# Real-life emissions from residential wood burning appliances in New Zealand

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# Executive Summary

A study, jointly funded by Environment Canterbury, Nelson City Council and the Ministry for the Environment (MfE) through the Ministry's Sustainable Management Fund (SMF), was conducted in 2003 and 2004 to determine real-life emissions from low emission residential wood burning appliances. The study aimed to:

- establish whether 3 g/kg is representative of real-life emissions for low emission wood burners, and if not,
- determine whether a conversion factor could be used to estimate real-life emissions for specific models of low emission wood burners using results from tests conducted under the Australian and New Zealand test standard (AS/NZS 4012/3), and to
- provide "scientifically defensible" emission data based on real-life emissions for use in air quality assessments in New Zealand (i.e. an emission factor for low emission wood burners as a class or classes of appliance).

The study was conducted in three stages. Real-life operation was simulated in a laboratory during Stage I to determine emissions under normal operating practices using merchant-supplied firewood. Stage II measured emissions from appliances installed in the field using the same prescribed firing method and fuel types used in Stage I. During Stage III, tests were conducted on the same field appliances as that used in Stage II but operated by the householder using their own firewood supply. A household survey was conducted during October 2003 to determine "typical" wood burner operation and fuel characteristics for Christchurch. Contaminants measured during Stage I included particulate matter (PM), volatile organic compounds (VOCs), poly-cyclic aromatic hydrocarbons (PAHs), formaldehyde and a range of elemental species. Only PM emissions were measured during Stages II and III.

The study provides the first field emissions for low emission wood burners in Australasia, and a base from which future studies may be developed. Although the study was initially designed to develop emission factors based on the relationships between the different test stages, and weighting these values by operator variables, relationships could not be developed. It was subsequently concluded that the Stage III emissions constituted the only emissions data from which inferences could be drawn about the performance of appliances "in-home".

However, the Stage III results were not necessarily representative of low emission wood burners as a class of appliances, and as such emission factors could not be developed. Nevertheless, the data indicate that in a "real-life" situation some appliances may well produce emissions that are substantially higher than the "real-life" emission factor commonly assumed for low emission wood burners. While further testing would be required before firm conclusions may be drawn, on the basis of this current evidence, the emission factor of 3 g/kg is likely to be an underestimate. The calculated median of 13.0 g/kg from the Stage III results, suggests that the emission factor currently used could be too low by a factor of up to 4 or 5. This value is not robust enough for use as an emission factor, but is useful for indicating the potential range of uncertainties associated with the factor used for forecasting emissions from these appliances.

The most important findings of the study were:

- Real-life emissions measured from the four low emission wood burners were substantially higher than the 3 g/kg emission factor typically used for this group of appliances.
- The AS/NZS 4012/3 test method is not indicative of real-life emissions because of the wide range of variables, behaviours and installations evident in the field. Thus, real life emissions could not be predicted from the AS/NZS 4012/3 test results.
- In order to develop a representative emission factor for low emission wood burners, a greater number of appliances, of the same model and within the same emissions class need to be tested.

The study also suggested that AS/NZS 4012/3 may not provide an adequate basis for air quality assessments of real-life emissions and thereby ensure low emissions in the field. To ensure that emission data used in air quality assessments can inform the development of strategies to achieve air quality standards, it is not sufficient to rely on AS/NZS 4012/3 test results for wood burners. Laboratory testing reflecting operational practices would provide greater assurance that appliances can be screened to achieve emission reductions in practice. Operational testing would provide greater assurance that air quality assessments based on laboratory tests are reliable.

To develop representative emission factors for wood burners, a comprehensive study, employing a similar methodology to that of Stage III, but encompassing a wider range of operational behaviours, installation and firewood variables is recommended. The use of a potentially unrepresentative emission factor could result in underestimates in air quality assessments of the reductions required to meet air quality objectives.

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# 1 Introduction

Air quality during the cooler, winter months is poor in many of New Zealand's towns and cities. The burning of wood and coal in enclosed solid fuel burners for residential heating has been identified as a key contributor to elevated contaminant concentrations in many areas (Environet Ltd, 2003a). Air quality guidelines and standards for particulate matter less than 10 microns ($PM_{10}$), fine particles ($PM_{2.5}$), and sometimes carbon monoxide (CO), are exceeded at this time of the year, as a consequence of increased emission loads and the presence of temperature inversions.

In response to community health concerns, regulatory authorities such as Environment Canterbury and Nelson City Council have introduced stringent air quality measures to target the key source of $PM_{10}$ emissions - residential heating. Issues relating to residential heating are also being addressed at a national level. A design standard has been introduced by Government to restrict the installation of residential wood burning appliances to those with particulate emissions less than 1.5 g/kg (as tested under the Australian and New Zealand Standards 4012 and 4013:1999, AS/NZS 4012/3).

The Canterbury Natural Resources Regional Plan – Air (NRRP) was notified in 2002 and at the time of writing commissioners were hearing submissions on Canterbury's Proposed Plan. The Plan includes measures that:

- reduce residential wood burner particulate emission limits from 1.5 g/kg to 1 g/kg with a thermal efficiency of more than 65% (authorised on the basis of emissions tested under AS/NZS 4012/3);
- bans the installation of residential wood burning appliances into new homes or those that do not currently use solid fuel for residential heating;
- bans the use of open fires from 2006; and
- requires the removal of solid fuel burning appliances after 15 years of use.

The measures developed for the NRRP were based on current and forecast wintertime particulate emission loads in Christchurch. Emissions were calculated for each solid fuel burning appliance by multiplying the quantity of fuel used by the emission factor[1] appropriate for that appliance. The particulate emission factors used were based on a combination of international and local emission testing data. The international data were derived from studies that measured the field performance of wood burners. That is, emissions discharged from installed wood burners under normal operating conditions. For low emission wood burners (i.e. those with authorisation particulate test emissions below 1.5 g/kg and also referred to as sub 1.5 g/kg appliances), field data were not available and an estimated emission factor of 3 g/kg was used.

The only emission data available for low emission wood burners are those provided during the authorisation process. However, the standard authorisation test is only designed to compare emissions performance between burners and does not represent emissions generated in practice. It does not include normal operating behaviours such as lighting and stoking the fire and standardised fuel is also used to eliminate the influence of fuel type, moisture, sap

[1] Emission factors are representative values that provide a measure of contaminant discharge for a specific type of activity and fuel consumption.

content and density. The 3 g/kg estimate for low emission wood burners attempts to take these variations into account. It also assumes there is a point below which improvements in authorisation test results do not correspond with real-life improvements, due to operator behaviour and fuel variations. Consequently, the 3 g/kg emission factor was applied to all appliances below 1.5 g/kg when assessing the effectiveness of measures developed for Canterbury's Proposed Plan.

The estimated value of 3 g/kg is used by Environment Canterbury to determine the contribution of low emission wood burners to current and future emission loads, and in the estimation of the number of appliances that may be sustained in an airshed. These assessments have increased in importance with the recent introduction of a National Environmental Standard (NES) for $PM_{10}$. The Standard only allows for one annual exceedence of 50 $\mu g/m^3$ (24-hour average) to be achieved by 2013. Regional authorities in non-complying areas will not be able to issue resource consents for any activity that may cause emission loads to deviate above a modelled downward trend-line.

Concerns regarding the applicability of the 3 g/kg emission factor, and the lack of actual real-life emission measurements from low emission wood burning appliances, led to the development of this study. The study, jointly funded by Environment Canterbury, Nelson City Council and the Ministry for the Environment (MfE), with in-kind contributions provided by Applied Research Services Limited (ARS Ltd) and several wood burner manufacturers, commenced in 2003 and was completed in 2004.

The study was conducted in three stages where:

- Real-life operation was simulated in a laboratory during Stage I to determine emissions under normal operating practices (using locally-sourced firewood).
- Emissions from installed appliances in Christchurch and Nelson were measured during Stage II, using the prescribed laboratory operation procedure developed during Stage I. A household survey was also conducted during this stage to determine typical residential wood burner operating practices and fuel variations in Christchurch.
- Emissions from the same field appliances were tested during Stage III, but the wood burners were operated normally by the householder, using their own firewood supply.

Contaminants measured during Stage I included particulate matter (PM), volatile organic compounds (VOCs), poly-cyclic aromatic hydrocarbons (PAHs) and a variety of elemental species, while only PM was measured during Stages II and III.

The study endeavoured to use the results to develop real-life emission factors for low emission wood burners. While the primary focus of this study was on these particular appliances, some testing of non-complying burners (i.e. with AS/NZS 4012/3 test emissions greater than 1.5 g/kg) was also conducted. As low emission wood burners are sold nationwide, results may also be applicable to other regions.

## 1.1 Study objectives

The objectives of this study are to:

- establish whether 3 g/kg is representative of real-life emissions for low emission wood burners, and if not,
- determine whether a conversion factor could be used to estimate real-life emissions for specific models of low emission wood burners using results from tests conducted under AS/NZS 4012/3, and to
- provide "scientifically defensible" emission data based on real-life emissions for use in air quality assessments in New Zealand (i.e. an emission factor for low emission wood burners as a class or classes of appliance).

This report provides an overview of residential heating emissions and regulation in New Zealand, presents the findings of this study, and discusses the degree to which the study objectives were achieved.

# 2 The residential wood combustion issue

Residential heating has been identified as an important source of $PM_{10}$, $PM_{2.5}$, and CO in many of New Zealand's towns and cities (Environet Ltd, 2003a). Other contaminants, including those classified as air toxics, are also discharged from residential wood heating. Therefore, it is important to ensure that emission estimates are robust and that residential wood combustion and its associated impacts on air quality are fully understood. This section provides background to residential heating issues in New Zealand, with a particular emphasis on Christchurch. It describes what is known about wood smoke emissions, composition and combustion processes, and discusses changes in the residential heating sector, the introduction of emission controls, and issues associated with the quantification of solid fuel burner emissions. Issues associated with coal combustion will not be discussed as the study focuses on residential wood burning appliances.

## 2.1 Wood smoke emission characteristics and influences

### 2.1.1 Primary chemical constituents and indicator species

Residential wood burners discharge a wide range of contaminants which vary depending on the type of appliance, burn rates, fuel moisture content, and the types of firewood used. The contaminants discharged under particular conditions and fuels are well documented in the literature and will not be discussed in detail (e.g. Fine *et al.*, 2002; Rogge *et al.*, 1998). Rather, this section will be limited to providing a brief overview of the main contaminants discharged during residential wood combustion and a short description of species that may be used as indicators/markers of residential heating for source apportionment purposes.

Emissions discharged from residential heating appliances during the combustion of wood, comprise both gases and particles. These include PM in the $PM_{10}$ and $PM_{2.5}$ size fractions, and gases including CO, carbon dioxide ($CO_2$), oxides of nitrogen ($NO_x$) and sulphur oxides ($SO_x$). Particulate and CO emissions are increased under incomplete combustion conditions, while $CO_2$ (the end product of combustion) increases under optimal combustion conditions.

Of the PM derived during wood combustion (on an enclosed residential wood burner), 96% is in the $PM_{10}$ size fraction and 93% in $PM_{2.5}$ (Houck and Tiegs, 1998). These particles are comprised of many chemical compounds with the main constituents being organic and elemental carbon. Data collected during the combustion of wood on an open fire, for example, indicate that as much as 74% of the $PM_{2.5}$ discharged was organic carbon and 1-18% was elemental carbon (Fine *et al.*, 2002). Salts and minerals include combinations of sodium, magnesium, potassium, calcium, zinc, ammonium, sulphate, chloride, carbonate and nitrate, depending on firewood species used (Houck and Crouch, 2002).

The organic compounds present in wood smoke also vary depending on combustion conditions and wood characteristics such as lignin content. Emissions may include simple hydrocarbons (C1 to C7), present as gases at ambient temperatures, or semi-volatiles that transfer to the particulate phase under ambient conditions (USEPA, 1992). Detailed source profiles including a wide range of organics are provided by Rogge *et al.* (1998), McDonald *et al.* (2000) and Schauer *et al.* (2001). The main PAH species present in wood smoke, as identified by McDonald *et al.* (2000) are acenapthylene, naphthalene, anthracene, phenanthrene, benzo(a)pyrene and benzo(e)pyrene. Other compounds include a variety of aldehydes, phenols, alcohols, ketones, carboxylic acids, methane, ethane, ethanol, acetaldehyde, acrolein, phenol, cresol, formic acid and acetic acid (Houck and Crouch, 2002). Organic compounds such as formaldehyde, benzene, toluene, xylene and PAHs, including the carcinogen benzo(a)pyrene, are known air toxics.

Health-based air quality guidelines and standards have been established in New Zealand to address ambient concentrations of some of these contaminants. Regional councils are required to monitor air quality in areas where these may be exceeded. This will be discussed in a later section.

Local information regarding the chemical composition of wood smoke is required for robust source attribution studies to be conducted in New Zealand. These studies are used in tandem with emission inventories to determine daily contributions of key sources to PM concentrations. Elemental source profiles are obtained from ambient chemical measurements and identified on the basis of the presence of wood smoke "indicator" species and/or actual direct source emission tests. Common indicator species suggested include water-soluble potassium and retene. The use of these species, however, is limited as they are not unique to residential wood combustion. Other biomass combustion sources discharge potassium, and motor vehicles have also been found to discharge retene (McDonald *et al.*, 2000). Substituted phenols and resin acids may be used for distinguishing between hard- and soft-wood species, but the most useful single compound appears to be the sugar, levoglucosan (McDonald *et al.*, 2000; Schauer *et al.*, 2001; Fine *et al.*, 2002). This constitutes a reasonably large fraction of the organic species and may prove to be a promising wood smoke indicator species. It is recommended that, if practical, future source apportionment studies analyse concentrations of this species in measured PM. Local profiles are still required, however, to determine whether the indicator species identified in the literature are valid for wood smoke emissions in New Zealand.

### 2.1.2 Combustion processes and the burning cycle

The quantity and composition of emissions is influenced by combustion processes and varies at different stages of the burning cycle. Combustion efficiency is lowest, and emission production greatest, when the rate of combustion is slow and flame intensity is low. Conversely, when combustion occurs at a higher rate, and the flame is of a greater intensity, secondary combustion[2] occurs and lower emissions are generated (Pechan, 1993a). The operating conditions, in particular, the amount of air available for combustion, also substantially impacts on particle size and the amount of PAHs discharged. Hueglin *et al.* (1997) indicates that PAH generation increases as the combustion air supply decreases.

The combustion process is initiated when firewood is ignited using an external heat source. As the wood is heated, the compounds that make up wood start to hydrolyse, oxidise, dehydrate and pyrolyse. Combustible volatiles, tarry substances and highly reactive carbonaceous char begin to form (Rogge *et al.*, 1998). The highest emissions are generated during this stage as the temperatures are insufficient to ignite the volatile gases. Eventually, the volatiles and the tarry substances do ignite and flaming combustion begins. The heat generated during flaming combustion evaporates any moisture contained in the wood, and drives out of the wood the volatiles arising from the combustion of resinous compounds and

[2] Unburnt volatiles released while the fuel is burning may be combusted in a secondary combustion zone, usually located towards the top of the firebox, where fresh pre-heated secondary air is introduced.

the decomposition products of the primary wood constituents – cellulose, hemicellulose and lignin. This continues until the amount of volatiles required to maintain flaming combustion decreases, and oxidation of the reactive char (or remaining charcoal) begins. Fixed carbon mixes with oxygen at high temperatures and $CO_2$ and heat are released. The heat is sufficient to continue the oxidising process and additional volatile compounds are produced, although at lower quantities than during the earlier stages of combustion (USEPA, 1993; Rogge *et al.*, 1998). This essentially describes a single burning cycle. In reality, batch-fed wood burners are fuelled on a frequent basis and new firewood is placed onto an existing bed of burning or glowing embers. Consequently, volatiles are simultaneously released from newly-loaded firewood while charring of the previous load occurs. Emissions under these conditions are dependent on the temperature of the firebox during refuelling, the length of time it takes for flaming combustion of the volatiles to occur, and the amount of air available.

Nevertheless, over a full operational cycle (i.e. from light-up to burn-out), there are three important phases to the operation of a residential wood burner. These are the light-up phase, the steady-state or intermediate phase, and the burn-out phase. The greatest emissions occur during the light-up phase as firebox and flue temperatures are not sufficient to ignite the volatiles released during combustion, as discussed above. These emissions are discharged from the chimney and condense in the cooler air (Todd, 2003a). The steady-state phase, characterised by high temperatures, is favourable for the emission and formation of aromatic compounds (McDonald *et al.*, 2000), although PM emissions are generally low. The final stage (the burn-out phase) is also characterised by low emissions from the remaining burning char. However, this is dependent on combustion parameters such as air supply and appliance type (Hueglin *et al.*, 1997). If the appliance is reloaded and the air control immediately adjusted to low in an attempt to achieve an overnight burn, the emissions are likely to increase. In this instance, the temperature of the firebox has decreased while volatiles are still being produced by the burning fuel (also referred to as the smouldering-phase).

### 2.1.3 Firewood species and characteristics

The composition of emissions discharged during residential wood combustion is also determined by the characteristics of the wood consumed. Wood is primarily comprised of celluloses (40-50% of dry wood weight), hemicelluloses (20-30%), lignin (20-30%) and other compounds including extractives (resin) and ash (Rogge *et al.*, 1998). The celluloses are polymerised polysaccharids made up of various combinations of carbon, hydrogen and oxygen. Lignin is composed of aromatic basic units and pyrolysis of these compounds leads to the discharge of PAHs (Hueglin *et al.*, 1997). Lignin content and composition varies by wood type and as such the PAHs discharged from the combustion of different wood fuels, also varies. The types of compounds present in hardwoods (angiosperms) versus softwoods (gymnosperms) are outlined in detail in McDonald *et al.* (2000) and Rogge *et al.* (1998). The lignin content is also responsible for the minor differences in calorific value on a mass basis, where softwoods have slightly higher calorific values (typically 19.2 MJ/kg) than hardwoods (typically 18.2 MJ/kg, Miller and Young, 1993). Variations in characteristics such as resin content within a given fuel type can also influence the level of emissions discharged during combustion (*pers. comm.*, Wayne Webley, ARS Ltd).

Fuel moisture and density are important wood characteristics as these determine the heat producing ability of the firewood on a volume basis. That is, a greater weight of hardwood fuel may be placed into a given firebox than for softwoods. In addition, hardwoods such as eucalypt tend to burn more slowly and produce a greater quantity of char than softwoods (*pers. comm.*, Wayne Webley, ARS Ltd). With regard to wood moisture, the higher the moisture content of the fuel, the more energy required to evaporate it off. This decreases the combustion efficiency and the heating value of the fuel leading to increased emissions (Rogge *et al.*, 1998). If the moisture content is too low, emissions may also increase as combustion is rapid and insufficient oxygen may be available to sustain complete combustion.

## 2.2 Residential heating methods and impacts on air quality in New Zealand

Cold temperatures are experienced throughout New Zealand during the winter season and, consequently, most households use some form of residential heating. Energy types available include coal, wood, bottled liquid petroleum gas (LPG), natural gas, diesel and electricity. The New Zealand census of population and dwellings provides statistics regarding residential energy use in New Zealand. A summary of these statistics is attached at Appendix 1.[3]

In 2001, 72% of households in New Zealand used electricity, 45% used wood, 28% used bottled gas, 13% used mains gas, 9% used coal and 5% used other energy types or did not heat their home. Coal, wood and electricity use was proportionally more popular in the South Island while natural gas had greater prominence in the North Island.[4] The use of wood for heating was most common in the West Coast region (77% of households) followed by the Tasman district (71%). Overall, wood use has declined with reductions evident across New Zealand.

Household surveys, conducted throughout New Zealand, are used to estimate the numbers of different types of appliances used for residential heating. Table 2.1 presents emission inventory data for Nelson (2001) and Christchurch (2002).[5] In all cases, householders were able to nominate multiple forms of heating used in the main living area of their house. Electrical heating is not presented as these appliances do not contribute to local air quality issues in New Zealand.

In areas such as Christchurch and Nelson, where air quality guidelines/standards are regularly exceeded, residential heating has been estimated to contribute to 84% and 78% of wintertime $PM_{10}$ emissions, in Christchurch and Nelson respectively (based on the most recent published emissions inventory data; Scott and Gunatilaka, 2004; Wilton and Simpson, 2001). Residential heating emissions in Christchurch (including all heating forms) also contributed 85% of daily wintertime $PM_{2.5}$, 48% of CO, 33% of $CO_2$, 6% of $NO_x$ and 4% of $SO_x$ emissions (Scott and Gunatilaka, 2004). Contributions to emissions of contaminants such as benzene are difficult to determine in New Zealand owing to the limited availability of reliable emission factor information. It would be expected, however, that residential heating would also be a significant source of CO and $PM_{2.5}$, and an important source of benzene and benzo(a)pyrene, in many of New Zealand's urban areas.

[3] NB: The census data relate to all methods used for residential heating. That is, multiple methods could be selected. Heating methods relate to the entire house, rather than main living area, which is the case for some emission inventories (e.g. Christchurch).

[4] Natural gas is less readily available in the South Island.

[5] New Zealand-wide data have been collected but could not be presented in this report as the study findings have yet to be released nationally.

