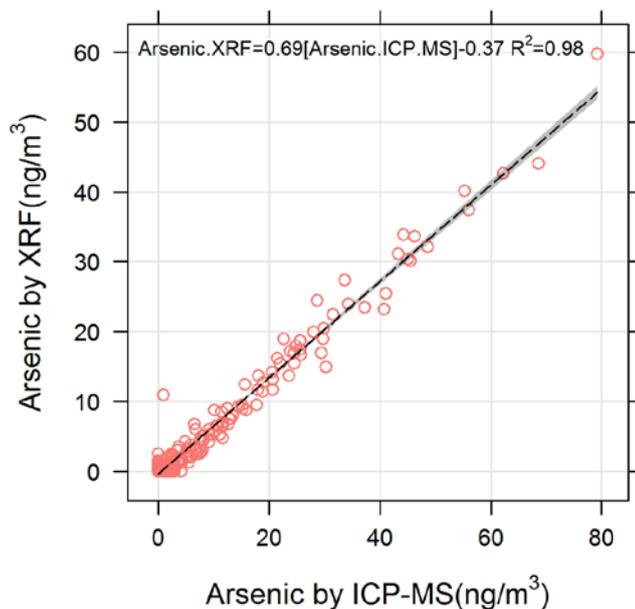


Figure 3.21: Scatterplot with linear fit for arsenic determined by XRF vs arsenic determined by ICP-MS (all data) with shaded area denoting 95% confidence interval

3.3.7.2 Relationships between trace elements based on correlations

A correlation matrix used to visualise the relationships between all pairs of the subset of XRF-determined elemental concentrations in PM₁₀ is presented in Figure 3.22. The correlation matrix employs hierarchical clustering to group variables that are most similar to each other as a way of identifying elements that may have a common source origin. Figure 3.22 shows two main clusters of strongly correlated elements (dark orange to red ellipses). The chlorine, sodium, magnesium and calcium cluster includes elements previously found to be associated with the “marine aerosol” receptor-modelled source and the clustering of black carbon, arsenic, zinc and

potassium is consistent with the “biomass burning” receptor modelled source (Davy et al., 2012).

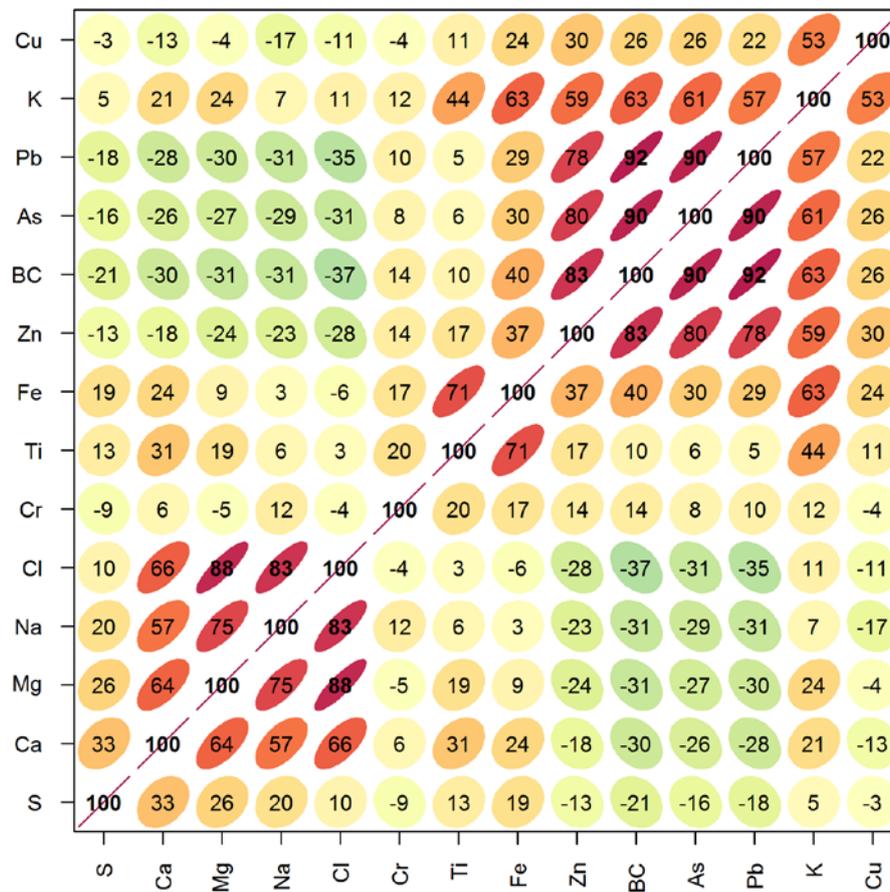


Figure 3.22: Correlation matrix showing relationships between XRF-determined elemental concentrations in PM₁₀ (all data)

Of note is the presence of lead in the XRF “biomass burning” cluster. Lead was not previously included in the receptor-modelled source “biomass burning” for Wainuiomata, or any of the other receptor modelling using particulate matter collected in other residential home heating areas of New Zealand where elemental concentrations were determined by Ion Beam Analysis. The reason for the absence of lead in receptor modelling is due to its high LOD of approximately 80 ng/m³ when determined by IBA (P. Davy, personal communication, February 23, 2015).

In the XRF data set, lead is most strongly correlated with black carbon ($r=0.92$), arsenic ($r=0.90$), followed by zinc ($r=0.78$). The concentrations of these three

elements were distinctly seasonal (Figure 3.23). Figure 3.24 shows the linear fit between lead and arsenic and lead and black carbon.

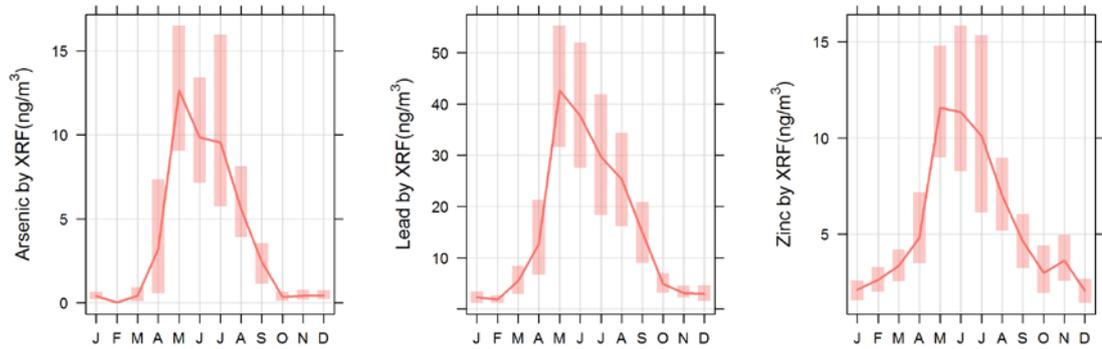


Figure 3.23: Seasonality in arsenic, lead and zinc average monthly concentrations determined by XRF with shaded areas showing 95% confidence intervals in the mean calculated through bootstrap simulations

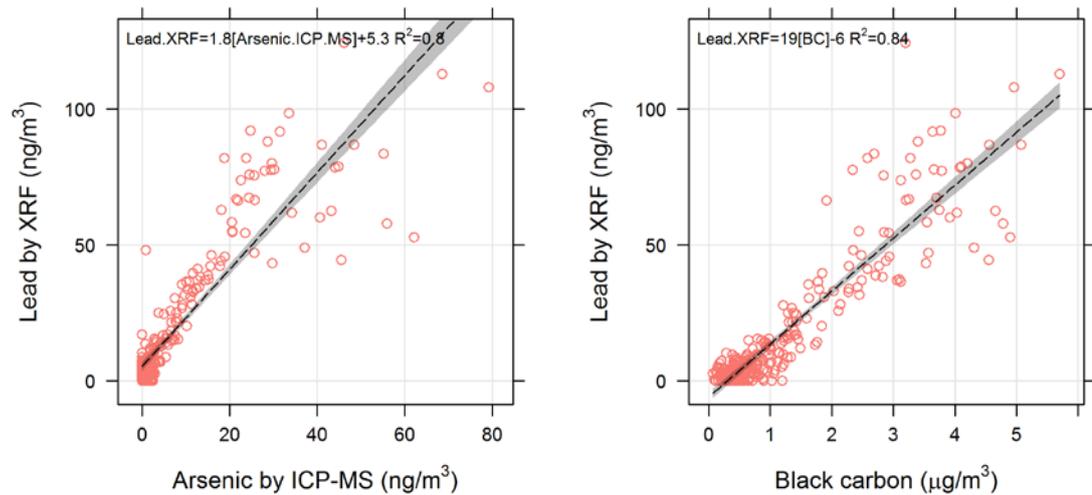


Figure 3.24: Scatterplots with linear fit for lead vs arsenic (left) and lead vs black carbon (right) for all data

The correlation of lead and arsenic concentrations in PM₁₀ measured by XRF was also found in Whangarei ($r=0.51$, $n=449$)(Davy & Ancelet, 2014) in urban Hamilton ($r=0.71$, $n=96$), from a regional council monitoring campaign¹².

Levels of lead in air have declined dramatically since the introduction of un-leaded petrol in 1996. Monitoring of lead in in urban Christchurch and Auckland during winter 2010 found levels of lead determined by ICP-MS in Total Suspended Particulate (TSP) of 21.0 and 7.0 ng/m³ (3-month winter average) (MfE, 2011c). The winter average concentration of lead in PM₁₀ determined by ICP-MS in Hamilton in 2011 was 4.4 ng/m³. The three-month winter average concentration of lead in air in Wainuiomata determined by XRF was 18.6 ng/m³. It is not clear how comparable these results are given the different analytical methods and particulate matter metric. However, these results are well within the national ambient air quality guideline of 200 ng/m³ (MfE, 2002).

In the Whangarei source apportionment study a possible explanation for lead co-varying with arsenic was the domestic burning of timber painted with lead-based paint (Davy & Ancelet, 2014). This is plausible as lead pigments were used commonly as house paints in New Zealand until the mid-1960s¹³, lead-based paint is almost certain to be present on pre-1945 paintwork, and it is likely to be present on pre-1980 paint work (MoH, 2012). Lead was not detected when particulate samples from a NES-compliant wood burner were analysed using IBA (Ancelet et al., 2011b). Failure to detect lead in particulate from wood burning may be due to the relatively high LOD for lead when determined by IBA. Lead was present in one sample of water-soluble extract of particulate matter obtained from real-life emissions testing of a pre-1984 wood burner that also contained high levels of arsenic attributed to the burning of 'old decking' that was presumably treated with CCA (Cavanagh, et al., unpublished manuscript). It is also possible that lead is emitted from burning of untreated natural wood as lead was detected in fine particulate emissions from domestic wood burners using softwoods (fir) in Oregon, USA, in samples collected on Teflon filters and analysed by XRF (Rau, 1989).

¹² Elemental concentrations for lead and arsenic in PM₁₀ determined by ICP-MS provided courtesy of Environment Waikato for monitoring undertaken from 5/11/2010 to 25/11/2011.

¹³ <http://www.resene.co.nz/comm/safety/lead.htm>

Zinc contributed to the receptor-modelled “biomass burning” source previously identified in Wainuiomata (Davy et al., 2012). Particulate matter emissions from a NES-compliant wood burner, burning untreated wood and analysed by IBA, determined an average emission factor of 0.3 mg/kg wood burnt for zinc (Ancelet et al., 2011b). Zinc occurs naturally in the cytoplasm of plants as it is an essential micronutrient for plant growth (Broadley, White, Hammond, Zelko, & Lux, 2007) and therefore zinc is evolved during combustion of wood.

3.3.7.3 Relationship between trace elements based on principal components analysis

Principal components analysis (PCA) using the XRF multi-elemental dataset, together with black carbon measurements was used as an alternative to the correlation matrix to visualise the structure of this multivariate data set. PCA was used to reduce the dimensionality to two components that collectively express much of the inherent variability in the multi-elemental data set. The raw data were scaled (i.e., standardised) as the concentration of some elements differed by an order of magnitude.

PCA used a subset of elements in the XRF data set plus black carbon. The elements chosen were sodium (Na), magnesium (Mg), sulphur (S), chlorine (Cl), potassium (K), calcium (Ca), titanium (Ti), chromium (Cr), iron (Fe), copper (Cu), zinc (Zn), arsenic (As) and lead (Pb). All of these elements, apart from lead, were identified by previous receptor modelling for Wainuiomata as contributing to particulate matter source factors, i.e., “biomass burning”, “marine aerosol”, “vehicle emissions”, “road dust”, “crustal matter” and “secondary sulphate” (Davy et al., 2012). Aluminium (Al) and silicon (Si) (also previously identified as contributing to source factors in Wainuiomata) were not present in the XRF data set as they were not able to be satisfactorily measured by XRF due to interference with the quartz filter media.

There were a sufficient number of samples (filters) for the number of variables used in the PCA to adequately characterise the dimensionality in these data: using 14 variables requires a minimum of 214 samples according to $n > p^2 + 3p + 1$ (where n is the

number of samples and p the number of variables) (Reimann, Filzmoser, Garrett, & Dutter, 2008).

The results of the PCA are presented as a bi-plot (Figure 3.25) that shows the relationships between the elements (via the loadings) and the observations (via the scores). In Figure 3.25 red arrows show the relationship of variables to the principal component (PC) axes, and the grey numbers show the projections of individual measurements onto the axes. PC1 was characterised by close association of black carbon, arsenic, lead and zinc and was assumed to represent the “biomass burning” source, which explained 35.6 per cent of the variability in the 13 elements plus black carbon. PC2 was characterised by the presence of chlorine, sodium, magnesium and calcium, and was taken to represent the “marine aerosol” source, explaining a further 23.6 percent of the variability in the data set. Together these two principal components explain 59% of the total variation in PCA data set. These results are consistent with the multi-element correlation plot (Figure 3.22).

The loose cluster of titanium, iron and potassium is not easily interpretable and may arise from a mix of sources such as road dust and crustal matter. Potassium is usually a marker of wood combustion and may also be associated with marine aerosol.

This method of examining relationships between elements for characterising possible sources is exploratory and is limited by the lack of certain important elements such as aluminium and silicon that are associated with crustal matter. Having size-resolved samples for PM_{25} and PM_{25-10} may also improve the ability to discriminate sources associated primarily with the fine fraction (e.g., combustion) or coarse fraction (e.g., crustal matter). The disproportionate number of samples for the winter months may also be influencing the results.

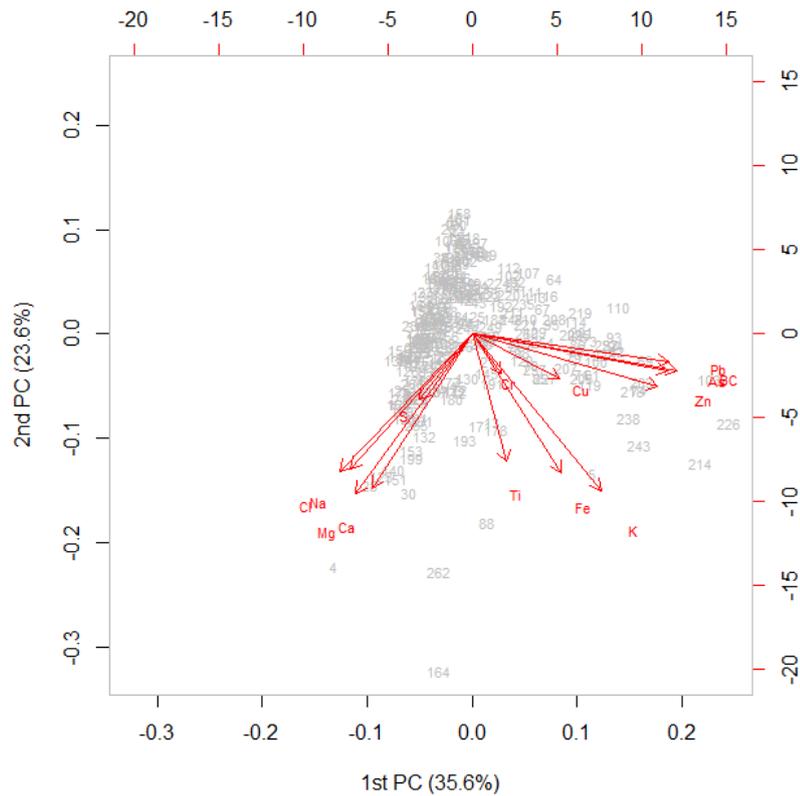


Figure 3.25: Bi-plot of PC1 and PC2 for 14 variables (n=279)

3.3.8 Dependence of arsenic concentration on $PM_{2.5}$ and meteorology

Ambient arsenic and $PM_{2.5}$ concentrations are a function of spatial and temporal variation in emissions sources and strength and the effectiveness of meteorological conditions for emission dispersal. Simple multiple linear regression was used to explore the relationship between arsenic concentrations, $PM_{2.5}$ and meteorology and to assess the amount of variation in arsenic concentrations that can be explained, jointly and individually, by $PM_{2.5}$ and meteorology.

During the winter months in Wainuiomata it is assumed that the majority of the measured $PM_{2.5}$ arises from domestic fires, with only a minor component arising from motor vehicles, secondary sulphate, marine aerosol and soils (Davy et al., 2012). There were no suitable estimations of $PM_{2.5}$ emissions from domestic fires (e.g., g/kg wood used or g per night per house) at fine enough temporal resolution available for modelling. A proxy for emissions based on the Wainuiomata home heating survey (Wilton, 2006) (average number of days per week using a fire by month of the year)

scaled to the proportion of households who reported using wood for heating in the Statistics New Zealand 2013 Census, did not provide sufficient resolution to account for day-to-day variability in number of fires being operated. Therefore PM_{2.5} concentrations were used in the modelling as a predictor variable to represent emissions from domestic fires. The modelling period was restricted to the months of April to September (inclusive) to reflect the winter domestic home heating season including the shoulder months, when fewer people were burning wood but arsenic was still detected.

Classification and Regression Tree (CART) analysis, a graphical based non-parametric approach, was used to evaluate suitable predictor variables. Candidate meteorological predictor variables identified as contributing significantly to variability in both arsenic and PM_{2.5} were the 0.1 quantile of 1-hour average ambient temperatures and the median 1-hour average wind speeds. These two variables are strongly linearly correlated with both arsenic and PM_{2.5} (Figure 3.17).

After testing of nested models with ANOVA, the model specified in Equation 3.3 ($R^2=0.90$, $n=141$) was selected as the most parsimonious for explaining the variation in mean arsenic concentration.

Equation 3.3:

$$E[As] = 1.90 + 1.95(PM_{2.5}) - 0.38 (\text{Temp}_{0.1\text{quantile}}) - 1.14(\text{WS}_{\text{median}})$$

Where:

$E[As]$ = Expected mean value for arsenic (ng/m^3)

$PM_{2.5}$ = Average $PM_{2.5}$ ($\mu\text{g}/\text{m}^3$)

$\text{Temp}_{0.1\text{quantile}}$ = 0.1 quantile of 1-hour mean temperature ($^{\circ}\text{C}$)

$\text{WS}_{\text{median}}$ = Median 1-hour wind speed (m/s)

This model reflects the positive correlation of arsenic with $PM_{2.5}$ and the negative correlation with temperature and wind speed, i.e., arsenic shows a similar pattern to $PM_{2.5}$ and the same tendency for higher concentrations at low temperatures and wind speeds. Model diagnostics showed no strong indications of failure in the model

residuals for the assumptions of normality or linearity in the response, and the three potential outliers were within Cook's distance and so were not exerting undue influence on the model fit. There was an indication of an upward trend in the standardised residuals with fitted values, although this may be due to the low number of relatively high arsenic concentrations being modelled and the influence of three model outliers.

As the predictor variables were correlated with each other, the individual contributions of PM_{2.5} and of meteorological factors to explaining variability in the mean arsenic levels according to the fitted model were explored. A linear regression of temperature and wind speed on PM_{2.5} revealed that 62% of mean PM_{2.5} (during the winter months when home heating sources are present) could be explained by the fitted model. Similarly, wind speed and temperature were found to explain 64% of the variation in mean arsenic levels.

An estimate of how much of the variation in arsenic is explained by PM_{2.5} independent of the impact of the meteorological variables was determined from the partial correlation coefficient of the residuals of these two models. A correlation coefficient of $r=0.88$ was obtained, equivalent to $R^2=0.77$, meaning 77% of the variation in mean arsenic (according to the fitted model) was explained by PM_{2.5} when the effect of meteorology on PM_{2.5} was removed.

The regression model does not have utility for predicting 24-hour arsenic concentrations from PM_{2.5} and meteorology due to wide spread in the magnitude and direction of the residuals. However, the purpose of linear models is to predict the mean response in arsenic for fixed values of both PM_{2.5} and of the meteorological variables. Therefore, based on the modelled relationship between arsenic and PM_{2.5} and meteorology in winter 2012 and 2013 (Equation 3.3), it was possible to predict the winter arsenic mean in 2014 from PM_{2.5} and meteorology observations in that year assuming that patterns in burning of treated timber remained unchanged.

Figure 3.26 shows the predicted winter mean for arsenic (May to September inclusive) using all the observations of PM_{2.5}, temperature and winds speed (i.e., not just days on which arsenic was measured) and the observed winter arsenic mean for the days that

arsenic was measured. The agreement between modelled arsenic and observed arsenic is reasonably close, providing confidence in the suitability of the linear model for predicting future arsenic concentrations. The predicted 2014 winter arsenic mean was similar to that observed/modelled in 2013 although lower than in 2012. This result is consistent with differences in the observed predictor variables between years. Observed wind speeds and temperatures were both lower in 2012 compared to 2013 and 2014. This could explain in part the higher concentrations of both arsenic and PM_{2.5} in the first year of monitoring, bearing in mind that a statistically significant difference in mean annual or winter arsenic was not found.

