Particles, Ozone and Air Toxic Levels in Rural Communities during Prescribed Burning Seasons

C.P. (Mick) Meyer[1], Fabienne Reisen[1], Ashok Luhar[1], Jenny Powell[1], Sunhee Lee[1], Martin Cope[1], Melita Keywood[1], Ian Galbally[1], Simone Linfoot[2], David Parry[3], Lachie McCaw[4], Kevin Tolhurst[5]

[1] CSIRO Marine and Atmospheric Research Aspendale Laboratories, Vic
[2] Department of Conservation and Climate Change, NSW
[3] Charles Darwin University, NT
[4] Department of Environment and Conservation, Science Division, Manjimup, WA
[5] Forest Science Centre, University of Melbourne, Creswick, Vic

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Enquiries should be addressed to: Dr. C.P. (Mick) Meyer
CSIRO Marine and Atmospheric Research
PMB1, Aspendale, Vic, 3187
Email mick.meyer@csiro.au
Phonr (03) 9239 4686

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Executive Summary

Information on air quality in rural Australia is sparse. While smoke from biomass burning such as the prescribed burning of forests, wildfires and stubble burning is often blamed as the major source of air pollution in regional centres, there is little data on the significance of its impact. This study monitored air pollutants at 4 sites in rural Australia between 2006 and 2008. Fine particle concentration ($PM_{2.5}$), ozone (O_3) and BTEX (benzene, toluene, ethylbenzene and xylene) were monitored for 12 months at Manjimup WA, Ovens VIC, and Casuarina NT. PM_{10} was monitored at Wagga Wagga, NSW.

The specific issues addressed in the study were:

1. The seasonal exposure of rural population centres (defined as centres outside the State capital cities with more than 1000 residents) to priority air pollutants resulting from prescribed burning or agricultural waste burning activities.
2. The contribution of emissions from prescribed burning to ambient ozone concentrations experienced by these communities; and
3. The extent to which ambient pollutants from prescribed burning penetrate into houses and, hence, the potential impact of smoke on the total (ambient and in-door) exposure levels of the resident population.

Principal findings

The monitoring program clearly shows that, on occasions, air quality in rural areas is significantly affected by smoke from biomass combustion sources. The main pollutant of concern was $PM_{2.5}$. Significant increases of $PM_{2.5}$ concentrations above background occurred at all sites during periods of wildfire, prescribed fire and crop residue burning activity leading to exceedences of the 24 h $PM_{2.5}$ AAQ NEPM standard on some occasions. The ozone NEPM standard was exceeded only during protracted forest wildfires. Concentrations of the BTEX species were low and never approached the NEPM standard at any site.

At Manjimup the four exceedences of the $PM_{2.5}$ NEPM Standard were caused by plume strikes, mostly from spring burns with one event in autumn from an unidentified source. During these days $PM_{2.5}$ exceeded the NEPM standard for 20% of the time. The site at Ovens in NE Victoria was similarly affected during the autumn prescribed burning season; during this month-long period smoke impacted the town on 12 days of which 7 exceeded the $PM_{2.5}$ NEPM standard. The impact of the 2006/7 wildfires had a substantial impact on Ovens causing exceedence of the

$PM_{2.5}$ NEPM standard on 13 of the 31days monitored during the event. Darwin air quality was affected by savanna fires throughout the dry season, particularly during June and July when daily mean $PM_{2.5}$ concentrations averaged 20 µg. m^{-3}. There were 3 exceedences of the NEPM standard during this period.

The emissions of two smoke tracers, laevoglucosan and non sea salt potassium (nss-K^+), are well characterised for forest fuels in Southern Australia. Analysis of particle samples from Ovens and Manjimup confirmed that more than 80% of the $PM_{2.5}$ recorded during the periods of wildfire and prescribed fire impact were from woody biomass combustion. Preliminary analysis of particle samples collected in Darwin also indicate that biomass combustion was the main source, however it is not possible to be more quantitative without detailed characterisation of the mission properties of Savanna fuels.

Wagga Wagga was more seriously affected by fine particle pollution than the other three sites. During autumn, particulate air quality at the Wagga monitoring station is among the worst of the NSW network. The correlation between the seasonal and diurnal timing of the exceedences and stubble burning activity within the regions strongly suggests that stubble burning is the cause.

Ozone is formed in smoke plumes by photochemical reactions involving NO and VOCs produced during combustion. Elevated ozone concentrations were only observed in plumes exposed to sunlight for several hours prior to impact. These conditions occurred during the wildfire season at Ovens, the spring burning season at Manjimup and in Darwin, but not during autumn prescribed burning periods in Ovens or Manjimup.

The observed concentrations of the BTEX air toxics were extremely low throughout the year at all monitored sites and less than 3% of the 1- year NEPM standards. Annual average concentrations of benzene, toluene, ethyl benzene and xylene were 0.074 ppb, 0.067 ppb, 0.014 ppb and 0.055 ppb at Manjimup and 0.104 ppb, 0.123 ppb, 0.013 ppb and 0.032 ppb at Ovens respectively. The maximum weekly average concentration of benzene, toluene, and xylene observed in this study were 1.1 ppb, 0.6 ppb and 0.3 ppb; these occurred at Ovens during wildfires in December 2006.These compare to the 1 year NEPM standards for benzene, toluene and xylene of 3 ppb, 100 ppb and 200 ppb respectively, and 1-day standards of 1 ppm and 250 ppb for toluene and xylene respectively. In Darwin, average concentration of benzene, toluene, ethyl benzene and xylene between July and September (mid to late dry season) were respectively 0.16, 0.69, 0.095 and 0.47 ppb. Although these are low relative to the NEPM standard, the low benzene to toluene ratio suggests that much of these VOCs were emitted from vehicle rather than biomass combustion sources.

Detailed modelling of smoke transport in the Northern Territory produced predictions of Darwin $PM_{2.5}$ surface concentrations that were consistent with the observations. Predictions of the spatial distribution of $PM_{2.5}$ air quality across the northern region of the Northern Territory showed that most of the region was seriously impacted each year. Hot spots of poor $PM_{2.5}$ air quality with up to 10 exceedences per month were predicted in an extensive region surrounding and to the south of Darwin in the early dry season, and up to 5 exceedences per month in central and SE Arnhem land in the late dry season. There was little interannual variation in the pattern of smoke impact for the modelled period, 2003 to 2007.

The impact of prescribed burning on the indoor air quality of residences depended on the duration of the smoke event and the ventilation rate of the houses. During short-duration events indoor and outdoor air quality was largely decoupled and indoor air quality was determined by household activities. During events that persisted for several days, which occurred at Booralite, Vic during wildfires in January 2007, indoor air quality was determined by external conditions coupled with management of household ventilation rate.

Policy implications and limitations

Biomass burning therefore clearly has a substantial impact on rural population centres in Australia. Prescribed burning impacts lead to $PM_{2.5}$ air quality that is comparable to or significantly poorer than Melbourne, and Launceston (a city noted for poor winter air quality due to woodheater emissions) and comparable to Sydney. During wildfires, air quality of impacted rural towns is almost 100% worse than the worst months in Sydney. Ozone is an issue only during summer wildfires and, to a lesser degree, in the spring prescribed burning season; there was little enhancement during the autumn prescribed burning. The concentrations of air toxics were not an issue, never approaching the air quality standards.

In regions of high fire activity dispersion modelling shows that impacts are protracted and extremely widespread. Therefore, it can not be assumed as is currently the case, that rural air quality is always good and that regional air quality monitoring and regulation is mostly unnecessary. For criteria pollutants other than particulates, the impacts are dependent on season, and duration of plume strike. There are a range of options for minimising impact including:

1. Limiting burning to months of lower solar radiation;

2. Scheduling ignition times to minimise both photochemical ozone formation and the risk of trapping smoke within the town airshed under a nocturnal inversion;
3. Planning burns to optimise smoke dispersion away from towns and valleys; and
4. through operational planning, minimising the duration of smoke impact so that rural populations can avoid exposure.

This study was a pilot study of necessarily limited spatial coverage designed primarily to investigate impacts of prescribed burning. The indoor air study was similarly limited by the short duration of the monitoring period (4-7 days), the small number of houses (4) and the consequently low probability of monitoring during a smoke event.

Areas of the current study that could usefully be strengthened or extended include:

- Characterising the emission properties of the smoke tracers laevoglucosan and nss-K^+ in order to quantify the proportion of $PM_{2.5}$ concentration attributable to biomass burning in the tropical savanna woodlands, grassland and cropping regions;
- Extending the smoke dispersion modelling to both prescribed fires and wildfires in Southern Australia. Time constraints prevented this from being completed in the current study;
- A more comprehensive measurement programme to explicitly target smoke infiltration rates and indoor exposure during the autumn, spring and wildfire seasons; and
- Extending annual or longer-term monitoring for particles and other criteria pollutants to other regions and towns in rural Australia;

Although the study was designed to address prescribed burning, wildfires were found to cause major impacts. The study of wildfire emissions requires a network of monitoring stations, that can be strategically and rapidly located during wildfire events.

The changes in smoke composition with plume ageing is currently poorly understood but potentially significant, particularly when smoke is transported long distances. Secondary pollutants such as ozone are formed during transport and therefore the duration and timing of plume dispersion is likely to affect the toxicity of the smoke that impacts rural towns. Smoke is also commonly transported into urban airsheds raising background concentrations of pollutant species, adding to the burden of industrial, vehicular and residential pollutant sources and adding to the formation of secondary pollutants. This needs to be quantified.

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List of Abbreviations

AAQ	Ambient Air Quality
AGO	Australian Greenhouse Office
AQMS	Air Quality Monitoring Station
ATD	Automatic Thermal Desorber
AVHRR	Advanced Very high resolution radiometer
BoM	Bureau of Meteorology
BTEX	Benzene, toluene, ethylbenzene and xylene
CADR	Clean Air Delivery Rate
CARP	Clean Air Research Programme
CDU	Charles Darwin University
CO	Carbon monoxide
CO_2	Carbon dioxide
DCC	Commonwealth Department of Climate Change
DEC, WA	Department of Environment and Conservation, Western Australia
DECC, NSW	Department of Conservation and Climate Change, New South Wales
DEWHA	Commonwealth Department of Environment, Water, heritage and the Arts
DSE	Department of Sustainability and Environment, Victoria
ETS	Environmental tobacco smoke
GC/FID	Gas Chromatograph with a flame ionisation detector
GCMS	Gas Chromatograph with a mass selective detector
GFED	Global Fire Emissions Database
HPLC	High Performance Liquid Chromatograph
I/O	Indoor to Outdoor concentration ratio
IPCC	Intergovernmental Panel on Climate Change
MCE	modified combustion efficiency
MWR	Monthly Weather Review
NEPC	national Environmental Protection Council
NEPM	National Environmental Protection Measure
NGGI	National Greenhouse Gas Inventory
NHMRC	National Health and Medical Research Council
NO	Nitric oxide
NO_x	Odd nitrogen oxides
$nssK^+$	non-sea-salt Potassium
NSW	New South Wales
NT	Northern Territory
O_3	Ozone
PAN	Peroxyacetyl nitrate
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
ppb	parts per billion (1 part in 10^9)
ppm	parts per million (1 part in 10^6)
Qld	Queensland

RFS	Rural Fire Service, New South Wales
RO_2	Peroxy radical
SA	South Australia
SOA	Secondary Organic Aerosol
TAPM	The Air Pollution Model
TAPM/CTM	The Air Pollution Model coupled with the Chemical Transport Model
Tas	Tasmania
TEOM	Taopered Element Oscillating Microbalance
ug	migrogram (10-6 g)
Vic	Victoria
VOC	Volatile Organic Compound
WA	Western Australia

1 INTRODUCTION

The impact of particulate pollution on the residents of rural communities is emerging as a significant issue. While generally, air quality in rural areas is considered good in comparison to cities there are occasions when this is clearly not the case. In cities, the sources of pollution are predominantly transport, industrial and to a smaller degree domestic activities, particularly domestic woodheaters. The low population density in rural areas means these sources are minor. Pollution mostly results from agriculture and forestry activities, and often involves smoke from biomass combustion. In these cases the pollution events can be severe, if short-lived and leave a strong perception of poor air quality in residents they impact. However quantitative information on long term air quality in rural regions is sparse.

The main sources of smoke in rural regions is produced by residential heating by wood stoves, prescribed burning of forests for forest management, agricultural waste burning, and wildfires. Prescribed burning is largely for fuel reduction purposes, but also include removal of logging slash and seed bed regeneration following logging. Agricultural waste burning is predominantly of cereal stubble for a range of crop management purposes including disease control, soil nitrogen management and to reduce the impacts on farm machinery. Wildfires, while rarely the direct result of forest management; can have a major and occasionally a protracted impact on air quality of large regions. The extensive fires in NE Victoria in the summers of 2003 and 2006 and Southern NSW in summer 2003, for example, adversely affected a large fraction of Victoria for several weeks.

Concern about reduced air quality, particularly from fine particle pollution has been increasing both locally and internationally over several decades as the associated health risks have become apparent. These derive from both acute and chronic effects of population exposure to fine particles and air toxics components contained in smoke. For the most part, the chemistry and health effects of polluted air have been determined for urban air masses which are largely influenced by industrial emissions vehicular

emissions and domestic woodheater emissions. Biomass burning produces a significantly different mix of reactive precursors to these urban sources and, therefore, the risk to rural air from photochemical ozone production from smoke precursors cannot be inferred from urban studies. There is robust evidence from international meta-analyses that increases in concentrations of particulate matter and fire-related toxins in the air are associated with moderate increases in mortality rates and morbidity levels of exposed populations (Brunekreef and Holgate 2002). In addition, most of the health effects calculated in existing meta-analyses are based on scenarios involving the burning of fossil fuels (i.e. urban air chemistry) rather than biofuels, and it remains to be seen whether similar effects are observed in scenarios involving smoke from biomass combustion.

Another issue is that the dose indicated by average ambient levels may not accurately reflect the actual exposure for any individual (Sarnat et al. 2006). People may actively avoid exposure to high smoke concentrations, or because of the task or location be unavoidably exposed to high smoke concentrations. For these reasons, there is often a poor correlation between ambient levels of air pollution and individual exposures despite the epidemiological evidence to support the relationship between air pollution and health consequences.

Therefore despite the possibility and the perception that smoke from biomass burning poses air quality and health risks in rural Australia, the issue has not been comprehensively investigated to date nor is it well understood (Dennis et al. 2002). The current understanding of air quality in rural Australia, and its drivers is in its infancy.

This project was designed to investigate the impact of smoke from prescribed burning in forests and woodlands, and from agricultural residue burning on the air quality of four population centres in regional Australia during 2007. Several aspects of air quality were considered. The project focused on two major components of air pollution: the mass concentration of particles less than 2.5 μm in diameter ($PM_{2.5}$), and ozone (O_3) which is both a criteria pollutant and primary indicator of reactive chemistry (e.g.

photochemistry). The toxic gas species benzene, toluene, ethylbenzene and xylenes were also investigated. The specific questions addressed in the study were:

1. what is the seasonal exposure to priority air pollutants of four rural communities each of which is in relatively close proximity to prescribed burning or agricultural waste burning activities.
2. To assess the contribution of emissions from prescribed burning on ambient ozone concentrations in these communities;
3. To assess the extent to which ambient pollutants from prescribed burning penetrate into houses, and hence assess the potential impact of smoke on the exposure levels of the resident population, and
4. To assess the factors that influence population exposure levels in order to inform policy development.

The four regions were NE Victoria and SW Western Australia, both of which are affected by prescribed burning of forests; Darwin, which is affected by the extensive burning that occurs annually in the tropical savanna woodlands; and Wagga Wagga NSW located in the Riverina region where extensive dryland and irrigated cereal cropping and stubble burning occurs These data are expected to be of use to rural communities that currently have relatively little information regarding their local air quality (particularly in relation to the health-based air quality measures) and to fire and land managers.

2 PROJECT DESIGN

The perception is that smoke from fires seriously degrades air quality in some regions of rural Australia for significant periods of time. The challenge is to quantitatively assess the air quality of rural regions in absence of biomass smoke against which the significance and the impact of smoke pollution can be determined. The first issue, therefore, is:

1. to characterise the seasonality of air quality in representative rural locations, particularly in the months when fire activity is absent or low, in order to establish the baseline of air quality;

2. to measure the frequency and duration of smoke plume strikes associated with fire events, and

3. to assess the impact of other smoke sources such as domestic woodheaters

The second issue is the extent to which rural residents are likely to be exposed to ambient smoke during fumigation events. Most residents, even those with outdoor occupations spend a large fraction of their time indoors, and in the event of a plume strike may actively seek to avoid direct exposure. This raises questions of:

1. Indoor air quality in the absence of ambient smoke; and,

2. The rate and extent of infiltration of ambient smoke into the residences.

The third issue is, in the event that fires pose a significant risk to air quality in rural population centres, how can the risks be managed and minimised without negative impacts from other environmental processes such as increased severity or frequency of wildfires, or negative consequences for rural industry.

The project was designed with three components, each addressing one of the issues.

1. The first component is monitoring programme in four representative regions of rural Australia to monitor local air quality for a minimum of a year. The air quality parameters assessed are (a) fine particle mass $PM_{2.5}$, which is the major component of smoke, (b) surface ozone concentration, which is a measure of the reactivity of the atmosphere and therefore the extent to which chemistry within smoke plumes lead to the formation of secondary products and (c) the concentration of the toxic Volatile Organic Compounds (VOCs) which can both pose a health risk and, more significantly for this study, are the precursors for reactive atmospheric chemistry. The class of VOCs measured in this study, BTEX, address both aspects. Additionally, in order to distinguish $PM_{2.5}$

associated with smoke from particulate pollution from other sources, particle samples were analysed for specific smoke tracers, non sea salt potassium (NSS K) and laevoglucosan.

2. The second component of this project is a small pilot study of the indoor air quality of four houses during a plume strike. This component was aimed to assess how closely the indoor and ambient atmospheres are coupled. Two approaches were used: monitoring of indoor and ambient air quality during a plume strike, and measurement of the key parameters that determine the rate of infiltration of ambient smoke into houses. The air quality indicators used in this component were $PM_{2.5}$ and BTEX.

3. Management of the air quality risks requires tools that can effectively predict the transport of smoke from source to town. These can be used to assess the spatial distribution of high risk areas given the current distribution and activity of current emission sources and to assess the impacts of alternative emission scenarios that might be introduced through policy instruments. The third component of this study applies the CSIRO atmospheric transport model TAPM to one of the study locations to assess (1) the accuracy of the transport predictions and (2) the spatial distribution of air quality impacts from fire within a large region. The aim is to extend this to the other 3 study regions. Two aspects are considered: (a) smoke transport and (b) the extent of chemical transformations within the smoke plume and within the population centres leading to secondary chemical products including ozone and secondary aerosol.

These three components were developed as follows:

1. The selection of four sites in rural Australia in regions impacted by smoke. These are (a) NE Victoria and (b) SW Western Australia, both of which are affected by prescribed burning of forests for fuel load reduction. (c) Darwin, NT, which is affected by the extensive burning that occurs annually in the tropical savanna woodlands; and (d) Wagga Wagga and Albury, NSW located in the Riverina region where extensive dryland and irrigated cereal cropping with associated stubble burning occurs.

2. Air quality monitoring stations were installed in two regions and existing stations were supplemented with adding additional instrumentation to measure an annual cycle of particulate and reactive air quality. New low-cost monitoring stations were installed in NE Victoria and SW WA. These stations comprised a continuous $PM_{2.5}$ monitor (which is not an air quality standard instrument), a $PM_{2.5}$ filter sampler (which is a standard ambient air quality method for $PM_{2.5}$ gravimetric mass, but which has low time resolution), passive BTEX samplers and a UV photometric ozone monitor, both of which are also Australian air quality standard methods. In Darwin, NT, an established monitoring programme for 24-h mean PM_{10} and $PM_{2.5}$ by Partisol, and continuous PM_{10} concentration by TEOM (both of which are Australian Standard methods for ambient air quality) was supplemented with a photometric UV ozone monitor and weekly sampling of BTEX by passive adsorbent tubes. Wagga Wagga and Albury, NSW, both have standard Air Quality Monitoring Stations for PM_{10} comprising TEOM and standard meteorological instruments. The Victorian and WA stations were installed in December 2006 and have operated continuously since that date. Ozone and BTEX monitoring was commenced in Darwin in April 2007 at the beginning of the fire season and continues to operate. Monitoring of particle chemical composition and BTEX commenced in Wagga in January 2008 and will not be presented in this report.

3. The indoor air quality study comprised the monitoring of indoor and ambient air quality in four houses in NE Victoria for a week during autumn 2007 during a prescribed burning programme. This was later supplemented by measurements of air exchange rate, which is the primary determinant of smoke infiltration into the indoor environment.

4. The third component of the study centred on the application of TAPM to predict the transport of $PM_{2.5}$ emitted by fires. This was aimed at predicting the hourly surface concentrations of $PM_{2.5}$ within the model domain from which the spatial distribution of air quality could be derived. The study concentrated primarily on the "top end" of the Northern Territory. Most of this region which consists of savanna woodlands which are extensively and regularly impacted by burning.

This component of the project assessed both annual and seasonal variation in smoke emissions and their impact on surface concentrations of $PM_{2.5}$ in the region. The air chemistry model, CTM, was used in conjunction with TAPM to assess the extent to which smoke contributes to ozone chemistry and secondary aerosol formation. These model predictions are constrained by comparison with surface observations from the monitoring programme. The focus of this component is the development of modelling approaches that, in the future, could be used to develop and assess management policies.

In combination the project design provides sufficient data to assess the initial question of whether prescribed burning and stubble burning cause significant negative impact on regional air quality and how to develop the support tools needed to assess and manage any impacts.

3 PRESCRIBED BURNING ACTIVITY IN AUSTRALIA

Biomass combustion in Australia is a major source of pollutants to the atmosphere. The extent of these emissions are estimated using standard methodologies defined in the Australia's National Greenhouse Gas Inventory methodology (NGGI) (AGO, 2007; Meyer, 2004) and the 2006 IPCC Guidelines (Eggleston et al., 2006) report emissions in four general categories: (1) fires in savanna and temperate grassland, (2) prescribed burning of forests, (3) wildfires in forests and burning of agricultural residues. In the Australian context savanna fires include fires in the tropical savanna woodlands of northern Australia, the arid grasslands of central Australia and temperate grasslands in southern Australia. Almost all fires in the tropical savanna woodlands and most fires in the arid rangelands are from anthropogenic causes; lightning is a minor source of ignition. Prescribed burning of forests is carried out by State Fire Authorities under prescription, mostly for reduction of forest fuel loads but also for silvicultural purposes. Wildfires are forest fires of all other classes. Fuel reduction burning is generally undertaken to limit the risk and severity of wildfires.

The mass emission of a pollutant species *i* (CO_2, CO, VOCs, $PM_{2.5}$, and other trace compounds) is estimated from the mass of fuel burned (M) the carbon content of the fuel (CC) and the emission factor (EF_i).

$$E_i = M * CC * EF_i \qquad (3\text{-}1)$$

In turn, the mass of fuel burned is estimated from the fire area (A), the fuel load (FL) and the burning efficiency (ξ)

$$M = A * FL * \xi \qquad (3\text{-}2)$$

Expressed in these terms the emission of any trace species is proportional to the emission of total carbon from the fire. From the annual statistics collated for the NGGI (Meyer, 2008), the annual emissions of total carbon from natural and forest fires in

Australia is very large, averaging 78 Mt C from 1983 to the present (DCC, 2008; Meyer, 2004), with large interannual variability (Figure 3-1). This quantity is similar to the total annual carbon emitted from the burning of fossil fuel in Australia (DCC, 2008). Savanna burning contributes 87 %, with wildfires and prescribed fires accounting for 10 % and 3 %. The savanna woodlands of the Kimberley and Arnhem Land contribute from 58 % to 88 % of total annual savanna emissions. Total prescribed burning emissions are relatively similar from year to year (Figure 3-2A), however emissions from wildfires are highly variable (Figure 3-2B); in extreme fire years (e.g.2003) emitting almost as much carbon to the atmosphere as savanna fires.

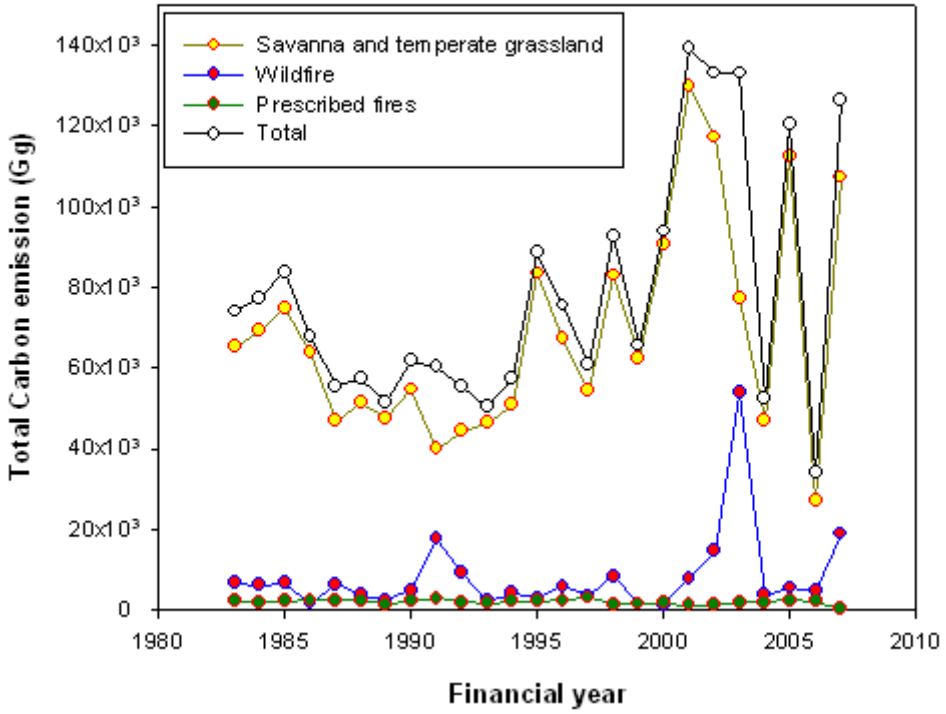


Figure 3-1 The time course of total carbon emissions in Australia between 1983 and 2007 from prescribed burning and wildfires in temperate forests, tropical savanna woodland and temperate grassland.

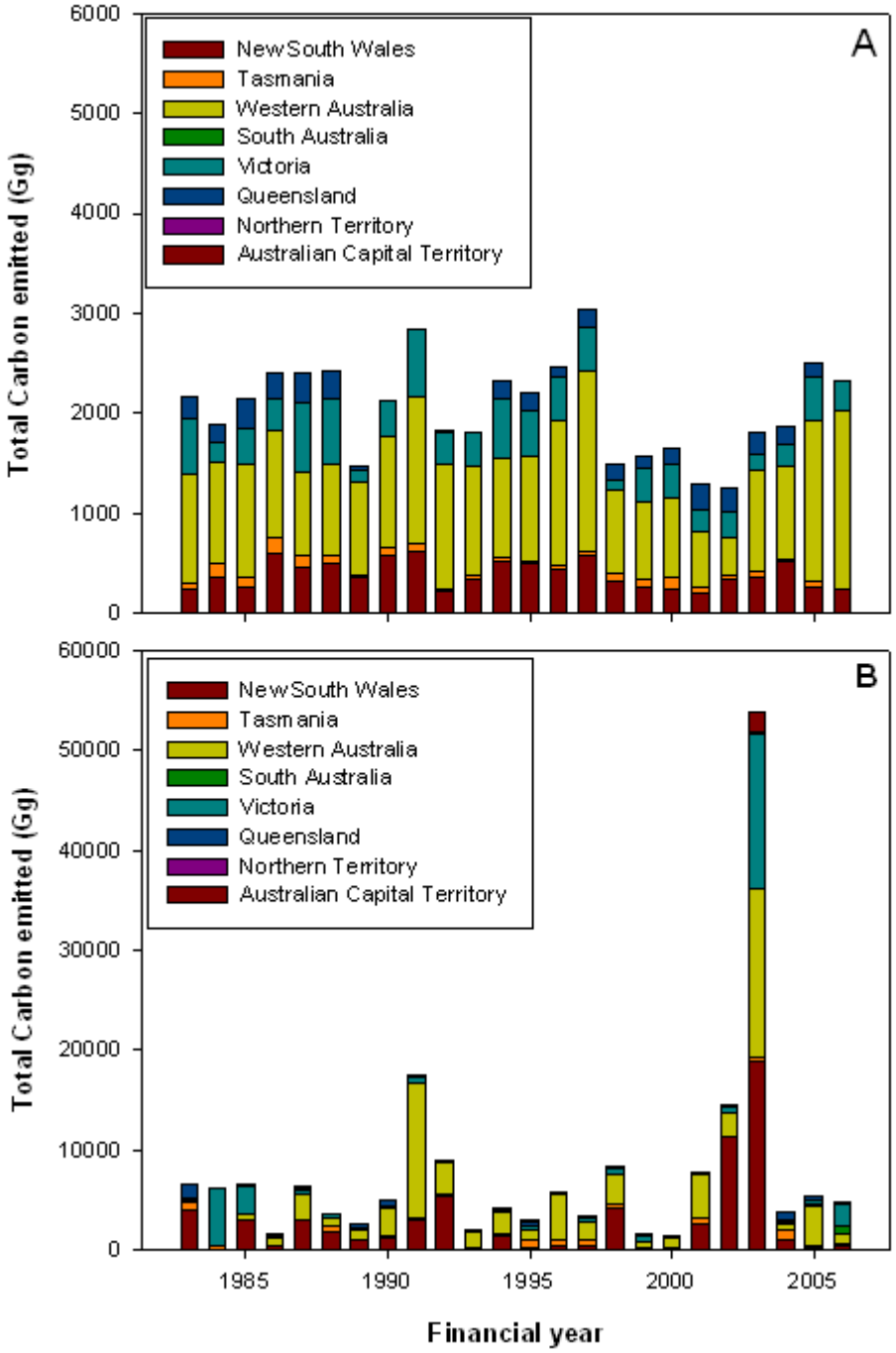


Figure 3-2: Total carbon emissions from (A) prescribed fires and (B) wildfires in Australia between 1983 and 2006. Source: Data supplied to the National Greenhouse Gas Inventory

Western Australia undertakes the largest prescribed burning programme in Australia (Figure3-2A), predominantly in the Jarrah and Karri forests in the SW region of the State and in some years their emissions exceed those from wildfires. Victoria and NSW

also have extensive prescribed burning programs; however these rarely contribute greater emissions than wildfires (Figures 3-2A, 3-2B).

Agricultural residue burning emits similar quantities of carbon as prescribed fires. Between 70 % and 80% is from the burning of cereal stubble.

Prescribed burning in forests occurs when the fuel moisture and weather conditions limit fire intensity and rate of spread; the aim is low intensity, cool burns sufficient to consume fine fuels on the forest floor without causing damage to the mid storey and trees. The period when these conditions occur varies between states. The seasonality of prescribed fire number and fire area in Victoria and WA are presented in Tables 3-1 and 3-2 respectively. In WA, prescribed burning was confined mostly to late autumn (May and June) and late spring/early summer (November and early December). In Victoria the timing of the fires was not identified in the statistics in a significant fraction of the cases. However for the cases where the timing of the fire was known prescribed burning occurred mostly in early autumn (late March and April,) and to a smaller extent in late spring (November to early December). These windows shift slightly depending on the start of the autumn and winter rains.

Table 3-1 Average monthly fire frequency (%) for the 2000–-2006 fire seasons in Victoria and the 2005-2008 fire seasons in SW WA, presented as number of fires. In the case of Victoria, dates were not identified for many fires in the data base and are listed as unspecified.

Month	Number (% per month)	
	Victoria 2000-2006	WA 2005-2008
January	0	3
February	1	2
March	7	8
April	30	14
May	10	14
June	0	1
July	0	0
August	0	2
September	0	10
October	0	15
November	0	12
December	7	18
Unspecified	46	

Table 3-2 Average monthly fire frequency (%) for the 2000–-2006 fire seasons in Victoria and the 2005-2008 fire seasons in SW WA, presented as number of fires. In the case of Victoria, dates were not identified for many fires in the data base and are listed as unspecified.

Month	Area (% per month)	
	Victoria 2000 to 2006	**WA 2005 – 2008**
January	0	7
February	0	3
March	6	1
April	24	9
May	9	17
June	0	3
July	0	0
August	0	1
September	0	3
October	0	23
November	1	24
December	8	9
Unspecified	52	

Fire activity in Northern Australia is determined by the monsoons. Currently fires take place throughout the dry season from late April to late November (Figure 3-3). Fire frequency typically increases throughout the season rising to a maximum in the late dry season in October (Figure 3-3A). The difference between low activity fire seasons (2005) and intense fire seasons (2004) is an increase in early-season fires and a reduction in late-season fires (Figure 3-3B).

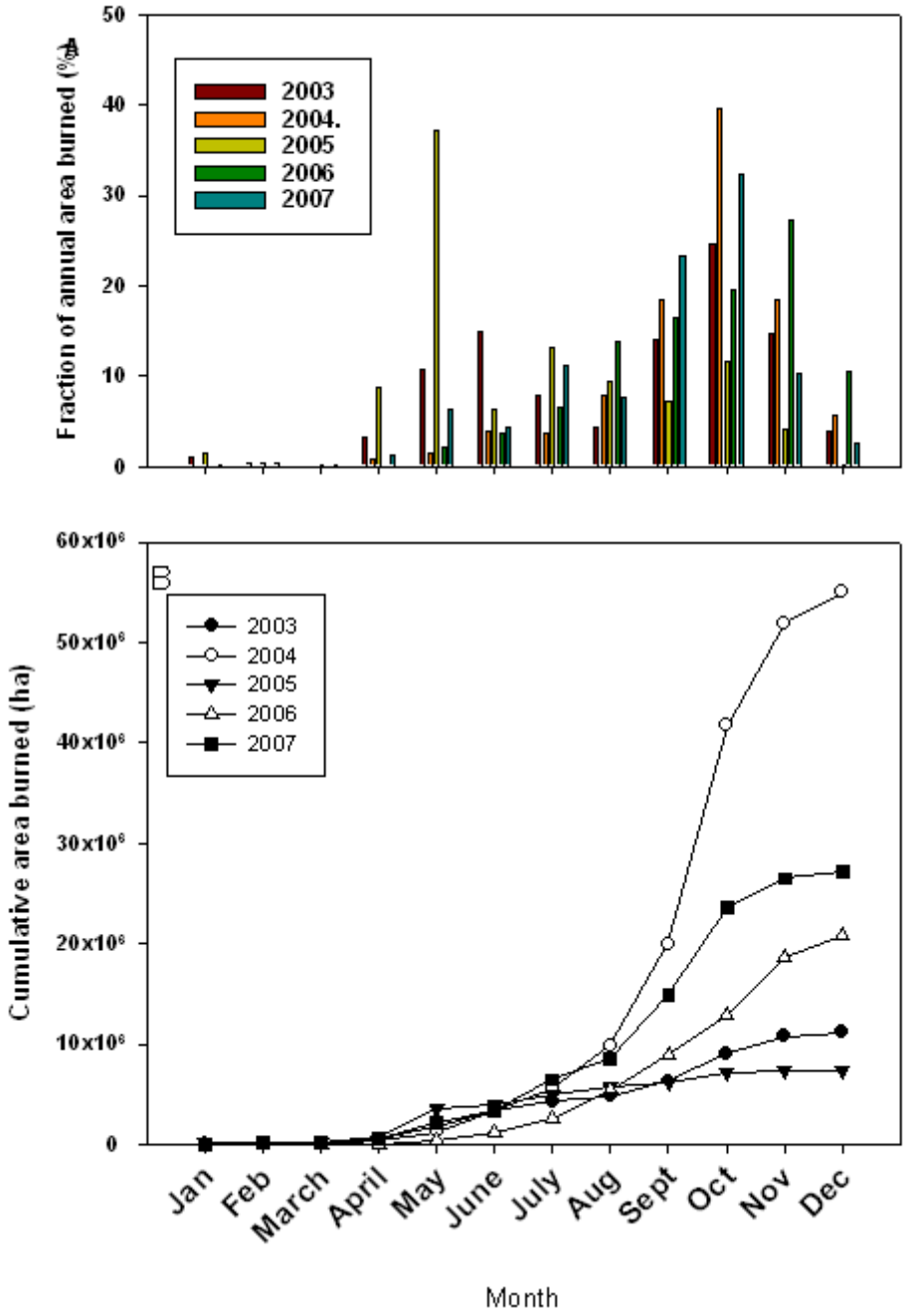


Figure 3-3: Seasonal distribution of fire activity in the Top-End for 2003-2007. (A) Fraction of total area burned annually; (B) cumulative total area

In summary, in Australia most of the fire activity and emissions occur in Northern Australia. It occurs with high frequency throughout the dry season to a maximum in October. Prescribed burning in southern Australia takes place in relatively narrow windows during autumn and spring. In Victoria the emphasis is on early autumn

burning, while in WA the burning takes place both in late autumn and late spring, slightly more in spring than autumn.

The difference in location, size and frequency of fires is likely to have a strong influence on the impact of smoke plumes on regional populations. The Northern fires season is protracted and extensive, while the prescribed burning season in Southern Australia is confined to two relatively narrow windows.

4 SELECTION OF STUDY LOCATIONS

The study addressed four locations in Australia, each with distinct characteristics. The criteria for site selection are a balance between the practical and optimal. The four regions chosen were the NT, where a large fraction of Australia's fire activity occurs with significant and protracted impact on surface air quality (Luhar et al., 2008), SW WA where the most active prescribed burning programme in Australia is centred, NE Victoria and the Riverina where there is substantial amounts of cereal stubble burning. An air quality site was selected in each of the regions which met the following criteria. Each site must:

1. be located within or near a town with a population of at least 1000 people;
2. be located within a region of substantial prescribed burning or other fire activity;
3. have a significant risk of smoke impact;
4. be reasonably close to research facilities for weekly maintenance, or
5. where possible, supplement existing air quality monitoring facilities.

Four locations which met these criteria were,

Casuarina, NT (the Charles Darwin University Campus)
Manjimup, SW WA (DEC regional office)
Ovens NE Victoria (DSE regional office), and
Wagga Wagga, NSW (DEC Air quality monitoring station)

The Casuarina site is located within Darwin, and is impacted by the prevailing SE winds that transport smoke from the fires in the woodlands to the SE. Figure 4-1, shows the area of the NT burned in 2006 and the location of towns with populations greater

than 1000. Darwin is ringed by fires activity. The current PM monitoring programme conducted by CDU was supplemented with ozone, BTEX and particle chemistry measurements

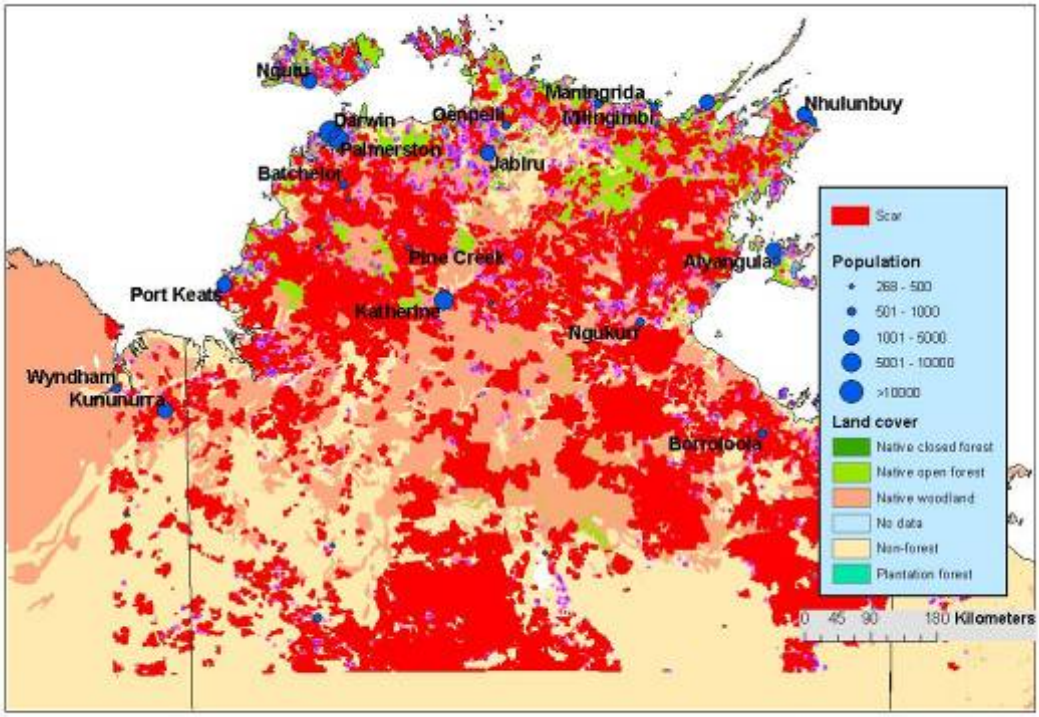


Figure 4-1 Fire area in the Northern Territory in 2007

The DEC, WA Regional office in Manjimup is located in the centre of the SW Karri and Jarrah forests from which the prescribed burning programme is conducted. The town is regularly impacted by smoke plumes from both near-field and distant fires. The town is located at an elevated and well ventilated site, where smoke masses disperse relatively quickly. Because fires are frequently distant from the town, smoke plumes range from fresh to aged, but unlikely to be from a combination of fire events. The location of Manjimup relative to areas of prescribed burning and other population centres is shown in Figure 4-2.

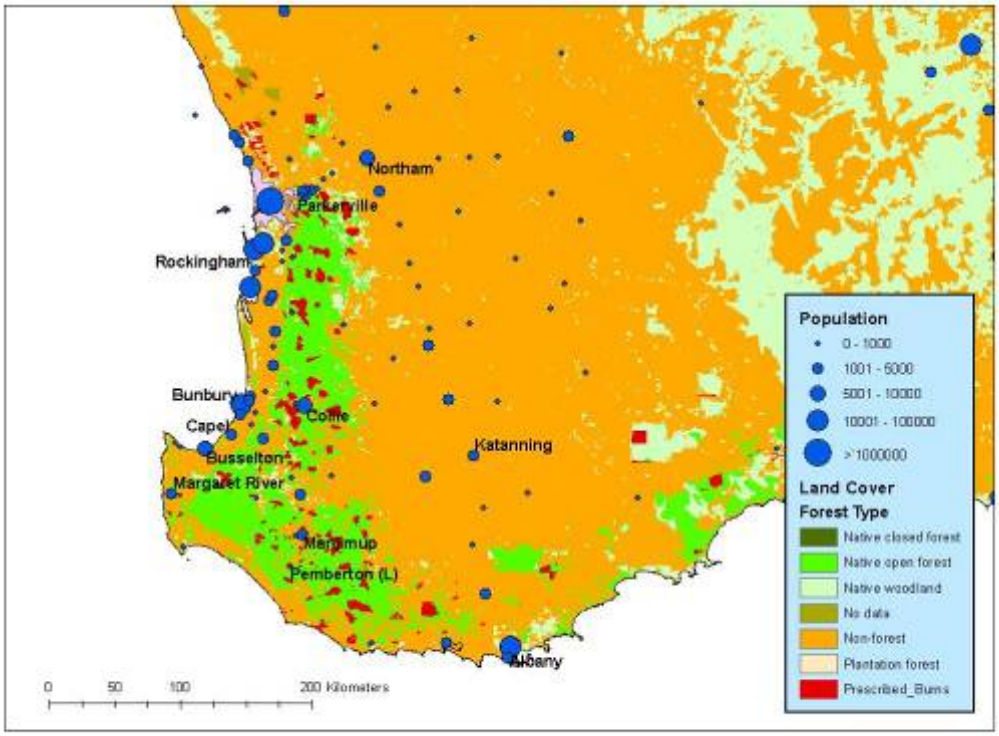


Figure 4-2 The spatial distribution of prescribed burning activity in southwest WA in 2005/06 and its proximity to population centres

The DSE, Vic regional office at Ovens, is located in the Ovens Valley approximately 10 km from the population centre of Myrtleford. The Ovens valley channels smoke emitted from fires in the surrounding forest to the plains to the West. Smoke tends to accumulate in the valley, dispersing slowly. In extreme case such as 2003 and 2006 wildfire seasons, dense smoke persisted in the valley for several weeks. This site is characterised by protracted fumigation events with smoke originating from a combination of different sources, either fresh or aged. Figure 4-3 shows the location of Myrtleford relative to surrounding forest, prescribed burning activity and population centres.

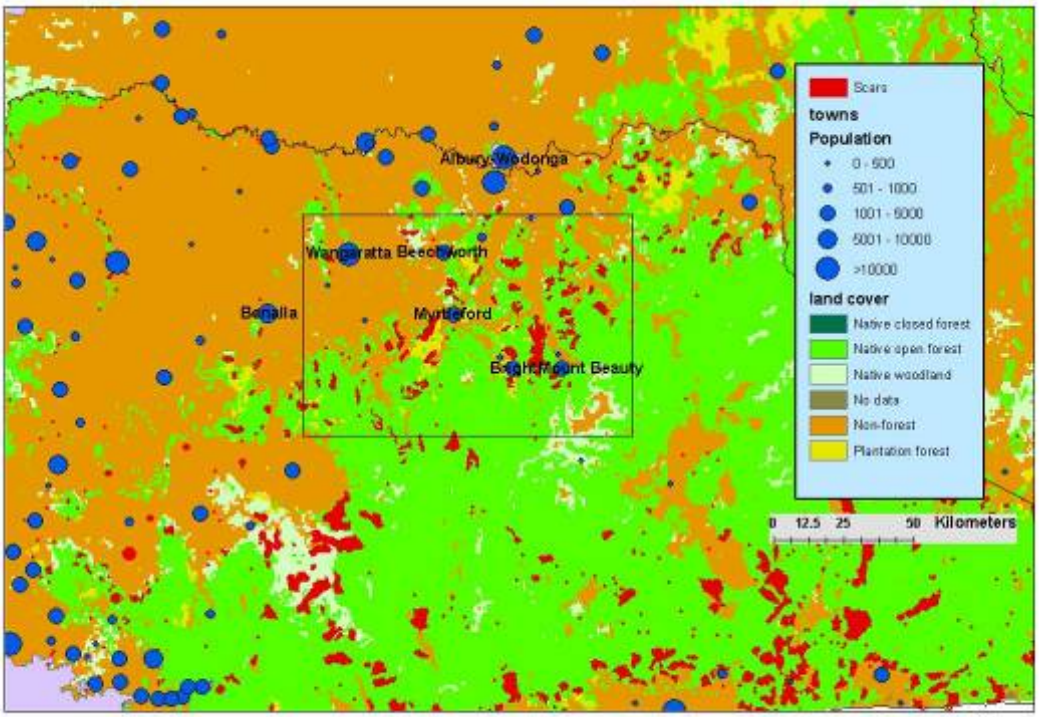


Figure 4-3 The spatial distribution of prescribed burning in NE Victoria for the 2005/06 fire season.

The air quality monitoring station operated by DEC, NSW is located to the west of the extensive cereal cropping region in the Riverina.

In summary, these four locations monitor distinctly different classes of smoke impact. Casuarina, NT monitors seasonal and persistent smoke impact in the tropics. Manjimup monitors fresh smoke plumes. Ovens monitors aged and persistent smoke plumes, and Wagga potentially monitors emissions from cereal stubble fires.

5 OBSERVED AIR QUALITY AT KEY SITES

5.1 Introduction

Air quality monitoring stations (AQMS) were installed at four locations within each region of key importance referred to in the previous section: Casuarina in the Northern Territory, Manjimup in Western Australia, Ovens in Central Victoria and Wagga Wagga in New South Wales.

Currently little information is available on air quality in rural communities where population exposures to bushfire smoke are likely to occur. Previous research of bushfire smoke impact on communities has focused on major bushfire events and has shown that the primary pollutant consistently exceeding air quality guidelines during bushfires was particulate matter. Particles in the air range from 0.01 μm to about 50 μm. Larger particles will fall out within hours, but smaller particles can stay suspended in the air for days or weeks until removed by rain. Particles can harm human health when inhaled. Current medical research shows that particle pollution can exacerbate existing respiratory symptoms and at high concentrations can cause such symptoms. Particles can also adversely impact cardiovascular health.

During bushfire events particularly during periods of intensive fires, large quantities of hazardous air contaminants are likely to be emitted and may exceed urban air quality guidelines. Exposures during prescribed burning are episodic and transitory.

Observations of key air pollutants over a 1-year sampling period allow us to address some of the issues including monitoring the seasonal exposure of communities to hazardous pollutants released during prescribed burning activities and assessing implications of such exposures in relation to existing (urban) air quality measures. The pollutants that were monitored included particulate matter, ozone and the VOC group benzene, toluene, ethylbenzene and xylenes (BTEX).

5.1.1 Air quality guidelines

It is well known that urban communities are exposed to a range of air pollutants associated with urban activities, especially vehicle emissions. Ambient air quality standards and goals for six criteria pollutants and five air toxics were set in the National Environment Protection Measure for Ambient Air Quality (AAQ NEPM) (NEPC 2003) and in the National Environment Protection Measure for Air Toxics (NEPC 2004) to control exposures to pollutants. The air quality guidelines relevant to this report are summarised in Table 5-1. The NEPM standard for PM_{10} is 50 $\mu g/m^3$ as a 24-hour average with a goal of no more than five exceedence days per site per year. The goal for particles is designed to allow for natural events such as dust storms and wildfires.

Table 5-1 National Environmental Protection Measures (NEPMs) for ambient air quality relevant to this study

Air contaminant	Average period	Maximum concentration	Maximum allowable exceedences
Photochemical oxidant (as ozone)	1 hour	0.10 ppm	1 day per year
	4 hours	0.08 ppm	1 day per year
Particles as PM_{10}	1 day	50 $\mu g/m^3$	5 days per year
Particles as $PM_{2.5}$	1 day	25 $\mu g/m^3$	Gather sufficient data for review in 2005
	1 year	8 $\mu g/m^3$	
Benzene	1 year	0.003 ppm	Gather sufficient data to develop standard in 2009
Toluene	1 day	1.0 ppm	Gather sufficient data to develop standard in 2009
	1 year	0.10 ppm	
Xylenes	1 day	0.25 ppm	Gather sufficient data to develop standard in 2009
	1 year	0.20 ppm	

5.1.2 Atmospheric processes

Wherever air pollution exists, three elements are important (1) emissions, (2) meteorology and topography and (3) atmospheric chemistry. While primary emissions are an important factor of air pollution, certain weather and topography can either mitigate the effect of pollutant emissions through dispersion or can keep emitted pollutants concentrated. If atmospheric reactions occur, secondary products, such as ozone and secondary aerosols are formed.

Particles are produced by combustion processes including vehicle emissions, industrial emissions and biomass burning. Particulate mass (PM) levels often display traffic

related peaks in urban areas in particular in the morning between 7:00 and 9:00, with a less pronounced traffic peak in the evening between 17:00 and 19:00.

Meteorology and topography play a major role in the dispersion of pollutants. Inversion layers are partly responsible for the diurnal pattern of air pollutants. The height of an inversion layer is lowest during the night and increases after sunrise to a maximum during the early afternoon. The shallow nocturnal boundary layer traps pollutants leading to enhanced surface levels during the night. After sunrise, the inversion height rises increasing the vertical mixing, and hence dilution of pollutants, resulting in a decrease in pollution levels. The location of towns and the surrounding topography will also influence wind conditions and smoke plume dispersion and are major factors contributing to pollution levels.

Additionally pollutants levels are governed by chemical and physical transformations that occur in the atmosphere. Ozone is formed from reactions of hydrocarbons emitted by anthropogenic and biogenic sources, NO and sunlight. This results in net photochemical formation of ozone during the day and is often highlighted in the diurnal pattern of surface ozone concentrations. There is also the potential for the formation of secondary aerosols.

5.1.3 Smoke events

Fine particles emitted from biomass burning are within a size range that scatters light most efficiently. Some of the effects include reduced visibility, but due to their small size range, the impact on human health is also significant. Smoke from biomass burning can produce a haze that may last for several days and may affect a wide area. In this study we define smoke events as occurring when $PM_{2.5}$ levels stay significantly above 25 $\mu g/m^3$ for several hours. In all cases these events were unambiguously different from the background $PM_{2.5}$ concentration timeseries. The measurements that were made, their measurement frequency, the duration of the record and the result of the analyses are listed in Table 5-2.

Table 5-2 Summary of air quality monitoring at stations established or supplemented by this project

Species	Air Quality Measuring Station			
	Casuarina	**Manjimup**	**Ovens**	**Wagga Wagga**
$PM_{2.5}$	TEOM continuous 2005 to Dec 07	Dustrack continuous Dec 06 to Dec 07	Dustrack continuous Dec 06 to Jan 08	TEOM continuous
$PM_{2.5}$ gravimetric mass	Partisol weekly Dec 06 to Sep 07 Analysed by NTU	Micro-Vol weekly Dec 06 to Dec 07 Average $PM_{2.5}$ = 9.5 ± 0.95 µg m^{-3} (n=55)	Micro-Vol weekly Dec 06 to Jan 08 Average $PM_{2.5}$ = 19.0 ± 1.9 µg m^{-3} (n=57) Max $PM_{2.5}$ = 212 µg m^{-3}	Dec 07 to present
$PM_{2.5}$ Smoke tracers		nssK^+ average = 0.04 ± 0.004 µg m^{-3} Laevoglucosan average = 0.7 ± 0.07 µg m^{-3}	nssK^+ average = 0.11 ± 0.01 µg m^{-3} Laevoglucosan average = 1.22 ± 0.12 µg m^{-3}	
Ozone	Monitor Labs Instrument continuous Mar 07 to Dec 07	TECO 49 Instrument continuous Dec 06 to Dec 07	TECO 49 Instrument continuous Dec 06 to Dec 07	
Benzene, Xylenes and Toluene (BTEX)	Adsorbent tubes weekly June 06 to Dec 07	Adsorbent tubes weekly Dec 06 to Dec 07	Adsorbent tubes weekly Dec 06 to Dec 07	

5.2 Methodology

5.2.1 Ozone

Ozone concentrations were measured using UV photometric ozone analysers. TECO model 49 analysers were installed at the Ovens and Manjimup sites, and a Monitor Labs Model 9810 analyser was installed at Casuarina, NT. Air samples were drawn from at least 5m above the surface through a Teflon inlet, via a zero-span unit and through the analyser. Analyser sensitivity was checked by calibration against an ozone transfer stand at the time of installation or immediately prior to deployment to the field location. In operation, the instruments were automatically zeroed for 10 minutes every 6 hours using sample air from which the ozone had been removed by a MnO_2 scrubber. Instrument span was checked daily by exposing the instrument to sample air in which the ozone concentration had been enhanced by exposure to UV radiation in the range 190 -200 nm produced by a low pressure mercury vapour lamp. These span checks were not calibrations but were used only to confirm normal and reproducible instrument response.