**Table 2-1 Types of fuels and residential heating appliances used on a winter's day in Christchurch (2002) and Nelson (2001)**

| Appliance type | Christchurch (2002) | | Nelson (2001) | |
|---|---|---|---|---|
| | Number | % | Number | % |
| Total open fires | 9375 | 8 | 1022 | 7 |
| Open fire (wood) | 9375 | 8 | 1022 | 7 |
| Open fire (coal) | 3860 | 3 | 149 | 1 |
| Total multi-fuel burner | 3840 | 3 | 601 | 4 |
| Multi-fuel (wood) | 3277 | 3 | 601 | 4 |
| Multi-fuel (coal) | 2518 | 2 | 253 | 2 |
| Total wood burner[1] | 40611 | 33 | 5784 | 40 |
| Pre-1990 wood burner | | | 2531 | 18 |
| Pre-1992 wood burner | 16361 | 13 | | |
| 1991-1995 wood burner | | | 1532 | 11 |
| 1992-2000 wood burner | 19737 | 16 | | |
| 1996-2000 wood burner | | | 1389 | 10 |
| Post-2000 wood burner | | | 332 | 2 |
| 2001-2002 wood burner | 4515 | 4 | | |
| Pellet burner | 563 | 0.5 | | |
| Gas burner | 36800 | 30 | 3729 | 26 |
| Unflued gas | 25464 | 21 | 3682 | 26 |
| Flued gas | 11336 | 9 | 66 | 0 |
| Oil burner | 1832 | 1 | 19 | 0 |

[1]The wood burner categories listed refer to appliances installed within the years nominated. Different categories were used for the Nelson and Christchurch surveys.

Ambient air quality guidelines and standards have been established nationally to provide a minimum level of protection for all New Zealanders. These are outlined in Table 2.2. The guidelines/standards are applicable in areas where people are exposed to contaminants over the relevant averaging period. All regional councils are required by the Resource Management Act (1991) to control the discharge of contaminants to air and to collect information about the state of the environment. Although it is not mandatory for regulatory authorities to monitor all these contaminants, it is in the best interests of the community to at least monitor those that may be potential issues.

**Table 2-2 Ambient air quality guidelines and National Environmental Standards (NES) in New Zealand**

| Contaminant | Status | Value | Averaging time | Permitted exceedences |
|---|---|---|---|---|
| Carbon monoxide | NES | 10 mg/m3 | 8-hour running average | 1 per year |
| Nitrogen dioxide | NES | 200 µg/m$^3$ | 1-hour average | 9 hours per year |
| Ozone | NES | 150 µg/m$^3$ | 1-hour average | 0 |
| $PM_{10}$ | NES<br>Guideline | 50 µg/m$^3$<br>20 µg/m$^3$ | 24-hour average<br>Annual | 1 per year<br>N/A |
| $SO_2$ | NES | 350 µg/m$^3$<br>570 µg/m$^3$ | 1-hour average<br>1-hour average | 9 hours per year<br>0 |
| Benzene (at 2002)<br>Benzene (at 2010) | Guideline<br>Guideline | 10 µg/m$^3$<br>3.6 µg/m$^3$ | Annual<br>Annual | N/A<br>N/A |
| Formaldehyde | Guideline | 100 µg/m$^3$ | 30 minutes | N/A |
| Acetaldehyde | Guideline | 30 µg/m$^3$ | Annual | N/A |
| Benzo(a)pyrene | Guideline | 0.0003 µg/m$^3$ | Annual | N/A |

The previous guideline for $PM_{10}$ (same value as the NES but zero exceedences) has been exceeded in many towns and cities across New Zealand. Environet Ltd (2003b) indicates that exceedences have occurred in at least 28 urban areas. The maximum concentration measured, 310 $\mu g/m^3$, was recorded in Christchurch in 1997. The number of annual exceedences at 2003 ranged from 1 in Lower Hutt and Whakatane, to 81 in Nelson. In most cases, the exceedences occurred during the winter months under low wind speed and temperature inversion conditions.

There are few exceedences of other guidelines and standards in areas where residential heating is a major source of emissions. CO is exceeded on occasion, and the annual guideline value for benzo(a)pyrene is exceeded by an order of magnitude, in Christchurch (Wilton *et al.*, 2002; McCauley, 2005). As the range of contaminants monitored around New Zealand increases, it is possible that the guideline for benzo(a)pyrene will be exceeded in areas other than Christchurch, and contaminants such as formaldehyde and acetaldehyde may also be revealed as potential air quality issues.

The recently established NES for $PM_{10}$ has significant implications for all non-complying areas. The standard of 50 $ug/m^3$ for $PM_{10}$ (24-hour average) with one allowable exceedence annually, must be attained by 2013. The most recent assessment conducted for Christchurch, for example, suggests that approximately 42,000 wood burning appliances and open fires must be removed by 2013 to reduce emissions down to appropriate levels (Scott, 2005).

Effective air quality control strategies, therefore, are required to adequately address these issues in New Zealand. Issues relating to the energy sector are significant, and influence the decisions made by consumers when determining which heating methods to install. Uncertainty regarding electricity supply must be reduced if the numbers of householders installing heat pumps, for example, are to increase. Long-term solutions are critical to the maintenance of warm, healthy homes and the reduction of emissions from residential heating.

## 2.3 Regulation of residential heating in Christchurch and the development of technology

Residential heating was first acknowledged as an issue in Christchurch as early as 1935, when a report published by the Sunlight League of New Zealand identified domestic chimneys as a principal air pollution source (Ayrey and Kingham, 2004).

Regulation of emissions from residential heating appliances, however, did not occur until the 1970s when the Clean Air Act (established in 1972) provided for the development of Clean Air Zones. The purpose of the zones was to restrict the types of fuels used for heating and the installation of solid fuel burners (residential wood and coal burning appliances) to those that met PM emission limits (authorised appliances), and to prohibit the installation of open fires. The first zone was established in 1977 and was expanded to cover a wider area in 1984 (including Rangiora and Kaiapoi (Ayrey and Kingham, 2004)).

Since that time, Environment Canterbury, given responsibility for managing air quality in the region by the Resource Management Act, 1991, has gradually tightened the PM emission limits for solid fuel burning appliances (from 5.5 g/kg to 1 g/kg), and introduced a thermal efficiency standard of 65%. In addition, the Proposed NRRP was notified in 2002, which bans the installation of solid fuel burning appliances into new homes (or homes that do not currently use solid fuel for heating), requires the removal and replacement of solid fuel burners after 15 years of use, and bans the use of open fires from 2006. An incentives and assistance programme was also introduced (the Clean Heat Project) to encourage householders to remove older, “dirtier” appliances and change to cleaner forms of heating.

Regulation is also occurring at a national level with the recent introduction of a design standard for wood burners by MfE. This is contained within the Resource Management (National Environmental Standards Relating to Certain Air Pollutants, Dioxins and Other Toxics) Regulations and was adopted in 2004. It limits, from 2005, the installation of any new wood burning appliance in urban areas of New Zealand to those with a PM test emission of 1.5 g/kg or less, and a thermal efficiency in excess of 65% (tested in accordance with AS/NZS 4012/3).

In response to increasingly more stringent emission limits for PM, residential wood burning technology has had to adapt and improve over the years. Wood burning appliances, in the simplest sense, essentially comprise an enclosed firebox (usually with a ceramic glass door located at the front), an air setting/burn rate controller, a baffle to regulate gas flow and combustion time, and a flue for the discharge of emissions produced during combustion. The fire is lit, generally using newspaper and kindling, and when the fire is established it is refuelled in batches (i.e. batch-fed) using loads of between 5 and 15 kg, depending on size (Todd, 2003a). During combustion, air enters the front of the firebox and travels over the firewood to provide oxygen for combustion. The primary air supply, however, is insufficient to allow the semi-volatile gases released during heating to fully ignite. A secondary combustion zone, usually present in appliances that do not employ catalytic technology (such as that used in the United States), is located at the top of the primary combustion chamber to provide improved combustion. The zone allows unburnt hydrocarbons and CO to mix with fresh pre-heated secondary air, to enhance combustion. The rate of pyrolysis is determined by the primary air supply and the efficiency of combustion by the secondary air supply zone (Gilmour and Walker; 1995).

Initially, technological improvements focussed on the primary air supply and the degree to which this could be turned down. On most modern appliances, the primary air supply can no longer be fully shut down. In more recent years, changes include:

- tamper-proofing the air supply control
- pre-heating combustion air to prevent quenching
- insulating the combustion chamber (e.g. with firebricks) to maintain firebox temperatures and prevent quenching on cool firebox surfaces
- maximising turbulence and mixing of gases and combustion air through firebox design
- improving heat exchange
- maximising residence time of combustion gases in the secondary chamber by improving firebox configuration and baffle locations (Todd, 2003a).

To date, substantial improvements have been made in authorised test emissions from solid fuel burning appliances in New Zealand. Environment Canterbury’s authorised wood burner list, for example, presently includes appliances with PM test emissions as low as 0.3 g/kg and 0.6 g/hr. The extent to which emission improvements in traditional batch-fed appliances under AS/NZS 4012/3 actually translate into improved real-life emissions is, however, uncertain. As discussed in Section 1, operational and fuel characteristics in a real-life situation are likely to have substantial impacts on emissions. Other concerns associated with modern wood burners include tamperability and durability issues. Removal of firebricks, attempted modifications to air supply controls, and inappropriate fuel use leading to appliance damage have been observed in Christchurch (*pers.comm.* M. Gaudin, Environment Canterbury). Nevertheless, wood burners are popular heating appliances as the wood can be obtained freely and the appliance may be operated without relying on electricity or gas supplies.

The most significant advance in wood burning appliance technology in recent years is the development of the pellet-fuelled heater. These appliances burn densified biomass pellet fuel in a specialised firebox (Pechan, 1993b). The fuel, comprising compressed wood formed into small pellets, is loaded into a hopper and automatically fed into the fire by an auger. The combustion rate is controlled by the amount of fuel fed into the fire rather than an air supply adjustment. These appliances are clean burning, efficient and make good use of waste wood products. The pellet fuel is standardised and substantially more uniform in composition than traditional firewood. This technology, which removes the impact of operator and wood fuel variability on emissions, is particularly promising. While some increase in emissions, relative to AS/NZS 4012/3 test emissions, are likely under real-life operating conditions, it would be expected that these would be less than those evident for traditional wood burners (Environet, 2004). However, the impacts of installation variables such as flue length and induced drafts, and pellets with a higher moisture content (e.g. absorbed during storage), on emissions is presently unknown.

## 2.4 The emission testing standard

Residential solid fuel burning appliances (wood and coal) have been emission tested in accordance with standard methodology for a number of years. Up until 1999, appliances in New Zealand were tested using the methodology outlined in NZS 7403:1992 and NZS 7402:1992. In 1999, Standards Australia and Standards New Zealand released the joint standards AS/NZS 4012, 4013 and 4014. These standards apply to “batch-fed” domestic solid fuel burning appliances primarily used for residential heating, with heat output rates less than 25kW. AS/NZS 4012 specifies the method for determining power output and efficiency, AS/NZS 4013 outlines the method for determining flue gas PM emissions and AS/NZS 4014 details the requirements for the various test fuels including softwoods.

Under AS/NZS 4012/3, appliances are installed in a calorimeter room and operated in accordance with the prescribed method. Nine tests are conducted in total, with three test runs at each separate burn rate (low, medium and high). Emissions are discharged into a dilution tunnel, the heat output and efficiency calculated, and the particulate emission presented as the average of the nine runs.

The standard test method is primarily used as a means of comparing inter-burner performance under standard conditions, with a key objective being repeatability of results in the laboratory. For this reason the test method is widely acknowledged as not providing results representative of emissions under real-life operating conditions. The ability of regulatory test methods to represent inter-burner performance on a real-life basis has been raised both overseas and in New Zealand (Gilmour and Walker, 1995; Houck and Tiegs, 1998).

The following list details the main requirements of AS/NZS 4012/3:

FUEL
- Must be of a specified quality (no bark or rotted sections), moisture content (16-20% wet weight), density (0.42 to 0.52 kg/L, no splits or voids), and ash content (less than 0.5% oven dry weight).
- The test fuel has specific length requirements and must be cut to an approximate cylindrical shape and have a diameter of between 75 and 110 mm.

FIRING REGIME
- The fire is lit and a bed of burning embers established prior to starting test measurements.
- An ember bed weighing 25% of the fuel load weight must be present when fuel is loaded.
- Fuel is loaded so that there is a specific gap between pieces and the sides of the firebox.
- The test fuel load is 16.5% of the fuel chamber usable volume.
- The appliance is tested at low, medium and high burn rates with at least three tests at each rate required. The appliance is operated on high when the fuel is loaded and remains on high until 20% of the test fuel mass has been consumed.
- Tests are conducted only over one burn cycle – that is when the mass of the test fuel is consumed to within ±0.5% of the mass of the test fuel load

Important variations from real-life operation include the exclusion of the light-up phase, use of potentially unrealistic fuel loads, testing under ideal combustion conditions only, and optimisation of firewood size and quality. An expert panel convened to review residential wood combustion technology in the United States agreed that emission values obtained during certification tests only roughly predict in-home performance. It was identified that the key reasons for the variation were burning conditions and fuel characteristic differences between the certification firing method (Method 28 in this instance) and the in-home test (Houck and Tiegs, 1998). These issues become particularly important when attempting to estimate emissions from residential heating appliances, as discussed in the following section.

## 2.5 Estimation of residential heating emissions

In areas where air quality is a potential issue it is necessary to identify and quantify the main sources contributing to contaminant concentrations. This may be conducted using receptor modelling techniques or, more commonly, by emission inventory. Emission inventories are generally used to collect information on discharges for a specific location and timeframe. In New Zealand, regular wintertime exceedences of 24-hour guidelines (and standards) led to the development of emission inventories where $PM_{10}$ emissions[6] (and other contaminants) are quantified for a daily wintertime period. Residential heating information is collected by a household survey, which determines the main heating appliance types used and provides fuel use estimates. Values known as emission factors are used to estimate emissions per appliance, which are aggregated into area-based residential heating emission totals.

Emission factors are essentially representative values that provide a measure of contaminant discharge for a specific type of activity and fuel consumption. Ideally emission factors are developed from a set of direct source emission tests and represent the long-term average of all

[6] Standard emission test results are reported for Total Suspended Particulate (TSP) and referred to in this report as PM. $PM_{10}$ is a subset of PM, with 96% of PM emissions from wood burners comprising $PM_{10}$ (Houck and Tiegs, 1998).

appliances within a set category. The results need to be representative of the wide and highly variable range of appliance operating conditions and practices employed under real-life conditions. The USEPA emission factors for residential heating, for example, were derived from and include:

- in-home and laboratory tests under actual operating conditions
- a wide cross-section of open fires and wood burners
- a large number of wood species
- a variety of moisture conditions

At the present time, data used to generate emission factors in New Zealand comprise various combinations of local and overseas emissions data (including the United States (US), Canada and Australia). While an acceptable range of data is available internationally for older, higher emitting appliances, specific research on lower emission appliances (e.g. sub 1.5 g/kg) is limited. Field-based residential heating emission data are virtually non-existent in New Zealand. Testing conducted in 1999, measured PM emissions from an open fire and two solid fuel burning appliances under real-life operating conditions. It included light-up emissions and merchant-supplied firewood but used a similar firing regime to that specified in AS/NZS 4012/3 (ARS, 1999a,b)

In the absence of locally-derived emissions data, USEPA emission factors have been used for older residential heating appliances, and emissions for the lower emission appliances estimated on the basis of their AS/NZS 4012/3 test emission. For appliances with test emissions below 1.5 g/kg, an emission factor of 3 g/kg was used to account for real-life operation variables such as light-up, refuelling, and locally-sourced firewood. It was assumed, on average, that operator and wood characteristics would offset any technological improvements for appliances with test emissions below 1.5 g/kg. Although, it is intuitively expected that emissions would be greater under real-life operating conditions than in the laboratory, it is uncertain if the 3 g/kg emission factor is representative of the current range of authorised appliances, where new appliances are available with test emissions as low as 0.3 g/kg.

The emission data presented in this report will be expressed in units of grams of PM produced per kilogram of firewood consumed (g/kg) and grams produced per hour of operation (g/hr). Emission factors are usually presented in these units and occasionally as grams produced per unit heat released (g/MJ). However, the latter is not suitable for expressing emissions discharged in a real-life situation as heat output is difficult to measure accurately in the field. There are distinct advantages and disadvantages to all three units, which are discussed in detail elsewhere (e.g. Mallett and Scott, 2002; Millichamp and Wilton, 2002).

Nevertheless, it is considered that g/hr (obtained from field measurements over several hours of operation) is the most useful and simplest measure for emission inventory purposes. To calculate daily residential heating emissions, an emission factor or rate (derived from a six-hour test cycle, for example) expressed in g/hr only requires additional information regarding the numbers of hours appliances are used for emissions to be calculated. This information may be readily collected by a household survey. The hours of appliance operation are easier for a householder to estimate than the actual kilograms of firewood used per operating cycle, which is required if an emission factor expressed in g/kg is used. A g/hr estimate from a six-hour test takes into account the quantity of fuel used and the efficiency of the appliance.

Also, g/hr relates directly to the environmental effect of the emissions produced, which is the reason for management intervention in the first place.

Whilst emission factors are essential for estimating emissions it is necessary to provide a note of caution regarding the application of these values. Emission factors are designed to represent average emissions and to provide an overall indication of the magnitude of discharges from various sources. These values, when combined with averaged activity rate and fuel consumption data, can only supply an approximate measure of the average contribution of emissions from a source sector. They are not intended to be precise and should not be used as anything other than averages.

It is essential that reliable, useful emission factors are developed in New Zealand. Different fuels and appliances are available from country to country and locally derived information is critical if accurate estimates are to be derived. This study attempts to address the applicability of the 3 g/kg emission factor for the current population of sub 1.5 g/kg appliances (referred to as low emission wood burners) authorised for installation in Christchurch, and to provide actual real-life emission data.

# 3 Study methodology

PM emissions from up to six residential wood burning appliances were determined under "real-life" operating conditions.

The study consisted of three key stages. These were:

- Simulating real-life operation in a laboratory using a prescribed operating procedure (Stage I, June to November 2003).
- Determining appliance performance in the field (in-home) using a prescribed operating procedure (Stage II, September to October 2003).
- Determining appliance performance in the field (in-home) when operated as normal by the householder using their own wood fuel supply (Stage III, July to September 2004).

This section outlines the sampling programme, the instrumentation and analytical methods used, briefly describes how the data were analysed and outlines limitations and uncertainties associated with the study.

## 3.1 Sampling programme

Prior to being installed into New Zealand homes, wood burners are required to be tested under AS/NZS 4012/3 and authorised on the basis of the emission test results (Section 2.4). As this is standard practice, the AS/NZS 4012/3 testing component is not explicitly described as a separate "stage" of the study in this section. However, for the purposes of the report, these tests will be referred to as Stage 0 or AS/NZS 4012/3 test emissions.

### 3.1.1 Stage I

Aim: To simulate "real-life" operating conditions in the laboratory and to determine the impact of those conditions on PM emissions relative to AS/NZS 4012/3 (Stage 0).

Five appliances (designated A to E) were tested in the AS/NZS 4012/3 test laboratory operated by ARS Ltd in Nelson from June to November 2003, using a prescribed method of operation. Initially the intention was to devise "good" operation and "bad" operation scenarios but preliminary trials demonstrated the difficulty of defining these. Rather, the approach adopted was to run a series of tests using low, medium and high burn rates, use merchant-supplied firewood (e.g. with knots, bark, resin etc), include light-up emissions and a six hour operating period. This would allow comparisons with the corresponding AS/NZS 4012/3 emissions at low, medium and high burn rates (where applicable).

A range of other variables, and their impact on emissions was also examined using Appliances A and B. A summary of the test runs conducted is provided in Section 3.1.4, and details of the analytes tested are given in ARS (2003d). A total of 30 tests were conducted on Appliance A, 21 on Appliance B, 7 on Appliance C, and 6 on Appliances D and E.[7] The tables in Appendix 2 outline various operating parameters applicable to these tests.

[7] The focus of the study was on low emission wood burners, therefore, a greater number of tests, including a wider range of variables was conducted on Appliances A and B. Budget constraints did not allow for the full range of tests to be conducted on all appliances included in the study.

The prescribed firing method for the low burn rate runs is attached at Appendix 3, as an example. While the same method was used for all appliances, the quantities given in the following description are specific to Appliance A. Five pieces of screwed-up newspaper were placed at the bottom of Appliance A's firebox. Six pieces of kindling were randomly loaded on top of the newspaper and the appliance set to a high burn rate. The sampler was started and the fire ignited. Once a good flame was established, medium sized pieces of locally-sourced pine firewood were loaded into the firebox (2.9 kg in total). The wood was burnt down to 25% of the full load size (0.7 kg) and refuelled using 2 medium-sized logs (2.9 kg in total). This process was repeated and when re-fuelling next occurred the burn-rate was adjusted to medium, left for five minutes and then set to low. This step was repeated continuously for the remainder of the five hour re-fuelling cycle with the final load inserted at five hours. Sampling continued for another hour to encompass burn-out time.