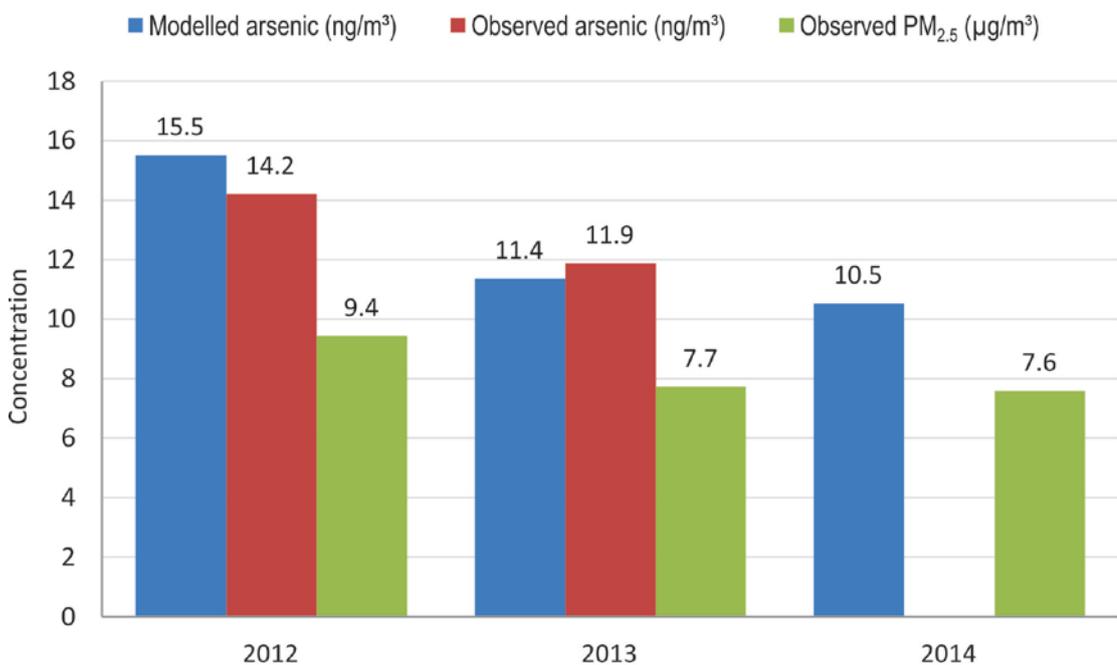


Figure 3.26: Predicted average winter arsenic using all observed PM_{2.5} and meteorological values

3.4 Synthesis of ambient air monitoring

Arsenic concentration (24-hour averages) in Wainuiomata air were strongly seasonal, with elevated levels observed during the main home heating period from May to August in 2012 and in 2013. Arsenic was also present, although at much lower concentrations, during the shoulder months of April and September when home fires are used by a smaller proportion of households. These patterns broadly match appliance use reported by Wainuiomata residents in a home heating survey in 2006 (Wilton, 2006). Arsenic was at very low levels or non-detectable for the rest of the year

when home heating is not being used and is most likely from a natural component of soils and crustal matter.

The annual average for arsenic was 6.8 ng/m^3 in the first year of monitoring and 5.3 ng/m^3 in the second. Therefore, based on the absolute value of the mean (an approach often used for regulatory purposes) arsenic failed to meet the national ambient air quality guideline of 5.5 ng/m^3 in the first year but was attained in the second year. However, as the 95 percent confidence intervals for each year included the guideline value of 5.5 ng/m^3 for arsenic in ambient air, results of this work can only be used to state that arsenic in air was at a concentration commensurate with and indistinguishable from the guideline value. The results cannot be used to indicate either compliance with, or exceedance of, the guideline value. This outcome of approximate statistical equivalence with the guideline indicates the presence of a potential issue and suggests the need to carry out a more detailed assessment of likely exposure to arsenic. Exposure and risk estimates are presented in Chapter 4.

Arsenic has been found in high levels in fine particulate matter evolved when CCA-wood is burnt in domestic wood burners (Abbott & Rogers, 1989; Cavanagh et al., unpublished manuscript) and therefore was originally suspected to be the principal source of arsenic due to temporal coincidence with home heating emissions throughout New Zealand (Cavanagh et al., 2012; Davy et al., 2011). Strong correlations of arsenic with other products of combustion (e.g., black carbon) and trace elements typically emitted from burning of untreated wood adds to the weight of evidence linking domestic fires as the source of winter arsenic.

The highest levels of PM_{10} , $\text{PM}_{2.5}$, black carbon and arsenic were measured on cold calm days. These weather conditions restrict the dispersal of domestic heating emissions and are therefore an important factor leading to PM_{10} exceedances throughout New Zealand (MfE, 2014). It is also possible that on cooler days there is an increased level of wood burning due to the inverse relationship between energy use and external temperature during the home heating season (Isaacs et al., 2010). Other factors, such as lifestyle and home occupancy patterns, play a role in timing of fire lighting, duration of burn and quantity of fuel used, independent of temperature

(Wilton & Bluett, 2012b). Therefore it can be difficult to disentangle the joint effects of human behaviour, emissions and meteorology on ambient air contaminant concentrations.

Day-to-day variability in arsenic was greater than that observed for PM_{2.5} and black carbon. This indicates that the general level of “biomass burning” and short term meteorological conditions did not fully explain the very high arsenic concentrations found on one day and not on another despite similar environmental conditions. A plausible explanation is the temporal and spatial variation in households burning CCA-wood on any given night. It appears likely that the number of residences burning treated timber is approximately proportional to the overall number of households using wood for heating. However, on any given night the number of residents using treated timber (and the proximity of their house to the monitoring site) randomly fluctuates within the airshed. Therefore, high arsenic concentrations may occur during the peak burning months of May, June and July (when appliance use is greatest), and when conditions are unfavourable for dispersion, and if one, two or more houses are burning CCA-wood instead of untreated wood, close to the monitoring site.

Linear modelling of arsenic using PM_{2.5}, wind speed and temperature as predictor variables showed utility for predicting the winter average for arsenic. The 1:2 day sampling frequency captured enough of the variability in arsenic to build a model that is useful for calculating a winter arsenic average for Wainuiomata assuming that burning behaviour relating to use of CCA-wood is relatively static. However, arsenic concentrations for a particular day, based on ambient PM_{2.5} and meteorology could not be accurately estimated by linear modelling. This may be better approached using a non-linear modelling technique or a model based on an underlying ‘extreme value’ probability distribution for arsenic.

Winter and annual averages across the airshed are likely to be similar. However, the day-to-day variability means that short term exposures (i.e., 24-hour duration) may differ markedly across the airshed depending on proximity to a household burning CCA-wood.

4 Health risk assessment

4.1 Introduction

This chapter includes all aspects of the research that relate to a health risk assessment designed to meet Aim 3 of this Thesis. Specifically this involves developing a site-specific exposure model to address potential risk of adverse health outcomes for residents of Wainuiomata from exposure to inorganic arsenic arising from the burning of CCA-wood within the airshed. It also includes an assessment of health risk due to exposure to arsenic from other environmental media, such as soils, and from background sources such as diet and smoking. Details of the methodology are provided in Section 4.2, results and their interpretation in Section 4.3, uncertainty in Section 4.4 and a synthesis in Section 4.5.

4.2 Methodology

4.2.1 Approach

The human health risk assessment methodology used here is based on the principle that the likelihood of adverse health effects from an environmental toxicant can be described as a function of an estimate of exposure to a toxicant and an estimate of its toxic properties, i.e., Potential health risk = $f(\text{metric of exposure, metric of toxicity})$.

The specific methodological approach adopted for the health risk assessment followed the US EPA risk assessment framework for “air toxics” described as any chemical substance in the air that could pose a risk to human and or environmental health. Whilst focused primarily on inhalation health risk, the US EPA methodology also considers risks posed by other exposure routes, where the air contaminant can accumulate in other environmental media such as soils, water, or plants.

The general framework consists of four inter-related steps, described below:

- i) *Problem formulation* – articulates the assessment question and develops a site-specific conceptual model to define the scope and nature of the health risk assessment.

- ii) *Exposure assessment* – quantifies the magnitude, frequency and duration of human exposure to an air pollutant for the exposure pathway(s) and route(s) under study.
- iii) *Toxicity assessment* – identifies the known adverse health effects associated with exposure to the air pollutant (hazard assessment) and the relationship between the amount of exposure and resulting response (dose-response assessment).
- iv) *Risk characterisation* – summarises and combines the findings of the exposure and toxicity assessments to characterise human health risk, both quantitatively and qualitatively.

Like various other US EPA guidance documents a three-tiered approach to risk assessment is recommended in which assessments may span a range of complexity levels depending on the degree of generic versus site-specific inputs and characterisation of uncertainty. The simplest Tier 1 “screening level” approach, using readily available generic inputs and conservative assumptions, has been adopted for this thesis. If the outcome of this relatively simple approach demonstrates insignificant risk then there is little justification for further investigation or management options. If on the other hand the risk appears to be relatively significant, then a higher tier of assessment is warranted to determine whether the risk is realistic or an artefact of the conservative assumptions in the Tier 1 assessment (US EPA, 2004).

Accordingly, a Tier 1 risk assessment process was undertaken based on the four step US EPA methodology for assessing human health risk due to toxic air contaminants emitted to air from stationary sources.

4.2.2 Assessment scope and conceptual model

The risk assessment scope was defined to meet research objective iv) Section 1.2.2 subject to the availability of data, methods and resources to support characterisation of potential exposure pathways. The assessment scope was restricted to the geographic boundary of the Wainuiomata airshed and to exposure of adult and child residents to arsenic from CCA-wood burning and from background environmental

sources. Occupational exposure or exposure to arsenic that might occur when people were outside the airshed was specifically excluded.

A site-specific conceptual model illustrating the exposure pathways and health effects to be evaluated was developed for the Wainuiomata airshed Figure 4.1. The primary source of emissions of inorganic arsenic to outdoor air was deemed to be the burning of CCA-treated timber by some residents as part of their domestic heating practice during the winter months. The potential exposure to arsenic in the indoor environment arising from infiltration of outdoor air and the contribution from indoor sources was included in the model for assessment. Potential indoor sources of arsenic are fugitive emissions from the wood burner during fire lighting and refuelling periods when CCA-wood is being used and environmental tobacco smoke (ETS) which has been shown to contain trace levels of arsenic.

As part of the literature review (Chapter 2), examples were found of non-inhalation pathways for airborne arsenic from industrial sources including incidental ingestion of contaminated indoor dust and consumption of home grown vegetables grown in soils containing deposited arsenic or arsenic deposited directly on leaf surfaces. The lack of study-specific measurements or models that could be adequately parameterised precluded inclusion of these pathways in the conceptual model.

Following combustion of CCA-wood, arsenic is preferentially retained in the ash (Abbott & Rogers, 1989). Therefore CCA-wood ash residue is considered a potential exposure media, depending on disposal practice, for accidental ingestion by children. Vegetables grown in soil that contains CCA-wood ash are known to uptake arsenic (Abbott & Rogers, 1989) but this exposure pathway was excluded from the conceptual model due to large uncertainties in assessing exposure and uptake from this pathway.

For comparative purposes the contribution of arsenic from other background sources, such as diet and active smoking was included in the conceptual model. Children's exposure to arsenic in dislodgeable arsenic residues from CCA-wood in playground structures was also included in the conceptual model as this exposure pathway has recently been examined in New Zealand (Read, 2003).

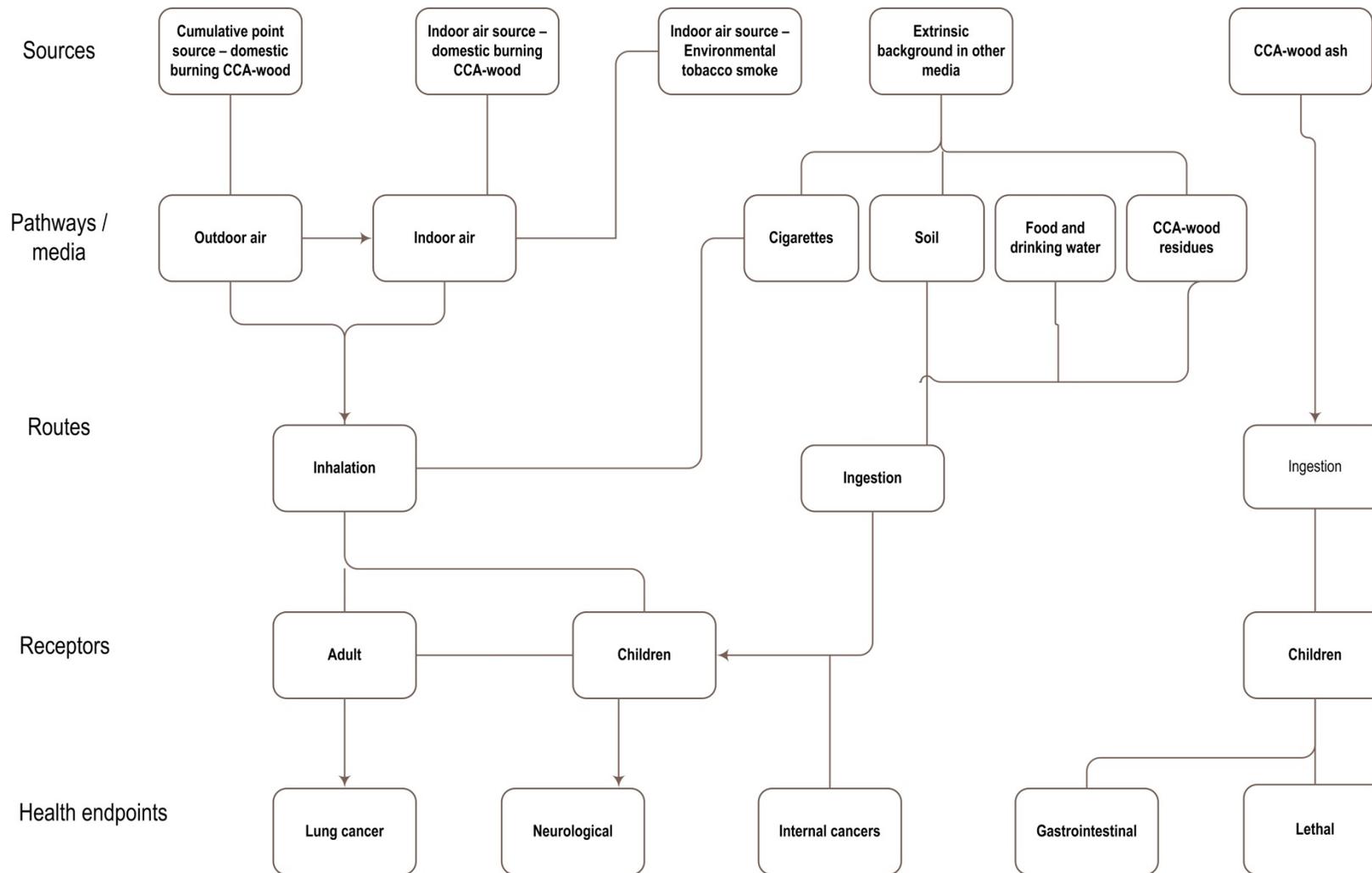


Figure 4.1: Site specific conceptual model for health risk assessment

4.2.3 Exposure assessment method

This section describes the methods used to estimate the magnitude, frequency and duration of an individual's and the population's intake of arsenic via the pathways and routes identified in the conceptual model. The outcome of the exposure assessment is a prediction of the likely dose received over a short time frame (acute exposure) and a long term or lifetime (chronic exposure). It is noted that for exposure to air toxics reference doses are specified as concentrations in air, and implicitly incorporate information about how much of an inhaled dose is absorbed. For this reason the exposure dose is estimated as the dose inhaled, rather than the portion of the dose that is absorbed. All exposure assessments were carried out by developing a Microsoft Excel spreadsheet that based on Equations 4.1 to 4.7 described in this section.

4.2.3.1 Characterisation of exposure setting

Wainuiomata has a usually resident population of 16,284 comprised of 5976 households of which 36% use wood to heat their main living area and 2% report using coal (Statistics New Zealand 2013 Census). This equates to 2151 dwellings using wood for home heating if the households who did not respond to this census question are assumed not do differ from those who did with respect to their home heating methods. The prevalence of households using CCA-wood on their home fires was estimated based on the results of the 2013 Masterton home heating survey (Sridhar & Wickham, 2013) that found that 16% of households that used wood burners, always, often or sometimes used decking or fencepost offcuts on their fires. This estimation is consistent with the 2012 Auckland home heating survey (Stones-Havas, 2014) which found 17% of wood burning households had used decking or fencepost offcuts on their fires. Therefore it was estimated that 344 households (i.e., 16% of 2151 dwellings) used CCA-wood at some time during the winter months in Wainuiomata.

4.2.3.2 Inhalation pathway assessment

The inhalation pathway exposure assessment was based on estimating arsenic intake for an individual (child and adult) for the following hypothetical household exposure scenarios:

- a) a non-smoking household
- b) where a least one member of the family smokes
- c) where CCA-wood is intermittently burnt inside the home
- d) where a least one member of the family smokes **and** where CCA-wood is intermittently burnt inside the home

Following the US EPA methodology, inhalation **exposure concentrations** (EC) were defined as the concentration of arsenic air in the breathing zone and no adjustment for other exposure parameters (e.g., body weight and inhalation rate) is required. Exposure modelling is recommended to account for time spent in different **microenvironments** (e.g., indoor, outdoor or in-vehicle) and where exposure concentrations vary.

For inhalation exposure assessment, acute exposure concentrations, defined as 24-hour or shorter exposure period, were derived from the maximum 24-hour concentration recorded during the outdoor monitoring campaign. Chronic exposure was defined as being continuously resident in Wainuiomata for six years (as a child), 20 years (as child and adult) and 75 years (as child and adult). Chronic exposure concentration was based on the 24-month average concentration from the single fixed outdoor air monitoring site. An exposure assessment was also made for a pseudo sub-chronic exposure based on being present in Wainuiomata for one winter period only. Concentrations of arsenic in indoor air were estimated using a mass-balance approach that takes into account infiltration of outdoor air and the contribution of indoor sources (see below).

The indoor environment is an important site for inhalation exposure as this is where people spend most of their time (Baker, Keall, & Au, 2007). Indoor arsenic levels were not measured in this study and no observations relevant to this assessment were found in the literature. Therefore a single compartment steady state mass-balance equation adapted from (Özkaynak, 1999; Özkaynak et al., 1997) was used to estimate

the indoor concentration of arsenic relevant to the hypothetical exposure scenarios as per Equation 4.1.

Equation 4.1:

$$C_{in} = C[indoor - ambient] + C[indoor - nonambient]$$

$$C_{in} = \frac{P \times ach}{ach + k} C_{out} + \frac{N_{cig} E_{smk} + T_{firebox} \times E_{firebox}}{(ach + k)Vt}$$

Where:

C_{in}	=	arsenic concentration indoors ($\mu\text{g}/\text{m}^3$)
C_{out}	=	arsenic concentration outdoors ($\mu\text{g}/\text{m}^3$)
P	=	building penetration factor (unitless)
ach	=	air exchange rate (hour^{-1})
k	=	arsenic decay or deposition loss coefficient (hour^{-1})
N_{cig}	=	number of cigarettes smoked per hour
E_{smkAs}	=	estimated arsenic emissions for cigarette smoking ($\mu\text{g}/\text{cig}$)
$E_{fireboxAs}$	=	estimated arsenic emissions CCA-timber burning ($\mu\text{g}/\text{min}$)
$t_{fireboxOpen}$	=	length of time firebox open while CCA-timber burning (min)
V	=	volume of living room (m^3)
t	=	model step time (1-hour)

The first term in Equation 4.1 describes the infiltration of ambient arsenic in $\text{PM}_{2.5}$ indoors. The penetration factor P is the fraction of arsenic particulate that penetrates the building envelope. The deposition rate k includes all processes, except air exchange, that result in a loss of particles (i.e., sedimentation, adsorption, and absorption).

The second term in Equation 4.1 describes the generation of particulate containing arsenic from indoor sources. Indoor emissions from smoking are specified in the equation to represent exposure to arsenic in environmental tobacco smoke (ETS) as opposed to direct inhalation from active smoking. N_{cig} represents the intensity of smoking (i.e., number of cigarettes per hour) and S_{smk} represents the quantity of arsenic emitted in sidestream smoke per cigarette. $S_{fireboxAs}$ represents the emissions from the burning of CCA-wood on a woodburner inside the home and $t_{fireboxOpen}$ represents the length of time that these emissions will be discharged into indoor air due to the firebox being opened.

The model input parameters selected are shown in Table 4.1 and the derivation of these values are described in the sections below.

Table 4.1: Indoor air model input parameters

Input parameter	Winter	Summer	Annual	Units
C_{out}	0.0146	0.001	0.061	$\mu\text{g}/\text{m}^3$
P	1	1	1	unitless
ach	1	0.7	0.85	ac/hr
K	0.12	0.12	0.12	decay/hr
E_{smkAs}	0.15	0.15	0.15	Arsenic $\mu\text{g}/\text{cig}$
N_{cig}	2	2	2	cig/hr
$E_{fireboxAs}$	16	0	16	Arsenic $\mu\text{g}/\text{min}$
$t_{fireboxOpen}$	1.667	0	0.1667	min
V	45	45	45	m^3

i) Air exchange rates and building penetration factor

Outdoor air reaches the indoor environment by three main routes: a) mechanical (e.g., air conditioning), b) natural ventilation (e.g., open windows) and c) infiltration (e.g., leakage through cracks around unsealed windows and doors). The air exchange rate is the sum of these three flow rates into a building divided by the interior volume (Nazaroff, 2004). Air exchange rates are expressed as number of air changes per hour, for example 1 air change per every two hours equals 0.5 hour^{-1} .

