Impacts of smoke from regeneration burning on air quality in the Huon Valley, Tasmania

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LIST OF ABBREVIATIONS

AAQ  Ambient Air Quality
DPI  Department of Primary Industries
EC  Elemental carbon
ED  Exposure Duration
EF  Emission factor
FT  Forestry Tasmania
NEPC  National Environmental Protection Council
NEPM  National Environmental Protection Measure
nssK+  non-sea-salt Potassium
OC  Organic carbon
PB  Prescribed burn
PB season  1 March to 30 April
PM  Particulate Matter
PM$_{10}$  Particulate Matter less than 10 micrometers in diameter
PM$_{2.5}$  Particulate Matter less than 2.5 micrometers in diameter
Tas  Tasmania
µg  microgram (10$^{-6}$ g)
WH  Wood-fuelled heater
WH season  15 May to 15 September
1. **SUMMARY**

The fine particulate matter (PM) in smoke is a well known health risk and subject to regulation by government. Elevated ambient PM concentrations are common in urban areas and are usually associated with emissions from vehicles and industry. In rural areas, air quality is generally good, however there are occasions when this is not the case. Rural pollution events mostly result from domestic, agricultural or forestry activities, and often involve smoke from biomass combustion. In the Huon Valley, Tasmania, smoke from prescribed burning has been subject to public debate. The commonly accepted view is that regeneration burning following logging operations is the major source of PM pollution; however, in the absence of reliable ambient air quality monitoring it is impossible to confirm the veracity of this perception. This study was designed to address the issue.

The major anthropogenic sources of PM in the Huon Valley are prescribed burning (PB), domestic wood-fuelled heaters (WH), windblown dust from roads, motor vehicles, and domestic waste incineration. To determine the contribution of these sources to the ambient surface concentration of PM in the Valley, two air quality monitoring stations were installed; one at a rural site, the Department of Primary Industry Research Station near Grove; and one in an urban area, Geeveston. The rural site was expected to be affected mostly by PB while the urban site was expected to be influenced by all anthropogenic PM sources. Ambient surface PM concentration was monitored continuously between March 2009 and November 2010; a period that includes the PB and WH seasons of both years, and the intervening summer when biomass combustion sources are minimal. The PB season is March and April, and the WH season extends from May to September.

Air quality at Grove was good. The mean annual PM$_{2.5}$ concentration was $6.2 \pm 1.5 \mu g \, m^{-3}$, there were no exceedences of the PM$_{10}$ Ambient Air Quality (AAQ) National Environment Protection Measures (NEPM) and only one exceedence of the advisory PM$_{2.5}$ NEPM. Although impacts from both PB and WH were detected, none was significant. Impacts from PB were associated with episodic and infrequent plume strikes. In contrast, increases in PM concentration due to WH emissions occurred regularly on most evenings during the winter. Maximum daily PM$_{2.5}$ concentrations in the 2009 and 2010 PB seasons were 10.2 and 15.9 $\mu g \, m^{-3}$ respectively; and in the WH seasons they were 15.0 and 12.4 $\mu g \, m^{-3}$. Anthropogenic source contribution to mean seasonal PM$_{2.5}$ concentrations was approximately 5% in the PB season and 4% and 24% in the 2009 and 2010 WH seasons, respectively.

Air quality in Geeveston was marginal. During the 20 month study, the 24 h PM$_{10}$ NEPM was exceeded on 8 occasions and the advisory 24 h PM$_{2.5}$ NEPM standard was exceeded on 99 occasions. Annual average PM$_{2.5}$ concentration was 12.7 $\mu g \, m^{-3}$, 71 % higher than the summer (i.e. background air) concentration of 7.2 $\mu g \, m^{-3}$. Plume strikes during the PB season were more frequent than at the rural site but most had minor impact. However, there were three significant PB events in 2009 each of approximately 50 h duration with mean PM$_{2.5}$ concentrations of 24, 25 and 40 $\mu g \, m^{-3}$, and one in 2010 that persisted for 162 h with a mean PM$_{2.5}$ concentration of 39 $\mu g \, m^{-3}$. The maximum daily PM$_{2.5}$ concentrations during the PB seasons were 44 and 71 $\mu g \, m^{-3}$ in 2009 and 2010. The WH season was persistently polluted. Average WH season PM$_{2.5}$ concentrations were 20 and 24 $\mu g \, m^{-3}$ in 2009 and 2010 respectively; and the mean daily maxima were 42 and 49 $\mu g \, m^{-3}$.

Therefore, although regionally the Huon Valley airshed is not polluted, there are locations within it that are severely impacted by smoke from anthropogenic biomass combustion. Contrary to public perception, most of the pollution in Geeveston was from wood-fuelled
combustion stoves within the urban boundary. During the 20 month study WH emissions were responsible for 77% of the anthropogenic PM pollution; compared to 11% from smoke plumes from PB activities, 4% from waste combustion and 8% from other sources.

Chemical analysis of PM confirmed that most anthropogenic PM was formed during biomass combustion. During the PB and WH seasons 95% of anthropogenic PM mass was PM$_{2.5}$ with a mass mode at 1μm. It was composed almost entirely of organic carbon and elemental carbon of which approximately 30% was one compound, the anhydrous sugar levoglucosan. Elemental carbon was 5 - 15% of total organic mass. Particle number density in air impacted by these sources was maximum between 100 and 200 nm. In contrast 38% of PM mass in background aerosol was in the 2.5 - 10 μm size fraction and was predominantly sea-salt. There was little evidence of significant amounts of PM from other anthropogenic sources in the PM$_{2.5}$ aerosol; there was little unaccounted mass that would indicate the presence of insoluble inorganic material (wind-blown dust); nor were there substantial quantities of elemental carbon that would indicate PM from vehicle emissions.

The potential impact on local residents of the high PM concentrations during the PB and WH season was assessed. WH pollution is largely a night-time issue in contrast to PB events which can also occur during the day. Due to the long persistence of high PM concentrations at night, indoor PM concentrations are unlikely to be substantially lower than outdoor ones. However plume strikes are usually of shorter duration and therefore it may be possible to avoid exposure to PB events by remaining indoors.

If health risk from exposure to PM is both dose and time dependent then the time course of PM concentration during pollution events will be as important as total average exposure. Comparison of the frequency distribution of mean hourly and mean daily PM concentrations indicate that dose/time related health risks will be more severe in the WH season than in the PB season.

Inventory estimates of annual PM emissions were at variance with the observed impact of WH and PB emissions on surface PM concentrations. Total annual PM emissions from PB events (8900 t/y) were 70 times larger than total annual emissions from wood-fuelled heaters (120 t/y) yet the impact of PB events on surface PM concentration was 1/7 of the impacts from WH. This difference is consistent with the modelled patterns of ventilation and dispersion in the airshed. PM from wood-fuelled heaters is emitted in the surface layer, predominantly at night when the mixing depth and wind speed are low, and consequently accumulates until morning when convective mixing of the lower atmosphere increases causing the PM to disperse. Prescribed burning events occur during the day when the ventilation rates are roughly 100 to 10,000 times larger than at night. The impacts of WH emissions in the urban area are therefore highly local, regular and persistent while impacts from PB events are highly episodic and transient. The severe PB pollution event in 2010 was associated with an atypically low daytime ventilation rate.

Current policy to manage the impacts from PB events on surface PM and associated health appears to be relatively effective. There was no evidence of widespread impact in the airshed, and localised impacts observed at Geeveston were small in comparison to other pollution sources. However there is potential for improvement to both the prescriptions and their implementation. It is possible that the surface impacts from PB events are associated not with the main smoke plume, which usually occurs when ventilation in the mixed layer is large and the buoyancy of the plume is sufficient to inject much of the emitted PM into the free troposphere, but with the smouldering-phase combustion which occurs at the end of the burn in the late afternoon and evening when mixing depths and ventilation rates are smaller and...
dispersion is reduced. Detailed characterisation of the plume behaviour and dispersion is required to develop and test atmospheric dispersion models that could assist in optimising the current prescription. Also, careful implementation of the current policy, particularly to exclude days when ventilation rate is marginal for good dispersion, could reduce the risk of plume strike on residential areas.

2. **SCOPE**

This document reports the results of the project “Impacts of smoke from regeneration burning on air quality in the Huon Valley, Tasmania” undertaken jointly between CSIRO Marine and Atmospheric Research and Forestry Tasmania through the Collaborative Research Agreement of 25\textsuperscript{th} June 2009.

3. **BACKGROUND**

Biomass burning is a major source of atmospheric pollutants. Its significance for air quality in urban jurisdictions has been studied for many years, and in many instances this has led to its regulation. In Australia, the focus has been primarily on the air quality of cities, however the impacts of smoke on rural air quality is also attracting attention (Meyer et al., 2008b; Reisen et al., 2011). Of the many chemical species that are emitted through wood combustion, the main component of concern has been particulate matter (PM). Most of the other toxic compounds emitted do not accumulate in concentrations that cause measurable health impacts; nor, with the exception of ozone, do secondary photochemical reactions produce gases in toxic concentrations. Even people whose occupations bring a risk of exposure to high wood smoke concentrations, such as rural fire-fighters, are rarely exposed to hazardous concentrations of compounds other than particulates (Reisen and Brown, 2006). However, the health risks from particulates are well known, and can be substantial. While larger particles deposit to the ground within hours, smaller particles can remain suspended in the atmosphere for days or weeks, and when inhaled, can cause acute and chronic health effects (Brunekreef and Holgate, 2002; Mazzoli-Rocha et al., 2010; Pope and Dockery, 2006). This study focuses solely on air quality with respect to particulates.

In Australia air quality is regulated through the Environment Protection Measures for Ambient Air Quality (AAQ NEPM), established by the National Environment Protection Council in 2003 (National Environmental Protection Council, 2003). The guidelines relevant to this report are summarised in Table 1. The NEPM standard for particulate matter with aerodynamic diameter less than 10 μm (PM\textsubscript{10}) is mandatory; however, currently the NEPM standard particulate matter with aerodynamic diameter less than 2.5 μm (PM\textsubscript{2.5}) is advisory. It was developed specifically to address the health risks resulting from fine particles, which unlike coarse particles, are not filtered by the nose and upper respiratory tract but are inspired deep within the lungs and, therefore, pose the greater risk to health. It is the standard directly relevant to this study.
Table 1  National Environmental Protection Measures (NEPMs) for ambient air quality relevant to this study

<table>
<thead>
<tr>
<th>Air Contaminant</th>
<th>Average Period</th>
<th>Maximum concentration</th>
<th>Maximum allowable exceedences</th>
</tr>
</thead>
<tbody>
<tr>
<td>Particles as PM$_{10}$</td>
<td>1 day</td>
<td>50 µg m$^{-3}$</td>
<td>5 days per year</td>
</tr>
<tr>
<td>Particles as PM$_{2.5}$</td>
<td>1 day</td>
<td>25 µg m$^{-3}$</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1 year</td>
<td>8 µg m$^{-3}$</td>
<td></td>
</tr>
<tr>
<td>Photochemical oxidant (as ozone)</td>
<td>1 hour</td>
<td>100 ppb</td>
<td>1 day per year</td>
</tr>
<tr>
<td></td>
<td>4 hours</td>
<td>80 ppb</td>
<td>1 day per year</td>
</tr>
</tbody>
</table>

Sources that contribute particulates to the air are numerous and the composition of the aerosol is varied. Constituents include sea-salt from marine aerosols, other inorganic and organic salts that are the endpoint of some atmospheric chemistry pathways (e.g. nitrate, ammonia and sulphates), insoluble inorganic mineral compounds particularly from wind-blown dust, elemental carbon and a wide range of organic compounds. There is also a wide range of sources, including marine, biogenic and anthropogenic. Among the anthropogenic class are industrial emissions, emissions from vehicles, particularly diesel-powered vehicles, and a wide range of biomass combustion sources including managed and unmanaged fires in natural ecosystems, agricultural waste burning, industrial furnaces, and domestic heating and cooking (wood-fuelled stoves). Distinguishing between the contributions of these sources can be difficult, particularly if PM concentrations are monitored by a single metric such as particle mass or number. Useful discriminating properties include physical and chemical properties of the aerosol particularly particle size and composition. Seasonal and diurnal variation in source emission rate, which is often reflected in variation in PM concentration, can also be useful. Wood-fuelled heaters, for example, are used mainly in the colder months and mostly at night or in the morning (Meyer et al., 2008a) in contrast to vehicular emissions that are maximum at the peak traffic times and vary little with season, or prescribed burning in forests, which occurs in a well-defined season mostly with documented occurrence and location.

The atmospheric PM concentrations that result from these emissions are determined by meteorology and dispersion. The two features of particular importance are (1) the strength and height of the inversion layer that separates the near-surface atmospheric layer (commonly called the mixed layer) from the free troposphere and (2) wind speed. The height of the inversion layer defines the thickness of the mixed layer into which PM is emitted. Typically it is lowest at night (<200m) increasing in depth as solar radiation drives convection to a maximum (1000 - 1500 m) in the afternoon. Wind speed determines the ventilation rate through this layer. A combination of low mixing height and low wind speed, which often occurs at night, can lead to large localised PM concentrations from comparatively small emission sources, while much larger emissions from daytime sources into a deeper and more ventilated mixed layer can lead to smaller PM concentrations that are rapidly dispersed. The severe pollution caused in Launceston by wood-fuelled heater emissions into a small, inversion-capped airshed is well documented and understood (Gras et al., 2001; Luhar et al., 2006; Meyer et al., 2008a). However, the impact on surface concentration of smoke pollutants from much larger sources, such as prescribed burning in forests can be more difficult to predict. Ground-level emissions

Impacts of smoke from regeneration burning on air quality in the Huon Valley, Tasmania
from forest fires sometimes behave as diffuse local emission sources while at other times the smoke is dispersed in a coherent plume impacting the ground along its path. For example, Reisen et al. (2011) found that prescribed burning in the Ovens Valley produced high surface concentrations of PM close to the source, while fuel-reduction burning in SW Western Australia produced plumes that affected the regional centre of Manjimup at substantial distances from the points of emission. Large or intense fires can produce plumes of sufficient buoyancy to rise through the surface boundary layer into the free troposphere, or, in extreme cases, into the stratosphere. In these cases, the smoke disperses with limited impacts on PM concentration at the surface. Hence, prediction of the impact of smoke from forest fires on the surface air quality in the surrounding region usually requires substantial knowledge of both the fire behaviour and local meteorology.