In general, the fuel used during this component of the study was "high resin" pine (also known as "old man pine) or "medium resin" pine, as identified by the test laboratory. The fuel was selected and supplied by a major wood merchant in Christchurch. Pine fuel, representative of the most common type used in Christchurch, was originally requested but during Stage I it became apparent that "old man pine" had been supplied. Although this pine had a high resin content, and produced higher emissions than pine with a lower resin content, it was considered appropriate for testing to continue using the fuel, as it was being sold in Christchurch, and was selected as "good" firewood by the supplier. Even when the issue was raised with the supplier, and a second load of firewood supplied, the wood provided was classified as "medium resin" pine by the test laboratory. The other firewood species investigated included macrocarpa (*Cupressus macrocarpa*), oregon (*Pseudotsuga menziesii*), gum (*Eucalyptus*), and "low resin" pine (*Pinus radiata*). Other variables tested were large wood logs (1.5 – 3kg), wet wood (~27% moisture on a wet weight basis) and dry pine off-cuts (~11% moisture).

Contaminants measured were PM, VOCs, PAHs, formaldehyde and a range of elemental species. These were not sampled during all test runs – see ARS (2003d) for more details.

### 3.1.2 Stage II

Aim: To determine the impact of in-home installation on emissions relative to those measured during Stage I and AS/NZS 4012/3 (Stage 0). This was achieved by comparing emissions tested in the laboratory using "low" and "medium" air settings with the equivalent tests conducted on the same appliances as installed in the field (i.e. in-home).

Householders living in Christchurch and Nelson, using the same appliances tested in the laboratory (Appliances A to E), were invited to participate in this study. However, correct identification of the appliances was difficult and installed appliances were not always identical to those tested in the laboratory. While Appliances A and B were identical to those tested in Stage I[8], Appliances C to E were either different models and/or fitted with hot water boosters, had secondary air bars or substantially longer flues than that tested in the laboratory. These variations are detailed in Appendix 2.

[8] The flue heights vared slightly where flues were 4.6 m in the laboratory, and 4.2 m for Appliance A and 4.8 m for Appliance B in the field.

Each appliance was operated by a technician from ARS Ltd and tested between September and October 2003, using the same prescribed firing method and firewood type (where possible) as Stage I. Although it was intended that emissions from Stage I and II would be directly comparable, with the only variation (ideally) being the difference between laboratory and home (e.g. room size, ventilation characteristics etc), this was not always possible. In addition, as budgets only allowed for six tests per appliance, the tests were limited to the low and medium burn rates and PM was the only contaminant measured during this component of the study.

A household survey of residential wood burner operation and fuel characteristics was also conducted during Stage II. A total of 513 wood burning households participated in the survey with a population sampling error of 4.3%. Full details of the methodology and results are found in Lamb (2003), although some results will be presented in this report. This survey provided information regarding typical wood burner operation in Christchurch.

### 3.1.3 Stage III

Aim: To determine the impact of householder operation and fuels on emissions performance relative to Stage I, II and AS/NZS 4012/3 (Stage 0).

Measurements were conducted on Appliances A to F during July and September 2004. The householders were requested to operate the appliance as they would normally, using their own firewood supply. While the sampling was being conducted, participants were asked to record on a form the weight of fuel loaded, time of loading and any changes in burn-rate settings. Sampling was undertaken both during the day and in the evening for a total of seven operating cycles. The method of operation and the fuels used varied considerably between households as discussed in later sections.

It should be noted that permission was not granted for sampling to continue at the 2003 location for Appliance D, so an alternative Appliance D was sourced with similar installation characteristics for the 2004 measurements.

The only contaminant measured during Stage III was PM. Greater details of the testing methodology adopted during Stage III, and the parameters recorded are presented in ARS (2004a).

### 3.1.4 Test summary

Table 3.1 summarises the different tests conducted on each appliance for each stage of the testing programme.

## 3.2 Instrumentation and analytical methods

### 3.2.1 Stage I

Appliances A to E were tested in the laboratory using a dilution tunnel test rig, as illustrated in Figure 3.1 (AS/NZS 4013: 1999). The appliances were fired using a similar procedure to that attached at Appendix 3. The hot gases produced during the test are discharged into the dilution tunnel and mixed with cold ambient air prior to measurement. The particles that form when gases cool and condense in ambient air are, therefore, included in the measurement. Sampling was conducted in accordance with AS/NZS 4013 where PM is collected onto preweighed glass-fibre filters. The filters were conditioned for at least 24-hours and weighed again to determine the gravimetric mass of the sample. Emissions in g/kg and g/hr were then calculated.

VOCs, PAHs and formaldehyde emissions were determined for each appliance at a single low and medium burn rate, and at a high burn rate for Appliances A and B. Samples for VOC analysis were drawn from the dilution tunnel into sampling tubes containing a bed of graphitised carbon/carbon molecular sieves, in accordance with USEPA Method TO-17. The samples were analysed using a thermal desorption-capillary gas chromatography/mass spectroscopy analytical procedure. The PAH samples were collected in accordance with NIOSH method 5515 into a tube containing a filter and XAD-2 resin. These samples were analysed for the 16 USEPA priority PAHs and retene (a potential marker species for softwood emissions) by gas chromatography. Formaldehyde samples were collected into a tube containing silica gel coated with 2,4 – dinitrophenylhydrazine, in accordance with NIOSH method 2016. These were then analysed by high performance liquid chromatography. The sampling/analytical methods used to determine emissions of these air toxics may be found in ARS (2003a).

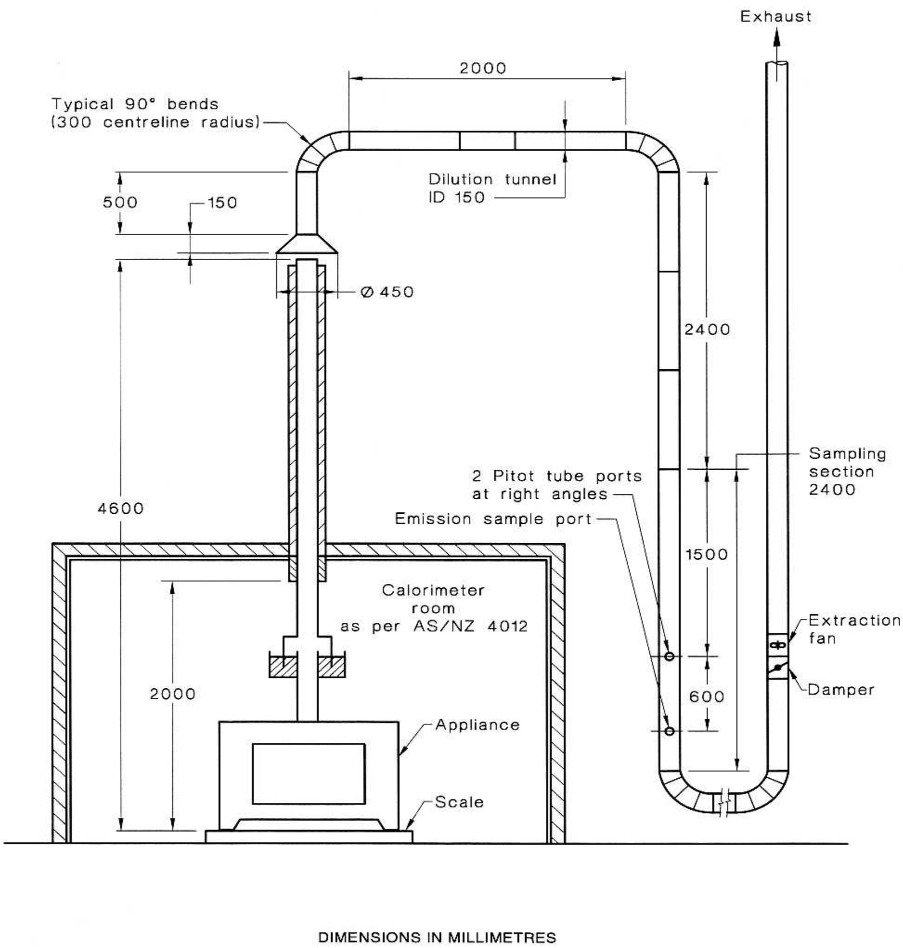


**Figure 3-1 Typical dilution tunnel facility (AS/NZS 4013:1999)**

**Table 3-1 Number of and types of fuels tested for each appliance at Stage I, II and III**

| | Stage I - Prescribed laboratory | | | Stage II - Prescribed in-home | | Stage III - householder fuel and operation | Operator survey |
|---|---|---|---|---|---|---|---|
| | Low burn rate | Medium burn rate | High burn rate | Low burn rate | Medium burn rate | Random | |
| Appliance A | High resin pine x 3<br>Low resin pine x 1<br>Oregon x 1<br>Macrocarpa x 1 | High resin pine x 3<br>Low resin pine x 1<br>Oregon x 1<br>Macrocarpa x 1<br>Gum x 3<br>Large high resin pine x 3<br>Wet high resin pine x 3<br>Dry pine offcuts x 3 | High resin pine x 3<br>Low resin pine x 1<br>Oregon x 1<br>Macrocarpa x 1 | High resin pine x 3 | High resin pine x 3 | full-cycles | Results on % households using wood of various types, sizes, operational settings etc. |
| Appliance B | High resin pine x 3 | High resin pine x 3<br>Gum x 3<br>Large high resin pine x 3<br>Wet high resin pine x 3<br>Dry pine offcuts x 3 | High resin pine x 3 | High resin pine x 3 | High resin pine x 3 | full-cycles | |
| Appliance C | Medium resin pine x 3 | Medium resin pine x 3 | Medium resin pine x 1 | Medium resin pine x 3 | Medium resin pine x 3 | full-cycles | |
| Appliance D | High resin pine x 3 | High resin pine x 3 | | Medium resin pine x 3 | Medium resin pine x 3 | full-cycles | |
| Appliance E | Medium resin pine x 3 | Medium resin pine x 3 | | Medium resin pine x 3 | Medium resin pine x 3 | full-cycles | |
| Appliance F | | | | | | full-cycles | |

PM collected from various appliances using a range of fuel types was also collected onto Teflon filters for analysis by Proton Induced X-ray Emission (PIXE). The samples were collected during Stage I test runs for Appliances B and C when pine (medium and high resin) and eucalypt firewood was used. These were taken at different stages of the operational cycle encompassing the light-up phase, and stable phase on low, medium and high burn rates. A total of 15 samples were analysed for black carbon (elemental carbon) and various chemical elements. Organic carbon, an important chemical constituent, could not be measured by this analytical technique.

### 3.2.2 Stage II and III

The equipment used for collecting emissions in the laboratory could not be used in the field. The AS/NZS 4012/3 dilution tunnel facility is not portable and is too cumbersome for an in-home application.

A review of portable samplers reported in the literature, including Oregon Method 41, the Southern Research Institute Wood Stove Dilution Sampling System and the OMNI Automated Emissions Sampler, was conducted by ARS (2003b). The method selected for this study was Oregon Method 41 (also known as the Condar Method). This method employs a dilution tube where flue gases and particles mix with ambient air prior to being collected on a series of filters.

A portable emissions sampler based on Oregon Method 41 was constructed and tested by ARS. The sampler was trialled in the laboratory, alongside the dilution tunnel system, while wood burner authorisation tests were being conducted. This allowed the two methods to be compared directly. The results of the 19 tests, presented in ARS (2003c), demonstrated a good relationship between methods with an $r^2$ value of 0.97. Field trials were also conducted to determine whether there were any practical issues that needed to be resolved prior to commencing sampling. Some blockages were experienced initially, but were eliminated by further adjustments to the sampler. Overall, the portable emissions sampler demonstrated compatibility with the AS/NZS 4012/3 dilution tunnel method and was viewed as appropriate for the in-home component of this study.

The gravimetric mass was calculated using the same method as described previously. The filters were conditioned and weighed prior to sampling, then re-conditioned and re-weighed after sampling. The gravimetric mass was then used to calculate emissions in g/kg and g/hr.

## 3.3 Data analysis

This report presents key results from Stage I, II and III relevant to the issues that impact on emissions, and the development of emission factors. The data, which were entered into Excel 2000 and Statistica 6.0 for analysis, are included at Appendix 4. Where averages were required, arithmetic means were used (unless otherwise stated). The relationship between AS/NZS 4012/3 and the various test stages was examined to determine whether or not emission factors could be developed from the data obtained during this study. The methodology for conducting these comparisons is detailed in the relevant sections.

## 3.4 Limitations/uncertainty

There are a number of limitations associated with the measurements conducted during this study. These are:

- A different sampler to that used in the laboratory was used in the home. Although trials conducted in tandem with the AS/NZS 4012/3 test rig demonstrated equivalency between methods, these measurements were made during standard authorisation tests. A similar comparison has not been made with wood burners operated using the "real-life" protocol, although, the testing laboratory indicates that equivalent results would also be expected under these conditions (*pers. comm.*, Wayne Webley, ARS Ltd).
- As with all emission testing results, there is significant level of variation inherent in the test data. Results under AS/NZS 4012/3 are obtained under closely controlled conditions and the average of nine runs represents the true average emission from that appliance (i.e. the average of many test runs). The standard deviation associated with the averaged result indicates the likelihood of the value representing the true average. ARS (1999c) estimate that the level of uncertainty (standard deviation in this instance) for the authorisation test is ±19% at the 95% confidence interval.[9] Assessment of the uncertainty of the laboratory measurements during Stage I indicated similar levels of uncertainty as the authorisation test. However, the uncertainty associated with the Stage II and III measurements could not be determined due to the low number of tests. The uncertainty is unlikely to be higher for the field-based measurements as a greater number of variables have been introduced and the conditions are less controlled. To reduce this uncertainty more test runs are required, which is not always practical or affordable.
- The ability of the study to provide representative emission factors was limited. The sample size was small (6 appliances) and while Stage III provided emissions under actual operating conditions in the field, it is uncertain how representative these emissions were of the residential wood burning population as a whole. A comparison of wood burner operation recorded during Stage III with the 2003 survey suggested that the appliances were operated differently to that of the wider population.

## 3.5 Study summary

Table 3-2 summarises key differences between the different stages. For more specific information about the appliances tested, installation characteristics and operational variables refer to Appendix 2.

[9] John Todd of Eco-Energy Options Pty Ltd also estimates the uncertainty associated with the AS/NZS 4012/3 test average to be ±20% at the 95% confidence level (*pers. comm.*John Todd, Eco-Energy Options Pty Ltd).

### Table 3-2 Study summary

| | Stage 0 | Stage I | Stage II | Stage III |
|---|---|---|---|---|
| **Brief description** | AS/NZS 4012/3 authorisation test | Prescribed method of operation employed in the laboratory to simulate real-life operating practices | Prescribed method of operation employed in the field (i.e. on installed residential appliances) | Householder operated appliance using own firewood supply |
| **Number of tests** | 9 | 6 – 9 | 6 | 7-8 |
| **Appliances tested** | A, B, C (1.2 g/kg), D, F | A, B, C (1.7 g/kg), D, E (1.3 or 1.8 g/kg)[10] | A, B, C (1.2 g/kg), D, E (1.3 or 1.8 g/kg)[10] | A, B, C (1.2 g/kg), D, E (1.3 or 1.8 g/kg)[10], F |
| **Test location** | Laboratory | Laboratory | Residence | Residence |
| **Testing equipment** | AS/NZS 4012/3 dilution tunnel facility | AS/NZS 4012/3 dilution tunnel facility | Portable emission sampler based on Oregon Method 41 | Portable emission sampler based on Oregon Method 41 |
| **Contaminants measured** | PM | PM, VOCs, PAHs and elemental species | PM | PM |
| **Run time** | One burn cycle - variable | 6 hours | 6 hours | As determined by householder |
| **Firing regime** | In accordance with AS/NZS 4012/3. Firebox already heated and ember bed established before testing. Wood loaded and sampling conducted over a single burn cycle for each burn rate. | Intended to simulate full-cycle operation. Testing is initiated at light-up, firewood loaded randomly into the firebox, and sampling ceases after six hours. Conducted in accordance with a prescribed procedure predominantly based on a low, medium or high burn rate. | Intended to simulate full-cycle operation. Testing is initiated at light-up, firewood loaded randomly into the firebox, and sampling ceases after six hours. Conducted in accordance with a prescribed procedure predominantly based on low, medium or high burn rate. | As determined by householder. |
| **Burn rate settings** | 3 low, 3 medium and 3 high | 3 low, 3 medium and three high, or 3 low and 3 medium | 3 low and 3 medium | As determined by householder |
| **Loading configuration** | Loaded with a specific gap between pieces and sides of the firebox | Random | Random | Random |

[10] The test laboratory was unable to determine the exact AS/NZS 4012/3 test emission for the Appliance E model tested during Stage I, and the Stage I appliance was different to that tested during Stage II and III.

| | Stage 0 | Stage I | Stage II | Stage III |
|---|---|---|---|---|
| **Load size** | 16.5% of the fuel chamber usable volume | approx. 8% of fuel chamber usable volume | approx. 8% of fuel chamber usable volume | As determined by householder |
| **Wood type** | Bark-free softwood with no rotted sections, and of a specified density and moisture content | Locally sourced firewood – medium and high resin pine | Locally sourced firewood – medium and high resin pine | Random – as determined by householder |
| **Wood moisture** | 16-20% wet weight | As supplied – 14-21% wet weight | As supplied – 14-20% wet weight | Random – as determined by householder |

# 4 Key contaminants and chemical constituents

This section presents the main results from Stages 0, I, II and III, with a particular focus on low emission wood burners (AS/NZS 4012/3 test emissions less than 1.5 g/kg). The ultimate aim of this study was to determine real-life emissions from this group of appliances. The results for Appliances D and E are only briefly discussed as there was uncertainty concerning the AS/NZS 4012/3 results for Appliance E during Stage I and II, and the comparability of emissions between Stages for Appliance D. This appliance was tested using high resin pine during Stage I and medium resin pine during Stage II, a hot water booster was present during Stage II but not in Stage I, and the Stage III appliance was located at a different residence to that tested during Stage II. Valid comparisons, therefore, are difficult for these appliances.

Key differences between appliances and stages are summarised in Appendix 2. Appliances A and B are directly comparable across all test stages (Stage 0, I, II and III), Appliances A, B and C across Stages 0, II and III and Appliances A, B, C and F across Stages 0 and III.

## 4.1 Particulate matter (PM)

The PM test results obtained during the study are discussed below and attached at Appendix 4.

### 4.1.1 Low emission wood burners

Figure 4-1 (a) and (b) present PM emissions for Appliance A in units of g/kg and g/hr.[11] The data provide a good indication of the magnitude of the variation in emissions generated during the different test stages.

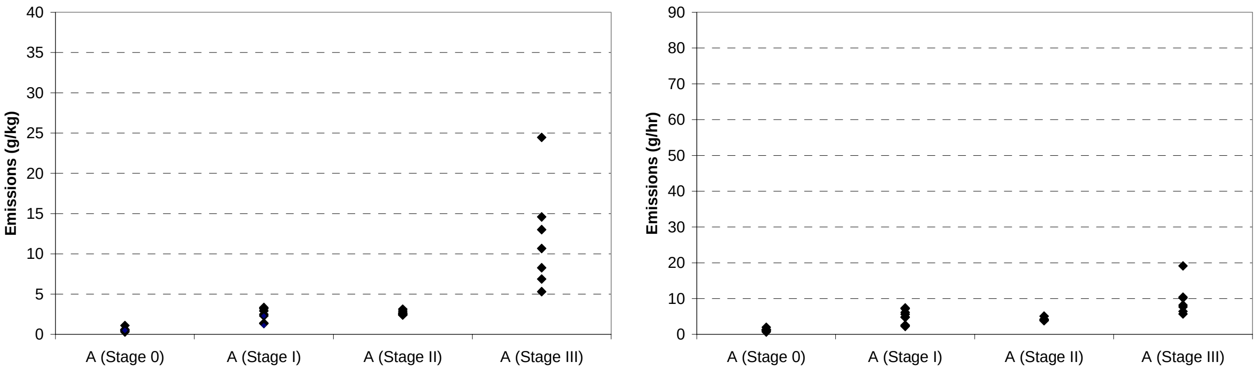


**Figure 4-1 a (left) and b (right) PM emissions for Appliance A measured during Stages 0, I, II and III in g/kg (a) and g/hr (b)**

Emissions presented in Figure 4-1 (a) were lowest during the authorisation test (Stage 0), slightly higher and more variable under simulated real-life operating conditions (Stage I), less variable under the same operating procedure applied in the field, and substantially greater and more widely distributed when the field appliance was operated by the householder (Stage III). Stage III emissions ranged from 5.3 g/kg to 24.5 g/kg, although most of the data was between 5 and 15 g/kg. The cause of the extreme value (24.5 g/kg) could not be determined from the technical data collected during the test. If this value is excluded, and the remaining

[11] Note that Stage II emissions are not directly comparable to Stages 0, I and III as high burn rates are not included.

measurements averaged and compared with averaged Stage 0 data, then emissions under real-life operating conditions were approximately 16 times higher than those discharged during the authorisation test (g/kg).

Of particular interest, is the magnitude of the difference between the Stage II and III data in Figure 4-1 (a). The key differences between tests were that during Stage II the appliance was operated using a prescribed method and fuel while during Stage III the householder operated the appliance using their own fuel. Variations in the fuel used (e.g. from high resin pine to pine off-cuts) and the use of a high burn rate during Stage III (only low and medium were tested during Stage II) were largely responsible for the differences observed.