In the New Zealand context research on airtightness of homes shows that air exchange rates are correlated with the building age and type of window joinery meaning newer homes are more airtight. All pre-1960 houses with strip flooring and unsealed timber windows are described as “draughty” with a nominal air exchange rate of 0.9 ach (range 0.7 to 1.7) and 1960s to 1980s houses range from 0.3 to 0.8 ach (Amitrano, Kirk, & Page, 2006; Bassett, 2001). A home heating survey of Wainuiomata residents found approximately 41% of respondents lived in houses built between 1965 and 1984 and 43% lived in pre-1964 housing stock (Wilton, 2005).

Based on the above information indoor model input values for residential air change rates are taken to be 0.7 for the winter months and 1 during the non-winter period when windows are expected to be more frequently open for natural ventilation.

ii) Indoor particle deposition loss rate

This parameter describes the rate at which particles are lost from indoor air through deposition onto room surfaces. The rate at which particles are lost or decay varies strongly with particle diameter (Nazaroff, 2004). For the purposes of this study it is assumed that most of the arsenic particulate would be in the sub 2.5 micron range as ambient air concentrations of arsenic are more strongly correlated with $PM_{2.5}$ concentrations than PM_{10} as found in Chapter 3.

Riley, McKone, Lai, and Nazaroff (2002) modelled empirical data to determine indoor deposition loss rates for PM as a function of particle size, ventilation conditions, and indoor surface area to volume ratios in residential and office environments. The results of this study suggest deposition loss rates of $0.11 \text{ (h}^{-1}\text{)}$ for a non-mechanically ventilated home and $0.13 \text{ (h}^{-1}\text{)}$ for a home with high natural ventilation could be appropriate input parameters. The mean of these two rates, $0.12 \text{ (h}^{-1}\text{)}$ was used as input for the mass-balance model in this work.

iii) Cigarette emissions

There is little available information on measured concentrations of arsenic in indoor air arising from environmental tobacco smoke (ETS) (i.e., after dilution with indoor air). One study that measured heavy metals in indoor air in public smoking and non-smoking areas (using low volume personal samplers) found that arsenic was elevated (0.1 to 1.0 ng/m^3 (mean 0.4)) in smoking areas and below the level of detection in non-smoking areas (Landsberger & Wu, 1995). Arsenic levels in sidestream smoke (emitted from the burning cigarette tip) range from 0.072 to 0.126 (water soluble) and from 0.122 to 0.178 (total element) $\mu\text{g/cigarette}$ (Behera, Xian, & Balasubramanian, 2014). Therefore, model input for arsenic per cigarette was taken as $0.15 \mu\text{g}$ - the mid-point between the lowest and highest total arsenic yield from Bereha et al. (2014). It was assumed that 20 cigarettes per day are consumed per person, and this is represented as two cigarettes per hour as a model input assuming only one person in the household smokes at a time. These inputs were used to estimate the additional arsenic levels that were likely to be present in smoking households only.

iv) Fugitive firebox emissions

Observational studies show that particulate and gases are emitted to indoor air from wood burning appliances when they are first lit and when the fire door is opened for addition of wood (Longley & Gadd, 2011; Salthammer, Schripp, Wientzek, & Wensing, 2014). Accordingly, where arsenic is present in particulate matter emitted from wood burning short term elevated arsenic concentrations indoors due to spillage from the wood burner when the fire box door is open could be expected.

The emission rate for arsenic arising from the burning of CCA-wood in a solid fuel appliance was estimated by Equation 4.2.

Equation 4.2:

$$E_{fireboxAs} = \left[\frac{EF_{As} \times FU}{D} \right] \div 60 \text{ seconds}$$

Where:

$E_{fireboxAs}$	=	Arsenic emitted ($\mu\text{g}/\text{minute}$)
EF_{As}	=	Emission Factor for arsenic ($\mu\text{g}/\text{kg}$)
FU	=	Fuel Use (kg/hour)
D	=	Duration of burning (hours/day)

An emission factor for arsenic of 278,000 ($\mu\text{g}/\text{kg}$) was chosen as it represents the mid-point of two experimental studies in which CCA-wood was burnt. Emission factors of 255,000 to 427,000¹⁴ ($\mu\text{g}/\text{kg}$) were derived from a New Zealand study in which CCA-wood was burnt in a domestic wood burner under a range of temperature conditions (Abbott & Rogers). An emission factor of 190,000 to 240,000 ($\mu\text{g}/\text{kg}$) was reported from simulated open burning of fresh and aged CCA-wood in the US (Wasson et al., 2005). A fuel use rate of 1.7 kg per hour was assumed based on previous home heating inventory work (Wilton, 2012). It was further assumed that half of the wood burnt (in the higher arsenic exposure household being modelled) was CCA-treated; providing a fuel use factor of 0.85 kg per hour. The burn duration was assumed to be 12 hours for a typical winter's day based on the Masterton home heating survey (Sridhar & Wickham, 2013).

¹⁴ Calculated by dividing total arsenic (g) in input fuel divided by 5 kg (amount of wood burnt) multiplied by percentage of arsenic found in flue gas.

Based on the above assumptions using Equation 4.2:, the arsenic emission rate was calculated to be 328 µg/minute. For the purposes of modelling indoor air concentration using Equation 4.1 it was assumed that only five per cent of arsenic emitted from the burning of CCA-wood in the firebox will be discharged into the room, i.e., the fraction that does not exit via the flue to outdoor air. Therefore the arsenic emission rate used for indoor modelling was 16 µg/minute, and this was taken to apply during the periods when the firebox door was opened in this case estimated to be 10 seconds 10 times per day (or 1.667 minutes in total).

4.2.3.3 Inhalation exposure characterisation

This section outlines the inhalation exposure modelling approach that used the monitored ambient air concentrations (2012 to 2013) and the modelled indoor air concentrations together with simple estimates of how people partition their time between indoor and outdoor locations (activity patterns) to derive time-weighted estimates of exposure concentration for the hypothetical scenarios outlined in Section 4.2.3.2.

Average inhalation exposure concentration for each scenario as a 1 year exposure and a 75 year or lifetime exposure was calculated using Equation 4.3.

Equation 4.3:

$$EC_A = \frac{1}{T} \left(\sum_j C_j \times t_j \right)$$

Where:

- EC_A = the adjusted average inhalation exposure concentration (µg/m³)
- T = total averaging time ($T = \sum t_j$; years)
- C_j = the average concentration for microenvironment j (µg/m³)
- t_j = time spent in the microenvironment j (years)

Estimates for New Zealand adults of time spent in the major generic microenvironments based on the 1998/1999 New Zealand Time Use Survey (NZTUS) in conjunction with international estimates of the split between indoor and outdoor time (Baker et al., 2007) are provided in Table 4.2. Estimates of children's time activity

patterns were taken from a 2010/2011 Canadian Human Activity Pattern Survey 2 (Matz et al., 2014). A cross check of the Canadian survey findings for daily time spent “In vehicle” were only slightly higher than the results of the New Zealand household travel report¹⁵ which found that 0 to 4 years were on average in transport for 40 minutes per day (4.66 hours per week) and 5 to 14 years were on average in transport for 38 minutes per day (4.37 hours per week) compared to the Canadian’s 1 to 4 years (46 minutes per day) and 5 to 11 years (49 minutes per day). Other patterns for the entire New Zealand and Canadian populations are broadly similar where these can be compared for the same categories. Therefore the Canadian children’s data was deemed suitable for use in the exposure assessment.

Table 4.2: Estimated proportions of time spent in main ‘microenvironments’

Microenvironment	NZ adults	Canada	Canada 1-4 years	Canada 5-11 years
Home – indoors	69.7%	69.9%	74.0%	71.3%
Home – outdoors	2.4%			
Transport – indoors	5.1%	5.3%	3.2%	3.4%
Transport – outdoors	0.1%			
Other places – indoors	19.0%	19.0%	15.3%	17.8%
Other places - outdoors	3.6%			
Outdoors		5.8%	7.6%	7.5%

4.2.3.4 Background exposure via inhalation

The US EPA risk assessment methodology defines background concentrations as the levels of contaminants that would be present in the absence of the source being evaluated, in this case arsenic arising from burning of CCA-wood. Background sources can be natural or anthropogenic in origin. The purpose of quantifying risks from background sources is to place the risk from a particular source in perspective by comparing the risk attributable to background against the risk attributable to the source being evaluated.

In this assessment as there are no known background sources of airborne arsenic (that is, the natural concentration of arsenic in air is negligible compared with the anthropogenic sources under consideration), the contribution of active smoking to arsenic inhalation exposure will be considered as a ‘pseudo-background’ source for

¹⁵ <http://www.transport.govt.nz/research/travelsurvey/reportsandfactsheets/>

evaluation and comparison with the risk attributable to exposure to arsenic from CCA-wood burning.

Cigarette smoke is a potential exposure route for arsenic due to trace levels of inorganic arsenic which may be present in the tobacco filler. Levels of arsenic in tobacco have reduced dramatically since the use of arsenate based pesticides has declined (Fishbein, 1976) but arsenic is still detectable in many brands of cigarettes (Behera et al., 2014; Caruso, O'Connor, Stephens, Cummings, & Fong, 2014; Marano et al., 2012). The arsenic level of 0.7 µg/cigarette used by Fowles and Bates (2000) to establish a cancer risk index for carcinogens in cigarettes to inform Ministry of Health harm reduction policy was considered to be unrealistically high. The arsenic level of 0.7 µg/cig was selected as midpoint of the range 0 to 1.4 µg/cig (Smith, Livingston, & Doolittle, 1997). However, a closer examination Smith shows that only 4 (non-filter) out of 38 brands produced mainstream smoke (the smoke inhaled during puffing) containing arsenic levels above 0.05 µg/cig. A search for more recent figures for arsenic yields in mainstream smoke found 0.0048 to 0.0184 µg/cig (Marano et al., 2012) and 0.036 to 0.104 µg/cigarette (Behera et al., 2014).

For the purposes of estimating lifetime cancer risk from exposure to arsenic in mainstream smoke in a way that can be compared to cancer risk attributable to ambient concentrations the yield of 0.05 µg/cig seems reasonable as this is the midpoint between the lowest and highest level reported in the (Marano et al., 2012) and (Behera et al., 2014) studies. Accordingly, inhalation exposure and risk for active smoking were calculated using Equation 4.4.

Equation 4.4:

$$ILCR = ADC_{Life} \times IUR$$

Where:

$ILCR$ = incremental lifetime cancer risk

ADC_{Life} = $\frac{ADC \times \text{number of years smoking}}{\text{Average life time}}$ (mg/m³)

ADC = $\frac{\text{Average daily concentration}}{\frac{\text{Arsenic yield per cigarette (mg)} \times \text{number of cigarettes/day}}{\text{volume of air inhaled perday (m}^3\text{)}}}$

IUR = unit risk factor (mg/m³)⁻¹

The exposure factors used in Equation 4.4 were 20 m³ volume of air breathed per day, 20 cigarettes per day consumed each containing 0.05 µg of arsenic by an adult daily for 35 years for an average lifetime of 75 years.

4.2.4 Ingestion pathway assessment method

This section describes the methods used to estimate the potential uptake of arsenic arising from incidental ingestion of CCA-wood ash by a child resident.

4.2.4.1 Media concentration

As found by the literature review (Chapter 2), when CCA-wood is burnt almost all the original amount of chromium and copper are retained in the ash while some of the arsenic is released into the gas phase. A range of arsenic concentrations found in residual ash are shown in Table 4.3.

Table 4.3: Reported concentration of arsenic (mg/kg) in CCA-wood ash

Arsenic concentration (mg/kg)	Description	Source
620	Ash from an indoor fire place where a mixture of CCA-treated pine and untreated wood had been burnt mixed with soil (US). Analysis method not stated.	McClanahan, Ames, & Murphy (2003)
1000	Ash from indoor pot belly stove where CCA-treated ply had been burnt over a winter period (US). Analysis method not stated.	Peters et al., (1984)
20,000 to 54,000	Ash from domestic wood burner trial where CCA-treated timber burnt (NZ). Analysis by AAS.	Abbott & Rogers (1989)
33,000 (8,900 to 45,000)	US wood-burning facilities. Conditions of incineration not stated.	Solo-Gabriele & Townsend, (1999)
11,080 to 99,300	Incineration of CCA-wood in an industrial furnace with analysis by neutron activation.	Solo-Gabriele et al., (2002)
69,000	Controlled burn of CCA-treated yellow pine, Alabama (US).	Harden & Pitt, (2013)
84,260	Simulated open burning of CCA-treated southern yellow pine 10-year old decking with analysis by WD-XRF (US).	Wasson et al., (2005)
85,900	As above but with analysis by ICP-MS.	Wasson et al., (2005)

The toxicity of CCA-wood ash residue to farm stock has been observed in New Zealand after cattle died following the ingestion of ashes from a household fire where CCA-treated pine had been burnt (Staples, 1965). Post-mortem results confirmed death was due to arsenic poisoning. An experimental trial established a lethal dose for sheep of

60 mg/kg for arsenic (As_2O_3) derived from CCA-wood ash (Staples, 1965). The human health effects of arsenic in ingested ash depend on its bioavailability, that is, the proportion of total arsenic in the ash that enters the blood stream via the gut after being swallowed. It was suggested by Staples (1965) that the relatively lower toxicity of arsenic in CCA-wood ash compared to the lethal dose for cattle of a soluble inorganic arsenic salt could be explained if only the water-soluble fraction of arsenic in CCA-wood ash was available for assimilation by sheep. Wood ash itself is likely to share some similarities with activated charcoal and be reasonably absorptive for arsenic and other trace elements. Orally administered activated charcoal itself has been widely used to reduce gastrointestinal absorption of various toxins and drugs (Neuvonen & Olkkola, 1988).

As approximately 20% of the arsenic trioxide in CCA-wood ash was water soluble this supports a bioavailability factor of 0.2 to be used in exposure modelling. However, due to interspecies uncertainties between human and animals and lack of specific analysis of CCA-wood ash, a conservative approach of using a bioavailability factor of 1 was adopted. This approach is consistent with the (MfE, 2011d) recommendation that a bioavailability factor of 1 is to be used when setting residential standards for arsenic in soils where there is no site-specific information.

4.2.4.2 Scenario evaluation

Residual ash from the burning of CCA-treated timber, if not appropriately disposed of, presents a risk if inadvertently ingested due to its potentially elevated arsenic levels. It is not an uncommon practice to put wood burner ash on the garden as a soil conditioner or fertilizer for lawns, vegetables and flowers¹⁶. Another practice is to leave the ash outside in a bucket for a period of time until it has cooled or convenient for disposal with the household rubbish as found in social context investigation in Chapter 5.

Young children aged one to six years who play outside, particularly those aged between two and three years, could come into contact with the ash disposed of

¹⁶ <http://www.nzwomansweekly.co.nz/health-home/gardening/ways-for-using-wood-ash-in-the-garden/>

outdoors. This age group exhibits frequent hand-to-mouth behaviour and exposure factors have been derived for daily incidental soil ingestion (MfE, 2011d). There is a very small subset of children who have pica behaviour. Pica is an eating disorder characterised by an appetite for non-nutritional substances such as, soil, clay, ash, metal etc.). Pica children therefore may face higher risk for exposure to CCA-wood ash that has not been appropriately disposed of, or left in an open fire or outside fire pit.

The amount of ash produced (and therefore the frequency of clearing the firebox and disposal of the ash) will vary depending on the frequency of fires and the efficiency of the wood burner. Due to the high uncertainty surrounding the circumstances of children's potential exposure, i.e., frequency and duration of contact with contaminated wood ash, the scenario to be evaluated is a one-off opportunistic contact with ash.

4.2.4.3 Ingestion exposure characterisation

The scope of this exposure assessment was limited to estimating a single episode of oral exposure for a child ingesting CCA-wood ash. Representative concentrations for arsenic in ash were sourced from studies where CCA-wood has been burnt in a domestic appliance, i.e., a lower end estimate of 620 mg/kg allowing for mixing with non-CCA-wood ash and for mixing with surrounding soil during outdoor disposal (McClanahan et al., 2003) and medium to high end estimates of 20,000 to 54,000 mg/kg (Abbott & Rogers, 1989).

A range of ash ingestion rates were used based on soil ingestion rates for a typical child of 50 mg/day, mild pica child of 100 mg/day (MfE, 2011d) and for pica children 1000 mg/day (US EPA, 2008) to 5000 mg/day (Calabrese, Stanek, Pekow, & Barnes, 1997). A trace level of ingestion (5 mg) was also included to represent transient exposure perhaps due to handling of ash followed by hand-to-mouth activity. These exposure factors are presented in Table 4.4. Equation 4.5 was used to calculate possible single episode intake rates of arsenic expressed as mg/kg BW-day that can then be compared to an acute toxicological reference value.

Equation 4.5:

$$ID_{ash} = \frac{C_{ash} \times IR}{BW}$$

Where:

ID_{ash}	=	Intake dose (mg/kg BW)
C_{ash}	=	Concentration of arsenic in ash (mg/kg)
IR	=	Intake rate (mg/day)
EF	=	Exposure frequency
BW	=	Body weight (kg)
BF	=	Bioavailability factor
CF	=	Conversion factor (10^{-6} mg/kg)

Table 4.4: Default exposure parameters used for standard residential setting

Exposure factors	Units	Low-end estimate	Mid-point estimate	High-end estimate
Concentration in ash	mg/kg	620	20,000	55,000
Intake rate	mg/day	5 to 50	100	1000 to 5000
Body weight (child 1-6)	kg	13	13	13
Bioavailability factor		0.2	-	1
Exposure frequency*		1		1

*EF equals 1, representing a daily exposure or a one-off acute exposure situation

4.2.4.4 Background exposure via ingestion

As identified in the conceptual model background sources of arsenic available for uptake via the ingestion pathway are diet and drinking water and incidental ingestion of soils that contain a natural trace component of arsenic. Childhood exposure to dislodgeable arsenic residues whilst playing on CCA-wood structures was also considered.

i) Soils

A survey of topsoils in the region un-impacted by human activity found arsenic¹⁷ ranging from 2 to 7 mg/kg and a sample in the Wainuiomata Scenic Reserve had a concentration of 3.0 mg/kg (URS, 2003). For this assessment the concentration of arsenic in soil from natural sources in residential Wainuiomata is assumed to be 4.0 mg/kg. This does not include areas of soil that may have elevated arsenic due to the

¹⁷ Total recoverable arsenic from nitric/hydrochloric acid digestion followed by ICP-MS (US EPA 200.2) on a dry weight basis.

presence of CCA-wood structures, e.g., soil underneath decking (Stilwell & Gorny, 1997).

The lifetime average daily intake was calculated as mg/kg BW-day according to Equation 4.6 using the default exposure parameters in Table 4.5 (adapted from MfE, 2011d).

Equation 4.6:

$$LADI = \frac{C_{soil} \times IF_{age-adjusted} \times (EF \times ED) \times CF \times BF}{EF \times ED}$$

Where:

<i>LADI</i>	=	lifetime average daily intake (mg/kg BW-day)
<i>C_{soil}</i>	=	arsenic concentration in soil (mg/kg)
<i>IF_{age-adjusted}</i>	=	combined adult and child ingestion rates and body weights
<i>EF</i>	=	exposure frequency (days/year)
<i>ED</i>	=	exposure duration (years)
<i>CF</i>	=	conversion factor mg/kg 10 ⁻⁶
<i>BF</i>	=	bioavailability factor

Table 4.5: Default exposure parameters residential soil ingestion

Exposure parameters	Units	Child (1-6yrs)	Adult	Age-adjusted
Concentration in soil	mg/kg	4	4	
Soil ingestion rate	mg/day	25	50	28.1 [#]
Body weight	kg	13	70	
Bioavailability factor	unitless	1	1	
Exposure frequency	days/yr	350	350	
Exposure duration	yr	6	14	
Averaging time - threshold	yr			75
Averaging time – non-threshold	yr			6

[#] in units of mg yr/kg day based on pro-rated child and adult soil ingestion and body weights for a 75 year lifetime to represent lifetime exposure for intake assessment purposes.

ii) Drinking water and food

The main contributors to ingestion intake of arsenic in New Zealand are food and drinking water (Read, 2003) (MfE, 2011e). The 2009 New Zealand Total Diet Survey (Vannoort & Thomson, 2011) found that weekly dietary exposures of inorganic arsenic ranged from a low of 1.3 µg/kg BW-week for 11-14 year girls, to a high of 3.1 µg/kg

BW-week for 6-12 month infants. Estimated daily intakes of inorganic arsenic by age group were calculated by dividing weekly mid estimates (i.e., limit of detection/2) by seven (Table 4.6). These dietary estimates are conservative as they were based on total arsenic (i.e., organic plus inorganic forms) with the assumption that 10% of total arsenic in fish/seafood is inorganic and 100% of arsenic in other food is inorganic. The focus on inorganic arsenic exists because organoarsenic compounds as found in seafood are regarded as being of very low toxicity. The main contributors to dietary inorganic arsenic are rice and rice products.