Regional pollution from biomass burning has become an issue of public concern in Tasmania and on occasions has been the topic of extensive news commentaries in the press, radio and television. Smoke plumes are frequently visible at large distance, and when ground-level smoke concentrations cause discomfort or trigger acute health responses, the events are vividly remembered. Current public perception is that smoke produced by forest management activities, particularly regeneration burning in native forest managed for timber production, is the principal cause of pollution events in rural towns and farmland, and a significant risk to health. Consequently, the Tasmanian Government is currently developing policies to monitor and regulate these events and, in response, Forestry Tasmania has implemented a management policy to limit the extent of regeneration and other prescribed burning so that the capacity of the airshed to absorb and disperse the smoke is not exceeded. While the regulatory and management policies are soundly based on expert knowledge, in the absence of measurements of smoke concentrations it is not possible to confirm whether the perception that regeneration burning is the main source of pollution is valid or whether the policy to limit the impacts of biomass burning on regional air quality is effective. To address the issue Forestry Tasmania commissioned this pilot project to monitor PM pollution at two sites in the Huon Valley, a region in which extensive regeneration burning is undertaken.

The objectives of the study were:

- To monitor the ambient surface concentration of PM in town and in rural farmland in the Huon Valley; and
- To assess the contribution of forest management burning to ambient PM concentration.

4. PROJECT DESIGN

4.1 Design principles

The fundamental concern in most pollution studies is the effect of anthropogenic emissions on population health. This is directly determined by the concentrations of pollutants in the ambient air to which people are usually exposed. The pollutant of concern in this study is PM and its measure is the mass concentration. There are many sources of PM, some are natural; some are anthropogenic; some are relatively inert remaining chemically or physically stable from the point of emission while others undergo transformations; and some are formed in the atmosphere from gas-phase chemical reactions and condensation. Most of the anthropogenic PM sources produce PM that is relatively stable. The source of primary concern for this study is forest regeneration burning. To quantify the significance of this source for health impacts we first need to define some measures.
If we assume that the mass of PM \( (M_i) \) observed in a volume \( (V) \) of surface air at any instant \( (t) \) and location in an airshed is the arithmetic sum of PM mass from all emission and formation sources \( (i) \) that has been transported to that point, then the PM concentration \( (C) \) is

\[
C(t) = \frac{1}{V} \sum_i M_i(t)
\]

i.e.

\[
C(t) = \sum_i C_i(t)
\]

Eq. 1

Where \( C_i \) is the component of \( C \) derived from source \( i \).

The impact \( (I) \) of the source on the total PM concentration is the contribution of \( C_i \) to \( C \) integrated over an appropriate time period.

\[
I = \frac{1}{t} \int_0^t C_i(t) \, dt
\]

Eq. 2

The activity of each emission source varies diurnally and seasonally and therefore for most cases the time interval over which this impact is usefully determined is at least a year. Therefore in order to measure the impact of an emission source on PM mass concentration it is necessary to

1. Monitor the ambient surface PM concentration at the location for a minimum of a year; and
2. Discriminate between the source components of PM mass \( (C_i) \).

At the other end of the spectrum, the activity of emission sources is usually assessed using emission inventories. Inventories take several forms, the most common of which estimates the total emission of a specified pollutant from a defined region and year and mostly lack any spatial or temporal information. Most pollution policies use regionally and annually aggregated inventory estimates as the primary metric of emissions. Total annual emission estimates are rarely well correlated with observed impacts on surface concentrations because the emissions are dispersed non-homogeneously within the airshed, and transported beyond the boundaries of the airshed. To a first approximation the lower atmosphere can be viewed as two well mixed compartments, the surface layer and the free troposphere with limited air exchange between the two. In this model, surface concentration is determined by the net flux \( (E_i) \) into the mixed layer, the depth of the mixed layer \( (Z_{\text{mix}}) \) and the mean horizontal wind speed \( (\bar{u}) \), averaged from the surface to \( Z_{\text{mix}} \). Therefore we might expect to observe that

\[
C_i = \frac{E_i}{(Z_{\text{mix}} \bar{u})}
\]

Eq. 3

However, due the complexities of turbulent mixing in the boundary layer, particularly at night when stable condition lead to stratification, equation 3 can, at best, provide only a partial description, and it should be more accurately written as

\[
C_i = f \left( \frac{E_i}{(Z_{\text{mix}} \bar{u})} \right)
\]

Eq. 1

Some sources produce plumes with sufficient buoyancy to rise through the mixed layer into the free troposphere or above and in this situation \( C_i \) is determined only by the fraction of the total emission that remains in the mixed surface layer. Hence, \( E_i \) is the net flux into the mixed surface layer and not the total flux at the ground surface. Current regeneration burning practices exploit
this feature. These burns are designed to develop intense fires which inject smoke into the free troposphere where smoke is dispersed without impacting the mixed surface layer. Only when the fine and coarse fuels are consumed and fire intensity diminishes are substantial amounts of PM injected into the mixed layer. If this strategy is achieved in practice we would expect to observe a large imbalance between the total PM emissions and surface impacts.

Ideally, observed surface impacts of each emission source are linked to monitored emissions activity through the application of a verified chemical transport model. However these models are complex and require accurate specification (temporally and spatially) of all emission sources and source injection heights. Unfortunately, these data are rarely readily available in complete form, and submodels that accurately describe fire intensity, plume rise and smoke injection height are still in development. However $Z_{mix}$ and $u$, can be determined using readily available meteorological models and Eq 4 provides a partial solution to the issue of linking emissions and impacts.

4.2 Experimental design

The Huon Valley was selected for the study because it has many features in common with other airsheds in Tasmania affected by forestry activity and some features that make it particularly suitable for an air quality investigation. It has a mix of forest, urban and rural landscapes, the main industries are forestry and farming, with almost no manufacturing or other large industry and therefore the range of emission source classes is small. Because the prevailing winds are from the west there are no large industrial or urban regions upwind of the airshed and therefore the background air is clean and its chemical composition is stable. Consequently, variations in the atmospheric composition of the airshed will be determined by the sources within the airshed, not by sources external to it. These benefits are coupled with the region’s logistical advantages of good roads, easy access from Hobart and good local facilities.

Biomass combustion appears to be the main PM emission source. The National Pollution Inventory (NPI) (SEWPAC, 2011) reports that the South west Tasmania airshed (which includes the Huon Valley) emitted 250 t PM$_{10}$ in 2009/2010. Domestic solid fuel burning emits the major proportion of this (59%) followed by forest fires (8%), and wood manufacturing industries (1%). Windblown dust from roads contributes a further 24% followed by emissions from fossil fuel combustion (motor vehicles, etc. 8%) and domestic waste incineration (1%). Although the NPI remains the accepted national database of pollutant emissions, it is not free from errors. The methodology for estimating forest fire emissions, which is sourced directly from the National Greenhouse Gas Inventory methodology, is appropriate for wildfires but it substantially underestimates the slash loads remaining after logging which are consumed in regeneration burning. However, this error merely increases the contribution of biomass combustion to total airshed PM inputs. The issue is discussed in detail in the next section.

Combining these two sets of information indicates that the Huon Valley has several properties that make the objectives of this study readily tractable. These are:

1. The urban, rural and forest regions in the airshed have distinctly different PM emission source profiles:
   - There are six towns in the Huon Valley. The main emission source in these urban areas is biomass combustion from wood-fuelled heaters, supplemented by a range of minor domestic emission sources, transport, wind-blown dust and waste combustion.
The rural region has a similar mix of emission sources. However, due to the very much lower dwelling density, biomass combustion from wood-fuelled heaters is likely to be a relatively minor source.

The western side of the airshed is comprised of forest. The only significant anthropogenic sources in this region are forest fires. These are mostly regeneration burns, although some fuel reduction burning is undertaken. There were no significant wildfire events in the airshed during the study period.

2. The two large biomass combustion sources, regeneration burning and wood-fuelled heater combustion are active in different seasons with limited overlap. The regeneration burning season is March and April, while wood-fuelled stove usage is mostly in winter and early spring (May to September).

3. In general, wood-fuelled heater emissions and regeneration burning emissions have different diurnal emission profiles. From the Launceston study (Luhar et al., 2006; Meyer et al., 2008a) and other surveys (Todd, 2008), wood-fuelled heater use occurs predominantly, but not solely, in the late afternoon and evening from 16:00 h to 23:00 h and in the morning before 09:00 h. Therefore most PM from wood-fuelled heaters is emitted into the nocturnal surface layer when atmospheric dispersion typically is poor. In contrast, regeneration burns are ignited in late morning or early afternoon to ensure that by late afternoon most of the fuel is consumed, fire intensity is low and the risk of fire escaping from the coupe is minimal and controllable. The deep convective mixed layer and higher mean wind speeds that occur in the afternoon favour dispersion of the smoke plumes.

4. PM from biomass combustion can be discriminated from road dust, transport emissions and other fossil fuel emissions, and from non-anthropogenic marine and biogenic emission by chemical composition and particle size. Combustion of biomass produces a relatively stable and unique chemical tracer, the anhydrous sugar levoglucosan. In contrast, road dust is comprised predominantly of large (>2.5 μm) inorganic, insoluble particles; fossil fuel emission sources produce aerosol rich in elemental carbon; and marine aerosol is composed mostly of soluble inorganic salts (sea salt) condensed into particles larger than 2.5 μm. Biogenic aerosol consists mostly of fine particles less than 100 nm diameter, rich in organic carbon and devoid of levoglucosan.

5. There are no large anthropogenic emission sources upwind of the airshed and therefore the only anthropogenic sources likely to affect surface PM concentration are located within the airshed.

This combination of diurnal, seasonal, spatial, chemical and particle size differences provides sufficient basis for discrimination between PM sources.

The specific questions required to answer the project objectives fully are:

1. Is there an air quality issue in the Huon Valley, i.e. does the air quality in the populated rural and urban regions of the Huon Valley comply with the current PM$_{10}$ and advisory PM$_{2.5}$ NEPM standards?
2. What are the relative impacts of the major anthropogenic PM sources on urban and rural ambient surface PM concentrations? and
3. What fraction of PM emitted from regeneration burning causes impacts on surface PM concentrations, i.e. how effective is the current policy designed to minimise the impacts of regeneration burning on surface air quality?

The following measurement programs should provide sufficient information required to answer these questions:
Program 1: Monitor continuously at two sites: one in an urban region and one in the rural region.

Both rural and urban sites are required to discriminate between local and regional pollution impacts.

Parameters monitored should include:
- 1 minute, continuous $[PM_{2.5}]$ by light scattering (e.g. TSI DustTrak);
- Weekly mean PM$_{2.5}$ and PM$_{10}$ gravimetric mass concentrations onto 1 μm Teflon filters. This provides (1) a measure of mean weekly PM$_{2.5}$ and PM$_{10}$ mass using an accepted standard method and the mass of PM in the size class 2.5 to 10 μm, and (3) an in situ standard calibration of the continuous PM$_{2.5}$ monitor;
- Weekly mean PM$_{2.5}$ gravimetric mass by continuous sampling onto quartz fibre filters;
- Anhydrous sugar (levoglucosan) and soluble ion content of PM$_{2.5}$ and PM$_{10}$; and
- Elemental carbon (EC) and organic carbon (OC) content of PM$_{2.5}$.

This provides the main observational data set required for the study.

Program 2: Determine the size resolved chemistry and particle number distribution of ambient surface PM during the regeneration burning season.

This provides detailed supplementary data to accurately discriminate between PM source classes in season when all sources are likely to be present.

Together (1) and (2) should directly answer the first two questions. This is because:
- Comparison between the urban and the rural time series will establish whether the two sites are similar during seasons when biomass burning sources are absent;
- Wood-fuelled heater density differs in rural and urban locations, therefore comparison between the urban and rural air quality time series in winter should establish the extent to which urban sources impact the rural region;
- Comparison between the urban and rural time series during the PB season will establish whether PM impacts from prescribed burning result from episodic plume strike, or whether prescribed burning causes widespread impacts throughout the airshed and
- Because the two biomass combustion sources and the remaining anthropogenic and non-anthropogenic sources can be discriminated by season and possibly also by site, the relative impact of biomass combustion emissions of surface air PM can be quantified by equation 2.

Program 3: Monitor surface meteorology at each site and estimate mixed layer depth ($Z_{mix}$), mean wind speed ($\bar{u}$) and hence ventilation rate using a regional atmospheric transport model.

These data facilitate an answer to question 3.
5. METHODS

5.1 Monitoring sites

Ambient measurements were conducted at two locations in the Huon Valley, Tasmania: Geeveston (43.16°S 146.93°E; elev. 115m) and Grove (42.98°S, 147.1°E; elev. 113m) between March 2009 and November 2010 (Figure 1). The Geeveston monitoring site was located within the township at the Forestry Tasmania (FT) Geeveston office. The Grove monitoring site was located at the Department of Primary Industries (DPI) Research Station, 1.6 km west of the township of Grove.

![Figure 1 Location of the air quality monitoring sites in the Huon Valley. The black squares show the location of the monitoring sites.](image)
Figure 2 shows the density of houses in the Huon Valley Statistical Local Area in 2006. The Geeveston monitoring station is located in the Geeveston census collection district which has 277 houses, 725 residents and a house density of 36 houses km$^{-2}$. The Grove monitoring site is located near the centre of Grove Census District which in 2006 had a population of 686 people dispersed over an area of 21 km$^2$ and a housing density of 12.5 houses km$^2$. The town of Huonville (area 5 km$^2$) with a population of 1184 people and house density of approximately 100 houses km$^2$ lies 5 km to the SW of Grove.
5.2 PM emission sources

To understand the air quality of the Huon Valley we need to review the 2009 and 2010 estimates of PM emissions from the three major biomass combustion sources; prescribed burning, wood-fuelled heaters and domestic waste burning.

Prescribed burning

In 2009 and 2010, Forestry Tasmania carried out a range of prescribed burns (Figure 3). Those relevant to the study (e.g. located close enough to impact on either the monitoring site at Geeveston or at Grove) are listed in Table 2.