Emissions, expressed as g/hr (Figure 4-1 b), follow a similar pattern to that illustrated in Figure 4-1 a. However, the difference between Stage II and III is not as marked as that indicated for emissions in g/kg. This is due to variations in fuel consumption rates between tests. Stage III emissions ranged from 5.7 to 19.1 g/hr with most emissions less than 10.5 g/hr. When the extreme value in Figure 4-1 (b) is excluded, emissions measured during Stage III in units of g/hr were approximately seven times greater than authorisation test emissions (g/hr).

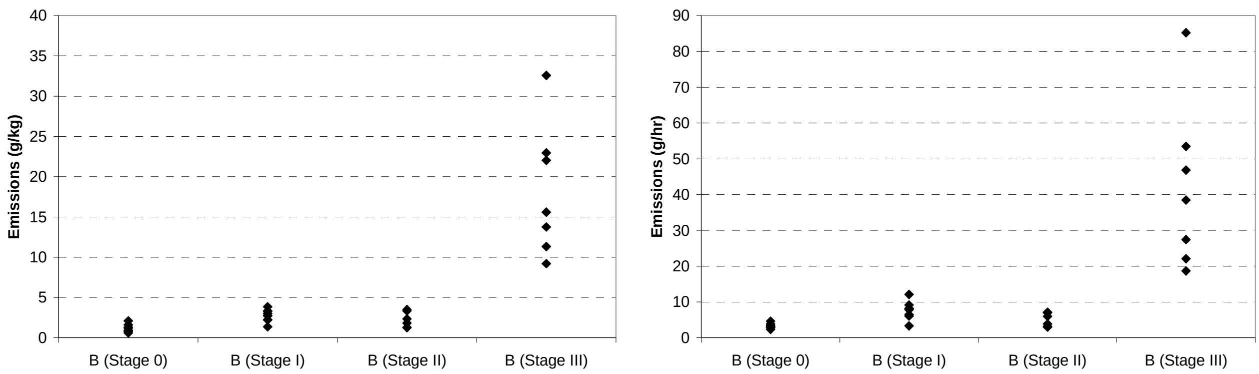


**Figure 4-2 a (left) and b (right) PM emissions for Appliance B measured during Stages 0, I, II and III in g/kg (a) and g/hr (b)**

Emissions presented in Figure 4-2 (a) and (b) for Appliance B followed a similar pattern between stages as that demonstrated by Appliance A. Emissions measured in the laboratory during Stage 0 and I, and in the field under prescribed operating conditions (Stage II) were substantially lower than those measured in the field under householder operation (Stage III). While Stage III emissions ranged from 9.2 to 32.6 g/kg and 18.7 to 85.2 g/hr, extreme values were evident. When these measurements were excluded, emissions measured during Stage III were approximately 14 times greater than authorised test emissions in g/kg and 11 times greater in g/hr.

A substantial difference in emissions (g/kg and g/hr) was also evident between Stage II and III suggesting that householder operation and fuel, again, were important influencing factors on emissions. During Stage III, high burn rates and split oregon logs with a moisture content of 19% were used. Conversely, during Stage II, tests were not conducted for high burn rates and high resin pine with a moisture content of 15% was used.

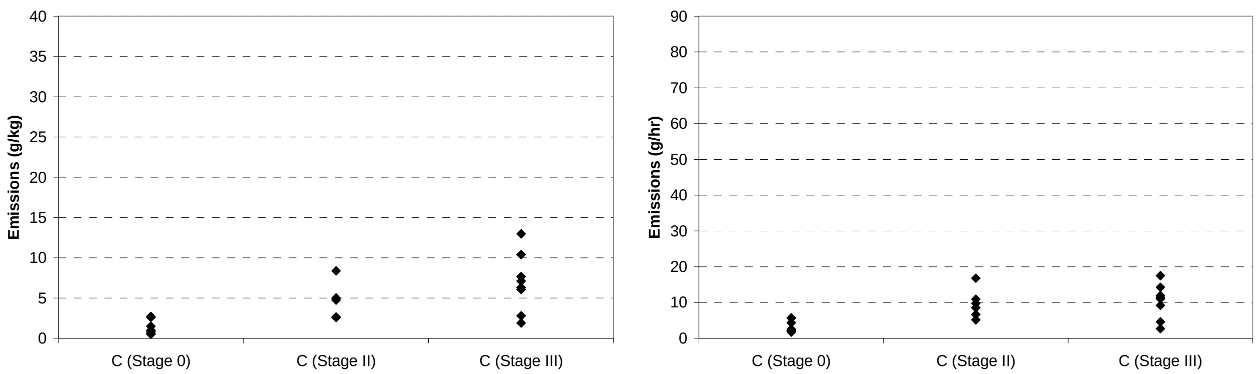


**Figure 4-3 a (left) and b (right) PM emissions for Appliance C measured during Stages 0, II and III in g/kg (a) and g/hr (b)**

The data presented in Figure 4-3 (a) and (b) are limited to Stages 0, II and III as a 1.7 g/kg Appliance C was inadvertently tested during Stage I, rather than the intended 1.2 g/kg model. The emissions increased between Stages although the magnitude of the difference between Stage 0 and III emissions was not as marked as that observed for Appliance A and B. Stage III emissions ranged from 1.9 to 13 g/kg and 2.7 and 17.5 g/hr, and were on average four times greater than authorised test emissions in g/kg and three times greater in g/hr. The householder operation stage had a substantially lower impact on emissions relative to that demonstrated by Appliances A and B.

A variety of factors may be responsible for these differences including the presence of a hot water booster and a longer flue than normal (7.5 m) during Stages II and III. The longer flue would increase the level of draught and could potentially enhance the ability of the appliance to cope with a wider range of householder operation and fuels. Alternatively, the appliance design may accommodate greater variations in operation and fuel types than that of other burners.

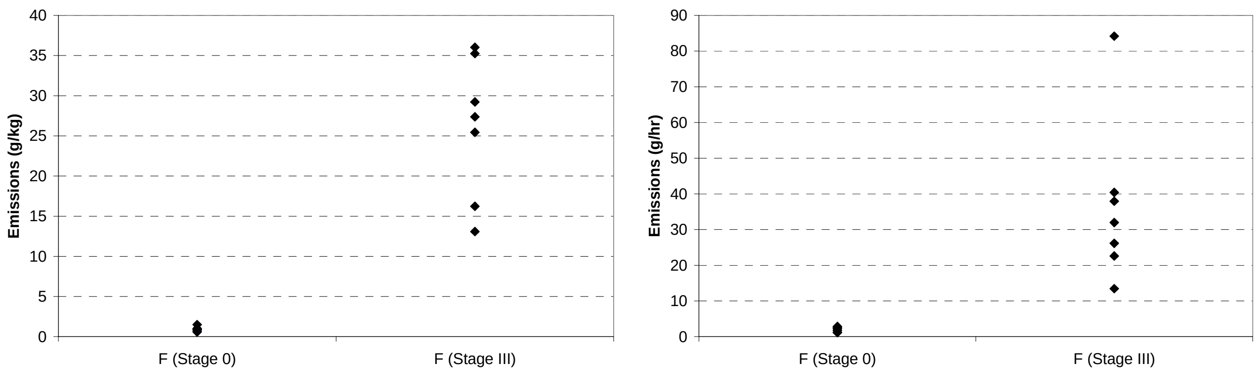


**Figure 4-4 a (left) and b (right) PM emissions for Appliance F measured during Stage 0 and III in g/kg (a) and g/hr (b)**

Emissions were only measured during Stage 0 and III for Appliance F (Figure 4-4 a and b). A substantial difference in emissions was observed between the two test stages with Stage III emissions ranging from 13.1 to 36 g/kg and 13.4 to 84.2 g/hr. The cause(s) of the extreme value present in Figure 4-4 (b) is uncertain. The use of firewood with a higher moisture content[12] and a shorter run-time (light-up and burn-down will have greater impact on emissions) than the other test runs, may be contributing factors.[13] Wood moisture was likely to be important as lower fuel consumption figures during Stage III (2.5 dry kg/hr during Stage 0 and 1.3 dry kg/hr during Stage III) suggested that a substantial amount of time was required to burn off the moisture in the firewood. This would lead to a cooler firebox and greater emissions. When the extreme value was excluded, the Stage III emissions were, on average, 27 times greater than authorised test emissions in g/kg and 15 times greater in g/hr.

### 4.1.2 Additional wood burners

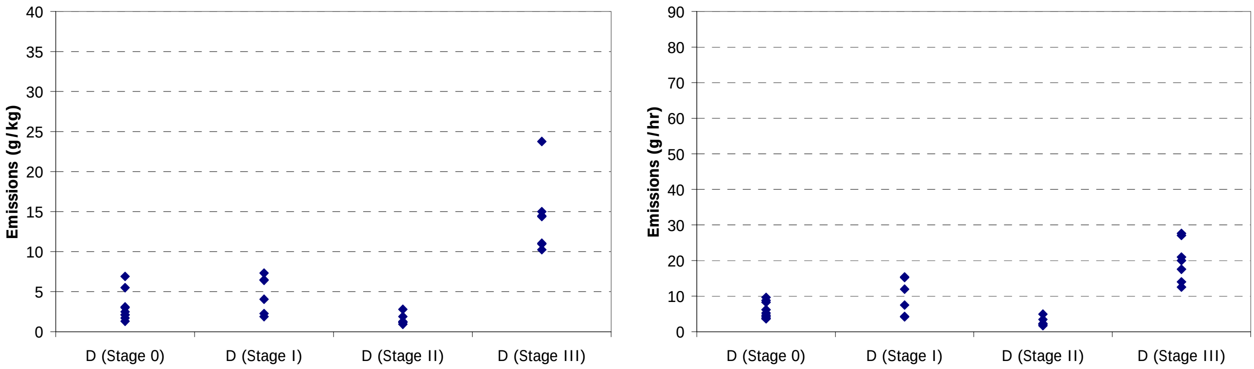


**Figure 4-5 a (left) and b (right) PM emissions for Appliance D measured during Stages 0, I, II and III in g/kg (a) and g/hr (b)**

While Appliance D was tested during all four stages, the Stage I appliance was not fitted with a hot-water booster, and the Stage III appliance was actually located in a different house to that used during Stage II (Appendix 2). Differences between Stages II and III are likely to be influenced by variations in the installation characteristics between households.

The pattern of emissions from Appliance D differed from those discharged from the other appliances tested, with Stage II emissions being lower than Stage 0. However, like all other appliances tested, emissions were greatest during Stage III. These ranged from 10.2 to 23.8 g/kg and 12.6 to 27.6 g/hr. When the extreme values were excluded, emissions under real-life operating conditions from this appliance were, on average, approximately four times higher than those discharged during the authorised test emission on a g/kg basis and three times higher on a g/hr basis.

[12] However, an average moisture content of 25% was quoted over all test runs for this appliance.
[13] The tests were conducted on this appliance fairly late in the burning season. A fresh supply of firewood was required for the tests and moderately wet wood was supplied (between 18 and 30% on a wet weight basis).

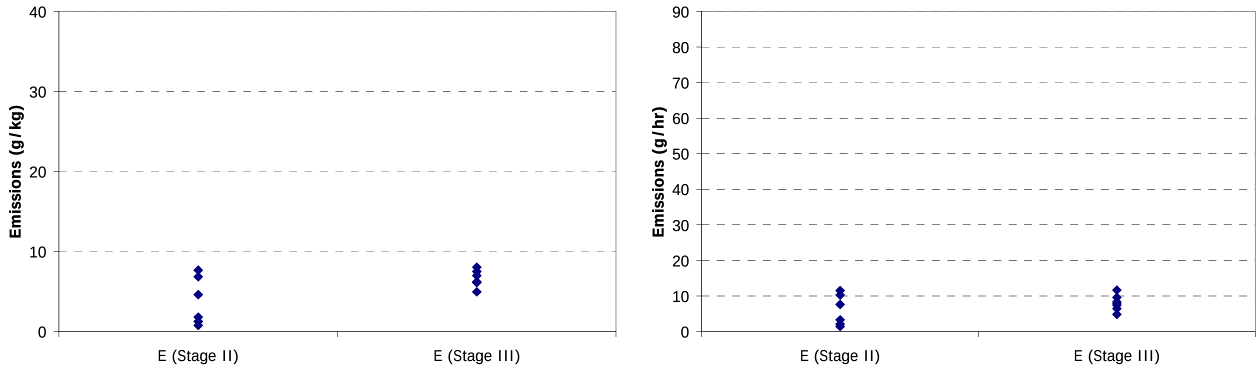


**Figure 4-6 a (left) and b (right) PM emissions for Appliance E measured during Stages II and III in g/kg (a) and g/hr (b)**

Figure 4-6 (a) and (b) presents emissions from Appliance E discharged during Stages II and III. There was considerable uncertainty regarding the Stage 0 emissions for the appliances tested and it was found that the wood burner tested during Stage I was different to that investigated during Stages II and III. Consequently, only Stage II and III emissions may be compared. However, it was uncertain whether this appliance was the 1.3 g/kg or 1.8 g/kg model. In this regard, results from these appliances are included in the "additional" rather than the "low emission wood burners" category.

The Stage III emissions ranged from 5 to 8.1 g/kg and 4.8 to 11.7 g/hr and were the lowest emissions produced by any appliance tested during this study. It was unclear from the technical data why this appliance performed so well in the field. However, it may potentially be due to the small load sizes, relatively frequent loadings and high control settings. Although comparisons between stages were limited to Stage II and III, it was possible to compare the Stage III emissions with those from other appliances (Section 4.1.3).

### 4.1.3 Variation between appliances based on average field performance (Stage III only)

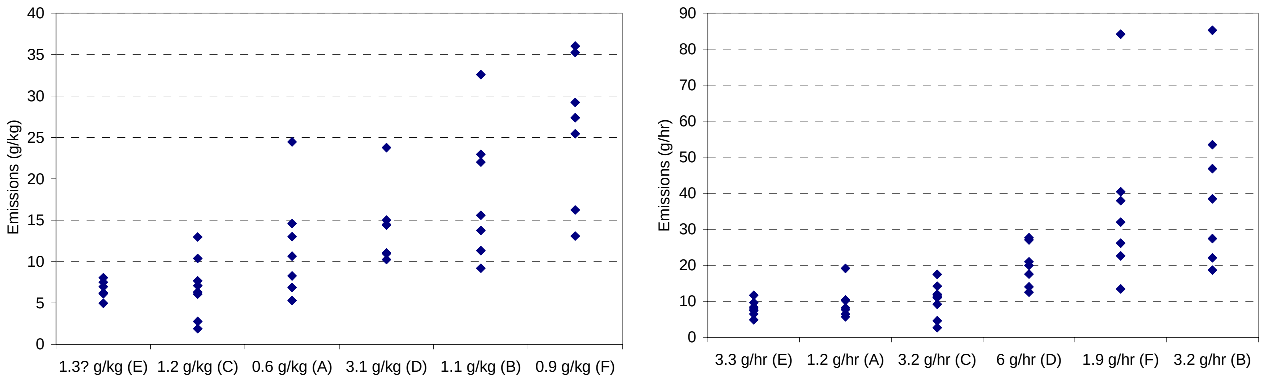


**Figure 4-7 a (left) and b (right) Emission results for Appliances A, B, C, D E and F ranked in order of average Stage III results (excluding extreme values)**

Figure 4-7 (a) and (b) present Stage III emission data for all appliances tested. The appliances are ordered from lowest to highest Stage III average emissions (excluding extremes) with the corresponding AS/NZS 4013/2 test emission (Stage 0) indicated on the x-axis.[14] These figures demonstrate that regardless of the units used to express emissions, relative performance between the standard laboratory test and home operation may be quite different.[15] This is not to say that with a larger sample size we would expect the same relative performance between the models (or emission classes) tested. However, these results, taken with the absolute differences between AS/NZS 4012/3 and "in-home" emissions performance for all burners in the study strongly suggest that when operator variables and installation characteristics are included, the AS/NZS 4012/3 test may not provide a reliable indication of the potential field performance of burners in either absolute or relative terms.

## 4.2 Air toxics

Air toxics including formaldehyde, VOCs, including benzene, and PAHs, such as benzo(a)pyrene were measured during Stage I of the study. Table 4-1 summarises key air toxics and presents the results together with data obtained from other residential wood burning appliance studies for comparative purposes. The averages of the low and medium runs, are shown, with the averages for Appliance A and B also including emissions at the high burn rate. The results presented for the other wood burner studies comprise a variety of laboratory and field data, using standard firing methods and real-life operating practices.

**Table 4-1 Key air toxic compounds found in wood smoke discharged from residential wood burning appliances (mg/kg), where n = number of runs**

| | Appliance A | Appliance B | Appliance C | Appliance D | Appliance E | Hedberg et al, 2002 | McDonald et al 2000 | Gras 2002 | Fisher et al 2000 |
|---|---|---|---|---|---|---|---|---|---|
| n | 3 | 2-3 | 2 | 2 | 2 | 4 | 8 | 53 | 30 |
| Test method | Formaldehyde - samples collected according to NIOSH method 2016 into a tube containing silica gel coated with 2,4 dinitrophenylhydrazine (DNPH). Analysed by high performance liquid chromatography (HPLC). VOCs - samples collected in accordance with USEPA method TO-17 into tubes containing graphitised carbon/carbon molecular sieves. Analysed by thermal desorption followed by gas chromatography/mass spectroscopy (GC/MS). PAHs - collected according to NIOSH method 5515 into a tube containing XAD-2 resin. Analysed by gas chromatography (see NIOSH method 5515). | | | | | ALDEHYDES: Sep-Pak DNPH-Silica cartridges, analysed by high performance liquid chromatography VOCs: BTX-monitor PAH: Munktell glass-fibre filters, PUF plugs, analysed by gas chromatography (GC-MS),Large residential heater, birch wood | PM- dilution sampler, PAH XAD/PUF cartridges/GCMS, VOCs - canisters/GCMS, PM2.5/XRF, OC/EC by TOR, 8 tests on a single appliance - possibly only one load of fuel. | Formaldehyde - DNPH trap, analysed by HPLC; VOCs - sampled into canisters, analysed by GC; PAHs - filter and PUF plug, analysed by HPLC, 53 samples from 4 appliances, AS/NZS 4012/3 firing method (ie one cycle), real and varied firewood | PAHs - XAD-2 resin cartridges, analysed by GC/MS, field tests (ie 11 installed wood burners |
| **Averaged emissions (mg/kg)** | | | | | | | | | |
| Formaldehyde | 70 | 126 | 652 | 390 | 774 | 422 | 246 | 24 | |
| Acetone | 5689 | 1236 | 1425 | 1929 | 1279 | 366 | 549 | | |
| 2-butenal | 23 | 22 | 27 | 4 | 20 | 2 | 36 | | |
| Benzene | 525 | 433 | 686 | 569 | 1094 | 1500 | 1190 | 200 | |
| Toluene | 96 | 131 | 140 | 94 | 178 | 740 | 320 | 81 | 138 |
| Xylene | 25 | 18 | 17 | 11 | 26 | 145 | 99 | 2 | 39 |
| Benzo(a)pyrene | 1 | 2 | 1 | 2 | 2 | 4 | 0.2 | 1 | 0.4 |
| Fluoranthene | 7 | 15 | 13 | 23 | 18 | 29 | 1.8 | 9 | 7 |
| Fluorene | 4 | 6 | 6 | 10 | 8 | 27 | 1.7 | 12 | |
| Phenanthrene | 27 | 38 | 33 | 66 | 48 | 99 | 7.4 | 10 | 20 |
| Anthracene | 4 | 5 | 5 | 9 | 7 | 19 | 1.4 | 2 | 4 |
| Pyrene | 6 | 10 | 10 | 18 | 14 | 26 | 1.5 | 5 | 6 |
| Retene | 5 | 26 | 4 | 4 | 6 | 0 | 0.02 | | |

[14] The Stage 0 emission for Appliance E was either 1.3 g/kg or 1.8 g/kg but, for the purposes of this assessment, it was assumed to be the 1.3 g/kg model.

[15] It is difficult to determine exactly why the appliances ranked in this manner. Multiple variables were involved including the presence of hot water boosters, and differences in the firewood, firing regimes and flue heights used. Emissions from Appliance F, for example, ranked worse than expected due to a combination of factors including the use of relatively wet firewood (25% moisture) and short burn-times (i.e. light-up and burn-down have a greater impact on the total emissions over short durations). Conversely, appliances such as Appliance C performed better than expected due to factors including a longer than normal flue, the presence of a secondary air bar, and the use of eucalypt firewood (which may be more suitable under those conditions). The best performer during Stage III was Appliance E and the technical data collected during testing provides no indication why this was the case.

Appliance A generated the highest acetone emissions at 5689 mg/kg. This value is extremely high when compared with that measured in other studies. As all appliances in this study discharged high acetone emissions, it suggests that either wood in New Zealand is substantially different to that elsewhere or, and more likely, that some contamination of the samples has occurred. Acetone is often used for cleaning sampling equipment and probes, and contamination is possible.