Table 4.6: Estimated daily dietary exposure to inorganic arsenic

Exposure parameters	25+ yr males	25+yr females	19-24 yr males	11-14 yr males	11-14 yr females	5-6 yr children	1-3 yr toddlers
Body weight (kg)	82	70	78	54	55	23	13
µg/kg BW-day	0.29	0.24	0.26	0.29	0.19	0.36	0.44
mg/kg BW-day	0.0003	0.0002	0.0003	0.0003	0.0002	0.0004	0.0004

These dietary figures include arsenic from both tap water and bottled water which were found to have a maximum arsenic concentration of 0.003 mg/kg which is equivalent to 0.003 mg/L: about three times lower than the New Zealand drinking-water maximum acceptable value (MAV) of 0.01 mg/L (MoH, 2008). The concentrations for arsenic in drinking water used in the dietary survey are likely to be representative of the Wainuiomata catchment.

The lifetime average daily dose (µg/kg BW-day) for a 75 year period was calculated by weighting the daily dietary survey estimates by life stage and taking the average between male and female.

iii) Surface residues CCA-wood structures in playgrounds

Children may be exposed to arsenic due to transfer of residues containing arsenic from the hands to the mouth following contact with the surface of CCA-wood structures, e.g., at playgrounds (Zartarian et al., 2006). A review commissioned by the Environmental Risk Management Authority (ERMA, now EPA New Zealand) found that although there was insufficient data in the New Zealand context to characterise health risk by this pathway with reasonable certainty, the estimated daily uptake based on

U.S. Consumer Product Safety Commission (US CPSC, 2003) methodology did not present any health concerns (Read, 2003).

In this work, the lifetime average daily intake was calculated as $\mu\text{g}/\text{kg BW}\text{-day}$ according to Equation 4.7 using the default exposure US CPSC (2003) parameters presented in Table 4.7.

Equation 4.7:

$$LADI = \frac{C_{hand} \times HT \times EF \times BF}{BW \times LT}$$

Where:

- LADI* = lifetime average daily intake (mg/kg BW-day)
- C_{hand}* = arsenic residue on palms ($\mu\text{g}/\text{hand}$)
- HT* = hand transfer rate to mouth (%)
- EF* = exposure frequency (days/year)
- ED* = exposure duration (years)
- BF* = bioavailability factor
- BW* = body weight (kg)
- LT* = lifetime (days)

Table 4.7: Default childhood exposure parameters CCA-wood surface residues

Exposure parameters	Units	Child (1-6yrs)	Source
Arsenic on hand	$\mu\text{g}/\text{hand}$	7.6	US CPSC (2003)
Hand transfer rate	%	43	US CPSC (2003)
Body weight	kg	13	MfE (2011d)
Exposure frequency	days/yr	156	US CPSC (2003)
Exposure duration	yr	5	US CPSC (2003)
Bioavailability	unitless	1	US CPSC (2003)
Lifetime	days	27,375 [#]	US CPSC (2003)

[#] 365 days x 75 years

4.2.5 Toxicity assessment method

The toxicity assessment identifies a) the critical health endpoints associated with exposure to arsenic (hazard identification) and b) associated **toxicity values** already developed and available the literature that numerically express the relationship between arsenic exposure and incidence of health effects in a population (**dose-response** assessment) appropriate for the exposure scenarios identified in the sections above.

Toxicity values are typically derived for **chronic** exposure (usually years to a lifetime) at relatively low concentrations and **acute** exposure (a short period of time such as minutes, hours or a day) at relatively high concentrations. **Sub-chronic** exposure refers to situations in which exposure occurs repeatedly over a period of time is longer than acute timeframe but significantly shorter than a lifetime exposure, e.g., daily exposure that continues for six months. For air toxics sub-chronic exposures are not commonly assessed. It is important to select a toxicity value that applies to the duration and frequency of the exposure concentration that is being evaluated for health risk.

Toxicity values generally fall into two categories – those for carcinogenic effects and those for non-cancer effects. Arsenic exposure can lead to cancer and non-cancer health endpoints depending on the exposure characteristics, i.e., magnitude and duration and route of exposure and the chemical form, particle size and solubility of arsenic taken into the body.

4.2.5.1 Toxicity overview

4.2.5.2 Carcinogenic effects

i) Evidence for carcinogenicity

Lung cancer is considered to be the critical effect resulting from arsenic inhalation exposure (WHO, 2013). Inorganic arsenic has been classified as a known human carcinogen by the International Agency for Research on Cancer (IARC) (Group 1) (IARC, 1998, 2012) and by the US EPA (Group A)¹⁸. The underlying mechanism for carcinogenicity has not yet been clearly identified (Hughes, 2002). Investigators have proposed multiple **modes of action** (MOAs) for carcinogenicity and there is ongoing debate about whether arsenic acts as a linear (non-threshold) or non-linear (threshold) carcinogen (Erraguntla, Sielken, Valdez-Flores, & Grant, 2012). In addition, there are several factors such as gender, ionising radiation, smoking, diet or genetic susceptibility which may act synergistically or as confounders influencing the dose-response curve for inhalation exposure and lung cancer (Tapio & Grosche, 2006).

IARC and US EPA both treat arsenic as a linear non-threshold carcinogen and therefore in developing health protection limits it is assumed that the dose-response relationship

¹⁸ <http://www.epa.gov/IRIS/subst/0278.htm#woe>

established for high exposure situations can be linearly extrapolated to the much lower concentrations found in other environmental exposure scenarios. Some authors question the applicability of using high-end occupational exposure dose-response relationships to evaluate health risk at the much lower exposures experienced by the general public (Lewis, Beyer, & Zu, 2015). A review of studies carried out to determine associations between lung cancer and residential exposure to arsenic (e.g., residents living near smelters) found these studies lacked reliable exposure data and therefore could not confirm or disprove whether relationships found between occupational exposure and health effects also applied to environmental exposures (EC, 2000).

The US EPA cancer risk assessment guidelines (US EPA, 2005) state that it is sometimes appropriate to apply an age adjustment factor for childhood exposures to account for the potential for early-life exposure to make a greater contribution to cancers appearing later in life, particularly for carcinogens with a mutagenic MOA. Arsenic shows little or no mutagenic potential (Tapio & Grosche, 2006) and therefore age adjustment factors were not applied in this assessment. However, there is some evidence from a Chilean study that found increased risks of lung cancer in young adults exposed to high levels of arsenic in drinking water during childhood (Smith et al., 2006).

ii) **Regulatory agency inhalation toxicological reference values**

The WHO, US EPA, California EPA and Texas Environmental Commission has published cancer **unit risk factors** (URF) for chronic arsenic inhalation (Table 4.8). URFs are defined as the excess or additional cancer risk occurring in a hypothetical population in which all individuals are exposed continuously throughout their lifetime to a mean concentration of $1 \mu\text{g}/\text{m}^3$ of the carcinogenic agent in the air they breathe.

These URFs have been used by agencies to set regulatory guidelines for ambient air to meet a prescribed level of risk of excess lifetime cancer cases in a population exposed to environmental concentrations. For example, based on the US EPA URF of 1.4×10^{-3} the corresponding environmental exposure concentration that represents 1 in 100,000 excess lifetime cancer risk is $2.3 \text{ ng}/\text{m}^3$ ¹⁹. Both WHO and US EPA caution that the URF

¹⁹ EC = 0.00001/0.0013/1000

is not equivalent to the true cancer risk, but represents plausible upper bounds which may vary widely according to the assumptions on which they are based. New Zealand's guideline for arsenic in ambient air of 5.5 ng/m³ was selected as a figure that was appropriate because it was within the range of three exposure concentrations derived from the US EPA, California EPA and WHO URFs for an "acceptable" risk level of 1 x 10⁻⁵ (Chido & Rolfe, 2000).

Table 4.8: Arsenic URFs published by agencies

Agency	Unit risk per 1 µg/m ³	Equivalent exposure concentration for 1 x 10 ⁻⁵ excess cancer risk
US EPA, IRIS (1984, unchanged following 1998 review)	4.3 x 10 ⁻³	2.3 ng/m ³
Californian EPA (1990)	3.3 x 10 ⁻³	3.0 ng/m ³
WHO (2002)	1.5 x 10 ⁻³	6.6 ng/m ³
Texas Commission on Environmental Quality (Erraguntla et al., 2012)	1.5 x 10 ⁻⁴	67 ng/m ³

The US EPA risk estimates for cancer (established in 1984) are based on the pooled results of studies at two US smelters (Montana and Tacoma) of adult male workers who were exposed to high levels of arsenic trioxide (US EPA Integrated Risk Information System²⁰). Another pooled unit risk estimate of 1.43 x 10⁻³ based on a reanalysis of the Tacoma smelter data and a study of Swedish smelter workers (Rönnskar) produced by (Viren & Silvers, 1994) underpins the current WHO URF of 1.5 x 10⁻³ (WHO 2002). A recent WHO review using studies published after 2005 found insufficient evidence to warrant changing their URF of 1.5 x 10⁻³ (WHO, 2013). Citing the findings of Smith, Ercumen, Yuan, and Steinmaus (2009) that arsenic toxicity depends on absorbed dose independent of whether it was inhaled or ingested, WHO (2013) note that if cancer risk estimates for arsenic in drinking water (calculated as a function of arsenic concentrations in urine) were transformed into equivalent inhaled levels of arsenic then the cancer risk for the general population at low-level arsenic concentrations would transform into a unit risk of 1 x 10⁻³, very similar to the 1.5 x 10⁻³ calculated from completely different data.

²⁰ <http://www.epa.gov/IRIS/subst/0278.htm>

More recently the Texas Commission on Environmental Quality (TCEQ) using updated cohort mortality data from the US smelter studies together with the Swedish cohort study developed a much lower risk factor of 1.5×10^{-4} that took into account the impact of age on lung cancer risk (Erraguntla et al., 2012).

The UK Expert Panel on Air Quality Standards (UK EPAQS) have adopted a threshold approach instead of the unit risk approach due to the inherent uncertainty in extrapolating from observed effects at high levels of exposure (i.e., occupational) to responses at the much lower concentrations commonly associated with environmental exposure. The approach used is to identify a level of exposure at which **no adverse health effects have been observed** (NOAEL) or the **lowest observed adverse effects level** (LOAEL) and apply safety factors to account for features, such as inter species differences, to protect susceptible groups in humans and for extrapolation from shorter duration occupational exposures to lifetime exposures. In the case of the EPAQS a LOAEL was derived from smelter occupational data and then divided by 10 to provide a notional NOAEL which was then divided by a further factor of 10 to allow for the greater exposure duration of the general public and then divided again by 10 to allow for the presence of susceptible groups within the general public. This led to a recommended ambient air guideline of 3 ng/m^3 (UK EPAQS, 2008). The European Commission has adopted a guideline of 6 ng/m^3 ²¹.

In this toxicity assessment the WHO URF was selected as appropriate for characterising the excess lung cancer risk following chronic or lifetime exposure as it has been recently reviewed (WHO, 2013). Following the precautionary approach of MfE (2011e) arsenic is considered to be a non-threshold carcinogen in absence of definitive evidence either way.

4.2.5.3 Non-cancer effects

i) Inhalation exposure route and toxicological reference values

Following acute inhalation exposure of workers to high levels of arsenic dusts and fumes gastrointestinal effects (nausea, diarrhoea, abdominal pain) as well as central and peripheral nervous system disorders have been reported (ATSDR, 2007). No

²¹ <http://ec.europa.eu/environment/air/quality/standards.htm>

deaths after acute arsenic exposure via inhalation have been reported (WBK & Associates, 2004).

Chronic inhalation exposure to arsenic is associated with irritation of the skin and mucous membranes, including dermatitis, conjunctivitis, pharyngitis and rhinitis (MfE, 2002).

For screening risk assessments of health risks associated with acute and chronic inhalation exposure to arsenic compounds US EPA (2004) recommends toxicity reference values published by U.S. NIOSH (National Institute for Occupational Safety and Health) and the California EPA ²².

The NIOSH defines the concentration of arsenic in air deemed to be **Immediately Dangerous to Life or Health** (IDLH) for occupational exposure to be 5 mg/m³ based on animal toxicity data. The US EPA (2004) recommends using the IDLH/10 of 0.5 mg/m³ which is one tenth of the IDLH and is approximately comparable to mild effects levels for 1-hour exposure²³.

The Office of Environmental Health Hazard Assessment (OEHHA) of the Californian Environmental Protection Agency (California EPA) has produced Reference Exposure Levels (RELs) for health risk screening assessments for residential exposure to stationary sources of hazardous air pollutants under California's Air Toxics Hot Spots and Toxic Air Contaminants programmes (OEHHA, 2003). A REL is defined as the concentration level at or below which no adverse health effects are anticipated for a specified exposure duration. RELs are based on the most sensitive, relevant health effect reported in the medical and toxicological literature and include generous margins of safety to protect the most sensitive individuals in the population (US EPA 2004). Exceeding a REL does not necessarily indicate that an adverse health impact will occur. Instead levels above the REL have an increasing, but undefined probability of resulting in adverse health effects, particularly in sensitive individuals (OEHHA, 2003).

An acute REL of 0.2 µg/m³ for a 4-hour exposure duration was established based on an animal study in which maternal arsenic exposure was associated with decreased foetal

²² <http://www.cdc.gov/niosh/idlh/7440382.html>

²³ <http://www2.epa.gov/sites/production/files/2014-05/documents/table2.pdf>

weight (Nagymajtenyi, Selypes, & Berencsi, 1985). An uncertainty factor of 1000 was applied to account for interspecies uncertainty and inter-individual variation. The California EPA notes that inorganic arsenic oxides in drinking water are considered to be developmental toxicants and reductions in foetal weight increases the risk of infant mortality (OEHHA, 2008).

The 8-hour and chronic REL for arsenic compounds in air of $0.015 \mu\text{g}/\text{m}^3$ is based on the critical effect of decreased intellectual function in 10 year old children (OEHHA, 2008). This REL is based on studies of children in developing countries being exposed to arsenic in drinking water (Tsai, Chou, The, Chen, & Chen, 2003; Wasserman et al., 2004). The California EPA has used route-to-route extrapolation to derive an inhalation REL from an oral LOAEL by applying a cumulative uncertainty factor of 30 to account for inter-individual variation and for infants who are potentially more sensitive and were not in the original study. In this case the 8-hour REL is taken to be equivalent to the chronic REL due to the possibility of repeated exposure and the relatively slow clearance of arsenic compounds from the body. The 8-hour REL is designed to reflect exposure due to chemical-emitting facilities that operate 8 hours a day, 5 to 7 days a week and to be matched to a concurrent exposure duration, for example exposure of children in schools. The chronic RELs are designed to reflect long-term exposure, i.e., 24-hours per day from 9 years to a lifetime duration at which adverse non-cancer health effects would not be expected. These RELs were treated as indicative only for health risk screening purposes.

ii) Oral exposure route and toxicological reference values

Non-cancer effects associated with oral exposure to arsenic include dermal lesions, skin pigmentation, keratosis, peripheral vascular disease and cardiovascular effects. Long term exposure is associated with the development of skin lesions which is a diagnostic criteria for arsenicosis. Based on a toxicity review by MfE (2011e) the lowest reported dosage associated with increased incidence of skin lesions is $0.0012 \text{ mg}/\text{kg}/\text{day}$ (Ahsan, Chemm, & Parvez, 2006).

Acute effects for high oral exposures include vomiting, diarrhoea, and gastrointestinal haemorrhage, and death may ensue from fluid loss and circulatory collapse or multiple

organ or tissue damage (ATSDR, 2007). The potentially lethal dose for arsenic via ingestion has been estimated to be between 1 and 3 mg/kg BW (ATSDR, 2007) and as low as 0.6 mg/kg BW (US EPA, 2012)²⁴. For this assessment an acute oral lethal reference level of 1 mg/kg BW was selected for risk characterisation for a child ingesting CCA-wood ash, to be consistent with the approach adopted by the Pattle Delamore Partners (PDP) calculation of acceptable soil concentrations for acute exposure to arsenic in soil for a contaminated site in Thames (Foote & Rumsby, 2011). Similarly, the NOAEL of 0.015 mg/kg BW-day for arsenic ingestion by children derived by Tsuji, Benson, Schoof, and Hook (2004) for acute exposure defined as less than 14 days was adopted.

4.2.5.4 Summary of selected toxicity reference values

Table 4.9 presents the dose-response values selected as appropriate inputs for the risk characterisation (Section 4.2.6).

Table 4.9: Toxicity reference values selected for risk characterisation

Duration	Scenario	Health-end point	TRV	Units	Metric	Source
Inhalation						
Chronic	Residential – annual lifetime	Cancer	0.0015	µg/m ³	URF	WHO (2000)
Sub-chronic	Residential – winter	Non-cancer	0.015	µg/m ³	EC	OEHHA (2008)
Chronic	Residential – annual lifetime	Non-cancer	0.015	µg/m ³	EC	OEHHA (2008)
Acute	Residential – sub 24-hour	Non-cancer	0.20	µg/m ³	EC	OEHHA (2008)
Acute	Residential – 1-hour	Mild effects	0.5	mg/m ³	EC	NIOSH (1994)
Oral						
Chronic	Residential background	Internal cancers	0.0086	µg/kg BW-day	Dose	MfE (2011e) [#]
Chronic	Residential background	Dermal	0.0012	mg/kg/day	Dose	ATSDR (2007)
Acute	Residential ash	Non-lethal effects in children	0.015	mg/kg/day	Dose	Tsuji et al. (2004)
Acute	Residential ash	Lethal effects	1	mg/kg BW	Dose	ATSDR (2007)

From Health Canada (2005) cited in MfE (2011e)

²⁴ <http://www.epa.gov/ttn/atw/hlthef/arsenic.html>

4.2.6 Risk characterisation method

The risk characterisation process integrated the results of the exposure modelling with the relevant toxicity values so that health risk could be quantified based on the time-weighted exposure concentrations for the selected exposure scenarios. All risk characterisation calculations were carried out using the Microsoft Excel spreadsheet developed for the exposure model in Section 4.2.3 utilising Equations 4.8 and 4.9 described below.

The cancer and non-cancer health risk estimates were presented as deterministic estimates of central tendency for a typical individual in each of the exposure scenarios together with the estimated number of people in the population of Wainuiomata who shared those exposure characteristics.

These risk estimates were based on point estimates of exposure and toxicity in line with a screening-level (Tier 1) analysis. There was insufficient information available on the probability distribution of exposure characteristics to adopt a probabilistic risk assessment approach.

4.2.6.1 Cancer risk estimates

Estimated individual cancer risk was expressed as the upper bound probability that a person may develop cancer over the course of their lifetime attributable to their exposure to arsenic under the hypothetical exposure scenarios under investigation. This predicted cancer risk is termed **incremental lifetime risk (ILTR)** as it is over and above the risk that individuals in the population already face due to non-arsenic related exposure – e.g., genetic pre-disposition or lifestyle factors. The ILTR for an individual was calculated by Equation 4.8.

Equation 4.8:

$$ILTR = EC_L \times IUR$$

Where:

<i>ILTR</i>	=	cancer risk to an individual (expressed as an upper-bound risk of contracting cancer over a lifetime)
<i>EC_L</i>	=	estimate of long-term inhalation exposure concentration for arsenic (assumed to be 75 years)
<i>IUR</i>	=	inhalation unit risk

4.2.6.2 Non-cancer risk estimates

The possibility of non-cancer adverse health effects were expressed as a unitless Hazard Quotient (HQ) estimated by dividing the estimate of the chronic inhalation by the appropriate **toxicity reference value** (TRV) as per Equation 4.9.

Equation 4.9:

$$HQ = \frac{EC_c}{TRV}$$

Where:

<i>HQ</i>	=	Hazard Quotient for an individual HAP
<i>EC_c</i>	=	estimate of inhalation exposure concentration
<i>TRV</i>	=	toxicity reference value, e.g., RfC

Based on the definition of the TRV, an HQ of less than or equal to one indicates that adverse non-cancer effects are not likely to occur, and thus can be considered to have negligible hazard. HQ's do not represent statistical probabilities of harm – they are a simple statement of whether and by how much an exposure concentration exceeds a particular TRV. As the reference values are designed to prevent the onset of harm, HQ values above 1 indicate that further more detailed investigation may be warranted to determine the true extent and nature of exposure and health risks.

4.3 Results and discussion

4.3.1 Inhalation exposure and risk characterisation

Table 4.10 presents the concentrations of arsenic found in outdoor and indoor air as determined by outdoor monitoring and estimated by the indoor microenvironment model for each of the exposure scenarios under evaluation, namely, a non-smoking and smoking residence and a non CCA-wood burning and a CCA-wood burning residence. In Table 4.10 “baseline” represents the exposure concentration attributable to outdoor levels of arsenic only and “+ ETS” represents the additional contribution of environmental tobacco smoke to the baseline level. Similarly, “+ CCA-wood burnt” includes the contribution of arsenic from CCA-wood burning inside the home to the exposure concentration. The indoor and outdoor concentrations were calculated for averaging times representing the three modelled exposure durations: annual, peak day (24-hour), and winter average (3-month).

Table 4.10: Estimated exposure concentration arsenic in air ($\mu\text{g}/\text{m}^3$) by microenvironment

Scenario	Annual	Winter	Winter peak
Ambient monitored	0.0061	0.0146	0.0790
Baseline	0.0068	0.0164	0.0885
+ ETS	0.0137	0.0245	0.0966
+ CCA-wood burnt	0.0695	0.7577	0.8298
+ ETS + CCA-wood burnt	0.0764	0.7658	0.8379

Table 4.11 shows 1-year exposure concentrations estimated for each scenario based on the microenvironment concentrations weighted by time spent indoor vs outdoors for an adult resident and hypothetical child (5-11 years) based on Table 4.2 and with children’s time patterns based on Canadian data (Matz et al., 2014). The exposure model is conservative (i.e. may slightly over-estimate exposure, and is not expected to under-estimate exposure) because it does not include periods in other microenvironments (such as working outside of Wainuiomata) where inhalation exposure to arsenic may be low or negligible, which would lower the overall estimate.