Figure 3 Location of Regeneration Burns (marked as red dots) conducted in the Huon Valley in Autumn 2009 and 2010. The black squares show the location of the monitoring sites.
Table 2: Area (in ha) of prescribed fires in the Huon Valley 2009 and 2010

<table>
<thead>
<tr>
<th>Date</th>
<th>Distance from Geeveston (km)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0-5</td>
</tr>
<tr>
<td>2009</td>
<td></td>
</tr>
<tr>
<td>11-Mar</td>
<td>0</td>
</tr>
<tr>
<td>12-Mar</td>
<td>0</td>
</tr>
<tr>
<td>20-Mar</td>
<td>0</td>
</tr>
<tr>
<td>21-Mar</td>
<td>0</td>
</tr>
<tr>
<td>1-Apr</td>
<td>0</td>
</tr>
<tr>
<td>2-Apr</td>
<td>0</td>
</tr>
<tr>
<td>7-Apr</td>
<td>0</td>
</tr>
<tr>
<td>8-Apr</td>
<td>37</td>
</tr>
<tr>
<td>9-Apr</td>
<td>0</td>
</tr>
<tr>
<td>13-Apr</td>
<td>0</td>
</tr>
<tr>
<td>14-Apr</td>
<td>0</td>
</tr>
<tr>
<td>22-Apr</td>
<td>18</td>
</tr>
<tr>
<td>23-Apr</td>
<td>0</td>
</tr>
<tr>
<td>21-May</td>
<td>0</td>
</tr>
<tr>
<td>Total</td>
<td>55</td>
</tr>
<tr>
<td>2010</td>
<td></td>
</tr>
<tr>
<td>4-Feb</td>
<td>0</td>
</tr>
<tr>
<td>25-Mar</td>
<td>0</td>
</tr>
<tr>
<td>28-Mar</td>
<td>0</td>
</tr>
<tr>
<td>31-Mar</td>
<td>0</td>
</tr>
<tr>
<td>1-Apr</td>
<td>0</td>
</tr>
<tr>
<td>5-Apr</td>
<td>0</td>
</tr>
<tr>
<td>6-Apr</td>
<td>0</td>
</tr>
<tr>
<td>7-Apr</td>
<td>0</td>
</tr>
<tr>
<td>16-Apr</td>
<td>0</td>
</tr>
<tr>
<td>17-Apr</td>
<td>0</td>
</tr>
<tr>
<td>18-Apr</td>
<td>0</td>
</tr>
<tr>
<td>20-Apr</td>
<td>0</td>
</tr>
<tr>
<td>Total</td>
<td>0</td>
</tr>
</tbody>
</table>

Impacts of smoke from regeneration burning on air quality in the Huon Valley, Tasmania
Annual emissions of PM from regeneration burning can be estimated using the standard algorithm presented in the NPI,

\[ E = A \times FL \times BEF \times EF \]  
\[ \text{Eq. 5} \]

Where:

- \( E \) = PM emission (kg)
- \( A \) = area of the logging coupe (ha)
- \( FL \) = fuel load (t ha\(^{-1}\))
- \( BEF \) = combustion completeness
- \( EF \) = emission factor for PM\(_{2.5}\) (kg t\(^{-1}\))

The area of each planned PB is provided by the responsible State agency, however, there are few published measurements of logging slash fuel loads, particularly for high productivity Eucalypt forests. The measurements at the Warra Long-term Ecological Research (LTER) site (Marsden-Smedley and Slijepcevic, 2001; Slijepcevic, 2001) provide some guidance. This site is within a high productivity old-growth forest, which, following logging, had an average fuel load of 560 t ha\(^{-1}\). Of this 54% of the mass was emitted in the regeneration burn. Although a fuel load of 560 t ha\(^{-1}\) is extremely high, comparison with other forests suggest that it is not exceptionally so. Total above ground biomass in similar mature wet sclerophyll forest in Victoria is approximately 650 t ha\(^{-1}\) (Grierson et al., 1992). From production and harvest area information (Forestry Tasmania, 2010) average forest yield in Tasmania is estimated to be approximately 215 t ha\(^{-1}\). Subtracting this from the total above ground biomass gives a total average residue after logging these high productivity forests of 435 t ha\(^{-1}\). Therefore the fuel load measured in the LTER study seems to be at the extreme end of the ecologically possible range.

The most commonly cited review of biomass combustion EFs (Andreae and Merlet, 2001) recommends PM\(_{2.5}\) EFs of 17 g kg\(^{-1}\) and 8.3 g kg\(^{-1}\) for temperate forests and grasslands respectively. These values are consistent with measured PM emissions factors in prescribed fuel reduction burns in Victoria, SE Queensland and SW Western Australia (Meyer et al., 2004).

Since the only fuel load estimate available is from the extreme end of the range, our estimates of PM\(_{2.5}\) emissions from PB will be at the upper limit of possible range. With this qualification we will assume that:

- broadcast burns have a fuel load of 560 t ha\(^{-1}\), and a BEF of 54%;
- windrow burns comprise mostly canopy residues and fuel > 7cm, which gives a fuel load for the LTER coupes of 450 t ha\(^{-1}\) and a BEF of 45% (Slijepcevic, 2001);
- the fuel load for variable retention coupes is 200 t ha\(^{-1}\), that
- the fuel load for both plantation and supplementary burns is 100 t ha\(^{-1}\); and
- the BEF for variable retention, plantation and supplementary burns is 45%.

Combining these factors with the 2009 and 2010 areas of PB in the Huon Valley in Equation 5, and recalling that this will give an upper limit, we estimate that PM emissions in these two years were 8645 and 9223 t PM respectively (Table 3). These are substantially greater than the NPI estimate of 20 t for PM\(_{10}\) emissions from South West Tasmania. However, the methodology prescribed by the NPI (EA, 1999) is inappropriate for regeneration burns. The parameters specified in the methodology were sourced from the National Greenhouse Gas Inventory Methodology (DEST, 1996), which estimates emissions for two classes of fires; fuel reduction burns and wildfires. The fuel loads appropriate for both these fire classes are much smaller than
the mass of residue remaining after logging, particularly in high production forests, and therefore the NPI emission estimate is, probably, substantially less than reality.

Table 3 Estimated PM emissions from Prescribed burning in the Huon Valley

<table>
<thead>
<tr>
<th>Burn Class</th>
<th>Burned Fuel (t ha⁻¹)</th>
<th>EF (kg PM t⁻¹)</th>
<th>Area (ha)</th>
<th>Emission (kg PM)</th>
</tr>
</thead>
<tbody>
<tr>
<td>FRB¹ Button Grass</td>
<td>8.5</td>
<td>8.3</td>
<td>380</td>
<td>6,779</td>
</tr>
<tr>
<td>HIB² Broadcast Burn - Native Forest</td>
<td>309.7</td>
<td>17.6</td>
<td>1127</td>
<td>6,140,236</td>
</tr>
<tr>
<td>HIB Broadcast Burn - Plantation</td>
<td>60</td>
<td>17.6</td>
<td>77</td>
<td>81,682</td>
</tr>
<tr>
<td>HIB Burn following VR logging</td>
<td>120</td>
<td>17.6</td>
<td>97</td>
<td>205,878</td>
</tr>
<tr>
<td>HIB Supplementary Broadcast Burn Native Forest</td>
<td>60</td>
<td>17.6</td>
<td>0</td>
<td>81,682</td>
</tr>
<tr>
<td>HIB Windrow Burning</td>
<td>202</td>
<td>17.6</td>
<td>614</td>
<td>2,190,710</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>2295</strong></td>
<td><strong>2561</strong></td>
<td><strong>8,645,285</strong></td>
<td><strong>9,229,953</strong></td>
</tr>
</tbody>
</table>

¹ FRB: Fuel Reduction Burn.  
² HIB: High Intensity Burn

Wood-Fuelled heaters

The PM emissions from wood-fuelled heaters can also be estimated with a few reasonable assumptions. Following the arguments of Luhar et al. (2006) and assuming that Launceston wood-fuelled heater use is similar to the Huon Valley we assume that 53% of dwellings used wood-fuelled heaters for domestic heating. The heaters are operated approximately 12 h per day between May and September (150 days) with an average fuel consumption rate of 2.6 kg h⁻¹. Meyer et al. (2008a) found that average PM emission factor for heaters was 10 g kg fuel⁻¹. Combining these values gives a yearly PM wood-fuelled heater emission from the Huon Valley of 121 t PM (Table 4). This is only slightly lower than the NPI estimate for 2009/2010 of 140 t PM for the larger Southwest Region.

Table 4 Estimated Annual Wood-fuelled heater Emissions in the Huon Valley

<table>
<thead>
<tr>
<th>Location</th>
<th>Area (km²)</th>
<th>Houses</th>
<th>Density (houses km⁻²)</th>
<th>PM emission (kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cygnet</td>
<td>4</td>
<td>294</td>
<td>73.0</td>
<td>7,292</td>
</tr>
<tr>
<td>Dover</td>
<td>4</td>
<td>189</td>
<td>49.9</td>
<td>4,688</td>
</tr>
<tr>
<td>Franklin</td>
<td>6</td>
<td>122</td>
<td>21.9</td>
<td>3,026</td>
</tr>
<tr>
<td>Geeveston</td>
<td>8</td>
<td>277</td>
<td>36.6</td>
<td>6,871</td>
</tr>
<tr>
<td>Huonville</td>
<td>5</td>
<td>441</td>
<td>97.7</td>
<td>10,939</td>
</tr>
<tr>
<td>Ranleigh</td>
<td>2</td>
<td>173</td>
<td>91.9</td>
<td>4,291</td>
</tr>
<tr>
<td>Rural</td>
<td>1,091</td>
<td>1,960</td>
<td>1.8</td>
<td>48,616</td>
</tr>
<tr>
<td>Forest</td>
<td>4,385</td>
<td>1,429</td>
<td>0.3</td>
<td>35,445</td>
</tr>
<tr>
<td><strong>Grand Total</strong></td>
<td><strong>5,502</strong></td>
<td><strong>4,885</strong></td>
<td><strong>2.4</strong></td>
<td><strong>121,168</strong></td>
</tr>
</tbody>
</table>
Other open combustion of biomass

During the fire permit period which occurs typically from November to March, with the exception of barbecues, campfires and incinerators, open burning can only be undertaken on issue of a permit. Outside this period open combustion of biomass on private land is mostly unregulated. Statistics of the extent of this activity in the Huon Valley outside the permit period are incomplete however anecdotal advice is that the biomass burning is common, particularly on agricultural holdings.

Clearly, PB and WH are the two dominant PM$_{2.5}$ sources in the Huon Valley. However, health impacts are determined not by total annual emissions, but by the PM$_{2.5}$ concentrations they produce in the surface air. Our challenge is to elucidate this processes that link the two.

5.3 Measurement methods

5.3.1 Ambient air monitoring

Weekly gravimetric measurements of PM$_{2.5}$ and PM$_{10}$ were made on samples collected using two low-volume aerosol samplers (Microvol-1100, Ecotech Pty Ltd, Knoxfield, Australia) fitted with respectively a PM$_{2.5}$ and PM$_{10}$ size-selective inlet in accordance with AS 3580.9.10:2006. The sample stream through the inlet and filter was maintained at a constant flow of 3 L min$^{-1}$ using a mass flow meter in the unit. Samples were collected on pre-weighed 47 mm Teflon filters (Teflo, Pall R2PJ047, 2 µm pore size). NATA-accredited gravimetric mass measurements on the pre-exposed and exposed filters were made using an ultra-microbalance (Model UMT2, Mettler Toledo, USA) with a specialty filter pan. Biweekly gravimetric measurements were made on samples collected between July and August 2009, March and April 2010 and June and August 2010.

Continuous measurements of PM$_{2.5}$ were made using a laser light-scattering detector (DustTrak model 8250, TSI Inc, USA), operated with a PM$_{2.5}$ impactor. Zeroing of the DustTrak was performed automatically every 6 hours. Particle mass concentration is determined by the amount of light scatter, based on a factory calibration factor (Standard ISO 12103-1 A1 dust). The relationship between light scattering and particle mass concentrations varies with particle source and particle size distribution and is usually nonlinear. Gravimetric mass measurements of PM$_{2.5}$ made on samples collected using a MicroVol 1100 were used to calibrate the PM$_{2.5}$ concentrations determined by the DustTrak. The relationship between the DustTrak data and the gravimetric measurements observed in this study is fitted by the relationship (Figure 4)

$$y = \frac{6.27x}{0.82 + x} + 0.24x$$

Eq. 2

Where:

$y$ = gravimetric PM$_{2.5}$ concentration and

$x$ = DustTrak PM$_{2.5}$ reading.

Above 8 µg m$^{-3}$ the relationship is approximately linear which is consistent with a single pollutant source class contributing most of the anthropogenic emissions of aerosol into the Huon Valley. In this report, all particle concentrations are reported in gravimetric units, and, where sourced from DustTrak data are calculated using the simpler linear relationship (Figure 4):

$$y = 0.25x + 5.52.$$  

Eq 3
Impacts of smoke from regeneration burning on air quality in the Huon Valley, Tasmania

Between March 2010 and November 2010, weekly samples of PM$_{2.5}$ were also collected on quartz filters which were analysed for organic and elemental carbon.

5.3.2 Size-resolved characterisation of regeneration burn aerosol

An intensive campaign was carried out in March and April 2010 during which microphysical and chemical properties of the aerosol were measured at Geeveston.

Twelve aerosol samples were collected using a micro-orifice uniform deposit impactor (MOUDI). The MOUDI is a 10 stage cascade impactor (effectively 12 stage when the inlet stage and final filter are included, as in this work) with the stages having 50% cut-points ranging from 0.056 µm to 18 µm in aerodynamic diameter. The MOUDI data are inverted using an inversion routine described by Keywood (Keywood et al., 1999).

Samples were collected on polycarbonate or tissue quartz filters depending on the analytical method to be applied (Table 5). Sampling duration was typically four days. Filters were analysed for gravimetric mass using the methodology described above, and a range of chemical composition including anhydrous sugars (levoglucosan), organic and elemental carbon, and soluble ions. Accurate size resolved gravimetric mass determination is possible only for samples collected on polycarbonate filters. Conversely, carbon analysis is only possible with samples collected on quartz filters.
Table 5  Details of MOUDI samples collected including substrate (PC = polycarbonate, TQ = tissue quartz) and analysis performed on substrates (mass = gravimetric mass, SI = soluble ions, AS = anhydrous sugars, C= organic carbon and elemental carbon). Fire activity: PB= Prescribed forestry burning, BB= other biomass burning.