Appliance B generated the highest retene emissions with a discharge of 26 mg/kg. This is high relative to the other results and may be due to the inclusion of the high burn rate in the average. Emissions of 2-butenal were greatest from Appliance C (27 mg/kg) with emissions of formaldehyde also high relative to the other appliances (652 mg/kg). The formaldehyde emissions from this appliance and from Appliance E (774 mg/kg) were also high compared to measurements conducted elsewhere. Fuel characteristics may be contributing factors in this instance as these two appliances were tested using fuel from a different batch of firewood to the other appliances.

Appliance D, together with B and C discharged 2 mg/kg of benzo(a)pyrene. These values were substantially higher than those measured by Fisher *et al.* (2000) and McDonald *et al.* (2000), were comparable to that measured by Gras (2002), and lower than data presented by Hedberg *et al.* (2002). Appliance D appeared to generate the highest emissions for many of the PAH species including fluoranthene, fluorene, phenanthrene, anthracene and pyrene, with Appliance A discharging the lowest emissions of these compounds. Appliance D generated the highest PM emissions, and Appliance A the lowest PM emissions, suggesting that the PAHs are related to PM emissions. This is consistent with the findings of Gras (2002) who found that PAHs were more abundant when PM emissions were high. Emissions of the various PAH species measured during this study were higher than those measured by Fisher *et al.* (2000) but lower than Hedberg *et al.* (2002). Emissions of the BTX species (benzene, toluene and xylene) were greatest from Appliance E. The benzene emission from this appliance (1094 mg/kg) was similar to that measured by Hedberg *et al.* (2002) and McDonald *et al.* (2000) but lower than Gras (2002).

The data presented in Table 4-1 demonstrate the variation evident in emissions of these contaminants from wood burning appliances. The air toxics results from this study are limited to Stage I. However, it may be expected that higher emissions of some of these contaminants, in particular the PAHs, would be observed in the field. This is a concern as the Stage I (laboratory) results are already higher than those observed in the field in the US (e.g. Fisher *et al.*, 2000). In addition, a recent study indicated that MfE’s annual guideline for BaP is exceeded in Christchurch by an order of magnitude (McCauley, 2005).

## 4.3 Elemental species

The elemental species measured during Stage I are presented in Figure 4-8. These samples were collected during the combustion of low resin pine and eucalypt firewood on Appliance A and C during Stage I. As the samples were collected for different time periods and did not encompass entire test runs the data could not be presented in mg/kg. Nevertheless, the relative abundance of various chemical species is indicated providing an elemental “source profile” which may assist with the identification of sources resolved by source apportionment studies (NB: organic carbon was not measured).

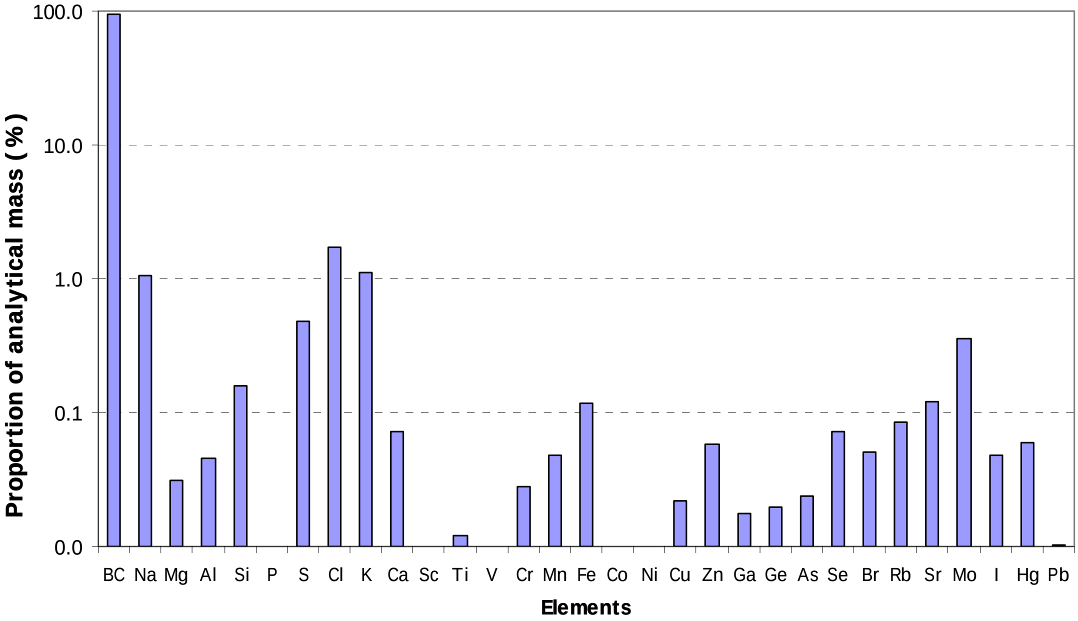


**Figure 4-8 Average elemental composition of wood smoke from two residential wood burning appliances**

The analytical mass was dominated by black carbon (elemental carbon) comprising 94.2% of the mass. Other key elemental species identified included sodium (1.1%), chlorine (1.7%) and potassium (1.1%). If organic carbon, had been measured it is likely, based on the results of other studies, it would constitute the most important chemical species, followed by elemental carbon (Gras, 2002; McDonald *et al*., 2000).[16]

[16] Measurements conducted by Gras (2002), for example, using locally available firewood (but appliance fired in accordance with AS/NZS 4012/3) found emissions of 1900 mg/kg of organic carbon and 700 mg/kg for elemental carbon.

# 5 Wood burner operation and fuel characteristics

A telephone survey of householders was conducted in October 2003, to determine methods of operating residential wood burning appliances and the types of fuels used. This information, provides excellent information on general wood burner operating practices in Christchurch. It allows the appropriateness of operation to be assessed and identifies those areas where improvement is required. Operator variables appear to have a substantial impact on emissions, and if wood burners are to continue to be used in Christchurch, it is essential that these are operated responsibly. The following sections discuss key results of the wood burner operation survey together with emission data obtained on these variables, collected during Stage I of the study. The results of the wood variable tests are included in Appendix 5.

## 5.1 Frequency of use and burn durations

The operation survey provided general information regarding the frequency of appliance use and burning durations. It was found that those households that use wood burners tend to do so for most of the winter. For example, 76% use their appliance for 5 to 7 days per week during June to August. Only 2.5% were actually influenced by the weather. During the week 57% used their appliance for six hours or less while 42% operated it for six hours or more. On the weekend, the hours of operation increased with 67% using the appliance for six hours or more, and 31% for six hours or less.

These results indicate that appliances are operated frequently and for varying durations. While some appliances may be used continuously (24-hours) most are operated for relatively short time periods. The fire would be allowed to burn out overnight and would be re-started the following morning or evening. If fires are typically lit during the cooler morning or evening hours, when temperature inversions are evident, these emissions will be trapped at the surface leading to contaminant accumulation and elevated $PM_{10}$ concentrations. Light-up emissions, therefore, may have a substantial impact on daily $PM_{10}$ concentrations around New Zealand.

Emissions discharged during the light-up phase and the stable-phase were measured during Stage I of this study. Two samples were collected during each operational cycle, with one encompassing the first hour of operation and the second including the following five hours. Light-up emissions were high relative to the stable-phase when dry pine off-cuts were used, and also for oregon, macrocarpa and low resin pine (between 2.6 and 4.2 times higher), although there was only one sample available for comparison for these wood types. The greatest start-up emission occurred when Appliance B was tested using eucalypt (13.6 g/kg). In all cases, light-up emissions were, on average, greater than the stable-phase emissions. This demonstrates their importance when considering the overall impact of emissions from wood burners.

## 5.2 Burn rates

The wood burner operation survey provided information that could potentially be used to design a representative firing regime for emission testing purposes, and highlighted those areas where operation could be improved.

In general, most wood burning householders were of the view that their fire was established within 10 minutes of light-up (77%). This is a potential issue as it suggests that householders may be loading up the fire earlier than they should – i.e. before the firebox and flue are sufficiently heated – leading to increased emissions during the light-up phase (or alternatively, householders misunderstood what was meant by “established”). However, a high air setting was used during light-up by the majority of householders (84%), which would result in lower emissions relative to other air settings. The main air setting (or burn rate) used throughout the operational cycle was medium (40% of households) followed by low (26% of households). It also appeared that householders establish their fire, turn it down to low or medium and do not adjust it for the remainder of the burn. Only 34% of householders adjusted their appliance to a high burn rate, for example, when refuelling. This is contrary to best practice which promotes the use of a high burn rate (or air setting) for 15 to 20 minutes after re-fuelling (Todd, 2003b).

PM emissions from Appliances A and B were collected under three different burning-rate regimes during Stages I and II (Table 5-1). Increasing burn rates coincided with decreasing emissions on a g/kg basis, but increasing emissions (g/hr) during Stage I. This indicates that Appliance A and B are operating more efficiently at higher burn rates, but because greater quantities of fuel are consumed, the emissions increase on an hourly basis.

It is uncertain whether this would also be found in the field (high burn rates were not investigated during Stage II) or in the emissions from a wider group of appliances. Rates of combustion are influenced by burner design characteristics such as firebox size, secondary air supply and heat retention capacity and each appliance may respond differently to the various burn-rates. For appliances where emissions increase at high burn rates (e.g. on a g/hr basis), it is likely that inadequate air supplies are available to sustain complete combustion. Conversely, for appliances that have lesser emissions at high burn rates, the combustion rates may be lower or better air supplies are available.

**Table 5-1 Emissions averaged for the low, medium and high burn rates for Appliances A and B**

| Low emission wood burners | Burn rate | PM emissions (g/kg) | | | PM emissions (g/hr) | | |
|---|---|---|---|---|---|---|---|
| | | Stage 0 | Stage I | Stage II | Stage 0 | Stage I | Stage II |
| Appliance A | L | 0.7 | 2.4 | 2.8 | 1.3 | 4.5 | 4.3 |
| | M | 0.5 | 2.3 | 2.6 | 1.1 | 4.8 | 4.1 |
| | H | 0.5 | 2.1 | | 1.2 | 5.1 | |
| Appliance B | L | 1.7 | 3.1 | 2.9 | 3.7 | 8.0 | 6.0 |
| | M | 1.0 | 2.2 | 1.6 | 3.0 | 5.8 | 4.0 |
| | H | 0.8 | 2.9 | | 3.1 | 9.2 | |

NB: L = low, M = medium and H = high

## 5.3 Overnight burns

The proportion of householders in Christchurch that re-load their appliance and adjust it to a low burn rate prior to retiring for the evening is 26%. However, the question may not have been answered honestly by all householders as many people are aware that this is an undesirable practice, and the proportion may in fact be higher. While the impact of overnight burning on emissions was not investigated during this study, emissions are likely to increase if the appliance is reloaded and immediately set to a low burn rate (Todd, 2003b). This is a practice that should be discouraged in any urban area, where wood smoke is a potential issue.

## 5.4 Firewood and load size

The survey also included questions on firewood loading practices. Approximately 52% of wood burning households loaded multiple pieces of firewood into the fire, with 33% loading a single piece.

Questions were asked about log size, with 49% of households used medium sized logs (weighing between 1.5 and 2kg), and 29% used large logs (weighing greater than 2 kg). The proportion of households using medium to large pieces of wood is high and could lead to inadequate burning conditions if the appliance is not hot enough or the air supply is insufficient. Larger pieces of firewood have less surface area than an equivalent load made up of more, smaller pieces of wood, and result in slower combustion (Todd, 2003b). The use of large wood pieces can also lead to appliance overloading. Ideally enough space must be left inside the appliance for the combustion gases to mix with air (Todd, 2003a). If this is insufficient, unburnt gases are discharged, condense into tar droplets and create smoke.

Combustion efficiency may also be reduced if the firewood pieces are too small. Wood that is too small combusts more rapidly and can become oxygen starved if the air supply is insufficient (Shelton, 1983). In addition, log loading geometry is an important factor with greater emissions produced when logs are placed parallel to the appliance door (*pers. comm.* Dr John Todd, Eco-Energy Options Pty Ltd).

Firewood size was one of the variables tested during Stage I of this study. When large firewood (1.5 to 3 kg) was used, emissions increased by approximately 50%. This is clearly an important variable, highlighting the importance of addressing firewood size through education and possibly regulatory means.

## 5.5 Firewood source and species

Firewood used in Christchurch is both purchased and available for free. In 2003, approximately 51% of wood burning households purchased all their wood, 30% obtained all of it for free and 19% did both (Lamb, 2003).

A wide variety of wood species is available for sale as firewood in Christchurch. These include pine, beech, macrocarpa, oregon, willow, eucalyptus, poplar, silver birch, manuka, larch, oak, and wattle. These are usually available as wood slabs, split firewood, off-cuts and demolition timber. Of those householders that purchased their fuel, pine was most popular - being purchased by 41% of householders. This was followed by macrocarpa (17%), oregon (16%) and eucalyptus (10%). Pine was also the most common self-sourced or free firewood with 41% of households that self-sourced their wood obtaining this fuel, followed by macrocarpa (11%), eucalyptus (10%) and off-cuts (8%).

Emissions discharged during the combustion of different firewood species were investigated during Stage I. These tests were conducted on Appliance A and B and included wood types such as high resin pine, low resin pine, oregon, macrocarpa and eucalypt. The results obtained are outlined in Appendix 5. The highest emissions were generated when burning eucalypt (5.7 g/hr on average) followed by high resin pine (5.3 g/hr on average), macrocarpa (4.7 g/hr – one sample only), oregon (2.8 g/hr – one sample only), and low resin pine (2.8 g/hr – one sample only).

Studies conducted elsewhere also indicate substantial variations in emissions depending on the wood type used. Environment Canada (2000) tested two types of wood (maple and spruce) on an older wood burning appliance and a USEPA certified appliance. Average emissions were greatest when spruce was used, especially when burnt in the older (conventional) appliance. The difference was not as marked for the certified appliance. McDonald *et al.* (2000) found emissions were greater for hardwoods than softwoods when burned in an open fire, and that the firewood species was an important determinant of the chemical compounds found in the wood smoke. In that study VOCs, PAHs, organic carbon, furans and CO were greatest during the combustion of hardwood. Retene and elemental carbon, on the other hand, were discharged in far higher quantities from soft woods. Retene is not normally found in emissions from hardwood species, and the elemental carbon discharge suggests improved combustion relative to the hardwood test runs. Rogge *et al.* (1998) provided an excellent discussion of the influence of the different wood constituents on the organic compounds discharged during combustion. Variations in emissions from the use of pine, oak and synthetic logs on an open fire were determined. PAHs were greatest from the synthetic logs followed by pine. Schauer *et al.* (2001) and Fine *et al.* (2002) also determined the chemical composition of wood smoke generated by different wood types. Schauer *et al.* (2001) found emissions were greatest from pine followed by eucalypt and oak, while Fine *et al.* (2002) observed substantially higher emissions from hardwood species known as yellow poplar and mockernut hickory.

The emissions from the combustion of different wood types vary considerably and are partially determined by the nature of the appliance used. The studies mentioned above were mostly conducted on open fires so it is difficult to say how these fuel types would operate on New Zealand appliances. Gras (2002), however, tested a variety of softwoods and hardwoods on Australian-designed appliances. Softwood on these appliances tested poorly relative to the hardwood species. Higher PM and PAHs were generated when softwoods were used. In general, hardwoods (as defined by density) which are usually more difficult to light and subsequently burn more slowly, generate greater emissions during the light-up phase, while pine lights more easily and burns faster producing higher emissions under slow burn rates (i.e. becomes oxygen starved; Todd, 2003a).

Appliance design can address these factors. Appliances are tested using softwoods in New Zealand, and hardwoods in Australia, as these are the dominant fuel types. Wood burner performance is optimised for these wood types. This is demonstrated in the results of this study and the Gras (2002) study, where emissions were greatest from eucalypt on the New Zealand appliances and pine on the Australian appliances. Wood type therefore may be important when attempting to educate householders about appropriate wood burner operation.

## 5.6 Firewood moisture

Firewood moisture information could not be collected during the household survey as it was conducted over the telephone. Previous work undertaken on this issue provides some information regarding typical moisture contents of firewood used in Christchurch.

A door-to-door survey of 200 households was conducted in 1999. The mean firewood moisture content (on a wet weight basis) was 16% with a minimum of 7% and a maximum of 35%. Although exact figures were not supplied for the proportion of households using wood at certain moisture contents, the percentiles indicated that at least 75% of the wood tested was within the required limit of 20% on a wet weight basis (25% dry weight; Lamb, 1999a).

A wood moisture survey of firewood merchants was conducted during the winter of 1998 (Lamb, 1999b). This found that 36% of the wood samples tested were greater than 23% on a wet weight basis (equivalent to the previous dry weight standard of 30%). These data suggest that although merchants may be selling a higher proportion of wood exceeding moisture standards, householders are covering and/or seasoning the wood prior to using it.

Wood moisture, and its impact on emissions, was investigated during Stage I (Appendix 5). The tests, conducted on Appliance A and B, indicated that emissions increased by 50% when wet pine (approx. 27% wet weight) was used in place of dry pine firewood (15%). Gras (2002) also examined the impact of wood moisture and found higher emissions were associated with the burning of green softwood (54% on a wet weight basis) and dry softwood (11-14% moisture), relative to fuels with moisture contents of 20 to 30%. Shelton (1983) indicates that high moisture decreases combustion efficiency as more energy is required to evaporate the water contained within the fuel. This lowers the temperature of the fire leading to less complete combustion (Todd, 2003b). Alternatively, if wood is too dry, combustion can be rapid and an insufficient oxygen supply can also lead to greater emissions. Fortunately the survey data suggest that most wood used by householders in Christchurch is within the required limits (<20% wet weight). However, if the firewood is too dry, or if new wood supplies are sought during winter, an increase in emissions could occur. These issues could potentially be addressed through education and/or better regulation of the firewood being used.

# 6 Estimating real-life emissions for low emission wood burners

## 6.1 Development of a conversion factor from AS/NZS 4012/3

One of the key objectives of the study was to determine whether a conversion factor could be used to estimate real-life emissions for specific models of low emission wood burners using results from tests conducted under AS/NZS 4012/3.

The simplest approach to developing a conversion factor is to establish the relationship between Stage 0 and III emissions using a regression analysis of measurements from a representative range of low emission wood burners. If a relationship can be established then the regression equation may be used to convert the AS/NZS 4012/3 test average for an individual appliance into a "typical" real-life emission estimate.

Alternatively, a staged approach may be adopted where the relationships between Stage 0 and I (i.e. from non-realistic operation and fuel to a more realistic mode), and Stage I and II (taking installation characteristics into account) are used to convert AS/NZS 4012/3 data into Stage II equivalents. The Stage II equivalents may then be weighted by "typical" operational characteristics and their impacts on emissions, to derive Stage III emissions representative of the wider population.

While the following section will discuss these two methods, it is important to note from the outset that that sufficient emission data must be available for realistic, robust and representative conversion factors to be derived. The number of low emission wood burners tested during this study was limited and only four were available to establish the relationship between Stages 0 and III, and only two for Stages 0 and I, I and II. Nevertheless, the methodology is discussed as it may be useful for future assessments.

### 6.1.1 Direct conversion

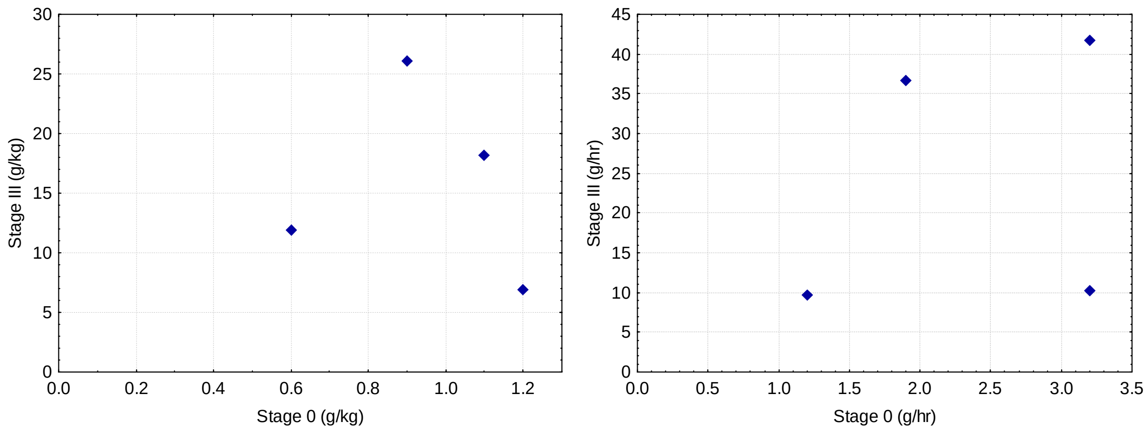


**Figure 6-1 a (left) and b (right) Averaged Stage 0 and III emissions for Appliances A, B, C, and F in g/kg (a) and g/hr (b)**

Figure 6-1 (a) and (b) present averaged Stage 0 and III emissions for the low emission wood burners tested during this study. Individual measurements could not be used as the Stage 0 test results were specific to a particular burn rate and could not be directly linked to a Stage

III emission. Rather, the nine Stage 0 emissions, and seven Stage III emissions, for each appliance were averaged.