Table 4.11: Estimated 1-year exposure concentration arsenic in air ($\mu\text{g}/\text{m}^3$) weighted by time spent in microenvironment by exposure scenario

Scenario	Annual	Winter	Winter peak
Child (5-11 years)			
Baseline	0.0051	0.0121	0.0655
+ ETS	0.0100	0.0179	0.0713
+ CCA-wood burnt	0.0497	0.5407	0.5940
+ ETS + CCA-wood burnt	0.0546	0.5465	0.5998
Adult			
Baseline	0.0049	0.0117	0.0636
ETS	0.0097	0.0174	0.0692
CCA-wood burnt	0.0486	0.5285	0.5803
ETS + CCA-wood burnt	0.0534	0.5341	0.5859

Lifetime-weighted exposure concentrations for arsenic in outdoor and indoor air are shown in Table 4.12 based on the most conservative assumption that a resident lives in Wainuiomata all their life, i.e., 75 years. The contribution of environmental tobacco smoke to lifetime weighted exposure concentration for residents is about the same as the contribution from arsenic in outdoor air. Using CCA-wood for firewood in the home substantially increases lifetime weighted exposure by a factor of 10 compared to indoor concentration arising from outdoor infiltration only.

Table 4.12: Estimated lifetime (75-year) weighted average exposure concentrations for arsenic in air ($\mu\text{g}/\text{m}^3$) by exposure scenario

Scenario	Duration (years) resident			Lifetime weighted average
	4	7	64	
No. years	4	7	64	75
Life stage	1-4 yrs	5-11 yrs	12-75 yrs	1-75 yrs
Baseline	0.0210	0.0354	0.3141	0.0049
+ ETS	0.0413	0.0697	0.6207	0.0098
+ CCA-wood burnt	0.2065	0.3482	3.1097	0.0489
+ ETS + CCA-wood burnt	0.2268	0.3825	3.4163	0.0537

4.3.1.1 Cancer risk characterisation

Figure 4.2 displays the relative estimated lifetime additional risk due to inhalation exposure, by exposure scenario. A resident who is not exposed to ETS and does not burn CCA-treated timber has an estimated risk of 0.9×10^{-5} (less than 1 in 100,000). In contrast, being exposed to ETS in the home marginally increases the lifetime risk (1.6×10^{-5}). Burning CCA-wood inside the home increases the theoretical risk to 7.5×10^{-5} which is above the tolerable excess cancer threshold of 1 in 100,000 (MfE, 2002). An

underlying assumption to this result is that a person is exposed continuously to the estimated long-term inhalation exposure concentration (based on 2012 and 2013 monitoring results) for their full lifetime, assumed to be 75 years. Based on estimated exposure concentrations, burning CCA-treated wood is likely to contribute about 10 times more arsenic to the indoor air than baseline levels, and about five times more arsenic than estimated from environmental tobacco smoke. These relative concentration magnitudes are also reflected in the relative risks as shown in Figure 4.2. The exposure model was also used to predict that an annual average arsenic concentration of 85 ng/m³ in outdoor air would be required to reach an excess cancer risk to 1 in 10,000 for the baseline exposure scenario.

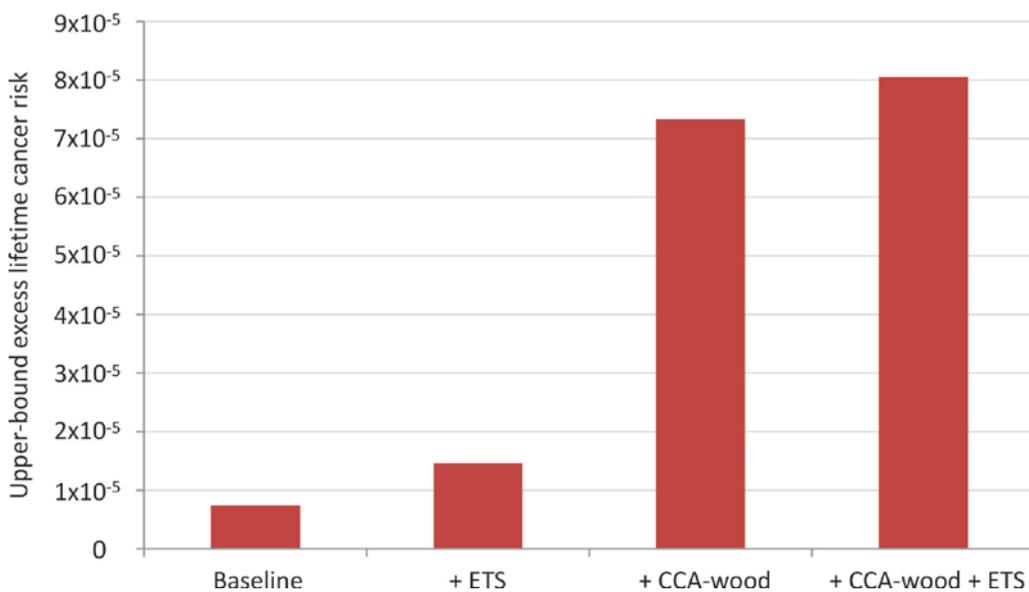


Figure 4.2: Individual excess carcinogenic risk for a resident attributable to lifetime arsenic inhalation based on IUR 1.5 x 10⁻³ per µg/m³

4.3.1.2 Non-cancer risk characterisation

Figure 4.3 displays the hazard quotients for lifetime exposure concentration (75 years) based on the California EPA chronic REL of 0.015 µg/m³ and the 24-hour peak exposure concentration for a 5 to 11 year old child based on the California EPA acute REL of 0.2 µg/m³. If CCA-wood is burnt inside the house the hazard quotients for both lifetime and short term exposures exceed 1 indicating potential for adverse health effects. If the mean winter exposure concentration was taken to represent a sub-chronic exposure, i.e., repeated 8-hour periods, and is compared to the California EPA chronic

REL of 0.015 µg/m³ then the HQ for the CCA-wood burning exposure scenario is 36 for a 5 to 11 year old and 38 for a 1 to 4 year old (not shown on Figure 4.3).

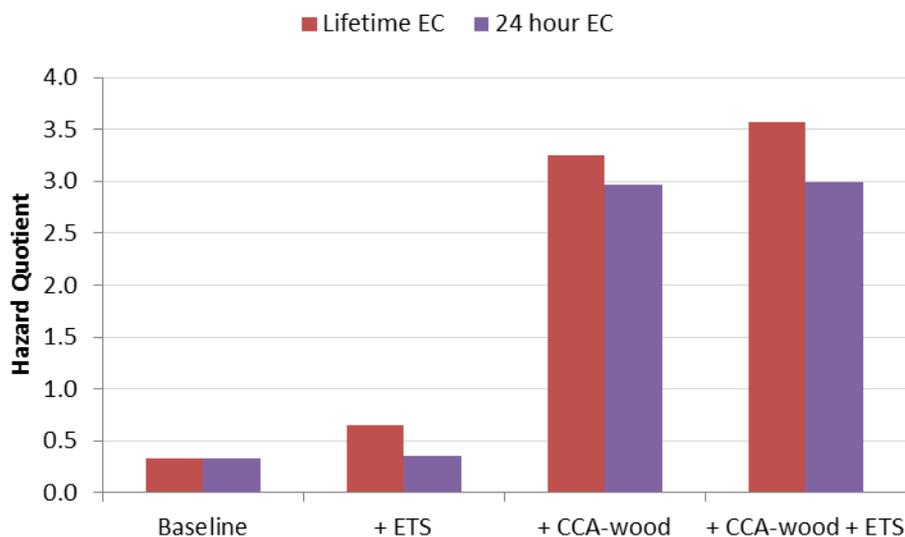


Figure 4.3: Hazard Quotients for non-cancer effects for lifetime exposure and 24-hour exposure based on 2012 to 2013 concentrations of arsenic in Wainuiomata air and other assumptions

Children were predicted to have the highest exposure concentration because they spent more time indoors. The number of children usually resident in Wainuiomata in each exposure scenario (mutually exclusive categories) was estimated by matching Statistics New Zealand 2013 Census data for individual smoking and ages of children with household data on method of home heating (Table 4.13).

Table 4.13: Number of children in Wainuiomata who live in a household with smoker(s) and/or where solid fuels are used for heating

No. children (1-4 years)	No. children (5-11 years)	Live in household with at least one regular smoker	Live in household that uses solid fuels for heating
360	616	NO	NO
251	530	YES	NO
370	461	NO	YES
178	303	YES	YES
1158	1911	Total	Total

Using the information in Table 4.13, the probabilities of children living in a smoking and/or wood burning house was calculated. The probability of a wood burning house using CCA-wood was estimated by multiplying the probability of wood burning ($p=0.36$) by the probability of using CCA-wood ($p=0.16$)²⁵ to get a probability of $p=0.058$. The number of households potentially burning CCA-wood was estimated as 344 which is the total number of households in Wainuiomata, 5976 x $p=0.058$. Table 4.14 then shows the number of children by age group in each exposure scenario calculated by multiplying the probability of each exposure scenario by the number of children that are usually resident in Wainuiomata. This information was then used to estimate the number of children who may be experiencing a higher level of health risk, indicated by an HQ above 1 for particular exposure scenario and exposure duration, compared to those who are less exposed (Table 4.15).

Table 4.14: Number of children by age group in each exposure scenario

Scenario	Children (1 to 4 years)		Children (5 to 11 years)	
	Probability	Number	Probability	Number
Baseline	0.74	854	0.69	1310
ETS only	0.22	251	0.28	530
CCA-wood only	0.018	21	0.014	27
ETS + CCA-wood	0.027	32	0.023	44
Total	1.00	1158	1.00	1911

Table 4.15: Number of children (1-11 years) and HQ by exposure scenario and exposure duration

Exposure scenario and duration	HQ	No. children (%)
3-month winter period and ETS	1.3	781 (25%)
3-month winter period and CCA-wood burnt in home with or without ETS	36-38	124 (4%)
24-hour period and CCA-wood burnt in home with or without ETS	3	

²⁵ The number of households burning CCA-wood in Wainuiomata was estimated 348 (i.e., 16% of 2175 wood burning houses; Section 4.2.3.1)

4.3.2 Ingestion exposure and risk

The potential intake dose of arsenic for a child ingesting CCA-wood ash under a range of exposure assumptions is presented in Table 4.16. These exposure assumptions are based on a combination of daily intake rate, concentration of arsenic in ash and percentage gastrointestinal bioavailability of arsenic (either 20% or 100%).

Table 4.16: Calculated intake dose (mg/kg BW) for one-off exposure to CCA-wood ash for a child at different ingestion rates

Exposure parameters	Trace	Typical	Mild Pica	Pica	Extreme Pica
Intake rate (mg/day)	5	50	100	1000	5000
Bioavailability 100%					
Low arsenic ash (620 mg/kg)	0.0002	0.0024	0.0048	0.0477	0.2385
Medium arsenic ash (20,000 mg/kg)	0.007	0.0769	0.1538	1.5385	7.6923
High arsenic ash (54,000 mg/kg)	0.0208	0.2077	0.4154	4.1538	20.7692
Bioavailability 20%					
Low arsenic ash (620 mg/kg)	0.00005	0.0005	0.0010	0.0095	0.0427
Medium arsenic ash (20,000 mg/kg)	0.0015	0.0154	0.0308	0.3077	1.5385
High arsenic ash (54,000 mg/kg)	0.0042	0.0415	0.0831	0.8308	4.1538

Depending on the amount ingested and the arsenic concentration in ash there is the possibility that the potentially lethal dose for arsenic of 1 mg/kg-BW could be consumed by a child who has access to ash. Figure 4.4 shows the theoretical relationship between ingestion rate and arsenic concentration that equate to a 1 mg/kg BW dose for a 13 kg child, assuming arsenic has 100% gastrointestinal bioavailability. Physically the maximum possible concentration of arsenic in ash, assuming arsenic is present in pure oxide form as As_3O_4 , is 778,369 mg/kg²⁶. Therefore, the value of 778,369 mg/kg was set as the upper maximum arsenic concentration in ash in Figure 4.4.

Based on the medium to high range of arsenic in CCA-wood ash concentrations reported in the literature review a child would need to ingest between 1000 to 5000 mg of ash (one fifth of a teaspoon upwards) to risk acute poisoning. Ingesting this quantity of ash is possible for a pica child. For acute non-lethal health effects sufficient

²⁶ As_3O_4 is 77.8% arsenic and 22.2% oxygen by weight

ash in the medium concentration range could be consumed by a non-pica child so that the NOAEL of 0.015 mg/kg-BW is exceeded.

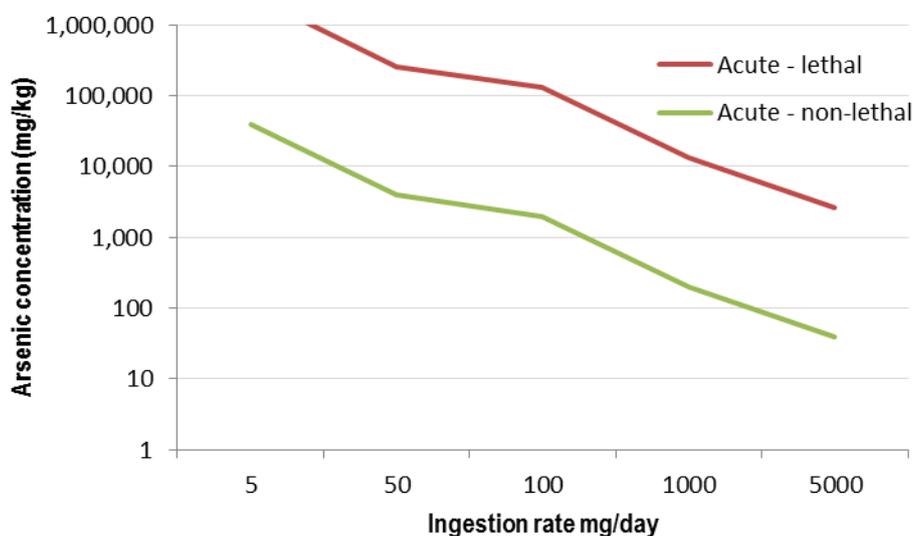


Figure 4.4: Arsenic dose (mg/kg BW) for a 13 kg child calculated from arsenic concentration in ash (assuming 100% gastrointestinal bioavailability) and ingestion rate required to reach acute toxicity reference values

4.3.3 Background exposure and risk

The estimated contribution to individual arsenic exposure from incidental ingestion of soils, diet and active smoking and the predicted excess lifetime cancer risk attributable to these 'background' sources is presented in Table 4.17. Lifetime cancer risk due to direct inhalation exposure of arsenic in cigarettes for a pack a day smoker was estimated as 3.6×10^{-5} . Note this is different to lifetime cancer risk for active smoking which is higher due to presence of other cancer promoting agents in cigarettes (i.e., a risk of 7.9×10^{-2} has been calculated from US mortality statistics (Fowles & Bates, 2000).

Table 4.17: Predicted lifetime excess cancer risk from exposure to 'background' sources

Sources	Dose	units	TRV	End point	Excess risk
Soil ingestion	0.00011	mg/kg BW-day	8.6×10^{-5}	Internal cancers	9.3×10^{-10}
Diet	0.00028	mg/kg BW-day	8.6×10^{-5}	Internal cancers	2.4×10^{-6}
Active smoking	0.024	$\mu\text{g}/\text{m}^3$	1.5×10^{-3}	Lung cancer	3.6×10^{-5}
CCA-wood residues ingestion	0.00716	$\mu\text{g}/\text{kg BW-day}$	8.6×10^{-2}	Internal cancers	6.2×10^{-5}

Figure 4.5 shows how the excess lifetime cancer risk for ‘background’ sources compares to that from baseline exposure to ambient levels of arsenic in air from burning of CCA-wood and ETS indoors.

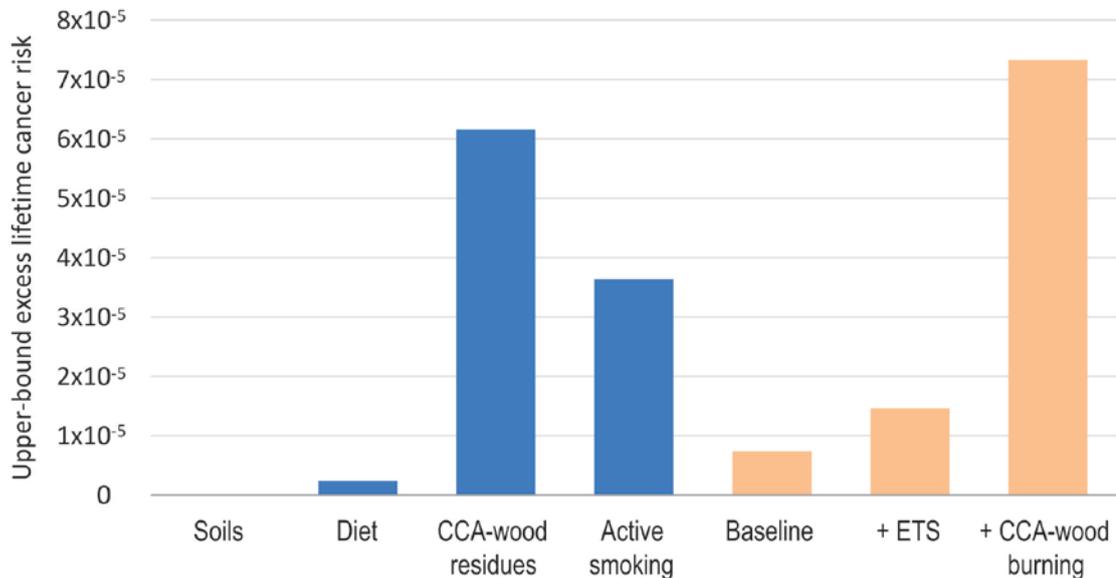


Figure 4.5: Estimated excess lifetime cancer risk by source

4.4 Uncertainty and sensitivity analysis

Air toxics health risk assessments should include a discussion of uncertainty and variability to be fully informative and to assist decision making (US EPA, 2004).

Variability refers to true heterogeneity or diversity that occurs within a population or sample. In this study outdoor arsenic concentrations were only measured at one location and therefore the variability in arsenic concentration across the wider airshed is unknown, but likely to be more variable than PM_{2.5}.

Uncertainty in health risk assessment occurs because of a lack for knowledge about specific factors, parameters or models underpinning the assessment. These types of uncertainty affect the confidence of any risk characterisation. Many parameter values (e.g., emission rates) may be both uncertain and variable. Uncertainty can often be reduced by further measurement or study, but intrinsic variability cannot be reduced, only better characterised (US EPA, 2004).

In this health risk assessment the largest source of uncertainty for the inhalation risk characterisation was the input parameters used to model the indoor levels of arsenic arising from the burning of CCA-treated timber inside the home. The uncertainty concerning both the typical quantity of CCA-wood burnt and the corresponding level of emissions that might enter the indoor environment during lighting and refuelling is very large. Due to this uncertainty it is not possible to assess the degree to which the health risks from this activity might be under or over-stated. To close the knowledge gap, in-situ testing and indoor monitoring would be required as well as further research into the exact amount and frequency of CCA-wood burning. This exercise would be a major undertaking given the inherent difficulty in characterising indoor air quality due to wood burning, due to inter-household variability.

As the risk characterisation was conducted using a deterministic exposure model, a sensitivity analysis was undertaken to quantify the effect of variability in the exposure model input values, where these were known on the ILCR and HQs for each scenario generated by plausible variation in the inputs.

The upper and the lower confidence levels for ambient arsenic monitoring data (Table 3.2) in the exposure model for generating exposure concentrations had very little effect on the ILCR and HQs for each exposure scenario. In terms of the modelled exposure concentration for arsenic in environmental tobacco smoke, use of the upper and lower estimates from the literature of arsenic yield in side stream cigarettes had only a marginal effect on the ILCR and HQs for exposure to second-hand cigarette smoke. The impact of number of cigarettes smoked per hour on exposure concentrations and risk was more significant. If there were four people smoking instead of one, then the exposure concentration and risk was more than doubled. The values selected for hourly air exchange were important for exposure concentrations resulting from indoor sources such as ETS and CCA-wood burning.

The unit risk factor (URF) selected for inhalation risk characterisation was the most important factor influencing the results. When the lowest Texas Commission on Environmental Quality (Erraguntla et al., 2012) URF was used (Table 4.8) the excess cancer risk was less than 1 in 100,000 for all exposure scenarios.

4.5 Synthesis

Of the various sources of arsenic that the general population are exposed to, people who burn CCA-treated wood have the highest potential exposure. People who do not burn treated timber themselves, but live in a community where treated timber is burnt by others leading to outdoor concentrations of the magnitude found in Wainuiomata, were predicted to have an excess lifetime cancer risk just within the tolerable limit of 1 in 100,000. A smaller number of residents who are also exposed to arsenic from indoor sources, such as second hand cigarette smoke and from using CCA-treated wood for firewood, were predicted to have an excess lifetime cancer risk eight times higher than the tolerable limit of 1 in 100,000.

The exposure model used to predict exposure concentrations does not specifically consider exposure to trace arsenic levels from active smoking. It was found that exposure to CCA-treated wood emissions originating inside the home were be about twice that received from by an active smoker. For context it is noted that the overall lifetime risk cancer from smoking, due to other carcinogens present in cigarettes, is about 1000 times higher than the theoretical risk from CCA-wood burning exposure inside the house.