<table>
<thead>
<tr>
<th>Sample Number</th>
<th>Sample ID</th>
<th>Date/Time On</th>
<th>Days sampled</th>
<th>Substrate</th>
<th>Analysis</th>
<th>Fire activity during sampling</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>A1</td>
<td>15-March 10:53</td>
<td>4</td>
<td>PC</td>
<td>mass, SI, AS</td>
<td>None</td>
</tr>
<tr>
<td>2</td>
<td>B1</td>
<td>19-March 11:18</td>
<td>4</td>
<td>PC</td>
<td>Mass, SI, AS</td>
<td>None</td>
</tr>
<tr>
<td>3</td>
<td>C1</td>
<td>23-March 11:59</td>
<td>3</td>
<td>PC</td>
<td>mass, SI, AS</td>
<td>None</td>
</tr>
<tr>
<td>4</td>
<td>E1</td>
<td>26-March 11:27</td>
<td>5</td>
<td>PC</td>
<td>mass, SI, AS</td>
<td>None</td>
</tr>
<tr>
<td>5</td>
<td>D1</td>
<td>31-March 11:55</td>
<td>6</td>
<td>PC</td>
<td>mass, SI, AS</td>
<td>Minor-PB</td>
</tr>
<tr>
<td>6</td>
<td>B2</td>
<td>06-April 13:21</td>
<td>3</td>
<td>TQ</td>
<td>C, SI, AS</td>
<td>Minor-PB</td>
</tr>
<tr>
<td>7</td>
<td>A2</td>
<td>09-April 12:22</td>
<td>4</td>
<td>TQ</td>
<td>SI, AS</td>
<td>PB &amp; BB</td>
</tr>
<tr>
<td>8</td>
<td>C2</td>
<td>13-April 12:19</td>
<td>3</td>
<td>TQ</td>
<td>C, SI, AS</td>
<td>PB &amp; BB</td>
</tr>
<tr>
<td>9</td>
<td>E2</td>
<td>16-April 16:51</td>
<td>4</td>
<td>TQ</td>
<td>C, SI, AS</td>
<td>PB</td>
</tr>
<tr>
<td>10</td>
<td>D2</td>
<td>20-April 11:42</td>
<td>3</td>
<td>TQ</td>
<td>SI, AS</td>
<td>PB</td>
</tr>
<tr>
<td>11</td>
<td>B3</td>
<td>23-April 10:13</td>
<td>4</td>
<td>TQ</td>
<td>SI, AS</td>
<td>PB</td>
</tr>
<tr>
<td>12</td>
<td>A3</td>
<td>27-April 13:47</td>
<td>3</td>
<td>TQ</td>
<td>C, SI, AS</td>
<td>BB</td>
</tr>
</tbody>
</table>

The 2010 regeneration burning season in the Huon Valley was very short; Forestry Tasmania ignited all fires on the 17th and 18th April. This resulted in smoke plume impacts in Geeveston between the 17 April and 22 April which corresponds to the sampling period for MOUDI samples 9, 10 and, partly, 11. However, biomass burning was also undertaken on private property by landowners. Smoke from these events was collected by samples 5, 6, 7 and 12.

5.4 Chemical analysis

Filters were analysed for water-soluble ions, the anhydrous sugars, levoglucosan and mannosan and total elemental and total organic carbon.

Water soluble ions and anhydrous sugars (levoglucosan) were extracted from the Teflon, or polycarbonate filters by adding 100 µl of methanol, to wet the hydrophobic filter, and 5 cm³ of de-ionised water produced from a Millipore Milli-Q Advantage 10 system. To prevent loss of soluble ions and anhydrous sugars from bacterial action, 50 µl of chloroform was added. Anion and cation concentrations were determined by suppressed ion chromatography (IC) with a Dionex ICS-3000 reagent free ion chromatograph. Anions were separated using a Dionex...
AS17c analytical column (4 x 250 mm), an ASRS-300 suppressor and a gradient eluent of 0.75 mM to 35 mM potassium hydroxide. Cations were separated using a Dionex CS12a column (4 x 250 mm), a CSRS-300 suppressor and an isocratic eluent of 20 mM methanesulfonic acid. Instrumental detection limits for anions and cations range from 0.006 to 0.026 µg filter⁻¹, which corresponds to an ambient detection level of 0.0002-0.0009 µg m⁻³, assuming a sample volume of 30 m³.

Anhydrous sugars were determined by high-performance anion-exchange chromatography with pulsed amperometric detection (HPAEC-PAD). Levoglucosan was separated from other anhydrous sugars such as galactosan and mannosan with a 4 mm x 250 mm CarboPac PA10 analytical column protected by a 4 mm x 50 mm CarboPac PA10 guard column. The instrumental detection limit for levoglucosan measured by the HPAEC-PAD technique is approximately 0.03 µg cm⁻³ in aqueous solution. This results in an ambient detection level of about 0.002 µg m⁻³, assuming an air sample volume of 30 m³. The development of the HPAEC-PAD technique is based on a previous study to measure levoglucosan in PM₂.₅ from biomass combustion (Engling et al., 2006a).

Carbon analysis was performed using a DRI Model 2001A Thermal-Optical Carbon Analyzer using the IMPROVE-A temperature protocol. Laser reflectance was used to correct for charring, since reflectance has been shown to be less sensitive to the composition and extent of primary organic carbon. Prior to analysis of filter samples, the sample oven was baked to 910 °C for 10 minutes to remove residual carbon. System blank levels were then tested until < 0.20 µg C cm⁻² was reported (with repeat oven baking if necessary). Twice daily calibration checks were performed to monitor possible catalyst degeneration. Replicate analysis was performed on approximately every 10th sample to within ± 10 %. The analyser is reported to effectively measure carbon concentrations between 0.05 – 750 µg C cm⁻², with uncertainties in OC and EC of ± 10 %.

Chemical analysis of the PM₂.₅ and PM₁₀ aerosols focused primarily on markers for wood smoke; specifically non-seasalt potassium, levoglucosan, total organic and total elemental carbon. Levoglucosan is a unique tracer for wood burning, and is not produced from any other known source (Simoneit et al., 1999). It is generally produced at high concentrations in biomass smoke and has been used in previous studies as a tracer for particle emissions from fires or burns (Alves et al., 2010; Engling et al., 2006b; Pio et al., 2008; Saarikoski et al., 2007; Saarnio et al., 2010; Sullivan et al., 2008; Ward et al., 2006) and residential wood burning (Caseiro et al., 2009; Jordan et al., 2006; Puxbaum et al., 2007; Schauer and Cass, 2000; Wang et al., 2006). Smoke from biomass burning is also known to be high in potassium, which can therefore be used as a chemical tracer for wood smoke (Zhang et al., 2010). However unlike levoglucosan, potassium has other significant sources including sea salt and soil dust. Non sea salt potassium (nssK⁺) in concentration units of µmol L⁻¹ was determined using the ratio of potassium to magnesium found in seawater (Millero, 1974) as shown in Equation 7;

\[
nssK^+ = K^+ \left( \frac{0.01058 \times Mg^{2+}}{0.11038 \times 0.5} \right) \tag{Eq 4}
\]

One other compound, oxalate, is also of interest as a potential tracer for biomass combustion. Oxalate can originate from primary emissions of biomass burning or as a secondary product from oxidation of aromatic hydrocarbons emitted by combustion of biomass or other material or produced naturally from biogenic sources. In rural areas the non-biomass sources are likely to appear as a relatively stable background source, possibly with seasonal variability which correlates with ambient temperature.
5.5 Modelling ventilation rate

Ventilation rate at Geeveston and Grove was calculated for every hour from March 2009 to November 2010 using the CSIRO atmospheric dispersion model TAPM. TAPM is a three-dimensional, prognostic meteorological and air pollution model (see (Hurley, 2006; Hurley et al., 2005); http://www.cmar.csiro.au/research/tapm for model details). Given the large-scale synoptic analyses as input boundary conditions for the horizontal wind components, temperature and moisture, TAPM simulates local scales at a finer resolution using a one-way multiple nesting approach, predicting local-scale meteorology. The synoptic input is supplied from the Australian Bureau of Meteorology’s GASP modelling system (given at 6-hourly intervals at approximately 100-km spaced grid points across Australia). Other inputs to the model include global databases of terrain height (given at a horizontal resolution of about 250 m for Australia), land use, and monthly sea-surface temperature. The performance of TAPM has been verified in a number of previous meteorological and dispersion studies (e.g. Hurley (2006). The model was run using the default settings, and four nested 40 x 40 grids of 20 km, 7 km, 3 km and 0.5 km spacing, centred on each monitoring site. Ventilation rate was calculated from the mixing layer depth ($Z_{mix}$) and the mean wind speed from the surface to the height of the mixing layer ($\bar{u}$).

5.6 Evaluating source impacts

The primary information we require for this study is the relative contribution of the major anthropogenic emission sources, both seasonally and annually, to ambient background concentration of PM$_{10}$ and PM$_{2.5}$.

The twenty months sampling period was separated into 3 seasons: 1 March to 30 April (PB season); 15 May to 15 September (WH season), and 16 September to 28 February (Spring/Summer). This seasonal separation was based on Forestry Tasmania’s burning programme, which mainly occurs in March and April; the domestic wood-fuelled heater season, which typically extends from late Autumn to early Spring; and the remainder of the year, when the main biomass combustion sources are fuel reduction burning and wildfires.

Hourly and daily PM$_{2.5}$ averages were calculated from the minutely DustTrak data corrected against gravimetric PM$_{2.5}$ concentrations. The seasonal averaged concentrations represent the arithmetic means calculated from either daily averaged concentrations (PM$_{2.5}$) or weekly/biweekly gravimetric measurements (chemical markers). The average ambient background concentration for each site was estimated from samples collected in spring/summer when there were long periods that were clearly free from the impact of biomass combustion.

In order to determine the contribution of prescribed burns, residential wood-fuelled heaters or other sources to the overall ambient PM load, two approaches were taken.
Approach 1:

The first approach assumes a fixed background PM$_{2.5}$ concentration all-year round against which the impact from biomass burning was assessed. We classify the sources $j$ contributing to ambient PM$_{2.5}$ into:

- Prescribed burns conducted by Forestry Tasmania and other state and local authorities (PB). These are documented in the Forestry Tasmania fire database (Table 2).
- Wood-fuelled heater emissions (WH)
- Biomass burning events other than Forestry Tasmania’s prescribed burns or wood-fuelled heater emissions (BB). These events correspond to time periods outside of the winter period when levels of the wood smoke tracer levoglucosan are high but no data is recorded for burning activities.
- Non-biomass burning events occurring when PM concentrations are above background levels but levoglucosan levels remain low.

The integrated PM$_{2.5}$ load from a source $j$ over the sampling period is given by

$$ PM_j (\mu g \ m^{-3} \ d) = \sum_{i=1}^{n} \left[ \left( \sum_{t_1}^{t_2} PM_{2.5} \right) - \bar{x}_{BG} \times (t_2 - t_1) \right] $$

Eq 5

Where:

- $\bar{x}_{BG}$ = average background PM$_{2.5}$ concentration which corresponds to the average ambient PM$_{2.5}$ concentration measured in spring/summer. The average concentration of PM$_{2.5}$ from November to February inclusive ($\bar{m}_B$) was 6.8 µg m$^{-3}$ at Geeveston and 6.4 µg m$^{-3}$ at Grove;
- $t_1$ = the start of an event;
- $t_2$ = the end of an event; and
- $i$ = the number of events recorded during the sampling period.

The contribution of a pollution source $j$ to the total (i.e. background and anthropogenic) PM load is obtained by dividing the integrated PM$_{2.5}$ load of source $j$ by the total PM load, e.g.

$$ Contribution \ of \ pollution \ source \ j (\%) = \frac{PM_j}{\sum_{i=1}^{n} PM_{2.5}} $$

Eq 6

where $\Delta T$ is the number of days within the sampling period (March 2009 to November 2010).

Approach 2:

In this approach we assume that Grove is representative of background air in the Huon Valley, excluding clearly identified local smoke plume events. The impact of biomass burning on ambient PM$_{2.5}$ concentrations measured at Geeveston is determined by subtracting the daily PM$_{2.5}$ concentrations measured at Geeveston from the daily PM$_{2.5}$ concentrations measured at Grove. This approach allows for seasonal variation in background aerosol concentration caused, for example, by secondary aerosol formation from atmospheric chemistry involving biogenic volatile organic compounds. It is a more conservative approach than Approach 1 because the local impact large-scale events that affect the aerosol concentration of an entire airshed, such as wildfires or extensive prescribed burning, will tend to be underestimated.
The integrated PM$_{2.5}$ load resulting from a pollution event at Geeveston is given by

$$PM_j(\mu g \ m^{-3} \ d) = \sum_{i=1}^{n}[\sum_{j=1}^{n}((Daily \ PM_{2.5})_{Geeveston} - (Daily \ PM_{2.5})_{Grove})_{j}] \ Eq \ 7$$

Where:

$t_1$ = the start of the event
$t_2$ = the end of the event; and
$i$ = the total number of events resulting from a pollution source $j$ during the sampling period.

Hence

$$Contribution \ of \ pollution \ source \ j \ to \ total \ PM \ load (\%) = \frac{PM_j}{\sum_{j=1}^{n}((Daily \ PM_{2.5})_{Geeveston})} \ Eq \ 8$$

Where $\Delta T$ is the number of days within the sampling period (March 2009 to November 2010).

6. RESULTS

Ambient monitoring was conducted over a 20-month period from March 2009 to November 2010, to sample two PB seasons, two WH seasons and one spring/summer season, in which there was no recorded forest fire. The summer/summer period is, in effect, a control.

6.1 Meteorology

Weather data was provided by the Bureau of Meteorology Automated Weather Station in Grove and a local weather station set up by Forestry Tasmania alongside the monitoring site in Geeveston.

There was virtually no difference in temperature between the two sites. Average monthly temperatures ranged between 6.6 and 17 ºC (Figure 5). Lowest temperatures were measured in June/July and highest temperatures were measured between January and March. Temperatures ranged from a mean daily maximum of 25 ºC in February to 12º C in July; mean daily minimum temperatures ranged from 12 ºC to 1 ºC for January and July.

The highest rainfall in 2009/10 occurred during winter.
Figure 5 Monthly average, minimum and maximum temperatures measured at Geeveston and Grove

Figure 6 shows the wind roses of mean hourly wind speed and wind direction data for Geeveston and Grove. There is little seasonal variation in wind direction at Geeveston. The town was ventilated mostly from the westerly airflow or, less frequently from easterlies; there were very few occasions where airflow was from the north or south along the axis of the valley. Almost all flows with wind speed above 2 m s\(^{-1}\) were from the WSW to WNW. Winds at Grove were slightly more variable. In summer and autumn wind was mostly from the southwest, particularly at wind speed above 2 m s\(^{-1}\); however there were long periods during the WH season in both years, but particularly in 2009, which were dominated by gentle wind flows, mostly at less than 2 m s\(^{-1}\) draining down the valley from the north.