A large scatter is evident in the data suggesting that a relationship does not exist between Stage 0 and III emissions. However, the number of samples is insufficient to determine whether or not this is the case. Results from a wider selection of appliances encompassing the full range of operational characteristics (fuel type, size, moisture content; and mode of operation such as burner settings), and installation variables (e.g. flue height, presence of secondary air bars and water boosters) would be required to determine whether a relationship exists.[17]

A wide range of tests is also necessary to ensure that the relationship is representative of "typical" operation rather than of a small sub-set of behaviours such as that from this study. It may be found that even when all variables are accounted for it is still difficult to establish a relationship because of the nature of the Stage 0 test methodology (i.e. short-burn time, optimal operation, exclusion of the light-up phase and the use of specialised fuel). It is not guaranteed that all appliances would respond in a proportional manner to these variables. Nevertheless, the advantages of a broad field study are two-fold. Not only would it be possible to definitively establish whether a relationship exists between the two stages, but it would also allow a representative emission factor to be developed.

### 6.1.2 Stage-by-stage conversion

As demonstrated in Section 6.1.1, a direct relationship could not be established between the Stage 0 and III measurements collected by this study. Thus, direct conversion of the data from Stage 0 to III was not possible. An alternative approach was investigated to assess whether a stage-by-stage conversion could be conducted, and the resulting data weighted by operational variables, to produce Stage III equivalents representative of "typical" operating behaviours.[18]

To determine whether a stage-by-stage conversion factor could be used for low emission wood burners, the relationships between Stage 0, I and II emissions were examined. If reliable relationships were determined then it would be possible to convert Stage 0 data into Stage I equivalents (to take real-life operation and locally sourced firewood into account) and then convert these into Stage II equivalents (allowing for the impact of a range of installation variables). The resulting value would not be representative of real-life emissions as it would be derived from emissions produced under prescribed operating conditions and fuel. To produce a more realistic value, the Stage II equivalent would then be weighted using a combination of data from the wood burner operation survey and results from the additional variable tests conducted during Stage I. The final estimate would comprise a Stage III equivalent, more representative of the wider population.

For this method to be successful the following is required:

- A sufficient number of appliances must be tested during Stage 0, I and II for relationships between stages to be developed

[17] Weighting of the Stage 0 emissions may also be required to take into account "typical" burn rates used in a particular area.
[18] A conversion factor developed on the basis of the relationship between Stage II and III would not be useful as a limited number of samples were available for Stage III encompassing only a small range of operations, installations and fuels.

- The appliances and installation characteristics must be identical between stages
- A wide range of variables must be tested in the laboratory using several wood burners
- A survey of householders must be conducted to determine “typical” operation

This study tested five appliances during Stages I and II, and six in Stage III. Only four of these appliances could be confirmed as low emission wood burners, and while the study made every attempt to limit the introduction of new variables between stages, to those specifically being investigated, this was not always possible. Installation characteristics varied on occasion and appliances tested in the field were not always those intended. Only Appliances A and B were tested in both Stages I and II (an earlier model C was tested in Stage I than during Stage II, and F was only tested during Stage III).

The study therefore provided a small dataset from which to conduct a comparative analysis of emissions. A single average was available per appliance (average of the combined low and medium burn rate results), and a relationship between stages could not be determined on the basis of two data-points. To increase the number of samples available, an attempt was made to conduct the analysis using two averages per appliance (the low and medium burn rate results were averaged separately – each based on three tests). While this increased the number of data-points to four, it was not a particularly valid approach as averages based on three test runs are unlikely to be representative of the “true” average for low and medium burn rate emissions. AS/NZS 4012/3, for example, indicates that nine test repeats are required to produce a representative average across all burn rates.

The study was also limited by the number of specific variables investigated. During Stage I, the impacts of fuel type (oregon, pine - low, medium and high resin – eucalypt and macrocarpa), size (small and large), and wood moisture (wet and dry) on emissions were determined. However, these tests did not encompass all fuel types and installation characteristics introduced during Stage III. For example, birch and native timbers were used, and hot water boosters and different flue lengths were evident during Stage III. While the installation characteristics would partly be taken into account when deriving the relationship between Stage I and II, it is uncertain how a combination of these variables (e.g. use of native firewood on an appliance fitted with a 7.5 m flue) would impact on emissions. As such, it would be difficult to adequately adjust Stage III data to produce a value representative of “typical” operation and installations. This method is reliant on establishing the impact of these variables on emissions for a wide range of appliances and situations, and would require more substantial testing than that conducted by this study.

As the relationships between stages, and the impact of different variables on emissions, were limited to data provided by two appliances, it was not worthwhile conducting the analysis. Nevertheless, the methodology proposed for conducting a stage-by-stage conversion is included in Appendix 6 for future reference.

## 6.2 Development of emission factors using study data

Another key objective of the study was to provide “scientifically defensible” emission data based on real-life emissions for use in air quality assessments in New Zealand (i.e. an emission factor for low emission wood burners as a class or classes of appliance). These air quality assessments include emission inventories and airshed capacity evaluations.

The study was initially designed to develop emission factors based on relationships between the different stages, and to weight these emissions by typical operating behaviours. Difficulties associated with maintaining consistency in appliances between stages, and the limited number of operation and fuel variables investigated, prevented such an analysis. It was subsequently concluded that the only genuine in-home emissions data were those provided during Stage III.

However, the Stage III results were not necessarily representative of low emission wood burners, as a class of appliances, as only four appliances in this category were tested and the results were based on a single household for each appliance type. Table 6-1 summarises the operational/fuel variables used by Stage III householders together with typical operation as determined by household survey. When the variables are compared, it is evident that Stage III operation differed from that deemed typical of the average householder in Christchurch.

**Table 6-1 Wood burner operation and fuel characteristics during Stage III, where n = sample size**

| Characteristics | Low emission wood burners Stage III | Typical operation (household survey) |
|---|---|---|
| n | 4 | 513 |
| Main wood burner setting[1] | | |
| Low | 18% | 30% |
| Medium | 6% | 44% |
| High | 76% | 26% |
| Wood size | | |
| Large wood (1.5 to 3 g/kg logs) | 75% | 33% |
| Small wood (<1.5 g/kg logs) | 25% | 67% |
| Wood moisture | | |
| Wet wood (~27% wet weight) | 25% | 1% |
| Dry wood (~ 11% wet weight) | 0% | 91% |
| Moist wood (11-27% wet weight) | 75% | 8% |
| Wood species | | |
| Gum (Eucalypt) | 25% | 16% |
| Oregon (Douglas fir) | 25% | 16% |
| Macrocarpa | 0% | 21% |
| Pine | 50% | 47% |

[1]For the in-home tests the proportion of time using the particular air setting is indicated rather than the main air setting used (not practical when only one or two appliances are tested).

Key differences between Stage III and typical (surveyed) operation were:

- some appliances were operated with significantly smaller or larger fuel loads
- less wood used but the logs were larger
- there were significant differences in the time between loads
- greater use of the high burn rate setting

In addition, the testing laboratory noted that during Stage III, higher flue oxygen levels and lower flue temperatures were observed relative to Stage II. It was suggested that the fires were burning cooler and less efficiently, possibly as a direct result of the larger fuel loads and the use of higher control settings (ARS, 2004a).

The range of behaviours exhibited is an important consideration as emission factors are representative values that provide a measurement of contaminant discharge for a specific type of activity and fuel consumption. These values must be representative of the wide and highly variable range of appliance operating conditions and practices employed under real-life conditions. A reasonable number of households (513) participated in the home heating survey and these “typical operation” numbers are more likely to be representative of average behaviour than that those demonstrated during Stage III (4 households). Consequently, it was inappropriate to establish an emission factor for low emission wood burners on the basis of the Stage III data.

However, the emission data do provide at least some indication of the potential magnitude of emissions discharged in the field from a small group of low emission wood burners, under householder-supplied fuel and appliance operation conditions. These currently constitute the only in-home emissions data available in New Zealand and suggest that the emission factor used to estimate emissions from low emission wood burners might be too low (3 g/kg). This may have substantial impacts on emissions forecasts conducted by air quality regulators, and it may be necessary for the potential impacts of this uncertainty to be estimated. More sampling of similar appliances in other households is needed before conclusions may be drawn about the way these results might be applied to the broader population and what an accurate emission factor might be.

# 7 Discussion

## 7.1 Real-life emissions from low emission wood burners

Prior to undertaking this study, there were no emission measurements available for installed residential wood burners in New Zealand or Australia. The only published in-home data were those derived from studies conducted in the US. These data formed the basis of the emission factors currently used for wood burners assessments in New Zealand. However, specific field data for appliances with very low AS/NZS 4012/3 PM emissions (e.g. less than 1.5 g/kg) did not exist and, a PM emission factor of 3 g/kg was assumed. As it was uncertain whether this value was applicable to the current mix of low emission wood burners, the real-life wood burner emissions study was commissioned. Consequently, this report presents the first field emission measurements from low emission wood burners in Australasia, and provides a base from which future studies can be developed.

The PM emissions measured during Stage III are the only genuine real-life emissions data collected during this study. Individual results for the low emission wood burners tested were variable with ranges from 1.9 to 36.0 g/kg and 2.7 to 85.2 g/hr (Appendix 4). Means of 15.5 g/kg and 24.1 g/hr, and medians of 13.0 g/kg and 17.5 g/hr were calculated. The ensuing sections discuss the applicability of the 3 g/kg PM emission factor, and the representativeness of Stage III emissions for this class of appliances.

The study also addressed emissions of contaminants other than PM. This included a variety of air toxics and elemental species. High PAH emissions coincided with high PM emissions. The measurements, which were conducted in the laboratory during Stage I, were greater than actual in-home measurements from appliances tested in the United States (Fisher *et al.*, 2000). The US appliances were rated higher (on an emissions basis) than the appliances used in this study. As PM measurements in the field were far greater than the Stage I emissions, it is likely that PAHs will be even more elevated in the field, under real-life operating conditions. Key elemental species detected in the wood smoke of two appliances tested during this study included black carbon, sodium, chlorine and potassium (organic carbon was not included).

## 7.2 Appropriateness of the 3 g/kg emission factor for low emission wood burners

The $PM_{10}$ emission factor currently used by regulators to represent low emission wood burners in emission inventories and to forecast emissions is 3 g/kg. As an emission factor, this value is required to represent the long-term average of all appliances in a set category. It needs to be representative of the wide and highly variable range of appliance operating conditions and practices employed under real-life operating conditions. The data provided by this study, while not covering anything like the full range of conditions and low emission appliances currently in-use, provides an indication of the potential difference in emissions that may be seen when the appliance is operated in the home compared to that obtained in the standard laboratory test.

The 3 g/kg emission factor is substantially lower than the mean emission of 15.5 g/kg derived during Stage III. However, the mean is not the best average for comparison as extreme values such as 36.0 g/kg distort the average. The median provides a better measure of average emissions from the low emission wood burners tested. The calculated median of 13.0 g/kg suggests that the emission factor currently used could be too low by a factor of up to 4 or 5. While the median may be useful for indicating possible uncertainties associated with emission factors, it is not sufficiently robust for use as an actual emission factor for emission inventory purposes. Emission data from the US, for example, suggests that the averages obtained by this study may be too high. The US study tested eleven certified non-catalytic wood burners in the field (30 runs) and reported, for example, mean emissions of 9.23±5.2 g/kg and 10.3±8.3 g/hr (Fisher *et al*., 2000).[19] Despite this concern, the results are useful for indicating the potential uncertainties associated with emission estimates for low emission wood burners for emission forecasting and airshed capacity assessments.

As a separate issue, when the 3 g/kg emission factor was first applied to sub 1.5 g/kg appliances for airshed management calculations, it was also assumed there was a point below which improvements in authorisation test results would not correspond with real-life improvements. This was because it was considered that under real-life operating practices, mode of operation and variations in fuels would offset any improvements achieved by the technology. This threshold value was considered to be an AS/NZS 4012/3 test emission of 1.5 g/kg. All appliances with authorised test emissions below 1.5 g/kg were assumed to discharge 3 g/kg under real-life operating conditions.

To determine whether this threshold applies, the Stage 0 averages were plotted against those obtained during Stage I and III. Figure 7-1 (a) and (b) demonstrate that when technology improves (on the basis of Stage 0 emissions), but all other factors remain constant (i.e. operation, firewood type and installation), a steady reduction in emissions is evident. That is, under simulated real-life operating conditions in a controlled laboratory environment, continual improvement in emissions may be expected. In contrast, when installation, operation and firewood type varies (Figure 7-1 d and e), Stage 0 provides a poor indication of the relative performance of appliances and continual improvement would not necessarily be expected. It is uncertain whether a threshold exists as the sample size is too small and only one appliance has Stage 0 emissions above 1.5 g/kg. Note, however, that an obvious threshold value could not be observed in the results of a US study where a greater number of appliances were tested (Figure 7-2).

Although it was not possible to determine whether a threshold value applies, it was clear that 3 g/kg is not representative of this group of low emission wood burners, and that the true average for low emission wood burners in this test is likely to be substantially higher than the value currently used.

[19]Median emissions were not available for comparison and it is highly probable that the results were influenced by substantially longer burn times than that used during the New Zealand study. Most appliances were operated in excess of 16 hours per day and at least four were used continuously over 24-hours. Also, one-week samples were collected over two to three weeks per appliance.

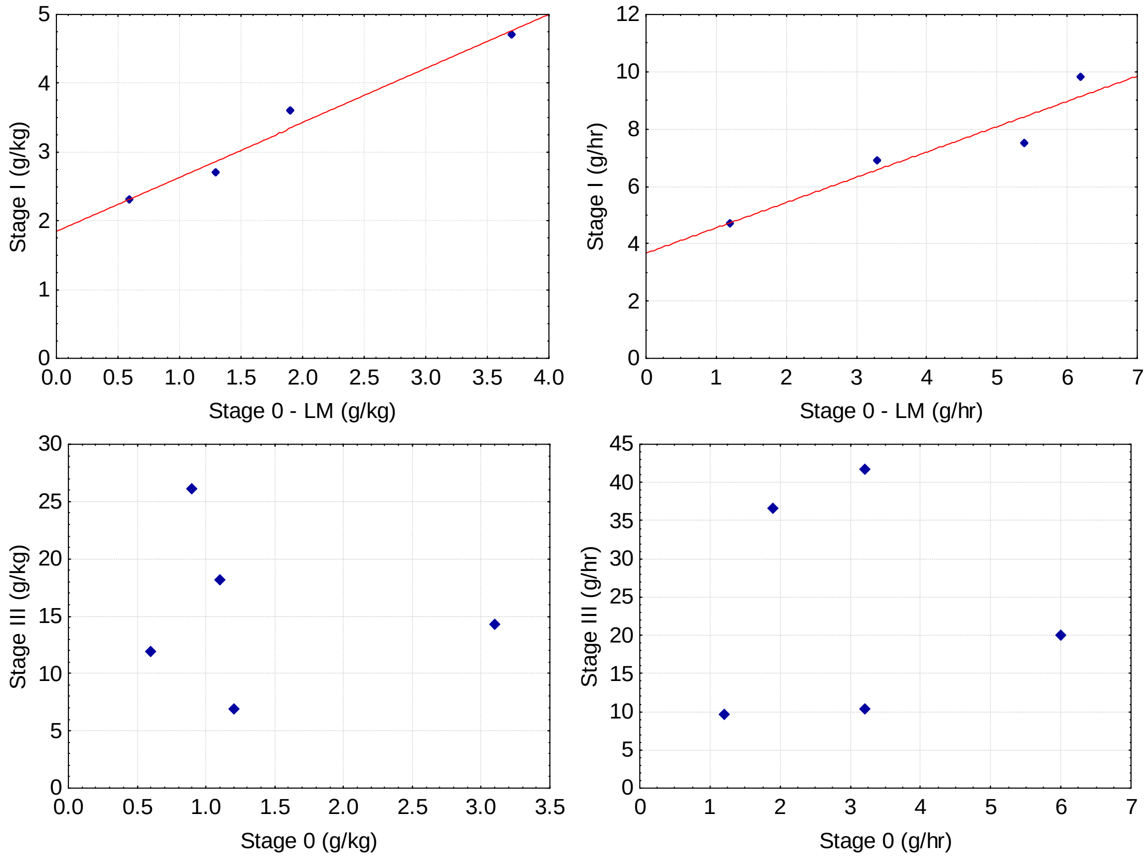


**Figure 7-1 a ( top left), b (top right), c (bottom left), d (bottom right) Relationships between Stage 0 and I (a & b), and 0 and III emissions (c & d).[20]**

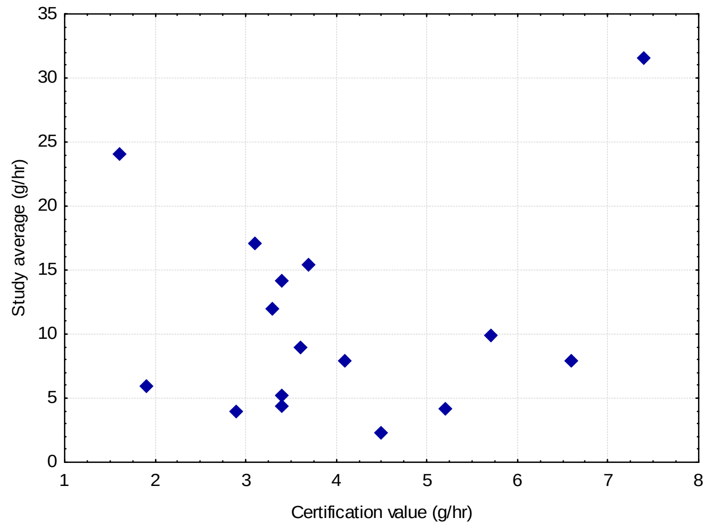


**Figure 7-2 Comparison of certification test emissions with real-life emissions in Klamath Falls and Portland, Oregon (Fisher et al., 2000)**

[20] NB: (a) and (b) include averaged Stage 0 emissions for low and medium burn rates, whereas (c) and (d) include low, medium and high burn rates. Also the 1.7 g/kg appliance is presented in Figures (a) and (b) whereas the 1.2 g/kg appliance is included in Figures (c) and (d).

## 7.3 Development of conversion and emission factors

The study investigated whether a conversion factor could be used to estimate real-life emissions from AS/NZS 4012/3 test data for specific low emission wood burners, and/or an emission factor to represent emissions from low emission burners as a class of appliances. The relationship between averaged Stage 0 and III emissions was assessed to determine whether direct conversion was possible from Stage 0 to III. However, the number of low emission wood burners tested was limited and only four were available for the analysis. As indicated in Figure 6-1, a wide degree of scatter was evident, suggesting that a relationship could not be found. If a greater range of appliances had been tested (encompassing a wide variety of installations, operations and firewood types), a relationship may have been found. Nonetheless, on the basis of the information provided by this study, a conversion factor could not be used to directly translate AS/NZS 4012/3 results into Stage III (real-life) emissions.

The relationships between Stages 0 to I, and I to II were also investigated to determine whether a step-wise approach could be adopted. The method proposed was to convert Stage 0 into a Stage I equivalent, then further convert this into a Stage II equivalent. This would constitute a "raw" emission value which could then be weighted on the basis of typical operation and fuel use characteristics in Christchurch to produce a Stage III equivalent value, representative of "typical" operating behaviours. However, for this method to succeed, a reasonable number of appliances needed to be tested, with identical models and installation characteristics between stages. The study made every attempt to limit variations between stages to those being investigated, but this was not always possible. Installation characteristics varied and, on occasion, the appliance being tested was different to that intended. Consequently, only two low emission wood burners were available for comparison between Stages 0 to I, and I to II. On the strength of this, the multiple conversion option could not be pursued further. Even if it had been possible to develop emission factors using this methodology, the data would be representative of Christchurch only. For the data to apply to other areas, each region/city would need to obtain operator and firewood information characteristic of their own region, and weight the data accordingly.

The original intention of the study was to determine relationships between stages and convert data into real-life emissions based on operator variables as described above. This could not be conducted due to difficulties in controlling variations between the different stages. Consequently, the only genuine in-home emission data provided by the study were those measured during Stage III. However, the ability of those results to represent the wide range of low emission wood burners, installations, methods of operation and fuel types used, was uncertain. The operational variables adopted during Stage III were compared to those typical of the wider population, as determined by household survey. It was found that the Stage III householders operated their appliances differently to that of the average householder. As the data were only representative of a small number of appliances, and the mode of operation may not be typical, it was not appropriate to develop emission factors from this dataset. These values only represent average emissions for a narrow range of appliances – contrary to the requirements for the derivation of emission factors.

Nevertheless, the data indicate that in a "real-life" situation, some appliances may well produce emissions that are very much higher than the real-life emission factor commonly assumed for sub 1.5 g/kg appliances. How representative the results of the study are is unknown, but they provide evidence that 3 g/kg may be an underestimate. Further monitoring of in-home emissions is required, and at a larger scale than that for this study, to derive valid emission factors.

## 7.4 Impact of operation, fuels and installation on emissions

The study directly investigated the impacts of different firewood variables on emissions during Stage I, and determined typical operating practices by household survey. The combined impacts of installations, operation and firewood types on emissions were determined indirectly, as inferred by the difference in emissions between Stages II and III.