5.2.2 DustTrak

The DustTrak (TSI Inc, USA) is a laser photometer that provides real-time measurements based on 90 degree light scattering. The DustTrak was operated with a $PM_{2.5}$ impactor. It was configured to automatically transmit aerosol mass concentration values via an analogue output voltage signal that is proportional to the displayed concentration. The analogue output has an output voltage range of 0-5 VDC with various scaling ranges. For any concentrations above the scaling range, the analogue output voltage remained fixed at the maximum of 5 Volts. Zeroing of the DustTrak was performed automatically every 6 hours.

The particle mass concentration is determined by the amount of light scatter, based on a calibration factor. The relationship between light scattering and particle mass concentrations varies with particle source and particle size distribution. Furthermore different responses were observed among individual DustTraks in particular at low particle levels. For these reasons, there is no single factor relating DustTrack detector signal to particle mass concentration; each instrument must be compared continuously with an independent gravimetric measure of particle mass concentration.

5.2.3 TEOM

Particles as PM_{10} were measured using a Tapered Element Oscillating Micro-balance (TEOM) in accordance with AS 3580.9.8.2001 (Standards Australia, 2001). The TEOM measurement system is based on the use of an oscillating tapered element, which has a replaceable filter attached to its end. The air sample is drawn through the filter and then through the stem of the tapered element, and as particulate mass collects on the filter element, the frequency of oscillation of the tapered element changes. The change in frequency provides a direct measure of the particulate mass collected over time. As flow controllers constantly measure the volume of air sampled, a direct and continuous measure of PM_{10} is obtained. The instrument provides continuous one-hour average data. Daily (24-hour) averages are calculated using these hourly data.

5.2.4 MicroVol

Particles as $PM_{2.5}$ were collected using a low-volume aerosol sampler in accordance with AS 3580.9.10:2006. The aerosol sampler used was a Microvol-1100 (Ecotech Pty Ltd, Knoxfield Australia) fitted with a $PM_{2.5}$ size-selective inlet containing a greased impaction plate. The sample stream through the inlet and filter was maintained at a constant flow of 3 lpm using a mass flow meter in the unit. The voltage output of the MicroVol was logged at 1-minute intervals, and used to determine the flow rate of the MicroVol during the sampling period. Samples were collected on 47 mm Teflo[1] filters (Pall R2PJ047, 2 μm pore size).

NATA-accredited gravimetric mass measurements on the pre-exposed and exposed filters were made in the aerosol mass laboratory (Accreditation Number 245) at the CSIRO Marine and Atmospheric Research Aspendale facility. Gravimetric mass on filters were determined using a Mettler UMT2 ultra-microbalance with a specialty filter pan. Electrostatic charging was reduced by the presence of radioactive static discharge sources within the balance chamber. The resolution of the balance was 0.1 μg, with a total uncertainty in aerosol mass at 95% confidence of 6.2 μg (U95%). When including the uncertainty in the MicroVol flow rate, the total uncertainty in mass concentration measurements U95% was 1.5 μg m^{-3} for 24-hour samples and 0.2 μg m^{-3} for 7-day samples.

5.2.5 Chemical analysis

The sampled filter samples were placed in heat sealed food grade plastic bags and stored at 4 ^{o}C until ready for analysis. Soluble ions and anhydrosugars (laevoglucosan) were extracted from the Teflo filters by adding 100 μl of methanol, to wet the hydrophobic filter, and 5 cm^3 of Milli-Q water to the plastic bags. Milli-Q water, having a resistivity of 18.2 mΩ.cm, was produced from a Millipore Milli-Q system, and was used in all sample and standard preparations. To prevent loss of soluble ions and anhydrosugars from bacterial action 50 μl of chloroform was added to the plastic bag.

[1] Teflo is a PTFE disc with a support ring

The bags were resealed and left to allow Soluble ions and anhydrosugars (laevoglucosan) to extract into the Milli-Q water.

NATA-accredited soluble cation measurements by ion chromatography were performed on the aerosol extracts in the Wet Chemistry Laboratory (Accreditation Number 245) at the CSIRO Marine and Atmospheric Research Aspendale facility. Aqueous potassium and magnesium concentrations were measured using a Dionex DX500 ion chromatograph. The chromatograph is equipped with a CS12A 250 mm × 4 mm analytical column and a CSRS Ultra II suppressor column. The ions are detected with a conductivity detector after they are separated using an isocratic eluent of 20 millimolar methane sulfonic acid. Potassium and magnesium standards were prepared by serial dilution of a NIST-traceable standard. Non seasalt potassium ($nssK^+$) in concentration units of $\mu mol.L^{-1}$ was determined using the ratio of potassium to magnesium found in seawater as shown in Equation 1

$$nssK^+ = K^+ - \left(\frac{0.01058 \times Mg^{2+}}{0.11038 \times 0.5} \right) \quad (1)$$

Extracts were analysed for laevoglucosan by one of two techniques. The $PM_{2.5}$ samples collected early in the study were analysed by a high performance liquid chromatography (HPLC) system consisting of a Dionex GP40 gradient pump, a Waters 717 autosampler, a Shimadzu System controller SCL-10A VP, a Shimadzu Evaporative Light Scattering Detector, (ELSD-LT), a Shimadzu Column Oven CTO-10AS VP and Shimadzu CLASS-VP chromatography software. The compound separation was performed with a 5 µm, 4.6 mm x 250 mm Alltech Prevail Carbohydrate ES column. The ELSD-LT operates by nebulising the mobile phase with nitrogen gas to produce a fine mist. Nebulised droplets then pass through a chamber where large droplets are siphoned off, leaving a fine mist which travels to a heated drift tube where the mobile phase evaporates. The analyte particles then pass through a detector chamber where light from a tungsten/halogen lamp is scattered and detected by an off -axis photomultiplier.

The extracts of samples collected later in the project were analysed with a Dionex ICS-3000 high performance anion exchange chromatograph with pulsed amperometric detection (HPAEC-PAD). The system consisted of an SP gradient pump, an AS auto-sampler, an EG eluent generator, a DC detector compartment with an electrochemical cell and Chromeleon chromatography software. Laevoglucosan was separated from other anhydride sugars such as galactosan and mannosan with a 4 mm x 250 mm CarboPac analytical column protected by a 4 mm x 50 mm CarboPac guard column.

The chromatographic conditions employed in the HPAEC-PAD technique included a flow rate of 0.5 ml min^{-1} and an injection volume of 25 µl. The peaks were separated isocratically in the first 10min with a mobile phase of 18 mM KOH, followed by a gradient from18mM KOH to 80mM KOH over 5min and then a 5min re equilibration at 18mM KOH. The eluent was generated by a Dionex EG and results in very low levels of carbonate contamination that enables a much shorter clean up and re equilibration of the column. The electrochemical detector was operated using waveform A in the integrated amperometric mode. The instrumental detection limit for laevoglucosan measured by the HPAEC-PAD technique is about 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 m^3. The development of the HPAEC-PAD technique is based on a previous study to measure laevoglucosan in $PM_{2.5}$ from biomass combustion (Engling et al., 2006).

Standard solutions of laevoglucosan and mannosan were produced from serial dilutions of stock standards, and ranged in concentration from ranged from 2.5 µg cm^{-3} to 50 µg cm^{-3}. Laevoglucosan (1,6 – anhydro-β-D-glucopyranose) was purchased from Sigma and mannosan (1,6 – anhydro-β-D-mannopyranose) was purchased from Carboller Inc. The aqueous concentrations of the extracts were calculated from a quadratic equation fitted to the standards in the case of HPAEC-PAD, and a point-to-point fit when the ELSD detector was employed.

Figure 5-1 is a chromatogram showing the separation of laevoglucosan and mannosan using the HPAEC-PAD system. The retention times for laevoglucosan and mannosan are 4.5 minutes and 6.2 minutes respectively.

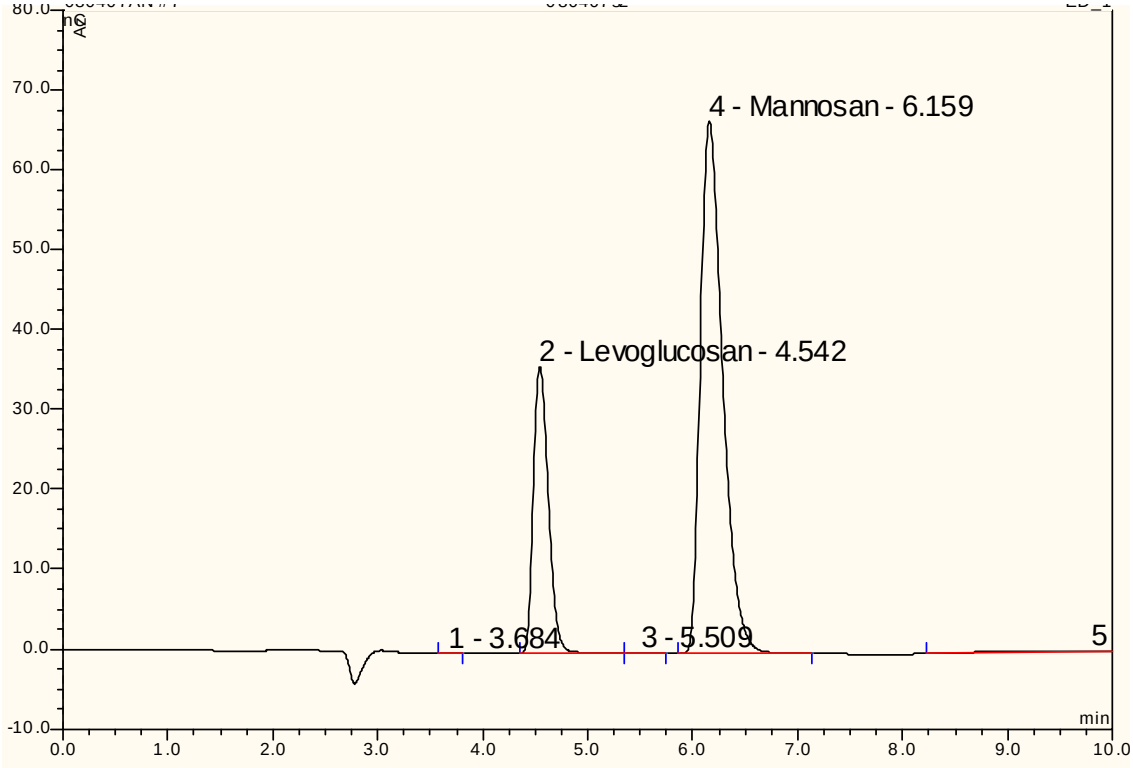


Figure 5-1 Chromatogram showing separation and retention times of laevoglucosan and mannosan using the HPAEC-PAD system

5.2.6 BTEX

BTEX samples were collected using a passive monitoring device (ISO 16017-2:2000) to provide an integrated measure of 7-day BTEX concentrations. This is considered to be an adequate time period to collect sufficient BTEX to quantify ambient outdoor concentrations. The BTEX passive sampler tubes were packed with the adsorbent Chromosorb^{TM} 106 (Supelco, USA). During sampling, one end of the tube remained sealed while the other end was exposed to the atmosphere using a diffusion cap. All measurements were made in duplicate and were accompanied by a field blank. The BTEX diffusive sampler tubes were cleaned before each sampling period, wrapped in aluminium foil and packed in clean metal tins, which were sealed with metal lids. Upon return, all sampler tubes were unpacked and the integrity of the seal of each cap was checked. The tubes were then analysed by using a PerkinElmer TurboMatrix™ 650 ATD (Automated Thermal Desorber) and a Hewlett Packard 6890A gas

chromatography (GC) equipped with a Flame Ionization Detector (FID) and a Mass Selective Detector (MSD). Compounds were identified by MS and quantified by FID.

5.3 Darwin

5.3.1 Description of site

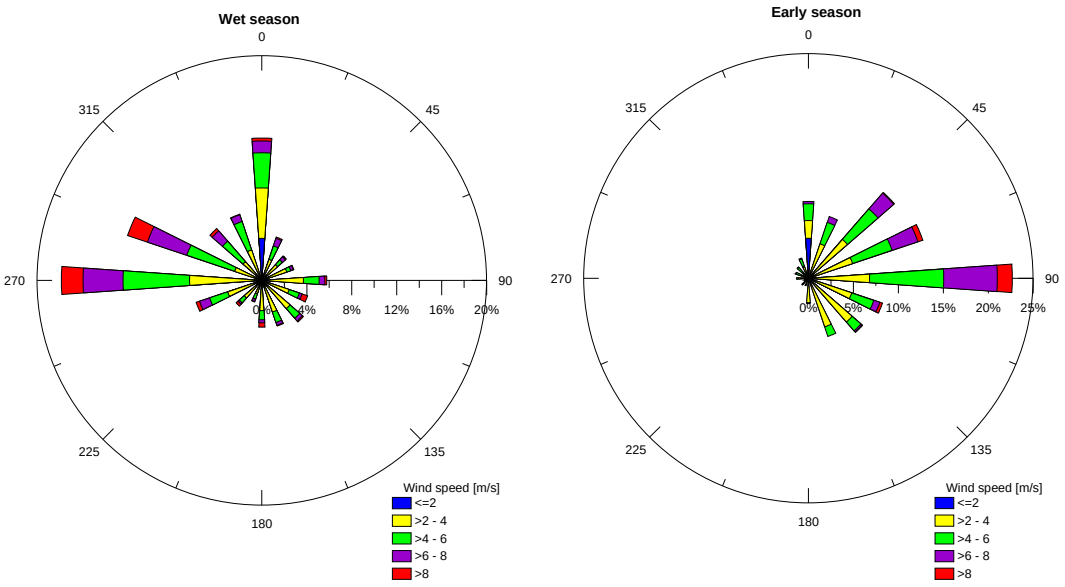
As mentioned in section 4, the monitoring site is located at the Charles Darwin University Campus in Casuarina within Darwin. The instrument set-up is shown in Figure 5-2.

Figure 5-2: Instrumentation set-up at Casuarina, Darwin NT: (a) inlet, (b) partisol sampler and (c) ozone analyser and TEOM

5.3.2 Meteorology

Monthly average temperatures at Darwin range from 23 to 29ºC, with October being the warmest and July the coldest month. Darwin has a wet and dry season with high rainfalls occurring between December and March. The burning season can be divided into early (April-May), mid (June-August) and late (September-November) season

Figure 5-3 shows the wind roses for Darwin-Airport by season using half-hourly wind speed and wind direction data. Lowest wind speeds are observed during early and late season with highest wind speeds during the wet and mid season. Wind patterns differ significantly between seasons. The wet season is dominated by westerly winds, whereas the early season is dominated by easterly winds. During mid season winds come from a south-easterly direction changing to a predominantly northern-north-westerly direction during the late season. The wind pattern will have a significant impact on smoke plume dispersion and the effects of prescribed burning activities on Darwin's air quality.



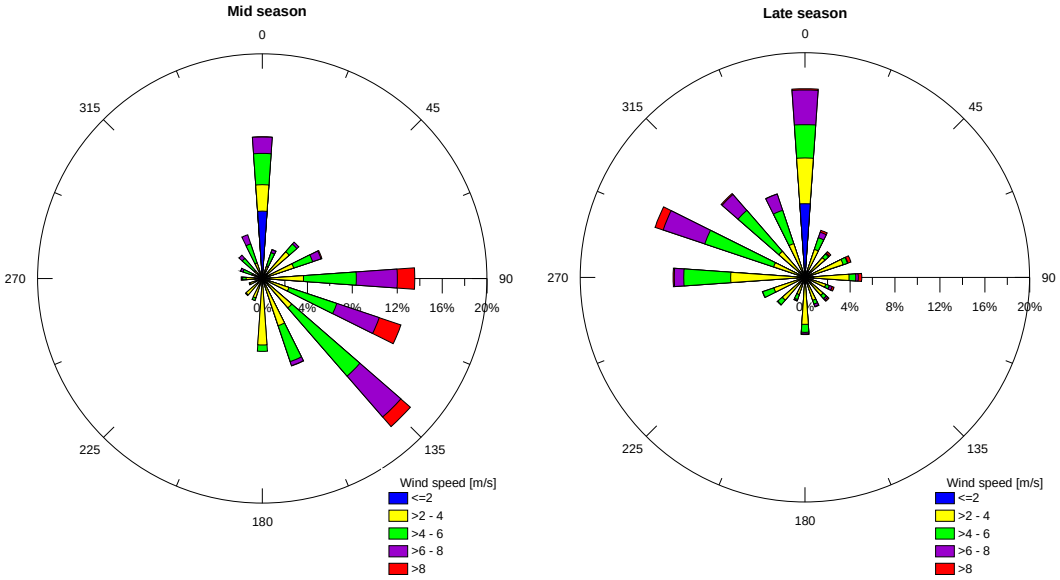


Figure 5-3: Half hourly wind direction and wind speed data by season measured at Darwin Airport

5.3.3 Observations

Measurements conducted at Darwin included

- continuous measurements of ozone between April 2007 and October 2008
- continuous daily gravimetric measurements of $PM_{2.5}$ and PM_{10} with a Partisol Dichotomous air sampler (January 2006 – September 2007) and continuous 10-minutely measurements of PM_{10} using a Tapered Element Oscillating Microbalance (TEOM) (January 2006 – December 2007), operated by Northern Territory University (NTU).
- weekly measurements of BTEX (April-December 2007).

5.3.3.1 Data quality

Ozone was measured continuously at 1-minute intervals, with zero checks conducted every 6 hours, and a span sample performed once per day. Because the signal was logged via a single-ended analogue-to-digital converter the monitor was configured with a positive zero offset. Zero values were stable throughout the period with little hourly or daily drift. Values averaged11.2 ppb with a standard error of 0.05 ppb.The ambient ozone readings were zero-corrected using the bracketing zeros. Unfortumately, the automatic zero-span was unintentionally deactivated between 17th December 2007

and 20th May 2008; for this period the average zero from the preceding 9 months was used to correct the data.

$PM_{2.5}$ and PM_{10} levels were measured daily using a Partisol sampler, which was decommissioned in September 2007. TEOM PM_{10} data were collected continuously between January 2006 and January 2008. There is a very good correspondence between the daily PM_{10} concentrations measured using the Partisol sampler and those measured using TEOM.; the data presented in this section is average PM_{10} from the two instruments.

5.3.3.2 Time series analysis

Figure 5-4 displays the time series of hourly concentrations (determined from averaging minutely data) of ozone measured at Casuarina, Darwin from April 2007 to October 2008. Hourly ozone concentrations ranged between 0 and 87 ppb, with an average concentration of 14 ppb. The 1-hour NEPM of 100 ppb was never exceeded during the sampling period.

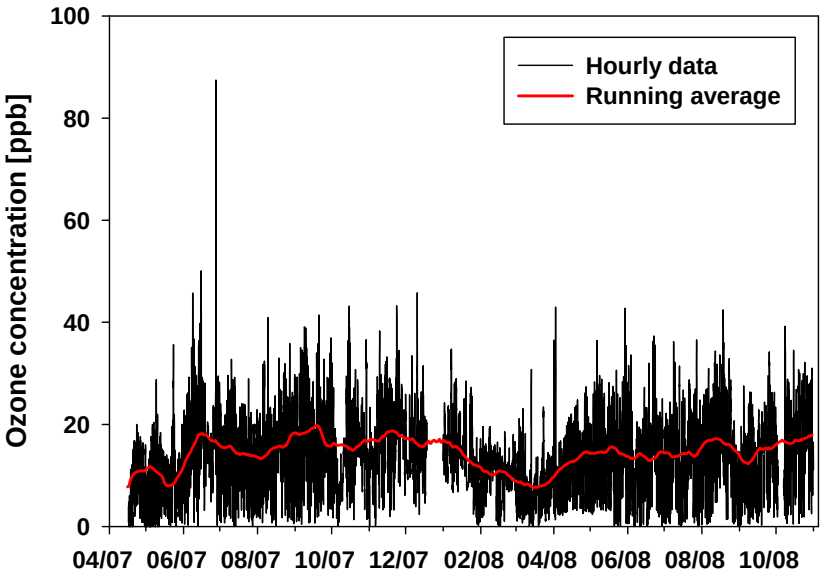


Figure 5-4 Time series of hourly ozone concentrations measured at Casuarina, Darwin between April 2007 and October 2008.

Figure 5-5 shows the daily $PM_{2.5}$ concentrations between January 2006 and September 2007 and daily combined PM_{10} concentrations between January 2006 and December 2007. For 2006 average $PM_{2.5}$ and PM_{10} concentrations were 7.6 and 15.6 $\mu g/m^3$, respectively. In 2007 the average concentrations of $PM_{2.5}$ and PM_{10} were 6.4 and 14.3 $\mu g/m^3$. Maximum daily $PM_{2.5}$ and PM_{10} concentrations were measured at 48 and 51.5 $\mu g/m^3$, respectively. The air quality NEPMs for $PM_{2.5}$ was exceeded on 5 days during 2006 and 3 days during 2007. There was one day in 2007 when the PM_{10} NEPM of 50 $\mu g/m^3$ was exceeded. All exceedences occurred between June and October.

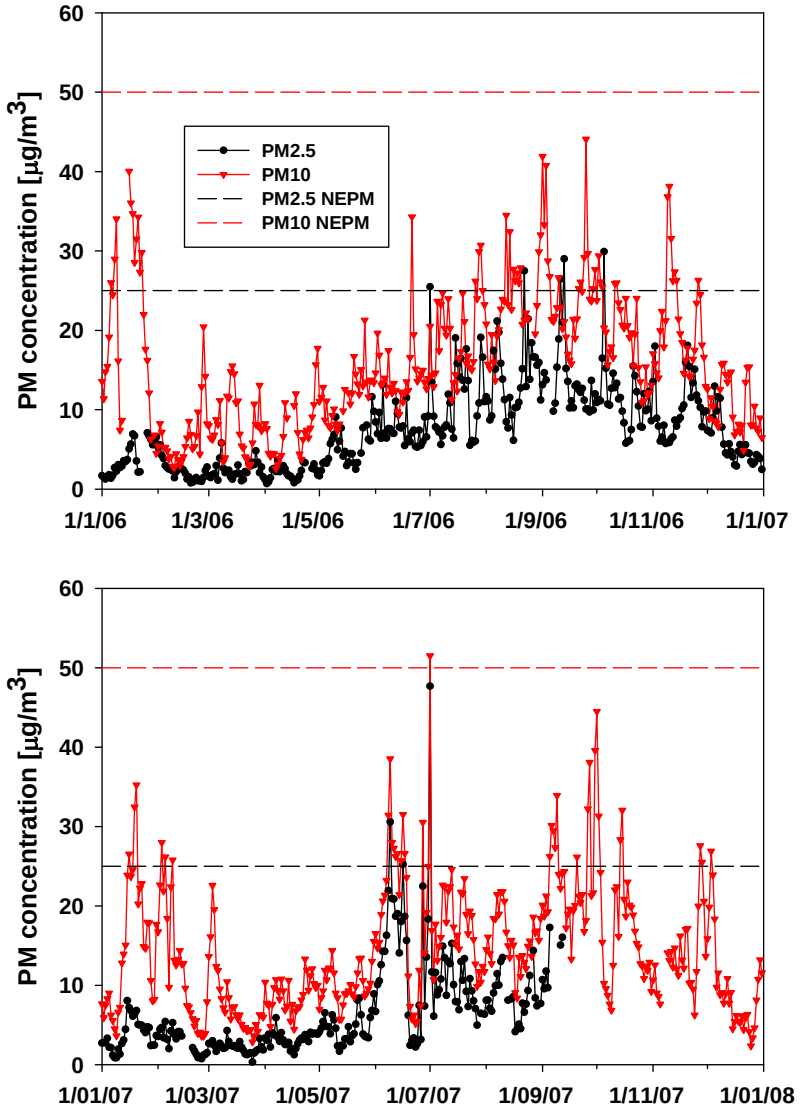


Figure 5-5 Time series of daily $PM_{2.5}$ and PM_{10} mass concentrations measured at Casuarina, Darwin (a) between January 2006 and December 2006 and (b) between January 2007 and December 2007.

5.3.3.3 Monthly distribution of Ozone and PM concentrations

As shown in Figure 5-6, during 2006, PM values were lowest in January-April increasing to maximum levels between August and November, and then again decreasing to lowest values in January-April. This is consistent with the fire season which begins early April at which stage PM levels started to increase and ends in November when PM levels started to decline.

During 2007, PM and ozone values followed the same trend until October, with maximum values measured in June and September and a slight decrease in August. Highest values of PM and ozone were observed during mid and late burning season (June-November). The minimum levels were recorded in February during the late wet season.

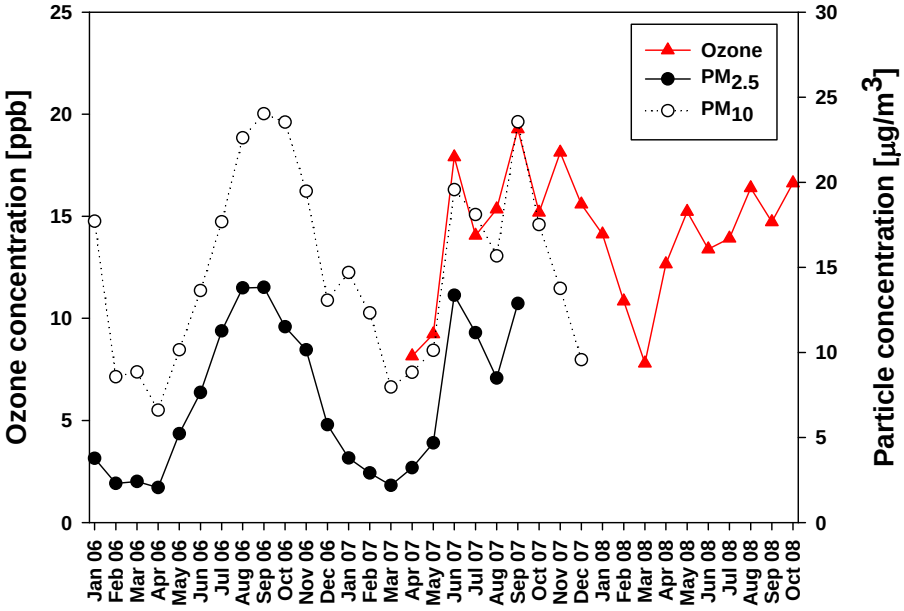


Figure 5-6 Monthly distribution pattern of surface ozone and PM mass concentrations measured at Casuarina, Darwin

5.3.3.4 Diurnal cycle of ozone and PM_{10}

Figure 5-7 shows the diurnal distribution of ozone and PM_{10} levels measured at Casuarina, Darwin between April 2007 and December 2007. There is a diurnal ozone cycle with maximum concentrations occurring during daylight hours and minimum

concentrations overnight, consistent with photochemical formation of ozone. Hourly PM_{10} concentrations display two distinctive peaks, one short peak in the morning at 8:00 and another broader peak in the late evening between 19:00 and 22:00. The morning peak is likely to be traffic-related and shows that vehicle emissions can be an important driver of Darwin's particle concentrations. The evening peak is less pronounced and may have a combination of sources, including vehicle emissions and bushfire smoke. Lowest PM_{10} levels are measured in the early hours of the morning and midday between 11:00-14:00, likely due to increased vertical mixing during the day. This is followed by a gradual increase in the afternoon to display maximum levels in the late evening.

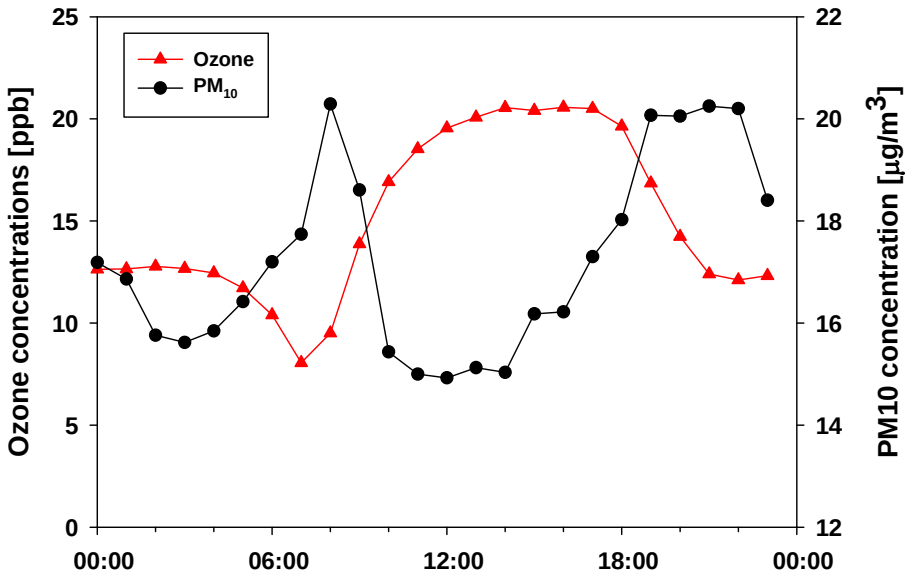


Figure 5-7 Diurnal cycle for ozone and PM_{10} measurements taken at Casuarina, Darwin

5.3.3.5 BTEX

Weekly measurements of BTEX were performed between July 2007 to September 2007, BTEX analysis was done by standard integration using retention time to identify compound peaks. Final quality control of the data has not been completed by the analytical laboratory at the time of the report. As a result, any peaks outside their retention times have been rejected for the analysis of this report and the BTEX data should therefore be considered preliminary. The average concentrations of benzene, toluene, ethylbenzene and xylenes measured between July and September 2007 were

0.16 ppb, 0.69 ppb, 0.095 ppb and 0.47 ppb, respectively. Concentrations were below the annual NEPM of 3 ppb for benzene and 100 ppb for toluene. Highest levels were observed for toluene, followed by xylenes, benzene and ethylbenzene. The average BTEX concentrations were lower than those measured in the cities of Melbourne (Torre et al., 2000) and Sydney (Linfoot and Freeman, 1998), slightly lower than those reported in Aspendale, VIC (Lawson et al., 2005), but similar to the BTEX levels reported for summer in Launceston (Galbally et al., 2004) (Table 5-3).

Table 5-3 Comparison of ambient BTEX concentrations in ppb measured at Darwin with concentrations measured in other locations

Location	Benzene	Toluene	Ethylbenzene	Xylenes
Darwin	0.16	0.69	0.095	0.47
Aspendale	0.3	0.8	0.1	0.6
Melbourne	1.1	1.8	0.2	0.8
Sydney urban	1.0	2.3	0.3	1.5
Sydney urban fringe	0.4	0.8	0.1	0.6
Launceston summer	0.23	0.59	0.08	0.45
Launceston winter	1.56	3.11	0.29	1.51

5.3.4 Smoke events

Emission estimates for the Top End for 2004 suggest that maximum emissions of $PM_{2.5}$ are associated with late dry season fires which occur between September and November. Early and mid dry season fires substantially small fractions of the annual total (Meyer et al., 2008a).

Figure 5-8 shows the daily $PM_{2.5}$ and PM_{10} distribution by season. The PM measurements for 2006 and 2007 shows that the higher daily concentrations are measured during the mid and late season, which is consistent for both years. Late season $PM_{2.5}$ data for 2007 is unfortunately not available.

During the wet and early season, about 80% of the $PM_{2.5}$ concentrations are below 5 $\mu g/m^3$ with another 15-20% between 5 $\mu g/m^3$ and 10 $\mu g/m^3$ and only a maximum of 3% of the daily $PM_{2.5}$ concentrations between 10 $\mu g/m^3$ and 15 $\mu g/m^3$. During the mid and late burning season, only about 34-50% of the daily $PM_{2.5}$ concentrations are below 10 $\mu g/m^3$. The mid season displays highest frequency in the range of 5-10 $\mu g/m^3$

whereas in the late season this shifts to concentrations ranging between 10 and 15 μg/m^3.

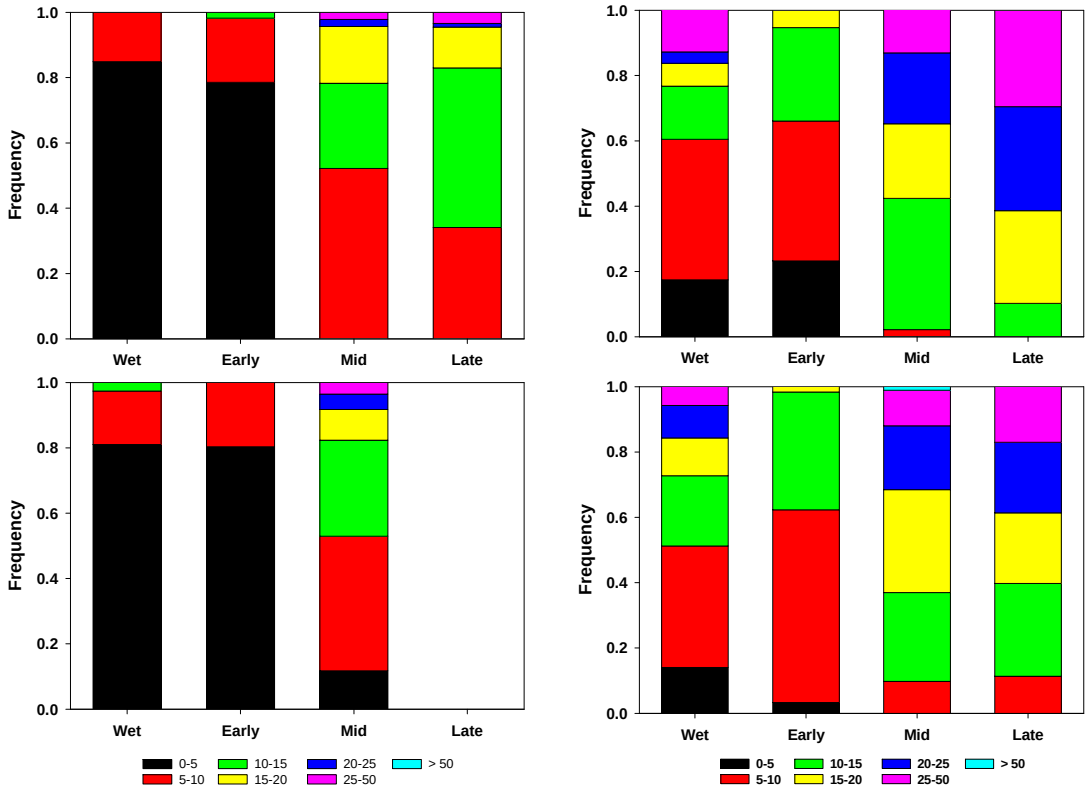


Figure 5-8 Daily $PM_{2.5}$ (left) and PM_{10} (right) concentration distribution by season for 2006 (top) and 2007 (bottom).

5.3.4.1 Chemical analysis

Filters collected for gravimetric mass concentrations were also used for the analysis of water-soluble ions and laevoglucosan, a wood smoke tracer. Although much is known for laevoglucosan content in smoke from forest fuels, better information is required for grassy fuels to enable assessment of source signature. The chemical species measured in the $PM_{2.5}$ aerosol can also be used to identify pollutant source contributions. Non sea-salt potassium ($nssK^+$) has two major particle sources, soil dust and smoke from biomass burning, known to be high in potassium and can therefore be used as a chemical tracer for wood smoke. $nssK^+$ was calculated using the known ratio of Mg^+ to K^+ in seawater.

Figure 5-9 shows the weekly time series plot of $PM_{2.5}$ concentrations and concentrations of laevoglucosan and $nssK^+$. Elevated laevoglucosan levels were measured between July and September 2004 and in July-August 2005, corresponding to the mid and late burning season. Lowest levels of laevoglucosan were observed between December and March which is the wet season. Laevoglucosan levels ranged from non-detectable to 1.4 $\mu g/m^3$. $nssK^+$ levels remained low (0 to 1.2 $\mu g/m^3$) with a major peak of 11.6 $\mu g/m^3$ observed in July.

The ratio of laevoglucosan to particle mass was lowest during the wet season increasing significantly during the mid and late burning season (Figure 5-10). $nssK^+$ displays a similar pattern.

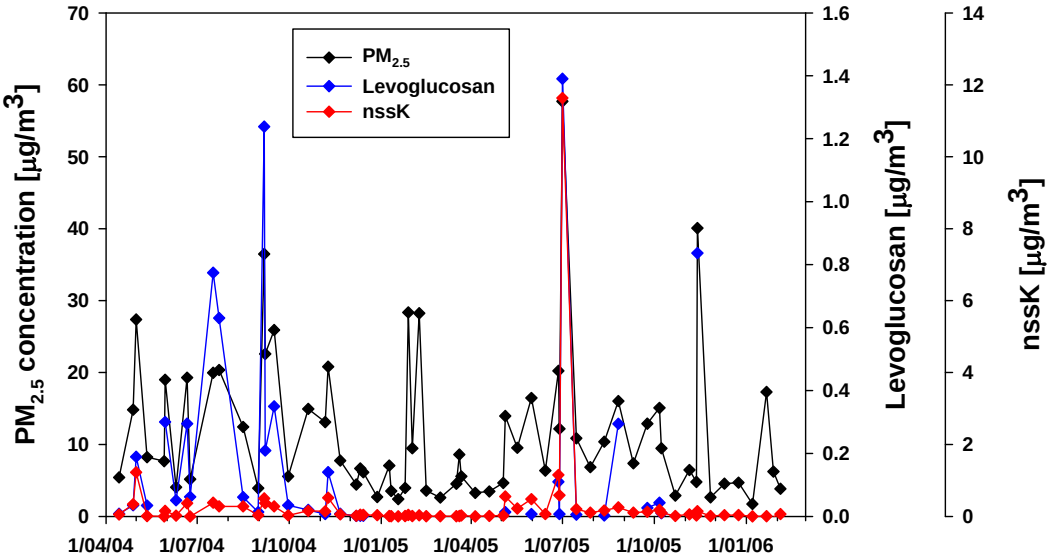


Figure 5-9 Weekly concentrations of $PM_{2.5}$, laevoglucosan and $nssK^+$ measured at Casuarina, Darwin.

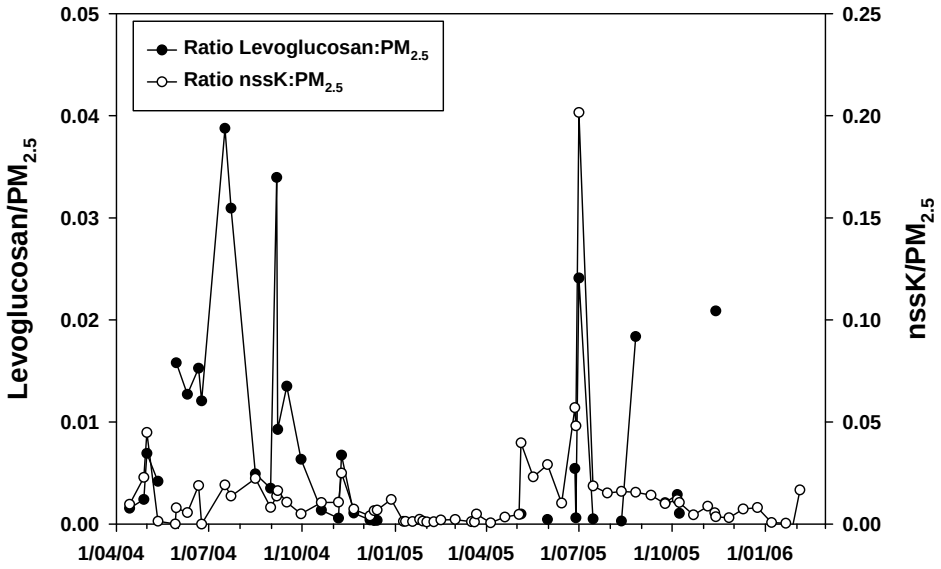


Figure 5-10 Weekly measurements of the ratios of laevoglucosan and $nssK^+$ levels to $PM_{2.5}$ mass concentrations measured at Casuarina, Darwin.

5.3.5 Summary

During the sampling period in Darwin, surface ozone concentrations never exceeded the 1-hour NEPM of 100 ppb. The $PM_{2.5}$ NEPM was exceeded on 5 days during 2006 and 3 days during 2007, whereas the PM_{10} NEPM was exceeded on one occasion in 2007. The diurnal pattern has shown that urban emissions are a major driver of particle pollution, to which biomass burning adds an additional load of particulate matter. There was a significant increase of particle levels during the mid and late burning season compared to the early burning season.

5.4 Manjimup

5.4.1 Description of site

The sampling site is located at the site of the regional office of the Department of Environment and Conservation in Manjimup, WA. Figure 5-11 shows the instrument set-up.

Figure 5-11 Instrument set-up at Manjimup, WA: (a) location of sampling station, (b) sampling box, (c) ozone analyser and (d) BTEX sampling set-up

5.4.2 Meteorology

Manjimup has a mild climate with cool wet winters and warm dry summers. Average monthly temperatures range between 11 and 19 ºC. Coldest months are June to September and warmest monthly averages were measured for December to February. Highest rainfall in 2007 occurred during winter (July-September).

Figure 5-12 shows the wind roses for Manjimup by season using hourly wind speed and wind direction data. Lowest wind speeds are observed during summer with highest wind speeds for winter. The summer is dominated by southerly winds, whereas winter is dominated by north-westerly winds. Autumn also displays southerly winds as well as

quite strong north-westerly winds. In spring winds are westerly. The seasonal wind patterns will have an impact on smoke plume dispersion and the effect of prescribed burning activities on Manjimup's air quality.

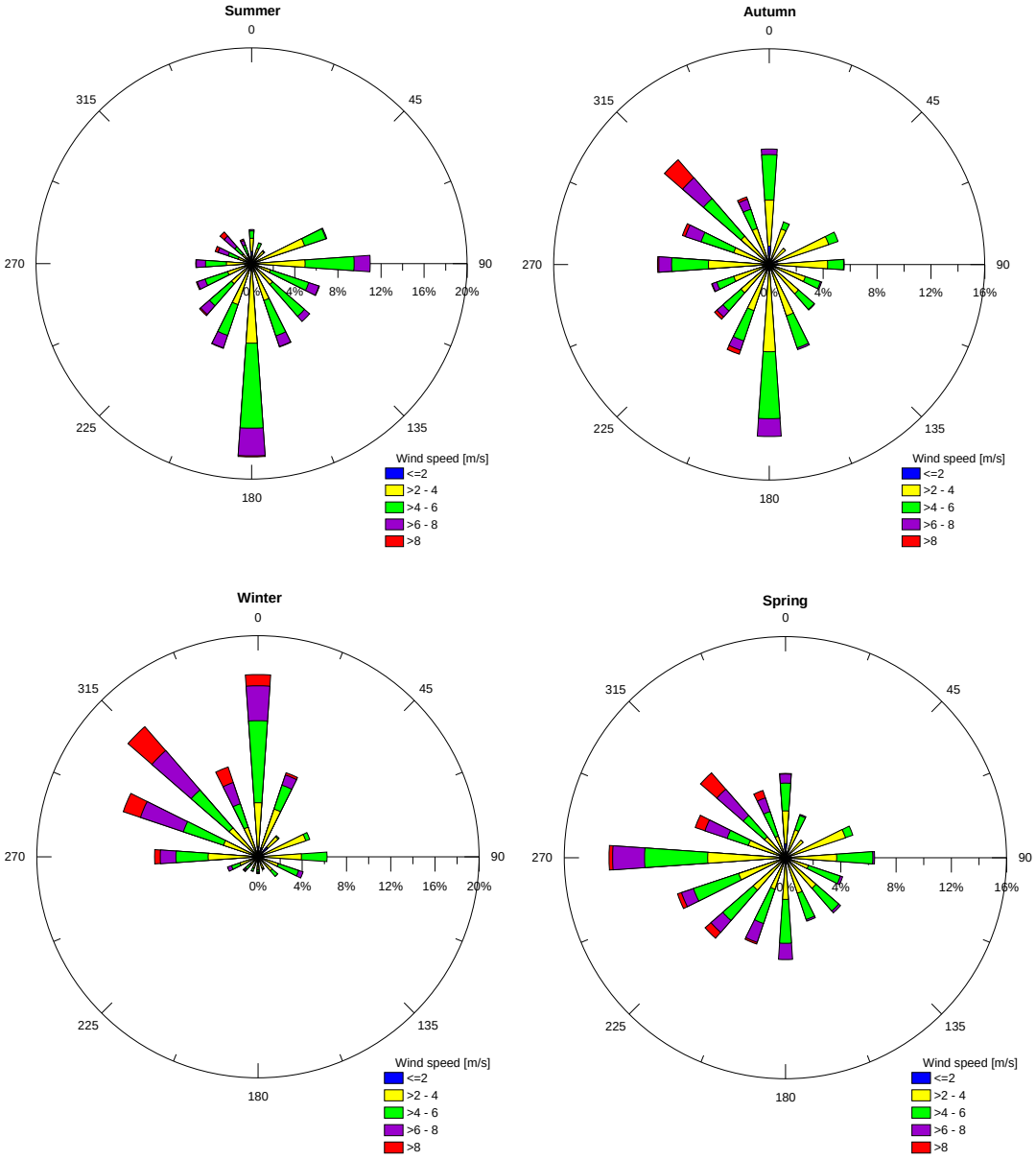


Figure 5-12: Hourly wind direction and wind speed data by season measured at Manjimup

5.4.3 Observations

Air quality measurements conducted at Manjimup, WA, included continuous measurements of ozone and $PM_{2.5}$, weekly mass concentrations of $PM_{2.5}$ and weekly measurements of BTEX.

5.4.3.1 Data quality

DustTrak issues

The DustTrak was configured to automatically transmit aerosol mass concentration values via an analog output voltage signal that is proportional to the displayed concentration. The scaling range was set-up at 0-1.0 mg/m^3, which was exceeded on May 19 for 2 minutes and on Nov 15 for approximately 1.5 hours.

The DustTrak was zeroed every 6 hours. However, zeroing was stopped on November 9, 2007.

There were several power outages, after which the DustTrak did not restart. As a result of this, there are missing data for several days.
Over the duration of the sampling period, the pump of the DustTrak also began to fail, resulting in low flows. The flow was set at 1.7 L/min to ensure particle collection of 2.5 μm diameter in size. The flow had dropped to 0.2 L/min measured on November 7, 2007. This drop in flow is likely to affect the size collection of particles, and could also have potentially affected the intake of particles. Unfortunately, it is not clear at which stage the pump failure began.

The DustTrak is factory-calibrated using standard ISO 12103-1 A1 dust (Arizona Road Dust), however, as discussed in section 5.2.2, differences in size distribution and particle characteristics between the Standard and bushfire smoke leads to differences between DustTrak readings and gravimetric measurements. Previous research has shown that for PM concentrations above 10 μg/m^3, the DustTrak over-recorded by a factor of 1.94 – 2.6 for indoor air (Heal et al., 2000; Ramachandran et al., 2000; Yanosky et al., 2002), by a factor of 2.7 (Kingham et al., 2006) and 3.0 (Chung et al.,

2001) for outdoor air, and by a factor of 3.17 (Trent, 2006) for smoke samples. At low particle concentrations (less than 10 μg/m^3), it is not clear whether a linear relationship is observed between DustTrak and gravimetric measurements.

In this study, weekly average gravimetric mass measurements of $PM_{2.5}$ from the MicroVol provided a continuous calibration of the DustTrak $PM_{2.5}$ readings. The calibration required an integration of the real-time DustTrak $PM_{2.5}$ concentration profile and calculation of average DustTrak $PM_{2.5}$ concentrations. These average $PM_{2.5}$ concentrations were then compared to the average gravimetric mass measurements of $PM_{2.5}$ collected on the filter of the MicroVol. The ratio of particle concentrations measured by the DustTrak to gravimetric mass concentrations ranged from 0.1 up to 2.7, the latter being measured during a smoke event, which is in agreement to previous data. In clean air the DustTrak underestimated PM concentrations. The DustTrak data was plotted against the gravimetric mass concentrations and a non-linear trend was observed (Figure 5-13). The data were fitted empirically with a power function

$y = 6.07\ x^{0.32}$ for $PM_{2.5} < 60\ \mu g/m^3$ and a ratio of 2.7 for $PM_{2.5} > 60\ \mu g/m^3$

All DustTrak data reported in this section are presented as corrected data.

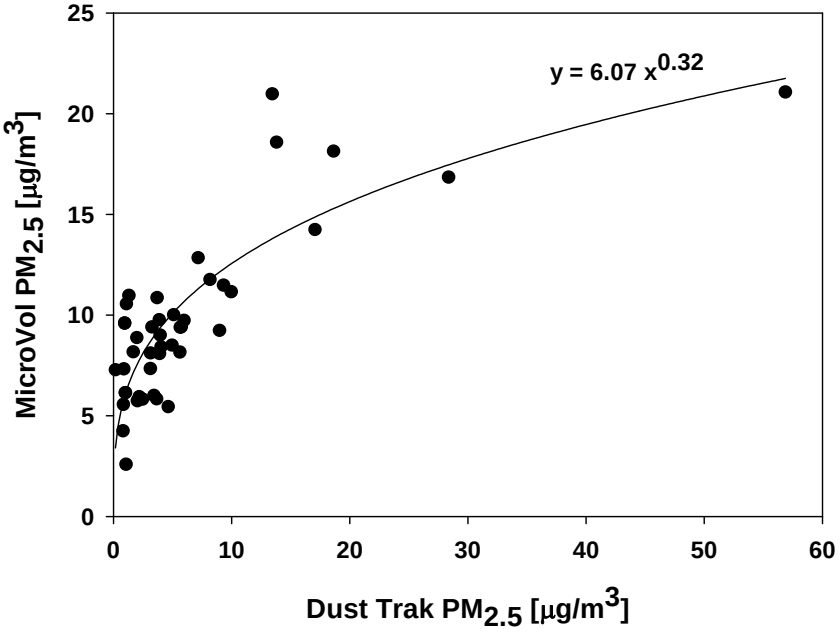


Figure 5-13 Calibration curve for DustTrak $PM_{2.5}$ measurements versus gravimetric $PM_{2.5}$ mass measurements made using a MicroVol

A similar pattern was also observed for long-term data collected using simultaneously a DustTrak and a TEOM at Ti Tree Bend air quality monitoring station in Launceston. as part of the Woodheater CARP project (CARP #16) between May and October 2007 (Mike Groth, pers comm.). This station is one of a network maintained and operated by the Department of Tourism, Arts and Environment, Tasmania. The Manjimup, weekly mean data together with data from Ti Tree Bend are plotted in Figure 5-14. At low concentrations the DustTrak underestimated PM_{10} concentrations at Ti Tree Bend, whereas at high concentrations, the DustTrak overestimated PM_{10} concentrations, This pattern is indicative of a combination of several sources with different light scattering characteristics. Sources likely to impact Ti Tree Bend include smoke from domestic woodheaters, transport sources, wind-blown dust and marine aerosol. The Manjimup data correspond closely to the Ti Tree Bend data: at high concentrations $PM_{2.5}$ composition is primarily smoke from fire events while at low concentrations other local and regional sources of PM predominate.

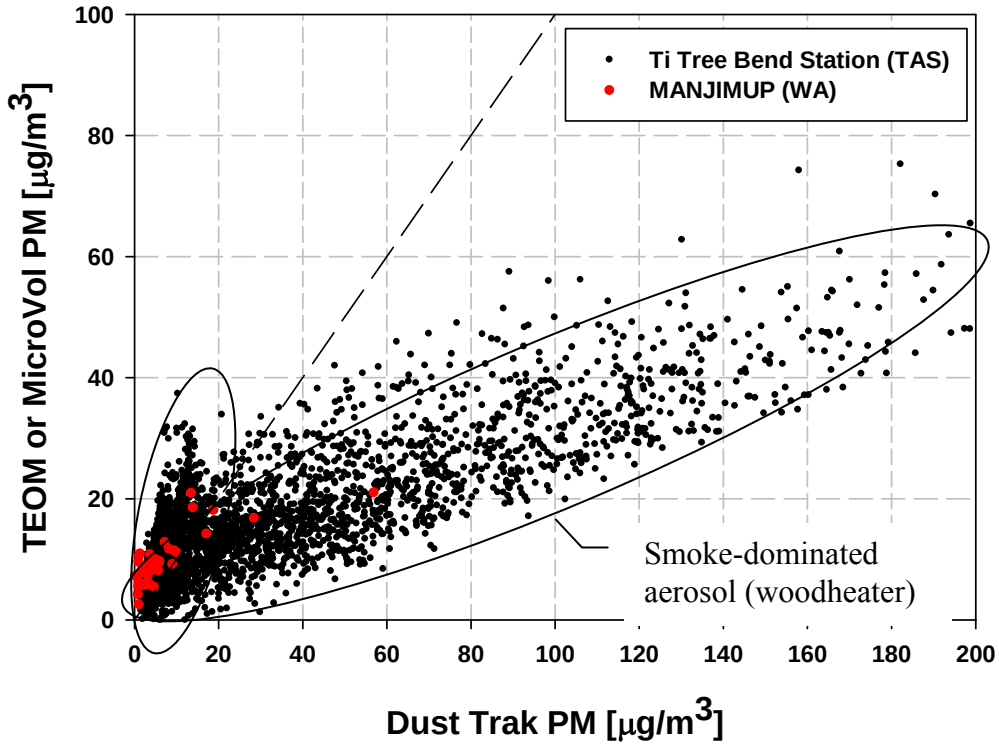


Figure 5-14 Relationship between TEOM and DustTrak PM_{10} concentrations at Ti Tree Bend monitoring station (TAS) and between MicroVol and DustTrak $PM_{2.5}$ concentrations at Manjimup, WA. Dotted line represents 1:1 ratio.