The dominant westerlies expose Geeveston to autumn fire activity in the forests to the west, while the easterlies, mostly, flush the airshed with clean marine air. Grove was far less susceptible to smoke impacts. During autumn, the south westerlies were almost invariably stronger than 2 m s\(^{-1}\) and hence smoke transport from forest burning to the SW is likely to affect Grove only under conditions of relatively high ventilation rate when the impact would be small. There were even fewer opportunities during the WH season for transport of smoke emitted from wood-fuelled heaters in Huonville at the low ventilation rates required to produce significant pollution events, and therefore, the monitoring site at Grove was well situated to sample the rural air in the Valley, relatively free from the influence of urban or forestry activity.
Figure 6  Wind roses for Geeveston and Grove. A,C,E,G WH season 2009, Spring/Summer 2009/2010, PB season 2010, WH season 2010 respectively at Geeveston and ; B,D,F,H WH season 2009, Spring/Summer 2009/2010, PB season 2010, WH season 2010 respectively at Grove. Blue < 2 m s^{-1}; yellow, 2-4 m s^{-1}; green, 4-6 m s^{-1}; red 6-8 m s^{-1}; purple > 8 m s^{-1}.
6.2 The annual time course of surface PM concentrations

Ambient PM mass was measured at Geeveston and Grove by two methods; by gravimetric mass determination using filter samples collected either weekly or biweekly, and continuously using a laser light-scattering detector (DustTrak model 8250, TSI Inc, USA) which records 1-minute average PM$_{2.5}$ concentrations. The gravimetric method is an accepted standard for PM mass measurement and establishes the calibration scale for the collocated DustTrak which provides the temporal resolution.

Average concentrations of PM$_{2.5}$, PM$_{10}$ and chemical tracers (levoglucosan, nssK$^+$ and oxalate) derived from the filter samples are summarised in Table 6 and Table 7.

Table 6 PM$_{2.5}$, PM$_{10}$, levoglucosan, nssK$^+$ and oxalate seasonal averages for Geeveston. The seasons are defined as: PB season (March-April), WH season (mid May-Mid September), Spring/Summer (Mid September-February), Annual (March 2009-March 2010). Uncertainty ranges are standard deviations.

<table>
<thead>
<tr>
<th>Geeveston</th>
<th>PB season</th>
<th>WH season</th>
<th>Spring/Summer</th>
<th>Annual</th>
</tr>
</thead>
<tbody>
<tr>
<td>PM$_{2.5}$ (µg m$^{-3}$)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Average</td>
<td>12.1±5.6</td>
<td>14.0±12.6</td>
<td>19.7±7.3</td>
<td>23.7±7.7</td>
</tr>
<tr>
<td>Hourly maximum</td>
<td>114</td>
<td>119</td>
<td>110</td>
<td>149</td>
</tr>
<tr>
<td>Daily maximum</td>
<td>44.3</td>
<td>71.1</td>
<td>41.6</td>
<td>48.5</td>
</tr>
<tr>
<td>Levoglucosan (µg m$^{-3}$)</td>
<td>1.95±1.50</td>
<td>1.65±1.56</td>
<td>4.42±2.34</td>
<td>5.49±2.04</td>
</tr>
<tr>
<td>nssK$^+$ (µg m$^{-3}$)</td>
<td>0.062±0.033</td>
<td>0.062±0.085</td>
<td>0.071±0.041</td>
<td>0.096±0.035</td>
</tr>
<tr>
<td>Oxalate (µg m$^{-3}$)</td>
<td>0.041±0.016</td>
<td>0.058±0.064</td>
<td>0.047±0.011</td>
<td>0.053±0.014</td>
</tr>
<tr>
<td>OC (%)</td>
<td>71.3</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>EC(%)</td>
<td>6.9</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>PM$_{10}$ (µg m$^{-3}$)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Average</td>
<td>21.2±13.2</td>
<td>18.6±13.2</td>
<td>23.3±8.7</td>
<td>29.1±6.1</td>
</tr>
<tr>
<td>Levoglucosan (µg m$^{-3}$)</td>
<td>2.25±1.61</td>
<td>1.71±1.54</td>
<td>4.54±2.48</td>
<td>6.02±1.99</td>
</tr>
<tr>
<td>nssK$^+$ (µg m$^{-3}$)</td>
<td>0.094±0.035</td>
<td>0.090±0.088</td>
<td>0.077±0.044</td>
<td>0.116±0.035</td>
</tr>
<tr>
<td>Oxalate (µg m$^{-3}$)</td>
<td>0.057±0.017</td>
<td>0.062±0.073</td>
<td>0.057±0.014</td>
<td>0.063±0.014</td>
</tr>
</tbody>
</table>
Table 7  PM$_{2.5}$, PM$_{10}$, levoglucosan, nssK$^+$ and oxalate seasonal averages for Grove. The seasons are defined as: PB season (March-April), WH season (mid May-Mid September), Spring/Summer (Mid September-February), Annual (March 2009-March 2010). Uncertainty ranges are standard deviations.

<table>
<thead>
<tr>
<th>Grove</th>
<th>PB season</th>
<th>WH season</th>
<th>Spring/Summer</th>
<th>Annual</th>
</tr>
</thead>
<tbody>
<tr>
<td>PM$_{2.5}$</td>
<td>µg m$^{-3}$</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Average</td>
<td>4.6±1.1</td>
<td>8.6±3.5</td>
<td>6.7±1.4</td>
<td>8.6±3.2</td>
</tr>
<tr>
<td>Hourly max</td>
<td>32.5</td>
<td>32.8</td>
<td>19.9</td>
<td>25.4$^a$</td>
</tr>
<tr>
<td>Daily max</td>
<td>10.2</td>
<td>15.9</td>
<td>15.0</td>
<td>12.4$^b$</td>
</tr>
<tr>
<td>Levoglucosan</td>
<td>0.24±0.08</td>
<td>0.31±0.32</td>
<td>0.59±0.44</td>
<td>0.92±0.46</td>
</tr>
<tr>
<td>nssK$^+$</td>
<td>0.025±0.008</td>
<td>0.041±0.047</td>
<td>0.030±0.021</td>
<td>0.059±0.034</td>
</tr>
<tr>
<td>Oxalate</td>
<td>0.036±0.016</td>
<td>0.048±0.051</td>
<td>0.031±0.016</td>
<td>0.036±0.014</td>
</tr>
<tr>
<td>OC (%)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>EC (%)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>PM$_{10}$</td>
<td>µg m$^{-3}$</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Average</td>
<td>9.4±2.1</td>
<td>10.3±3.7</td>
<td>7.7±1.5</td>
<td>9.1±2.4</td>
</tr>
<tr>
<td>Levoglucosan</td>
<td>0.54±0.39</td>
<td>0.30±0.31</td>
<td>0.59±0.47</td>
<td>0.94±0.39</td>
</tr>
<tr>
<td>nssK$^+$</td>
<td>0.050±0.021</td>
<td>0.045±0.047</td>
<td>0.032±0.022</td>
<td>0.062±0.026</td>
</tr>
<tr>
<td>Oxalate</td>
<td>0.053±0.015</td>
<td>0.045±0.050</td>
<td>0.032±0.016</td>
<td>0.039±0.014</td>
</tr>
</tbody>
</table>

* The highest averaged hourly and daily PM$_{2.5}$ concentrations were measured at 157 µg m$^{-3}$ and 28.7 µg m$^{-3}$ respectively. These peak concentrations were due to a windrow burning of a grubbed orchard approximately 500 m from the monitoring site.

In the spring/summer season the PM concentration in Geeveston was marginally higher than in Grove; the mean differences were 7% for PM$_{2.5}$ and 16% for PM$_{10}$, but these were not statistically significant. We conclude that in the absence of biomass burning both sites were comparable.

At Grove, the average PM$_{2.5}$ and PM$_{10}$ concentrations in the PB and WH seasons differed little from their values in the spring/summer season (Table 7). The highest daily PM$_{2.5}$ concentration recorded at this site, 28.7 µg m$^{-3}$, occurred in June 2010 and was caused by windrow burning of a grubbed apple orchard approximately 500 m from the monitoring site. Excluding this event the highest mean daily PM$_{2.5}$ concentration was measured at 18 µg m$^{-3}$. However at Geeveston there were large differences. In the PB season the mean PM$_{2.5}$ and PM$_{10}$ concentrations were respectively 76% and 60% higher than in summer and the WH season was even more polluted, PM$_{2.5}$ was 166% above background in 2009 and 220% in 2010. For PM$_{10}$, the differences were slightly lower; 88% in 2009 and 135% in 2010 due to the slightly higher background PM$_{10}$ level (Table 6). The highest daily PM$_{2.5}$ concentration of 71 µg m$^{-3}$ measured in April 2010 at Geeveston was caused by regeneration burning. While this event was also observed at Grove, it was substantially attenuated at 16 µg m$^{-3}$.

The gravimetric measurements of PM$_{2.5}$ and PM$_{10}$ were found to be linearly correlated (Figure 7). In background air approximately 40% of particle mass was found in the PM$_{2.5}$ to PM$_{10}$ size fraction. The slope of 1.06 indicates that most of the additional PM found in the PB and WH seasons was PM$_{2.5}$, and probably from a uniform source. If so, then the linear regression shown in Figure 7 can be used to estimate PM$_{10}$ concentration from the continuous PM$_{2.5}$ time series recorded by the DustTraks.
Figure 7 Relationship between gravimetric PM$_{10}$ and gravimetric PM$_{2.5}$ measurements

The daily average PM$_{2.5}$ and PM$_{10}$ concentrations measured between March 2009 and November 2010 at the two sites are presented in Figure 8. The Figure shows in detail the seasonal variation presented in Table 6 and Table 7. Geeveston, the urban site, is strongly and persistently impacted in the WH season; less so and more episodically during the PB season. Grove, the rural site, is only slightly impacted, mostly during the WH season, but also by occasional pollution events at other times of the year.
Figure 8 Daily averaged PM$_{2.5}$ and PM$_{10}$ concentrations measured at Geeveston (a) and Grove (b) between March 2009 and November 2010

The seasonal distribution of daily PM$_{2.5}$ concentrations for Geeveston and Grove is shown in Figure 9. At Grove, 78-98% of daily PM$_{2.5}$ concentrations were below 10 µg m$^{-3}$, with minimal seasonal variation. At Geeveston we observed a large seasonal variation. In spring/summer the majority of days (97.5%) had daily PM$_{2.5}$ concentrations less than 10 µg m$^{-3}$, and no days experienced concentrations above the advisory NEPM standard of 25 µg m$^{-3}$. The highest proportion of days above the standard occurred in the WH season (21-56%) followed by the PB season (6.6%). In the PB and WH seasons the proportion of days with concentrations less than 10 µg m$^{-3}$ was 3-29% and 56-65% respectively, much lower than the proportion of days observed for spring/summer. Days with concentrations above 50 µg m$^{-3}$ were only observed during autumn 2010 (1.6%).
Figure 9 Seasonal distribution of daily PM$_{2.5}$ concentrations for Geeveston (top) and Grove (bottom). The seasons are defined as follows: PB season (March-April), Winter WH season (Mid May-Mid September), Spring/Summer (Mid September-February).

The monthly distribution frequencies of 24-h mean PM$_{2.5}$ is shown in Figure 10. The distributions are relatively symmetrical around the median for most months but occasionally are skewed by episodic extreme pollution events. At Geeveston, the amplitude of the annual cycle was approximately 15 µg m$^{-3}$. The lower boundary of the envelope (i.e. 5th percentile) also varied seasonally; a broad minimum occurred between October and March and the maximum occurred in July indicating a seasonal cycle in the background air concentrations might also be occurring at this site. The upper limit of the envelope (the 95th percentile) followed the same pattern, minimum in summer, maximum in July, but with double the amplitude of the median. The highest values were observed in April 2010 when the distribution was strongly skewed by the regeneration burning events.
At Grove, PM$_{2.5}$ concentrations remained low throughout the year with small monthly variations. The amplitude of the annual cycle was 3 µg m$^{-3}$ with maxima in both summer and winter. Excluding the episodic pollution events of November 2009, April 2010 and June 2010, the range of the monthly PM$_{2.5}$ concentration distribution at Grove varied from less than 1 µg m$^{-3}$ to ~5 µg m$^{-3}$.
Figure 11 shows the diurnal distributions of the 1 h mean PM$_{2.5}$ concentrations at Geeveston and Grove. At both sites the diurnal distribution is generally characterised by an increase in particle levels in the afternoon, peaking at about midnight, then gradually decreasing early morning, with an additional peak at around 8am. The lowest concentrations occur near midday. The morning peak is most distinct during the late autumn and winter. Minimal diurnal variations are observed in summer and spring and the amplitudes of the cycles were larger at Geeveston than at Grove.

These diurnal variations can be attributed to the atmospheric mixing, which is limited under a low stagnant night-time inversion, and to residential heating by wood-fuelled stoves during evening hours. The daily cycle of wood-fuelled heater emission rates of PM$_{10}$ and the resultant atmospheric concentrations measured in Launceston (Meyer et al., 2008a) are similar to the cycles observed at Grove and Geeveston. The amplitude of the diurnal variation at Geeveston (≈30 µg m$^{-3}$ PM$_{2.5}$) is 6-fold higher than at Grove, which is broadly consistent with the difference in building densities surrounding the two locations and hence the strength of the WH emission source.
Figure 11: Diurnal distribution of PM$_{2.5}$ concentrations measured at Geeveston (top) and Grove (bottom). Note differences in scale.
6.3 Chemical composition of PM$_{2.5}$ and PM$_{10}$ aerosols

In the previous section we presented the temporal and diurnal patterns of PM concentrations, which help in discriminating between biomass combustion sources. Analysis of the chemical composition of PM will further aid in confirming this discrimination and assist in identifying other anthropogenic PM sources that may be contributing to total ambient PM.