It was found that wood burning appliances in Christchurch were typically used for relatively short durations (i.e. non-continuous), increasing the potential impact of light-up on daily PM emissions. Light-up emissions determined during Stage I were approximately 3 to 4 times higher on average than those discharged during the stable-phase.[21]

Most householders operated their appliances on low or medium burn rates with very few attempting overnight burns. A relatively high proportion of householders used medium to large pieces of wood and tests conducted during Stage I indicated that emissions can double when large wood is used in place of smaller pieces. Appliance overloading and a consequent increase in emissions may also occur.

The impact of fuel species on emissions was investigated with the greatest emissions discharged when eucalypt was used, followed by high resin pine, macrocarpa and oregon. The number of householders using wet wood was not determined during this study but previous work suggested that a relatively low number of households in Christchurch use wet wood. This is encouraging as emission tests conducted during Stage I found that wet wood could double emissions.

The results presented in Figure 4-1 to 4.6 (a) and (b) illustrate the variation in emissions between different test stages – where possible - providing an insight into the potential impact of operation and firewood types on emissions. These demonstrate:

- Stage 0 to I – the impact of introducing a realistic mode of operation including light-up and the use of locally-sourced firewood.
- Stage I to II – the impact of different installation characteristics such as flue height on emissions (mode of operation did not vary from Stage I and the appliance was operated by a technician).
- Stage II to III – the impact of householder operation using their personal firewood supply.

The introduction of a more realistic burning cycle (six hours, inclusion of light-up and refuelling) and the use of locally sourced firewood increased emissions during Stage I relative to Stage 0. Averaged emissions for Appliances A, B, C and D presented in Figure 7-1 (a) and (b) were substantially higher than those indicated for Stage 0 and the regression line suggests that factors such as the use of locally-sourced firewood and the inclusion of light-up may account for approximately 1.5 to 2 g/kg and 4.5 to 5 g/hr of emissions.

[21] Light-up emissions include all PM collected during the first hour of sampling.

The change in emissions from Stage I to II is less readily understood as inconsistencies were observed with some Stage II emissions increasing relative to Stage I, and some decreasing. Further testing would be required to determine exactly why this occurred. It is likely that the step from laboratory to field testing was not in itself a single variable, but the difference is probably of only limited impact.

Of greater interest, is the magnitude of the difference in emissions from Stage II to III. Emissions for all appliances, with the exception of Appliance C, were substantially higher during Stage III although C, on average, was still higher for III compared to II at least on a g/kg basis. This suggests that the mode of operation and the fuels used by the householders were responsible for the marked increase in emissions. On the other hand, it may be that the installation and appliance design characteristics of some appliances were more accommodating of sub-optimal modes of operation and unfavourable firewood characteristics, than some other appliances. The latter may have been the case for Appliance C as the change in operation and fuels between stages did not increase emissions to the same extent as that of the other appliances. While it was not possible to determine which particular factor or combination of factors led to the lower emissions, this important area of uncertainty may need to be addressed in future research.

In a more general sense, actual appliance operation during Stage III highlighted some important issues. Householders tended to use less fuel, larger logs and, in some cases, wetter wood for this part of the study. ARS (2004a) suggested that fires were burning cooler, and less efficiently, as a result of the fuel load size and the use of higher control settings. Research conducted in the US reveals that common household practices such as the use of inadequately seasoned firewood, large firewood loads, and insufficient kindling and light-up time (i.e. loading up earlier than appropriate) can have a major impact on emissions (*pers. comm.* Paul Tiegs, OMNI Environmental, Oregon, USA).

What is certain is that wood burner operation, installation and firewood characteristics have a substantial impact on emissions. To address this, new technologies would need to accommodate a wide range of operational behaviours, flues and firewood variables without substantially increasing emissions. The test method (AS 4012/3) used to authorise wood burners in New Zealand does not currently consider the impact of these variables on emissions. Therefore, emission improvements in the laboratory will not necessarily result in improved field performance.

## 7.5 Issues associated with AS/NZS 4012/3

A key regulatory strategy for reducing solid fuel burner emissions, to date, has been to gradually tighten the allowable emission limit for appliances (as tested under AS/NZS 4012/3). The impact of operational variables outlined in this study (i.e. loading, fuel type, fuel moisture content, installation characteristics) on actual emissions suggest that this may not be sufficient to achieve the desired emission reductions.

Ideally, lower laboratory test emissions should correspond with lower real-life emissions. The results suggest that AS/NZS 4012/3 provides a good indication of inter-burner performance in the laboratory under simulated real-life operating conditions, but not of actual emissions measured in the field, when multiple variables relating to operation, fuel type and installation are introduced.

Overall, all appliances tested performed poorly in the field, relative to their laboratory-tested models. It was difficult to pinpoint exactly which variable(s) were responsible for these differences, and further work is necessary in this regard. Nevertheless, this study suggests that when mode of operation, firewood and installation characteristics vary between appliances, the AS/NZS 4012/3 test, as it currently stands, is unlikely to provide a reliable indication of performance in the field.

Although only a small number of appliances were tested, overseas research suggests that this finding is not unique. A study conducted by Fisher *et al*. (2000), also suggests that emission improvements, as demonstrated by standardised emission tests, may not necessarily be observed in practice (Figure 7-2).

The study shows that simply including locally-sourced fuel, light-up and refuelling phases at different burn rates in a laboratory setting does not necessarily provide an indication of field performance, as demonstrated by the difference in the relationships between Stage 0 and I, and 0 and III. In view of this, there is a case to be made for encouraging the development of appliances that are more accommodating of real-life variables (e.g. a range of firewood types, sizes, moisture contents; low temperature operation; and different installation characteristics).

Laboratory testing reflecting operational practices would provide greater assurance that appliances can be screened to achieve emission reductions in practice. Operational testing would provide greater assurance that air quality assessments based on laboratory tests are reliable. Further research is needed to identify the range of variables that determine an appliance’s emissions performance in the field.

## 7.6 Consideration of additional measures

It may be necessary to consider introducing additional measures to address wood burner operation in New Zealand. These could include combinations of effective, targeted educational measures, and regulation and enforcement relating to the nature of firewood sold and used, and the emission of excessive smoke from residential chimneys. These measures would focus on variations in performance identified in this study. Greater compliance monitoring may also be required with regard to the appliances available for sale. A study undertaken in Australia, for example, found that of 47 residential wood burning appliances available for sale, 55% had one or more serious design faults that could affect performance, and of 12 of these tested in the laboratory, 7 failed to meet AS/NZS 4012/3 PM emission limits (Department of Environment and Heritage, 2004). Limited shop floor audits have been conducted in New Zealand, and more work in this area may be required.

From an educational point of view, an issue is the difficulty of determining appropriate operator behaviours, which vary according to the appliances and fuels used. Todd (2003b) defines correct wood burner operation as the appropriate use of air controls, correct loading of firewood into the appliance (i.e. avoid over-loading), use of dry firewood and maintenance of the appliance. Further work in this area may be needed to develop better guidance for householders. The results of this study, and the US study mentioned in the previous section, indicate that firewood size, moisture and loading practices are important areas that must be addressed.

The development of technology and products that remove the influence of operation and fuel variations on emissions should be encouraged. For example, pellet burning appliances have increased in popularity in New Zealand. Recent emission tests conducted on a single pellet burner using a similar methodology to that adopted for Stage I of this study, measured average emissions of 0.98 g/kg (Environet Ltd, 2004). Using the same averaging method, and assuming that pellet burners are operated in a similar manner to residential wood burning appliances (in terms of burn rates), the equivalent emissions for Appliance A and B were 2.3 g/kg and 2.7 g/kg, respectively. The pellet burner used less fuel in a six-hour cycle than Appliance A and B, and Environet Ltd (2004) estimates that Appliance A and B discharged at least four times the amount of PM produced by the pellet burner, over a six-hour cycle. This demonstrates, albeit on the basis of tests conducted on a single pellet burner, that removal of fuel and operator variables as is effectively achieved with an appliance such as a pellet burner, can result in substantial emission reductions in a laboratory.

Light-up emissions were also indicated as significant during this study, and could be reduced by encouraging the use of firelighters. In addition, emissions could potentially be reduced by facilitating the development of markets for, and the production of, densified wood logs. Work in the US, for example, has demonstrated significant emission improvements when compressed wood logs are used (Houck and Tiegs, 1998). Further work on this issue is required, however, to determine whether emission reductions are achieved in practice, and if production and use of the logs are viable in New Zealand. Use of these logs could be encouraged on high pollution nights.

# 8 Conclusions

This study provides the first field performance data for low emission wood burners in New Zealand and Australia, and provides a base from which future studies may be developed. Although it was not possible for emission factors to be developed from the data, the magnitude of the emissions measured during the study suggested that the current value used to represent low emission wood burners, 3 g/kg, in some circumstances, may underestimate real-life emissions by a factor of up to 4 or 5.

A conversion factor could not be developed for estimating real-life emissions from the AS/NZS 4012/3 test emissions, as a relationship between the authorised test emissions and real-life emissions could not be established. A range of operational behaviours, firewood and installation variations impacted on the field performance of the individual appliances. To obtain reliable and representative emission factors for low emission wood burners, a comprehensive study, employing a similar methodology to that of Stage III, but encompassing a wider range of operational behaviours, installation and firewood variables is required.

The study also suggested that AS/NZS 4012/3 may not provide an adequate basis for air quality assessments of real-life emissions. A combined approach of laboratory and operational testing may provide greater assurance that appliances will achieve required emission reductions in practice.

# 9 Recommendations

- A programme is developed to provide a representative population-wide average of real-life emissions from wood burners. Similar methodology to that employed during Stage III would be required, where the tests are conducted on installed appliances and operated by the householder using their own fuel. This should include a number of pellet burners so

that emission factors could also be developed for these appliances. The emission factors could then be used for emission inventories and other air quality assessments.
- An investigation into the variables that influence field emissions is required.
- Methods to ensure better field operation could be considered such as education and encouraging the development of technologies and products that remove operational and fuel variation influences, such as pellet burners, more user-friendly wood burners, and the use of firestarters and densified wood logs.

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# Appendix 1 – Data from the 2001 New Zealand Census

| Area and Year | Fuel Type Used to Heat Dwellings | | | | | Fuel Type Used to Heat Dwellings | | | | Percent of total |
|---|---|---|---|---|---|---|---|---|---|---|
| | Electricity | Mains Gas | Bottled Gas | Wood | Coal | Solar Power | Other Fuel(s) | No Fuels Used in this Dwelling | Total Dwellings | |
| | | | | | | | | | | |
| Regional Councils | | | | | | | | | | |
| North Island | | | | | | | | | | |
| Northland Region | | | | | | | | | | |
| 1996 | 30,504 | 705 | 10,014 | 25,275 | 1,608 | 444 | 345 | 1,836 | 45,492 | 56% |
| 2001 | 29,115 | 1,098 | 14,658 | 24,696 | 1,362 | 579 | 333 | 2,394 | 47,610 | 52% |
| Auckland Region | | | | | | | | | | |
| 1996 | 275,388 | 28,794 | 61,125 | 127,083 | 31,647 | 2,559 | 2,883 | ###### | 337,551 | 38% |
| 2001 | 281,091 | 46,149 | 92,802 | 117,390 | 21,717 | 3,771 | 3,225 | ###### | 371,118 | 32% |
| Waikato Region | | | | | | | | | | |
| 1996 | 77,940 | 20,484 | 33,726 | 60,438 | 14,862 | 675 | 1,167 | 2,238 | 117,636 | 51% |
| 2001 | 75,549 | 24,615 | 40,260 | 58,680 | 10,677 | 960 | 1,359 | 3,009 | 123,945 | 47% |
| Bay of Plenty Region | | | | | | | | | | |
| 1996 | 51,864 | 4,965 | 23,892 | 39,009 | 3,795 | 693 | 783 | 1,833 | 76,941 | 51% |
| 2001 | 50,994 | 7,491 | 30,135 | 38,403 | 2,880 | 1,014 | 993 | 2,454 | 83,616 | 46% |
| Gisborne Region | | | | | | | | | | |
| 1996 | 9,438 | 2,067 | 3,759 | 9,300 | 396 | 78 | 90 | 234 | 14,421 | 64% |
| 2001 | 8,700 | 2,334 | 5,067 | 9,039 | 387 | 120 | 93 | 291 | 14,673 | 62% |
| Hawke's Bay Region | | | | | | | | | | |
| 1996 | 34,599 | 2,805 | 19,458 | 32,349 | 2,400 | 318 | 423 | 642 | 49,398 | 65% |
| 2001 | 33,228 | 3,405 | 22,266 | 32,112 | 2,094 | 489 | 456 | 762 | 50,631 | 63% |
| Taranaki Region | | | | | | | | | | |
| 1996 | 21,939 | 15,195 | 6,483 | 16,677 | 1,374 | 219 | 183 | 447 | 37,491 | 44% |
| 2001 | 19,209 | 15,774 | 7,914 | 16,620 | 1,071 | 309 | 234 | 648 | 37,710 | 44% |
| Manawatu-Wanganui Region | | | | | | | | | | |
| 1996 | 50,055 | 28,188 | 18,414 | 37,086 | 3,891 | 414 | 432 | 873 | 78,627 | 47% |
| 2001 | 46,479 | 28,173 | 22,035 | 36,855 | 3,168 | 558 | 504 | 1,221 | 79,134 | 47% |
| Wellington Region | | | | | | | | | | |
| 1996 | 117,219 | 38,148 | 31,701 | 54,855 | 10,938 | 885 | 1,173 | 1,830 | 143,970 | 38% |
| 2001 | 118,020 | 43,713 | 37,689 | 53,445 | 8,196 | 1,179 | 1,254 | 2,964 | 152,073 | 35% |
| Total, North Island | | | | | | | | | | |
| 1996 | 668,946 | 141,351 | 208,566 | 402,072 | 70,917 | 6,294 | 7,473 | ###### | 901,530 | 45% |
| 2001 | 662,388 | 172,752 | 272,823 | 387,240 | 51,549 | 8,979 | 8,451 | ###### | 960,510 | 40% |
| | 69% | 18% | 28% | 40% | 5% | 1% | 1% | 3% | | |
| South Island | | | | | | | | | | |
| Tasman Region | | | | | | | | | | |
| 1996 | 9,450 | 42 | 3,285 | 10,077 | 1,017 | 192 | 120 | 210 | 13,695 | 74% |
| 2001 | 9,675 | 75 | 4,272 | 10,647 | 768 | 246 | 159 | 243 | 15,078 | 71% |
| Nelson Region | | | | | | | | | | |
| 1996 | 12,051 | 72 | 4,443 | 8,040 | 1,413 | 174 | 138 | 234 | 14,856 | 54% |
| 2001 | 11,862 | 111 | 5,238 | 8,178 | 900 | 222 | 183 | 324 | 15,606 | 52% |
| Marlborough Region | | | | | | | | | | |
| 1996 | 10,947 | 42 | 2,940 | 9,243 | 1,668 | 141 | 144 | 114 | 13,701 | 67% |
| 2001 | 10,902 | 90 | 4,026 | 9,480 | 1,152 | 180 | 201 | 174 | 14,805 | 64% |
| West Coast Region | | | | | | | | | | |
| 1996 | 7,479 | 42 | 2,202 | 9,048 | 7,908 | 78 | 105 | 96 | 11,760 | 77% |
| 2001 | 6,723 | 75 | 2,823 | 8,868 | 7,578 | 90 | 132 | 120 | 11,577 | 77% |
| Canterbury Region | | | | | | | | | | |
| 1996 | 152,019 | 711 | 36,558 | 95,166 | 30,777 | 1,368 | 1,833 | 1,044 | 171,033 | 56% |
| 2001 | 151,791 | 1,530 | 55,716 | 92,109 | 17,367 | 1,722 | 2,688 | 1,839 | 181,740 | 51% |
| Otago Region | 84% | 1% | 31% | 51% | 10% | 1% | 1% | 1% | | |
| 1996 | 60,183 | 327 | 8,844 | 41,616 | 24,327 | 486 | 828 | 363 | 66,843 | 62% |
| 2001 | 59,001 | 546 | 14,682 | 42,471 | 21,768 | 627 | 1,149 | 561 | 68,850 | 62% |

# Appendix 2 – Test parameters

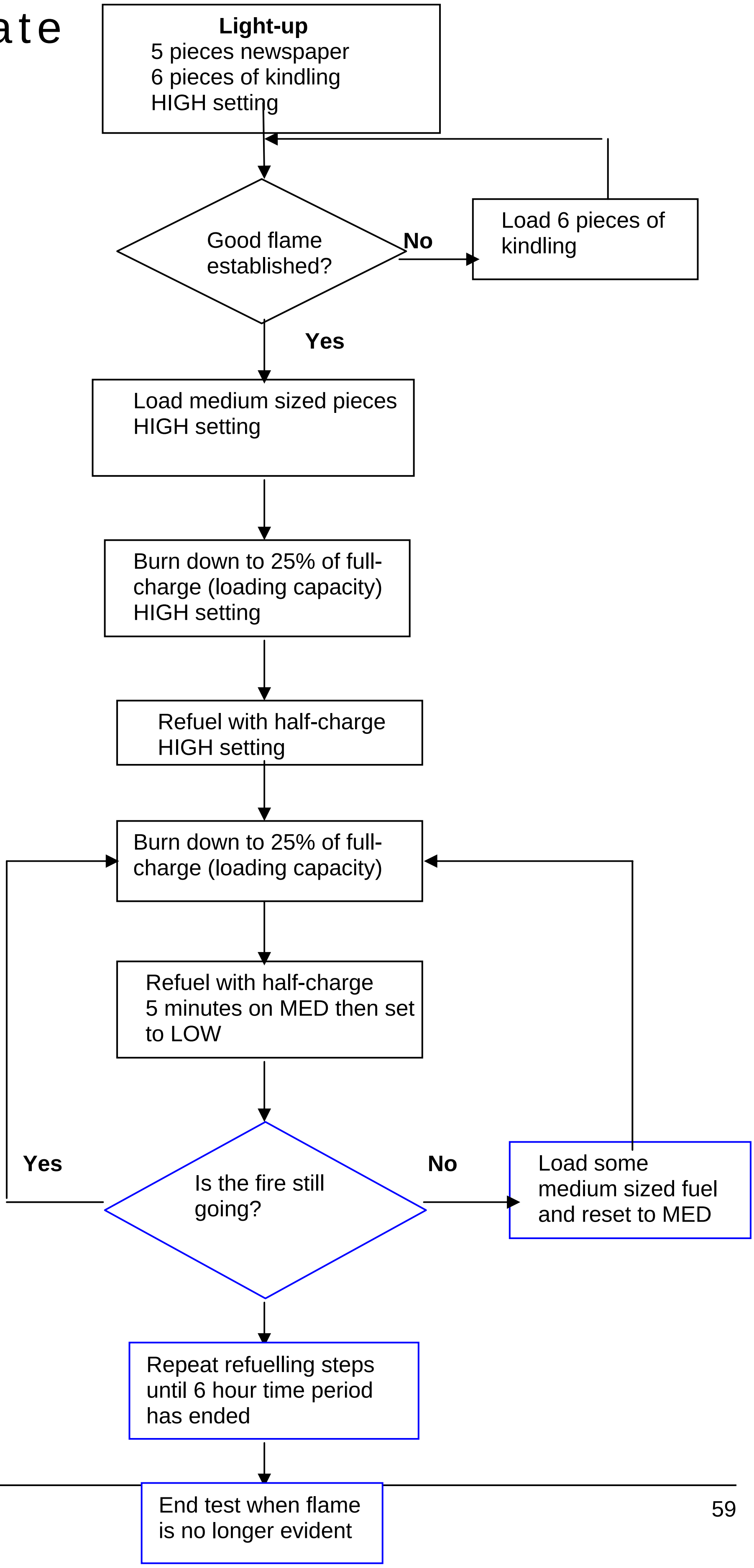
| APPLIANCE A | STAGE 0 | STAGE I | STAGE II | STAGE III |
|---|---|---|---|---|
| Number of test runs | 9 | 9 | 6 | 7 |
| Control settings (low, medium, high) | 3L, 3M, 3H | 3 L, 3 M, 3H | 3L, 3M | 18% L, 82% H |
| Flue height | 4.6 | 4.6 | 4.2 | 4.2 |
| kW rating | 7.6 | 7.6 | 7.6 | 7.6 |
| Firebox size (litres) | 30.2 | 30.2 | 30.2 | 30.2 |
| Hot water booster | N | N | N | N |
| Secondary air bar | N | N | N | N |
| Wood type | Pine | High resin pine | High resin pine | Pine - offcuts |
| Target wood loads (kg - wet weight) | 2.9 | 1.5 | 1.5 | 1.3 |
| Kindling number | | 6 | 6 | |
| Kindling size (kg - wet weight) | | 0.25 | 0.25 | |
| Medium log number | 3 | 2 | 2 | 1-2 |
| Medium/normal log size (kg - wet weight) | | 0.75 | 0.75 | |
| Average moisture content (%) | 16-20 | 15 | 14.9 | 16 |
| Average fuel consumption (Total dry kg/hr) | 2.3 | 2.1 | 1.3 | 0.9 |
| Average run time | 78 | 360 | 360 | 348 |
| Average emissions g/kg | 0.6 | 2.3 | 2.7 | 11.9 |
| Average emissions g/hr | 1.2 | 4.8 | 4.2 | 9.7 |
| **APPLIANCE B** | **STAGE 0** | **STAGE I** | **STAGE II** | **STAGE III** |
| Number of test runs | 9 | 9 | 6 | 7 |
| Control settings (low, medium, high) | 3L, 3M, 3H | 3L, 3M, 3H | 3L, 3M | 4%L, 12%M, 84%H |
| Flue height | 4.6 | 4.6 | 4.8 | 4.8 |
| kW rating | 9.1 | 9.1 | 9.1 | 9.1 |
| Firebox size (litres) | 49.2 | 49.2 | 49.2 | 49.2 |
| Hot water booster | N | N | N | N |
| Secondary air bar | N | N | N | N |
| Wood type | Pine | High resin pine | High resin pine | Oregon - split logs |
| Target wood loads (kg - wet weight) | 4.6 | 2.8 | 2.8 | 2.7 |
| Kindling number | | 6 | 6 | |
| Kindling size (kg - wet weight) | | 0.46 | 0.46 | |
| Medium log number | 4 | 2-3 | 2-3 | |
| Medium/normal log size (kg - wet weight) | | 0.93-1.4 | 0.93-1.4 | 1-2 |
| Average moisture content (%) | 16-20 | 14 | 15.4 | 19 |
| Average fuel consumption (Total dry kg/hr) | 3.1 | 2.8 | 1.9 | 2.5 |
| Average run time | 96 | 360 | 360 | 427 |
| Average emissions g/kg | 1.1 | 2.7 | 2.3 | 18.2 |
| Average emissions g/hr | 3.2 | 7.7 | 5 | 41.7 |