Gravimetric mass measurements

Gravimetric measurements of $PM_{2.5}$ were made using a low-volume MicroVol-1100 (Ecotech Pty Ltd, Australia). The flow rate of the MicroVol was calibrated at the beginning of the sampling period against a voltage output. The voltage output of the MicroVol was logged at 1-minute intervals, and used to determine the flow rate of the MicroVol during the sampling period. The voltage output remained stable throughout the sampling period.

Very low mass concentrations were measured in June/July, which could not be explained. DustTrak data was used to correct for those measurements.

Ozone monitor

Ozone was measured continuously at 1-minute intervals, with zero checks conducted every 6 hours and span measurements performed once a day. Span samples were consistent during the monitoring period. As for the Dusttrak power outages have resulted in missing ozone data for several days. Additionally, the filter removing any potential particles entering the ozone analyser was blocking the flow in November, and as a result ozone data are missing for this period

5.4.3.2 Time series analysis

Figure 5-15 displays the time series of hourly concentrations (averaged from minute data) of ozone and $PM_{2.5}$ measured at Manjimup, WA from December 2006 until December 2007.

Hourly ozone levels ranged from 0 ppb to 63 ppb, with an average of 20 ppb. The maximum hourly ozone concentration of 63 ppb was measured in March 2007. The 1-hour NEPM of 100 ppb was not exceeded during the sampling period.

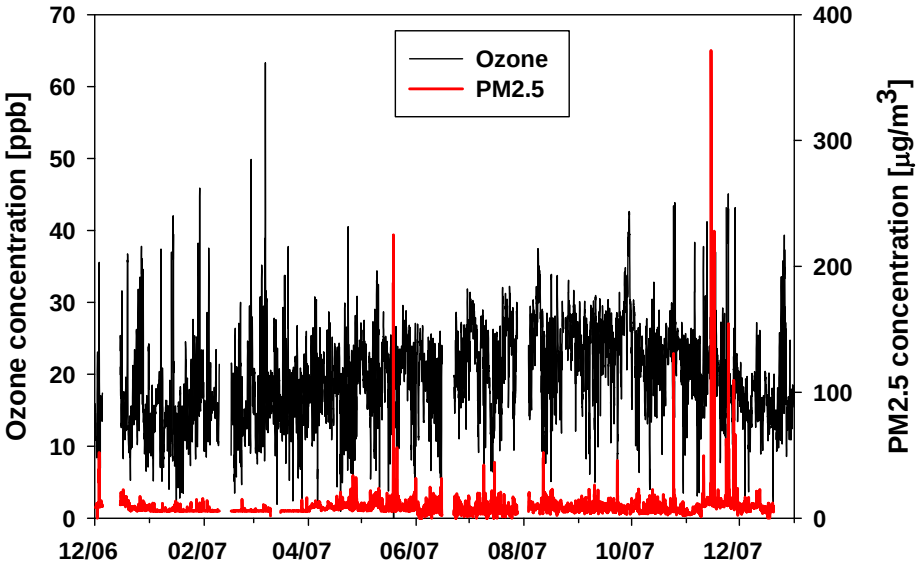


Figure 5-15 Time series plot of hourly surface ozone and $PM_{2.5}$ concentrations (averaged from minute data) measured at Manjimup, WA between December 2006 and December 2007.

Hourly $PM_{2.5}$ levels ranged from 0 to 372 µg/m^3 (calibrated against gravimetric measurements), with an average of 8.7 µg/m^3. Daily average concentrations ranged from 0 to 87 µg/m^3. The NEPM $PM_{2.5}$ Advisory is set at 25 µg/m^3 which should not be exceeded more than 5 days per year. This accounts for unusual high emissions such as emissions during bushfires or dust storms. The standard (using the corrected DustTrak data) was exceeded for 1 day in May and for 3 days in November. Daily $PM_{2.5}$ levels were elevated in November (Figure 5-16), which was due to prescribed burning in the area. A few high concentrations were also measured in August and October. Days with exceedences will be analysed in section 5.4.4 to assess the duration and intensity of the smoke events.

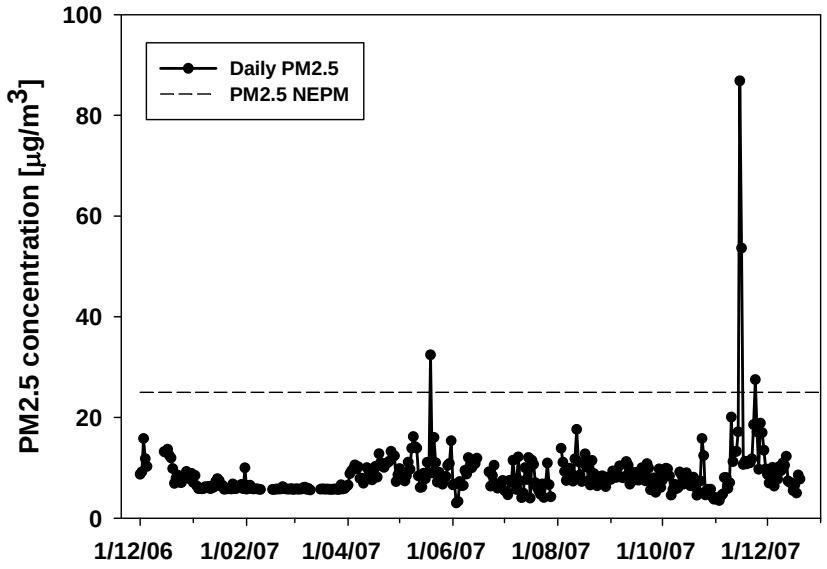


Figure 5-16 Time series plot of averaged daily $PM_{2.5}$ concentrations measured at Manjiump between December 2006 and December 2007.

5.4.3.3 Monthly distribution of Ozone and $PM_{2.5}$ concentrations

The ozone concentrations show a monthly cycle with minimum ozone levels during January increasing to display maximum levels in August/September (Figure 5-17). This is consistent with surface ozone measurements conducted at Cape Grim which showed seasonal ozone maximum in winter and minimum in summer and is probably due to the short transport distance of 50 km from the coast during which ozone destruction can occur at the soil surface. The seasonal pattern in marine surface air is attributed to enhanced photochemical destruction in the moist marine boundary layer in summer (Ayers et al., 1992).

$PM_{2.5}$ concentrations showed distinctive peaks in May 2007 and November 2007, with other smaller peaks in December 2006 and August 2007. The elevated concentrations in May and November coincide with the autumn and spring prescribed burning season. Minimum $PM_{2.5}$ concentrations were measured from January to March, as a result of low fire activities in the area, but also consistent with a greater mixing depth in summer compared to winter. High levels of $PM_{2.5}$ during winter suggest domestic wood burning activities but may also be due to a more stable atmosphere suppressing vertical mixing.

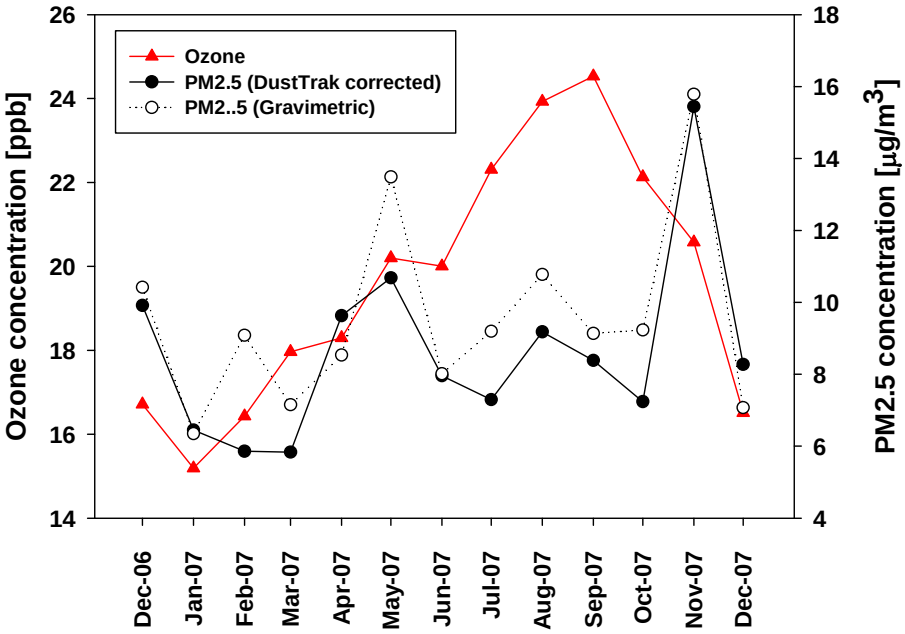


Figure 5-17 Monthly distribution pattern for surface ozone and $PM_{2.5}$ concentrations measured at Manjimup, WA.

5.4.3.4 Diurnal cycle of ozone and $PM_{2.5}$

Figure 5-18 shows the average diurnal pattern of hourly concentrations of ozone and $PM_{2.5}$. As expected, the diurnal cycle of ozone concentrations displays maximum concentrations during midday and minimum concentrations at night, which is due to increased photochemistry during the day.

Excluding any smoke events, particle levels display maximum values overnight, decreasing through the early hours of the morning and increasing between 6:00 and 8:00. Particle concentrations then drop to reach lowest values during midday. This is followed by a gradual increase until midnight.

Diurnal changes can be attributed to the atmospheric mixing, but also to pollutant sources. In general morning and afternoon peaks are due to traffic. In the case of Manjimup, the sharp increase after about 16:00 with maximum levels at 18:00 can be attributed to smoke events in the area, as the peak was less pronounced when PM levels of smoke events were excluded from the calculation of the diurnal distribution. This will be discussed further in section 5.4.4.

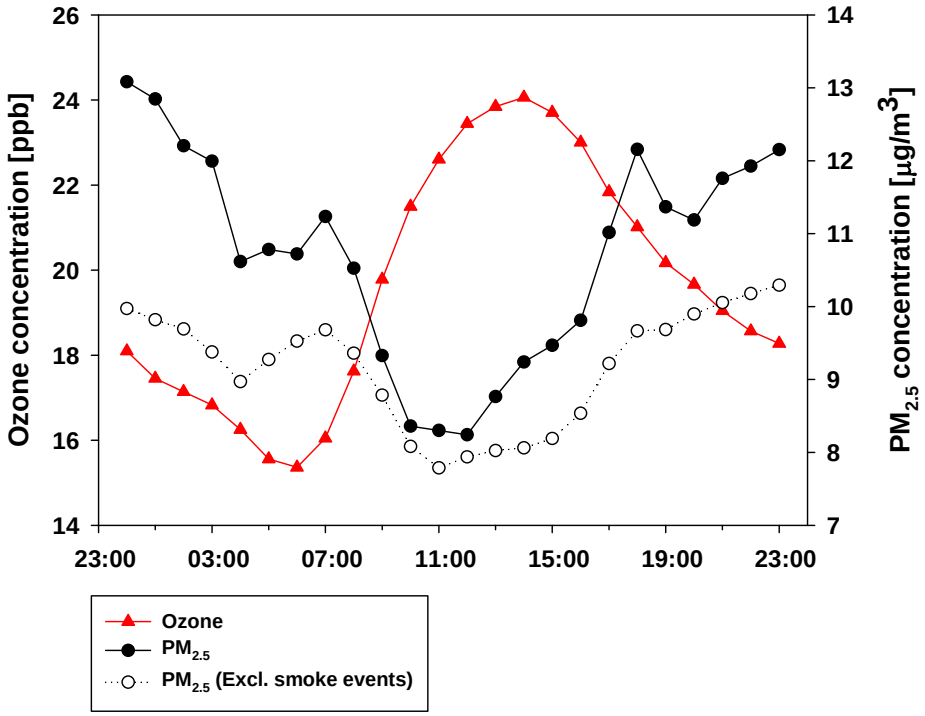


Figure 5-18 Diurnal distribution cycle of surface ozone and $PM_{2.5}$ concentrations measured at Manjimup, WA.

5.4.3.5 BTEX

Weekly measurements of BTEX were performed between December 2006 to December 2007, however total analysis has only been completed for data collected between December 2006 and May 2007. Data for May 2007 until December 2007 should be considered preliminary. Highest average concentrations over the 1-year period were observed for toluene, closely followed by benzene, xylenes and ethylbenzene (Table 5-4). Maximum levels for BTEX were measured at 0.354 ppb, 0.487 ppb, 0.092 ppb and 0.190 ppb respectively. The levels were well below the NEPM of 3 ppb for benzene and 100 ppb for toluene. The annual BTEX averages were lower than the values of BTEX reported in Aspendale (Lawson et al., 2005), Melbourne (Torre et al., 2000), Sydney (Linfoot and Freeman, 1998) and Launceston (Galbally et al., 2004) (Table 5-4).

Table 5-4 Comparison of ambient BTEX concentrations in ppb measured at Manjimup with concentrations measured in other locations

Location	Benzene	Toluene	Ethylbenzene	Xylenes
Manjimup	0.1	0.12	0.02	0.04
Aspendale	0.3	0.8	0.1	0.6
Melbourne	1.1	1.8	0.2	0.8
Sydney urban	1.0	2.3	0.3	1.5
Sydney urban fringe	0.4	0.8	0.1	0.6
Launceston summer	0.23	0.59	0.08	0.45
Launceston winter	1.56	3.11	0.29	1.51

The weekly concentrations (average of two samples) of BTEX observed throughout the 1-year period are presented in Figures 5-19 and 5-20.

The BTEX concentrations are highest in winter and lowest in summer, which is consistent with reduced vertical mixing during winter. The increase is most pronounced for benzene and toluene, similar to that reported in a study conducted in Launceston (Galbally et al., 2004). Overall benzene levels are high compared to toluene levels. Previous studies have shown that in urban areas benzene to toluene ratio is about 0.4-0.6 (Lawson et al., 2005; Galbally et al., 1999). At Manjimup, the annual average ratio was 0.9. The elevated benzene to toluene ratio can be explained by wood smoke emissions which are relatively rich in benzene. In fact wood smoke displays about 3 times higher levels of benzene than toluene (Environment Australia, 2002). The highest ratio was measured during summer when toluene levels seem to be exceptionally low. High levels of benzene were measured during autumn and winter which may be due to autumn prescribed burning and domestic wood smoke use during winter. Even though extensive prescribed burning was conducted around Manjimup during spring, benzene levels remained mainly below toluene levels.

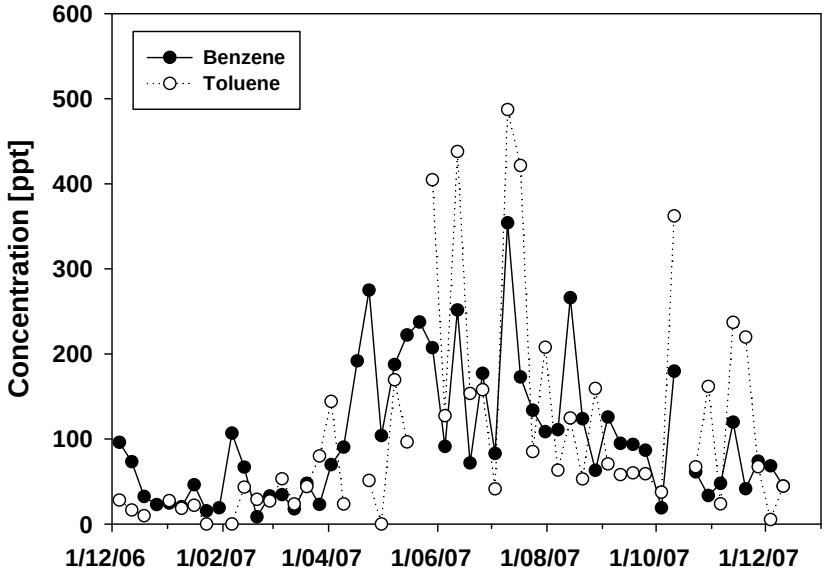


Figure 5-19 Weekly concentrations of benzene and toluene measured at Manjimup, WA between December 2006 and December 2007.

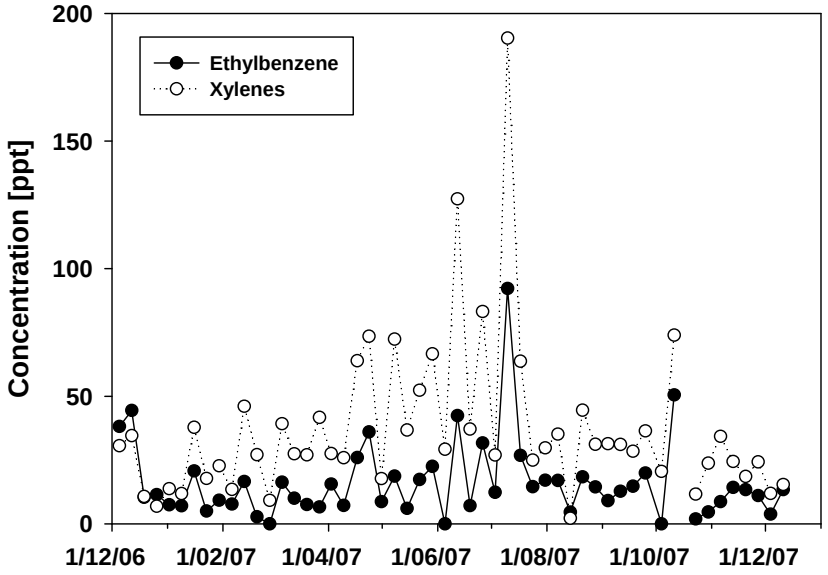


Figure 5-20 Weekly concentrations of ethylbenzene and xylenes measured at Manjimup, WA between December 2006 and December 2007.

5.4.4 Smoke events

In southern Western Australia prescribed burns are carried out in spring and autumn when weather conditions are mild and fire behaviour is moderate which makes it easier

to manage the fire. Outcomes from both prescribed burning season can be quite different due to differences in fuel moisture (Ward, 1999). Spring burning is undertaken when fuels are still reasonably moist from winter rains enabling low fire intensity burns. Autumn burning is undertaken when ground fuels are beginning to get wet as a result of rains. The fuels are still relatively dry compared to fuels in spring resulting in moderate intensity fires.

Four major smoke events were identified in this study. Three were in spring: October 24th (event 1), Nov 14th – Nov 16th (event 2) and Nov 23rd-Nov 29th (event 3). These events were most probably from prescribed burning activities undertaken by DEC, WA. The locations of prescribed burns that may have been the smoke source are shown in Figure 5-21. Ignition times for these burns mostly occurred between 11 AM and 2 PM although one burn commenced at 4 PM. Event 4 occurred in autumn on May 19th and was most probably due to forest management activities in a private plantation to the north of Manjimup.

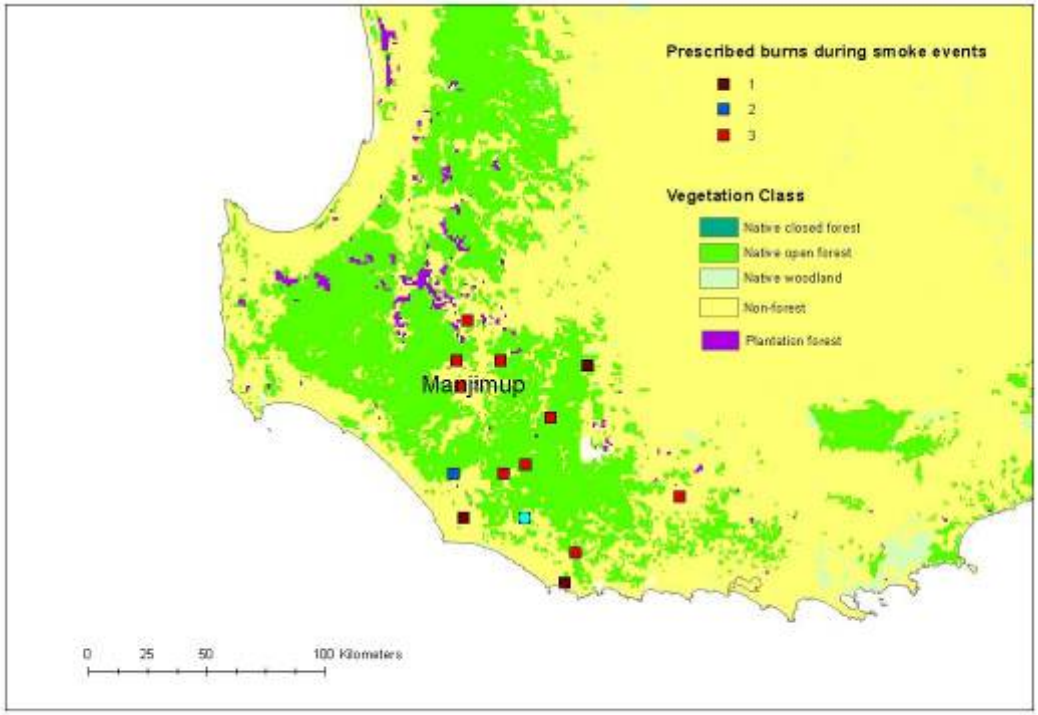


Figure 5-21 Location of Prescribed burns conducted by DEC, WA during the time of recorded smoke events at Manjimup

5.4.4.1 Chemical analysis

Filters collected for gravimetric mass concentrations were also used for the analysis of water-soluble ions and laevoglucosan, a wood smoke tracer. The chemical species measured in the $PM_{2.5}$ aerosol can be used to identify pollutant source contributions. Non sea-salt potassium ($nssK^+$) has two major particle sources, soil dust and smoke from biomass burning, known to be high in potassium and can therefore be used as a chemical tracer for wood smoke. $nssK^+$ was calculated using the known ratio of Mg^+ to K^+ in seawater.

Figure 5-22 shows the weekly time series plot of $PM_{2.5}$ concentrations and the two wood smoke markers, laevoglucosan and $nssK^+$. Weekly gravimetric $PM_{2.5}$ mass concentrations displayed elevated levels at three occasions, notably in May, August and November. At all three occasions, laevoglucosan and $nssK^+$ levels were elevated, suggesting the presence of smoke. The elevated levels in May and November coincide with autumn and spring prescribed burning. The high particle levels and presence of elevated concentrations of biomass burning tracers observed in August suggest domestic wood burning activities.

Laevoglucosan levels ranged from non-detectable to 4.6 $\mu g/m^3$. The highest concentration was measured in August coinciding with the highest particle levels measured during the winter. Levels of $nssK^+$ ranged from 0.001 to 0.27 $\mu g/m^3$. The concentrations were particularly elevated in the weeks of October 19-26 and of November 9-16. During both weeks, smoke from prescribed burns had travelled to Manjimup resulting in increased concentrations of $PM_{2.5}$ and ozone.

The ratio of laevoglucosan to particle mass was lowest during summer increasing four-fold during autumn and five-fold during winter and spring (Figure 5-23). This is consistent with biomass burning activities during autumn and spring (prescribed burning) and domestic wood burning during winter. Unlike laevoglucosan, $nssK^+$ displays low ratio to PM mass concentrations during winter time and very high ratios during spring.

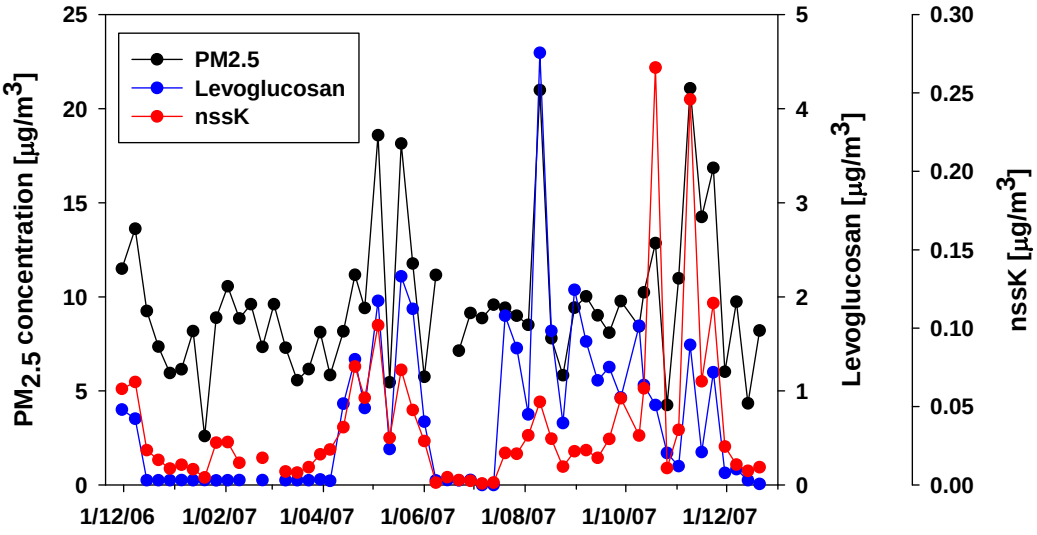


Figure 5-22 Weekly concentrations of $PM_{2.5}$, laevoglucosan and $nssK^+$ measured at Manjimup, WA.

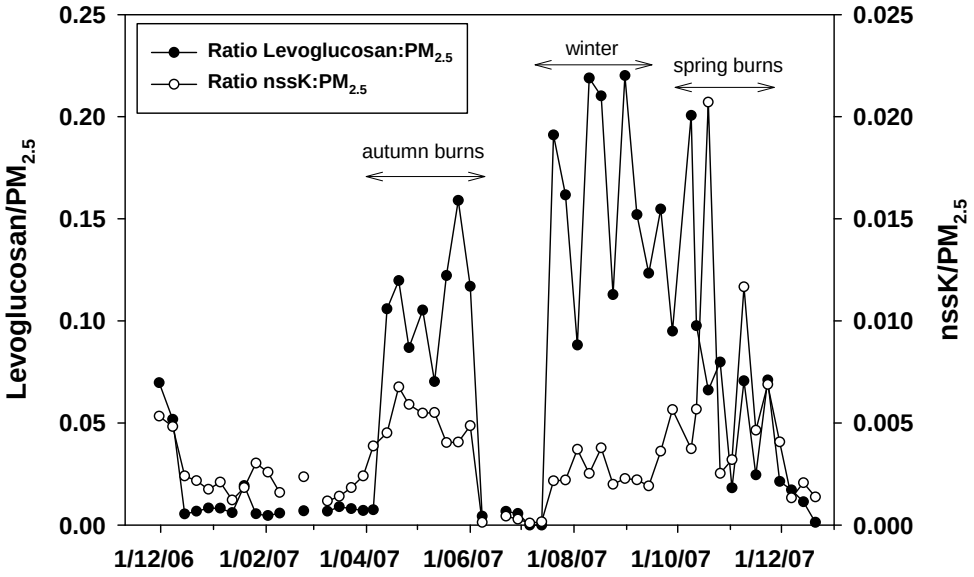


Figure 5-23 Weekly measurements of the ratios of laevoglucosan and $nssK^+$ levels to $PM_{2.5}$ mass concentrations measured at Manjimup, WA.

The contribution of biomass burning to the particle load was assessed by calculating the average $PM_{2.5}$ concentration from weekly samples containing no laevoglucosan and the average $PM_{2.5}$ concentration derived from samples containing laevoglucosan. For autumn and spring, the ratio of samples with and without laevoglucosan was 1.9:1, suggesting that samples affected by wood smoke had a $PM_{2.5}$ concentration that was almost twice as high as the particle concentration of non-wood smoke samples.

Figure 5-24 shows the distribution of daily $PM_{2.5}$ levels by season and suggests that although Manjimup was not affected by fires over summer, the prescribed burning season did affect Manjimup's s air quality. During summer, about 90% of measured daily $PM_{2.5}$ levels were below 10 $\mu g/m^3$. In winter, this dropped to about 62%. For both summer and winter, only 1% of the daily $PM_{2.5}$ concentrations ranged between 15 and 20 $\mu g/m^3$, and none of the daily $PM_{2.5}$ values exceeded the NEPM of 25 $\mu g/m^3$.

In autumn and spring, higher daily $PM_{2.5}$ concentrations were observed with 70% and 62% of the values being below 10 $\mu g/m^3$ and 4-7% of daily values ranging between 15 and 20 $\mu g/m^3$. For both seasons, daily $PM_{2.5}$ values exceeded the NEPM of 25 $\mu g/m^3$, with 1% of the daily values for autumn and 3% of the daily values for spring.

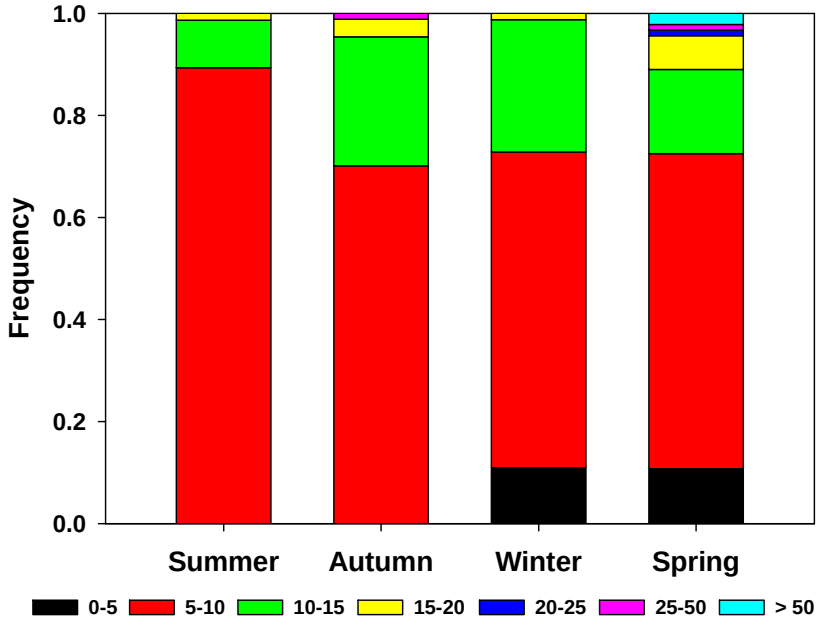


Figure 5-24 Hourly $PM_{2.5}$ seasonal distribution

5.4.4.2 Prescribed burning

Prescribed burning is the likely cause of elevated $PM_{2.5}$ concentrations observed in autumn and spring. Biomass burning is known to emit large amounts of PM, which can travel over large distances and affect the air quality of downwind communities. During autumn, the daily $PM_{2.5}$ NEPM of 25 $\mu g/m^3$ was exceeded once, whereas 3 days of exceedences were observed in spring. The continuous measurements of $PM_{2.5}$ and

ozone enabled to assess the duration, intensity and frequency of smoke plumes affecting Manjimup.

Figure 5-25 shows the time series plots for ozone and $PM_{2.5}$ concentrations at 4 different time periods of excessive haze (hourly PM levels exceeded 25 µg/m^3 over an extended period of time). The 4 periods relate to an autumn prescribed burn in May (3 days), and spring burning in October and November (1 day, 3 days and 6 days respectively). Elevated $PM_{2.5}$ levels coincide with high levels of ozone, which is consistent with an aged plume. At and near the fireground, ozone is destroyed through titration by NO, which is emitted in substantial quantities during biomass burning.. However ozone formation also occurs in smoke plumes by slower photochemical reactions involving NOx, VOCs, and UV radiation and therefore significant increases in ozone concentration are frequently observed in smoke plumes during long distance transport.

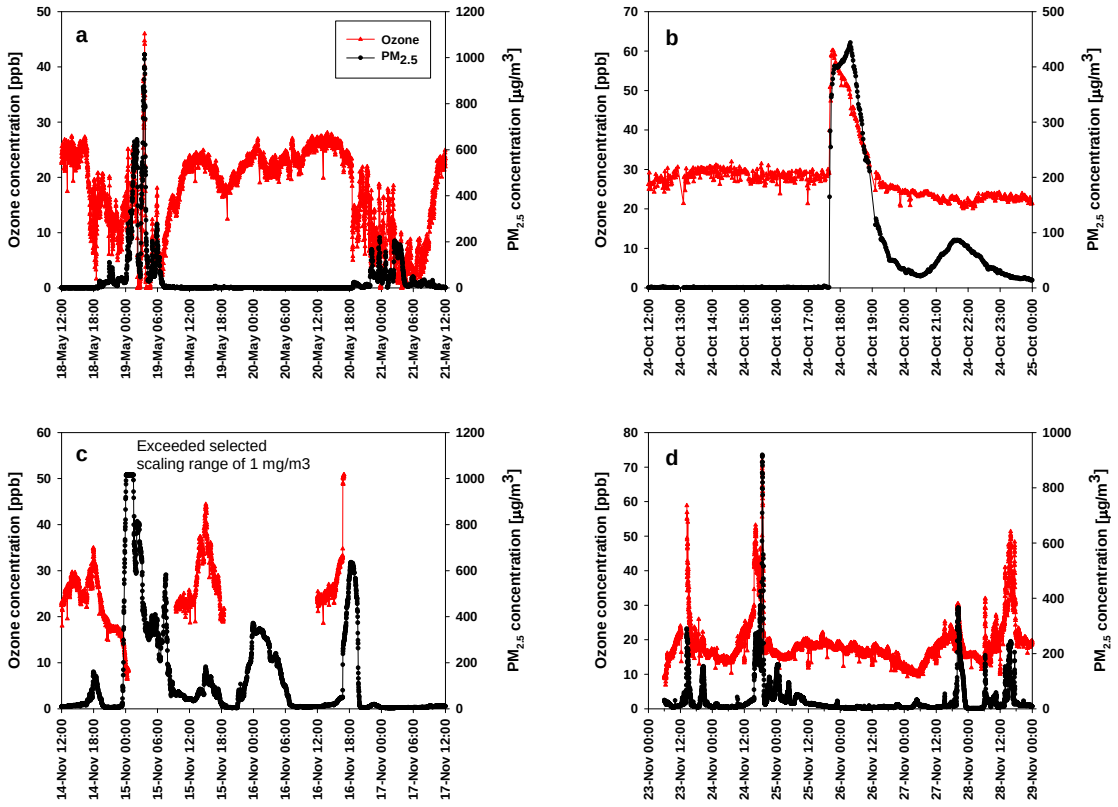


Figure 5-25 Time series plot of ozone and $PM_{2.5}$ concentrations for (a) autumn burn, (b), (c) and (d) spring burns

Prescribed burns are lit during late morning, and often continue smouldering through the night. Depending on the distance from Manjimup the arrival of the smoke plume is typically from late afternoon to early evening allowing up to 6 hours for photochemical ozone production. During smoke events, surface concentrations of both ozone and $PM_{2.5}$ increased significantly above background with highest levels generally occurring in the late evening. This is further highlighted in Figure 5-26 which displays the diurnal cycle of $PM_{2.5}$ levels measured for those days when hourly $PM_{2.5}$ concentrations exceeded 25 µg/m^3. There is a distinct difference between autumn and spring. For the autumn burning season, the diurnal pattern displays maximum levels overnight between 01:00 and 0:300 and minimum levels between 11:00 and 16:00.,. During spring burns, the first peak occurs near 18:00 however the concentration again increases later in the evening as the atmospheric mixing height decreases. Minimum levels are measured between 9:00 and 13:00.

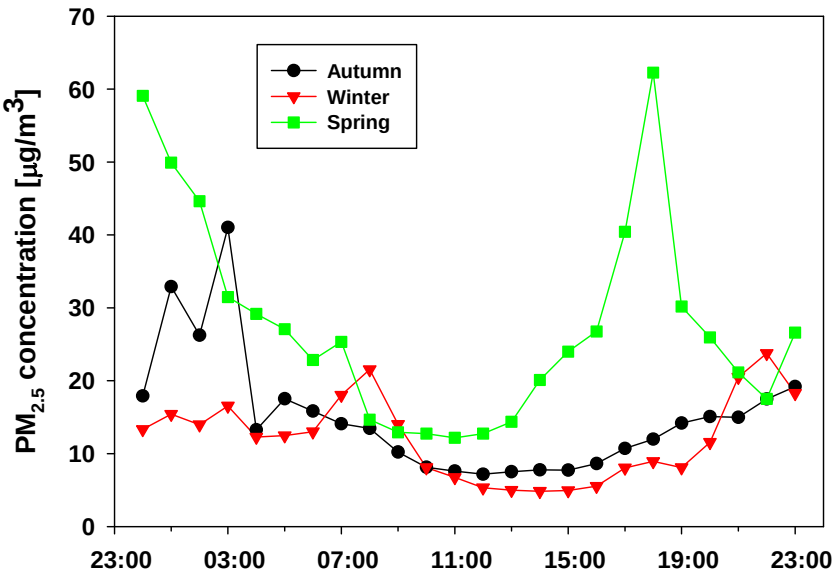


Figure 5-26 Daily variations of $PM_{2.5}$ concentrations according to season

Duration

In May (see Figure 5-25 a), the first smoke plume reached Manjimup on May 19 at about midnight. $PM_{2.5}$ levels gradually increased until about 2:15 to reach 650 µg/m^3. Levels quickly dropped, but then increased again at 3:00 and peaked at around 3:30

(1000 μg/m^3). There was a sharp decrease again followed by two additional peaks at 5:00 (200 μg/m^3) and 6:00 (280 μg/m^3). $PM_{2.5}$ reached low levels at about 7:00. A second plume reached on May 20 at about 22:00 and lasted for 6.5 hours. Maximum value reported during that time was 230 μg/m^3.

On October 24 (Figure 5-25 b), a smoke plume from a burn nearly 60 km from Manjiump, reached the town at about 17:30 increasing $PM_{2.5}$ levels 400 fold and doubling ozone levels. The concentrations remained elevated for about 2.5 hours. There was another smaller peak at about 21:30.

Extensive prescribed burning was conducted between November 14 and November 30 around Manjimup. During this period, smoke plumes travelled to Manjimup resulting in elevated particle levels on 8 days. Ozone data was intermittent due to a blocked filter.

The first episode in November lasted for 3 days from November 14-16 (Fig 5-25 c). A first peak occurred at 18:00 followed by a larger peak at midnight, with $PM_{2.5}$ concentrations increasing by 2 orders of magnitude and exceeding 1 mg/m^3. PM levels remained elevated for 2.5 hours. Another peak was observed between 7:30 and 9:00, which is likely to be traffic related. $PM_{2.5}$ levels dropped after 9:00, but remained high during the day with an average concentration of 60 μg/m^3. A smaller peak occurred again at about 15:00 with concentrations returning to normal on November 15 at about 18:00, but then increasing again between 21:00 and 7:00. Another peak was observed on November 16 between 16:45 and 20:00.

The second episode in November occurred between November 23 and November 29, with 1 day of low exposures (Fig 5-25 d). The first peak occurred on November 23 at 14:00 increasing $PM_{2.5}$ concentrations to 300 μg/m^3. Levels remained high for about 1.5 hours. Another 1-hour peak (concentrations of 150 μg/m^3) occurred between 20:00 and 21:00. A second smoke plume arrived on November 24 at 15:40, initially rising $PM_{2.5}$ levels 10 fold and doubling ozone concentrations. $PM_{2.5}$ and ozone concentrations peaked at 18:45, with $PM_{2.5}$ levels around 900 μg/m^3 and ozone levels at 70 ppb. Ozone

levels dropped back to background at about 19:30, however $PM_{2.5}$ concentrations remained high (30-160 μg/m^3) until 5:00. The next plume arrived on November 27 at 19:30, peaking at 360 μg/m^3, and dropping to low concentrations at about 23:00. This was followed by another increase for about 1 hour, doubling ozone concentrations and increasing $PM_{2.5}$ concentrations 10 fold. High levels were again measured on November 28 between 13:00 and 18:00.

Intensity

Figure 5-27 shows the distribution of PM concentrations from smoke events in May and November. In general, higher concentrations were measured in November compared to May. In May, as shown in Figure 5-27, about 65% of PM levels are below 10 μg/m^3, whereas in November PM levels below 10 μg/m^3 only made up 30-40% of the PM levels measured during that period. This is consistent with low intensity fires during spring, when fuel moisture is higher. Smouldering fires emit higher levels of particles. In autumn fires are of higher intensity resulting in lower particle emissions compared to spring burns.

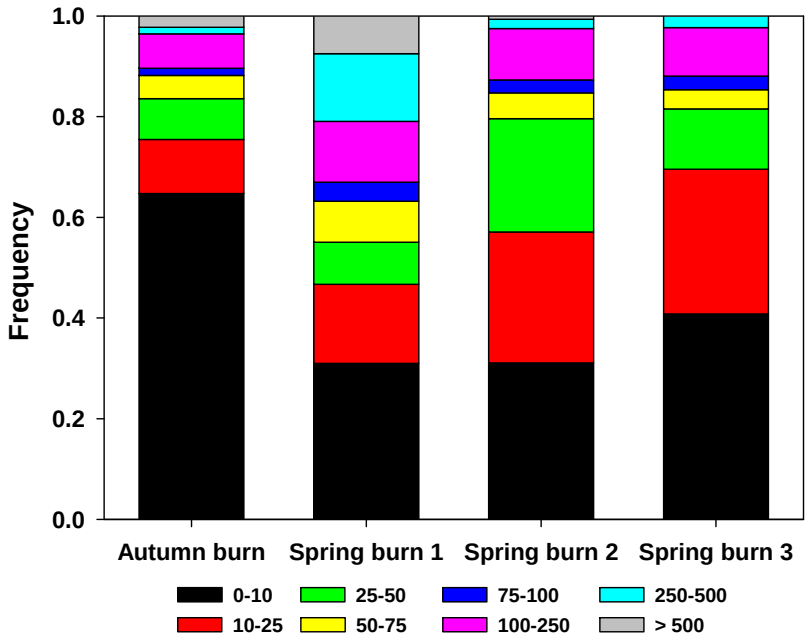


Figure 5-27 Distribution of PM concentration for smoke events in autumn and spring. Autumn burn, spring burn 1, spring burn 2 and spring burn 3 correspond to the periods of May 18-21, Nov 14-17, Nov 23-25 and Nov 27-29, respectively.

5.4.4.3 Winter

High particulate levels in conjunction with elevated wood smoke tracers were measured in winter. As shown in Figure 5-26 $PM_{2.5}$ levels are 3 times higher at night than during daytime. The diurnal pattern is influenced by daytime atmospheric mixing, but also by night time wood burning activities. Concentrations peak at 8:00 (likely to be a traffic peak) and 22:00. Figure 5-28 shows the time series plot of $PM_{2.5}$ and temperature over winter. The $PM_{2.5}$ peaks occurred in the morning, except for the peak observed on August 12 which occurred at 22:00. The elevated $PM_{2.5}$ levels measured during the night of August 12 coincide with the lowest temperatures measured at Manjimup during winter, with a low of 2.5 ºC and also coincide with the highest levels of laevoglucosan. As can be seen in Figure 5-29 $PM_{2.5}$ levels increased at 21:00 and remained elevated through the night.

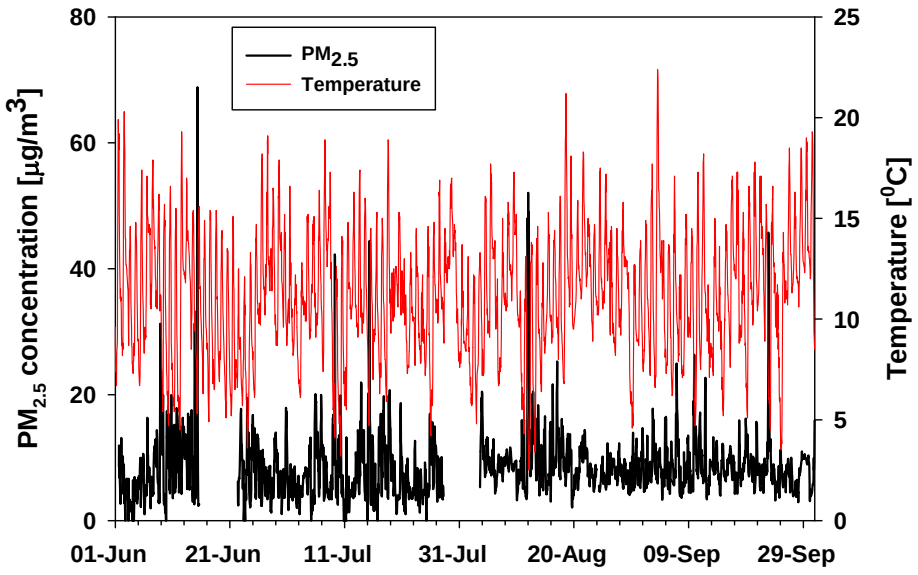


Figure 5-28 Time series plot of hourly $PM_{2.5}$ concentrations and temperature measured at Manjimup, WA.

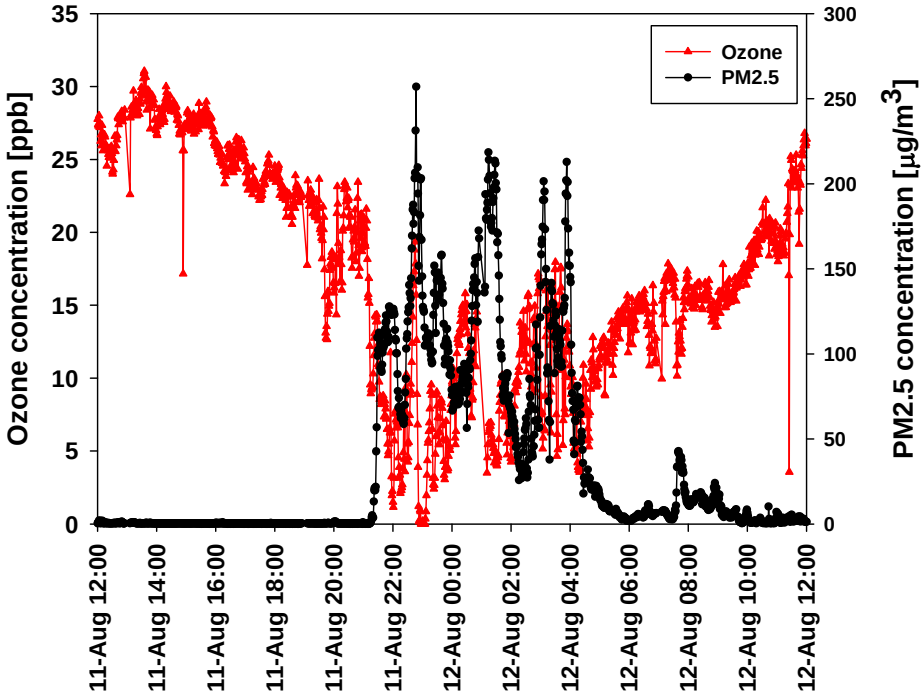


Figure 5-29 Time series plot of minutely averaged surface ozone and $PM_{2.5}$ concentrations measured at Manjimup, WA during a high PM episode in winter.

5.4.5 Summary

At Manjimup elevated $PM_{2.5}$ levels were monitored in November coinciding with the spring prescribed burns as well as in May as a result of autumn prescribed burns. The effect of biomass burning almost doubled average $PM_{2.5}$ concentrations. During the prescribed burning season, the daily $PM_{2.5}$ NEPM was exceeded for a total of 4 days during the year. One exceedence day was reported on May 19 and three exceedence days (Nov 15, 16 and 24) were reported during the spring prescribed burning activities. Using the definition of smoke events defined in section 5.1.3, Manjimup recorded excessive haze on 15 days between December 2006 and December 2007. During autumn we recorded 3 days of smoke events (3%) and during spring 10 days (11%). There was one day in spring when $PM_{2.5}$ levels stayed above 25 µg/m^3 for 14 hours. During the majority of smoke events (62%), elevated particle levels (> 25 µg/m^3) were recorded over a 1-3 hour period. Looking at the diurnal pattern of $PM_{2.5}$ concentrations measured during the prescribed burning periods, elevated levels usually occurred during

the night, except for the spring prescribed burns at Manjimup. There is another distinctive peak at 18:00 ($PM_{2.5}$ about 60 μg/m^3).

Surface ozone concentrations never exceeded the 1-hour NEPM of 100 ppb, even though significant ozone increases were monitored during smoke plume events. The time series of surface ozone and $PM_{2.5}$ concentrations measured during smoke events have shown simultaneous increases of both pollutants. This is consistent with an aged smoke plume impacting on Manjimup's air quality. Plume ageing occurs during long-range transport when photochemical reactions in the plume form ozone and secondary aerosols.

5.5 Ovens

5.5.1 Description of site

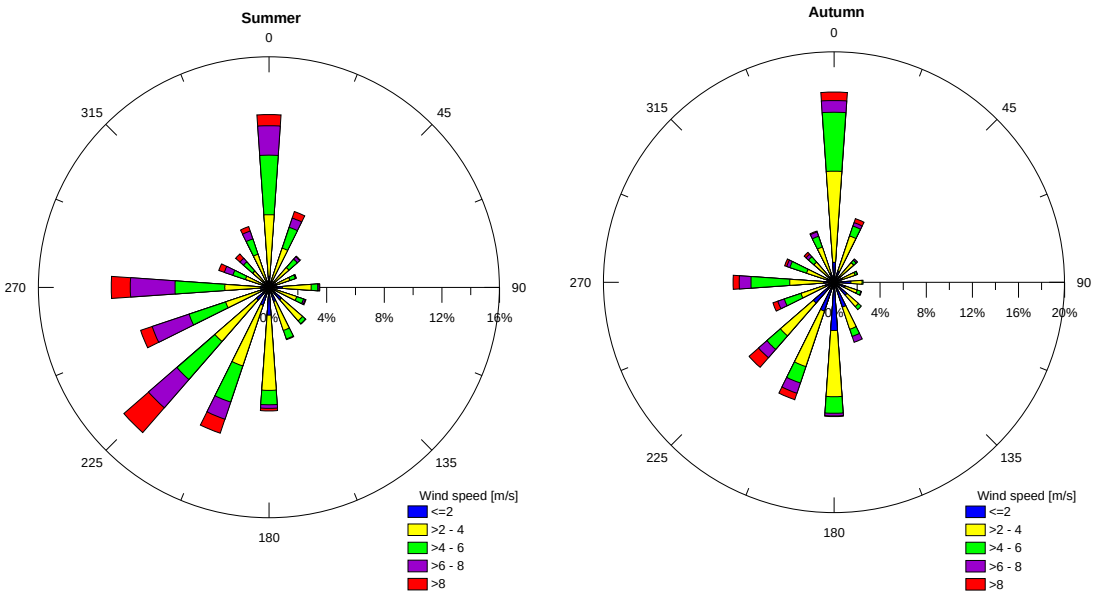
The sampling site is located at the site of the regional office of the Department of Environment and Sustainability at Ovens. Figure 5-30 shows the instrument set-up.

Figure 5-30 Sampling location and instrument set-up at Ovens, VIC

5.5.2 Meteorology

Average monthly temperatures range between 7 and 25 ºC. This range is greater than that observed at Manjimup. There is a significant drop in monthly average temperatures between February and June and the wettest month is May.

Figure 5-31 shows the wind roses for Wangaratta by season using half-hourly wind speed and direction data. Wangaratta is approximately 55 km west of Ovens and the closest meteorological station to Ovens. Lowest wind speeds are observed during autumn and winter with highest wind speeds measured in summer and spring. Between seasons, the wind patterns are similar (unlike wind Manjimup and Darwin). This is due to the topographical location of Ovens, the Ovens Valley oriented in a northwest-southeast direction and surrounded by the Victorian Alps in the southeast. The topography of the area and dominant wind direction has an impact on the smoke plume dispersion and on the local air quality.



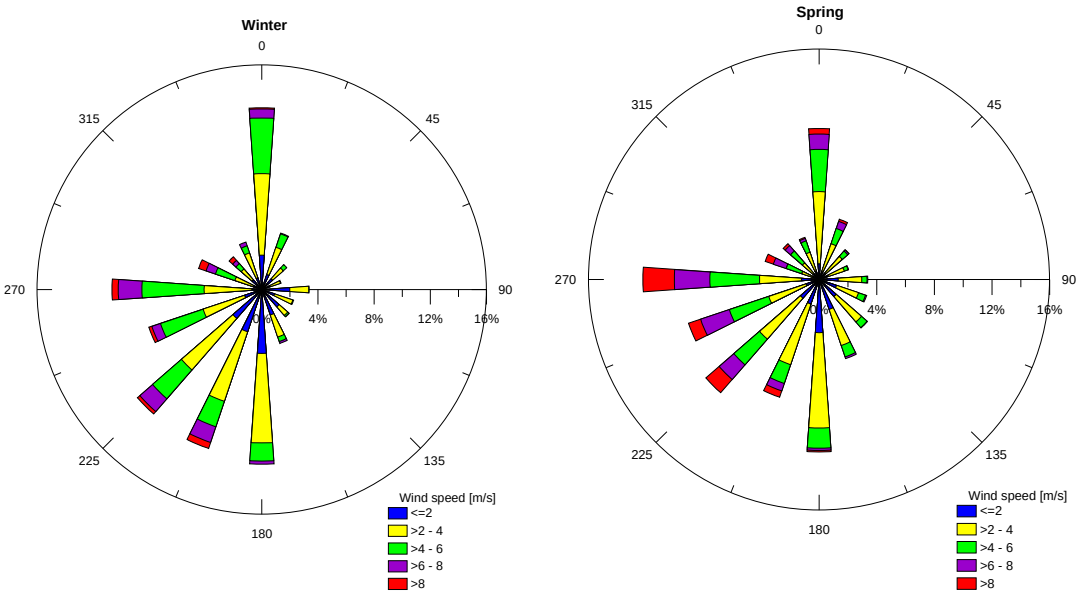


Figure 5-31 Half-hourly wind direction and wind speed data by season measured at Wangaratta

5.5.3 Observations

Measurements conducted at Ovens included continuous measurements of ozone and $PM_{2.5}$, weekly mass concentrations of $PM_{2.5}$ and weekly BTEX measurements.

5.5.3.1 Data quality

DustTrak

Similar to Manjimup, the DustTrak was configured to automatically transmit aerosol mass concentration values via an analogue output voltage signal that is proportional to the displayed concentration. At Ovens the scaling range was initially set-up at 0-10.0 mg/m^3 to ensure that high aerosol concentrations emitted from the Great Alpine fires would be measured. This scaling range was dropped to 0-1 mg/m^3 mid January and again dropped to 0-0.1 mg/m^3 end of October. Data was also digitally logged at 15-minute intervals from mid December 2006 until mid January 2007 and mid August 2007 until end of December 2007.

Over the duration of the sampling period, no DustTrak data was collected between January 15 to 22, from February 26 to April 12 and from May 14 to 21. On October 24th, data-logging records began displaying sudden increases in the $PM_{2.5}$

concentrations. These discontinuities occurred either once during midday or once in the morning and once in the evening and most likely represent instrument artefacts rather than actual increases in $PM_{2.5}$ levels.