Children's predicted lifetime excess cancer risk due to ingestion of residues containing arsenic during contact with CCA wood, for example on playground structures, was calculated to be 6.2×10^{-5} excess lifetime cancer risk when calculated using default USA input parameters. This is only slightly less than the 7.2×10^{-5} excess lifetime cancer risk estimated to arise from indoor exposure as a result of CCA-wood being burnt inside the home. It must be stressed that this number is very uncertain and likely to be an upper estimate.

Non-cancer related health risks are unlikely to exist for children who do not live with smokers and where CCA-treated wood is not burnt in the home. However, hazard quotients for winter and 24-hour average exposure to children in homes where CCA-wood is used were in the mid-30s based on the California EPA's reference screening level. This suggests that further investigation may be warranted to determine the true extent and nature of children's exposure and health risks as a result of winter time

exposure to arsenic inside the home. 124 children (4% of resident population) were estimated to be in this hazard quotient category assuming the 16% of wood burning households use CCA-treated timber (equates to 344 houses).

Ash of CCA-treated wood is likely to pose a serious hazard to children. If ingested in modest quantities it has the *potential* to cause fatality, if it is assumed that the arsenic content of the ash residue is 100% gastrointestinally available.

5 Social behavioural context investigation

5.1 Introduction

This chapter includes all the components of the research that directly relate to the investigation of the social context surrounding wood burning for home heating focused on aspects related to the practice of using CCA-wood in the community. One of the primary motivations for this thesis is to investigate the feasibility of improving air quality (and hence reducing adverse health risk) by discouraging the use of treated timber for firewood for home heating. This intention is described by Aim 4 and its objectives in Section 1.2.2. In this Chapter, details of the methodology are provided in Section 5.2 with the results presented in Section 5.3 and discussion of the results in Section 5.4.

5.2 Methodology

5.2.1 Philosophical stance

The social context investigation followed a **constructivist** approach to belief about knowledge or epistemology that reality is constructed by social, historical and individual contexts, so there is no absolute shared truth. Social constructivism does not deny an objective reality of the natural world – but is concerned with the everyday interactions between people and how they use experience and language to construct their understanding of the environment around them (Burningham & Cooper, 1999). A common goal of social constructivism is to understand the world of lived experience from the perspective of those who live in it (Andrews, 2012) and in doing so in this work it may provide a fresh perspective for understanding and management of air pollution risks (Bickerstaff & Walker, 2003).

5.2.2 Theoretical lens

It could be argued that the burning of treated timber in domestic setting is an **environmentally significant** behaviour because of the potential for air toxics (specifically inorganic arsenic) to be emitted to both indoor and outdoor air leading to the involuntary exposure of other community members.

Approaches to changing environmentally significant behaviours, in particular those concerning the relationship between residential energy consumption patterns and environmental sustainability, are roughly on a continuum from those that are framed at the individual level (**agency**) to those that are framed at the social level (**structural**) (Wilson & Dowlatabadi, 2006). At the agency end of the spectrum, economic or psychological approaches view the individual as an autonomous decision maker, albeit subject to external influences (Stern, 2000) and at the other end there is the sociological perspective that individual decisions are constructed or determined by wider social and technological systems (Shrove, 2010).

In this thesis the context of burning treated timber in the household is examined through the lens of **practice theory**. Theories of practice have arisen from a sociological perspective that questions the relevance of individual decision models and instead emphasises the socially and technologically embedded nature of decision making. Practice theory encompasses a range of theoretical approaches that commonly focus on patterns of human activities as the unit of analysis as well as the role of the material world in influencing how activities are carried out. A useful description is provided by (Reckwitz, 2002):

A 'practice' (Praktik) is a routinized type of behaviour which consists of several elements, interconnected to one other: forms of bodily activities, forms of mental activities, 'things' and their use; a background knowledge in the form of understanding, know-how, states of emotion and motivational knowledge. A practice – a way of cooking, of consuming, of working, of investigating, of taking care of oneself or of others, etc. – forms so to speak a 'block' whose existence necessarily depends on the existence and specific interconnectedness of these elements, and which cannot be reduced to any one of these single elements.

Practice theory approaches have been applied within the field of consumer studies (Shove & Pantzar, 2005; Warde, 2005) and more recently within energy studies, for example standby appliance energy use (Gram-Hanssen, 2009), impacts of power cuts on heating practices (Rinkinen, 2013), smart metering and energy demand (Strengers, 2010).

In this thesis a practice theory inspired approach was used to explore the stability of, variation between households and potential for change in wood burning practice in the community. Following the approach of Gram-Hanssen (2010, 2011), who studied household regulation of indoor climate and consequent differences in energy consumption, the conceptual framework for analysis of home heating practice was based on exploring how the following four interdependent elements might work together to maintain or change a practice over time.

i) Practical understandings, embodied habits and know-how

This element describes how people have learned to do, say and appreciate certain things in a way that unconsciously becomes part of their habits. This learning stems from carrying out other practices and socialisation in the environment in which they were brought up. In the wood burning context this would encompass how people come to operate their burners – such as regulating the heat output, stoking the fire, getting rid of ash etc.

ii) Institutionalised knowledge and explicit rules

This element takes into account influence of knowledge obtained from different institutions and channels, for example, electrical appliance star efficiency ratings or information campaigns. It is stressed that there is no simple causal correlation between knowledge and practices.

iii) Engagements and meanings

This element reflects practices may have different meaning for the people who carry them out and there is an underlying goal or reason for performing the practice. For example, to save money and/or achieve a desirable level of thermal comfort.

iv) Technologies and materials

This element describes the role of technology, materials and infrastructure in configuring how practices are carried out. For example, specific design of the wood

burning appliance and all the material aspects home heating, such as dwelling layout, level of insulation, use of chainsaws, splitting axes etc.

These four elements of practice form a structure that collectively sustains a practice and at the same time these elements are sustained and developed by people who “perform” the practice (Gram-Hanssen, 2010).

5.2.3 Data collection

The constructivist philosophical stance aligns with the theoretical framework which seeks to explore peoples lived experiences of using wood for home heating and how they might come to burn treated timber or know of other people who do and what types of approaches, if any, might work to discourage the practice. Therefore, insights may be achieved by finding out how people talk about using wood for home heating, burning treated timber and air pollution.

Focus groups were used as the primary method to obtain insights into people’s home heating practices, in particular the context surrounding selection and use of wood fuels. The focus group method has three distinct features (Denscombe, 2007):

- a) the session has a specific focus, with the group discussion being centred on a phenomenon or experience familiar to the participants;
- b) interaction within the group is emphasised as a way of generating information;
- c) a moderator facilitates the group discussion and interaction.

5.2.3.1 Focus group participant recruitment

Following a constructivist approach, recruitment and selection of participants was not constrained to random sampling techniques to reduce bias, but favoured the use of pre-existing groups in order to provide data that approximates ‘natural’ conversations and everyday interactions (Kitzinger, 1995).

The research population was people who live in the ward of Wainuiomata and use wood for home heating. Wainuiomata was chosen because air quality monitoring was being concurrently undertaken in this community. Therefore, air quality monitoring

results could inform the focus group discussion. Participants were recruited purposively through approaches to not-for-profit community groups based in Wainuiomata. Approximately ten different community organisations were contacted by telephone or visited in person and those who expressed an interest were provided with an invitation letter, an information sheet and participant consent form (Appendices 8.1, 8.2, 8.3).

Attendance at the focus groups was incentivised by offering a \$100.00 donation to the community group or charity of choice for each member that participated (up to a maximum of 10 per particular community group). Incentives were thought necessary because attending and participating in a focus group requires an investment on the part of the individual giving up their personal time. The incentive served as a stimulus to attend the session, rather than opt out on the night and communicates to the participants that the focus group is important (Krueger & Casey, 2009). Furthermore, incentivising participants assists fund raising of community groups in return for people giving up their time and energy to participate in the research. It also avoids financially rewarding individuals which may bias response in some way.

Using existing community groups to recruit participants from their members and associates was seen as a cost and time-effective way of reaching a pool of potential participants. Community groups were able to use existing communication networks – e.g., through existing meetings or email mailing lists. A further advantage of this method was the potential to tap into existing groups of people who have some common ground– e.g., people from the same Church or sports organisation - and therefore simulate a degree of ‘natural’ interaction. This method also encourages participation from those on low incomes and beneficiaries – whose voices are not often heard. In this respect, the Wainuiomata Community Centre offered to find participants for two of the focus groups. The Centre runs various social programmes, e.g., budget advice, *Whanau n Arms* (free fruit and vegetable distribution scheme) and Korkiri Marae Health and Social Services and provides support workers for Whanau (families) in difficult circumstances and was able to recruit participants from their contacts and staff members. The Wainuiomata Community Centre was therefore able to provide participants that may otherwise have been difficult to recruit. This recruitment

strategy was designed to provide a wide cross-section of the community to maximise variation in participant's experiences while aiming keeping the final groups relatively homogeneous.

The following organisations agreed to provide members or associates to take part in the focus groups:

- Scouts
- Community Centre
- Kapa Haka Group
- Playcentre
- Kohanga Reo

5.2.4 Ethics and privacy

The research design methodology was designed to meet the major ethical principles outlined in the *Code of ethical conduct for research, teaching and evaluation involving human participants* (Massey University, 2010), in particular:

- respect for persons;
- minimisation of harm to participants, researchers, institutions and groups;
- informed and voluntary consent;
- respect for privacy and confidentiality;
- the avoidance of unnecessary deception;
- avoidance of conflict of interest;
- social and cultural sensitivity to the age, gender, culture, religion, social class of the participants; and
- justice.

Ethical approval for the qualitative research was obtained from the Massey University Human Ethics Committee on 19 July 2012 (HEC: Southern A, Application 12/32).

Informed and voluntary consent was obtained from each participant by providing an information sheet about the study and obtaining written approval before the focus group. Participants were given the option of withdrawing their consent at any time and this was reiterated at the focus group meeting.

Measures to protect participant privacy were a written assurance that the participant's name and the community group that they belonged to would be kept confidential and not referred to in any material that was published about the findings of the focus group. Participants were also asked to sign a non-disclosure statement to say that they would keep focus group discussions and participant's names confidential. This was reiterated during the focus group session. In reality, some of participants would know each other, i.e., those from the same community organisation, and may know other people at the focus group as Wainuiomata is a relatively small town. It could not be guaranteed that participant confidentiality would be maintained as it was not possible to ensure that participants would actually not repeat what was said in the group at some future time.

The facilitator and transcriber also signed confidentiality agreements (Appendix 8.8).

Participants were told that they would be sent transcripts of the discussion and given the opportunity to make amendments to the transcription of their spoken words.

There was a potential conflict of interest with the author being both an employee of Greater Wellington Regional Council (GWRC) and a student researcher. GWRC provided the funding for the research project – paid for the donations to community groups, for the hire of the meeting venue and for the transcribing of the digital recordings of the focus group sessions. GWRC is the regulator for discharges of contaminants to air in the region and has the ability to take enforcement action against people who burn treated timber. At each focus group meeting it was explained that the work was being undertaken by a student researcher who was also employed by GWRC and therefore the information gathered at the focus group meeting might be used in a general way to inform future education campaigns.

All electronic documents relating to the focus group research were stored on a password protected file system at GWRC. The electronic records were deleted and the physical records destroyed on 31 March 2015.

5.2.4.1 Focus group setting and timing

Four focus groups were held in August 2012, near the end of the winter wood burning season. The reason for this was to ensure participants would have had recent experience of using wood and observations about winter temperatures, their degree of comfort and possibly experience of winter air quality. Four focus group meetings were held over a two week period during August 2012 at the Wainuiomata Community Centre (Table 5.1).

Table 5.1: Focus group dates and participant numbers

Focus group	Date	Duration	Participants
1	20 th August 2012	7.30pm to 9pm	15
2	23 rd August 2012	7.30pm to 9pm	8
3	27 th August 2012	7.30pm to 9pm	8
4	29 th August 2012	9.30am to 11am	8

5.2.4.2 Focus group facilitation

The focus groups were facilitated by a professional mediator, Karen Bell (Enviro Solutions Ltd). Having a facilitator made it easier to listen more attentively to the group discussion and interaction and to take notes without the distraction of trying to run the meeting. It was also possible to record down who was speaking, to whom, to aid the identification of participants during the transcription process. In addition, particular ideas and interactions that occurred during the meeting salient to understanding the context of the burning of treated timber were noted down.

Refreshments were made available for participants throughout the meeting in case people had not had a chance to have their dinner before the meeting. Furthermore, the sharing of food to show hospitality, respect and build rapport is an important aspect of Maori culture.

5.2.4.3 Focus group discussion format

The focus groups were run using a semi-structured format. A list of grouped and ordered questions was prepared ahead of time by the author and used by the facilitator to guide discussion (Appendix 8.4).

Before discussions commenced the facilitator proposed a set of ground rules for how the discussions would be conducted (Appendix 8.5). Participants were invited to comment and to suggest other ground rules. At the beginning of each session as an ice-breaker the participants were asked to introduce themselves and their community group and tell the group how long they had lived in Wainuiomata and how they heated their homes. Participants were then asked to explain to the group in more depth about their individual and family wood burning practices from sourcing wood through to how they disposed of their ash. Participants were also asked to talk about what they saw as the pros and cons of using wood for home heating.

As it was not known beforehand the degree of awareness of the issue of burning treated timber the focus group began with a general look at wood burning and home heating before becoming more specific about treated timber use. Accordingly a large amount of ancillary information was obtained which may or may not be directly relevant to the key research questions but would provide context.

As a stimulus for discussion participants were shown samples of wood – both treated, untreated timber and natural wood (pine and gum) and asked how they could tell which wood was treated and which wood was untreated. Participants' views on air quality were sought as well as their opinions on the most effective way of educating others not to use treated timber on their fires. The facilitator was provided with a list of potential questions that could be used depending on feedback received from the group.

All participants were asked to fill in a demographic questionnaire (Appendix 8.6). This questionnaire was anonymous and was for the purpose of determining broad similarities or differences between groups and the general population bearing in mind that the purpose of the research method was not to obtain a representative sample.

5.2.5 Data analysis

5.2.5.1 Transcriptions of audio recordings

Transcription is the process of transforming an audio recording to text media. Data in text form are much easier to analyse than sound recordings - although it is recognised that during transcribing not everything is captured and a 'selective' account of the focus group discussion and interaction is produced (Davidson, 2009). As a way of recording some of the unspoken content of the focus group, e.g., general atmosphere, demeanour of the participants and nature of some of the interaction, field notes were taken during the meeting and after each meeting impressions of what happened in each session and what had been learnt were written down as memos.

The digital recordings of focus groups 1, 2 and 3 were transcribed by Mary Hart (Validatis Research Ltd). An "intelligent verbatim" transcription method was used in which all words were transcribed except for short pauses (less than 5 seconds), "ums" and "ers", and repeated words. The final transcriptions identified statements made by the facilitator and myself. Contributions from participants were identified by their initials and those that could not be identified, as male or female. Not all the spoken words could be transcribed as some words were unintelligible or involved people talking over the top of each other, therefore the transcription process did not capture the entire discussion. The transcription does however indicate where laughter occurred as people were speaking.

The transcriptions prepared by the professional transcriber were checked against the original recordings for accuracy. Consequently some minor changes were made to the transcriptions, where spoken words had been incorrectly transcribed. Transcriptions were sent to focus group participants who were asked to read the transcript and make any changes or corrections to the record of their spoken words. Only one participant requested that a change be made. Participants were asked to sign a release form (Appendix 8.7) for use of their transcription.

5.2.5.2 Coding of verbal transcriptions

A systematic approach to analysing the transcripts was undertaken using a coding process that assigned an identifying label to each segment of text in the focus group transcriptions. A coding scheme was developed based on all the focus group transcript text data, including notes taken at Focus Group 4. The final coding typology is shown in Table 5.2 Some of these code categories generated from the *data corpus* were particularly salient to the research objectives (Section 1.2.2) concerning CCA-treated timber. This is an artefact of the semi-structured nature of the focus group discussions which ensured that data relating to the research objectives was collected. The final coding structure was arranged to reflect the elements of practice framework inspired by practice theory as outlined in Section 5.2.2.

5.2.5.3 Thematic analysis

The transcriptions of the focus group discussions were analysed thematically. Thematic analysis is a method for identifying, analysing and reporting patterns or themes observed in the group discussion text (Braun & Clarke, 2006). A theme captures something important about the data in relation to the research question and represents some level of meaning within the data set determined by researcher judgement. It is not a summary of topics of discussion. Thematic analysis conducted within a constructivist framework does not focus on individual motivations or psychologies, but instead seeks to theorise the socio-cultural contexts, and structural conditions that enable the individual accounts that are provided (Braun & Clarke, 2006).

5.3 Results

5.3.1 Focus group demographics

Some demographic details and methods of home heating used provided by focus group attendees are summarised in Table 5.3. Maori who make up 28 per cent of Wainuiomata residents were well represented in focus groups 1 and 4. Overall there was a bias towards female participants who made up 70% of the attendees.

Table 5.2: Coding typology for focus group transcripts

Category or node	Code(s)	Description
Context of home heating (Settings)	Appliance/fuel types	Fuel type (gas, electricity, solid fuels) and appliance type (heat pump, wood burner etc.)
	Changes	Switching methods (e.g. from gas to wood burning)
Practice surrounding home heating with wood	Selection , collection & handling	Types of wood used and where and how obtained
	Using the fire	Circumstances surrounding the use of the fire, who prepares, when, how often etc.
	Ash disposal	Handling and fate of ash
	Chimney care	Frequency of chimney sweeping etc.
Meanings associated with home heating using wood	Satisfaction with heating	Are people warm enough? (level of thermal comfort)
	Pros of using wood	Perceived advantages
	Cons of using wood	Perceived disadvantages
	Health concerns	Attributed to heating method or lack of heat
	Environment and air quality	Indoor, outdoor and climate
Knowledge & awareness of treated timber use and impacts	Differentiation of wood types	Knowing the difference between wood types
	Impacts of burning treated timber	Environmental effects, health effects, effects on flue/firebox
	Interventions that might work	Ways to influence people not to use treated timber

5.3.2 Themes identified from focus group transcripts

5.3.2.1 Different practices for warming the home and acquiring fuel

Following introductions, the focus group participants were asked to tell the group in turn how exactly they heated their homes and to describe their practices around sourcing and using wood for fuel and disposal of ash. Participants shared information

Table 5.3: Focus group participant demographics

	Total	Gp 1#	Gp2	Gp 3	Gp 4
Participants	39	15	8	8	8
Gender					
Male	10	2	3	3	2
Female	26	10	5	5	6
Age range					
<20	2	2			1
20-30	6	3	1	1	1
31-40	7	2	1	1	3
41-50	9		4	4	1
51-60	7	3	1	2	1
61-70	1				1
Ethnicity					
NZ European	10		5	5	
Maori	16	7	1	1	7
NZ European/Maori	1				1
NZ European/Maori/Samoan					
Maori/Moriori	1	1			
Samoan/Dutch	1			1	
Cook Is Maori	2	2			
Home heating methods					
Electricity	2	2		1	
Bar heater	3	3			
Oil column	3			1	2
Heat pump	3	3		1	2
Fan heater	2	1	1		2
Log burner	20	3	7	7	3
Open fire	2	2		1	2
Pellet burner					
Gas bottle	3	3			
Gas mains	5	1	1	2	1
Household situation					
Three generations	1	1			
Two adults	1	1			1
Family with school children	14	4	4	2	4
Family with adult children	10	1	2	5	2
Boarder	2	2		1	
Flatting	1				1
Live alone	1		1		

#15 participants but only 12 forms handed in

with the group as individuals through “telling their stories” rather than interacting with each other to generate fresh insights.

Participants reported using a variety of types of heating methods and appliances with only one person reporting being solely dependent on wood for home heating. Time patterns for using fires for home heating versus other methods (e.g., heat pump or

gas) largely depended on how cold it was, family occupation patterns (e.g., at home or out during the day) and the influence of home layout (e.g., single storey or double storey) on heat penetration throughout the house.

When asked about what they found to be the advantages and the disadvantages of wood heating, many participants spoke about the feeling of “*cosiness*” created by the fire. For a couple of participants the fire had meaning for sustaining family connections “*it creates a sense of Whanau*” and for another re-enacting and reproducing family routines around fires from childhood.

Most people who used wood for heating were convinced that the quality of heat produced was qualitatively superior to other forms of heating, such as gas or heat pumps, because they found the heat to be hotter, dryer and longer lasting. In this way fire better achieved their household thermal comfort needs as it provided more warmth where and when it was needed.

Interestingly, the cost of using wood was seen as both a positive and negative aspect. For those participants who self-collected their wood, “*it’s free*” or “*it’s cheaper*” and “*I like not paying for electricity*” was the one of the first reasons given for using wood. Those who purchased their wood were more circumspect, noting that wood was expensive and premium woods (such as macrocarpar and gum) were unaffordable. Some wood purchasers found the initial large financial outlay for buying wood at the beginning of the season, rather than pay as you go with electricity and gas, to be a major impediment. Although one participant noted that with buying wood in advance there was more certainty about future cost as opposed to receiving a large power bill in winter.

The participants who sourced their own wood frequently reported a positive feeling of self-sufficiency in that they could continue with their heating and cooking activities during power outages. One male participant said he liked the exercise and being outdoors during the felling, chopping and stacking of wood. Most wood burners commented that although keeping a fire was hard work and created mess inside the house it was “worth” the effort.