| APPLIANCE C | STAGE 0 | STAGE I | STAGE II | STAGE III |
|---|---|---|---|---|
| Number of test runs | 9 | 6 | 6 | 8 |
| Comment | 1.2 g/kg appliance | 1.7 g/kg appliance | 1.2 g/kg appliance | 1.2 g/kg appliance |
| Control settings (low, medium, high) | 3L, 3M, 3H | 3L, 3M | 3L, 3M | 48%L, 13%M, 39%H |
| Flue height | 4.6 | 4.6 | 7.5 m | 7.5 m |
| kW rating | 9 | 10.4 | 9 | 9 |
| Firebox size (litres) | 45.6 | 56.3 | 45.6 | 45.6 |
| Hot water booster | Y | N | Y | Y |
| Secondary air bar | N | N | Y | Y |
| Wood type | Pine | Medium resin pine | Medium resin pine | Eucalypt - split logs |
| Target wood loads (kg - wet weight) | 4.3 | 2.2 | 2.2 | 3.2 |
| Kindling number | | 6 | 6 | |
| Kindling size (kg - wet weight) | | 0.38 | 0.38 | |
| Medium log number | 4 | 2-3 | 2-3 | 1-2 |
| Medium/normal log size (kg - wet weight) | | 0.75-1.3 | 0.75-1.3 | |
| Average moisture content (%) | 16-20 | 20.7 | 14.5 | 21 |
| Average fuel consumption (Total dry kg/hr) | 3.3 | 2.1 | 1.8 | 1.5 |
| Average run time | 101 | 360 | 360 | 647 |
| Average emissions g/kg | 1.2 | 3.6 | 4.7 | 6.9 |
| Average emissions g/hr | 3.2 | 7.5 | 9.6 | 10.3 |
| **APPLIANCE F** | **STAGE 0** | **STAGE I** | **STAGE II** | **STAGE III** |
| Number of test runs | 9 | | | 7 |
| Control settings (low, medium, high) | 3L, 3M, 3H | | | 100%H |
| Flue height | 4.6 | | | 4 |
| kW rating | 7.1 | | | 7.1 |
| Firebox size (litres) | 33.3 | | | 33.3 |
| Hot water booster | N | | | N |
| Secondary air bar | N | | | N |
| Wood type | Pine | | | Pine - split logs |
| Target wood loads (kg - wet weight) | 3.1 | | | 1.8 |
| Kindling number | | | | |
| Kindling size (kg - wet weight) | | | | |
| Medium log number | 5 | | | 1 |
| Medium/normal log size (kg - wet weight) | | | | |
| Average moisture content (%) | 16-20 | | | 25 |
| Average fuel consumption (Total dry kg/hr) | 2.5 | | | 1.3 |
| Average run time | 78 | | | 223 |
| Average emissions g/kg | 0.9 | | | 26.1 |
| Average emissions g/hr | 1.9 | | | 36.7 |

| APPLIANCE D | STAGE 0 | STAGE I | STAGE II | STAGE III |
|---|---|---|---|---|
| Number of test runs | 9 | 6 | 6 | 7 |
| Comment | | | | Same as Stage II but different house |
| Control settings (low, medium, high) | 3L, 3M, 3H | 3L, 3M | 3L, 3M | 3.8 |
| Flue height | 4.6 | 4.6 | ?? | 36%L, 30%M, 33%H |
| kW rating | 6.8 | 6.8 | 6.8 | 6.8 |
| Firebox size (litres) | 40.1 | 40.1 | 40.1 | 40.1 |
| Hot water booster | Y | N | Y | Y |
| Secondary air bar | N | N | N | N |
| Wood type | Pine | High resin pine | Medium resin pine | Pine and native timbers - split logs |
| Target wood loads (kg - wet weight) | 3.8 | 1.7 | 1.7 | 2.3 |
| Kindling number | | 6 | 6 | |
| Kindling size (kg - wet weight) | | 0.29 | 0.29 | |
| Medium log number | 4 | 2 | 2 | 1-2 |
| Medium/normal log size (kg - wet weight) | | 0.87 | 0.87 | |
| Average moisture content (%) | 16-20 | 14 | 13.6 | 21 |
| Average fuel consumption (Total dry kg/hr) | 2.2 | 2.2 | 1.6 | 1.4 |
| Average run time | 113 | 360 | 360 | 411 |
| Average emissions g/kg | 3.1 | 4.7 | 1.5 | 14.3 |
| Average emissions g/hr | 6 | 9.8 | 2.8 | 20 |
| **APPLIANCE E** | **STAGE 0** | **STAGE I** | **STAGE II** | **STAGE III** |
| Number of test runs | 9 | 6 | 6 | 7 |
| Comment | 1.8 g/kg appliance | Uncertain - either 1.8 g/kg or 1.3 g/kg appliance | Different to Stage I model - Uncertain - either 1.8 g/kg or 1.3 g/kg appliance | Same Stage II appliance |
| Control settings (low, medium, high) | 3L, 3M, 3H | 3L, 3M | 3L, 3M | 100%H |
| Flue height | 4.6 | 4.6 | ?? | 3.5 |
| kW rating | 7.6 | uncertain | uncertain | uncertain |
| Firebox size (litres) | uncertain | 33.3 | uncertain | uncertain |
| Hot water booster | N | N | N | N |
| Secondary air bar | N | N | N | N |
| Wood type | Pine | Medium resin pine | Medium resin pine | Pine and silver birch - split logs |
| Target wood loads (kg - wet weight) | 3.9 | 1.9 | 1.9 | 1.5 |
| Kindling number | | 6 | 6 | |
| Kindling size (kg - wet weight) | | 0.32 | 0.32 | |
| Medium log number | 4 | 2 | 2 | 1-2 |
| Medium/normal log | | 0.95 | 0.95 | |

| | | | | |
|---|---|---|---|---|
| size (kg - wet weight) | | | | |
| Average moisture content (%) | 17 | 20 | 19.2 | 16 |
| Average fuel consumption (Total dry kg/hr) | 2 | 2.2 | 1.4 | 1.2 |
| Average run time | 103 | 360 | 360 | 360 |
| Average emissions g/kg | 1.8 | 3 | 3.8 | 6.6 |
| Average emissions g/hr | 3.3 | 6.4 | 6.1 | 8 |

# Appendix 3 – Example firing procedure – Low burn rate



# Appendix 4 - Test data summary

| Low emission wood burners | PM emissions (g/kg) | | | | PM emissions (g/hr) | | | |
|---|---|---|---|---|---|---|---|---|
| | Stage 0 | Stage I | Stage II | Stage III | Stage 0 | Stage I | Stage II | Stage III |
| **Appliance A** | 0.5 | 2.5 | 2.9 | 24.5 | 1.1 | 4.8 | 3.9 | 19.1 |
| | 1.1 | 3.3 | 3.1 | 13.0 | 2.0 | 6.2 | 5.1 | 7.7 |
| | 0.5 | 1.3 | 2.4 | 6.9 | 0.9 | 2.5 | 3.8 | 5.7 |
| | 0.6 | 2.3 | 2.6 | 10.7 | 1.4 | 4.8 | 4.2 | 6.4 |
| | 0.3 | 3.2 | 2.5 | 5.3 | 0.7 | 7.2 | 4.1 | 10.2 |
| | 0.6 | 1.4 | 2.7 | 14.6 | 1.4 | 2.6 | 4.2 | 8.2 |
| | 0.4 | 2.9 | | 8.3 | 1.1 | 7.4 | | 10.4 |
| | 0.5 | 2.3 | | | 1.3 | 5.6 | | |
| | 0.5 | 1.1 | | | 1.3 | 2.2 | | |
| Average | 0.6 | 2.3 | 2.7 | 11.9 | 1.2 | 4.8 | 4.2 | 9.7 |
| Standard deviation | 0.2 | 0.8 | 0.3 | 6.4 | 0.4 | 2.0 | 0.5 | 4.5 |
| **Appliance B** | 1.6 | 2.7 | 3.5 | 9.2 | 3.5 | 8.2 | 7.0 | 53.5 |
| | 2.1 | 3.3 | 1.8 | 32.6 | 4.6 | 8.0 | 3.9 | 85.2 |
| | 1.3 | 3.3 | 3.3 | 22.0 | 3.0 | 7.9 | 7.1 | 46.8 |
| | 0.9 | 2.2 | 1.3 | 15.6 | 2.9 | 6.1 | 3.0 | 27.4 |
| | 1.2 | 3.0 | 1.3 | 23.0 | 3.2 | 8.1 | 3.2 | 38.5 |
| | 0.9 | 1.4 | 2.3 | 13.8 | 2.8 | 3.3 | 6.0 | 22.1 |
| | 0.9 | 2.7 | | 11.3 | 3.9 | 9.2 | | 18.7 |
| | 0.6 | 3.8 | | | 2.3 | 12.1 | | |
| | 0.8 | 2.2 | | | 3.0 | 6.5 | | |
| Average | 1.1 | 2.7 | 2.3 | 18.2 | 3.2 | 7.7 | 5.0 | 41.7 |
| Standard deviation | 0.5 | 0.7 | 1.0 | 8.1 | 0.7 | 2.4 | 1.9 | 23.1 |
| **Appliance C** | 1.0 | | 8.4 | 7.7 | 2.1 | | 16.8 | 11.9 |
| | 2.7 | | 2.6 | 6.1 | 5.7 | | 5.2 | 11.0 |
| | 2.6 | | 4.9 | 6.3 | 5.7 | | 9.8 | 9.2 |
| | 0.9 | | 2.6 | 7.1 | 2.5 | | 6.7 | 11.4 |
| | 1.5 | | 4.8 | 10.4 | 4.4 | | 10.9 | 14.2 |
| | 0.8 | | 5.0 | 1.9 | 2.2 | | 8.5 | 2.7 |
| | 0.6 | | | 13.0 | 2.2 | | | 17.5 |
| | 0.6 | | | 2.8 | 2.1 | | | 4.6 |
| | 0.5 | | | | 1.8 | | | |
| Average | 1.2 | | 4.7 | 6.9 | 3.2 | | 9.6 | 10.3 |
| Standard deviation | 0.9 | | 2.1 | 3.6 | 1.6 | | 4.1 | 4.8 |
| **Appliance F** | 1.5 | | | 13.1 | 2.9 | | | 13.4 |
| | 0.6 | | | 25.4 | 1.1 | | | 32.0 |
| | 0.9 | | | 27.4 | 1.8 | | | 37.9 |
| | 1.0 | | | 35.2 | 2.4 | | | 40.4 |
| | 1.0 | | | 29.2 | 2.5 | | | 26.1 |
| | 0.8 | | | 36.0 | 1.8 | | | 84.2 |
| | 0.6 | | | 16.2 | 1.2 | | | 22.6 |
| | 0.6 | | | | 1.2 | | | |
| | 0.7 | | | | 2.3 | | | |
| Average | 0.9 | | | 26.1 | 1.9 | | | 36.7 |
| Standard deviation | 0.3 | | | 8.8 | 0.6 | | | 22.9 |
| **Mean** | **1.0** | **2.5** | **3.2** | **15.5** | **2.4** | **6.2** | **6.3** | **24.1** |
| **Median** | **0.8** | **2.6** | **2.7** | **13.0** | **2.2** | **6.3** | **5.1** | **17.5** |
| **Minimum** | **0.3** | **1.1** | **1.3** | **1.9** | **0.7** | **2.2** | **3.0** | **2.7** |
| **Maximum** | **2.7** | **3.8** | **8.4** | **36.0** | **5.7** | **12.1** | **16.8** | **85.2** |
| **Standard deviation** | **0.6** | **0.8** | **1.7** | **9.9** | **1.3** | **2.6** | **3.5** | **21.4** |

| Additional wood burners | PM emissions (g/kg) | | | | PM emissions (g/hr) | | | |
|---|---|---|---|---|---|---|---|---|
| | Stage 0 | Stage I | Stage II | Stage III | Stage 0 | Stage I | Stage II | Stage III |
| Appliance C (1.7 g/kg) | 1.8 | 4.4 | | | 4.3 | 9.1 | | |
| | 2.2 | 4.3 | | | 5.5 | 8.4 | | |
| | 1.5 | 3.8 | | | 4.1 | 6.9 | | |
| | 1.5 | 3.0 | | | 4.4 | 6.8 | | |
| | 2.2 | 2.9 | | | 7.9 | 6.9 | | |
| | 2.0 | 3.3 | | | 6.2 | 7.1 | | |
| | ?? | | | | ?? | | | |
| | ?? | | | | ?? | | | |
| | ?? | | | | ?? | | | |
| Average | 1.9 | 3.6 | | | 5.4 | 7.5 | | |
| Standard deviation | 0.3 | 0.6 | | | 1.5 | 1.0 | | |
| Appliance E | | 3.6 | 4.6 | 6.2 | | 7.8 | 7.6 | 7.9 |
| | | 4.9 | 7.7 | 5.0 | | 10.3 | 10.3 | 6.5 |
| | | 3.1 | 6.8 | 7.0 | | 6.0 | 11.5 | 8.4 |
| | | 2.3 | 0.8 | 6.1 | | 5.4 | 1.4 | 11.7 |
| | | 1.7 | 1.3 | 8.1 | | 4.0 | 2.1 | 7.4 |
| | | 2.2 | 1.8 | 6.2 | | 4.8 | 3.3 | 4.8 |
| | | | | 7.5 | | | | 9.6 |
| Average | | 3.0 | 3.8 | 6.6 | | 6.4 | 6.1 | 8.0 |
| Standard deviation | | 1.2 | 3.0 | 1.0 | | 2.3 | 4.3 | 2.2 |
| Appliance D | 6.9 | 4.1 | 1.3 | 14.4 | 9.7 | 7.5 | 2.0 | 17.6 |
| | 3.0 | 6.5 | 2.8 | 11.0 | 4.2 | 12.0 | 4.9 | 12.6 |
| | 5.5 | 2.3 | 1.2 | 15.0 | 8.8 | 4.3 | 2.3 | 21.0 |
| | 2.1 | 7.3 | 1.9 | 11.1 | 4.6 | 15.4 | 3.4 | 27.0 |
| | 3.1 | 6.4 | 0.9 | 14.5 | 6.2 | 15.2 | 1.7 | 20.0 |
| | 1.7 | 1.9 | 1.1 | 23.8 | 3.7 | 4.2 | 2.3 | 27.6 |
| | 2.5 | | | 10.2 | 8.3 | | | 14.0 |
| | 1.3 | | | | 3.6 | | | |
| | 2.1 | | | | 5.3 | | | |
| Average | 3.1 | 4.7 | 1.5 | 14.3 | 6.0 | 9.8 | 2.8 | 20.0 |
| Standard deviation | 1.9 | 2.3 | 0.7 | 4.6 | 2.3 | 5.1 | 1.2 | 5.9 |

# Appendix 5 – Fuel variation test results

| Variables | Appliance A | | Appliance B | |
|---|---|---|---|---|
| | g/kg | g/hr | g/kg | g/hr |
| Large high resin pine | 4.9 | 9.4 | 2.6 | 6.1 |
| | 5.8 | 11.9 | 3.1 | 7.4 |
| | 4.7 | 9.4 | 7.0 | 15.9 |
| Wet high resin pine | 6.3 | 12.5 | 4.9 | 11.8 |
| | 4.1 | 9.1 | 3.2 | 7.4 |
| | 4.8 | 9.7 | 4.0 | 9.4 |
| Dry pine offcuts | 1.7 | 4.2 | 1.6 | 5.3 |
| | 1.6 | 3.7 | 1.6 | 5.1 |
| | 1.8 | 4.5 | 1.7 | 5.4 |
| Eucalypt | 1.4 | 3.0 | 1.8 | 5.0 |
| | 3.1 | 6.5 | 3.4 | 8.9 |
| | 2.1 | 4.4 | 2.2 | 6.4 |
| Low resin pine | 1.5 | 3.2 | | |
| | 1.3 | 2.7 | | |
| | 1.1 | 2.7 | | |
| Oregon | 1.7 | 3.0 | | |
| | 1.2 | 2.8 | | |
| | 1.3 | 2.8 | | |
| Macrocarpa | 2.6 | 4.6 | | |
| | 2.3 | 4.8 | | |
| | 2.0 | 4.5 | | |

# Appendix 6 – Stage-by-stage conversion

The following methodology is only appropriate where a sufficient number of wood burners have been tested and the appliances and their installation are identical between Stages 0, I and II. In addition, information is also required regarding "typical" wood burner operation and fuels used, and the impacts of these on emissions.

1. Calculate the "raw" emission value for a particular low emission wood burner (base case)

Import all available Stage 0, I and II testing data for low emission wood burners into a statistical package such as Statistica 6. Compile matrix and scatter plots of Stage 0 and I data, and Stage I and II data. Review the $r^2$ values and regression lines and if the $r^2$ values are greater than 0.8, the regression equations may be used to convert the Stage 0 data for the appliance to Stage I and II equivalents. The Stage II equivalent forms the "raw" emission value for this appliance and may be used to estimate real-life emissions from this appliance for other parts of New Zealand.

To ensure that the emission value is representative of wood type, size, moisture and typical appliance operation in Christchurch, or any other area, it must be adjusted by statistics collected on those variables in the area of concern. The final value should then be evaluated against the field emission tests to determine whether the conversion and weighting process produces realistic numbers.

2. Calculating the real-life estimate

This is based on emission testing results and a household survey. During Stage I and II, several variables were kept constant including log size (0.5 kg), wood moisture (15% wet weight), wood type (high resin pine) and low and medium burn rates. The "raw" emission value (base case) is reliant on these variables. The value may be adjusted to a Stage III equivalent by (1) determining the impact of different variables on emissions (conducted during Stage I), and (2) determining how many households use different operating and firewood variables (by household survey).

During Stage I, the impact of a number of variables on emissions from Appliance A and B were evaluated. These included burn rates, firewood type, size and moisture content. The relative difference between emissions produced under these conditions and base case was determined. Table 1 presents the proportional impact of these variables on emissions, as measured during this study. This assumes that the relative differences will be maintained regardless of appliance used.

A householder survey was also conducted during the study to determine the proportion of households in Christchurch using each variable type. These percentages are presented in Table 1. These values, may be used to calculate the proportional increase in emissions based on a variety of average operating behaviours and fuel variations in Christchurch.

**Table 1 Impact of operator and fuel variables on emissions and the proportion of householders in Christchurch operating in accordance with these variables**

| Characteristics | Impact of each variable on emissions - relative change from base case | Proportion of households using each variable type - "Typical" operation[1] |
|---|---|---|
| Main wood burner setting | | |
| Low | +36% | 30% |
| Medium | 0% | 44% |
| High | +22% | 26% |
| Wood size | | |
| Large wood (1.5 to 3 kg logs) | +147% | 33% |
| Small wood (<1.5 kg logs) | 0% | 67% |
| Wood moisture | | |
| Wet wood (~27% wet weight) | +140% | 1% |
| Dry wood (~11% wet weight) | -12% | 91% |
| Moist wood (11<>27% wet weight) | 0% | 8% |
| Wood moisture | | |
| Gum (eucalypt) | +23% | 16% |
| Oregon (Douglas fir) | -26% | 16% |
| Macrocarpa | +21% | 21% |
| Pine | 0% | 47% |

[1]Where there were other categories in the survey that were not tested for during the study the proportion of households were divided up evenly between those that were tested

It was estimated that typical operation and fuel variations in Christchurch would result in a 41% increase in base case emissions. On this basis, a raw emission value of 4.7 g/hr, for example, would increase to 6.6 g/hr. This constitutes the Stage III equivalent values weighted by average wood burner operation and installations in Christchurch.