Due to sampling difficulties using the DustTrak, gravimetric mass concentrations will be used in the discussion of PM levels, and DustTrak concentrations will only be used as qualitative data during major smoke events, such as the wildfire season and the prescribed burning season, to assess the intensity and the duration of the PM impact on the community.

For those periods, as mentioned under section 5.4.3.1, gravimetric mass measurements of $PM_{2.5}$ made using the MicroVol were used to calibrate the $PM_{2.5}$ concentrations determined by the DustTrak. The calibration required an integration of the real-time DustTrak $PM_{2.5}$ concentration profile and calculation of average DustTrak $PM_{2.5}$ concentrations. These average $PM_{2.5}$ concentrations were then compared to the average gravimetric mass measurements of $PM_{2.5}$ collected on the filter of the MicroVol. An adjustment factor of 3.0 was used for the bushfire period, which is in agreement with other studies. For the prescribed burning season, the DustTrak underestimated particle concentrations compared to the gravimetric mass measurements. The weekly DustTrak data was plotted against the weekly gravimetric measurements made using the MicroVol and the following equation was used to correct the DustTrak data ($y = 2.69\ x + 10.9$) (Figure 5-32).

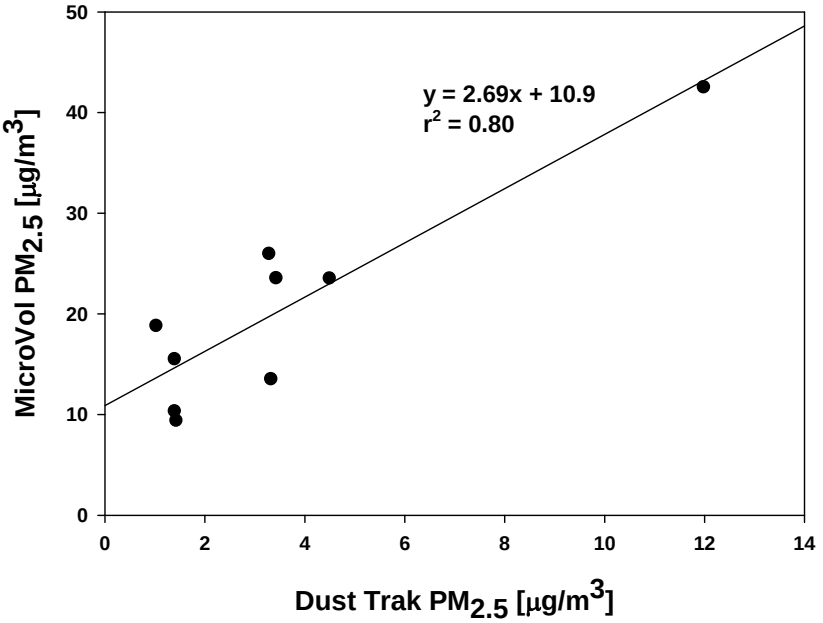


Figure 5-32 Calibration curve for DustTrak $PM_{2.5}$ measurements versus gravimetric $PM_{2.5}$ mass measurements made using a MicroVol

5.5.3.2 Time series analysis

Figure 5-33 displays the time series of hourly concentrations of ozone (averaged from minute data) measured at Ovens between December 2006 and January 2008. Hourly ozone concentrations ranged from 0 to 155 ppb, with an average of 20 ppb. The maximum 4-hour running average ozone value was measured at 139 ppb. Ozone air quality guidelines were exceeded during the severe wildfire events of summer 2006/2007 (Figure 5-34). The hourly NEPM of 100 ppb was exceeded on 13 occasions between December 19 and 23, while the 4-hour NEPM of 80 ppb was exceeded 34 times during that time.

Hourly $PM_{2.5}$ levels ranged from 0 to 2.23 mg/m^3 (corrected using an adjustment factor of 3.0), with maximum levels largely exceeding air quality guidelines during December 2006 and January 2007 (Figure 5-34). The daily $PM_{2.5}$ NEPM was exceeded on 13 days between December 13 and January 14. The actual number of exceedence days is likely to be higher during the 2006/2007 bushfire season, since measurements at Ovens did

not commence until approximately 2 weeks after the start of the Great Alpine fires, and ended on January 15.

Excluding the summer wildfires, hourly ozone concentrations ranged from 0 to 85 ppb, with an average of 19.5 ppb. None of the ozone levels exceeded the 1-hour ozone NEPM of 100 ppb. During the prescribed burning season, $PM_{2.5}$ levels ranged from 0 to 263 μg/m^3 (corrected using the adjustment equation previously discussed). Seven days of exceedences of the $PM_{2.5}$ NEPM were observed during the autumn prescribed burning season between April 20 and May 13 (Figure 5-34).

The bushfire period as well as the prescribed burning season will be analysed in more detail in section 5.5.4.

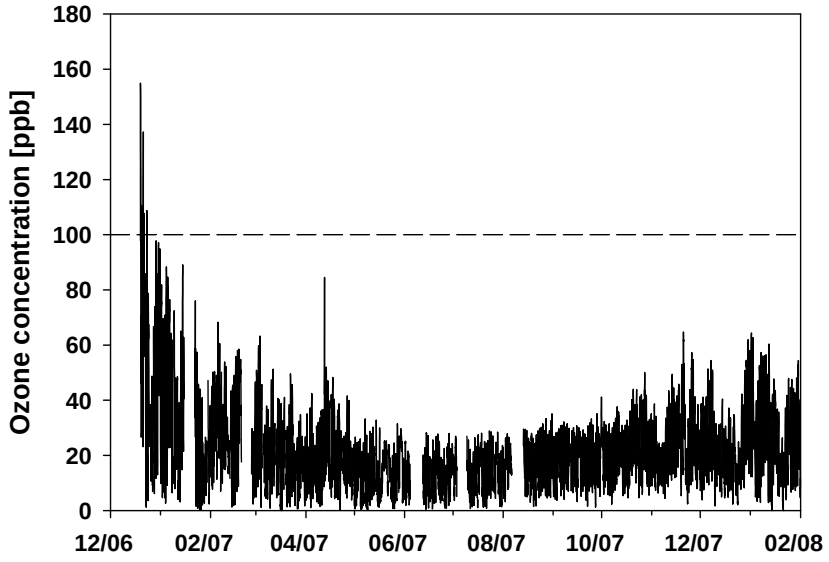


Figure 5-33 Time series plot of minutely averaged surface ozone concentrations measured at Ovens, VIC between December 2006 and January 2008.

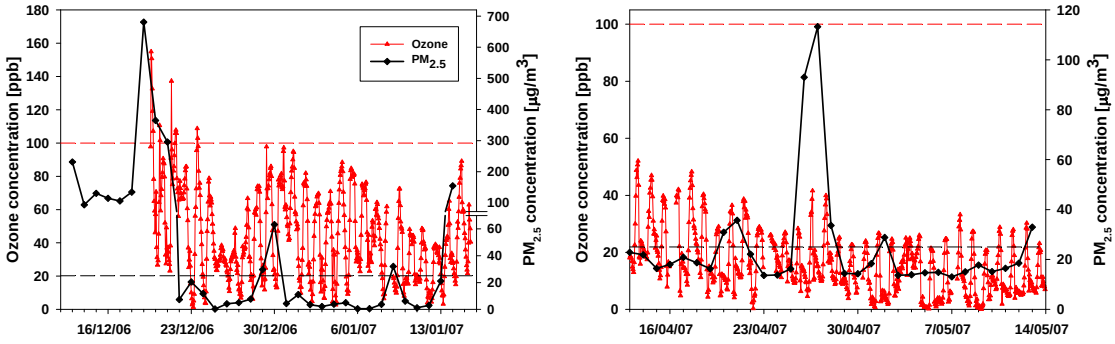


Figure 5-34 Time series plot of hourly ozone and daily $PM_{2.5}$ concentrations measured during the wildfire events in summer 2006/2007 and the autumn prescribed burning season in 2007.

5.5.3.3 Monthly distribution of Ozone and $PM_{2.5}$ concentrations

Figure 5-35 shows the monthly distribution of ozone and $PM_{2.5}$ concentrations. Ozone concentrations were maximum (>150 ppb, with an average of 35 ppb) in December 2006 and minimum concentrations of 12 ppb in June 2007. Lowest concentrations were measured between May and July. After July, ozone concentrations gradually increased to 25 ppb measured in January 2008.

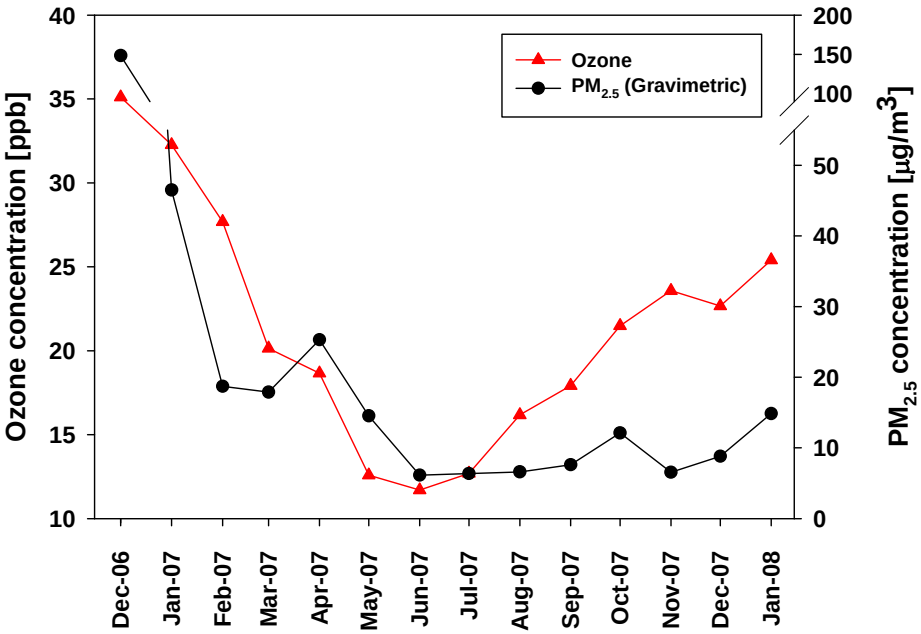


Figure 5-35 Monthly distribution pattern of ozone and $PM_{2.5}$ concentrations measured at Ovens, VIC.

This seasonal cycle contrasts with Manjimup where maximum ozone levels occurred during winter The monthly cycle observed in Ovens is more consistent with polluted urban regions, with an increase of ozone during summer, associated with increased photochemical activity during the warmer summer months and decreased photochemistry in winter. Unlike Manjimup, Ovens is 200 km inland and air masses reaching Ovens most likely have traversed long distances over land where ozone chemistry is dominated by NO_x and VOC emissions from biogenic and other land-based sources.

The high concentrations in December were matched by very high $PM_{2.5}$ concentrations. $PM_{2.5}$ concentrations decreased significantly from 150 to about 20 $\mu g/m^3$ in February/March 2007 and peaked again in April (25 $\mu g/m^3$) during the prescribed burning season. $PM_{2.5}$ concentrations remained low (6-8 $\mu g/m^3$) between June and November, expect for an additional peak in October (12 $\mu g/m^3$), and then gradually increased to 15 $\mu g/m^3$ in January 2008.

5.5.3.4 Diurnal cycle of ozone and $PM_{2.5}$

As can be seen in Figure 5-36, ozone concentrations vary diurnally with photochemistry and turbulent mixing from the free troposphere. The highest $PM_{2.5}$ concentrations were observed at night and early morning, dropping to lowest concentrations in the afternoon. This diurnal cycle is due to inversions being much lower at night trapping the pollutants closer to the ground.

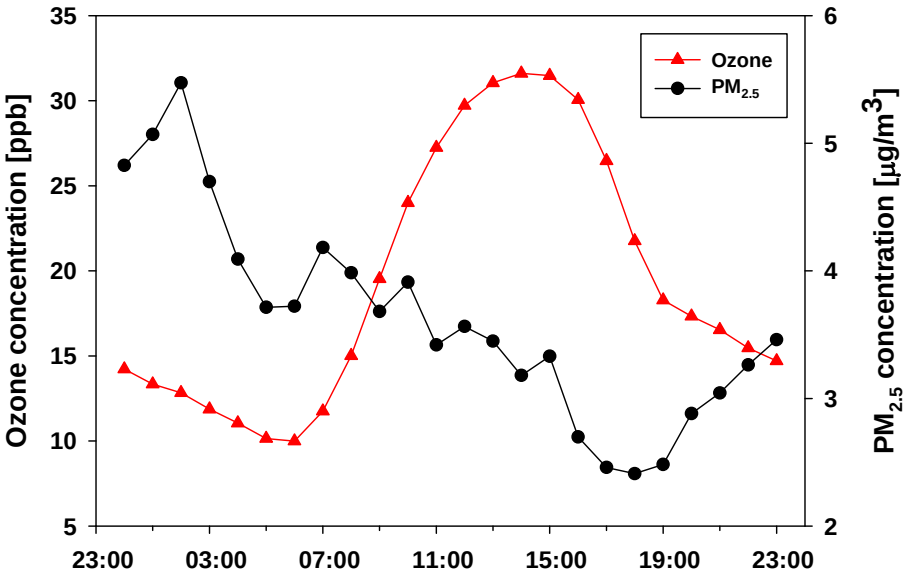


Figure 5-36 Diurnal distribution pattern of ozone and $PM_{2.5}$ concentrations measured at Ovens, VIC.

5.5.3.5 BTEX

Weekly measurements of BTEX were performed on samples collected between December 2006 and December 2007. As shown in Table 5-5, toluene shows the highest average annual concentration, followed by benzene, xylenes and ethylbenzene. Maximum concentration of toluene, benzene, xylenes and ethylbenzene were 1.06 ppb, 0.615 ppb, 0.075 ppb and 0.286 ppb respectively, well below the NEPM of 3 ppb for benzene and 100 ppb for toluene and lower than the values reported in other studies.

Table 5-5 Comparison of ambient BTEX concentrations in ppb measured at Ovens with concentrations measured in other locations

Location	Benzene	Toluene	Ethylbenzene	Xylenes
Ovens	0.11	0.12	0.01	0.03
Ovens (excl wildfires)	0.07	0.1	0.01	0.02
Aspendale	0.3	0.8	0.1	0.6
Melbourne	1.1	1.8	0.2	0.8
Sydney urban	1.0	2.3	0.3	1.5
Sydney urban fringe	0.4	0.8	0.1	0.6
Launceston summer	0.23	0.59	0.08	0.45
Launceston winter	1.56	3.11	0.29	1.51

The weekly concentrations (average of two samples) of BTEX observed throughout the year are presented in Figure 5-37 and 5-38.

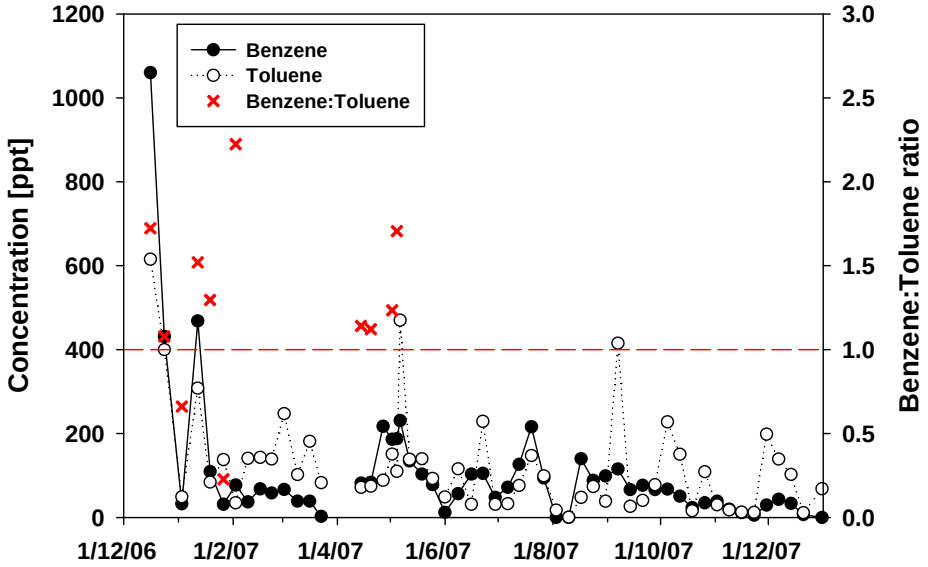


Figure 5-37 Weekly concentrations of benzene and toluene measured at Ovens, VIC between December 2006 and December 2007. Also displayed on the graph is the ratio of benzene vs. toluene for the bushfires in 2006/2007 and the autumn prescribed burning season.

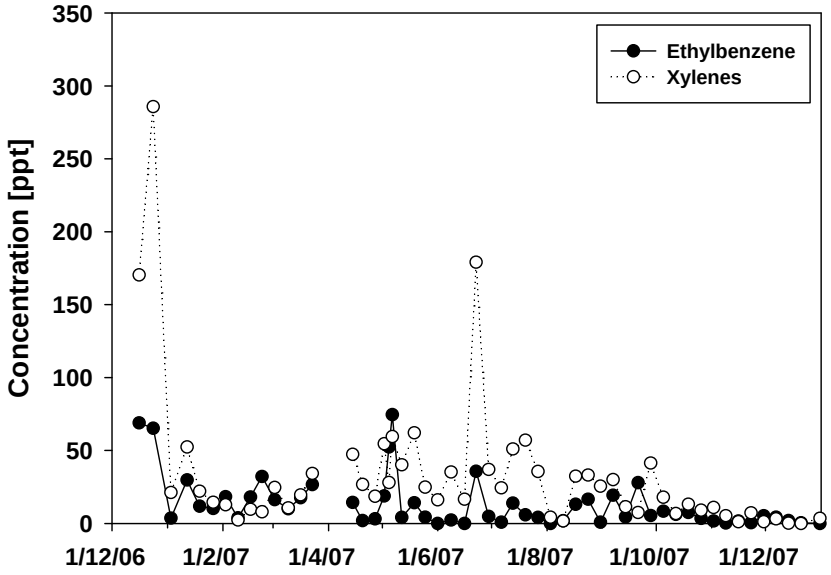


Figure 5-38 Weekly concentrations of ethylbenzene and xylenes measured at Ovens, VIC between December 2006 and December 2007.

Highest BTEX concentrations were measured during the Great Alpine fires in December 2006. Benzene levels were higher than toluene levels observed in autumn during the prescribed burns and throughout winter. The annual benzene to toluene ratio

was calculated at 0.7-0.8 which is about twice as high as the ratio measured in Aspendale (Lawson et al., 2005) and in Launceston during summer (Galbally et al., 2004). Winter measurements in Launceston (influenced by woodsmoke from domestic woodheaters) resulted in benzene to toluene ratio of 0.5 which is still below the ratio measured in this study.

5.5.4 Smoke events

Ovens was affected by an intense bushfire season in December 2006 and January 2007 and by a range of prescribed burns conducted in autumn 2007.

5.5.4.1 Chemical analysis

Figure 5-39 shows the time series of weekly $PM_{2.5}$ concentrations as well as the levels of laevoglucosan and non sea-salt potassium $nssK^+$, both markers of wood smoke between December 2006 and the beginning of December 2007, and from the end of January 2007 until January 2008 (so that the samples effected by the severe bushfires in December 2006 and January 2007 are excluded).

The weekly gravimetric measurements of $PM_{2.5}$ show three distinctive peaks over the sampling period, with high $PM_{2.5}$ concentrations corresponding to high concentrations of laevoglucosan and non sea-salt potassium. The first period occurred during the Great Alpine fires from December 2006 to January 2007. During that time period, particle levels greatly exceeded air quality guidelines, reaching daily levels as high as 681 $\mu g/m^3$, which is 27 times higher than the $PM_{2.5}$ NEPM. The other period of high $PM_{2.5}$ levels coincides with the prescribed burning season in April/May. As for the wildfires, concentrations of laevoglucosan and $nssK^+$ are high, confirming the source of PM as being bushfire smoke. Another $PM_{2.5}$ peak in October is not associated with elevated levels of laevoglucosan or $nssK^+$, indicating that the PM may have originated from other sources than wood smoke.

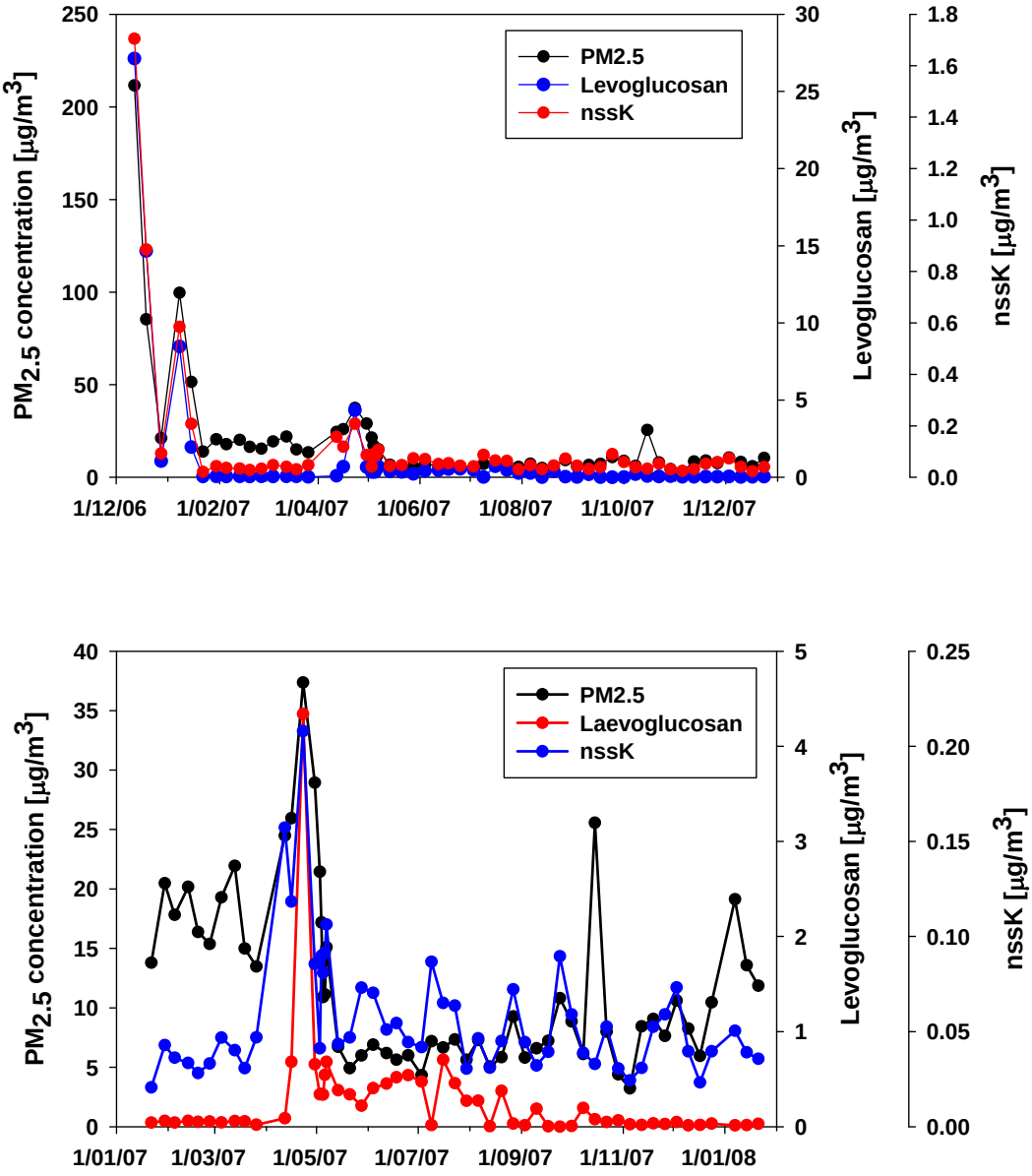


Figure 5-39 Weekly concentrations of $PM_{2.5}$, laevoglucosan and $nssK^{+}$ measured at Ovens, VIC (a) between December 2006 and December 2007 and (b) between February 2007 and January 2008.

5.5.4.2 Bushfire season

In the summer of 2006/2007, Victoria was affected by the longest recorded fires in Victoria's history. The fires were started by lightning on 1 December 2006 in the Alpine regions of eastern Victoria and burned an area of 1,048,238 hectares over a 69 days period. Due to its location within the Alpine region, Ovens was severely affected by the bushfires.

Figure 5-40 shows the time series of hourly ozone and $PM_{2.5}$ concentrations during December 2006 and January 2007. Monitoring began about 12 days after the fires had started and the high pollutant levels indicate that Ovens was already severely impacted by the smoke.

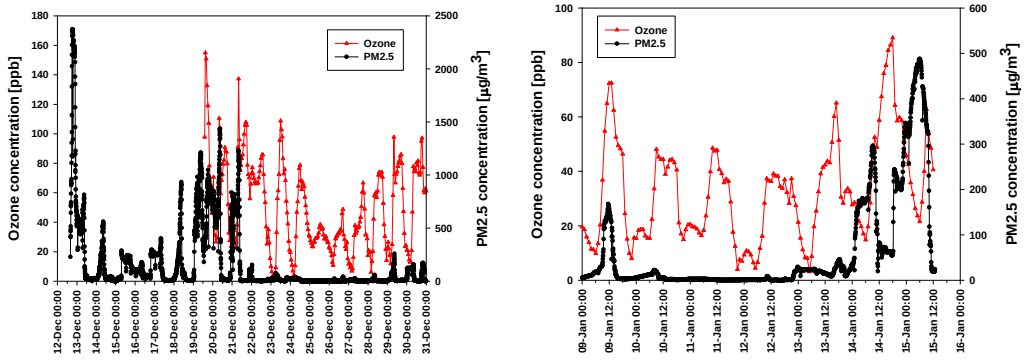


Figure 5-40 Time series plot of hourly concentrations of ozone and $PM_{2.5}$ measured at Ovens VIC during the 2006/2007 bushfires.

As can be seen on Fig 5-41 about 30% of the hourly $PM_{2.5}$ concentrations between December 12 and January 15 were above 25 µg/m^3 and 1% of those values exceeded 1 mg/m^3. As can be seen on the time series plot (Figure 5-40), the majority of very high $PM_{2.5}$ concentrations occurred in the initial part of the monitoring period, between 12 and 22 December. Hourly $PM_{2.5}$ levels exceeded 1 mg/m^3 on December 12 and December 19-21. On December 12, peak levels occurred between the hours of 17:00 and 22:00, whereas for the other 3 days, the very high levels were measured in the morning between 8:00 and 10:00. These high episodes were followed by a range of episodes of low $PM_{2.5}$ concentrations (hourly levels below 50 µg/m^3). These low episodes were observed almost every day between December 13 and 21 and occurred during the day generally between the hours of 10:00/11:00 and 1:00/2:00. This may be a result of mixing processes in the planetary boundary layer venting surface emissions to higher levels during the day and hence diluting pollutants.

The high level episode (December 12-21) was followed by a low level episode which lasted from December 22 to January 14. The majority of hourly $PM_{2.5}$ concentrations were below 50 µg/m^3, with higher concentrations (maximum hourly levels of 125-200

μg/m^3) measured on December 29 (7:00-9:00), December 30 (1:00-9:00 and 19:00-22:00) and January 9 (10:00-13:00). On January 15, hourly concentrations increased to 480 μg/m^3.

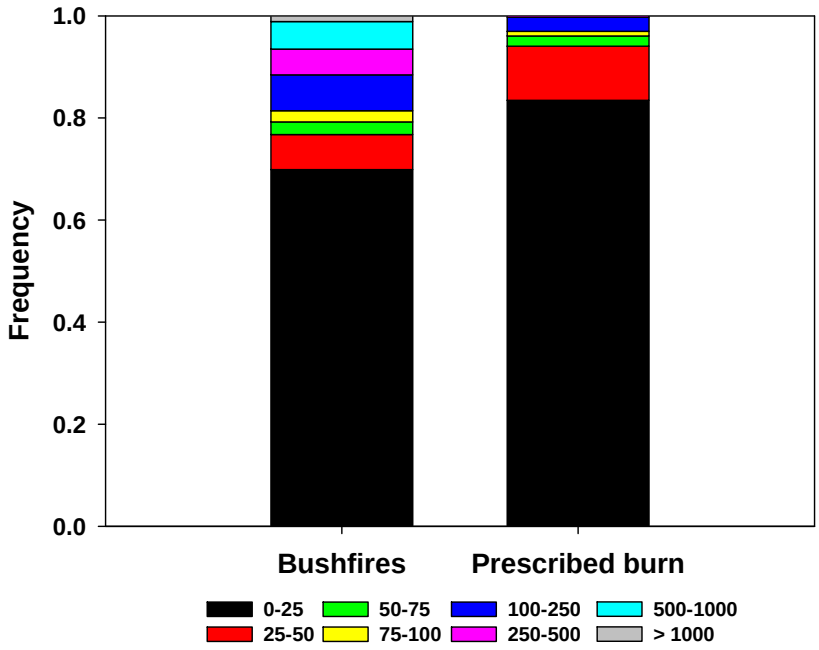


Figure 5-41 Distribution of PM concentration for bushfires and autumn prescribed burns.

5.5.4.3 Prescribed burns

In Victoria, the majority of prescribed burns are conducted during autumn by the Department of Sustainability and Environment (DSE). Generally weather conditions are milder, the fire behaviour is less intense and the fuel is moist from initial rains at the end of the summer. All these factors make it easier to control the fire. In autumn 2007 however due to the severe drought, prescribed burning was reduced as the fuel moisture was too low to safely conduct the burns and also to avoid the risk of severe scorching of the forest canopy as the fire intensity would be too high.

A few prescribed burns were conducted around Ovens (Figure 5-42) during the period of mid April to beginning of May. Additionally, Hancock plantations conducted a range of slash and heap burns in logged pine plantations close to Rosewhite, Buffalo River and Merriang South. Exact dates for the prescribed burns have not been specified

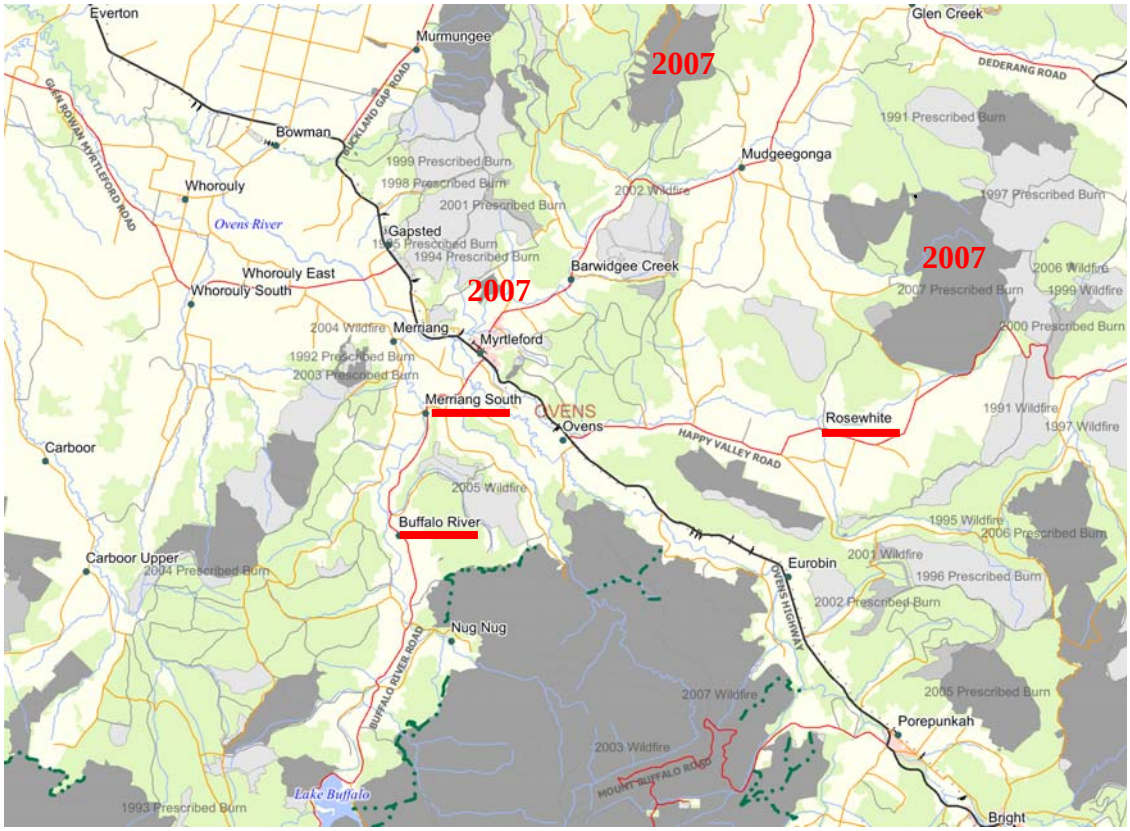


Figure 5-42 Location of prescribed burns and slash burns in logged pine plantations around Ovens

Elevated $PM_{2.5}$ levels were observed during April and May 2007. Prescribed burns were carried out in the week of April 23-29, which resulted in the smoke plume shown on Figure 5-43. Pine plantation burn-offs were carried out between April 23 and May 5.

Ozone concentrations followed a diurnal trend with lowest levels measured between the hours of 17:00 and 6:00 and highest levels during the day.

Duration

The first $PM_{2.5}$ peak was measured on April 21 when levels quadrupled from 30 to 120 $\mu g/m^3$. Ozone levels remained high for another 3 hours and began to decrease by 19:00. The highest impact was observed on April 26. A short peak (195 $\mu g/m^3$) was monitored at 12:30. After that concentrations gradually increased until 22:00 to peak at 268 $\mu g/m^3$. This was followed by a gradual decrease to 90 $\mu g/m^3$ (April 27 8:00). There were a few peaks between 8:00 and 13:00 at which stage $PM_{2.5}$ concentrations dropped to 40 $\mu g/m^3$ and increased again at 17:30 by 3 times. This was followed by a decrease to 40 $\mu g/m^3$

and an increase to 126 µg/m^3. On April 28 at 2:00, PM levels gradually decreased to reach 20 µg/m^3 at 11:00. There was no change in ozone levels during that time suggesting that the plume was not aged

Additional peaks were observed at the beginning of May. On May 1 $PM_{2.5}$ levels increased to about 70 µg/m^3 (occurring in the evening) and a gradual increase in ozone concentrations was observed on May 2 at about 18:00. Unlike the April plume, the ozone concentrations increased overnight from 5 to 25 ppb. Ozone levels returned back to the normal diurnal cycle on May 4.

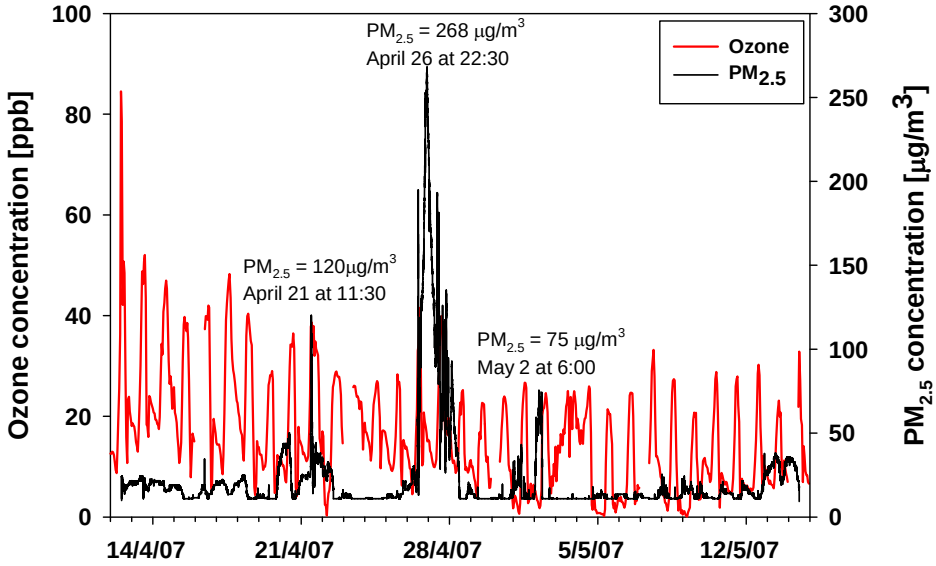


Figure 5-43 Time series plot of hourly concentrations of ozone and $PM_{2.5}$ measured at Ovens VIC during the autumn prescribed burns

Intensity

Figure 5-41 shows the hourly distribution of PM concentrations for the prescribed burn period between April 13 and May 13. The majority of PM levels (83%) were below 25 µg/m^3, 11% were between 25 and 50 µg/m^3 and 3% of the measured $PM_{2.5}$ concentrations were in the range of 100-250 µg/m^3. The maximum hourly $PM_{2.5}$ level was measured at 268 µg/m^3.

5.5.5 Summary

The bushfire events of summer 2006/2007 had a significant impact on Ovens air quality, increasing average $PM_{2.5}$ concentration 5-fold. The high $PM_{2.5}$ concentrations measured during the bushfires are consistent with levels reported in other studies that have looked at air quality impacts of major wildfire events. During the summer period, both ozone and $PM_{2.5}$ NEPMs were largely exceeded.

Elevated $PM_{2.5}$ concentrations were also measured in April/May, in association with prescribed burns in eucalypt forests and slash burns in logged pine plantations around Ovens. Figure 5-42 shows the location of the prescribed burns conducted by DSE and also a range of pine plantation burn-offs done by Hancock Plantations during the week of May 3-7.

Between April 12 and May 13, 12 days of smoke events were recorded with a monthly average in April exceeding the daily NEPM. Between April 13 and April 30, there were 5 exceedence days (April 20, 21, 26, 27, 28), with an average concentration of 61.3 $\mu g/m^3$. Non exceedence days during that month averaged 18 $\mu g/m^3$. There were 2 additional exceedence days recorded in May (May 2 and 13).
At Ovens, smoke events lasted between 1 hour and up to 24 hours. 24-hour day periods above 25 $\mu g/m^3$ were observed on two days during autumn. About 45% of days considered as smoke event days had smoke periods lasting between 2 and 5 hours, and 66% of days recorded smoke periods lasting between 11 and 24 hours. Looking at the diurnal pattern of $PM_{2.5}$ concentrations measured during the prescribed burning periods, elevated levels usually occurred during the night.

Surface ozone concentrations never exceeded the 1-hour NEPM of 100 ppb during autumn. Ozone levels were not always affected by the prescribed burning activities. This suggests that Ovens is exposed to fresh smoke plume, which would be consistent for burns in close proximity to the monitoring station, as well as to aged smoke plume consistent with haze from burns remaining for longer periods in the valley.

5.6 Discussion

The seasonal and diurnal pattern of measured ozone and particle concentrations along with the chemical composition of the particles allowed us to make qualitative suggestions about the sources of aerosol and about the contribution of prescribed burning to the area's air quality. Smoke from prescribed burns has been shown to be an important source of particle loading at each site, as shown by the elevated levels of wood smoke tracers coinciding with high levels of PM concentrations.

Tables 5-8 and 5-9 summarize the average monthly and annual concentrations of ozone and $PM_{2.5}$ over the 1-year sampling period. The data show elevated monthly average $PM_{2.5}$ concentrations during the periods of prescribed burning, with a lesser effect on monthly ozone concentrations, with the exception of Darwin. In Darwin, monthly PM and ozone concentrations are higher during the mid and late burning season.

5.6.1 Particle concentrations

Annual average concentrations of $PM_{2.5}$ were highest for Ovens which was due to the intensive bushfires in the Victorian Alps. Due to the intensity and irregularity of the Victorian bushfire season, measurements from that period will be excluded for the discussion and focus will be on the effects of prescribed burning activities on the air quality of rural communities. Excluding the bushfire season, annual average concentration at Ovens dropped down to 11.9 $\mu g/m^3$, which is still about 20% and 27% higher than the annual averages measured at Casuarina and at Manjimup respectively (Table 5-9). Compared with urban air quality data (Table 5-6), annual averages measured at Ovens and Manjimup are lower than annual average measured at Sydney, but similar to those measured in Melbourne and Launceston. While Melbourne and Launceston both show maximum $PM_{2.5}$ concentrations during winter, at Manjimup and Ovens highest monthly averages are recorded during autumn (Ovens, if excluding bushfire season) and spring (Manjimup).

Table 5-6 Monthly and annual averages of $PM_{2.5}$ concentrations measured in Melbourne, Sydney, Launceston, Manjimup and Ovens.

Month	Melbourne	Sydney	Launceston	Manjimup	Ovens
Jan 07	9.3	18	6	6.4	47
Feb 07	6.3	19.5	3	9.1	19
Mar 07	6.3	18	5	7.2	18
Apr 07		19	7	8.5	**25**
May 07		20	12	13.5	14.5
Jun 07	**15.5**	23	**20**	8	6.2
Jul 07	7.6	25	17	9.2	6.4
Aug 07	8.5	24	9	10.8	6.6
Sep 07	6.3	18.5	7	9.1	7.6
Oct 07	7.3	21.5	4	9.2	12
Nov 07	6.2	26.5		**15.8**	6.6
Dec 07	13.9	21		7.1	8.8
Annual	*8.7*	*21*	*9*	*9.5*	*15*

Overall, monthly average concentrations measured at Ovens were elevated compared to the other sites, in particular during the summer and autumn. Winter and spring concentrations were low at Ovens compared to Manjimup. Each elevated monthly average seems to coincide with burning activities.

5.6.2 Ozone

The 1-hour and 4-hour NEPM of 100 ppb and 80 ppb of ozone was not exceeded at any sites (excluding bushfire events for Ovens) (Table 5-8).

Annual average concentrations of ozone were the same for Ovens and Manjimup, with the lowest annual average measured at Darwin. Maximum 1-hour ozone concentrations were similar between Ovens and Darwin, and Manjimup recorded the lowest 1-hour maximum ozone concentration.

Even though ozone levels seem to increase at Manjimup with the arrival of smoke plumes, the monthly ozone monthly averages are less affected than $PM_{2.5}$ averages. The ozone concentrations measured in May and November are similar to those measured in the previous and following months. The same is true for Ovens, although the 1-hour maximum ozone concentrations measured in April was the second highest and the highest if excluding bushfire season.

At Darwin, monthly ozone concentrations seem to follow the same trend as monthly $PM_{2.5}$ concentrations, with highest averages measured in June and September similar to that observed for $PM_{2.5}$ concentrations.

In comparison to urban air quality data (Table 5-7), annual averages measured at Ovens and Manjimup are lower than annual average measured at Sydney, but similar to ozone concentrations measured in Melbourne.

Table 5-7 Monthly and annual averages of ozone concentrations measured in Melbourne, Sydney, Manjimup and Ovens.

Month	Melbourne	Sydney	Manjimup	Ovens
Jan 07	21	35.5	15	32
Feb 07	16	39.5	16	28
Mar 07	18	34	18	20
Apr 07	14	32.5	18	19
May 07	12	26.5	20	13
Jun 07	8	21	20	12
Jul 07	13	24.5	22	13
Aug 07	13	29.5	24	16
Sep 07	18	35.5	24.5	18
Oct 07	20	39	22	21.5
Nov 07	22	40.5	21	24
Dec 07	24.5	35.5	16.5	23
Annual	*17*	*33*	*20*	*20*

5.6.3 BTEX

BTEX concentrations were very low both at Ovens and Manjimup, with levels much below NEPMs and also much lower than levels measured in urban areas.

5.6.4 Chemistry – Topographical location

In this study, the relationship between $PM_{2.5}$ and ozone constructions varied between locations and fire type. In summary:

- At Darwin, elevations in $PM_{2.5}$ were associated with elevated concentrations of ozone during the dry season.
- At Ovens, when bushfire smoke impacted the measurement site in December 2006 and January 2007, both ozone and $PM_{2.5}$ concentrations were extremely high and exceeded NEPMs. However short-term variations of $PM_{2.5}$ and ozone frequently were uncorrelated. This suggests a mixture of fresh and aged smoke plumes. In smoke of uniform age, ozone and $PM_{2.5}$ are expected to be directly correlated, positively in the case of aged plume, where there has been sufficient time for ozone formation and negatively in the case of fresh plumes due to

titration of ozone by NO. A lack of correlation indicates a mixture of fresh and aged smoke. Due to the severity of the fires over the summer, haze was trapped in the valley over extended periods of time, resulting in some photochemical formation of ozone. In May 2007, elevated ozone and $PM_{2.5}$ were associated with smoke from the prescribed burns, suggesting that either aged smoke plume originated from burns further away or pollutants were trapped in the valley and photochemical ozone formation occurred.

- Elevated ozone concentrations associated with increased $PM_{2.5}$ concentrations were observed at Manjimup in association with prescribed burning smoke, however, smoke from domestic woodheater emissions displayed low ozone concentrations in association with elevated $PM_{2.5}$ concentrations.

From the associations described here the age of smoke plumes effecting the different locations can be summarised as follows: Darwin bushfire smoke in the dry season was aged; Ovens bushfire smoke was a mixture of fresh and aged, mainly due to its topographical location; Manjimup prescribed burning smoke was aged; Manjimup domestic woodheater smoke was fresh.

The relationship of ozone concentrations with elevated particle concentrations in a smoke plume is an indication of the age of the plume. In the fire combustion process particles and nitric oxide, NO, are produced. In a fresh smoke plume or close to the source of the fire, the NO concentrations are sufficiently high to destroy (or titrate out) the ozone present. This process is described by the reaction 3 in the ozone production destruction reaction scheme below. As the plume ages, other photochemical processes take place within the plume, and from the volatile organic compounds present, organic peroxy radicals, RO_2, are formed. These RO_2 radicals oxidise NO to NO_2, competing with the ozone production process (reaction 3) and favouring the ozone production process (reaction 4). Thus in a traverse of an aged smoke plume, an increase in ozone concentration is found along with the increase in particle concentration.

$O + O_2 + M \rightarrow O_3 + M$[2] (1)

$NO_2 + h\nu$[3] $\rightarrow$ NO + O (2)

$O_3 + NO \rightarrow NO_2 + O_2$ (3)

$RO_2 + NO \rightarrow NO_2 + RO$ (4)

In other studies, modelling of ozone destruction and formation in smoke plumes during long range transport over the North Atlantic (Real et al., 2007) showed that large increases in O_3 were mainly due to PAN loadings in the plume.

In other work, elevated ozone concentrations measured in Atlanta, Georgia, associated with a smoke plume from prescribed burning in February 2008 (Lee et al., 2008) were associated with enhanced concentrations of isoprenoid (isoprene and monoterpenes) and other VOCs measured in the smoke plume. The enhanced ozone concentrations were associated with the oxidation of these compounds which also resulted in the presence of secondary organic aerosol oligomeric species in $PM_{2.5}$ samples. Greatly enhanced isoprenoid concentrations have been observed for a range of vegetation (Alessio et al., 2004). While the behaviour of biogenic emissions varies between different plant species, exposure to fire and the associated elevated temperatures has been found to induce bursts of monoterpenes from vegetation that do not normally emit VOC's resulting in temporarily elevated loads of monoterpenes.

2 M is any "body" with mass eg primarily nitrogen or oxygen molecules particles, trace gas molecules, and surfaces of large objects. M absorbs energy from the reaction as heat; necessary for combining O and O2 into O3.

3 hv is the ultraviolet portion of solar radiation

Table 5-8 Monthly and annual averages of hourly ozone concentrations in ppb, maximum 1-hour ozone levels in ppb and number of NEPM exceedances (exc.) measured at the 3 sites between December 2006 and December 2007.

		Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	*Annual*
Casuarina	**Ave**					8.2	9.2	17.9	14.1	15.4	19.3	15.2	18.1	15.6	***14.8***
	Max					20	35.6	**87.4**	32.8	40.9	41.4	43.2	43.2	45.7	***87.4***
	Exc.					0	0	0	0	0	0	0	0	0	***0***
Manjimup	**Ave**	16.7	15.2	16.4	18.0	18.3	20.2	20.0	22.3	23.9	24.5	22.1	20.6	16.5	***19.8***
	Max	37.8	45.9	49.9	**63.3**	40.5	34.4	31.9	32.3	37.5	42.7	43.9	45.1	39.4	***63.3***
	Exc.	0	0	0	0	0	0	0	0	0	0	0	0	0	***0***
Ovens	**Ave**	35.1	32.3	27.7	20.1	18.7	12.6	11.7	12.7	16.2	17.9	21.5	23.6	22.7	***19.8***
	Max	**155**	**89.1**	68.2	63.3	84.5	33.2	28.2	28.9	35.1	41.2	50.1	64.7	62.0	***89.1***
	Exc.	**13**	0	0	0	0	0	0	0	0	0	0	0	0	***0***
NEPM															***100***

Table 5-9 Monthly and annual averages of daily $PM_{2.5}$ concentrations in $\mu g/m^3$, maximum daily $PM_{2.5}$ levels in $\mu g/m^3$ and number of NEPM exceedances (exc.) measured at Casuarina for 2006 and 2007, and at Manjimup and Ovens between December 2006 and December 2007.

		Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	*Annual*
Casuarina 2006	**Ave**		3.8	2.3	2.4	2.1	5.2	7.6	11.3	13.8	13.8	11.5	10.2	5.7	***7.5***
	Max		7.1	5.6	5.8	3.6	11.6	13.2	25.5	27.5	29.0	**29.9**	18.1	12.9	***29.9***
	Exc.		0	0	0	0	0	0	**1**	**1**	**2**	**1**	0	0	***5***
Casuarina 2007	**Ave**		3.8	2.9	2.2	3.2	4.7	13.4	11.2	8.5	12.9				
	Max		8.1	5.4	4.3	5.9	8.9	30.6	47.7	14.4	17.3				
	Exc.		0	0	0	0	0	**2**	**1**	0	0				***3***
Manjimup	**Ave**	10.4	6.4	9.1	7.15	8.5	13.5	8.0	9.2	10.8	9.1	9.2	15.8	7.07	***9.5***
	Max	15.8	10.0	6.5	6.6	13.3	32.4	12.0	12.2	17.6	11.2	15.8	**86.8**	12.3	***86.8***
	Exc.	0	0	0	0	0	**1**	0	0	0	0	0	**3**	0	***4***
Ovens	**Ave**	148	46.5	18.7	17.9	25.3	14.5	6.14	6.37	6.61	7.61	12.1	6.55	8.82	***14.8***
	Max	**681**	155			113	33.0								
	Exc.	11*	2*			5	2								
NEPM															***25***

6 IMPACTS OF STUBBLE BURNING

6.1 Introduction

PM_{10} concentrations are measured at a network of stations in the Sydney, Illawarra and lower Hunter regions and in some rural centres - Albury, Wagga Wagga, Bathurst and Tamworth. Of the rural sites Wagga Wagga has the highest number of NEPM exceedence days and the highest average concentrations.

The major sources of anthropogenic particle emissions in the Sydney region are industry, the commercial and domestic sectors, and motor vehicles. Domestic solid-fuel heating makes up a significant proportion of commercial and domestic emissions, and diesel vehicles are the major contributor to motor vehicle emissions.

In rural areas, such as Wagga Wagga and Albury, likely significant sources include wildfires, prescribed burning, dust storms, agricultural emissions from stubble burning and wood heating. The relative contribution of each of these sources is unknown. CSIRO began sampling at DECC's Wagga Wagga monitoring site in December 2007. Results from this project will assist in reconciling the particle exceedences at Wagga Wagga, as additional instrumentation and analysis will determine the particle composition and indicate whether the particles are from dust or smoke sources.

6.2 Methodology

6.2.1 Sampling locations in the Riverina

Wagga Wagga is located about 400 km to the south-west of Sydney in the region called the south-west slopes and the Murrumbidgee River. There are no other significant urban centres in the region and no significant industrial sources of air pollution. The urban centre has a population of 50,634 (ABS, 2002). The Wagga Wagga air quality monitoring site is located on the corner of Morgan and Murray Streets. The location and topography of the region are presented in Figure 6-1.

Albury (and its Victorian counterpart Wodonga) are located some 450 km to the south-south-west of Sydney straddling the Murray River. Albury itself is bounded by elevated ground to the north, east and south of the Murray River on the southern boundary between the city and the higher ground. The populated area extends into the higher ground rather than to the west where there is less elevation. There are few significant industrial sources of air pollution in the region. The population of Albury is 42,458 (ABS, 2002). The Albury air quality-monitoring site is located in Jelbert Park, at the corner of Kaylock Road and Cambourne Street.

The DECC, NSW AQMS stations was supplemented for this study with a MicroVol fitted with 47mm stretched Teflon filters and a PM_{10} size-selective inlet to match the size fraction detected by the TEOM. The filters were changed weekly and analysed for soluble ion and laevoglucosan concentration.

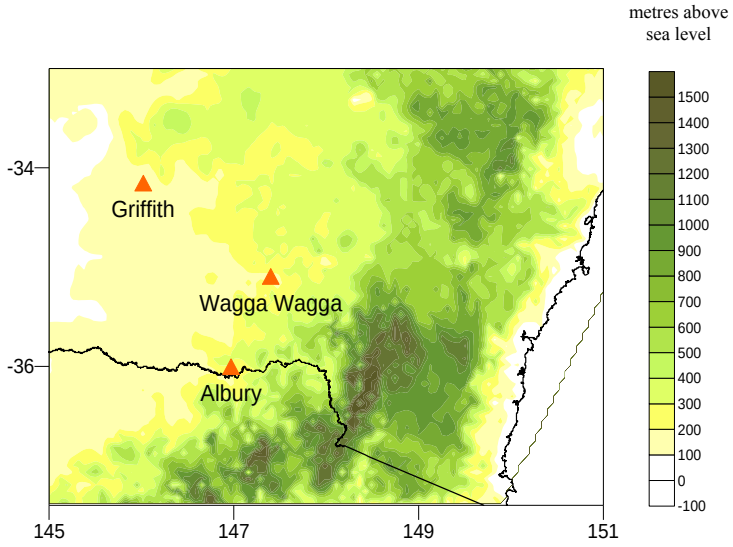


Figure 6-1: Topography of the region and location of the site

6.3 Results

6.3.1 Air Quality (PM_{10}) in NSW

PM_{10} concentrations are measured at a network of stations in the Sydney, Illawarra and lower Hunter regions. In addition, monitoring is required to determine whether the AAQ NEPM standards and goals have been met within populated areas of more

than 25 000. Initial campaign monitoring of PM_{10} has also been undertaken in the rural centres of Albury, Wagga Wagga, Bathurst and Tamworth.