6.3.1 Time series analysis of biomass burning tracers

At both sites there was large seasonal variation in levoglucosan concentration; it was highest in the WH season and minimum in the spring and summer (Table 6 and Table 7). WH season average concentrations were six times higher than summer average concentrations, indicating a significant woodsmoke contribution to the WH season particle loading. At Geeveston elevated levoglucosan levels were also measured during the autumn PB season. Weekly measured levoglucosan concentrations ranged from 0.054 to 10.9 µg m$^{-3}$ for PM$_{2.5}$ and from 0.069 to 9.91 µg m$^{-3}$ for PM$_{10}$ at Geeveston and from 0.018 to 1.83 µg m$^{-3}$ and 0.024 to 1.65 µg m$^{-3}$ for PM$_{2.5}$ and PM$_{10}$, respectively at Grove. Seasonally averaged levoglucosan concentrations were in general higher than those observed in other sites impacted by woodsmoke across the US and Europe, where average concentrations ranged from 0.003 to 2.98 µg m$^{-3}$ (Caseiro et al., 2009; Jimenez et al., 2006; Kleeman et al., 2008; Nolte et al., 2001; Pio et al., 2008; Puxbaum et al., 2007; Saarikoski et al., 2007; Ward et al., 2006; Zhang et al., 2010). Only in Fresno, California were maximum winter levoglucosan levels similar to levels observed at Geeveston (Schauer and Cass, 2000).

Figure 12 shows the weekly or biweekly time series measurements of the biomass burning tracers levoglucosan, nssK$^+$ and oxalate for PM$_{2.5}$ samples collected at Geeveston and at Grove. Similar trends are observed for levoglucosan and nssK$^+$, with high concentrations measured between April and July and low concentrations measured in spring and summer. The levoglucosan and nssK$^+$ concentrations were also reasonably correlated ($r^2 = 0.62-0.68$) at both sites suggesting their common emission sources. At Geeveston there was one outlier with much higher nssK$^+$ to levoglucosan ratios compared to the remaining data points. This may be a result of either higher nssK$^+$ input from burns, higher content of soil dust entrained in the smoke plume or degradation of levoglucosan in the smoke plume during transport (Hennigan et al., 2010; Hoffmann et al., 2010). The measurement was taken between 16-20 April 2010 during a smoke plume impact. Excluding this particular data point from the data set, we obtain a correlation between levoglucosan and nssK$^+$ of $r^2 = 0.73$. Levoglucosan and nssK$^+$ concentrations were on average 2-8 times higher at Geeveston than at Grove. This is mainly due to the spatial location of both monitoring sites. Geeveston and its surrounding localities have a population of about 1600 compared to 700 at Grove and therefore will have a higher frequency of residential wood burning. Furthermore, the monitoring site was located within the township where impact of wood-fuelled heater smoke on ambient air quality would be higher during winter. Grove displayed higher nssK$^+$ to levoglucosan ratios compared to Geeveston, suggesting a potential additional source of K most likely soil dust. The Grove monitoring site is located alongside an agricultural experimental field with a likely higher input of soil dust compared to Geeveston.
Figure 12  Time series analysis of hourly PM$_{2.5}$ concentrations and weekly or biweekly gravimetric PM$_{2.5}$, levoglucosan, nssK$^+$ and oxalate concentrations. The top panel highlights smoke plume events observed at Geeveston (GE1-GE4) and Grove (GR1-GR4).

The seasonal trend and the spatial variation was less pronounced for oxalate and almost no correlation was observed between levoglucosan and oxalate ($r^2 = 0.17$). A distinctive peak for

Impacts of smoke from regeneration burning on air quality in the Huon Valley, Tasmania
all three biomass burning tracers was only recorded in April 2010, resulting from smoke impacts of regeneration burning. This indicates distinctly different behaviour of oxalate from that of levoglucosan or nssK⁺. During smoke plume events we observe primary emissions of oxalate from biomass burning, consistent with the elevated levels of biomass burning tracers, levoglucosan and nssK⁺. But at most other times the elevated levels of oxalate are probably the result of secondary formation by gas-phase atmospheric chemistry.

6.3.2 Size distributions of PM chemistry and number and mass during the 2010 PB season

Between March and May 2010 the standard instrument suite at Geeveston was supplemented with two high-resolution analysers to investigate in greater detail the changes in aerosol composition during the transition from summer, through the PB season to the WH season. These instruments provide size resolved particle mass concentration, particle number concentration and chemistry.

Size-resolved chemical composition

The time series of PM₁₀ and PM₂.₅ show a significant increase in particulate matter associated with prescribed burning activities during April 2010. In this section we discuss the effect of this activity on the aerosol mass and chemical size distributions observed at the sampling site at Geeveston.

The 2010 regeneration burning season in the Huon Valley was short; regeneration burns were conducted from 25 March to 20 April. Fire restrictions were lifted on 9 April 2010 allowing landholders to undertake burning on private property without a requirement for authorisation from the Tasmania Fire Service, and anecdotal advice indicates that a significant amount of non-forest burning occurred on the 10th and 11th of April. The 16th-20th April proved to be ideal for regeneration burning and it was these fires that had a major impact on the air quality of Geeveston and to a minor extent on Grove. MOUDI samples 1-6 were collected prior to the lifting of fire restrictions; samples 7 and 8 were collected in the period where both forestry and non-forestry burning was active; samples 9-11 were collected during the period of active regeneration burning near Geeveston and sample 12 was collected after the cessation of the 2010 PB season (Table 2).

The size-resolved mass distributions for the 5 samples collected between 15 March and 6 April 2010 are shown in Figure 13. No forest burning was registered during the collection periods of samples 1 and 2, and prescribed burning during the collection of samples 3 and 4 was conducted 85-110 km and 43 – 73 km NW of Geeveston respectively, well outside the wind trajectory to the town (Table 2). The concentrations of PM₁₀ and PM₂.₅ in these samples range between 9-15 µg m⁻³ and 5-9 µg m⁻³ respectively and their mass distributions are dominated by a coarse mode that peaks at between 5 and 10 µm. These distributions are typical of a non-polluted atmosphere in which sea-salt is the major source of coarse mode particles, and secondary particles (i.e. particles which form by atmospheric chemistry, and are represented here by non-sea salt sulphate), are the major components of the fine particle mode, less than 2.5 µm in diameter (Figure 14A).

Sample 5 was collected between 31 March and 6 April following the commencement of the Forestry Tasmania’s regeneration burning programme for 2010. During this period 19 prescribed burns (470 ha total area) were completed within a radius of 30 km from Geeveston.
(Table 2) and while their impact on PM$_{2.5}$ concentration was small, increasing average PM$_{2.5}$ concentration to 13 µg m$^{-3}$, the combustion aerosol is clearly evident in the chemical composition. The mass size spectrum (Figure 13) is broad and flat; the largest mode occurs at 1 µm with a smaller mode at 6 µm. The 6 µm mode in the mass distribution corresponds to sea-salt K$^+$ (estimated here from Na$^+$ concentrations) and the 1 µm mode corresponds to levoglucosan (Figure 14B).

![Figure 13](image_url)  
Figure 13 Gravimetric mass distribution as a function of particle diameter for samples 1-5
Figure 14  Size resolved chemical composition of low concentration ambient air samples. A: Gravimetric mass, sea-salt and nss-SO$_4$ (indicator of secondary aerosol measured in Sample 1 (Background air). B. Gravimetric mass, sea-salt and levoglucosan (indicator of biomass burning) in Sample 5 (smoke impacted air).

Impacts of smoke from regeneration burning on air quality in the Huon Valley, Tasmania
Figure 15  Size distribution for all samples of: A. Levoglucosan and B. Sea-salt.
The size distribution of levoglucosan and sea-salt of all MOUDI samples are compared in Figure 15. Highest concentrations of levoglucosan (by a factor of 3 or more) were observed in sample 9, the main smoke plume event, while highest concentrations of sea-salt were observed in samples 1-4 and 12. The size distributions of levoglucosan are dominated by a single mode around 1.5 µm, with some variation which could be due to particle condensation associated with smoke aging. In all samples the size distribution of sea-salt centred on a single coarse mode of 6 µm.

The time series of the size resolved levoglucosan concentration is shown in detail in Figure 16. Levoglucosan is present in the aerosol throughout the period and, mostly, high concentrations correlate with high particle number density and the PB events listed in Table 2. The trend in the particle size mode distribution is clearly evident.

Samples 6 - 12 were collected on TQ filters for the determination of OC and EC and hence mass size distributions for these samples are not available. The size-resolved distribution for OC and EC for samples 6, 8 and 9 (smoke impacted samples) is shown in Figure 17. Similar to the size distribution of levoglucosan, the distributions are dominated by single modes centred about
1.5 - 1.8 µm for OC and 1.1 - 1.6 µm for EC. Highest concentrations of both were observed in sample 9.

Figure 17 Size distribution of organic carbon (OC) and carbon (EC) in samples 6, 8, 9 and 12, all of which were impacted by smoke from regeneration burning.

**Summary**

Summarising these data, what we observe in the 2010 PB season is a transition from background aerosol consisting of marine and biogenic PM to aerosol that is predominantly biomass combustion PM. The chemical fingerprint of marine/biogenic aerosol is:

- A mass spectrum with mode of 5 – 6 µm;
- High sea-salt content with a mode at 5 – 6 µm; and
- Low levoglucosan content

The chemical fingerprint of biomass combustion aerosol is:

- A mass spectrum with mode of 1 – 2 µm;
- High levoglucosan content with a mode at 1-2 µm;
- High organic carbon content with a mode at 1-2 µm; and
- An EC/OC mass fraction of 5 - 20%.
Microphysical Properties

The transition from background to biomass combustion PM is evident in more detail in the high resolution time series of particle size distribution. The size spectrum of ambient PM at Geeveston was measured throughout the 2010 PB season at 5 minute intervals using a Scanning Mobility Particle Sizer (SMPS). This instrument separates PM into classes of different particle diameters based upon electrical mobility. The number concentration in each class is then measured after condensation growth using a condensation particle counter.

Three sequences of size-resolved particle concentrations are presented here to illustrate the contrasting characteristics of clean and smoke impacted air. Figure 18 covers the period immediately prior to the commencement of prescribed burning when Geeveston was ventilated with clean background air. Particle number concentrations are low throughout, with some relatively short episodes at night and in the early morning when concentrations increased. The size mode of these episodes ranges between 20 and 100 nm. The second period (Figure 19) shows the aerosol number concentration during the first week of prescribed burning when PM$_{2.5}$ concentrations remained low. While the number concentrations are substantially higher, again mostly at night, the particles are small with a mode near 50 nm. In contrast Figure 20 shows the size distribution of aerosol particles during the severe smoke event (16 April to 18 April). The concentrations are more than an order of magnitude higher than the second period, the mode is substantially larger and particle growth is clearly evident increasing from 93 nm to about 157 nm in 12 h at a growth rate of 5.3 nm h$^{-1}$. Particle growth can be caused by condensation of either organic gases or water onto particles, however, because the condensation sink (CS) for water vapour during this event (Pirjola et al., 1999), is extremely high (40 s$^{-1}$) compared to the monthly average (2 s$^{-1}$), we can conclude that the growth observed here is due mainly to humidity. The details of the phenomena are described by Pirjola et al. (1999). Therefore, the smoke impacting Geeveston in these events (Figure 19 and Figure 20) is relatively fresh.
Figure 18  Size distribution of aerosol particles measured with the SMPS from 23rd March to 25th March during the collection of MOUDI sample 3, a very clean period.
Figure 19  Size distribution of aerosol particles measured with the SMPS from 26th March to 29th March during the collection of MOUDI sample 4. Low concentrations of smoke.
Figure 20    Size distribution of aerosol particles measured with the SMPS from 16th April 01:00 AM to 18th April 18:00 PM during the collection of MOUDI sample 9. Very high smoke concentrations. The black dots correspond to the mean diameter of the particle mode during the growth period.

6.3.3 The annual cycle of levoglucosan

We now return to the specific smoke tracer, the anhydrous sugar levoglucosan. This compound is produced solely by the thermal decomposition of cellulose. It was once thought to form a relatively consistent fraction of emitted PM mass and to be chemically stable, at least for many days. If this were the case then it would be possible using only the levoglucosan content to determine what fraction of any aerosol mass concentration was produced by biomass combustion. Recent work has shown that it is neither chemically stable in the long term (Hennigan et al., 2010; Hoffmann et al., 2010), nor is it emitted as a constant fraction of PM mass (Meyer et al., 2008a). Nevertheless it remains a valuable tracer, particularly when assessing fresh smoke. The time series of the levoglucosan fraction of PM$_{2.5}$ mass for Geeveston and Grove for the weekly and biweekly filter samples is shown in Figure 21. At both Grove and Geeveston, the levoglucosan fraction increases from near zero in late summer to a maximum in the WH season. At Grove, the maximum fraction is 20% in early June 2009, averaging 15% through June and July 2009 and 12% in the complete WH season of 2010. The fractions at Geeveston are higher; they average 24% in June and July 2009 and the 2010 WH season.

The relationship between levoglucosan and PM$_{2.5}$ concentration from the PB and WH season filters is described by a rectangular hyperbola in which the levoglucosan fraction approaches an
asymptote as the fraction of biomass combustion PM$_{2.5}$ in the total aerosol increases (Figure 22). The levoglucosan fraction from well oxygenated combustion of Eucalypt fuels in wood-fuelled heaters averages 25% in a range of 17% - 30% (Meyer et al., 2008a) which is similar to the maximum values measured at Geeveston in the WH season. The lower fraction observed, particularly in the PB season could be caused by dilution with the background PM or could be caused by differences in combustion chemistry as was observed in Launceston wood-fuelled heater emissions (Meyer et al., 2008a). If it is the former then we would expect to find a consistent levoglucosan composition in the anthropogenic component of surface PM at all times through the WH and the PB seasons. This appears to be the case (Figure 21C).