While there were a diverse range of wood burning practices, four general approaches to acquiring wood for the home heating became apparent:

(i) Buy in advance and opportunistic free top-ups

These people organised and purchased their wood before the onset of winter by the cord or trailer load from commercial firewood merchants (e.g., Ablaze), from Trade-Me or from fund-raising ventures, such as Scouts. Some participants reported topping up their purchased supplies as and when free sources came available. For example, demolition waste or materials obtained by extended family members through their occupations – e.g., garden waste (trimmed branches), construction waste (decking off-cuts), left over wood from recycled furniture – or top-up with pallets left out for collection by commercial premises. Others obtained extra wood from contacts (friends or family) from trees felled on farms.

Female FG2: *“...we’d get a cord of pine or something like that. And then we get bits on top of that as well. So we get, like for instance at the moment we’ve got an old tree we cut down a couple of years ago, just for using smaller sticks and stuff. And we actually got quite a lot this year from the college in Moohan Street that they pulled down. We got a couple of trailer-loads from there.”*

(ii) Drip-feed pay as you go for small amounts

Two participants bought their wood from the supermarket or the warehouse in small lots. One participant said that this was because of the convenience and lack of mess, the other was due to budget.

Female FG1: *“But financially I’ve been buying my wood from New World, the kindling and the wood. Because now that it’s just me I just do it when I can.”*

(iii) Free used wood material and incidental home waste

These people did not normally buy wood but used pallets that are left stacked outside commercial premises with signs stating *“free firewood – help yourself”* and so on. Other materials from around the home were also burnt such as newspapers,

cardboard boxes, egg cartons and plastic bags. This practice was described as warmth creation as opposed to waste disposal.

Facilitator: *“And do you use fire wood?”*

Female FG1: *“Oh no, we just go ring up the bro, he brings a whole truck from Mainfreight. Pallets.”*

(iv) Self-sufficient wood collectors

Another distinct group were participants that sourced and gathered their own firewood from standing or felled trees, either on their own property or from council land or from other contacts. These people did not ‘pay’ for wood but used their own time, effort and resources (often with help from friends and family) to source their firewood.

Male FG2: *“The self-sufficiency thing is definitely what I said before, on the lifestyle block we’ve got. We have actually been planting trees in anticipation for using a wood burner.”*

5.3.2.2 Ash disposal practice

Participants reported a range of practices for handling the ash from their fire. Many kept the ash in a bucket outside for varying lengths of time. Some put the ash straight outside in the garden and others waited until the ash was cool and then it was disposed of along with household rubbish. People were not always consistent with how they dealt with their ash, for example:

Female FG2: *“Well I normally put it in the bin. If I’m lazy I throw it in the garden.”*

5.3.2.3 Ability to discriminate treated from untreated timber

Having the practical competence to assess whether wood is treated is an important pre-requisite for being aware that treated timber is being burnt. In order to explore people’s awareness and understanding focus group participants were asked if they could tell the difference between treated wood and untreated wood from actual examples of the following wood types:

- untreated building timber
- natural wood (pine and gum) logs
- pine cones
- trellis
- pallet
- CCA-treated building timber offcut
- boron-treated building framing offcut

In focus group 2 and 3 viewing the wood samples generated much discussion as participants talked about the different kinds of wood. In focus group 3, the discussion was dominated by one individual who was very knowledgeable about the details of timber treatment, such as the different grades of treated timber (H3, H4, H5, H6 etc.) due to his occupational experience.

Male FG: *“...and that’s H, what’s called H4 or H5 ground retention timber, which is highly treated with arsenic. The other one you just picked up before is what they put under timber when it goes to treatment, when they package it, for the dunnage, and that would have some treatment in it. It’s not treated for any specific reason. That’s a fillet piece, which again would be treated...”*

In focus group 1, there was also one individual with a forestry background who was able to inform the other members of the group about the differences in the wood, but with much less technical detail than the person in focus group 3. Other people admitted not knowing that there was a difference.

Female FG1 *“And so funny how you all said treated or non-treated, I’ve never known that exists [laughter]...my old man he didn’t understand you know, he wasn’t a scientist...”*

Female FG2 *“I’ve heard along the lines of treated versus untreated. I wouldn’t have a clue. Not that I don’t care...I don’t know about it.”*

A small number of participants commented on visual appearance, for example: *“greenish tinge”* and one other on the weight of the wood – if it is heavy then it is likely to be a hard wood (e.g., kwila) that does not need to be treated for outside use. Others

reported that they had been taught in childhood by their parents not to use treated wood on the fire through visual cues.

Female FG1 *“When I was growing up I was always told never touch any type of wood that has different colours on how it should look”*.

A small number of participants reported that they could tell whether burning wood was treated with CCA through its smell, for example: *“if you open the [firebox] door it smells chemically”* or the colour of the flames *“like a witch fire”*.

Most of the focus group participants were unclear about how to tell the difference between treated and untreated wood, and the nuances of differences between CCA-treated wood, boron-treated wood, stained or painted wood. For some it was not something that they had ever consciously thought about. For some people, size and shape of the wood was the criteria for whether it was suitable for wood burning.

5.3.2.4 Awareness of practice of burning treated timber

None of the participants reported knowingly using treated timber on their fires. There were two participants who reported burning non-wood materials on their fires such as cardboard boxes, newspaper and plastics.

Female FG1: *“Or if I run out of wood I end up burning the plastic bags and newspaper and everything else just to keep warm.”*

One participant was adamant that he had smelt burning treated timber from some houses on many occasions whilst he was out running during the evening.

Male FG2: *“I go for a run at night, I can smell the houses that are burning it. I get about four or five areas I run past, and you smell it. It’s not necessarily the same places.”*

One other person (Female FG3) reported that treated timber was all her mother’s neighbour burnt. There were some past incidents reported by participants that showed that they had burnt waste materials that may or may not have included treated timber.

Female FG1 *"...we had just moved into this house and they had heaps of paint pails and wood and stuff like that. But I noticed that when I started burning it, this real bad odour came out. And it started affecting my asthma and I ended up in hospital."*

One focus group participant held an overtly negative view from personal experience about the practice of burning CCA-wood in the community as he himself smelled burning treated timber whilst out running:

Male FG2: *"...you instantly know that's not good, and try to get away as quick as you can."*

One other focus group participant said she could tell when treated timber is being burnt as it smells *"poisonous"* (Female FG4).

Overall there was low awareness of the practice of burning treated wood in the community, but there was supporting evidence that CCA-wood was believed to be used in the community from two participants. A possible explanation provided by one of the participants for the general lack of interest and or awareness is the role of technology. Most people use enclosed wood burners and so are physically isolated from the smell of burning CCA-wood compared to that produced from an open or outdoor fire.

Male FG2: *"With the log burners these days, they just chuck the wood in and they won't smell it because they have the door closed. In the old days you'd actually smell it when you were burning treated timber."*

Furthermore, as the odour is emitted out the chimney at night when most people are inside their homes and therefore they are not confronted with acrid smoke that might be produced by a neighbouring chimney.

No "common social understanding" about using CCA-timber on the fire was evident from the focus group conversations. Only those with embodied understanding from

past family learning or occupational expertise had strong negative perceptions about burning treated timber.

5.3.2.5 Knowledge of emissions, smoke, air pollution and health

Only two participants (one in focus group 2 and one in focus group 3) were specifically aware that arsenic was released when CCA-treated timber was burnt.

Male FG3: *“Well it’s got arsenic in and rubbish like that, in the treated timber. And it will burn out the flue quicker. Being an ex-carpenter, and working with timber, I sort of know a bit about wood.”*

Male FG2: *“One because you put in the air, all the stuff, the arsenic and all the rest of [unintelligible] goes up in the atmosphere. Secondly it destroys your fireplace and flues. Big time.”*

The link between burning treated timber and damage to the fire box and/or flue was also picked up by one other participant:

Female FG2: *“... when we bought our fire we were instructed not to use treated wood in it. [after probing] ... I think it was more looking after the fire was more his angle [rather than the emission of hazardous chemicals].”*

Again, for the majority of the participants there was little awareness that burning CCA-wood released arsenic into air and that there might be health impacts from this practice, apart from those who had institutionalised knowledge from occupational situations.

This finding is not surprising given the general lack of recognition amongst participants about the connection between wood burning and release of emissions (as smoke) and health effects. Participants had practical knowledge that chimneys in winter produced smoke and this was usually “blown away” by the wind and that the presence of smoke in the valley meant that it was or was going to be a cold and windless night. Wood smoke was generally perceived as a normal and unremarkable part of the winter environment (*“it’s just smoke...”*) configured by the temperature and topography. For

some participants wood smoke had positive meaning or engagement through the sense of smell and nostalgia, for example:

Female FG1: *"I love the smell of wood smoke".*

Female FG1: *"And for me it brings back memories and all sorts of stuff..."*

When framed as air pollution, wood smoke was described by some participants as "smog" and was a problem for others elsewhere, for example in Christchurch, Auckland and in the Hutt Valley:

Male FG2: *"So I don't think it's like Christchurch where they had a smog problem. Because we've got such a high wind level, here, it's not that bad."*

Male FG3 : *"I think when I left for work in the mornings, you know, early, then you get up to the top of the hill then you look back at the valley, it's quite clean, but when you look over the Hutt, I'd say about Naenae upwards, it's just chock-a-block."*

When asked about sources of air pollution in their community, the majority of participants reported that car fumes were the major source of air pollution in their community, for example:

Female FG1: *"...you'd go for a walk along the main roads and you can smell the exhaust..."*

Although both home heating using wood and driving are everyday activities that produce emissions to air, the emissions from motor vehicles were top of mind when talking about air pollution.

None of the participants viewed outdoor wood smoke as having a negative impact on health. There was however, one individual who articulated the role of fire lighting practice on emissions, for example:

Female FG2: *"I had an older lady living next door to us for quite some years, and I'm not sure what she burnt on her fire, but out of the flue at the top there was tar dripping down at times. And I'm sure she must have thrown her veggie scraps. And all sorts of*

stuff got stuck on there and burnt. And the smoke that came out of her fire – we actually had to, in the winter we'd have that whole side of the house, we wouldn't open a window. So that was quite an issue. But now these new people are there and they don't burn the fire in that way and it's not an issue any more. So I think the way people use their fire really makes a difference to the quality of the air, the smoke, that comes out, the emissions and things like that. And it sort of makes a real difference on how it's burnt and how it's used I think."

Another participant commented on the role of technology in reducing emissions from fires:

Female FG1: *"...it's a double burner which burns things twice, so it cuts down the emissions."*

Only one participant questioned the relationship between indoor and outdoor air in the context of burning treated timber.

Female FG2: *So is it that it's bad for you inside the house? That you're burning it. Or is it bad for outside?*

Although participants did not attribute health effects to outdoor smoke, those with existing health conditions (mainly in focus group 1) appeared to readily attribute their health problems to home heating practices based on the views (presented as 'facts') by other members of the group, for example, when discussing the use of pine as firewood:

Male FG1: *"Yeah, be very careful with your fire when it's wet because you can blow it up sort of thing."*

Female FG1: *"But it's all right when it's dry?"*

Male FG1: *"When it's dry. The only reason I know is because I used to work in a forest. So, yeah, pine. Macrocarpar, most of that's dry – don't put it in an open fire when it's wet because if you do you can get sick."*

Female FG1: *“That would explain why I got sick because we got that wood and it was wet.”*

It appears that how the indoor environment is heated (or lack of heating) is important for how people might explain their health problems. Indoors not outdoors is where health is created, for example, “Gas heating makes me feel sick” or “the heat pump clogs her [child] up” and for one participant asthma and fire use were connected as follows:

Female FG1: *“Oh, only because I’m asthmatic. My kids are asthmatic. And two of my kids are chronic asthmatics as well. So it just depends. Like with my daughter, she’s the one that’s real bad, so if I have the wood burner going she’s really good. But then if it’s too hot she’ll get asthma, and I reckon that’s the fumes from the wood really. I’m sure there’s a bit of that in there.”*

However, people made sense of health effects based on their personal exposure inside their homes as it related to their experiences. Participants appeared to be unaware of the connection between long term exposure to combustion products and premature death (in particular for heart disease sufferers) instead focusing on the what they saw as the immediate or short term consequences for respiratory health.

When the topic of long term cancer risk from arsenic exposure was introduced there was little reaction from participants. In focus group 3 the ensuing discussion was somewhat fatalistic:

Female FG3: *“If the findings were: these levels of arsenic were too high, what would be done to fix it?”*

Male FG3: *“Is it a bit like closing the gate after the horse has gone?”*

Male FG3: *“It’s a bit late for us, but it’s the young ones.”*

People could accept ‘scientifically’ that there was arsenic in air as shown by the council measurements but it didn’t necessary cause great concern because there wasn’t any

first-hand experience of health impairment or symptoms that might corroborate this 'danger'.

5.3.2.6 Supporting and discouraging – using treated timber on the fire

Although there were no first-hand experiences of burning treated timber reported, participants shared their opinions about the circumstances they thought might lead to the use of treated timber for others.

A widely held view was that people would use treated timber out of ignorance. Treated timber offcuts are seen as attractive because they are free and conveniently sized and therefore do not require further chopping. The high cost of wood was given as a reason for people topping up their supplies with any free wood that becomes available no matter what its appearance. The ready availability of waste wood from commercial premises or being given away by others as a form of disposal was also mentioned as encouraging the use of treated timber, for example:

Female FG3: And people that have renovated at home. They'll put their stuff on the front lawn with a free sign.

Male FG2: But I think that's why a lot of people burn pallets and treated wood, it's just convenient to pick up. Go to the local chippies who just chuck it in the bin and they can just take it out. And it's ready for them to chuck on the fire. They don't have to split it and cart it around. And they probably don't know the difference between treated and untreated, it's all wood to them.

Male FG2: "A lot of people can't afford to go and get pine and what-not, so they've got to take what they can get and a lot of it's off the building sites or the bins that they go to. And they probably don't like it but they've probably got no choice."

Female FG4: "Whanau pick up wood from wherever they can find it. Pallets are popular including the ones that have blue all over them."

The view was also expressed that keeping warm for some would be more important than avoiding using treated timber - even if the consequences of burning were known.

Female FG3: *“I think as times get tighter too. The economy. People are just not going to care. Their focus is: I’ve got to be warm. I’m either warm and use my fire with treated wood or my power bill goes up and I can’t afford to pay it. So I think as times become more desperate. As I said you quite often see trailer loads of pallets, and you know that they’re treated, strapped on to the trailer.”*

Many focus group participants appeared to lump together all waste wood as treated wood, including pallets. Pallets are not treated with CCA, and unless painted or stained do not present a health risk when burnt. It was not clear therefore whether motivations ascribed to people using pallets as a fire wood source would be the same as for people using CCA-treated timber. There may very well be socioeconomic reasons for using pallets, although it was noted by focus group participants that pallets are not conveniently sized for fire wood use and require the use of a skill-saw to break down and storage space. Interestingly, a lower proportion of people in the 2013 Masterton home heating survey reported using pallets compared to those who reported using decking and building offcuts, although this might reflect differences in local availability of these two wood sources.

The other key supporting factor identified was ‘ignorance’ and this could be remedied through mass media communications, education through schools and targeted campaigns such as council road shows and displays in supermarkets.

Female FG1: *Seeing it and hearing it. I think those are just the two ways you’re going to grab people. So commercials, TV, and radio. People don’t want to read but they’ll look.*

Female FG3: *The bad thing is, like, well we’re relatively young, but we only found out by chance that it was bad to burn treated wood. Nobody told us or educated us. And I think there needs to be some education out there.*

The possibility of targeting older people who read the newspaper as a way of bringing about inter-generational education was also proposed by one participant:

Female FG1: *“Wainui hard out read their newspapers and local rag or whatever. Because they want to stay informed. They would like to be informed with the knowledge. So that would be really great because most of our olds actually read it. It will remind them to remind their mokos [children] that if this looks funny, don’t touch it.”*

The place of education in schools was robustly discussed in Focus Group 2 as this was yet viewed by one participant (a teacher) as yet another pressure to be imposed on a low decile schools already inundated with responsibilities for educating children about nutrition, sun-smart etc. And in any rate *“in a community like ours children don’t have the control”* at home to influence wood choices.

An alternative way to discourage the use of treated timber was to provide affordable good quality wood through community wood schemes, for example a gum tree plot at the school or working bees on council-owned forestry land.

One participant saw an important step in discouraging the use of treated timber would be to reduce the supply and availability in the community by educating building firms and manufacturers to dispose of all their offcuts appropriately and to make a subsidy available for this.

5.4 Discussion

The focus groups revealed that Wainuiomata householders, who use wood for home heating, were convinced that this method of heating produced superior heat compared to other methods. This finding is consistent with a national study of internal household temperatures which found that households using enclosed wood burners were warmer than those using other forms of heating (Isaacs et al., 2010). Wood burning households also reported a sense of emotional wellbeing associated with cosiness, comfort and nostalgia of fire. This finding has also been reported in other studies (Hine, Marks, Nachreiner, Gifford, & Health, 2007). Financial aspects around buying wood, such as, power bills or the high up-front cost of purchasing a winter supply of wood, were also an important part of people’s engagement and meaning ascribed to using wood for home heating. Routines around wood collection (buying in

advance, self-collecting, topping up, chopping, stacking, carrying wood, fire lighting and cleaning up, depend on household circumstances such as family demographics and housing stock. Free wood, mainly pallets left outside by commercial premises, were popular and readily available. A small number of people burnt non-fire wood materials such as cardboard egg cartons. Overall, using wood for home heating was seen to require physical effort and sacrifice of convenience for the achievement of cosiness, thermal comfort and resilience.

One focus group participant's personal account of smelling CCA-wood burning outdoors was consistent with the air quality monitoring results (Chapter 3), which showed high variability in night-to-night arsenic levels that was not fully explained by the variation in measured fine particulate levels. Specifically, the participant whilst running at night observed different houses on different nights burning treated timber. If the houses burning treated timber on a particular night are close to the monitoring station, this could explain the particularly high arsenic concentration recorded at that station on that night.

There was generally low awareness of the practice of burning CCA-treated timber in the community. There were a very small number of people who actively avoided burning CCA-treated timber. In terms of practice theory, embodied know-how from people's prior family learning was one way people identified (and thereby avoided) treated timber. The other way of knowing was through occupational experience, such as being a carpenter or working in the forestry sector. Some participants were not even aware of the concept of treated versus untreated timber and the majority could not tell the difference when presented with a range of treated and untreated wood samples. Therefore, strategies that seek to inform people about the dangers of using CCA-treated timber will not be effective if people are then unable to identify treated timber during their fuel selection process.

Apart from the minority of CCA-wood burning avoiders, there was no knowledge that burning CCA-wood lead to emissions of arsenic. Arsenic is a well-known iconic poison due to its historical use as a homicidal agent (Hughes et al., 2011). Whether knowledge of arsenic emissions during burning of CCA-wood is in itself enough to discourage the

use of CCA-wood burning, should people be able to identify the wood as CCA-wood, was not specifically discussed by the focus groups. It is possible that if it were more widely known that burning CCA-wood releases arsenic then this would be sufficient to bring about social censoring of the practice. A Danish study of household heating practices describes how the social culture surrounding burning of painted and processed wood had shifted over a period of 30 years from being seen as a sound form of waste management to it now being a clear breach of social norms to use “inappropriate kinds of fuel” that produce poisonous smoke in a wood burner (Petersen, 2008).

Generally, traffic and not wood smoke was perceived as the main source of air pollution in Wainuiomata. This is at odds with GWRC’s viewpoint that is informed by the findings of a local source apportionment study, which showed that vehicle emissions contribute only a very small amount to observed particulate matter (Davy et al., 2012). Furthermore, other residential areas in the region that are similar to Wainuiomata have low levels of nitrogen dioxide (a surrogate measure for traffic air pollution) (NZTA, 2014). Participants’ views about sources of air pollution were consistent with a national environmental perceptions survey in 2013 that found most people (88%) attribute motor vehicles and transport as the main causes of degraded air quality (Hughey et al., 2013).

Statistical associations between mortality rates and estimated long term exposure to outdoor air pollution (PM₁₀) in New Zealand have been shown to be significant (Hales, Blakely, & Woodward, 2010). Although health impacts are the underpinning rationale for national standards for PM₁₀ and guidelines for arsenic in air, this knowledge has not been incorporated into the everyday context of wood burning. There has been little publicity around the issue of health, wood smoke and compliance with national standards in the Wellington region and to date, no regulation introduced in the region to control the emissions from domestic fires. Therefore the issue of wood smoke emissions and health does not form part of the public narrative, apart from the commonly held perception that Wellington’s wind blows the air pollution away. Air pollution problems are considered to be elsewhere, such as Christchurch winter smog. The ‘air pollution problem’ constructed by the scientific view of wood burning

'pressures' leading to levels of pollutants in outdoor from which harmful health effects for the community can be quantitatively predicted is quite different from the socially constructed view of wood burners as an everyday cosy, family-friendly way of providing a superior form of home heating compared to other methods. Furthermore, people who burn wood generally have positive emotional response to wood smoke and so do not consider it to be air pollution or to have negative health effects (Hine et al., 2007).

A widely held view from the focus groups was that other peoples' use of CCA-treated timber would be because they could not afford wood and would burn whatever came to hand. However, no members of the focus group had direct personal experience to support this view for burning CCA-wood. Although there was some evidence that people burnt non-firewood materials if they ran out of wood and finance was a contributing factor. A cross tabulation of Masterton home heating survey data²⁷ found no statistically significant differences in income bands between respondents who reported using decking and building offcuts and those who did not. It not clear what role economic situation actually plays as a supporting factor for use of CCA-timber but it was used by participants as a way of making sense of the behaviour and therefore provided context for their suggested approaches for discouraging its use.