Figure 6-2 shows the number of exceedences of the NEPM daily standard for PM_{10} from DECC's monitoring sites in NSW. Wagga Wagga had the highest number of exceedence days over the 5-year period from 2003 to 2007, recording a total of 148 exceedence days. Albury had the second highest total of 59 days, followed by the Sydney region with 44 exceedence days.

Wagga Wagga recorded the highest number of exceedences of the standard each year, with the exception of Albury in 2003, and did not meet the NEPM goal in any year. Albury met the goal, of no more than 5 exceedence days a year, in 2004 and 2005.

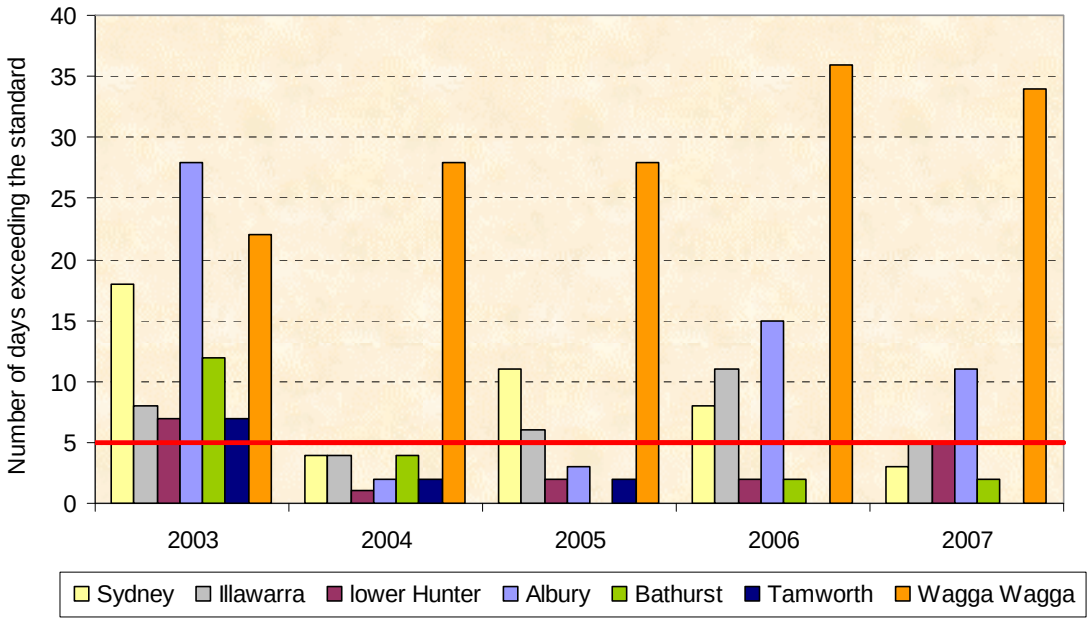


Figure 6-2: PM_{10} NEPM exceedence days in NSW (2003 to 2007)

Figure 6-3 provides a statistical summary of the daily PM_{10} concentrations at the four regional centres. Note that the PM_{10} concentrations have been plotted on a log scale.

Albury had the highest maximum PM_{10} concentration (921 $\mu g/m^3$), followed by Wagga Wagga (837 $\mu g/m^3$). Wagga Wagga had the highest average concentration (26.7 $\mu g/m^3$), then Albury (20.9), Tamworth (17.6) and Bathurst (16.9 $\mu g/m^3$).

While there are some extreme concentrations, 95% of the data at Wagga Wagga falls below 60 $\mu g/m^3$ and below 45 $\mu g/m^3$ at the remaining sites. The few extreme

concentrations impact on the average concentrations resulting in an average that is greater than the median at all sites, particularly at Albury.

The median concentrations at Albury (14.9 µg/m^3), Bathurst (14.0) and Tamworth (16.0) are similar with the highest median concentration at Wagga Wagga (21.4 µg/m^3).

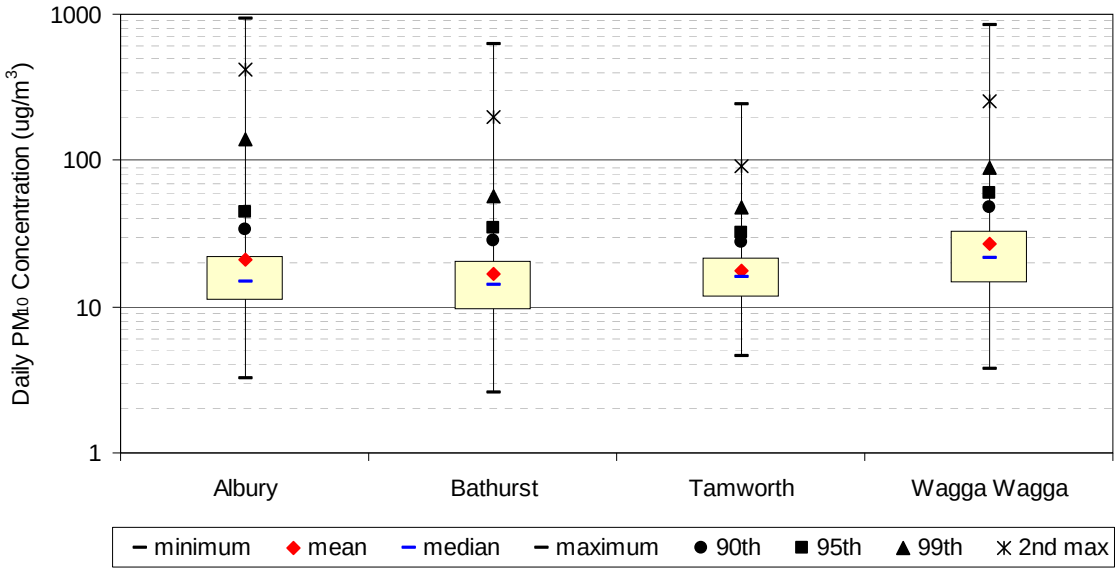


Figure 6-3: Statistical summary of daily PM_{10} data from all sites (2003-2007)

Figure 6-4 presents the distribution of daily PM_{10} concentrations at each site as a percentage, for example 10% of daily PM_{10} concentrations at Tamworth are greater than 27 µg/m^3 compared to 47 µg/m^3 at Wagga Wagga or alternatively 90% of all daily PM_{10} concentrations at these sites are less than these concentrations.

Wagga Wagga recorded the highest concentrations overall with Albury recording a small percentage of daily averages greater than 100 µg/m^3. Tamworth recorded the lowest concentrations overall.

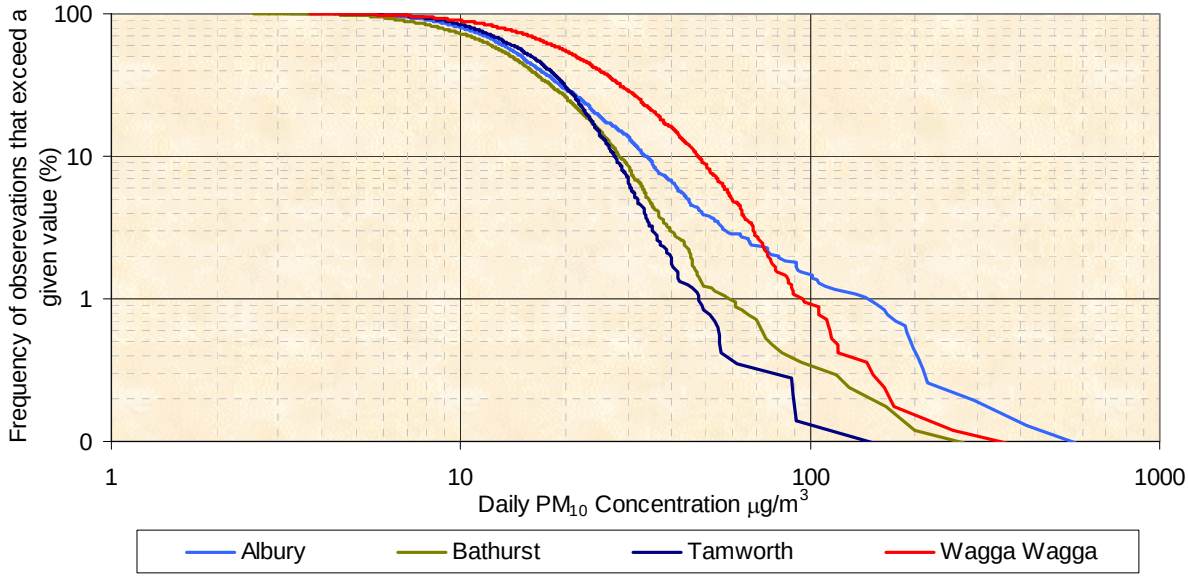


Figure 6-4: Relative frequency Daily PM_{10} (2003 to 2007)

6.3.2 PM_{10} in the Riverina

PM_{10} concentrations at Albury and Wagga Wagga have been analysed in further detail as these sites recorded the highest averages and highest number of exceedence days.

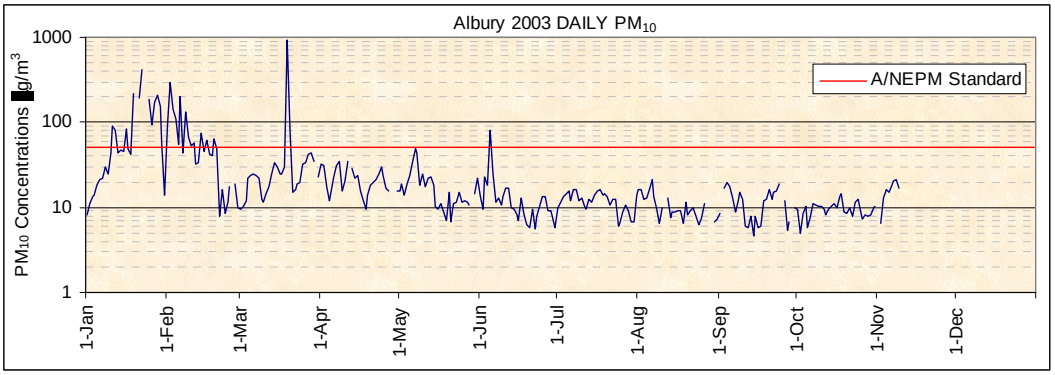
6.3.2.1 Daily PM_{10} concentrations

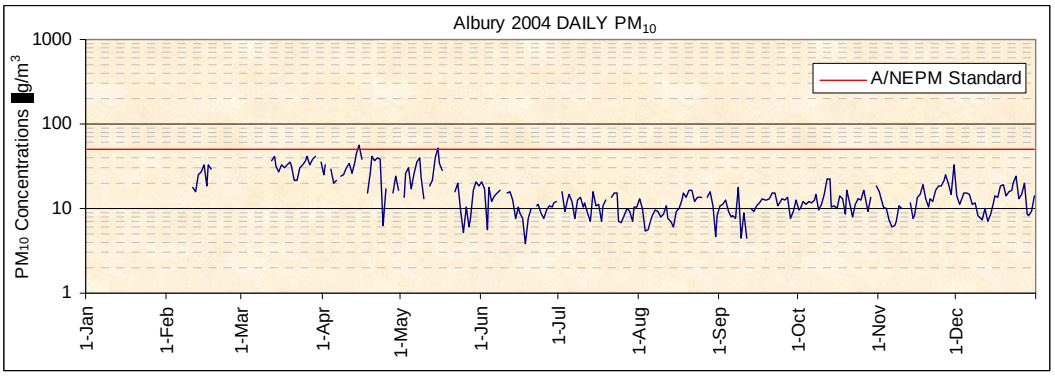
Time series analysis

Figure 6-5 presents daily PM_{10} concentrations for Albury and Wagga Wagga from 2003 to 2007. These figures show that the higher concentrations occur in the autumn and summer months.

The elevated concentrations in January 2003 at both Albury and Wagga Wagga were the result of major wildfires in Canberra and NSW. The impact of the Victorian wildfire emergency can be seen in data from December 2006 to February 2007 at each site.

The highest 24-hour concentration at Wagga Wagga and at Albury occurred on the 19th March 2003 and was the result of a major dust storm throughout NSW.


Albury 2003 DAILY PM10
PM10 Concentrations µg/m3
1000
100
10
1
A/NEPM Standard
1-Jan
1-Feb
1-Mar
1-Apr
1-May
1-Jun
1-Jul
1-Aug
1-Sep
1-Oct
1-Nov
1-Dec


Albury 2004 DAILY PM10
PM10 Concentrations µg/m3
1000
100
10
1
A/NEPM Standard
1-Jan
1-Feb
1-Mar
1-Apr
1-May
1-Jun
1-Jul
1-Aug
1-Sep
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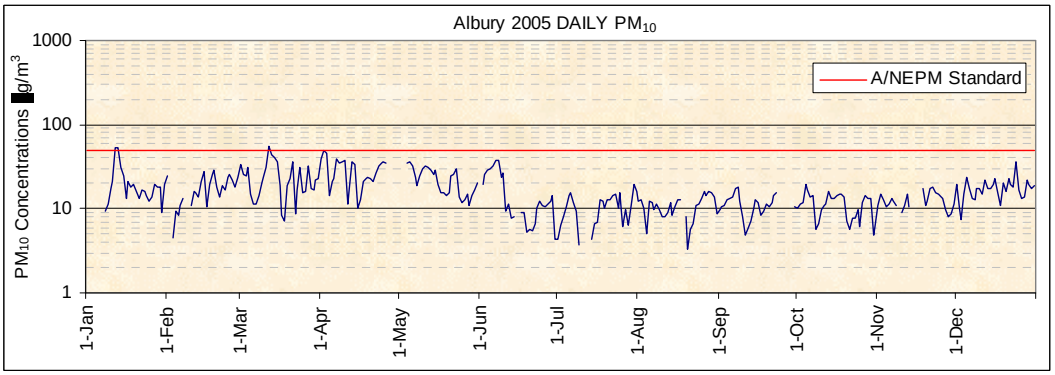
Albury 2005 DAILY PM10
PM10 Concentrations µg/m3
1000
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1
A/NEPM Standard
1-Jan
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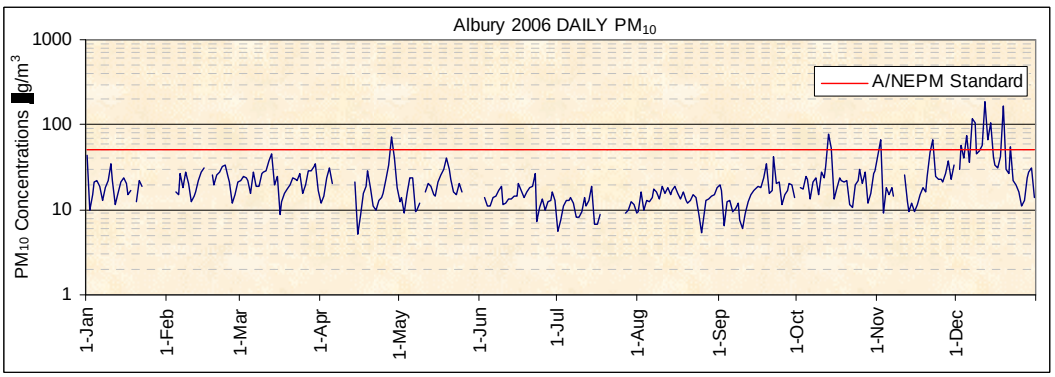
Albury 2006 DAILY PM10
PM10 Concentrations µg/m3
1000
100
10
1
A/NEPM Standard
1-Jan
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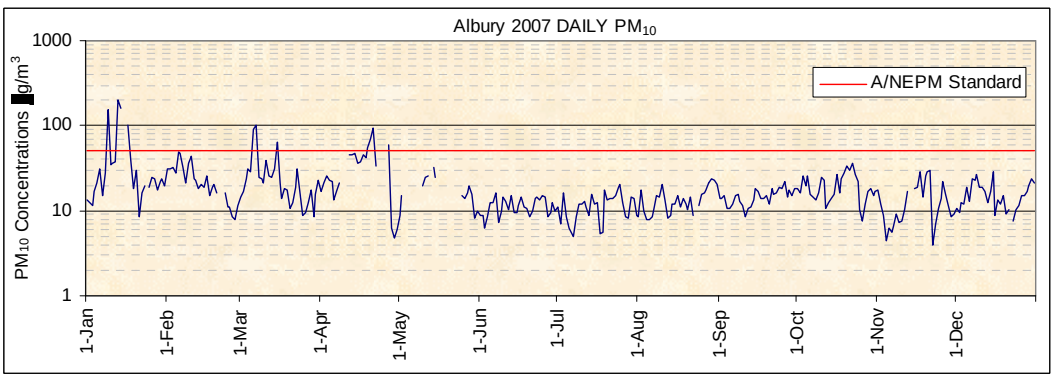

Albury 2007 DAILY PM10
PM10 Concentrations µg/m3
1000
100
10
1
A/NEPM Standard
1-Jan
1-Feb
1-Mar
1-Apr
1-May
1-Jun
1-Jul
1-Aug
1-Sep
1-Oct
1-Nov
1-Dec

Figure 6-5a: Time series of PM_{10} at Albury

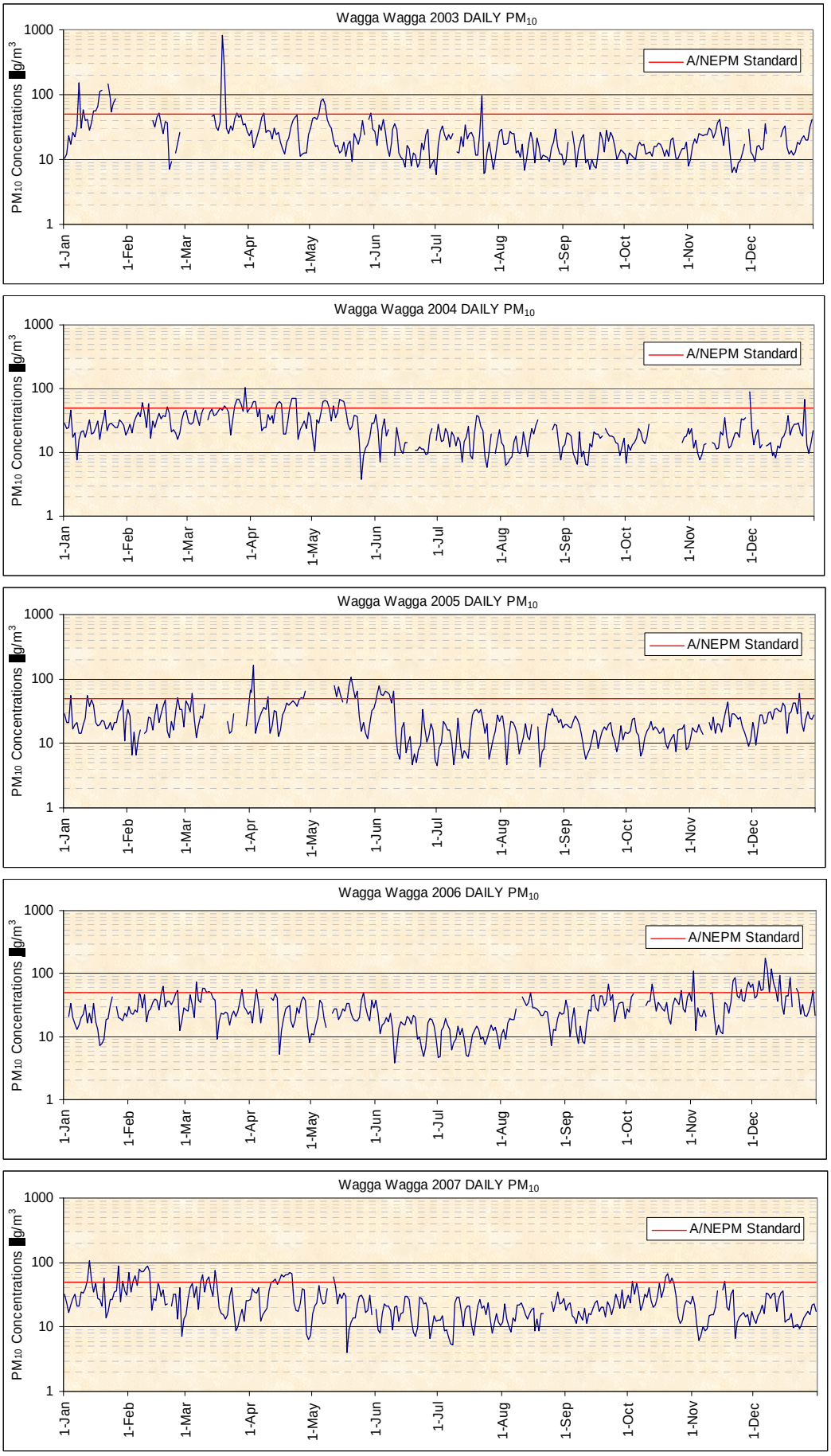


Figure 6-5b: Time series of PM_{10} at Wagga Wagga

Annual distribution of daily PM_{10} data

Figure 6-6 shows the distribution of daily PM_{10} at Albury and Wagga Wagga by year from 2003 to 2007. There was a greater difference in these measures between the years at Albury than those at Wagga Wagga. The highest mean and percentiles occurred in 2003 at Albury.

Wagga Wagga shows little difference between the years for the percentiles, median and averages.

As seen in the time series plots, the highest maximum 24-hour concentration recorded in 2003 at both sites was the result of a major dust storm.

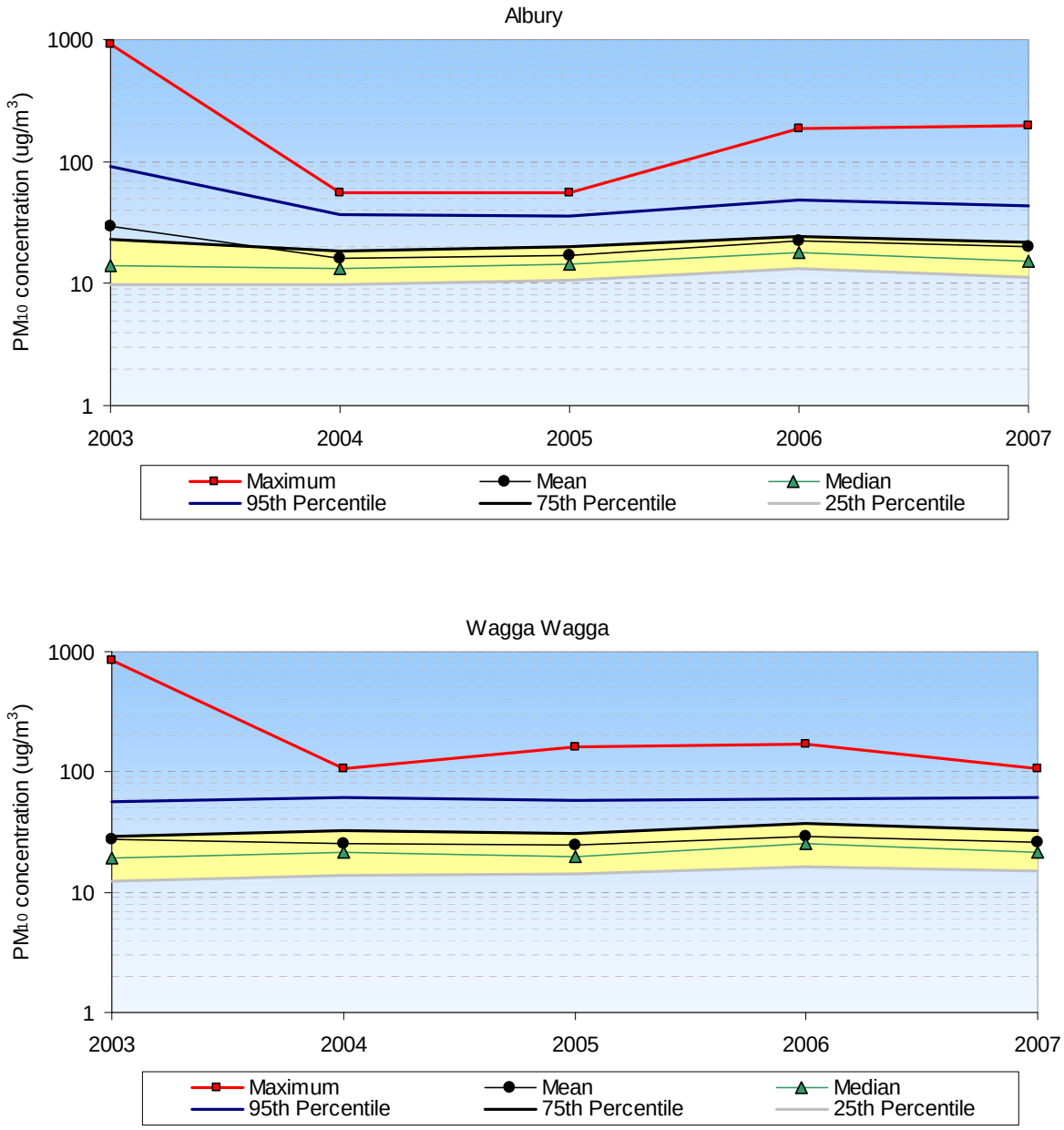


Figure 6-6: Annual Distribution of daily PM_{10} at Albury and Wagga Wagga (2003 to 2007)

Seasonal distribution of daily concentrations

Seasonal differences in the daily PM_{10} are shown in Figure 6-7. At both Albury and Wagga Wagga the lowest average, median and percentiles occurred in winter followed by spring. At Wagga Wagga the highest average and median PM_{10} concentrations occurred in autumn. At Albury the highest average occurred in summer and the highest median occurred in autumn.

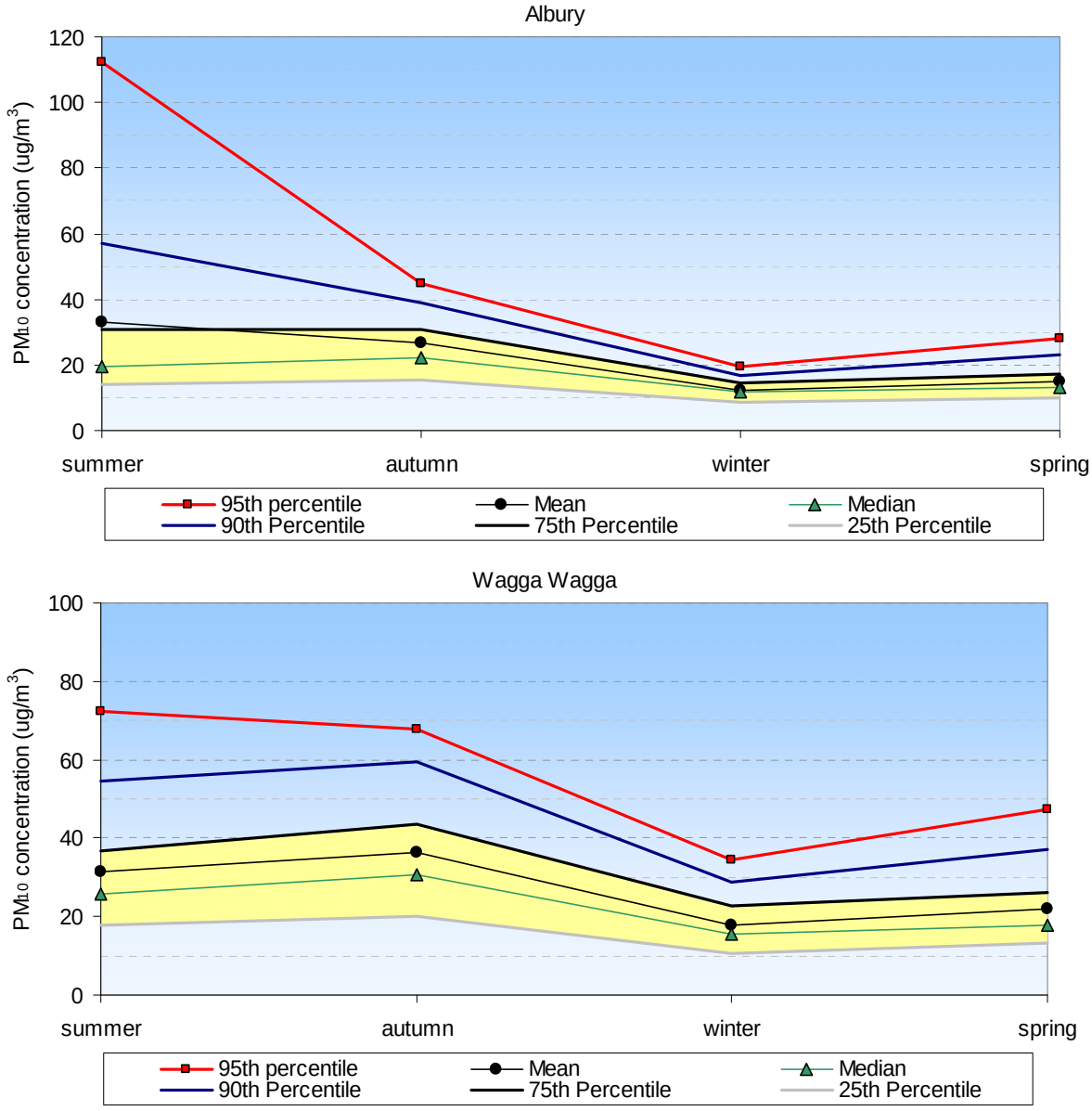


Figure 6-7: Seasonal distribution of PM_{10} 24-hour averages Albury and Wagga Wagga 2003 to 2007

Figure 6-8 presents the seasonal distribution of all daily PM_{10} concentrations from 2003 to 2007. The category greater than 50 $\mu g/m^3$ is equivalent to an exceedence of the NEPM standard.

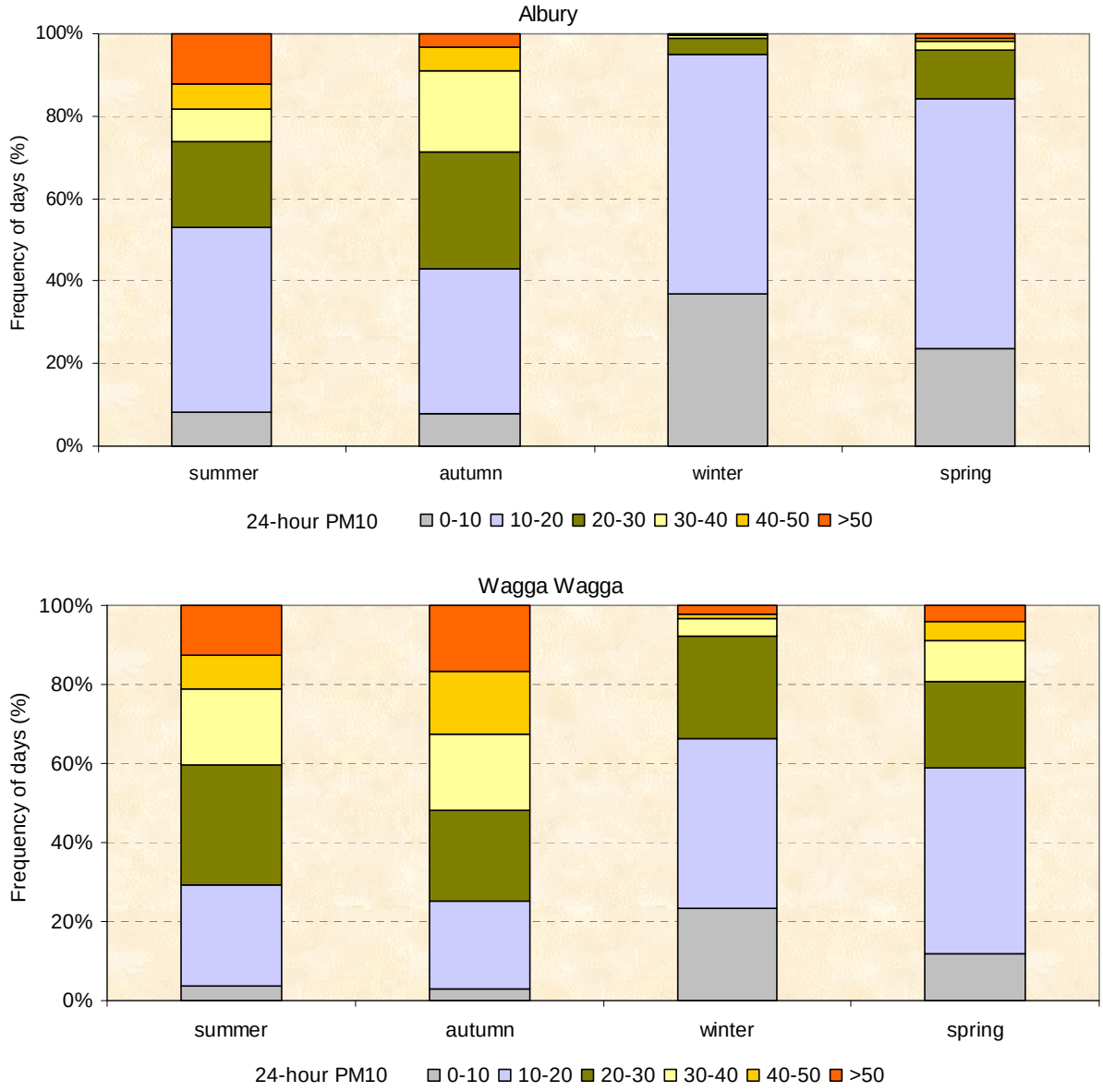


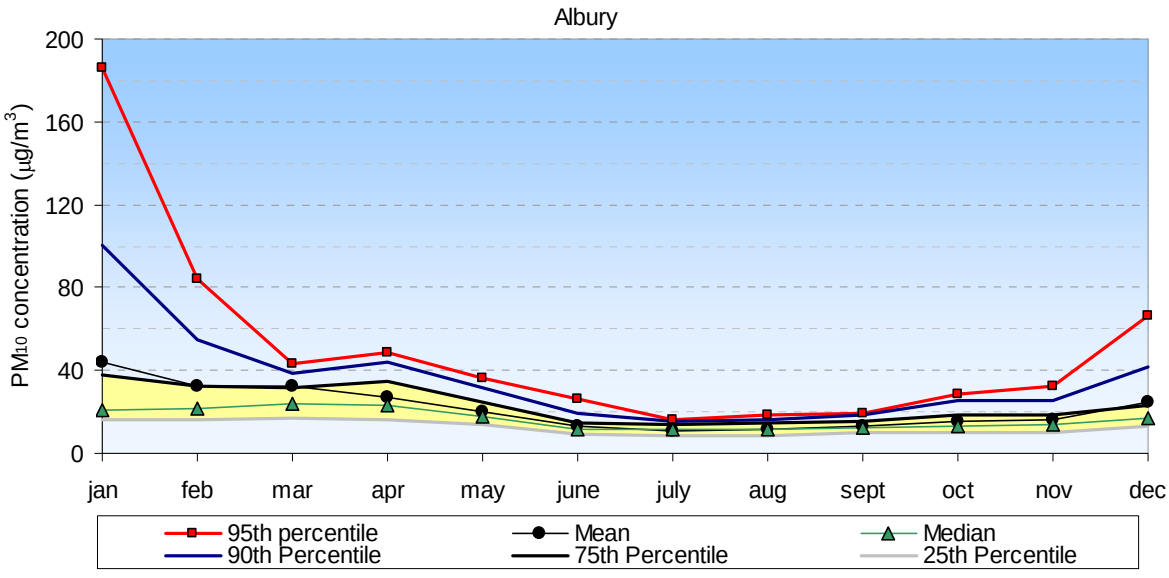
Figure 6-8: Daily PM_{10} distribution Albury and Wagga Wagga (2003 to 2007)

At Albury less than 1% of days experienced concentrations above the standard in winter (1 day) and spring (4 days). In addition the majority of days in winter and spring had daily concentrations less than or equal to 20 µg/m^3 (95% of days in winter and 84% in spring). The highest proportion of days above the standard occurred in summer with 12% of days (41 days) with concentrations above the standard followed by autumn with 3% of days (13 days). In summer and autumn the proportion of days with concentrations less than 20 µg/m^3 is much lower (53% in summer and 43% in autumn).

A similar pattern was found at Wagga Wagga with the distribution for winter and spring being similar with the lowest proportion of days with PM_{10} concentrations above the standard (2% in winter and 4% in spring). In winter and spring the majority of days had daily concentrations less than or equal to 20 µg/m^3 (66% of days in winter and 59% in spring). Autumn and summer had the highest proportion of days with concentrations above the standard with 17% in autumn (equivalent to 68 days) and 13% in summer (53 days). In summer and autumn the proportion of days with concentrations less than 20 µg/m^3 is much lower (39% in summer and 25% in autumn).

Monthly distribution of daily concentrations

The seasonal pattern is further explored in Figure 6-9 presenting daily PM_{10} data by month. At Albury the highest average, 90th and 95th percentile occurred in January and the highest medians occurred in March and April. At Wagga Wagga the highest average, median and 25th percentile occurred in March, while the highest 90th percentile occurred in May and the highest 95th percentile occurred in January.



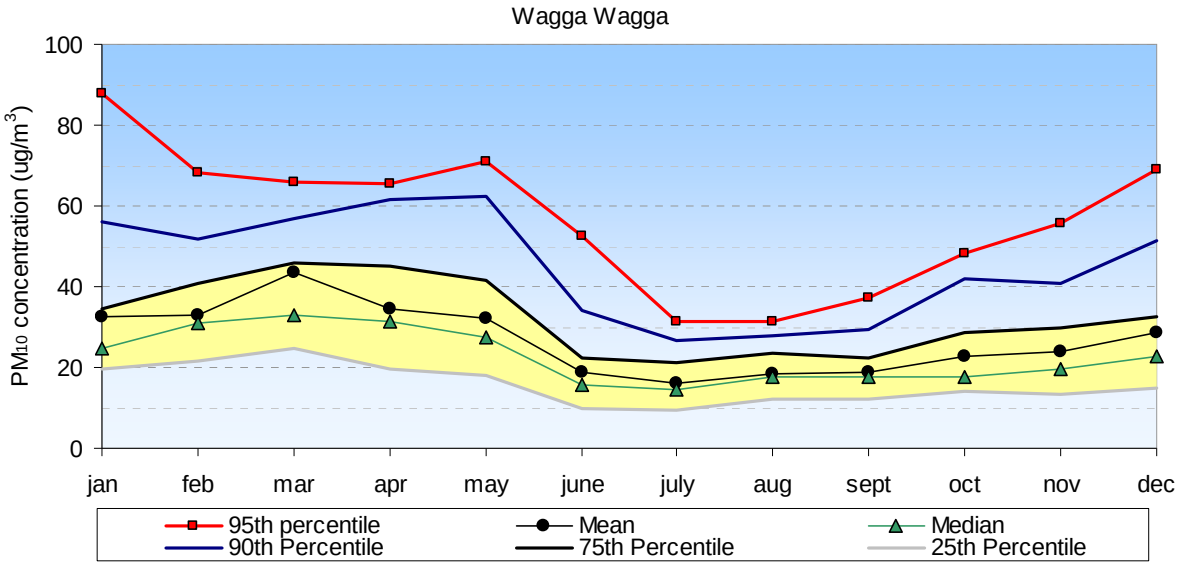


Figure 6-9: Monthly distribution of PM_{10} 24-hour averages at Albury and Wagga Wagga (2003 to 2007)

Figure 6-10 shows the number of exceedence days between 2003 and 2007 for each month. Of note are the very few exceedences in the cooler months of July and August. The highest number of exceedence days at Wagga Wagga were recorded in the autumn months of April and May. In Albury the highest number of exceedence days were recorded in January and February.

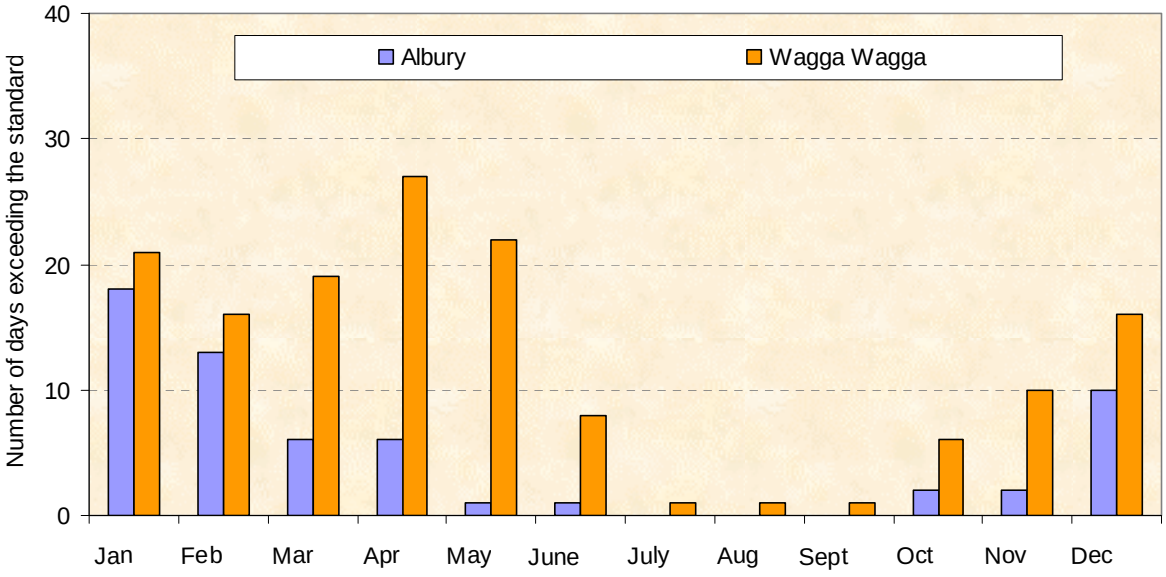


Figure 6-10: Monthly distribution of PM_{10} 24-hour exceedence days at Albury and Wagga Wagga (2003 to 2007)

6.3.2.2 Hourly PM_{10} concentration analysis

Diurnal distribution

Figure 6-11 shows the diurnal distribution of the hourly PM_{10} data at Albury and Wagga Wagga. At both sites the highest average hourly values occur in the evening around 8pm. There is also another peak earlier in the day at 8-9 am however these concentrations are lower than the evening peak.

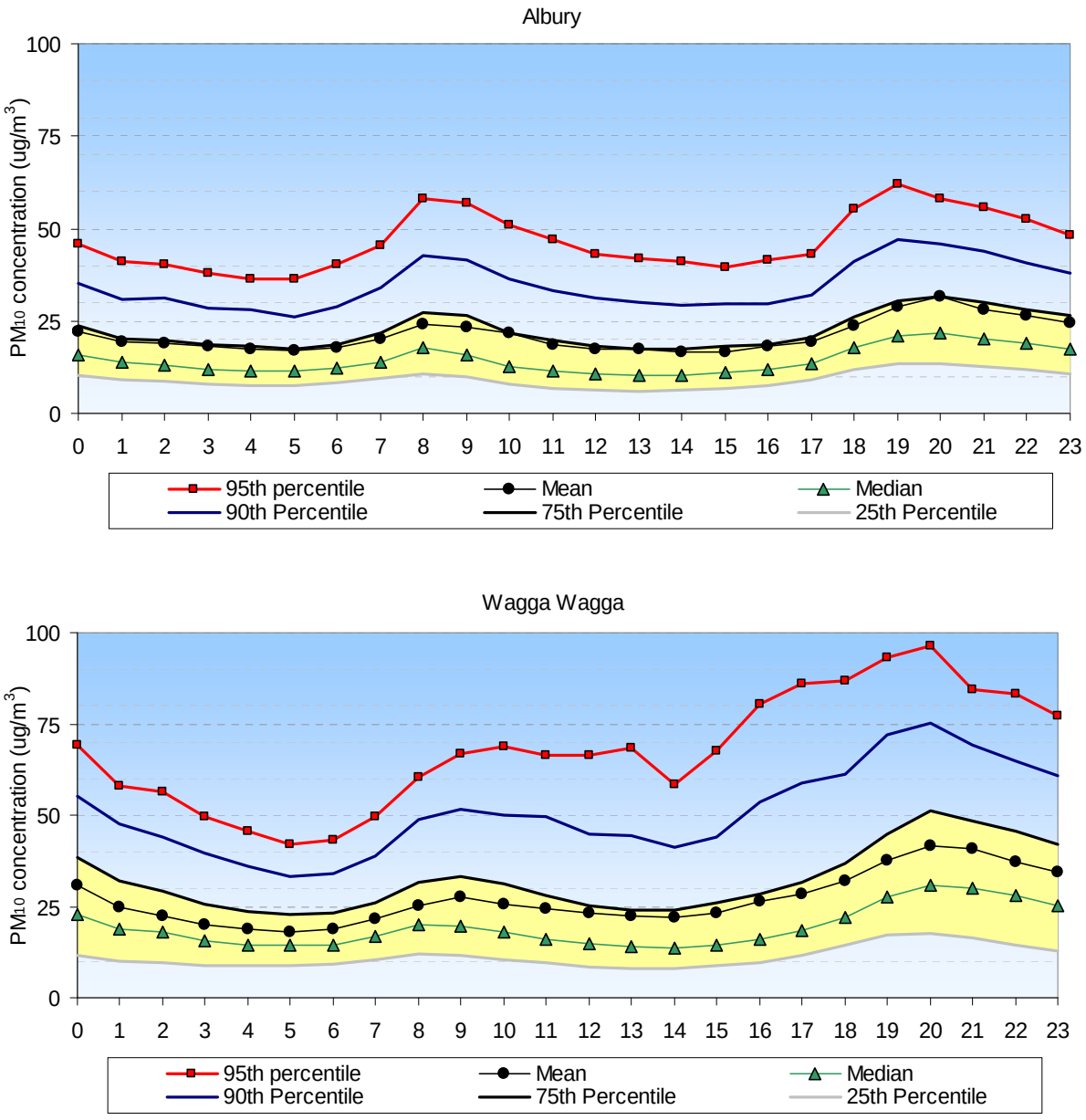


Figure 6-11: Diurnal distribution of hourly data at Albury and Wagga Wagga (2003 to 2007)

Figure 6-12 presents the total number of hours where PM_{10} concentrations exceed both 50 and 100 µg/m^3 for each hour of the day. There are more hours above these two concentrations at Wagga Wagga than at Albury. These higher hourly

concentrations occurred most frequently from 7pm to midnight with the highest number of hours above 50 μg/m^3 occurring at 7pm at Albury and 8pm at Wagga Wagga. The highest number of hours above 100 μg/m^3 occurred at 9am at Albury and 8pm at Wagga Wagga.

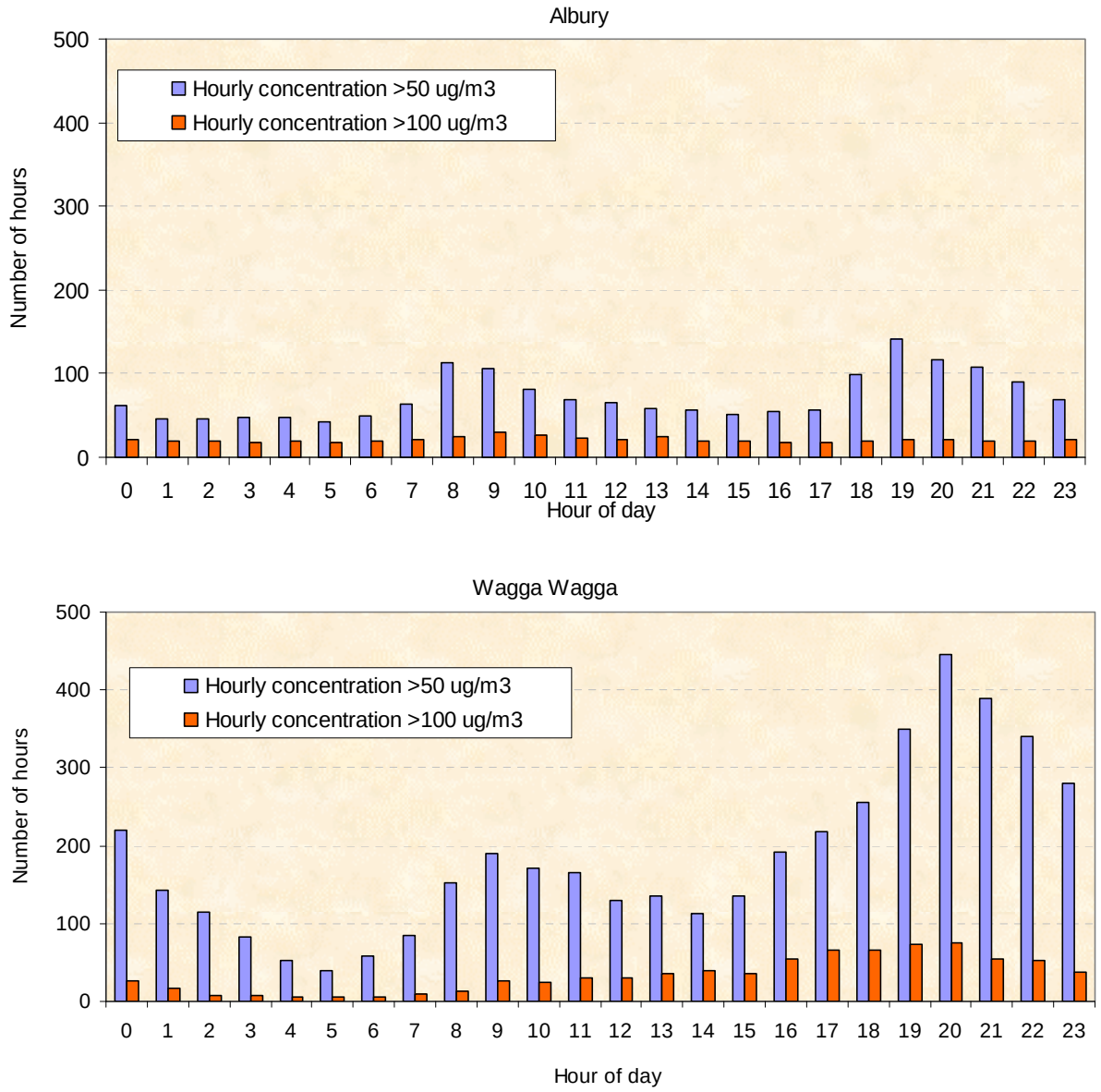
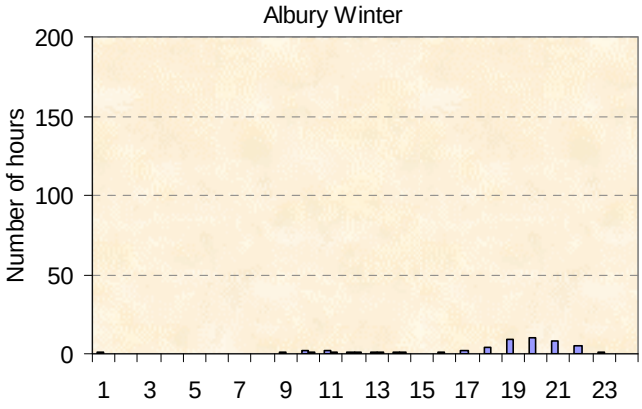
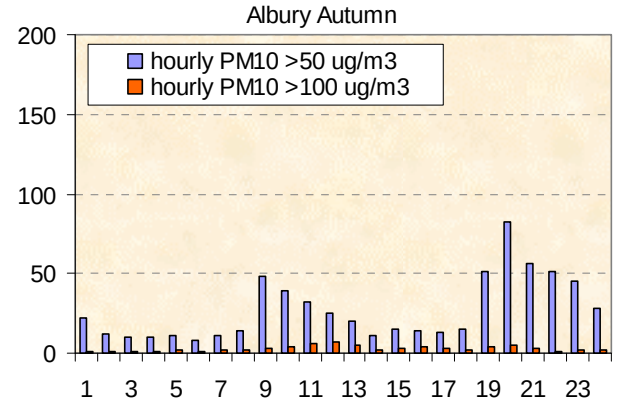
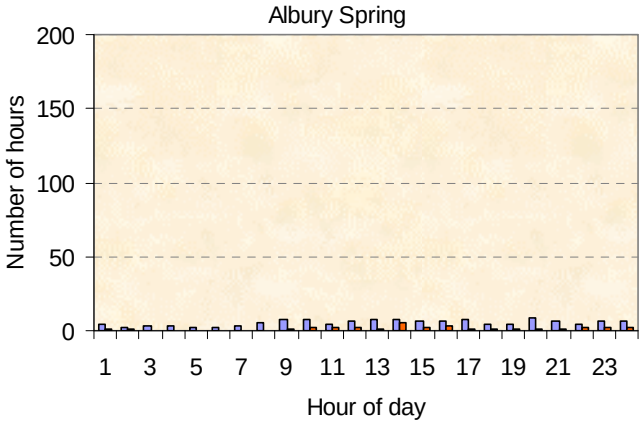


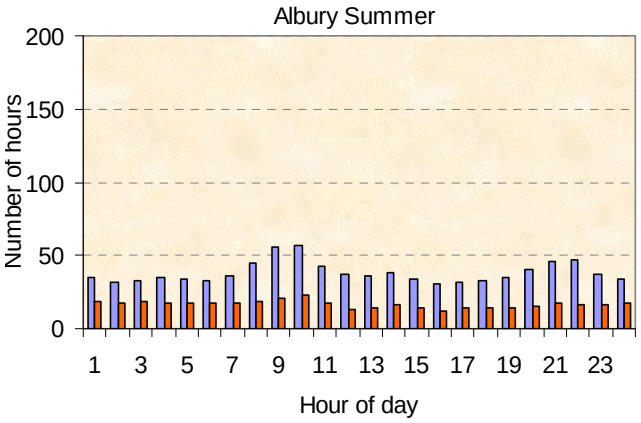
Figure 6-12: Number of hourly concentrations above 50 and 100 μg/m3 at Albury and Wagga Wagga (2003 to 2007)

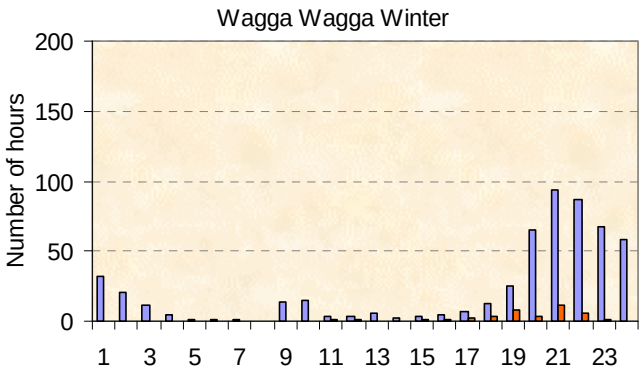
Figure 6-13 presents the distribution of these same data by season. It can be seen that there are more hours at these higher concentrations in autumn and summer for both Albury and Wagga Wagga. In autumn and winter at Wagga Wagga the higher concentrations occur later in the day, peaking around 8-10 pm. In autumn at Albury the higher concentrations occur at 8pm and another peak around 9am. In spring and particularly in summer there is a more even distribution across the day.