Figure 21 Time series of levoglucosan fraction (%PM$_{2.5}$ mass) for A: Grove and B: Geeveston and C: levoglucosan fraction of anthropogenic PM$_{2.5}$ at Geeveston during the PB and WH seasons. Anthropogenic PM$_{2.5}$ is estimated by Equation 8.
The relationship between the fraction of levoglucosan in PM$_{2.5}$ mass and PM$_{2.5}$ concentration for the autumn and winter filters collected at Geeveston. The data is fitted with the function

$$F_{lev} = \frac{41.3 \times PM_{2.5}}{18.1 + PM_{2.5}} \quad (r^2 = 0.67)$$

### 6.3.4 EC/OC data

The EC content of PM can be diagnostic of the emission source. PM emitted by vehicles usually contains a large fraction of EC. Chow (Chow et al., 2004) report chemical profiles for a wide range of sources in California and find that the EC (expressed as a fraction of total carbon (TC) content) in road dust, soil, vehicle exhaust and vegetation fires averages 13%, 3.6%, 37% (19 - 49%) and 14% (7 - 24%) respectively. The EC:TC ratios in work environments with high densities of diesel powered vehicles range from 21% to 63% (Groves and Cain, 2000). Analysis of the PM$_{2.5}$ quartz filters collected between March 2010 and July 2010 at Geeveston and Grove showed that carbonaceous material accounted for 76 ± 22% of the PM$_{2.5}$ mass, with the major component being organic carbon. The EC:TC ratios for Geeveston and Grove were 7 – 23% (average 14%) and 6 – 40% (average 14%) respectively (Figure 23). These ratios indicate that the contribution of fossil fuel combustion to ambient PM$_{2.5}$ in the Huon Valley is minor.
6.3.5 Aerosol chemical mass balance

The final task is to combine these analyses to produce the chemical mass balance of PM$_{2.5}$ comprising all the water-soluble ions, organic matter and elemental carbon. Organic carbon (OC) measured by the Thermal/Optical Carbon Analyzer quantifies the mass of carbon on the filter. OC is converted to organic matter by multiplying the measured concentration of OC (µg C m$^{-3}$) by a factor of 1.4 which is the estimated organic molecular weight per carbon weight (Turpin and Lim, 2001). These chemical components accounted on average for 90 ± 10% of the total PM$_{2.5}$ mass. Therefore there was little unaccounted mass that would indicate the presence of insoluble inorganic material (wind-blown dust) or substantial quantities of elemental carbon that would indicate vehicle emissions. At Geeveston, the chemical composition of PM$_{2.5}$ was dominated by organic matter (Figure 24). At Grove, in addition to organic matter, water soluble ions also constituted an important fraction of PM$_{2.5}$ mass.
6.4 Ventilation rate

The concentration of PM in the mixed layer is determined by both the activity of the emission source and the atmospheric dispersion processes, particularly the ventilation rate and volume of the mixed layer. The two parameters that define these are the mixing layer depth ($Z_{\text{mix}}$) and the mean wind speed ($\bar{u}$). These can be estimated, as described in section 5.5.

The TAPM modelled diurnal patterns of the monthly averaged mixing layer depth ($Z_{\text{mix}}$) at Geeveston and Grove for 2009 and 2010 are shown in Figure 25. The monthly profiles are similar at both sites. Mixing depth is low at night increasing during the day to a maximum in the early afternoon then diminishing again to the night time height by early evening. Lowest mixing depths are observed in winter between May and July.
This pattern is pertinent to both WH and PB emissions. At the time when wood-fuelled heaters are most active, $Z_{\text{mix}}$ is minimum, PM accumulates and concentrations rise. It also determines the impact of PB emissions. Prescribed burns are usually ignited near midday when the mixing depth is maximum to ensure that the smoke disperses quickly. However fire intensity, the parameter that determines buoyancy and plume strength, decreases as the fire progresses and careful management is required to ensure that the fire burns faster than the mixing depth diminishes. If the fire continues burning through the night, then the impact of PB emissions on source PM concentration will be large.

![Figure 25: Diurnal pattern of modelled depth of mixing layer at Geeveston for 2009 (top) and 2010 (bottom)](image-url)
The seasonal time course of $Z_{mix}$ at Geeveston is shown in Figure 26b. Both maximum and minimum $Z_{mix}$ vary significantly by day and by season; the daily minimum depth of the mixing layer ranged from 25 to 500 m while the daily maximum depth of the mixing layer ranged from 120 to 2500 m. Ventilation rate is the product of $Z_{mix}$ and $\bar{u}$ and the positive correlation between the two results in a daily variation in the ventilation rate of 3-4 orders of magnitude (Figure 26c). The range in minimum and maximum daily ventilation rate for the PB seasons were 6-6000 m$^2$s$^{-1}$ and 1400-38000 m$^2$s$^{-1}$ respectively and for the WH seasons were 2-6000 m$^2$s$^{-1}$ and 1000-37000 m$^2$s$^{-1}$ respectively. We find that the low emissions rates (comparative to PB) occur at night when the ventilation rate is also minimum and the PB emissions take place when ventilation rate is maximum. The ratio of PB emissions to WH emission is 70, substantially less than the ratio of daytime and nighttime ventilation rate which is usually 500-1000. Hence despite the very large PM emissions from PB, the impact on surface concentrations typically is small. But there is also a large day to day variability in this ratio, from 5 to 3000; and there are occasions when there is a significant risk of surface impacts from regeneration burns. Fortunately most days of low ventilation rate are excluded by the prescription rules.
Comparison of Figures 26a and 26c show that days of high PM\(_{2.5}\) concentration in the PB season were usually days of relative low maximum ventilation rate, while days of high PM\(_{2.5}\) concentration in the WH season were days of low ventilation rate at night.

### 6.5 Detailed analysis of smoke plume strikes

In this section we discuss the effect of prescribed burning on the urban and rural monitoring site and analyse in more detail the smoke plume events.

Smoke events resulting from biomass burning sources are primarily characterised by high levels of PM\(_{2.5}\) with concurrent increases in concentrations of biomass burning tracers. We define these events as occurring at Geeveston when 24-hour running average PM\(_{2.5}\) concentrations exceeded 25 µg m\(^{-3}\) and/or hourly PM\(_{2.5}\) concentrations exceeded 40 µg m\(^{-3}\) and levoglucosan were significantly elevated (> 2 µg m\(^{-3}\)). At Grove, events were defined as 24-hour running average PM\(_{2.5}\) concentrations exceeded 10 µg m\(^{-3}\) and/or hourly PM\(_{2.5}\) concentrations exceeded 25 µg m\(^{-3}\). In all cases these events were unambiguously different from the background PM\(_{2.5}\) concentrations.

#### 6.5.1 Geeveston

Figure 27 shows the hourly PM\(_{2.5}\) concentrations measured at Geeveston highlighting the biomass burning episodes in autumn each year. During autumn, four distinct smoke plume events were identified; three in autumn 2009 (13-14 April (GE1), 22-24 April (GE2) and 28-30 April (GE3)) and one in autumn 2010 (15-20 April (GE4)) The air quality and meteorological data for the smoke episodes are shown in Table 8. Potential sources for the smoke episodes GE1, GE2 and GE4 are regeneration burns carried out by Forestry Tasmania (Table 2). However, no fire activity data was registered by Forestry Tasmania that could potentially have caused the smoke plume event occurring at the end of April 2009 (GE3). The elevated PM\(_{2.5}\) concentrations could either be caused by burns on private land or potential residential wood burning. A very minor smoke plume event detected at Geeveston on 21 March 2009 was probably associated with three small prescribed burns (Table 2) located to the north and northwest of Geeveston.

![Figure 27 Time series of hourly PM\(_{2.5}\) concentrations during autumn smoke plume events at Geeveston](image_url)
The smoke events persisted, on average, 50 hours with one exception; GE4, the single large event in 2010 lasted 162 hours. During the autumn smoke episodes, average PM$_{2.5}$ concentrations ranged from 24 to 40 µg m$^{-3}$, with maximum hourly PM$_{2.5}$ levels peaking at 50 to 119 µg m$^{-3}$ and maximum daily PM$_{2.5}$ concentrations ranging from 25 to 71 µg m$^{-3}$. A significant increase in hourly PM$_{2.5}$ concentrations (six to ten fold) generally occurring in the afternoon/early evening was observed for each smoke plume event. Hourly PM$_{2.5}$ concentrations exceeded 25 µg m$^{-3}$ for 38-69% of the time (the highest percentage being observed for GE2) and exceeded 50 µg m$^{-3}$ for 0-27% of the time (the highest percentage being observed for GE4).

For comparison, the duration of high PM periods in the WH seasons varied from 36 hours to 27 days and PM$_{2.5}$ concentrations were similar to those observed during the autumn smoke plume events. The mean hourly maxima ranged from 75-149 µg m$^{-3}$, which is similar to the PB season events. The mean daily maxima varied between 19 and 49 µg m$^{-3}$ which is only marginally lower than the mean daily maxima for the four PB season events (Table 8). Therefore although the peak mean hourly PM concentrations during PB and WH events were similar the latter were more persistent and had a greater impact on surface PM concentration. The anthropogenic component of the WH events contributed 40.8% to total integrated PM concentration compared to the 4.1% contribution from the PB events.

Figure 28 shows the hourly PM$_{2.5}$ concentrations measured during the April 2010 smoke plume event in conjunction with wind speed and ventilation rate. This PB event was conducted at a time of relatively low ventilation rate and the smoke concentrations predictably were large. Interestingly, the highest concentrations occurred at night from plume strike which suggests either that plume dispersion at the emission site during the afternoon was poor, or that the smoke impacting Geeveston was emitted later in the day when the ventilation rate was even less. Almost all the fires responsible for GE4 were less than 10 km from Geeveston (Table 2), and plumes from intense burns should have penetrated the inversion layer (Figure 28).
Figure 28  Time series analysis of average hourly PM$_{2.5}$ concentrations measured during the April 2010 prescribed burning event and modelled hourly wind speed and ventilation rate.
Table 8  Air quality and meteorological data during smoke plume episodes, winter episodes and the summer (smoke-free) period at Geeveston

<table>
<thead>
<tr>
<th>Geeveston</th>
<th>PB season 2009</th>
<th>PB season 2010</th>
<th>WH season 2009</th>
<th>WH season 2010</th>
<th>Summer</th>
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<tr>
<td></td>
<td>GE1</td>
<td>GE2</td>
<td>GE3</td>
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<td>GE4</td>
</tr>
<tr>
<td>Duration (hrs)</td>
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<td>52</td>
<td>57</td>
<td>162</td>
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<tr>
<td>PM$_{2.5}$ ($\mu$g m$^{-3}$)</td>
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<td>To seasonal PM (%)</td>
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<td>285</td>
<td>260</td>
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</table>

1. Calculated using equation 9 with hourly PM$_{2.5}$ data
2. Missing weather data between 8/6/2010 and 28/07/2010; at Grove total rainfall measured between 12/05/2010 and 15/07/2010 was 44.4 mm
6.5.2 Grove

Four smoke plume events were recorded at Grove; 13-14 April 2009 (GR1), 7-11 November 2009 (GR2), 1 April and 17-20 April 2010 (GR3) and 21-22 June 2010 (GR4) (Figure 29). The air quality and meteorological data for the events are shown in Table 9. The smoke plume event in November was characterised by four distinctive peaks, with maximum PM$_{2.5}$ concentration ranging from 23.5 to 43.5 $\mu$g m$^{-3}$. The smoke plume in June was produced by the burning of the windrowed residues of an apple orchard felled the previous year. This fire was less than 1 km from the monitoring site. During this event PM$_{2.5}$ concentration increased 16-fold higher than the normal background concentration of 10 $\mu$g m$^{-3}$ to a peak of 158 $\mu$g m$^{-3}$. However, the event was short-lived and concentrations were above 50 $\mu$g m$^{-3}$ for only 4 hours.

The smoke episodes impacting the monitoring site in April 2009 and 2010 were the result of the prescribed burns which also impacted the Geeveston site, however, compared to Geeveston the impacts were minor (Figure 29). The large 2010 event that fumigated Geeveston had relatively little impact further up the valley. The peak mean hourly concentration of 40 $\mu$g m$^{-3}$ PM$_{2.5}$ occurred near midday on 18 April and thereafter the event consisted of a slowly increasing background concentration to approximately 20 $\mu$g m$^{-3}$ PM$_{2.5}$. This increase probably reflects the general level of the PM$_{2.5}$ pollution that accumulated in the valley rather than being a direct plume strike.

![Figure 29](image-url) Time series of the hourly PM$_{2.5}$ concentrations during a smoke plume event caused by regeneration burns in autumn 2009 (left) and autumn 2010 (right)
Table 9  Air quality and meteorological data during PB season plume episodes, WH season episodes and the summer (smoke-free) period at Grove

<table>
<thead>
<tr>
<th>Grove</th>
<th>PB season 2009</th>
<th>PB season 2010</th>
<th>WH season</th>
<th>Summer</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>GR1</td>
<td>GR2</td>
<td>GR3</td>
<td>season</td>
</tr>
<tr>
<td>Duration (hrs)</td>
<td>19</td>
<td>70</td>
<td>65</td>
<td></td>
</tr>
<tr>
<td>PM$_{2.5}$ ($\mu$g m$^{-3}$)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Average</td>
<td>13.2</td>
<td>16.4</td>
<td>15.7</td>
<td></td>
</tr>
<tr>
<td>Day (10AM-6PM)</td>
<td>10.1</td>
<td>17.9</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Night (8PM – 8AM)</td>
<td>21.3</td>
<td>14.5</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hourly Max</td>
<td>32.5</td>
<td>43.5</td>
<td>32.9</td>
<td></td>
</tr>
<tr>
<td>Daily Max</td>
<td>10.0</td>
<td>17.7</td>
<td>15.8</td>
<td></td>
</tr>
<tr>
<td>Hrs &gt; 25 $\mu$g m$^{-3}$</td>
<td>1</td>
<td>16</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td>Hrs &gt; 50 $\mu$g m$^{-3}$</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Levoglucosan ($\mu$g m$^{-3}$)</td>
<td>0.61</td>
<td>1.18</td>
<td></td>
<td></td>
</tr>
<tr>
<td>nssK$^+$ ($\mu$g m$^{-3}$)</td>
<td>0.042</td>
<td>0.183</td>
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<td>Anthropogenic contribution</td>
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<tr>
<td>To seasonal load (%)</td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>To total load (%)</td>
<td>0.1</td>
<td>0.7</td>
<td>0.6</td>
<td></td>
</tr>
<tr>
<td>Temperature (°C)</td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Average</td>
<td>12.2</td>
<td>17.2</td>
<td>11.9</td>
<td></td>
</tr>
<tr>
<td>Hourly Min</td>
<td>2.3</td>
<td>6.2</td>
<td>5.6</td>
<td></td>
</tr>
<tr>
<td>Hourly Max</td>
<td>23.1</td>
<td>31.4</td>
<td>23.6</td>
<td></td>
</tr>
<tr>
<td>RH (%)</td>
<td>71</td>
<td>74</td>
<td>89</td>
<td></td>
</tr>
<tr>
<td>Wind speed (m/s)</td>
<td>2.50</td>
<td>1.54</td>
<td>1.04</td>
<td></td>
</tr>
<tr>
<td>Wind direction (°)</td>
<td>160</td>
<td>117</td>
<td>89</td>
<td></td>
</tr>
<tr>
<td>Total Rainfall [mm]</td>
<td>0.2</td>
<td>0</td>
<td>1.0</td>
<td></td>
</tr>
</tbody>
</table>

1 Calculated using equation 9 with hourly PM$_{2.5}$ data
6.6 Summary of results

Returning to the original objectives it is now clear that:

- At the rural site in Grove, PM concentrations remained low throughout the year with slight increases observed during winter and autumn. At the urban site in Geeveston we observed a similar pattern, but with a much greater amplitude. Grove has very low PM$_{2.5}$ surface concentrations all year round, Geeveston experiences persistently high PM$_{2.5}$ concentrations in winter and episodic smoke plume strikes in the autumn.