Focus group findings in the light of practice theory suggest the following elements maintain the domestic burning of CCA-wood:

- Practical understandings and embodied know-how that lead to the avoidance of treated timber are only present in a minority of people who have specialist occupational learning, such as builders or wood workers.
- Knowledge that CCA-wood releases arsenic when burnt is not widely held as there has been no effective proactive local or national campaigns designed to educate the public and to discourage the burning of CCA-wood, or that establish the connection between burning and release of arsenic into the air that people breathe.

²⁷ UMR survey commissioned by GWRC for developing a domestic heating emissions inventory

- Burning of CCA-wood by others is understood as a supplementary fuel for people who cannot afford to maintain supplies of clean burning wood and for whom keeping warm is more important than environmental considerations.
- The offering of free wood waste for others to take away and use is a widespread and socially accepted practice that is unregulated.
- The technology of an enclosed wood burner means that the distinctive smell of burning CCA-wood is not detected inside the home environment where people are present.

Community options to discourage the use of CCA-wood on home heating fires are targeted educational campaigns situated in the community context, at the local supermarket or in the local paper. Television was seen by some as an effective way of reaching a wide range of people. Changing the structural (determinants) context of wood supply and demand through community wood schemes that provide affordable wood and reducing availability of CCA-wood by subsidising appropriate disposal from building sites was also seen as an important discouraging factor.

A practice theory approach for long term discouragement of the practice of burning CCA-wood could aim to foster the development of “common social understandings”, that is, the ‘right’ and ‘wrong’ ways of doing things, analogous to social norms (Strengers, 2010). In this way meanings and engagements around the use of CCA-wood need to be shifted away from a discussion around fuel poverty to that concerning potential toxicity of emissions inside the home. It has been established in a Danish study that people’s knowledge of levels and toxicity of particulate emissions from wood burning could be spread through publicly communicated narratives based on social networks and face-to-face interaction between households and professionals, such as wood-burning stove dealers and chimney sweeps (Petersen, 2008). However, for this to work in the New Zealand context the communication narrative would need to start with a broad approach about the potential for toxic emissions from burning all processed woods, due to the difficulties in specifically identifying CCA-wood.

5.5 Limitations

The findings of the social context investigation were based on purposefully selected focus groups that may not capture the diversity of social understandings and practices surrounding the use of CCA-wood in the context of home heating. There were no participants who had direct experience of burning treated timber and therefore the perspective of those carrying out this practice was not directly observed. There was also a numerical bias towards female participants in the focus groups. It is not clear how this might affect the validity of the findings.

6 Conclusions and recommendations

6.1 Conclusions

This work quantifies the level of arsenic and other co-measured variables in air to assess regulatory compliance with the health guideline and to determine the source of arsenic in air. A simple screening-level air toxics risk assessment process has been undertaken using conservative exposure assumptions in order to establish the likelihood of harm due to contact with arsenic in air and in other sources. Community perspectives and understanding of the environmental and health impacts of burning of CCA-treated timber, and the social aspects that might support or discourage this practice, have been partly determined.

Across two years, comparatively high concentrations of arsenic were observed in Wainuiomata on some winter days. Expressed as annual averages, the 95% confidence interval estimates of arsenic concentrations included the guideline value of 5.5 ng/m^3 and therefore could not be used to indicate either compliance or non-compliance with the guideline value. It is evident that compliance with annual average guidelines for $\text{PM}_{2.5}$ and PM_{10} is not sufficient to ensure compliance with the annual average guideline for arsenic. The winter average arsenic concentration in Wainuiomata could be reasonably well predicted based on knowledge of three correlated variables: daily winter $\text{PM}_{2.5}$, wind speed and temperature. Simple source attribution techniques based on correlation between arsenic and other measured physical and chemical variables supports the hypothesis that domestic wood fires are the principal source of elevated arsenic in air. Evidence from the social context investigation indicates that CCA-treated timber is being burnt in the community by some households providing corroboration for the findings of the source attribution.

Results of quantitative exposure modelling and risk assessment suggest that the potential community health impacts are not significant where exposure is limited to outdoor arsenic and to outdoor arsenic that infiltrates into the indoor environment, where “not significant” is defined as an additional lifetime cancer risk of less than 1 in 100,000 and a hazard quotient less than 1. Being exposed to second hand smoke inside the home doubles an individual’s potential arsenic exposure via the inhalation route

and marginally increases the additional lifetime cancer risk to 1.5 in 100,000. In contrast, for an active smoker, the additional cancer risk attributable to inhaled arsenic from cigarettes is about three times higher than for a non-smoker who is only exposed to arsenic in air arising from outside the home. In comparison, the estimated lifetime cancer risk for children attributable to ingesting surface arsenic residues, resulting from contact with CCA-wood structures in playgrounds, was about six times higher than the risk from arsenic in outdoor air derived from community burning of CCA-treated timber.

Individuals residing in households where CCA-treated wood is used for firewood were predicted to have the highest level of health risk. The additional arsenic released inside the home from this practice pushes the lifetime excess cancer risk up to about 7 in 100,000. Although this potential incidence is still low, these results suggest that for households where CCA-treated timber is burned, lifetime excess cancer risk is higher than the risk that would normally be deemed to be acceptable.

Of more potential concern are the non-cancer health risks, namely decreased intellectual function due to short term or winter (sub-chronic) arsenic exposure for sensitive children who live in homes where CCA-wood is burnt and the inhabitants smoke. New Zealand does not have any health guidelines for short-term exposure to arsenic apart from work place safety standards that are not applicable in this situation. Based on California EPA reference exposure levels (RELs), it is suggested that there may be some level of risk for a small number of Wainuiomata children (approximately 4%) whose predicted inhalation exposure led to hazard quotients above 1. This finding suggests that the greatest risk to children's health from inhalation of arsenic may arise from indoor exposure as a result of burning CCA-wood. It is acknowledged that this finding is preliminary as RELs are likely to be extremely conservative and there is large uncertainty in the modelled concentrations for indoor sources of arsenic. However, efforts should be made to discourage the practice of CCA-wood burning as a precautionary measure to protect against exposure to indoor sources of arsenic.

Overall, the greatest potential for acute health risk for children was found to be posed by accidental or incidental ingestion of ash arising from the burning of CCA-wood.

Acute poisoning could arise following ingestion of one fifth of a teaspoon upwards. The social investigation found that some people placed of their firebox ash outside in the garden or left it in a bucket for some time to cool before disposal. This would mean that if CCA-wood had been burnt then ash heavily contaminated with arsenic could be accessible to a toddler or young child.

In terms of the social environment, there was little awareness about the impacts on air quality of burning of CCA-treated timber and that the practice might have negative health implications. Being able to tell the difference between CCA-treated timber and non-treated timber was only possible for those with prior understanding due to occupational learning or from their family upbringing. It was a commonly held perception that either fuel poverty or ignorance could lead to the use of CCA-treated timber. However, the social investigation was inconclusive regarding the role of household's financial situation in the use of treated waste wood for firewood.

6.2 Recommendations

In keeping with the design of this research project, recommendations arising from the work can be divided into new or outstanding scientific or technical questions, and potential means of addressing risks at a social level or through implementation of policies.

i) Technical questions

The toxicity of arsenic depends on its valence state; as arsenic (III) is more toxicologically active than arsenic (V). Therefore, determining the relative proportions of arsenic (V) and arsenic (III) in particulate matter evolved, and in ash produced, when CCA-treated timber is burnt could assist with evaluating health risks due to inhalation and ingestion exposure. Synchrotron X-ray techniques could be used to determine oxidation and molecular structure of arsenic in particulate matter.

The modelled indoor arsenic concentration and subsequent exposure estimates produced by this work have a high degree of uncertainty. The actual extent of indoor exposure to arsenic arising from outdoor and indoor-derived contributions from

combustion of CCA-treated timber can only be more accurately established through indoor air monitoring and collection of data on people's activity patterns.

The actual level of health risk posed by ingestion of arsenic in CCA-wood ash and of soils, into which ash containing arsenic has been mixed, requires measurements of arsenic bioaccessibility and bioavailability in these media.

Spatial variability in arsenic across New Zealand is not fully understood. Wainuiomata was selected as a small representative wood-burning community in New Zealand. It is unlikely that the findings in this work are unique to Wainuiomata, and some previous evidence exists that burning of CCA-treated wood occurs in other New Zealand towns and cities. Future work could be carried out on this topic to better define the absolute scale of CCA-wood burning in New Zealand, and to determine whether the practice is more common in some areas than others, and determine the social and economic factors that contribute to this activity.

More research is required to establish the appropriate toxicity reference value for arsenic in air that matches the winter-time exposure characteristics of the New Zealand context.

ii) Review of policy controls

This work has demonstrated that regulatory controls to prevent burning of CCA-treated wood are either only partially effective or ineffective, because a proportion of householders continue to burn treated wood irrespective of a prohibition on the activity and there is little awareness of the potential harmfulness of emissions. The highest-risk activity associated with the burning of CCA-treated wood relates to the fate of the arsenic-enriched ash, which if improperly disposed of may pose a direct and acute hazard to children. Potential exposure and risks associated with children coming into contact with CCA wood ash far outweigh those that occur through contact with play equipment and were not considered in ERMA's review of CCA wood.

These findings suggest the need for a high-level review of the policy and regulatory framework which permits the manufacture, use and disposal of CCA-treated wood in

New Zealand, to determine where risks might be best managed. It is therefore recommended that these findings be brought to the attention of the Ministry for the Environment and New Zealand Environmental Protection Agency for their consideration in the context of their oversight of policies and regulations that permit the on-going manufacture, use and disposal of CCA-treated wood in New Zealand.

iii) Development of practical risk reduction strategies

Due to the potential for inhalation exposure to arsenic inside the home when CCA-wood is burnt and ingestion risk to children from ash, it is recommended that social marketing approach could be initially used to raise the level of awareness through a highly visible and simple campaign that is locally situated. For example, a billboard could be sited on the main road into and out of Wainuiomata that displays a large photo of treated timber offcuts with a simple message such as “Burning waste wood poisons your home’s air – stick to natural wood instead”. Such messages would need to be strongly visible so people could see what “treated timber” looks like and so they could then make an informed decision about its use.

Following any social marketing initiatives to discourage the use of CCA-wood burning in Wainuiomata, further investigation would be required to evaluate its effectiveness in raising awareness, reducing the practice and ultimately resulting in ambient air arsenic concentration reducing to background levels. Further monitoring of arsenic in outdoor air would be needed to confirm reduction in concentrations.

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8 Appendices

8.1 Focus group invitation letter



MASSEY UNIVERSITY
COLLEGE OF SCIENCES
TE WĀHANGA PŪTAIAO

Community groups

Invitation to your members to participate in focus group meeting about wood burning practices and air quality

I am a part-time student at Massey University carrying out a project to investigate air quality, wood collecting, and burning practices in Wainuiomata as part of a Master of Philosophy (Environmental Health) degree. I also work at Greater Wellington Regional Council and am responsible for regional air quality monitoring and reporting.

I would like to invite you to approach your members to see whether any of them are interested in attending a focus group meeting of 6-10 people in mid to late August 2012 to discuss their personal experiences of choosing and burning wood for home heating.

I am seeking to recruit up to 10 people from your organisation who use wood for home heating. I am contacting a number of community groups based in Wainuiomata and hope to recruit up to 40 participants in total. Focus group participants will be selected on a first come basis. The focus groups will be held in the English language and participation will require basic written and verbal skills.

Please note participation in the project is totally voluntary and you are under no obligation to accept this invitation. If you wish to be involved, could you please pass on the enclosed information sheet to your members. I am able to provide an electronic copy if you wish to use email. Please note that the name of your community group and of the participants will be kept confidential and will not be referred to in any material that is published which contains the findings of the focus groups.

In return for identifying potential focus group participants, Greater Wellington Regional Council will make a one-off financial contribution to a fundraising project your community group is carrying out. Your community group will receive a donation for each participant who takes part in a focus group meeting.

If any of your members are interested could they please contact me either by phone or email. Once a member makes contact with me I will answer any questions they might have, check their availability for one of the four focus groups dates and arrange to get their signed consent form agreeing to participate in the focus group.

If you have any questions at all please contact me on the number below.

Yours sincerely

A handwritten signature in purple ink that reads "T. Mitchell".

Tamsin Mitchell

Tel: 830 4025 or 560 4746 (after hours)
Email: tamsin.mitchell@gw.govt.nz

8.2 Focus group information sheet



MASSEY UNIVERSITY
COLLEGE OF SCIENCES
TE WĀHANGA PŪTAIAO

Focus groups information sheet

Topic: Wood burning and air quality in Wainuiomata

Introduction

My name is Tamsin Mitchell and I'm a part-time student at Massey University carrying out a project to investigate air quality, wood collecting and burning practices in Wainuiomata as part of a Master of Philosophy (Environmental Health) degree. I also work at Greater Wellington Regional Council and am responsible for regional air quality monitoring and reporting.

Project Description

The project aims to discover the factors that influence peoples' choice and use of wood for indoor fires as some kinds of wood produce more harmful contaminants when burned than others. Not-for-profit community groups have been asked to nominate up to 10 potential volunteers from their members (or associates of members) for a focus group meeting. In return, Greater Wellington Regional Council will make a financial contribution to any fundraising project the community group is carrying out. Potential focus group members need to be people who use wood for home heating and will be selected on a first come basis. Please note the focus group meetings will be held in English.

Invitation to attend focus group meeting

I would like to invite you to be part of one focus group meeting to discuss your experiences of choosing and burning wood for your home heating fire. Four focus groups, each with 6-10 participants, are planned for the following dates:

Monday 20 th August 7pm to 9pm	Thursday 23 rd August 7pm to 9pm
Monday 27 th August 7pm to 9pm	Wednesday 29 th August 9.30am to 11.30am

If you are interested in being involved in a focus group or want more information please call me on 830 4025 during workdays before 2.30 pm or on 560 4746 after hours, or leave a voice message, or email me at tamsin.mitchell@gw.govt.nz

Once you contact me I will answer any questions you may have, let you choose one of the four dates that suit you best for attending the meeting, and arrange to collect your signed consent form (see Privacy below).

How will the focus groups be run?

A professional facilitator will run the focus group and, before the meeting starts, ground rules around the meeting will be introduced and discussed and modified if required. The group discussions will be guided by a set of general topics relating to the use of wood for indoor fires and peoples' views on what wood is acceptable to burn. The discussions will be recorded digitally and written notes will be taken. The focus group meetings will be two hours long and be held in the evening in the Wainuiomata Community Centre. Refreshments will be provided. A transcript of the focus group discussions will be sent to you and you will have the opportunity to make changes to any parts of this record that relate to your personal contribution.

What will happen to the focus group results?

The focus group findings will be used to investigate the relationship between air quality and the nature/types of wood used in indoor fires. Results will be used for my student research and may also be used in the future to help develop council strategies for promoting wood burning practices that minimise health risks. All records relating to the focus group discussions and participant consent forms will be stored in a password-protected part of the Greater Wellington Regional Council document management system. All records will be destroyed upon completion of the research.

8.3 Focus group participant consent form



MASSEY UNIVERSITY
COLLEGE OF SCIENCES
TE WĀHANGA PŪTAIAO

Wood burning practices and air quality in Wainuiomata

FOCUS GROUP PARTICIPANT CONSENT FORM

I have read the Information Sheet and have had the details of the study explained to me. My questions have been answered to my satisfaction, and I understand that I may ask further questions at any time.

I agree not to discuss anything that was said in the Focus Group with anyone who did not take part in it. I agree to take part in this study under the conditions set out in the Information Sheet.

Signature: _____ Date: _____

Full Name - printed _____

Please provide your contact details:

Address	
Telephone	
Community Group	

I will attend the following meeting at the Wainuiomata Community Centre (please circle one)

- Monday 20th August – 7pm to 9pm
- Thursday 23rd August – 7pm to 9pm
- Monday 27th August – 7pm to 9pm
- Wednesday 29th August – 9.30am to 11.30am

Tē Kōwhiri
ki Pūrehuroa

Institute of Food, Nutrition and Human Health
PO Box 756, Wellington 6140, New Zealand T 04 801 5799 F 04 801 4994 www.massey.ac.nz

8.4 Focus group questions

Wood burning practices and air quality in Wainuiomata

FOCUS GROUP TOPICS & KEY QUESTIONS

1. **Attitudes**

- Are you able to heat your home so that you and your family are comfortable during winter?
- Reasons for burning wood (eg, availability/cost/heat output/aesthetics/security of supply)
- What are the advantages/disadvantages of using wood for heating?

2. **Behaviours**

- Where do you source wood? (purchased/self-collected/family/friends/church/waste eg skips, building sites/pellets)
- When do you get your wood? (as needed, the year before etc)
- Types of wood (pre-split or rounds). Species (pine/gum etc)
- What type of person in your household takes the responsibility for getting wood supplies (where, when, how)
- How is wood stored (covered, uncovered?) and for how long before used (seasoning)
- Handling – split before use? (where, when, how)
- What type of fire place (enclosed, wetback etc)
- Fire-lighting - (when and how)
- Materials used to start fire (newspaper, kindling, pine cones)
- Non-firewood material burned (eg magazines, rubbish, old furniture, off cuts)
- Do you damp down fire so it burns all night and can be re-started in the morning?
- How do you dispose of ash?
- Do you get your chimney swept? (when and how often?)

3. **Beliefs & awareness**

- How do you rate air quality in Wainuiomata? (good, poor, not an issue?) compared to elsewhere
- What are the main sources of air pollution in Wainuiomata (your neighbourhood?)
- Are you concerned about chimney smoke? (health, smell, aesthetic)
- Does wood smoke affect health or well being in your family/community?
- Who do you look to in the community to be responsible for protecting air quality?
- Can you light a fire in a way that minimises smoke?
- Do some burning materials cause more harmful air pollution? (wet wood, plastics, treated or painted timber etc)
- How can you tell the difference between 'safe' burning materials and non-safe (eg treated timber vs non-treated timber)
- Do you know about the EECA subsidies for insulation and clean heat?

8.5 Focus group ground rules

Focus Group ground rules

1. We agree to take turns speaking and not interrupt each other.
2. We agree to listen respectfully and sincerely try to understand each other.
3. We recognise that, even if we do not agree, each of us is entitled to our perspective.
4. We will point out if we feel the meeting organisers are not being impartial.
5. We agree to follow the process as led by the meeting facilitator.
6. We don't discuss specifics about what occurred in the meeting, outside of the meeting (Who said what stays in this room).
7. The official record of the discussions will be circulated to all participants who will have the opportunity to make changes to their comments.

8.6 Focus group demographic questionnaire

Focus group participant information

Male / female

Please circle

Age group

Please tick which age group you are in

- | | |
|----------------------------------|----------------------------------|
| <input type="checkbox"/> < 20 | <input type="checkbox"/> 51 – 60 |
| <input type="checkbox"/> 20 – 30 | <input type="checkbox"/> 61 – 70 |
| <input type="checkbox"/> 31 -40 | <input type="checkbox"/> 71 – 80 |
| <input type="checkbox"/> 41 – 50 | <input type="checkbox"/> 80+ |

Ethnicity

Please tick which group or groups you belong to

- | | |
|--|---|
| <input type="checkbox"/> NZ European | <input type="checkbox"/> Niuean |
| <input type="checkbox"/> Maori | <input type="checkbox"/> Chinese |
| <input type="checkbox"/> Samoan | <input type="checkbox"/> Indian |
| <input type="checkbox"/> Cook Island Maori | <input type="checkbox"/> Other (please state) _____ |
| <input type="checkbox"/> Tongan | |

Home heating methods for main living area

Please tick as many as apply

- Electricity
- bar heater
- oil column
- heat pump
- fan heater
- log burner
- open fire
- pellet burner
- gas heater - bottle
- gas heater - mains supply
- other (please state) _____

Which of the following best describes you and your household situation?

- Single person below 40 living alone
- Single person 40 or older living alone
- Young couple without children
- Family with oldest child who is school age or younger
- Family with an adult child still at home
- Couple without children at home
- Flattening together
- Boarder

8.7 Focus group transcript release form



MASSEY UNIVERSITY
COLLEGE OF SCIENCES
TE WĀHANGA PŪTAIAO

Wood burning practices and air quality in Wainuiomata

AUTHORITY FOR THE RELEASE OF TRANSCRIPTS

I confirm that I have had the opportunity to read and amend the transcript of the focus group that I participated in.

I agree that the edited transcript and extracts from this may be used in reports and publications arising from the research.

Signature: **Date:**

Full Name - printed

Signature: **Date:**

Full Name - printed

Signature: **Date:**

Full Name - printed

Signature: **Date:**

Full Name - printed

Signature: **Date:**

Full Name - printed

Te Kunenga
ki Pūrehuroa

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8.8 Focus group facilitator and transcriber confidentiality agreements

Wood burning practices and air quality in Wainuiomata
FOCUS GROUP FACILITATOR'S CONFIDENTIALITY AGREEMENT

I Karen Sarah Bell (Full Name - printed) agree to not to disclose anything discussed in the Focus Groups.

Signature: Karen Bell. Date: 23/08/2012

Wood burning practices and air quality in Wainuiomata
TRANSCRIBER'S CONFIDENTIALITY AGREEMENT

I Mary Hart (Full Name - printed) agree to transcribe the recordings provided to me.

I agree to keep confidential all the information provided to me.

I will not make any copies of the transcripts or keep any record of them, other than those required for the project.

Signature: Mary Hart Date: 24/9/12