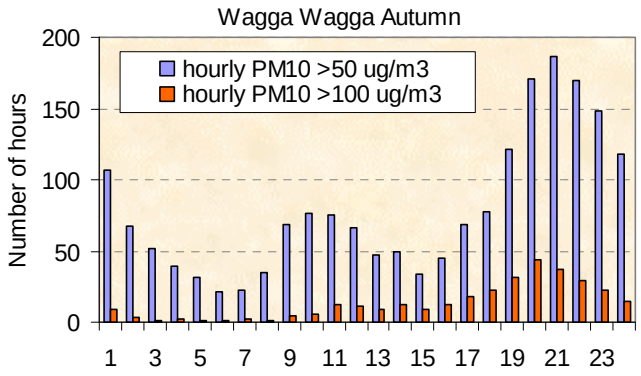

Albury Winter
Number of hours
200
150
100
50
0
1 3 5 7 9 11 13 15 17 19 21 23

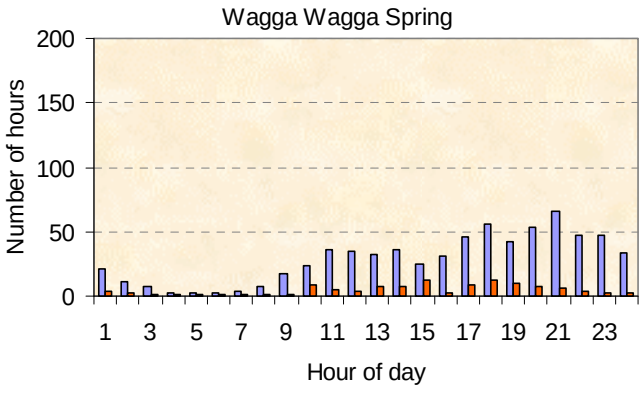

Albury Autumn
hourly PM10 >50 ug/m3
hourly PM10 >100 ug/m3
200
150
100
50
0
1 3 5 7 9 11 13 15 17 19 21 23


Albury Spring
Number of hours
200
150
100
50
0
1 3 5 7 9 11 13 15 17 19 21 23
Hour of day


Albury Summer
Number of hours
200
150
100
50
0
1 3 5 7 9 11 13 15 17 19 21 23
Hour of day


Wagga Wagga Winter
Number of hours
200
150
100
50
0
1 3 5 7 9 11 13 15 17 19 21 23


Wagga Wagga Autumn
hourly PM10 >50 ug/m3
hourly PM10 >100 ug/m3
Number of hours
200
150
100
50
0
1 3 5 7 9 11 13 15 17 19 21 23


Wagga Wagga Spring
Number of hours
200
150
100
50
0
1 3 5 7 9 11 13 15 17 19 21 23
Hour of day

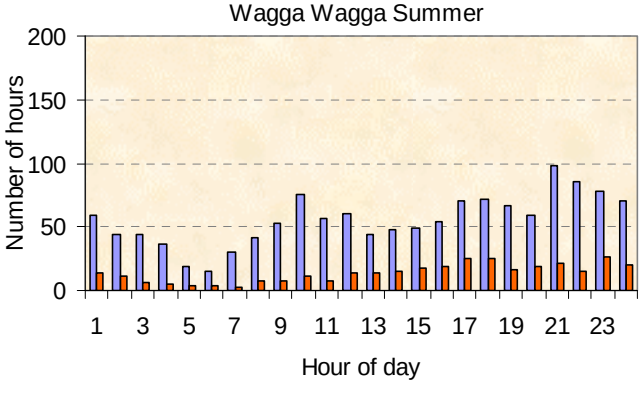
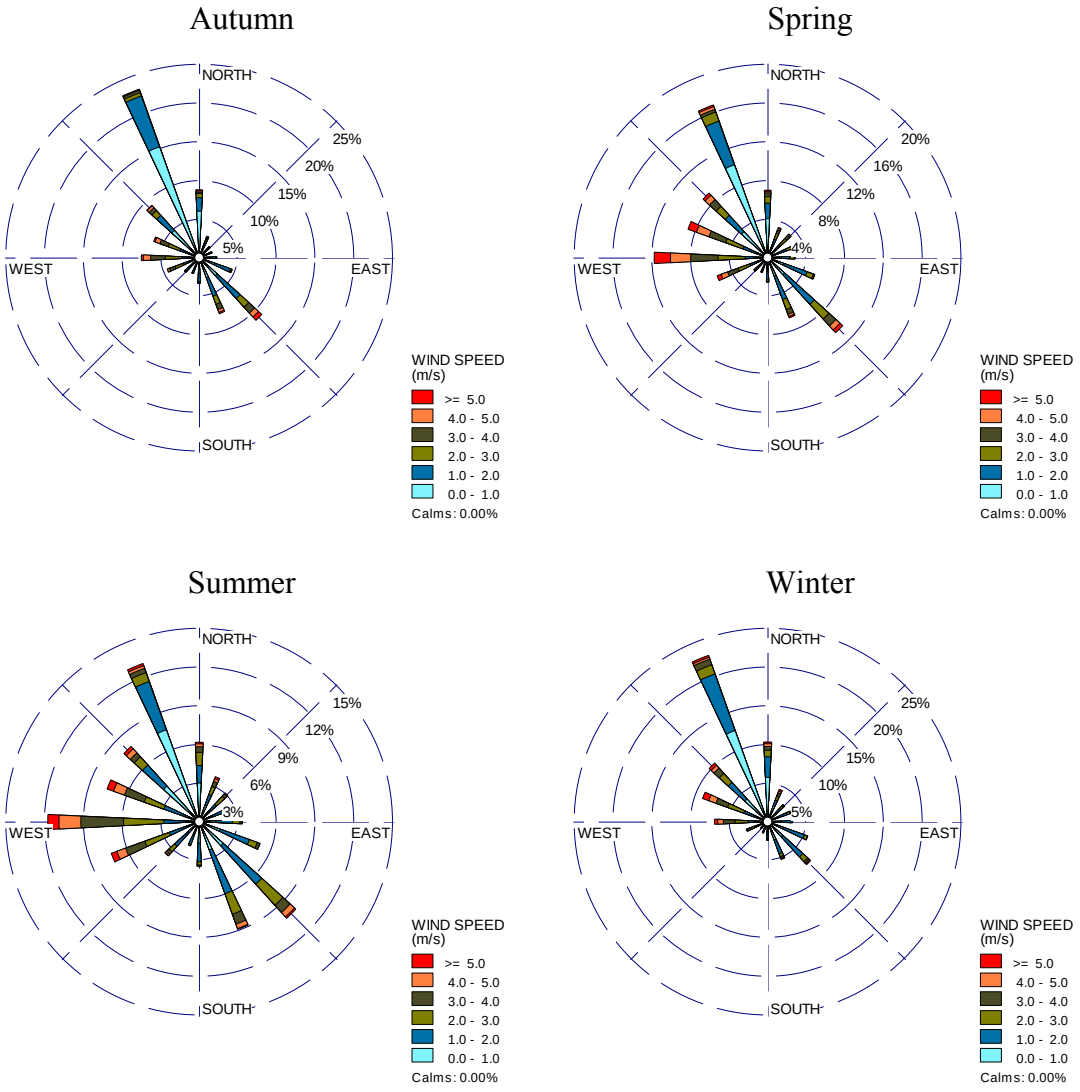

Wagga Wagga Summer
Number of hours
200
150
100
50
0
1 3 5 7 9 11 13 15 17 19 21 23
Hour of day

Figure 6-13: Seasonal distribution by hour of number of hourly concentrations above 50 and 100 μg/m3 at Albury and Wagga Wagga (2003 to 2007)

Wind roses

Figure 6-14 present wind roses for Albury and Wagga Wagga by season using hourly wind speed and direction data. At both locations higher wind speeds occur more frequently during the spring. In comparison the autumn months are associated with lower wind speeds. At Albury north-westerlies are the dominant wind direction particularly in autumn and winter.

Albury



Wagga Wagga

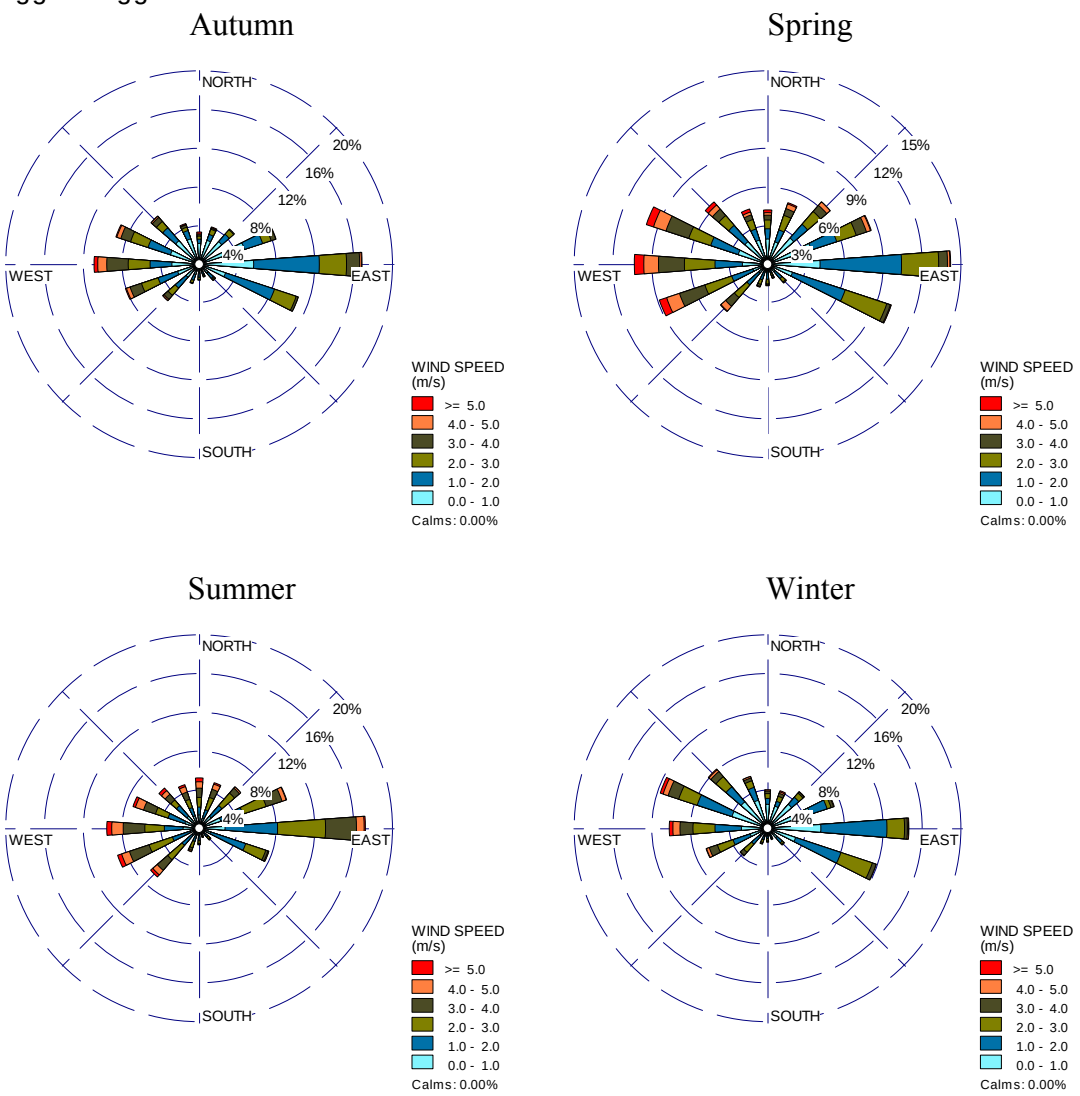


Figure 6-14: Wind roses by season Albury and Wagga Wagga (2003-2007)

Figure 6-15 shows the associated wind speed category for each hour with a PM_{10} concentration above 50 and above 100 $\mu g/m^3$. The majority of hours with these higher concentrations also had the lowest wind speed category. For example at Wagga Wagga almost 50% of hours with PM_{10} greater than 50 $\mu g/m^3$ occurred with a corresponding wind speed of less than 1 metre per second.

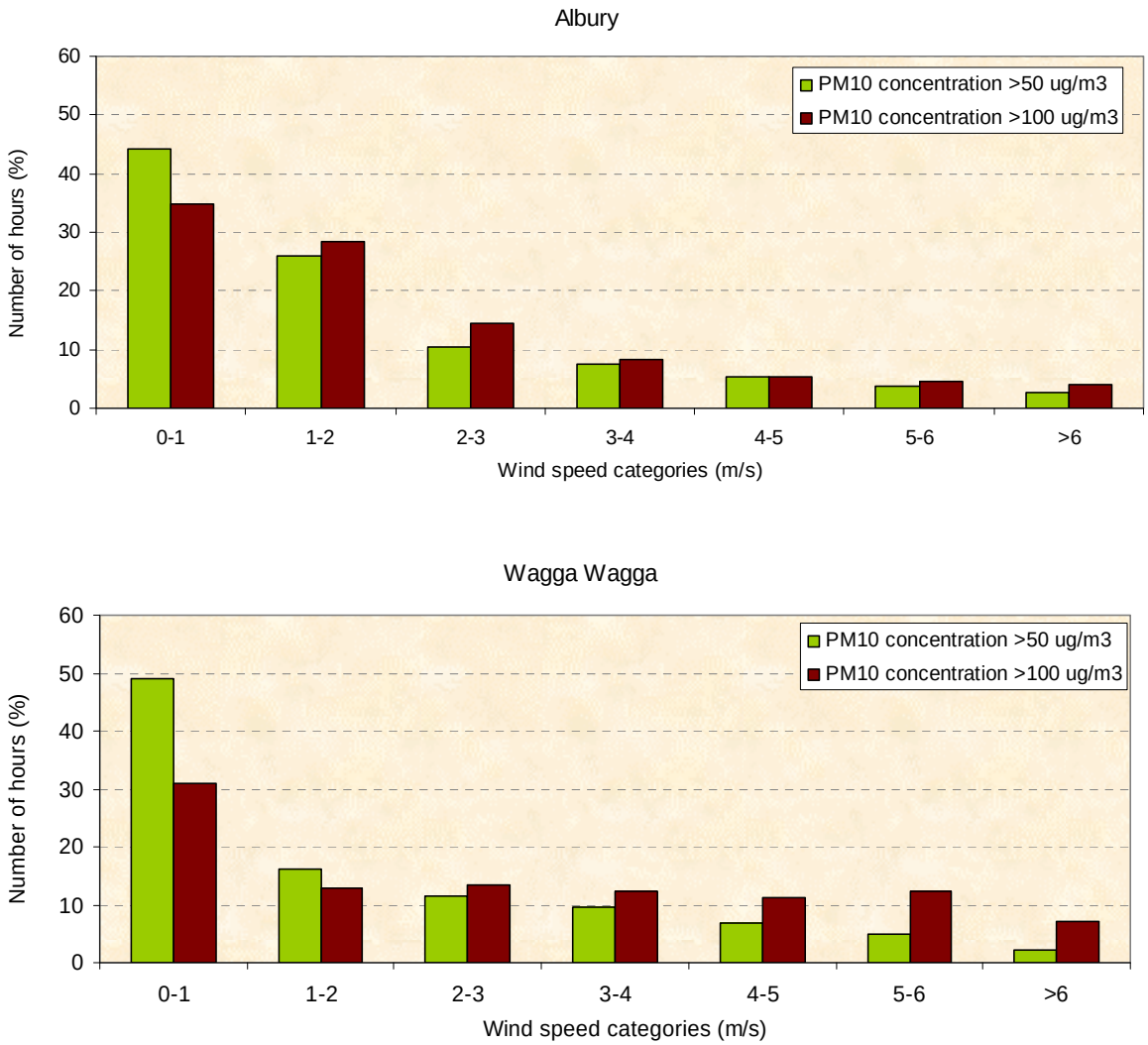


Figure 6-15: Percentage of hourly PM_{10} concentrations above 50 and 100 μg/m3 in each wind speed category

6.4 Discussion

6.4.1 Particle sources in the Riverina

In the Sydney region the major sources of anthropogenic particle emissions are industry, the commercial and domestic sectors, and motor vehicles. Domestic solid-fuel heating makes up a significant proportion of commercial and domestic emissions, and diesel vehicles are the major contributor to motor vehicle emissions.

In rural areas, such as Wagga Wagga and Albury, likely significant sources include wildfires, prescribed burning, dust storms, wood heating and agricultural emissions from stubble burning. While the relative contribution of each of these sources is

unknown, both the seasonal patterns found in the measured PM_{10} data as well as other external data sources can assist in determining the likely sources.

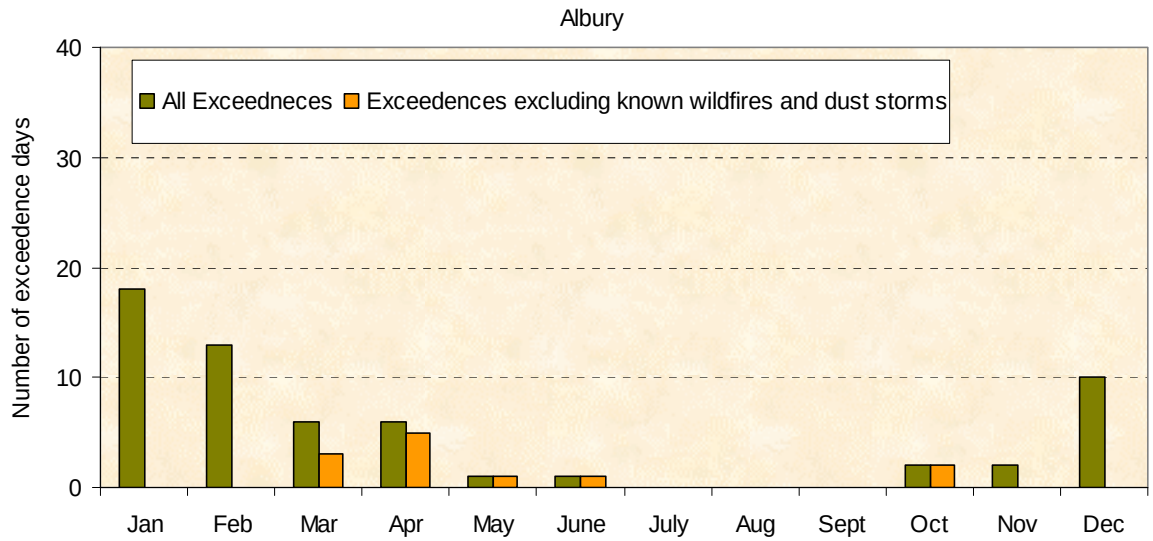
6.4.1.1 Dust storms

Days influenced by dust-storm events were identified using the Bureau of Meteorology's Monthly Weather Reviews (MWR) for New South Wales and Victoria (BoM, 2003-2007a and b). In addition records from the DECC's regional offices identified some further events.

The Bureau of Meteorology reports dust storm events based on reports from a network of observers, rather than instrumentation. It is therefore possible that overnight and other events may be missed.

Using this data record known dust storms were identified and removed from the data set (Figure 6-16), noting that it is possible that there are further days remaining.

While it is possible there are further events that have not been identified, regional dust storm events generally occur under higher wind speeds and the majority of hourly PM_{10} greater than 50 $\mu g/m^3$ were associated with low wind speeds (Figure 6-15).



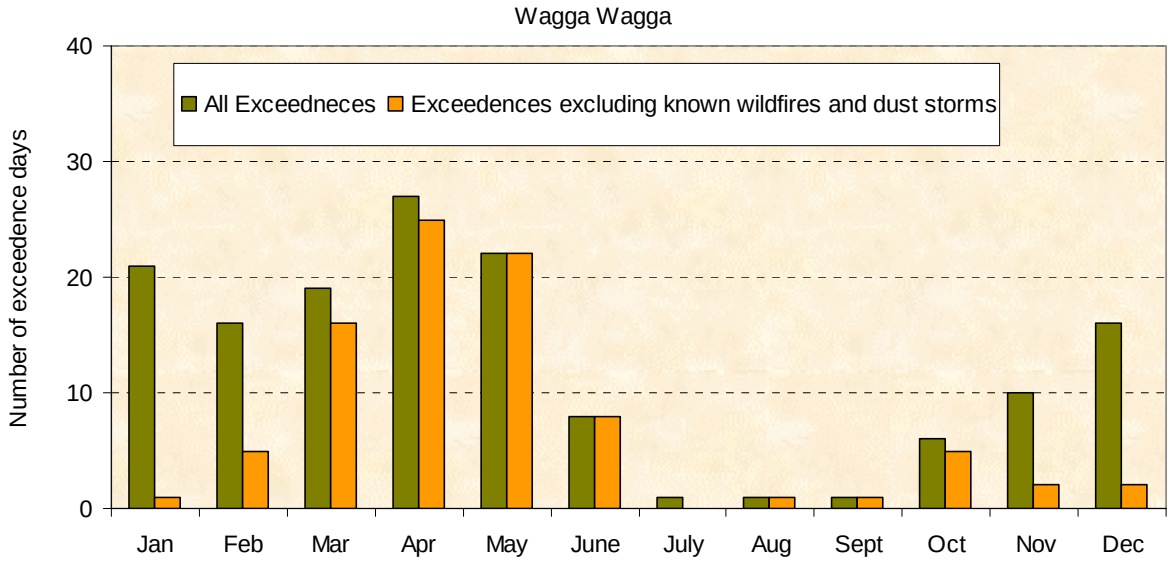


Figure 6-16: Exceedences excluding wildfires and dust storms

6.4.1.2 Wildfires

Days where wildfires were likely to have had an impact on air quality were identified using the Bureau of Meteorology's Monthly Weather Reviews (MWR) for New South Wales (BoM 2003-2007a and b) and information from fires in the national parks.

The following issues in identifying these types of events should be noted:

- Until recently wildfire and prescribed burning data records were spread across the state being held by regional fire district offices, councils and other agencies. A centralised database was recently developed which will assist in obtaining accurate and comprehensive data for the future. This data set is not yet available.
- While a fire may be reported as being under control, smouldering logs may continue to burn for days or weeks, hence a decision is required on the length of time after a fire is under control before it no longer impacts on air quality at a site.
- A fire may be reported close to a monitoring site, however if the wind is blowing smoke away from the site there may be no impact at the site.
- This type of assessment is subjective in terms of determining how close does a fire needs to be to a site before an impact on air quality is assumed.

Figure 6-16 presents the both the total data set and the remaining exceedence days after the known wildfires and dust storms were removed. At Wagga Wagga many of the summer exceedences were removed due to wildfire occurrence. At Albury all of the exceedences in January, February and December were found to have wildfires or dust storms likely to have impacted on the particle concentrations. Few events were identified during the autumn months, leaving the highest number of exceedences at both sites.

6.4.1.3 Prescribed burning

Prescribed burning traditionally occurs from autumn to spring, outside of the summer bushfire danger period. With Kosciuszko National Park to the east of Albury and Wagga Wagga, it is possible that smoke from larger prescribed burns in the park may impact on air quality under certain wind conditions. There were prescribed burns reported as being conducted in Kosciuszko National Park and the surrounding area during the period 2003 to 2007. It would be expected that an impact would be seen under predominantly easterly flow. Based on wind directions it is possible that these burns had an impact on PM_{10} on some of the days only.

6.4.1.4 Agricultural sources

Figure 6-17 shows the land use for the Riverina region. Near Wagga Wagga the land use is predominantly dryland agriculture, particularly to the northwest and southwest. Near Griffith there is a higher proportion of irrigated agriculture, which is mainly rice.

Wheat and rice are summer crops that are harvested from February to June. After the crops are harvested, farmers burn the remaining part of the plant above the ground (stubble). Burning of wheat stubble has been popular with farmers for controlling diseases or weeds and making planting easier.

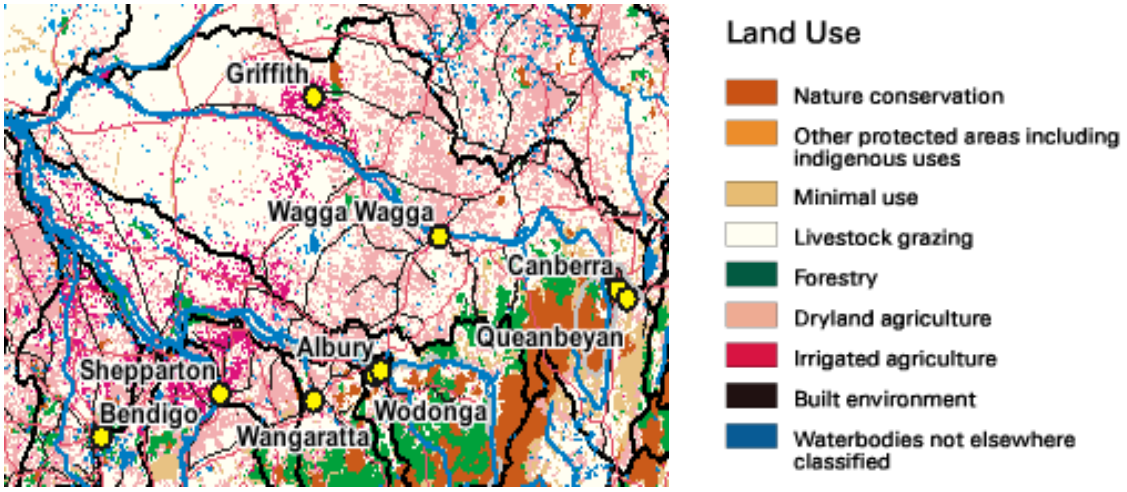


Figure 6-17: Land use in the Riverina (Source: NLWRA 2001)

Burning permits

Landowners are required to obtain a permit from the Rural Fire Service to burn during the period 1 November to 31 March each year (during the bushfire danger period). The permit is issued for a one-week period and the permit holder is required to contact the Rural Fire Service (RFS) on the day they plan to burn within that period.

This data set has the following limitations:

- stubble burns completed after the end of March do not require permits and are therefore not recorded
- stubble burning permits do not contain the size of the burn
- for privacy reasons the location of the burn is not available

Hotspot analysis

In lieu of these records, satellite data can be used to detect hotspots on particular days and at certain locations. This data set is limited by the inability to determine both the size and type of fire (ie it can be the result of a wildfires, prescribed burn or an agricultural burn).

Figure 6-18 shows the location of all hotspots identified using MODIS satellite data for the period 2003 to 2007 by season. Hotspots occurring in the national park to the east were excluded as they were likely due to wildfires or prescribed burning. There is a clear seasonal pattern with nearly 20 times more hotspots identified in the autumn months than the summer months and 25 times more than in spring. In autumn the

hotspots are generally to the west of Wagga Wagga. The highest number of hotspots identified occurred in April then May. The majority of hotspots identified in the winter months, shown in Figure 6-18, occurred in June (Figure 6-19).

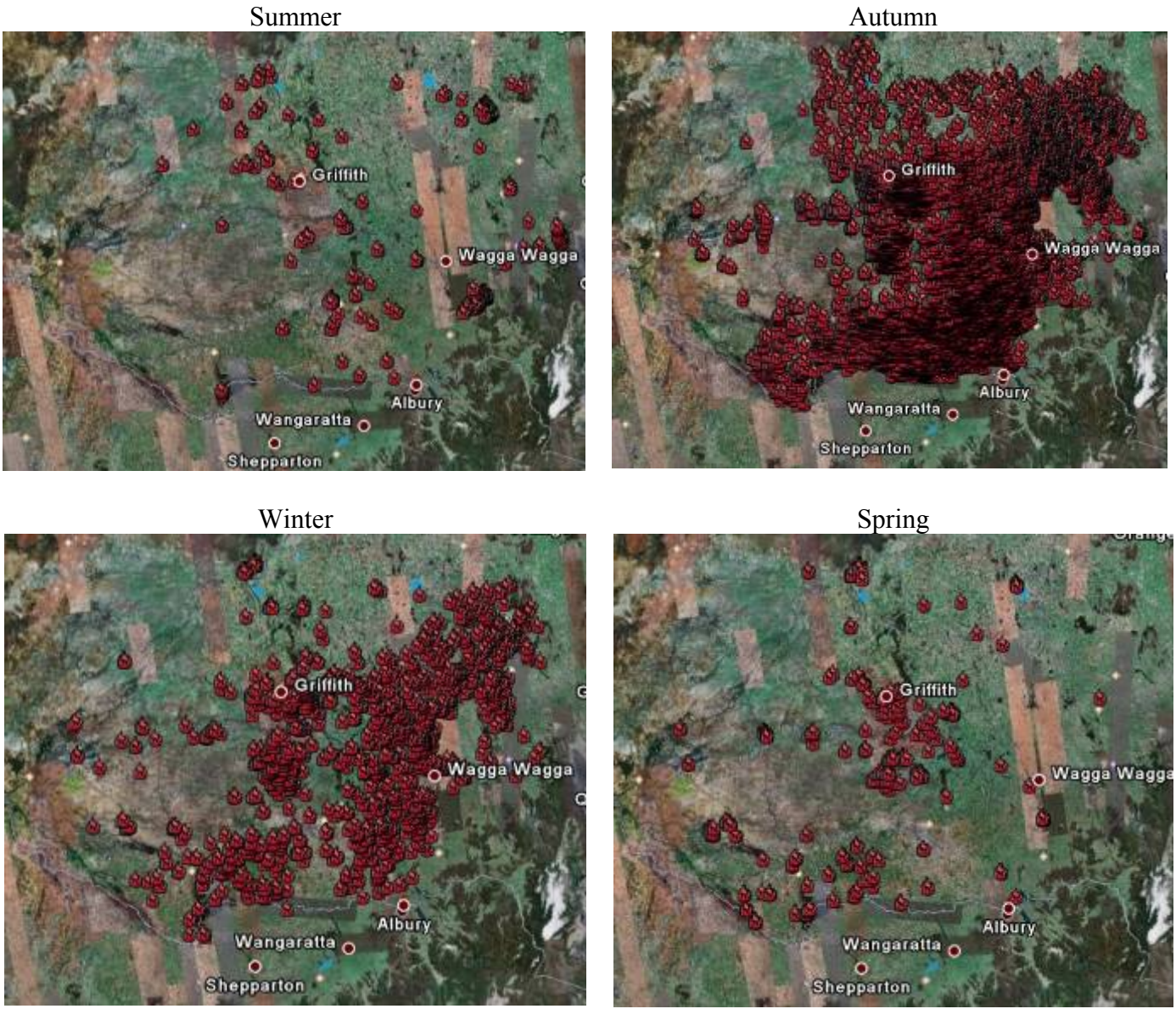


Figure 6-18: Hotspots identified from MODIS satellite data by season (2003 to 2007) (Source: MODIS satellite data)

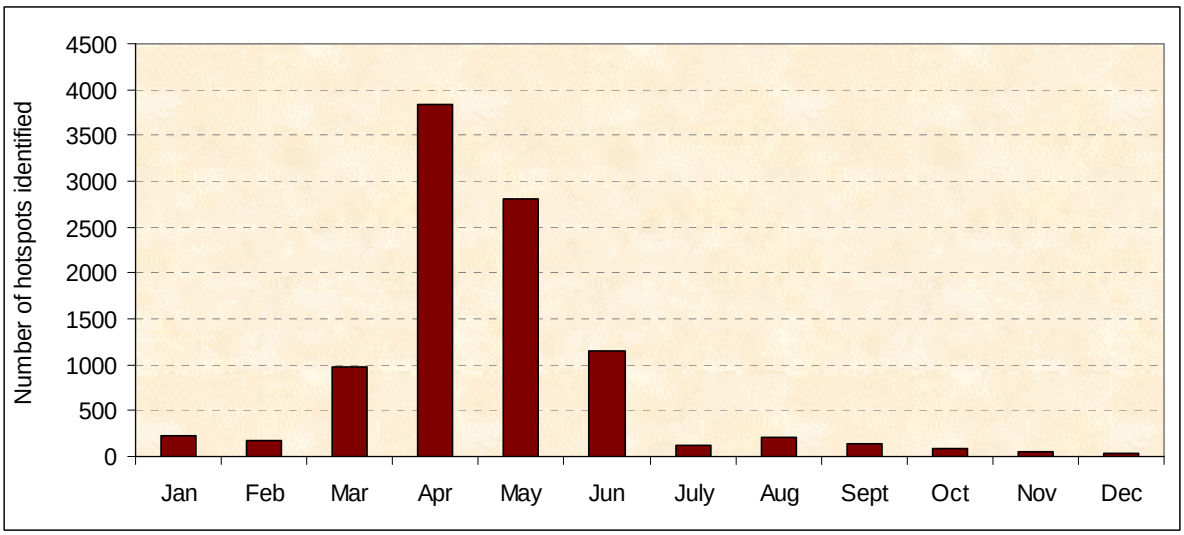


Figure 6-19: Number of hotspots identified from MODIS satellite data by month (2003 to 2007)

Pollution roses of hourly PM_{10} concentrations at Wagga Wagga and Albury plotted with the corresponding hourly wind direction are presented in Figure 6-20 for a range of hours (0-5am, 6-11am, 12-5pm, 6-11pm). The autumn months only are presented as these were the months with the higher PM_{10} concentrations particularly at Wagga Wagga and the months with the highest number of hotspots identified. The pollution roses for summer, winter and spring are presented in Appendix 1.

Albury

During autumn at Albury the predominant wind direction overnight (18 to 23 and 0 to 5 roses) is from the NNW. This is consistent with drainage flows from the elevated topography in that direction. Winds during the period hour 12 to 17 are variable.

Wagga Wagga

Across all seasons the morning pollution roses (0-5 and 6-11am) show a dominant easterly and east south easterly component as a result of drainage flows from the elevated topography to the east of Wagga Wagga. There are few southerly winds across all months and all times of the day.

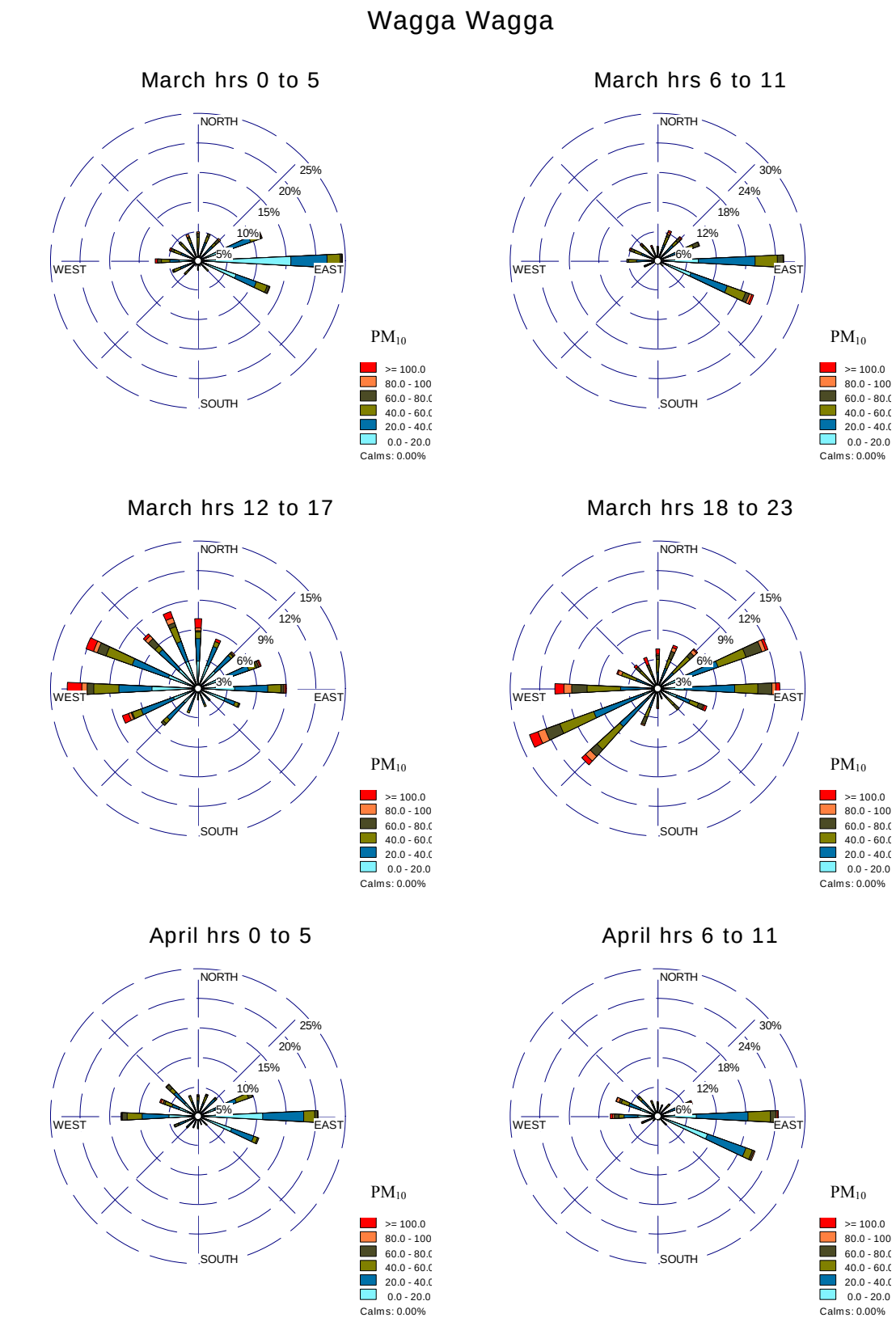
In the autumn months the predominant wind direction in the afternoon is westerly and in the evenings from the west and the east. The higher concentrations are associated with these afternoon and evening wind directions.

This is consistent with the results of the autumn hotspot analysis in Figure 6-18 showing several hotspots to the west of the site.

In summer the afternoon roses (Appendix 1) from hours 12-17 are dominated by westerly and north-westerly flows. Later in the day the predominant wind directions are south-westerly and north-easterly. The higher hourly PM_{10} concentrations are also associated with winds from these directions for each of the afternoon hour ranges.

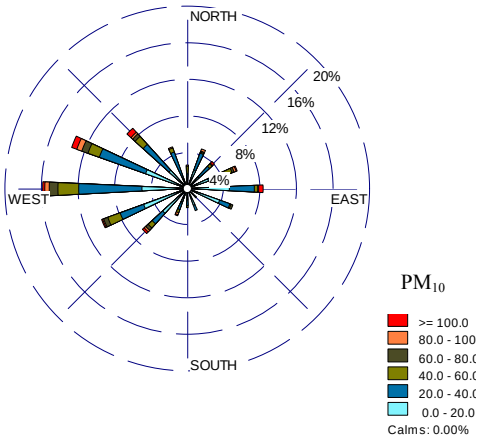
Concentrations are lower in winter than other seasons and the wind direction is fairly evenly spread across easterly and westerly sectors particularly in the afternoon. In spring the higher concentrations are generally associated with westerly winds.

Figure 6-20: Pollution roses for Wagga Wagga and Albury (autumn months only)

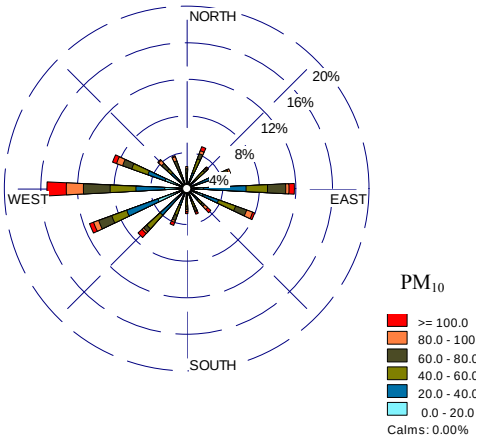


Wagga Wagga

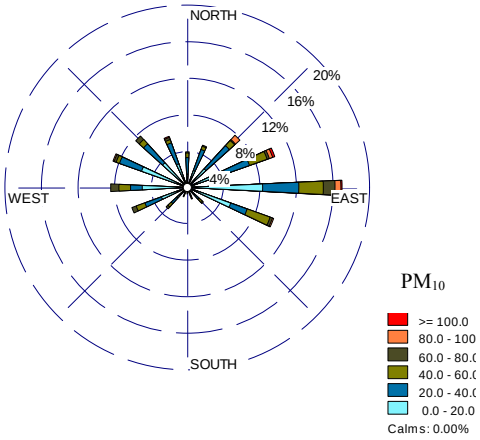
April hrs 12 to 17



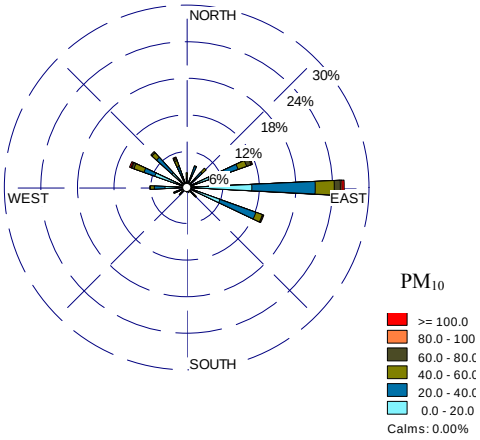
April hrs 18 to 23



May hrs 0 to 5



May hrs 6 to 11



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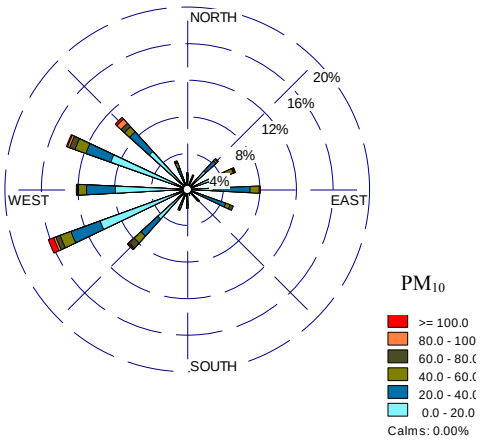

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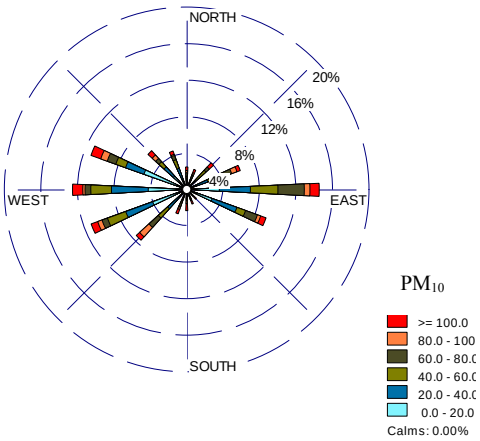
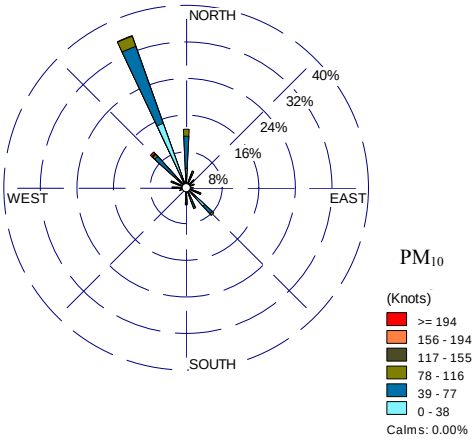
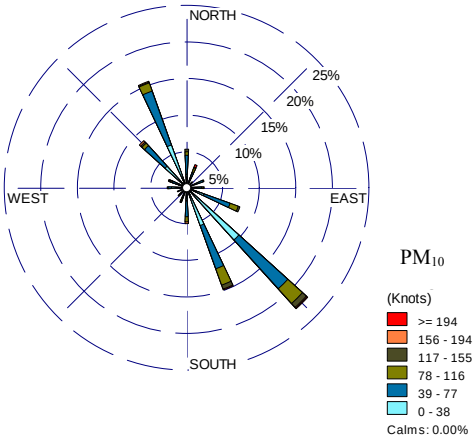


Albury

March hrs 0 to 5



March hrs 6 to 11



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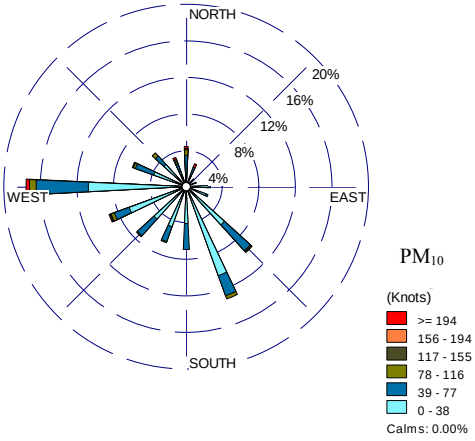

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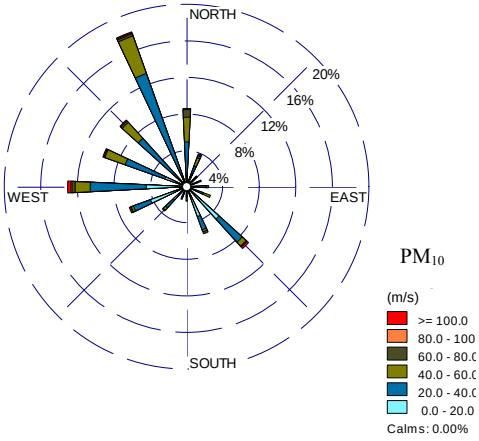

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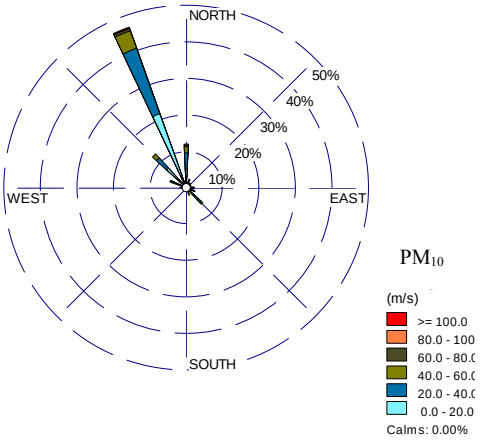

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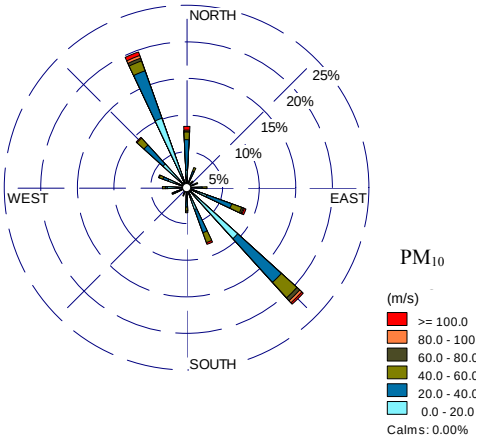
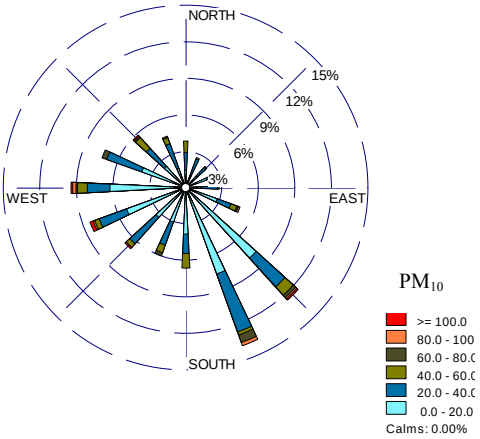
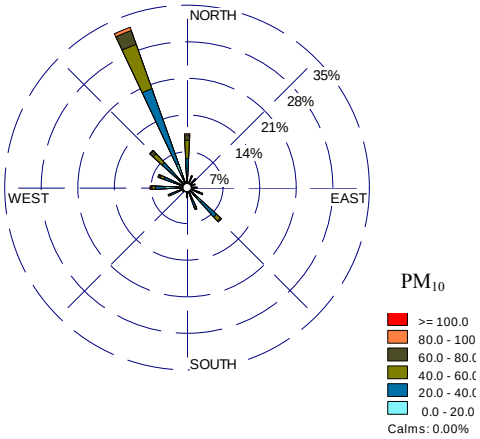


Albury

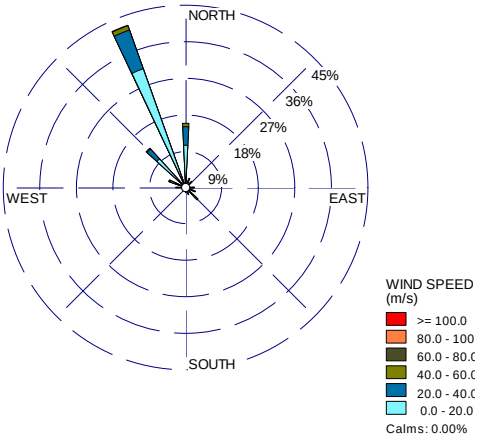
April hrs 12 to 17



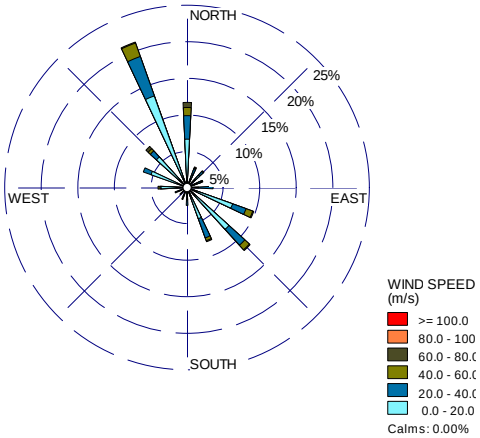
April hrs 18 to 23



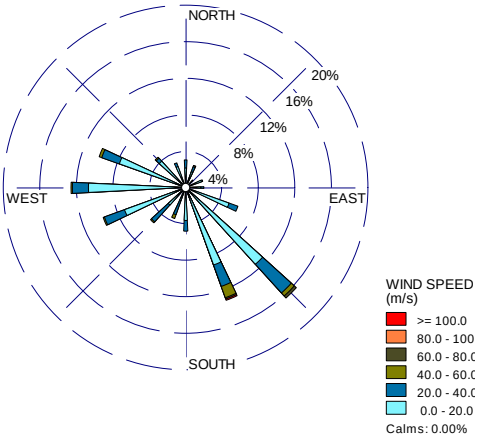
May hrs 0 to 5



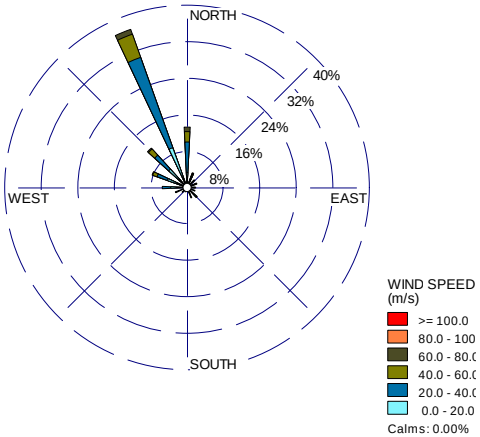
May hrs 6 to 11



May hrs 12 to 17



May hrs 18 to 23



6.4.1.5 Woodheaters

Domestic woodheaters are used in the Wagga Wagga and Albury regions. Figure 6-21 presents the average daily maximum, minimum and 9am and 3pm temperatures for each month averaged over the five-year period 2003 to 2007 at Wagga Wagga. The remaining exceedence days after removing dust storms and wildfires are also shown. July is the coolest month and therefore likely to be the time of increased woodheater usage. The PM_{10} data shows however that there were no exceedences during July in this adjusted data set. Data presented in section 3 also suggest lower overall daily concentrations during winter in Wagga Wagga and Albury, so it is therefore unlikely that woodheaters are a significant source of particles in these regions.

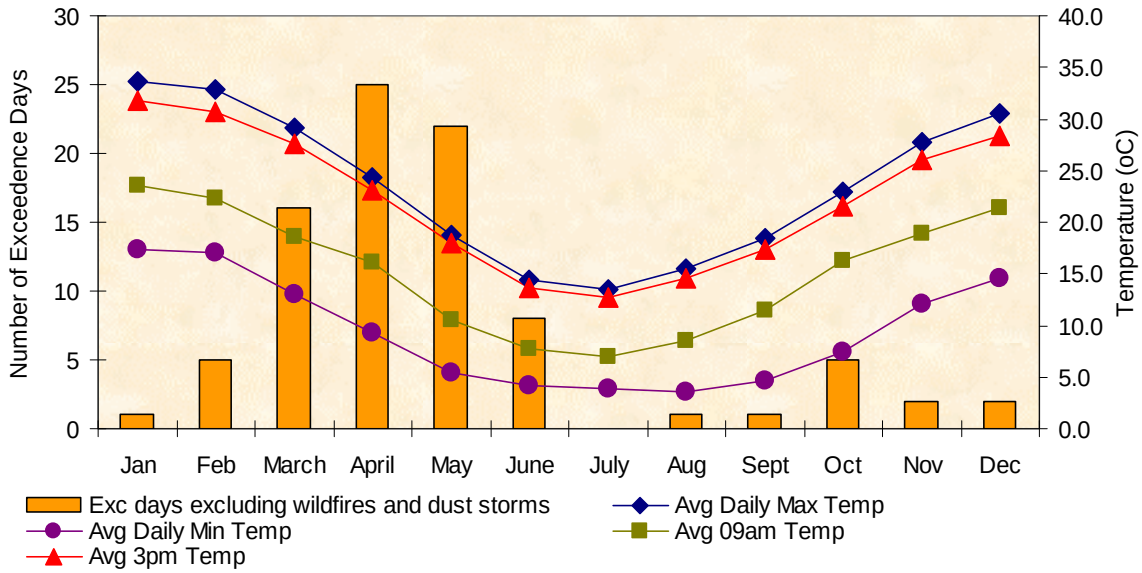


Figure 6-21: Temperature by month at Wagga Wagga and adjusted exceedence days (2003-2007)

6.4.1.6 Local sources near the Wagga Wagga monitoring site

When the Wagga Wagga monitoring site was commissioned in 2001, the surrounding area was covered in grass. The recent drought conditions have led to the grass dying off leaving bare soil. There is therefore the possibility of raised dust leading to increased PM_{10} concentrations caused by either higher wind speeds or cars using the area on days with limited dispersion.