- Analysis of the chemical composition of both PM$_{2.5}$ and PM$_{10}$ confirmed major source classes of ambient PM. The background PM contained both inorganic and organic constituents indicating both marine and biogenic sources. In contrast to the background PM, the chemical composition of the anthropogenic PM was consistent with a single source class; biomass combustion. There was no evidence from PM composition or size spectra that mineral dust or fossil fuel combustion are significant PM sources in the Huon Valley.

- In the PB season the biggest impacts can be correlated with PB activity relatively close to the monitoring sites. However there is no evidence from these events of extensive accumulation of PM throughout the Valley; on the contrary, the impacts have the characteristics of smoke plumes that pass through the sites and disperse at a rate dependent on ventilation rate. Although the fires were ignited in the day, the greatest impacts on ambient air quality occurred at night when the ventilation rate was low.

- The protracted periods of high PM$_{2.5}$ concentration occurred in the WH season. The diurnal pattern of PM$_{2.5}$ concentration suggested that the source was mostly active in the evening and morning. The impacts on background PM concentration at Grove and Geeveston are consistent with the differences in house density, which suggests that WH impact is a highly localised phenomenon. Despite the close proximity of the Grove site to two urban areas (Grove 1.6 km to the east and Huonville 5 km to the south) no urban impacts were detected in the PM$_{2.5}$ record.
7. DISCUSSION

7.1 Air quality in the Huon Valley

During the 20-months monitoring program we recorded 99 exceedence days of the advisory PM$_{2.5}$ NEPM and eight exceedence days of the PM$_{10}$ NEPM at Geeveston, the urban site. The rural site at Grove only recorded one exceedence day of the advisory PM$_{2.5}$ NEPM and no exceedences of the PM$_{10}$ NEPM. At Geeveston out of the eight exceedence days of the PM$_{10}$ NEPM, three occurred during the PB season (one in 2009 and two in 2010) and five in the 2010 WH season. The majority (92%) of the advisory PM$_{2.5}$ NEPM exceedences occurred in the WH season. These statistics suggest that the air quality at Geeveston is marginal, but mostly during the WH season.

Although Geeveston and Grove are located within the same airshed, there are large differences in PM concentrations between the two sites. Unlike other studies that have shown that PM$_{2.5}$ average concentrations in an airshed are uniform (Ward et al., 2004), the findings from this study suggest that PM$_{2.5}$ measurements are primarily influenced by local sources rather than by regional pollution. Because there are no anthropogenic sources upwind of the Huon Valley airshed and because Grove and Geeveston have very similar aerosol composition in the spring and summer it is reasonable to assume that any seasonality in the background aerosol will affect Grove and Geeveston equally. Therefore Grove can be used as a background site against which the smoke impact on the air quality in Geeveston can be assessed.

The analysis approach 1 which was applied in Section 6 assumes that the summer PM concentration contains no anthropogenic material and therefore is the background PM. We assume that its concentration remains constant throughout the year and calculate the impact of the PB and WH sources relative to that reference. Because the PM concentration at Grove appears to be relatively free from anthropogenic influences, Approach 2 (Equation 11), we use the PM concentration at Grove to allow us to calculate the source impact (I) without assuming that the background PM concentration has no seasonality. It is a conservative approach that will underestimate I if the assumption isn’t completely valid, but it improves the temporal accuracy. Maintaining our assumption that the anthropogenic sources are temporally separate, we can define them as follows:

1. any days with elevated concentrations of daily PM and weekly wood smoke tracers on which Forestry Tasmania was conducting a burn were classified as Forestry Tasmania burns (PB)
2. any days with elevated concentrations of daily PM and weekly wood smoke tracers observed between mid May to mid September were classified as wood-fuelled heater emissions (WH) if Forestry Tasmania was not conducting burns
3. any other days on which concentrations of both PM and wood smoke tracers were elevated were classified as other burns (BB)

The results are presented in Figure 30 and Table 10.
Impacts of smoke from regeneration burning on air quality in the Huon Valley, Tasmania

At Geeveston, the contribution to the total ambient PM$_{2.5}$ load was 6% from Forestry Tasmania regeneration burns (PB), 39% from wood-fuelled heater emissions (WH), 2% from other burns (BB) and 4% from other non biomass burning sources (nBB). This suggests that wood-fuelled heater emissions contributed about seven times more particles to the ambient PM$_{2.5}$ load at Geeveston compared to smoke from prescribed burns conducted by Forestry Tasmania. During the 20 months sampling period regeneration burns contributed 11.2% to the additional burden of ambient PM$_{2.5}$ (above background) at Geeveston compared with 76.5% from wood-fuelled heaters. Burns conducted by local landholders (4%), vehicle emissions, emissions from local industry and other sources of fine aerosol (8%) account for the remainder. Therefore despite the
70-fold larger PM emissions from PB compared with WH, the impact on the airshed from the PB activities is very small.

The difference in relative impact of PB and WH events is entirely consistent with our knowledge of dispersion. Wood-fuelled heaters are local emission source active mostly in the evening and, consequently, their diurnal impact on surface PM concentration is greatest at night and minimal during the day (Figure 31B). This pattern closely resembles the PM emission cycle measured \textit{in situ} on WH in Launceston (Meyer et al., 2008a), with the exception of the morning peak at 07:00. The lack of a daytime impact could be explained equally by increasing ventilation rate and reducing emission rate. The relationship between nighttime PM concentration and ventilation rate shows that, mostly, high concentrations occur under conditions of low ventilation (Figure 32), that the daytime relationship is less consistent and that there are no occasions in the day when we find high PM concentrations and low ventilation rate indicating that daytime WH activity is low.

![Figure 31](image.png)

**Figure 31** Number of hourly averaged PM$_{2.5}$ concentrations above 25, 50 and 100 µg m$^{-3}$ at Geeveston during autumn and winter
The diurnal impact of PB emissions on surface PM concentration is not what we expect. Prescribed burns are daytime events when ventilation rates are usually large, yet the impacts occur mostly, although not totally, at night (Figure 31A). We can’t rule out the possibility that WH sources are also active, and the similarity between the April diurnal cycle of PM$_{2.5}$ concentration to May, June, July and August both at Geeveston and Grove (Figure 11) suggests this may be partly true. However, an alternative possibility is that the impacts are from PB plumes that develop in the late afternoon and evening when ventilation rate diminishes and/or by smouldering burns that emit low levels of smoke into the nocturnal boundary layer. This needs to be investigated further through characterisation of the smoke plume events and detailed modelling of plume dispersion. If the explanation is valid, then there may be further scope for improving the management of PB impacts by optimising fire duration to minimize the risk of late afternoon and evening smoke emissions.

### 7.2 Population exposure and health impacts

There are two issues that might lead to different health impacts from PB and WH pollution events of identical severity. The first issue relates to the concentration time course of the pollution event and the second to whether houses and buildings can protect their occupants from exposure to smoke.

#### 7.2.1 Health impacts

Smoke plume events are often associated with large peaks in concentrations of short duration. If, as is sometimes argued, the health impact of PM$_{2.5}$ has a non-linear dose response then over the course of a day the impact from short periods of high PM concentrations may be more severe than sustained exposure to lower concentrations even though both events have the same...
24-h average concentration. These scenarios can be assessed by comparing the frequency distribution of surface PM$_{2.5}$ concentrations.

Figure 33 Frequency distribution of hourly and daily PM$_{2.5}$ concentrations measured at Geeveston during autumn prescribed burning period and winter period
The comparison between the PB and WH seasons for 2009 and 2010 presented in Figure 33 (left panels) shows that a similar pattern occurs in the frequency distribution of 1-h mean concentrations for the PB and WH seasons of both years. The majority (60-80%) of hourly PM$_{2.5}$ concentrations fall in the 0-10 $\mu$g m$^{-3}$ class with increasingly fewer events in each of the higher concentration bins; however the high concentration bins in the WH seasons have higher frequencies than the corresponding PB season bins. If the health response is either linear, or has an increasing slope i.e. if the health impact increases faster than the concentration, then the WH seasons which have a higher frequency of high concentration events than the PB season, will have the greater health impact. This is also the case average daily PM$_{2.5}$ in 2009 (Figure 33, right panels), however the frequency distributions for the 2010 PB and WH seasons are sufficiently different that it is impossible to determine which season is likely to have the greater daily risk to health without detailed knowledge of the health response function.

7.2.2 Population exposure

The second issue that potentially can affect health risk from high ambient PM$_{2.5}$ is the degree of personal exposure. WH pollution events occur almost exclusively at night and therefore the exposure risk depends on how much PM diffuses indoors to affect the residents. PB plume strikes can also occur during the day when there is a possibility that people are outdoors and may be exposed directly to the smoke plume. The question we need to resolve is to what extent buildings can attenuate exposure to pollution events.

Indoor PM$_{2.5}$ concentrations are influenced by indoor sources of PM$_{2.5}$, air exchange rate which defines the ingress of outdoor PM$_{2.5}$ and egress of indoor PM$_{2.5}$, and PM$_{2.5}$ deposition. To calculate indoor PM$_{2.5}$ concentrations from outdoor PM$_{2.5}$ concentrations, we assume that a house is represented as a single well-mixed zone, the air flow through the building shell is constant over the simulation period and there are no indoor PM$_{2.5}$ sources. The indoor PM$_{2.5}$ concentration is described by the differential equation:

$$\frac{dC_{in}(t)}{dt} = P \cdot a \cdot C_{out}(t) - (k + a) \cdot C_{out}(t)$$  \hspace{1cm} Eq. 12

Where:
- $C_{in}$ is Indoor PM$_{2.5}$ concentration
- $C_{out}$ is Outdoor PM$_{2.5}$ concentration
- $P$ is Penetration factor (assumed to be 1 for PM$_{2.5}$)
- $a$ is Air exchange rate (h$^{-1}$)
- $k$ is Decay rate (h$^{-1}$)

A number of Australian studies have measured air exchange rate in residences in several States and across a variety of house characteristics. Building age is a major factor influencing air exchange rates. In general older houses have higher infiltration rates compared to newer homes. A study conducted by Physick (Physick et al., 2008) used values for air exchange rates of 0.6 h$^{-1}$ for residences built before 1990 and 0.3 h$^{-1}$ for residences built after 1990. This was based on the fact that significant changes in infiltration rates occurred after the implementation of the 1990 building code, which removed the requirement for fixed permanent ventilation. In newer homes air exchange rates measured can be as low as ~0.15 h$^{-1}$. When external windows and doors are opened, air exchange rates in excess of 3 h$^{-1}$ were measured.
Biomass burning particles tend to have a bimodal distribution with a dominant fine mode peak at 0.5 μm and a small coarse particle mode at about 10 μm (Keywood et al., 2000). Decay rates of 0.12 h⁻¹ were measured indoors by Abt (Abt et al., 2000) for particle sizes of 0.3 to 0.5 μm and we will use this value for the decay rate in the calculations of indoor PM₂.₅ concentrations.

Figure 34 shows the time variations in measured outdoor and calculated indoor PM₂.₅ concentrations for the autumn generation burn smoke plume event (15-22 April 2010). The maximum PM₂.₅ concentrations are an order of magnitude lower indoors than outdoors. In this situation exposure to peak PM₂.₅ concentrations can be avoided by remaining indoors with windows and doors closed. When the plume has dissipated, opening the windows and ventilating the house rapidly removes any infiltrated PM.

![Figure 34](image)

Figure 34  (a) Averaged 1-minute PM₂.₅ concentrations measured outdoors at Geeveston during smoke plume event in April 2010; (b) and (c) calculated indoor PM₂.₅ concentrations measured during peak outdoor PM₂.₅ concentrations. Three different exchange rates were used: pre-1990 and post-1990 homes and an air exchange rate of 1.5 h⁻¹ representative of a leaky residence.

The options are fewer in the WH season. Nighttime pollution events are persistent therefore indoor and outdoor air equilibrates to an indoor: outdoor PM₂.₅ concentration ratio that depends on air exchange rate. The solutions to Equation 12 for the WH season cases are shown in Figure 35. The PM₂.₅ concentration is attenuated by a significant amount only in well sealed houses.
However the issue may be more academic than real because:

- most of the PB smoke impacts occurred at night (Figure 31); and
- people spend almost all their time indoors. A survey in 1997 of population daily activity patterns found that over 90% of peoples’ time, including recreational, domestic and work time, is spent indoors (ABS, 1997), 57% at home.

Assuming a 90% indoor air exposure and 10% outdoor air exposure, the average exposure of a person during the April 2010 smoke plume event was 30.5 $\mu$g m$^{-3}$ compared to an exposure of 22.4 $\mu$g m$^{-3}$ during the 2010 winter. While short-term PB events such as those observed during April 2010 can be partly avoided by remaining indoors, this option provides minimal protection from persistent WH pollution.

### 7.3 Smoke management strategy

We return to the question of the effectiveness of the smoke management strategy. Prescribed burns emit 99% of total anthropogenic PM emissions yet the annual impact of that smoke on Geeveston was 14% of that caused by the wood-fuelled stoves which emitted the remaining 1% of PM. The largest of these events were caused from fires less than 15 km from the town on a day of low ventilation. The impacts appeared to be localised rather than extensive, and mostly occur at night. Smoke impacts at Grove were minor. This combination of facts suggests that the current management strategy is relatively effective. Most of the smoke is dispersed through the airshed without major impact at the surface.
The same combination of facts suggests that implementation of the management strategy could also be improved. Much of the impact was caused by fires relatively close to Geeveston on days of marginal ventilation. More significantly, the current understanding of PB plume formation and dispersion is not complete and there is a possibility that some of the impacts were caused by smoke emitted into a diminishing mixing layer in late afternoon and early evening. Further, the probability of plume strike across the airshed clearly is non random because Grove was unaffected by fires less than 10 km upwind while Geeveston was fumigated in similar events. Improving the knowledge of these processes could aid in predicting areas at risk from plume strike, assist in the design and management of the burns and reduce PB smoke impacts.
REFERENCES