The monitoring site is also located next to clay tennis courts, which are 'bagged' daily all year round, which may contribute to elevated PM_{10} concentrations over a short duration. Given that the higher concentrations often occur over several consecutive hours and the seasonal patterns found, it is unlikely that the tennis court maintenance practices are contributing to the seasonal pattern of the higher number of exceedence days in autumn at Wagga Wagga.

6.4.1.7 Particle chemical composition

Particle chemical composition was determined on MicroVol filters collected weekly between December 2007 and July 2008. Corresponding validated PM_{10} TEOM data was not available yet for this period. Figure 6-22 shows the weekly time series plot of $PM_{10,}$ laevoglucosan and $nssK^+$ concentrations. As discussed above, this measurement period includes the summer and early autumn, when wind blown dust is likely to be a significant regional source of PM, the crop stubble burning season and the early winter, when woodheater usage in Wagga is likely to be a significant local PM source.

Between December and April, $nssK^+$ levels followed the same trend as PM_{10} levels, although laevoglucosan levels remained low. This suggests that the likely particle source is dust. Between April and June, both laevoglucosan and $nssK^+$ levels were elevated, suggesting the presence of smoke.

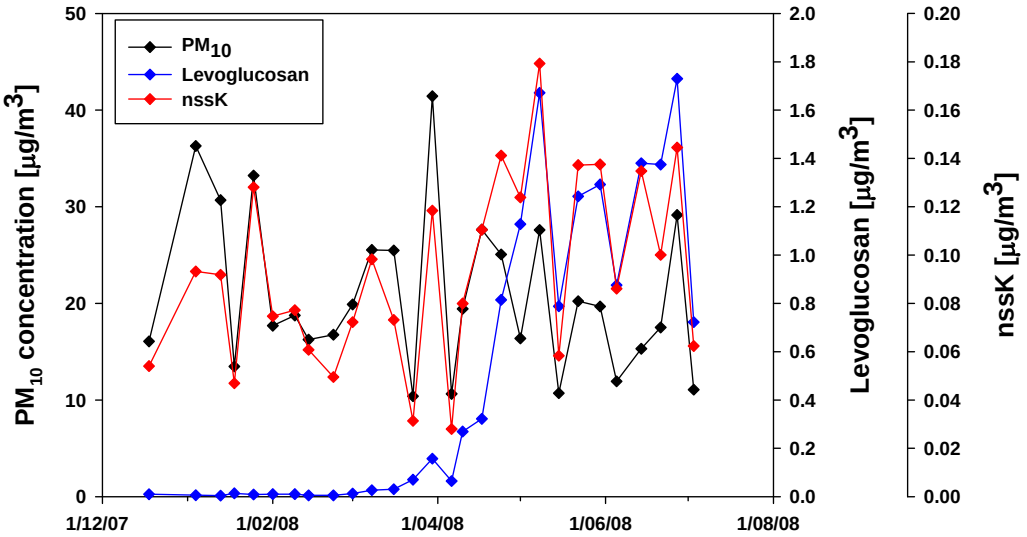


Figure 6-22 Weekly concentrations of PM_{10}, laevoglucosan and $nssK^+$ measured at Wagga Wagga, NSW.

The hotspots for January to June 2008 are shown in Figure 6-23. The number of hotspots identified from MODIS satellite data was highest in April (66%) followed by May (16%). June only recorded 1% of the number of hotspots identified between January and June 2008.

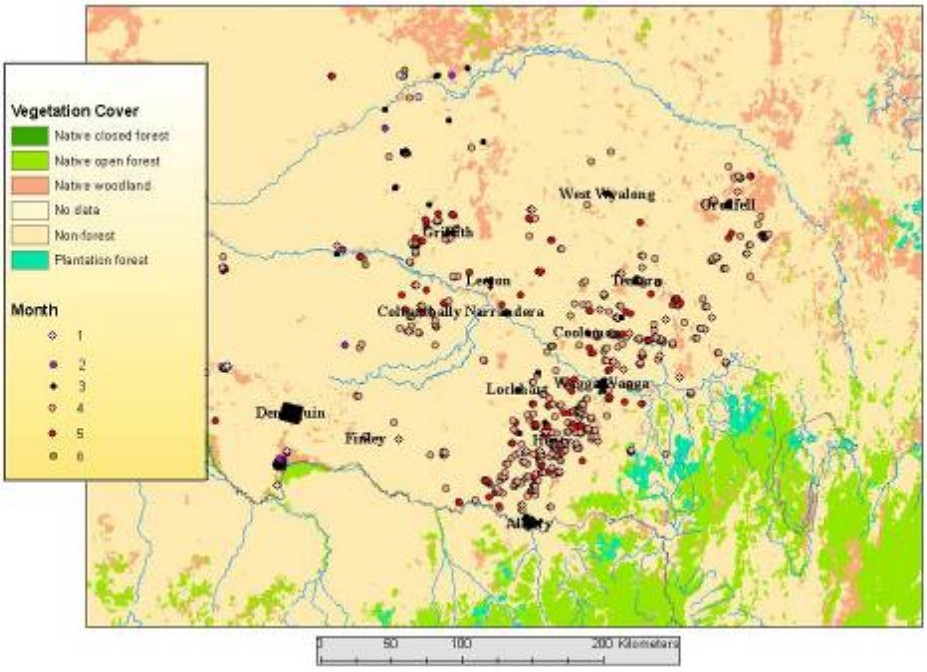


Figure 6-23 Hotspots identified between January and June 2008

Particle levels were high in April, consistent with the high number of hotspots identified in April; however only a small increase was observed for laevoglucosan levels. Currently laevoglucosan content from biomass burning in forest fuels is well known, but less information is available on laevoglucosan fraction for grassy fuels. PM emitted from combustion of forest fuels and wood typically has a laevoglucosan content ranging from 10% to 20% and averaging near 18% (Meyer et al., 2008b). The laevoglucosan content of grassy fuels while thought to be in the range of 5-10% is currently very poorly defined, particularly for Australian crops (R. Gillett, personal communication). Better information on laevoglucosan content in grassy fuels is needed to assess its source signature and the contribution of stubble burning to observed PM mass concentration.

A more significant increase in laevoglucosan levels was observed in May and levels remained elevated until July indicating that up to half the particulate mass may be from biomass combustion. As shown in Figure 6-24, the ratio of laevoglucosan to particle mass was also significantly higher between May and July. Since fire activity was low during those months, the elevated laevoglucosan levels and high laevoglucosan and $nssK^+$ to PM ratios may be due to other smoke sources.

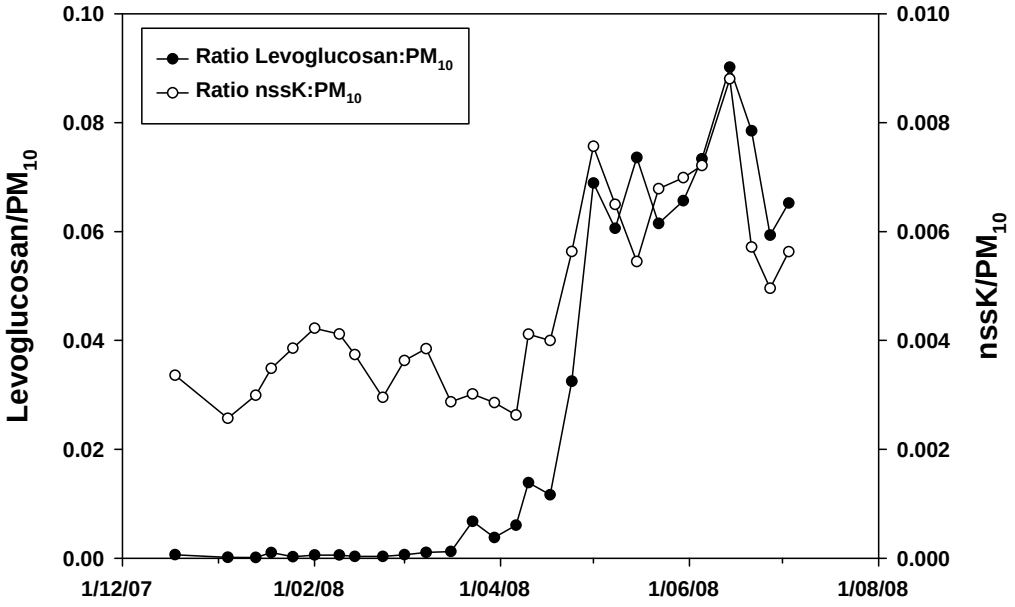


Figure 6-24 Weekly measurements of the ratios of laevoglucosan and $nssK^+$ levels to PM_{10} mass concentrations measured at Wagga Wagga, NSW.

7 IMPACT OF PRESCRIBED BURNING ON INDOOR AIR QUALITY

7.1 Introduction

A survey of how people spend their time in 1997 revealed that over 90% of people's time is spent indoors, much of which is spent at home (Table 7-1). Monitoring was undertaken to understand the impact of prescribed burning on indoor air quality in the home in comparison to the impact from other activities on indoor air quality. Indoor and outdoor levels of $PM_{2.5}$ and BTEX in houses during periods of prescribed burning were measured. These levels were compared to typical levels found in urban and rural Australian residences in the absence of prescribed burning. A mass balance equation describing indoor and outdoor influences on indoor quality was also used to understand whether strategies such as changing the ventilation state of the house can reduce impacts of prescribed burning on indoor air quality.

Table 7-1. Time budget for Australian persons 15 years and older, 1997, including weekends and weekdays.

Environment	Minutes/Day	% of Day
Home (personal- & child-care, domestic & unpaid	820	57
Work (employment and education)	199	14
Shopping (goods and services)	29	2
Recreation and social activity	262	18
Transit (associated with all environments)	73	5
Outdoors (domestic, recreation and social activity)	54	4

Source: ABS 1997

7.2 Methodology

7.2.1 Indoor air quality measurements

To assess the influence of smoke plumes on indoor air quality, sampling was performed at residences during a period of scheduled prescribed burning in the Ovens region. Sampling was simultaneously performed in 4 houses over 5 days from 2 May to 7 May 2007. The locations of monitored residences are shown in Figure 7-1 and Table 7-2.

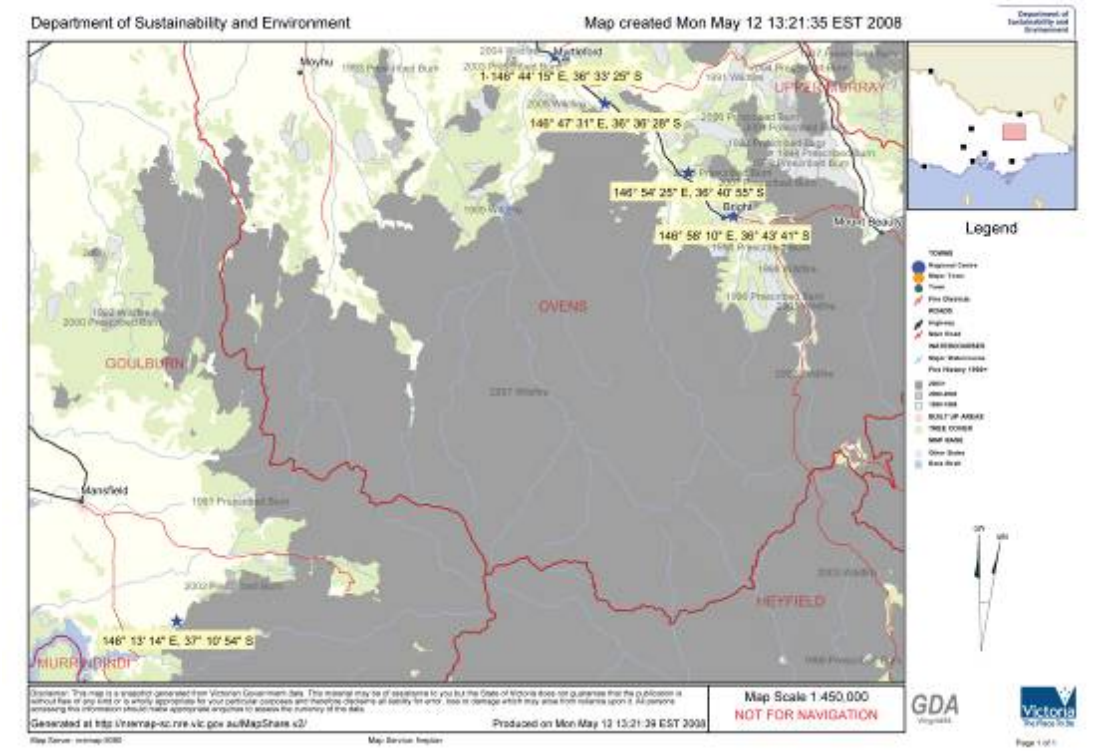


Figure 7-1 Locations of sites for indoor and outdoor monitoring programs in Victoria.

Table 7-2: Locations of sites for indoor and outdoor monitoring programs in Victoria.

Ovens DSE	Boorolite	Bright	Myrtleford1	Myrtleford2	Porepunkah
146° 47' 31" E	146° 13' 14" E	146° 58' 10" E	146° 43' 24" E	146° 44' 15" E	146° 54' 25" E
-36° 36' 28" S	-37° 10' 54" S	-36° 43' 41" S	-36° 33' 26" S	-36° 33' 25" S	-36° 40' 55" S

7.2.1.1 $PM_{2.5}$ measurements

Low-volume *Microvol-1100* (Ecotech Pty Ltd, Australia) gravimetric samplers fitted with $PM_{2.5}$ impactors were used to collect 24-hour $PM_{2.5}$ aerosol on filters and to determine mass concentration according to AS/NZS 3580.9.10:2006. One sampler was located in the living/family room of the residence. A second sampler was located outside the residence.

Continuous $PM_{2.5}$ was also logged within each residence using a *DustTrak* (TSI Inc., USA) instrument, which measures the degree of scattering across a path of light caused by particles. The instrument response was corrected using a linear regression between gravimetric $PM_{2.5}$ measured within each residence and the *DustTrak* logger output for the Myrtleford 1 (R^2 0.99, n=5), Myrtleford 2 (R^2 0.92, n=4), Bright (R^2 0.94, n=5) and Porepunkah (R^2 0.98, n=5).

Continuous 5-minute $PM_{2.5}$ data from the outdoor site located at the DSE Field Office at Ovens was used to quantify smoke plume behaviour throughout the prescribed burn period. The instrument response was corrected using a linear regression (R^2=0.82) between gravimetric $PM_{2.5}$ measured at that site and the *DustTrak* logger output.

The sampled filters were subsequently analysed for mass concentration and tracers of woodsmoke (non-seasalt potassium and laevoglucosan) in the soluble fraction of the aerosol. Details of the analytical method are outlined in section 5.2.5

7.2.1.2 BTEX

BTEX measurements were also conducted in each of the 4 residences using the same method as described in section 5.2.6. For sites 1 and 4, two BTEX samples (in duplicates) were taken over the sampling period, one sampling over a 1-day period and

the other sampling over a 4-day period. At sites 2 and 3, three BTEX samples (in duplicates) were collected, the first one over a 1-day period and the other 2 over a 2-day period.

7.2.1.3 Activity monitoring

Whilst no diary was recorded by the residents, the residents were interviewed after the completion of sampling. Observations were also recorded by technicians changing the daily-sample filters.

7.2.1.4 Measurements during bushfire season

Additional sampling was performed at a residence in Boorolite during 3 periods in the December 2006/January 2007 Victorian wild fires. The availability of only one *DustTrak* required moving the sampler between indoors and outdoors to sample both environments. The sampler was predominantly left outdoors to quantify smoke plume behaviour during the wild fires and periodically was taken indoors briefly for periods of about 10 minutes to determine the indoor to outdoor $PM_{2.5}$ ratio. As there were no gravimetric measurements available to correct the DustTrak data, all concentrations greater than 10 $\mu g/m^3$ were divided by a factor of 3.0 (Chung et al., 2001). The residents recorded observations such as wind and smoke plume conditions outdoors, particle-generating activities within the residence and the ventilation state of the house.

7.2.2 Ventilation rate measurements

Infiltration rates for each residence were measured using a carbon dioxide (CO_2) release method described in Dunne et al. (2006). The method involved filling residence with 5000 ppm of CO_2 and measuring the rate of decay in indoor CO_2 concentration using three *QTrak* monitors (TSI Inc.), logging at 1-minute intervals located in separate rooms around the house. Measurements were performed with all external door and windows closed and with all internal doors open (except for the toilet), allowing the interior to be treated as a single well-mixed zone. An additional monitor was placed outside to correct for contributions from outdoor CO_2. Upon completion of the CO_2 release, the residence was vacated, closed up and left for 3 hours. To prevent indoor

sources of CO_2 during testing, the residence was vacant and no combustion sources were operated.

7.3 Results

7.3.1 Indoor measurements during prescribed burning

Indoor monitoring was performed in 4 residences during the April/May 2007 period from 2 to 7 May. Unfortunately prescribed burning in the Ovens Valley only occurred on the first day of indoor monitoring. For this day, the impact of smoke on IAQ was greatest at the Bright residence, which was unoccupied for the period and thus had no additional indoor sources of PM. For the remainder of the indoor monitoring period, there were no local emissions from prescribed burns thus indoor air quality during these periods was the product of non-prescribed burning sources only.

Table 7-3 displays the data collected during the indoor air monitoring.

Table 7-3 Range of average indoor $PM_{2.5}$ [$\mu g/m^3$], outdoor $PM_{2.5}$ [$\mu g/m^3$], I/O ratio, laevoglucosan (LG) to $PM_{2.5}$ ratio and BTEX concentrations [ppb]

Compound	Bright	Myrtleford 1	Myrtleford 2	Porepunkah	DSE site
Indoor $PM_{2.5}$, 24-h	12 – 29	31 – 54	15 – 24	14 – 89	
Outdoor $PM_{2.5}$, 24-h	13 – 32	13 – 17	11 – 22	12 – 20	
I/O ratio, 24-h	0.6 – 0.9	1.9 – 3.2	0.7 – 1.4	0.7 – 7.3	
Indoor $PM_{2.5}$, 5-min	8 – 59	14 – 140	12 – 104	11 – 520	
Outdoor $PM_{2.5}$, 5-min	N/A	N/A	N/A	N/A	
I/O ratio, 5-min	N/A	N/A	N/A	N/A	
Indoor LG/$PM_{2.5}$	ND-0.054	ND-0.018	ND-0.032	ND-0.009	
Outdoor LG/$PM_{2.5}$	ND-0.058	ND-0.09	ND-0.09	ND	
I/O LG ratio	0.9-1.2	0.2	0.4		
Benzene	0.5		0.2-0.4		0.2
Toluene	0.5-1.1	1.5-2.0	0.4-0.6	4.5-5.7	0.1-0.5
Ethylbenzene	0.3-7.1	0.7-1.2	0.3-0.5	5.4-7.5	0.05-0.07
Xylenes	0.2-3.2	0.7-1.1	0.2-0.5	10.0-14.2	0.03-0.06

N/A Insufficient data collected, ND – non detected

Daily indoor $PM_{2.5}$ exceeded the 24-h advisory NEPM of 25 $\mu g/m^3$ for 5 days in the Myrtleford 1 residence, 2 days in the Porepunkah residence and 1 day in the Bright

residence. Indoor levels were in general higher than outdoor levels both for $PM_{2.5}$ and BTEX.

As displayed in Figure 7-2, daily indoor/outdoor (I/O) ratios for $PM_{2.5}$ exceeded one for 5 days in the Myrtleford 1 residence, 3 days in the Myrtleford 2 residence and 2 days in the Porepunkah residence. Highest I/O ratios were measured at the Porepunkah residence with lowest ratios measured at the Bright residence which was unoccupied during the sampling period. The higher indoor levels of pollutants at the other residences suggest particulate contributions from indoor sources.

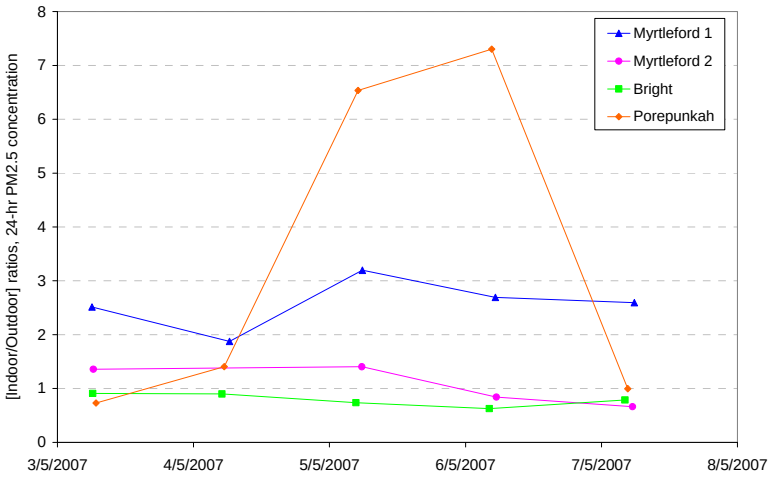


Figure 7-2 Indoor/outdoor $PM_{2.5}$ ratios concentrations at 4 houses in the Ovens Valley, May 2007.

7.3.1.1 Time series analysis of indoor $PM_{2.5}$

Continuous measurements of indoor $PM_{2.5}$ concentrations were used to identify time periods when elevated levels occurred and to assess the activities leading to high particulate levels. The time series of $PM_{2.5}$ concentrations are shown in Figures 7-3 to 7-6 for each residence.

Indoor $PM_{2.5}$ concentrations were elevated at the residence at Myrtleford site 1 with the highest peak levels occurring at night (Figure 7-3). Indoor sources were identified in the Myrtleford 1 house as environmental tobacco smoke (ETS) and incense. The occupants smoked inside the residence and intermittently burned incense to cover the smell of the

ETS. Tobacco and incense smoke odours were observed by technicians changing the filters.

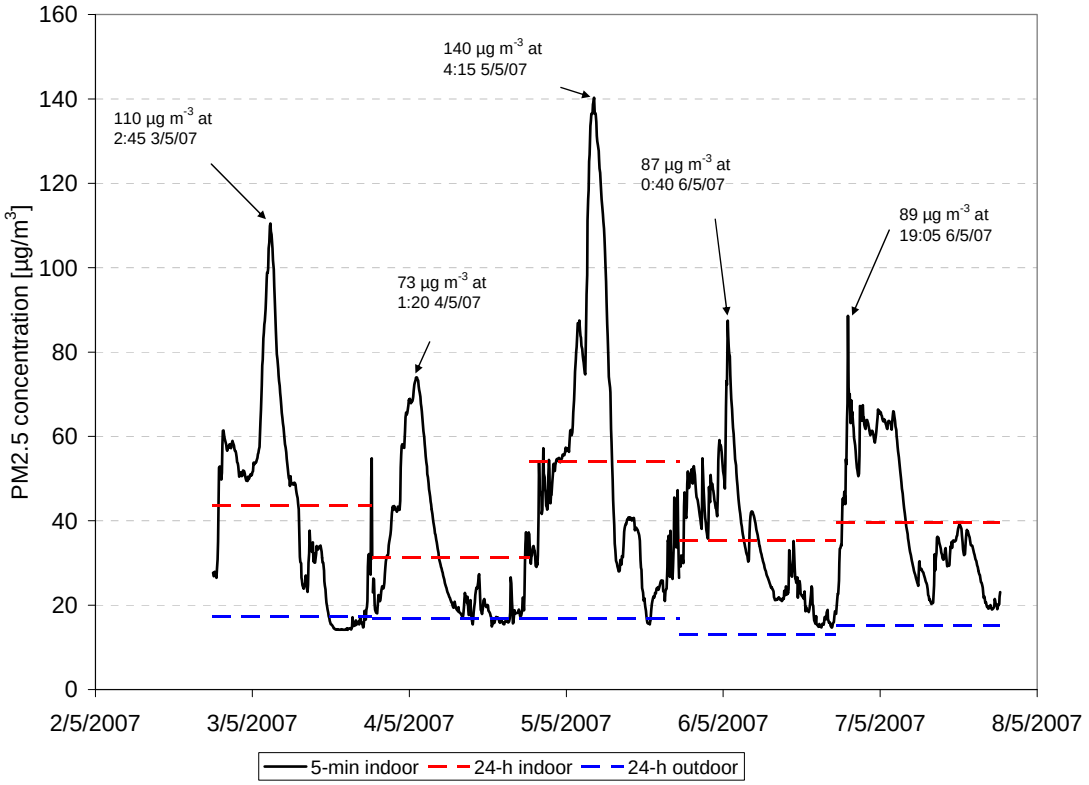


Figure 7-3 Indoor and outdoor $PM_{2.5}$ concentrations at a smoking residence at Myrtleford site 1, May 2007.

The major indoor source identified in the Myrtleford 2 residence was an open fire place in the same room as the indoor sampler (Figure 7-4). Wood smoke odour was observed by technicians changing the filters on May 3. The sampling equipment was moved to an adjacent room on the second day, resulting in a drop of $PM_{2.5}$ levels. Very low $PM_{2.5}$ levels were monitored on the 4th and 5th day, when the residence was vacant.

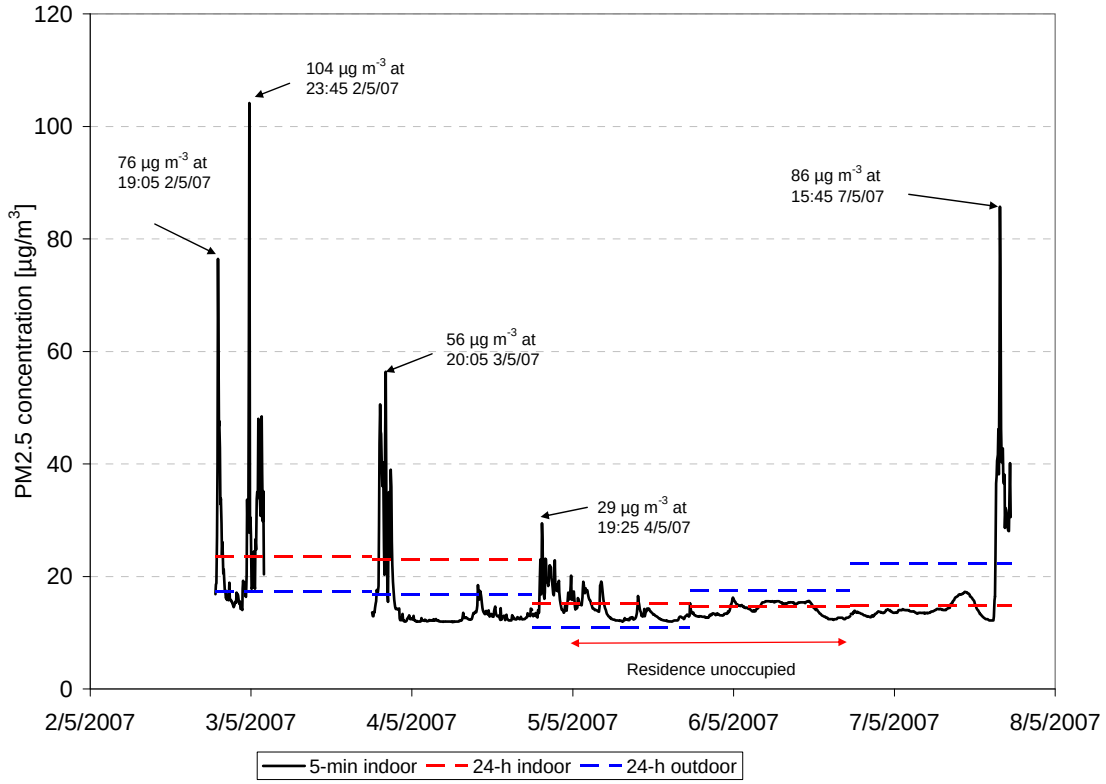


Figure 7-4 Indoor and outdoor $PM_{2.5}$ concentrations at a residence using a woodheater at Myrtleford site 2, May 2007.

Very high short-term $PM_{2.5}$ peaks were measured at the Porepunkah residence with the majority of peak concentrations occurring between 19:00 and 20:00 (Figure 7-5). When questioned about the events at a later date, the occupants recalled they had dinner guests and that it was possibly related to cooking. High BTEX levels were also measured at this residence; the source of these is unknown. A high-PM peak of short duration that occurred at 9:10 in the morning was also unexplained. The rapid decay suggests the source was very close to the sampler indoors. Low laevoglucosan concentrations in the aerosol samples suggest that the high BTEX and PM levels were not from wood-combustion sources inside or outside the residence.

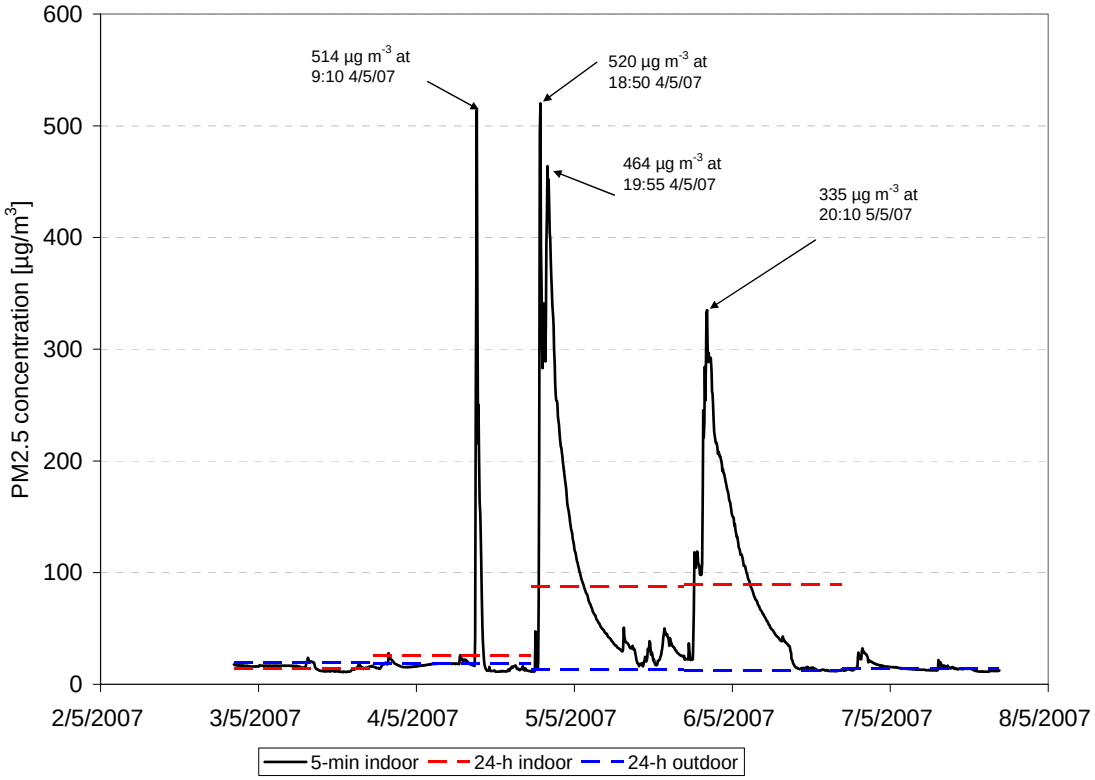


Figure 7-5 Indoor and outdoor $PM_{2.5}$ concentrations at a residence in Porepunkah, May 2007.

Indoor concentrations were lowest at the Bright residence which was unoccupied during the indoor monitoring (Figure 7-6). On the first day of monitoring, the indoor concentration exceeded the 24-h $PM_{2.5}$ advisory NEPM. The Bright residence was unoccupied during the indoor monitoring and I/O ratios remained below 1 (0.6-0.9), indicating the high indoor $PM_{2.5}$ originated outdoors. Both the indoor and outdoor samples had elevated laevoglucosan concentrations of >1 µg m^{-3}, which suggests high indoor $PM_{2.5}$ resulted from penetration wood-smoke from outdoors, from the prescribed burns and/or from wood-heater emissions.

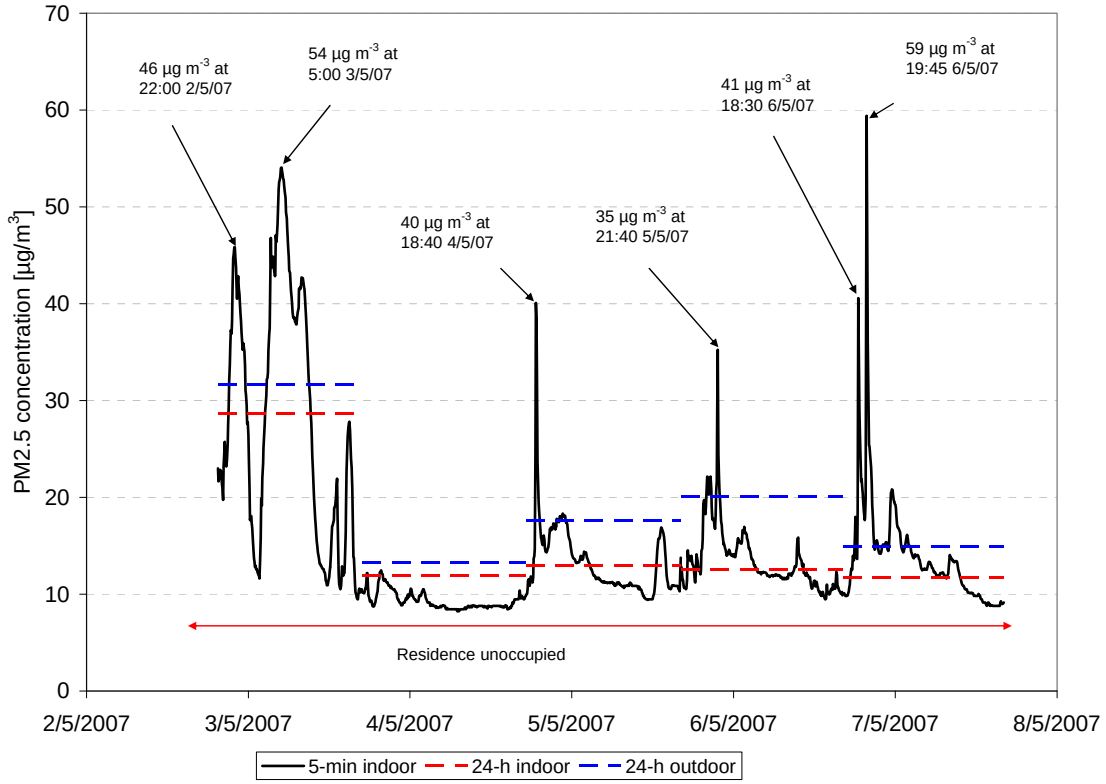


Figure 7-6 Indoor and outdoor $PM_{2.5}$ concentrations at a residence in Bright, May 2007.

Influences on these houses have been found to include emissions from cooking, fire-place, incense, cigarettes, activities causing resuspension and penetration from outdoor sources, including smoke from wood fires on the first day of sampling.

7.3.2 Ventilation rate measurements

Ventilation of the house influences the rate and degree of exchange between indoor and outdoor air. To determine the minimum ventilation rate of the house, also known as the infiltration rate, air exchange measurements were performed on each of the residences. The rate of decay of CO2 for each residence is shown in Figure 7-7. Note that the ventilation of the house is only measured over a 3-hour sampling period and only considers ventilation when all external openings were closed as tightly as possible. Three of the four residences had a small number of windows that could not be completely closed due to warping of the frames, resulting in a permanent opening to outdoors of between 0.02 and 0.34 m^2. The measured air exchange rates for the residences ranged from 0.29 to 0.9 h^{-1}, with the highest exchange rates associated with the highest area of permanently-open window.

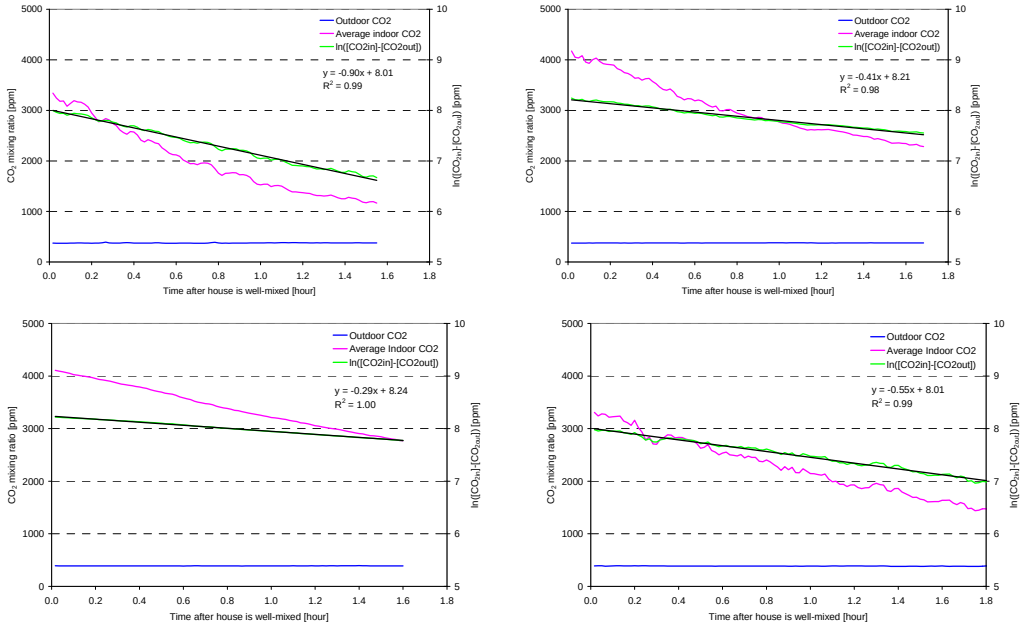


Figure 7-7 Ventilation measurements in the a) Porepunkah, b) Bright, c) Myrtleford 1 and d) Myrtleford 2 residences.

7.4 Discussion

7.4.1 Model

Factors affecting the influence of a smoke plume on indoor air can be expressed in terms of a mass-balance equation outlined in Dunne et al 2006 and in many previous papers (Dockery and Spengler 1981, Raunemaa et al 1989, Thatcher and Layton 1995, Freijer and Bloemen 2000). The change in concentration of indoor pollutants is influenced by

- Indoor sources of pollutants
- Air exchange rate which defines the ingression of outdoor pollutants and egression of indoor pollutants
- Decay rate of pollutants

Over a 24-hour measurement period, the mass balance equation can be simplified if under steady state conditions to be:

$$\overline{C}_{i,j} = \overline{C}_{o,j}\left[\frac{Pa}{a+K_j}\right] + \left[\frac{S_j}{V\left(a+K_j\right)}\right]$$

Where

$$a = \frac{Q}{V},\ K_j = \frac{v_{d,j}A_s}{V}$$

a = air exchange rate (h^{-1})

Q = air flow rate ($m^3.h^{-1}$)

V = interior volume of furnished building (m^3)

P = penetration factor if pollutant j into building (1=100% penetration, assume 1 for $PM_{2.5}$)

A = total surface area of furnished building interior (m^2)

$\overline{C}_{i,j}$ = average indoor i concentration of pollutant j ($g.m^{-3}$)

$\overline{C}_{o,j}$ = average outdoor o concentration of pollutant j ($g.m^{-3}$)

v_j = deposition velocity of pollutant j ($m.h^{-1}$)

K_j = decay rate of pollutant j ($m.h^{-1}$)

$S_{i,j}$ = emission rate of indoor i pollutant j source ($g.h^{-1}$)

Major indoor sources and their influence on indoor air quality will be discussed in section 7.4.2, whereas the influence of air exchange rate and decay rate of pollutants on indoor air quality will be covered in section 7.4.3.

7.4.2 Indoor air quality

The indoor measurements conducted in the 4 residences at Ovens identified a range of indoor $PM_{2.5}$ sources including emissions from Environmental Tobacco Smoke (ETS), burning incense, combustion in an open fire place and from cooking and contributions from outdoor sources, including smoke from wood fires on the first day of sampling. Combustion activities such as cooking, burning candles or incense and smoking have been found to be the dominant indoor sources of fine particles (0.1-2.0 μm). A study conducted in 136 homes across NSW suggested that the predominant source of PM_{10} exposure in homes for around 25% of the population was ETS (Sheppeard et al., 2006).

In homes free from tobacco smoke, the contribution of woodheaters, both operating in the home and in the region was the predominant influence on PM_{10}.

The most comprehensive set of $PM_{2.5}$ measurements in Australia was made as part of an NHMRC-funded study by Monash University, Department of Epidemiology and Preventative Medicine (Abramson M.: Personal communication). The study included 121 measurements of 24-hour averaged $PM_{2.5}$ within residences in the electorate of Goldstein, Melbourne from August 2003 to February 2005. Eleven households had indoor 24-hour $PM_{2.5}$ concentrations higher than the ambient air NEPM $PM_{2.5}$ advisory of 25 $\mu g\ m^{-3}$ (24 hour average). For 103 measurements where simultaneous indoor and outdoor $PM_{2.5}$ was measured, 86% of the indoor/outdoor ratios were greater than one, with a geometric mean indoor/outdoor ratio of 1.7. This suggests that the majority of households have significant indoor sources of $PM_{2.5}$, which result in higher exposure to $PM_{2.5}$ indoors than outdoors (Abramson M.: Personal communication).

An intensive investigation of $PM_{2.5}$ was carried out in a single non-smoking home in Melbourne over 7 weeks during winter 2000. During periods of inactivity outdoor $PM_{2.5}$ strongly correlated with indoor $PM_{2.5}$ concentration (Powell and Ayers, 2007). For periods of inactivity and no occupancy the indoor/outdoor $PM_{2.5}$ concentration ratio was 0.79. This ratio increased 0.93 during household activities, and increased further to 1.96 during periods of cooking and cleaning.

Many of activities carried out in a house (e.g. heating, cooking, walking, cleaning, burning candles) result in short-term elevated particle concentrations, adding to peak exposure, consistent to what has been observed in the Porepunkah residence. For example, He et al. (2004) measured the concentration of $PM_{2.5}$ associated with cooking activities in 15 homes in Brisbane and $PM_{2.5}$ concentrations showed an increase over the background by 3, 30 and 90 times during smoking, grilling and frying, with elevated concentrations persisting at least one hour after the start of cooking.

Overall, the $PM_{2.5}$ levels measured in the residences at Ovens are typical of what is found in other Australian households. The five indoor measurements from each of the four houses measured in the Ovens region are comparable with 50th and higher

percentile measurements performed in the NHMRC-funded study, shown in Figure 7-8. The outdoor measurements from each of the houses in the Ovens region fall into the top 20% of measurements, but as the Ovens measurements were performed over one week compared to the Melbourne measurements performed over 19 months, they are not truly comparable.

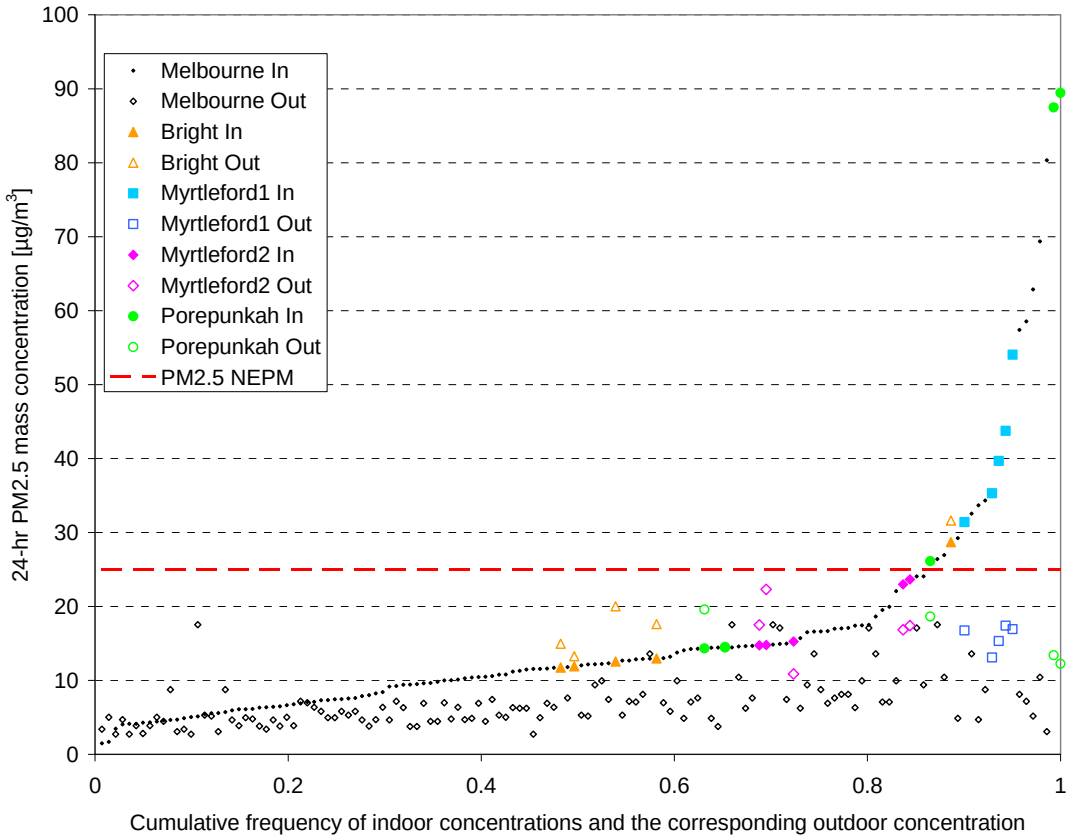


Figure 7-8 Indoor and outdoor $PM_{2.5}$ concentrations measured in residences at Ovens compared to those measured in Melbourne residences.

The two sets of data can be normalised by using the Indoor to Outdoor ratios, shown in Figure 7-9. The even distribution of I/O ratios from the Ovens houses compared to the range of the I/O from the NHMRC shows that these houses look typical in their indoor concentrations.

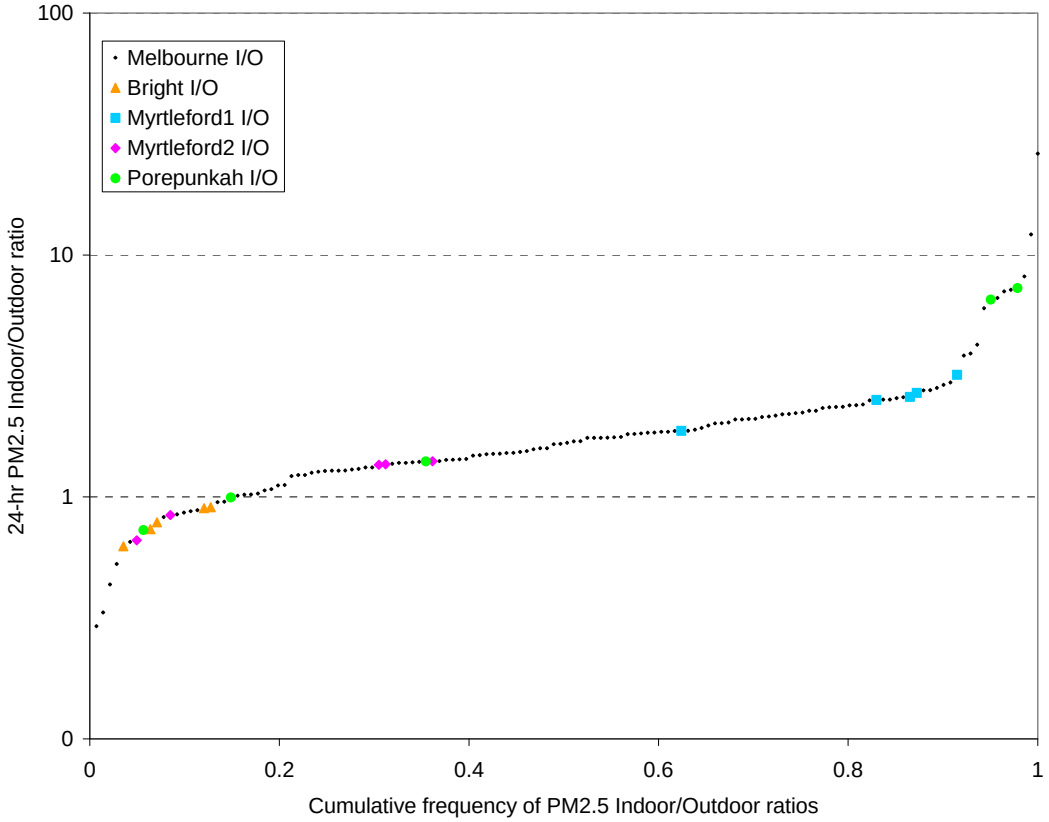


Figure 7-9 Indoor/outdoor $PM_{2.5}$ ratios measured in residences at Ovens compared to those measured in Melbourne residences

7.4.3 Outdoor penetration

The effect of outdoor pollution sources on indoor air quality is influenced by the air exchange rate as well as the deposition velocity of the pollutants. In the case of smoke particles, which have a peak mass size distribution at 0.3-0.5 micron, the dominant removal process is ventilation. This is also the easiest removal process to control with a house by controlling the state of external openings to the house. Both processes will be discussed below.

7.4.3.1 Ventilation

Ventilation is recognised as a significant influence on indoor air quality. Ventilation influences indoor concentrations by allowing mixing of outdoor air with indoor air. This process can act to dilute indoor concentrations from indoor sources and it can add to

indoor concentrations by bringing in pollutants from outdoor sources. Ventilation rate is expressed in air changes per hour (ach or h-1).

Infiltration is defined as the air exchange between outdoor air and indoor building air when the building in its closed up state thus the air exchange occurs through cracks, spaces and fixed ventilators in the building shell. Natural ventilation is defined as air exchange between the building interior and exterior through the same processes as infiltration and additionally through controllable openings such as vents, windows and doors.

The dynamics of infiltration and natural ventilation rely on a pressure differential between inside and outside air caused by external air advection or density differences due to temperature gradients between indoors and outdoors. Thus infiltration and ventilation rates vary according to meteorological conditions outside, temperature differentials between inside and outside and whether windows and doors are open. Natural ventilation is commonly used in single- and double-storey residences in Australia and may include some mechanical ventilation such as extraction fans in the kitchen, bathroom and toilet.

In this study, air exchange rate measurements were conducted in December 2007 over a 3-hour sampling period, with doors and windows closed and no mechanical ventilation. Air exchange rates are likely to vary under different meteorological conditions, in particular during windy conditions. Table 7-4 shows air exchange measurements performed in Australian residences, including those performed in this study. The air exchange rates measured in this study are similar to those measured in other studies. In general older houses have higher infiltration rates of 0.4 to 0.5 h^{-1}. Natural ventilation when external windows and doors are opened can be much higher, with air exchange rates in excess of 3 h^{-1}. When no manual ventilation is used, air exchange rates measured can be as low as ~0.15 h^{-1} in newer houses. Newer houses tend to have lower infiltration rates because the 1990 Building Code of Australia removed the requirement for fixed permanent ventilation (for improved energy efficiency).

Table 7-4 Air exchange measurements performed in Australian residences, including those performed in this study listed in *italics*.

Study description	Ventilation type	Air changes per hour, h^{-1}	Measurement method	Author
20-yr house	*0.00 m^2 openings*	*0.29 Myrtleford 1*	*CO_2 release*	*Current study*
5-yr bungalow	*0.02 m^2 openings*	*0.41 Bright*		
30-yr house	*0.02 m^2 openings*	*0.55 Myrtleford 2*		
40-yr house	*0.34 m^2 openings*	*0.90 Porepunkah*		
Houses, Melbourne	Infiltration, no wind	0.33	Pressurization	Biggs cited in Brown 1997
	Canberra estimate	0.44	Estimates are calculations based on typical meteorology	
	Sydney estimate	0.55		
	Hobart estimate	0.55		
	Melbourne estimate	0.57		
20-yr house, Melbourne	Natural (winter)	0.23 ± 0.03	CO_2 release	Dunne et al 2006
	Natural (summer)	1.4 ± 0.1		
43 houses, Sydney	Infiltration, winter	0.9 (0.2 - 2.3)	SF_6 tracer release	Ferrari et al 1988
Houses<5 yrs Sydney	Infiltration, winter	0.33	Tracer gas release	Ferrari 1991 cited in Brown 1997
116 houses	Natural, heater use	1.1 (0.1 - 3.8)	CO_2 depletion	DEH 2004
9 new houses, Perth 1985	Infiltration	0.05 – 0.41	Tracer gas release	Harrison cited in Brown 1997
14 houses, Brisbane	Infiltration	0.61 ± 0.45	CO_2 depletion	He et al 2005
	Natural	3.00 ± 1.23		
5-yr house	Infiltration, winter	0.19	CO_2 release	Physick et al 2008
5-yr apartment		0.30		
75-yr house		0.36		
40-yr house		0.39		
40-yr house		0.39		
30-yr unit Melbourne	Natural, 7 weeks, winter	0.50	Modelled	Powell & Ayers 2007
1 house, Adelaide	Infiltration	0.3 - 0.6	CO depletion	Steer et al 1990
	Window & doors open		SF_6 tracer release	
28 houses, Brisbane	Natural - all houses	1.44	Modelled	Yang et al 2004
	Houses pre-1990	1.76		
	Houses post-1990	1.32		

7.4.3.2 Deposition velocity

The mass of particles from biomass burning tend to have a bimodal distribution dominated by a fine-particle mode with a median diameter of 0.3 μm and a coarse particle mode with a median diameter larger than 10 μm [WHO 1999]. A similar distribution was observed in Australia by Keywood et al (2000), with a dominant fine-mode peak at 0.5 μm and a small coarse particle mode at about 10 μm (Figure 7-10). The measurements were made in Launceston, Australia in wintertime, when the airshed was impacted by wood-heater emissions.

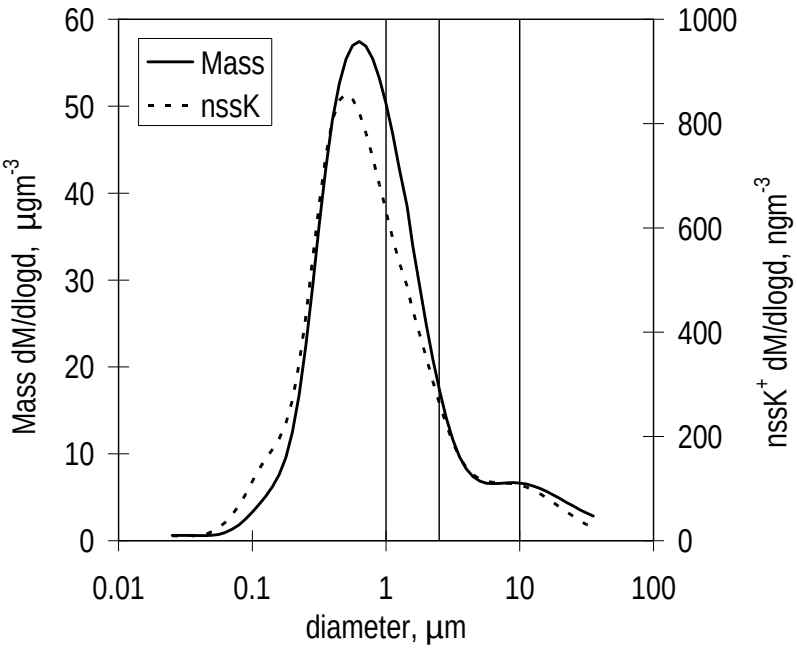


Figure 7-10: High mass sample from Launceston during winter, with $nssK^+$ used as the woodsmoke tracer, peaking at an aerodynamic diameter of 0.5 µm (Keywood et al 2000).

Using the steady state equation displayed in section 7.4.3 for the Bright residence which had no $PM_{2.5}$ indoor sources and an air exchange rate of 0.41 h^{-1}, the decay rate was found to be 0.12 h^{-1}, which is comparable to decay rates measured indoors by Abt et al (2000) for particle sizes of 0.3 to 0.5 micron. It can be assumed that when the aerosol mass is dominated by particles of smaller diameter, deposition occurs to all interior surfaces (Nazaroff and Cass 1989). Using the measured surface area to volume ratio for the Bright residence of 1.48, the mean deposition velocities calculated for each of the 5 days of measurements ranged from 0.08×10^{-4} to 0.46×10^{-4} $m.s^{-1}$, which are consistent with velocities described in Fogh et al (1997).

7.4.4 Indoor measurements during bushfires

Measurements of $PM_{2.5}$ using a *Dust Trak* were performed over 14 days at a house near Boorolite during the 2006/2007 wild fires in the Goulburn Valley, Victoria. These measurements can be used to understand the influence of fire smoke plumes on indoor air quality and on the activity patterns of the occupants. The results are presented in Figures 7-11 to 7-13, which show outdoor $PM_{2.5}$ on time scales of 5 minutes and spot measurements of indoor $PM_{2.5}$, also on time scales of 5 minutes.

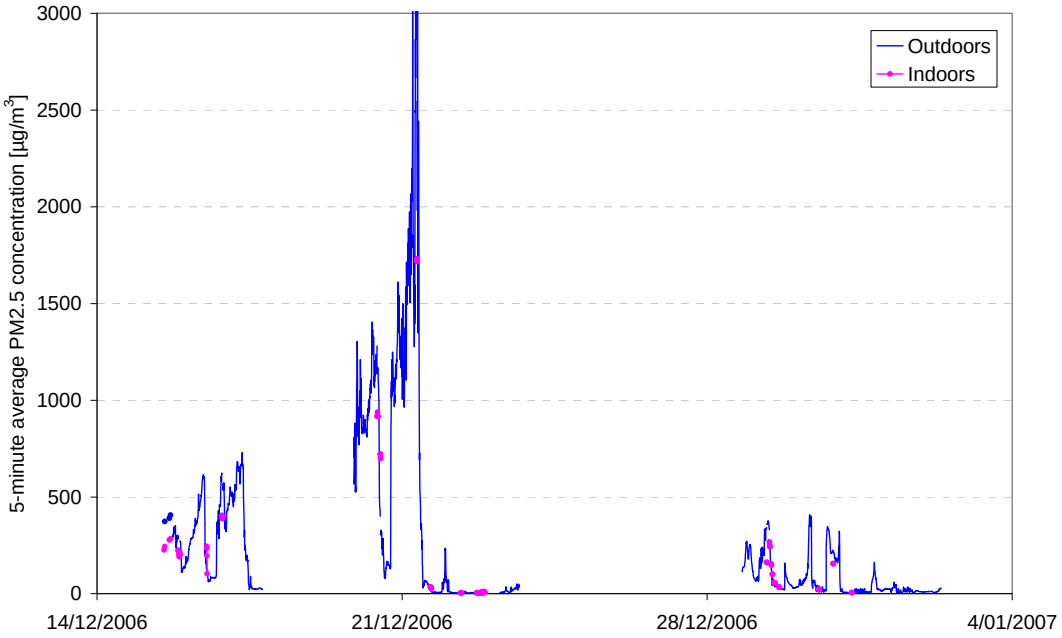


Figure 7-11 Outdoor and indoor $PM_{2.5}$ at a residence in Boorolite during 3 wild fire periods in December 2006 and January 2007.

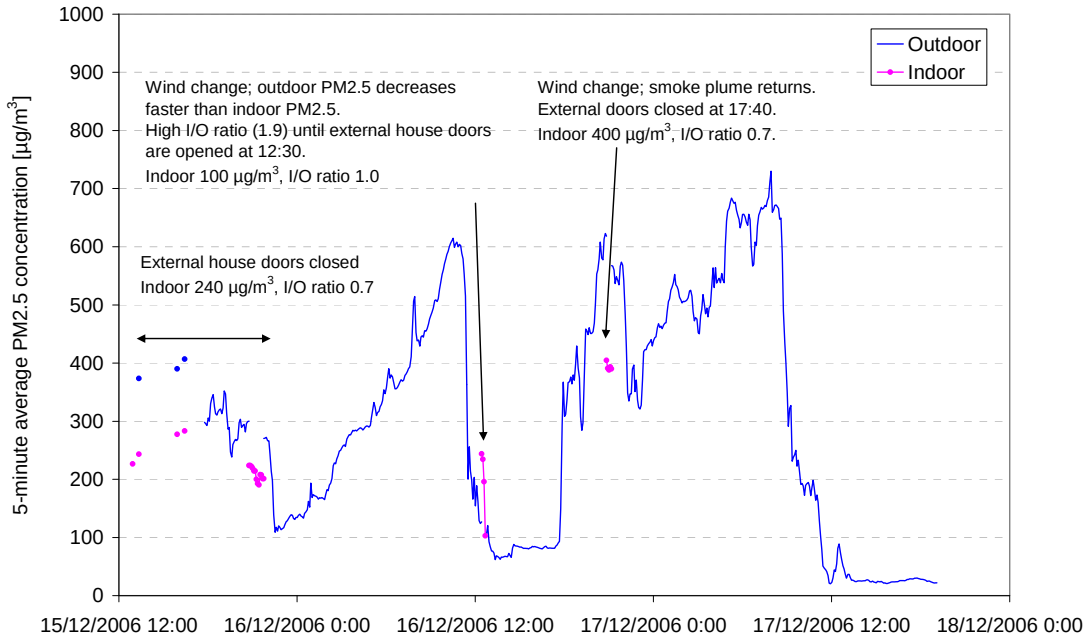


Figure 7-12: Outdoor and indoor $PM_{2.5}$ at a residence in Boorolite during wild fires, 15 to 18 December 2006.

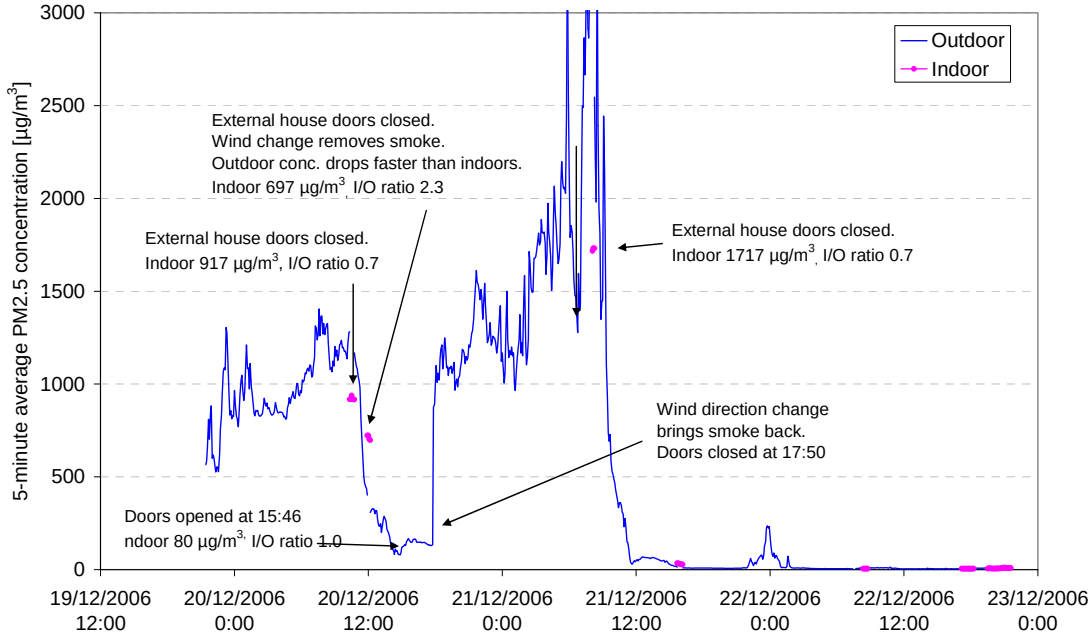


Figure 7-13: Outdoor and indoor $PM_{2.5}$ at a residence in Boorolite during wild fires, 19 to 23 December 2006.

A record of activity by the occupants revealed that the occupants closed all external windows and doors when the smoke plume was present outside. Spot measurements of

the indoor concentrations show that when the house was in its closed-up state and the indoor and outdoor concentrations were in equilibrium (not changing significantly over the time interval measured), the indoor/outdoor ratio remained at 0.7. This figure is typical of the levels you would expect when indoor and outdoor $PM_{2.5}$ are in a steady state condition with air exchange rates of about 0.3 to 0.5 h^{-1} and decay rates of 0.1 to 0.2 h^{-1} and there are no indoor sources.

When changes in indoor and outdoor concentrations were large compared to the 5-minute average measurement period, the I/O ratios were much more variable. There were two occasions during the fires when the I/O was much greater than 1 because the wind blew the smoke plume away but the house ventilation was low, resulting in higher concentrations indoors than outdoors. The occupants reported that in these conditions, they would open all external doors and windows to ventilate the house, so that the indoor concentration dropped to the outdoor concentration in minutes.

7.5 Strategies

It is reported that up to 75% of particle pollution can penetrate into houses with doors and windows closed and an air-conditioning system on recycle mode (WHO, 1999). This is similar to the I/O ratio of 0.7 measured at the Booralite residence with closed-up windows and doors. During periods of dense smoke, staying indoors with windows and doors closed will reduce and delay infiltration of smoke, reducing exposure to short-term peak smoke levels outdoors. Jamriska and Morawska (2003) found that smoke particles exhibited deposition loss rates of 0.2–0.4 h^{-1} and estimated, for a building with natural ventilation and an air change rate of 1.3 h^{-1} (which would be common for Australian housing), that indoor particle concentrations would be about 20% lower than outdoor concentrations. However, when plume moves away and outdoor concentrations decrease significantly, the indoor concentrations remain elevated. Opening door and windows during those periods will reduce indoor concentrations.

The Environmental Health Branch of the NSW Department of Health funded a survey of the Albury community shortly after the January/February 2003 bushfires to

determine the health effects of the bushfire smoke and the effectiveness of public health interventions. A total of 389 interviews were included in the analysis.

The study estimated that 30,470 people modified their daily activities during the period of high smoke levels (70% per cent of the Albury population). Of those people who modified their daily activities, the most frequent response was to have to have stayed inside as much as possible (88%) followed by reducing outdoor activities (54%) and closing windows and doors (44%). These figures equate to 62%, 38% and 31% of the total Albury population, respectively.

A small number of people limited exposure to the smoke by leaving the area. Whilst this may be a useful strategy during a periods of few burns and short burn durations, periods of frequent prescribed burning and potentially longer plume dispersal times (burns are often scheduled for periods of calm weather) leaving the area may not be a viable strategy.

Another option is to stay indoors with external doors and windows shut and operating an air cleaning device. Stand-alone room air-cleaning devices have become commercially available in recent years, especially for removal of fine particles. While it is important that these have a high one-pass efficiency (f %) for removing the particle sizes of interest, the capability to deliver a sufficient volume of air (Q, $m^3\ h^{-1}$) relative to the space (V, m^3) in which the device operates is of greater importance. Brown (1999) termed this the 'effective air cleaning rate (R, h^{-1})' where:

$$R = f.Q/100.V$$

and showed that even for a device with low f (21%), it was possible to reduce respirable particle levels in a space by nearly 80% if the device provided an R value 3 times higher than the ventilation rate of the space. A similar approach is now recommended in US standards where a 'clean air delivery rate' (CADR) is determined for commercial air-cleaning devices, with a minimum performance criteria for CADR to reduce particle levels in room air by 80% (Shaughnessy and Sextro, 2005). This corresponds to a

CADR of approximately 4 room volumes per hour for typical ventilation rates. Particle level reduction by 90% is possible by increasing CADR to 10 room volumes per hour, but this small increase in removal is at the expense of a 2.5 times larger and noisier device, and a CADR that delivers 80% reduction is considered a practical limit to room air cleaning (Shaughnessy and Sextro, 2005). This is currently not a common practice in Australia, but may emerge as a viable option in the future.

8 REGIONAL PREDICITION OF AIR QUALITY IMPACTS FROM PRESCRIBED BURNING

8.1 Introduction

Every year, biomass burning releases large quantities of aerosol that are significant at both regional and global scales. In terms of global total carbon emissions, it is estimated that biomass-burning emissions are approximately one third to one half of those from fossil fuel combustion, and that savannah fires are responsible for about 50% of the total emissions from biomass burning (Williams et al., 2007). According to a database (*viz.* GFEDv2) used by van der Werf et al. (2006), the global annual biomass burning emissions of total particulate matter and $PM_{2.5}$ (particulate matter with an aerodynamic diameter of 2.5 μm or less), averaged over the years 1997–2006, were 50 Tg and 37 Tg, respectively. Of these, the Australian contributions were 5.5% and 4.6%, respectively. Most of the Australian emissions result from wild fires and prescribed fires in the savannah regions of tropical Australia (Kasischke and Penner, 2004).

Spatio-temporally resolved fields of aerosol emissions due to biomass burning are essential for modelling and assessing their impact on air quality, climate and weather, and several researchers have developed such fields at global scale using a combination of satellite and ground-based measurements and semi-empirical methods (e.g., Ito and Penner, 2004; van der Werf et al., 2006). Recently, biomass burning emissions in northern Australia were considered in a two-part study by Meyer et al. (2008) and Luhar et al. (2008) who recognised that current inventory estimates of such emissions were not constrained by observations of concentrations of emitted species and were not

available at a sufficiently fine spatio-temporal resolution so as to be able to use them for assessing their impact of emissions on air quality.

Meyer et al. (2008) devised a scheme to construct a high-resolution (1 h, 1 km) inventory of aerosol emissions for the Top End (the monsoonal top half of the Northern Territory and a domain of around 1000 km × 1000 km) for the 2004 dry season (April–November), and validated the emissions using a three-dimensional meteorological and transport model called TAPM (Hurley et al. 2005) coupled with a variety of field measurements. Essential inputs to the emission calculations were fuel-load distribution, satellite-based measurements of fire scars, yielding burnt areas, and hotspots, providing timing information on daily basis. It was shown that hotspots without associated fire scars must be taken into account in order to produce credible aerosol fields. Prediction of emissions at hourly time resolution was enabled by assigning a diurnal variation based on a version of the McArthur fire danger meter (an empirical measure of expected fire behaviour for a standard fuel type). The transport modelling showed that emissions leaving the study region are largely advected to the west over the Timor Sea towards the Indonesian archipelago from April to September, shifting to the south-west during October–November.

Luhar et al. (2008) used the above aerosol emission rates in TAPM, assuming no secondary aerosol generation, and compared the modelled aerosol concentrations and aerosol optical depths with ground-based and satellite measurements, obtaining satisfactory results, thus effectively verifying the validity and accuracy of the emission calculation methodology of Meyer et al. (2008). The model hindcast numerous exceedences of the advisory maximum $PM_{2.5}$ exposure limit across the study region, with large areas in excess of 30 exceedences during the study period.

8.2 Present Modelling

This work follows the technique proposed by Meyer et al. (2008) and Luhar et al. (2008) and determine aerosol emissions from the Top End for the period 2003–2007 (5 years), providing information of the inter-annual variability of emissions patterns and consequently of air quality regarding particulate matter. We also use new and better

modelling options (e.g. gravitational settling and deposition) for aerosol concentration calculation than used in the previous study.

Recently measured aerosol concentrations for the year 2007 are used to further validate the techniques followed. The following describes the procedures followed, data analysis and modelling results.

8.3 Fuel load distribution

In the tropical savanna region of Australia, the fire season begins in early April, soon after the end of the monsoon period, and ends in November. The northern Australian flora is categorised as savanna woodland, consisting of trees with leaf litter and annual grass in between the trees. The dominant burning in the region is that of fine fuel (i.e. grass, leaf litter and twigs less than 6 mm in diameter). The same spatial distribution of fuel load ((kg C ha^{-1}) for the Top End as derived by Meyer et al. (2008) at a grid resolution of 1 km × 1km was used. The main tool used in deriving the fuel load is a semi-empirical model known as VAST (Vegetation and Soil carbon Transfer) developed by Barrett (2002). It is a biogeochemical production model relating the main drivers of production, intercepted radiation, temperature, soil moisture, rainfall, and vegetation class, to biomass and soil pools of carbon. The estimated spatial distribution of fuel in the Top End ranges between 1–3 tonnes C ha^{-1}, with the highest values located in the north-west and in some parts of the north-east, and the lowest loading located in the south (see Figure 8-1). The fuel load is coupled with satellite data for fire scars and hotspots in order to determine the amount of fuel burned within a grid cell.

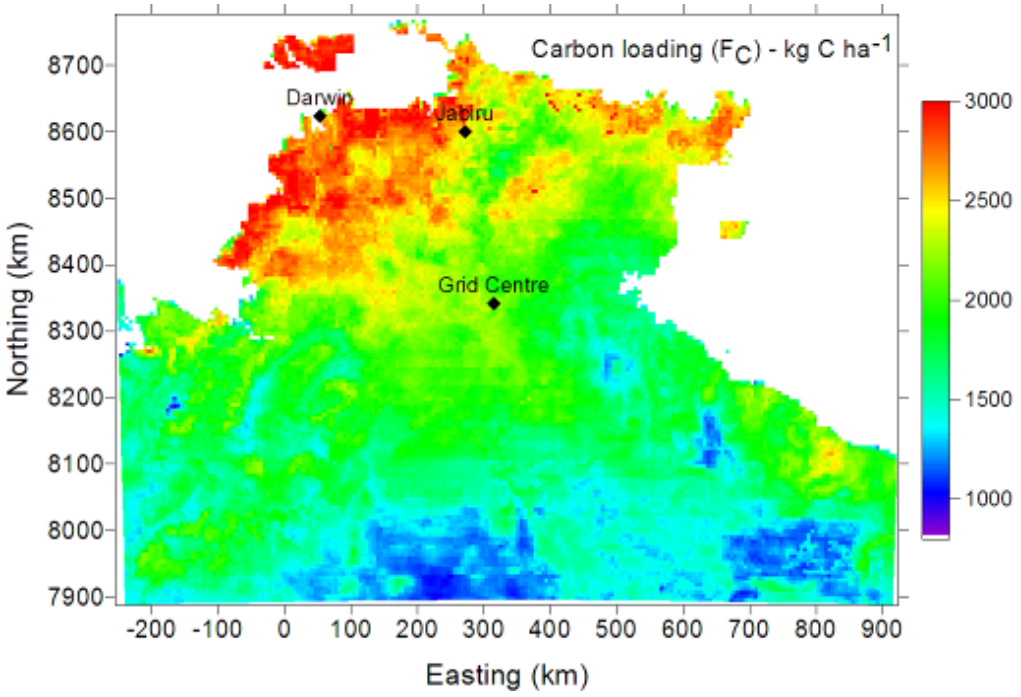


Figure 8-1: Spatial distribution of available fuel carbon loading (F_C) (kg C ha^{-1}) in the Top End of Australia at a resolution of 1 km × 1 km.

8.4 Distribution of daily fire emissions

The following satellite data for the Top End for the period 2003–2007 (5 years) were analysed to determine the fire emissions on a daily basis:

- Fire scars (or burned areas) at a grid resolution of 1 km × 1 km as reported by the Department of Land Information (DLI) of Western Australia (now Landgate; see http://www.rss.dola.wa.gov.au/newsite/noaaql/NOAAinfo.html; http://firewatch.dli.wa.gov.au), based on NOAA-AVHRR (Advanced Very High Resolution Radiometer) satellite images.
- Hotspot data at 1 km × 1 km resolution from Sentinel— a national bushfire monitoring system managed by Geoscience Australia — based on MODIS (Moderate Resolution Imaging SpectroRadiometer) on the Terra/Aqua satellites. Current data are available on-line at https://acres.ga.gov.au/modis_data.

The fire-scar data were available at a frequency of about once every ten days, whereas the hot-spot data were available at a frequency of once per day. A detected hot spot

indicates an actively burning fire within a grid cell at the time of satellite overpass, without providing information on the size of the fire. The above data are generally consistent with each other, but occasionally there may be cases when hot spots are detected with no associated fire scars, and similarly fire scars with no hot spots. Such discrepancies between hot spots and fire scars arise where fire-scar areas are below the resolution of AVHRR, where terrain is complex and fire scars are difficult to identify, or where fire intensity is low and the infrared emission from hot spots falls below the satellite detection threshold.

8.4.1 Fire scars with or without hotspots

Burned areas were identified using the fire-scar data. However, the fire-scar data are available only about once every ten days, without any knowledge as to when the burning actually took place and how it varied from day to day. To apportion a daily variation to a fire scar for a particular grid cell on a given date, the hot-spot data were used as follows. Within each fire-scar grid cell, the total number of hot spots occurring within a 31-day window consisting of 25 days prior to the fire-scar date and 5 days after this date was determined. Fire scars are identified from reflectance differences between successive pairs of images. The date assigned to the scar is that of the second image of the pair. This image was not always the image closest to the time of the fire, but the image in which the scar first became obvious. Hotspots coincident with scars provided far more accurate dates of fire occurance. It was rare for an area to burn more than once in any calendar year. The fraction of the fire-scar area occurring on a particular day within this window was determined as the number of hot spots on that day divided by the total number of hot spots within the 31-day window. Therefore, the daily fuel burned (mass/area/day) within the fire-scar grid cell is equal to this fraction multiplied by the fuel load. If there were no hot spots in a fire scar within the above 31-day window, then it was assumed that all the burning took place at the midpoint of the interval between AVHRR images i.e., on the day five days prior to the fire-scar date.

8.4.2 Hotspots without fire scars

A significant number of hot spots have no corresponding fire scars (termed as anomalous hot spots). Meyer at al (2008) found that anomalous hotspots must be taken into account in order to produce credible aerosol fields. They included such hot spots in the emission calculation methodology by assigning a burnt area to an anomalous hot spot. The anomalous hot-spot area data were gridded at a resolution of 1 km × 1 km. It was assumed that a hot spot for a particular date emits for 24 hours on that date.

An emission factor of 4.54 g $PM_{2.5}$ per kg of dry fuel burned for savanna and grassland was applied (Ito and Penner 2004).

8.5 Apportioning hourly variation

The calculated daily emissions of $PM_{2.5}$ were converted into hourly averaged emissions by applying the same diurnal function as derived by Meyer et al. (2008) based on a version of the McArthur fire danger meter (an empirical measure of expected fire behaviour for a standard fuel type). McArthur's fire danger meters estimate a fire danger index that is directly related to fire behaviour (e.g. chances of a fire starting, its rate of spread, flame height and difficulty of suppression) according to various combinations of air temperature, relative humidity, wind speed, and fuel characteristics. The computed diurnal function typically shows that the fire behaviour is less severe during the night, reaching a minimum shortly before sunrise. Subsequently, there is a sharp increase in the severity of fire behaviour, reaching a peak in early to late afternoon and then decreasing.

8.6 Calculated emission fields

Figure 8-2 presents the total emissions of $PM_{2.5}$ from the Top End for the years 2003–2007 calculated from the above methodology. A clear inter-annual variation is evident: the highest emissions were in the year 2004, followed by a dip in 2005 and an increasing trend for the subsequent two years. The second highest emissions were for 2007.

Table 8-1 gives the estimated total amounts of dry matter burned and carbon released into the atmosphere, in addition to the total $PM_{2.5}$ emissions plotted in Figure 8-2. The total carbon emission for the full year 2004 is 69.3 Tg, which can be compared to 67.6 Tg obtained by Meyer et al. (2008) for April–November for the same year. As noted by them, this figure is in remarkable agreement with the bulk estimate of 64.3 Tg derived using the NGGI (National Greenhouse Gas Inventory) methodology (AGO, 2007). Emissions of aerosols and other species can be calculated from the dry matter burned or from the carbon emitted by applying appropriate emission factors.

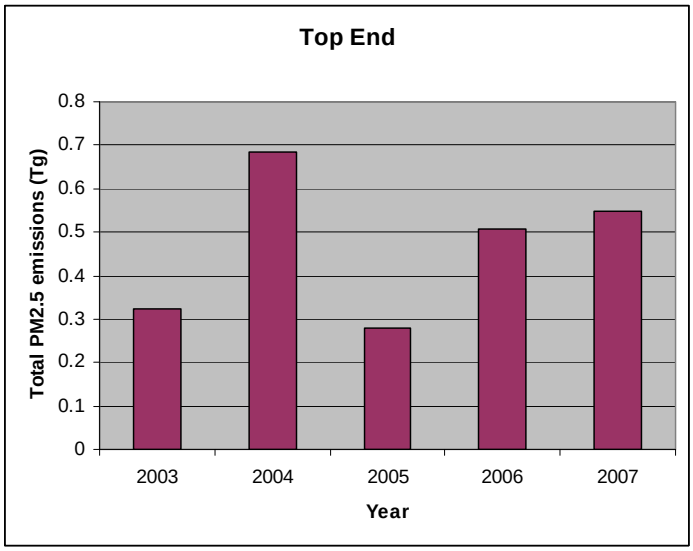


Figure 8-2: Calculated total emissions of $PM_{2.5}$ (Tg) from the Top End for the years 2003–2007.

Table 8-1. Estimates of biomass combustion in the Top End.

Year	Dry matter burned (Tg)	Carbon released (Tg)	$PM_{2.5}$ released (Tg)	GFEDv2 Carbon released[1] (Tg)
2003	71.6	32.9	0.325	23.5
2004	150.7	69.3	0.684	57.5
2005	61.2	28.2	0.278	23.2
2006	111.9	51.5	0.508	45.0
2007	120.7	55.5	0.548	-

[1]Based on figures from the GFEDv2 database (see van der Werf et al. (2006); data downloadable from http://ess1.ess.uci.edu/~jranders/data/GFED2).

For comparison purposes, Table 8-1 also presents the total amounts of carbon released from the Top End for the years 2003–2006 determined from the Global Fire Emissions Database (GFEDv2) given at a spatial resolution of 1° × 1° and a temporal resolution of 8 days (see van der Werf et al., 2006; data downloadable from http://ess1.ess.uci.edu/~jranders/data/GFED2). The GFEDv2 emission methodology also makes use of satellite data for burned areas and a biogeochemical model for fuel loads. Our estimates are consistently higher than those from GFEDv2 by 13–28%, but the similarity between the annual variations of the emission from the two datasets is remarkable, considering the fact that the two methodologies are independent with uncertainties in fuel loads and other combustion parameters, and differences in the way the satellite data have been used. However, note that our approach involves a much finer spatial and temporal resolution (1 km, 1h) with the emissions constrained by ambient aerosol concentration measurements.

Time series of the estimated daily $PM_{2.5}$ emissions (tonnes per day) presented in Figure 8-3 for the five years show that biomass burning starts in April and ends in November-December. The most intense emissions occur from September onwards, except for the low emission year of 2005 when they occur in April-May.

Figure 8-4 to Figure 8-8 present the calculated spatial distribution of $PM_{2.5}$ emission density (kg ha^{-1}) over two-monthly periods for the selected five years. The emissions are negligible during January–February, with a small increase in biomass burning during March–April in the western/north-western part of the Top End (except for the period in 2006 during which the emissions are negligible). During May–June, emissions increase, but are largely confined to the north-west of the region. Emissions extend to the east and north-east during July–August. For September–October, except of the year 2005, southern areas also burn, but their emission intensities are not as high as those in the north. During November–December, the fire activity in the north subsides, but large areas in the south continue to release aerosols (except for the year 2005).

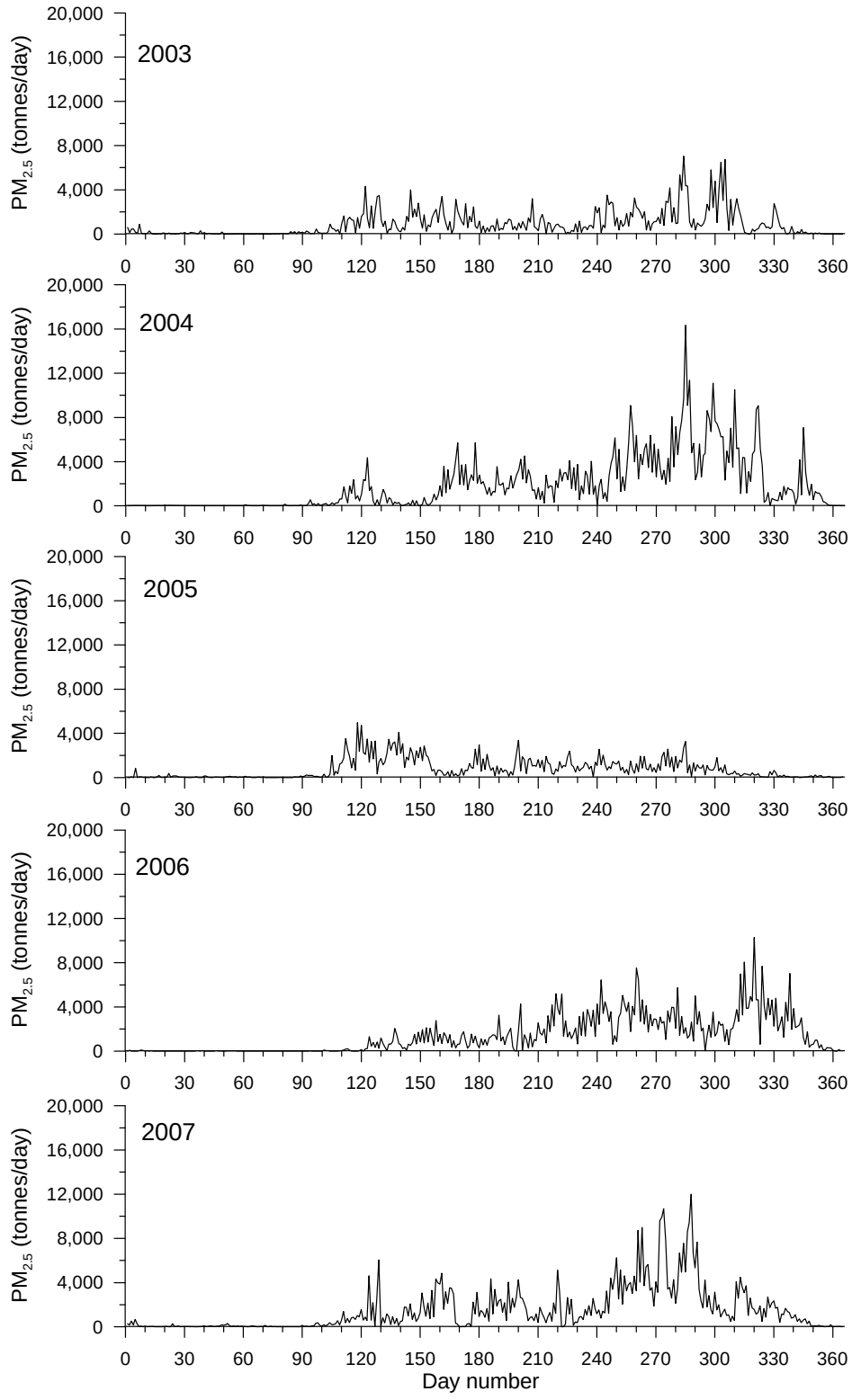


Figure 8-3: Calculated time series of daily $PM_{2.5}$ emissions (tonnes per day) from the Top End the years 2003–2007.

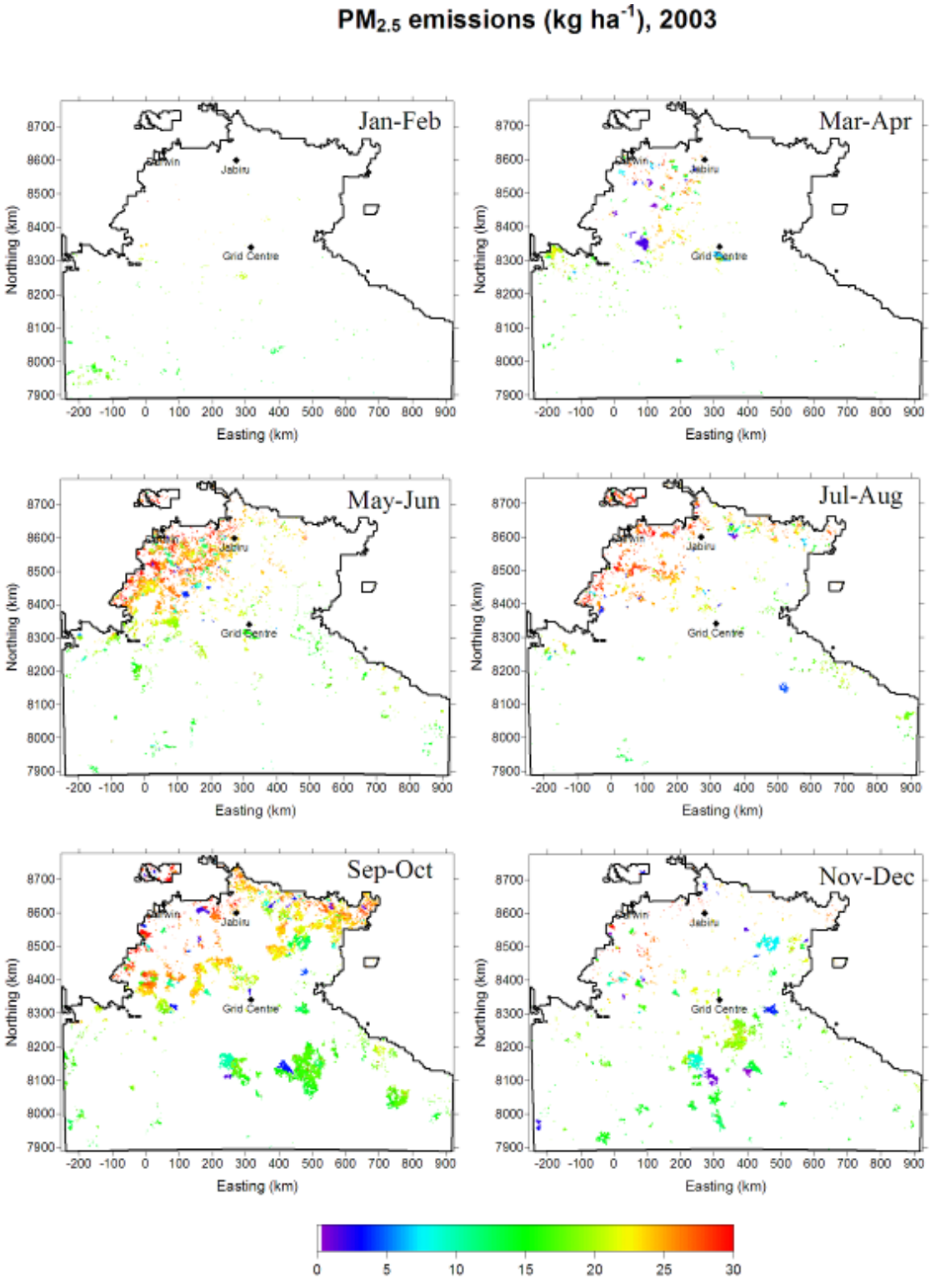


Figure 8-4: Spatial distribution of total $PM_{2.5}$ emissions (kg ha^{-1}) from biomass burning in the Top End at a resolution of 1 km × 1 km for two-monthly periods of the year 2003.

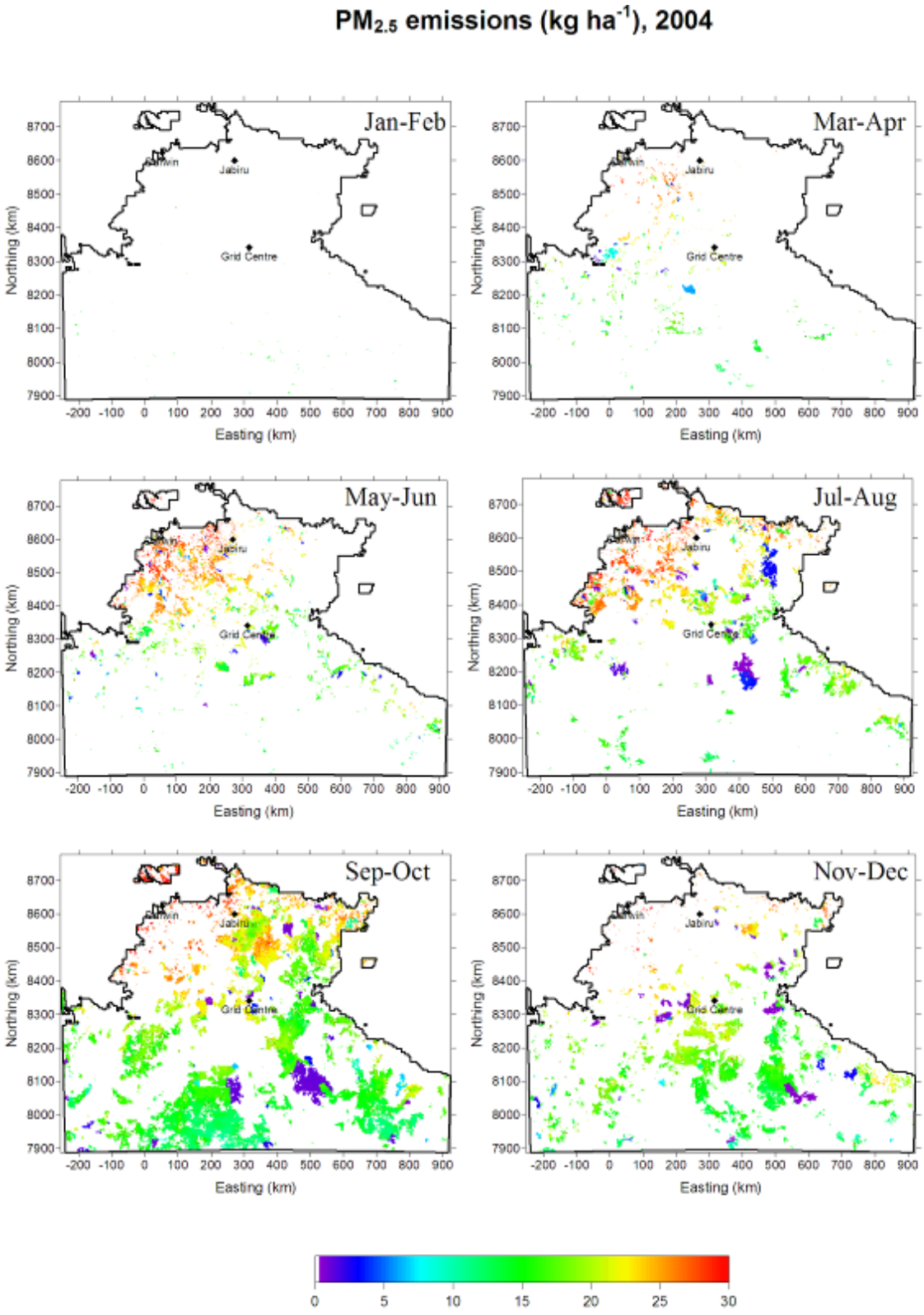


Figure 8-5: Spatial distribution of total $PM_{2.5}$ emissions (kg ha^{-1}) from biomass burning in the Top End at a resolution of 1 km × 1 km for two-monthly periods of the year 2004.

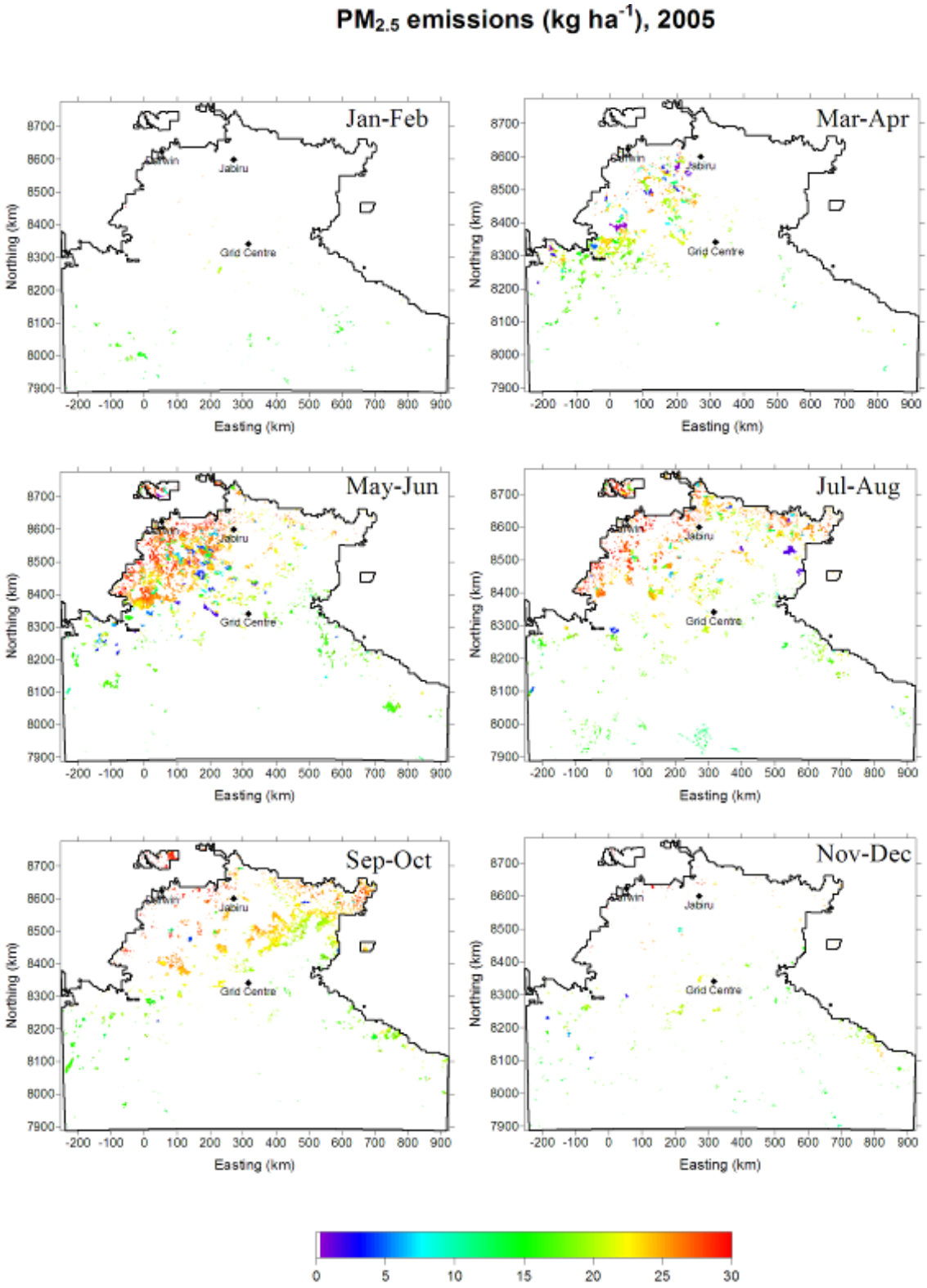


Figure 8-6: Spatial distribution of total $PM_{2.5}$ emissions (kg ha^{-1}) from biomass burning in the Top End at a resolution of 1 km × 1 km for two-monthly periods of the year 2005.

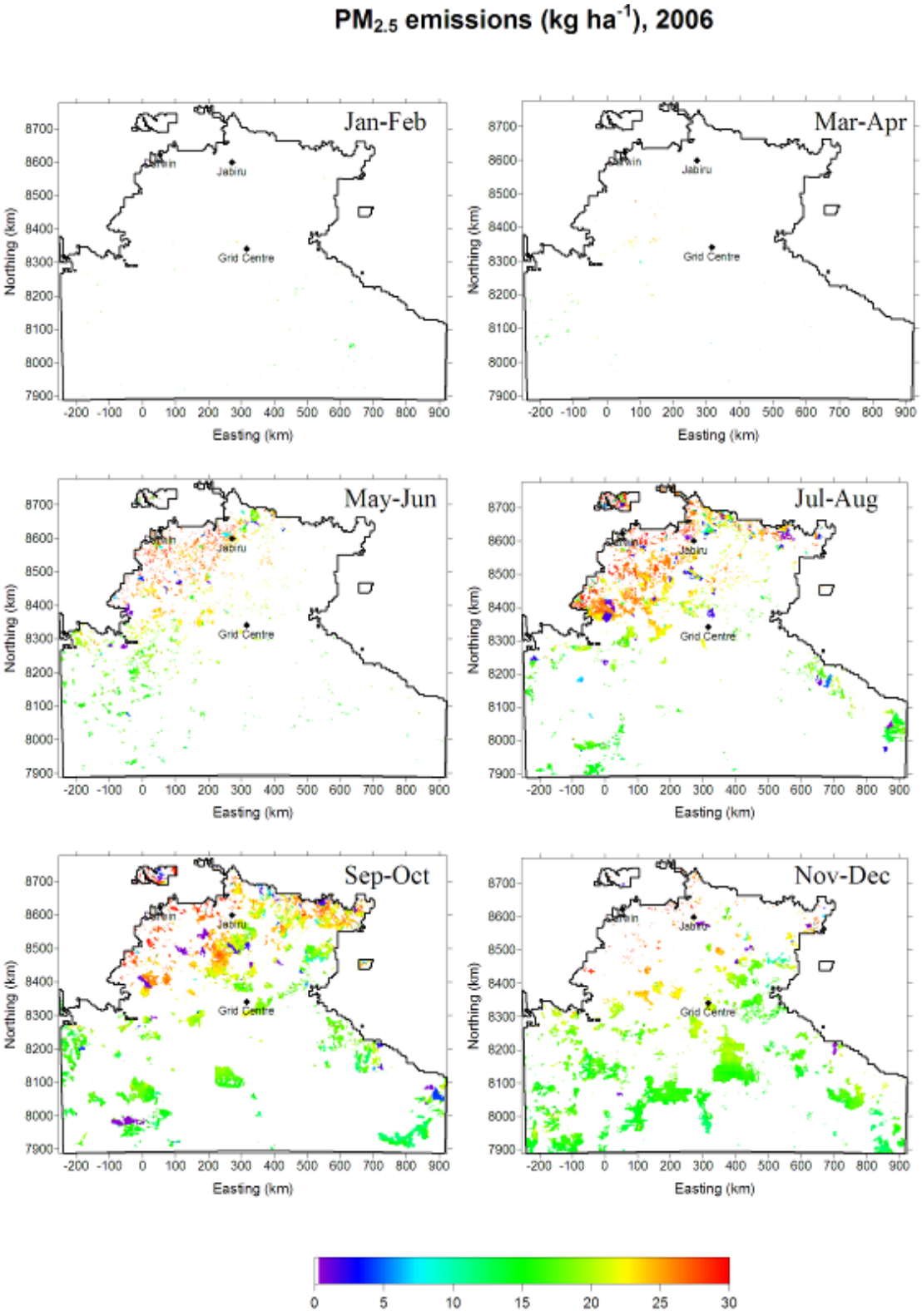


Figure 8-7: Spatial distribution of total $PM_{2.5}$ emissions (kg ha^{-1}) from biomass burning in the Top End at a resolution of 1 km × 1 km for two-monthly periods of the year 2006.

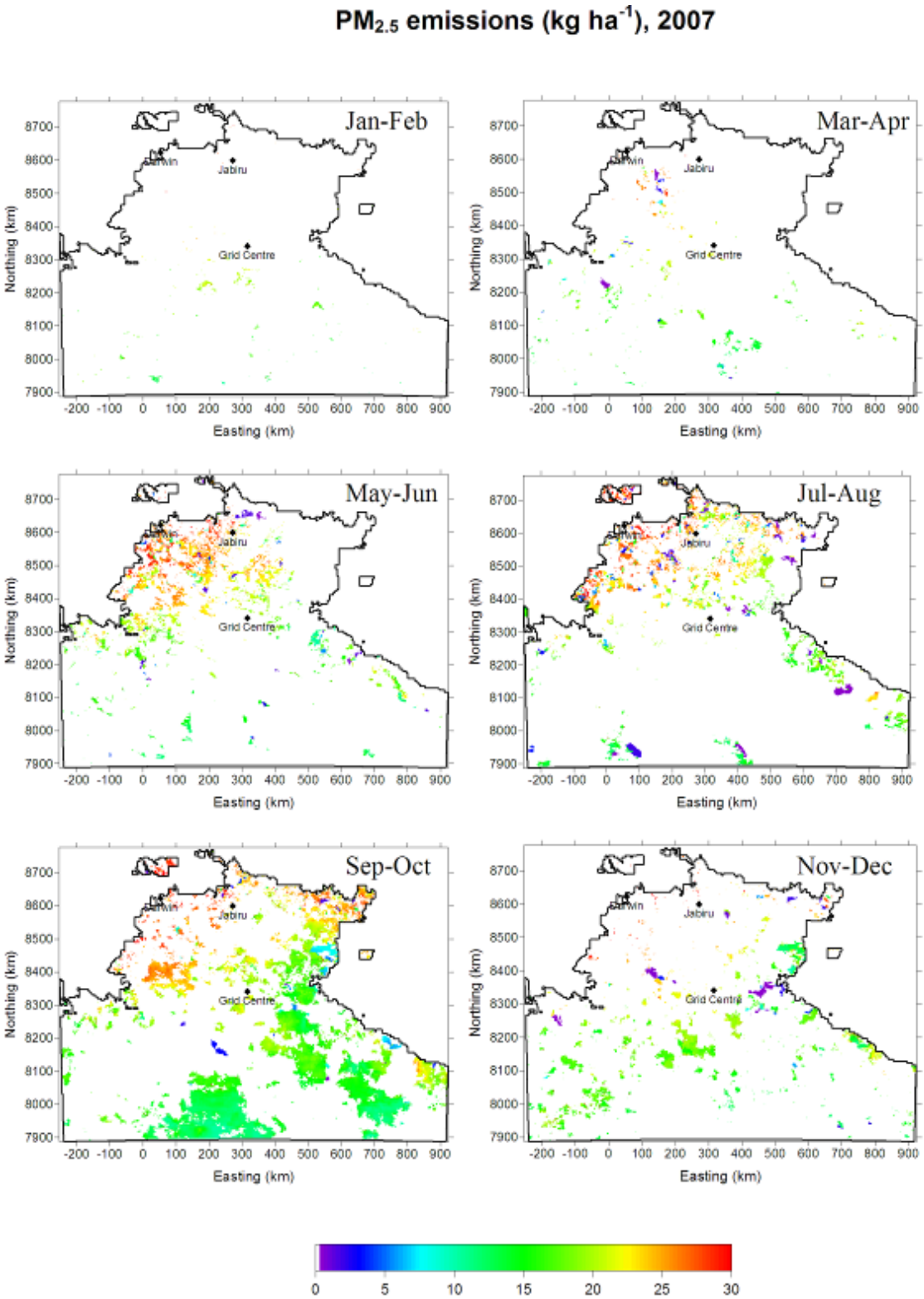


Figure 8-8: Spatial distribution of total $PM_{2.5}$ emissions (kg ha^{-1}) from biomass burning in the Top End at a resolution of 1 km × 1 km for two-monthly periods of the year 2007.

8.7 Modelling aerosol concentrations

In this section, we apply a three-dimensional prognostic meteorological and transport model called TAPM (Hurley et al. 2005) to predict aerosol concentrations across the Top End. In an earlier study, TAPM was validated for meteorology and dispersion for the dry season of 2004 by Meyer at al. (2008) and Luhar et al. (2008).

8.7.1 TAPM

TAPM (v3.5) is a three-dimensional, prognostic meteorological and air pollution model (see 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, typically down to a resolution of 1 km, such as sea breezes and terrain induced flows. Other inputs to the model include global databases of terrain height, land use, and monthly sea-surface temperature.

The meteorological component of TAPM solves a set of momentum equations for the horizontal wind components, the incompressible continuity equation for the vertical velocity in a terrain-following coordinate system, and scalar equations for potential virtual temperature, specific humidity of water vapour, cloud water and rain water. Pressure is determined from the sum of hydrostatic and optional (not used here) non-hydrostatic components. The model incorporates explicit cloud microphysical processes. A vegetative canopy, soil and urban land-use scheme is used to predict energy partitioning at the surface, while radiative fluxes, both at the surface and at upper levels, are also included. Boundary conditions for the turbulent fluxes are determined by Monin-Obukhov surface-layer scaling variables and parameterisations for stomatal resistance. The turbulence levels in TAPM are determined by solving prognostic equations for turbulent kinetic energy and eddy dissipation rate. We neglect any modifications to atmospheric motions as a result of biomass burning.

The air pollution transport component of TAPM consists of an Eulerian grid-based set of conservation equations for species for determining a spatially explicit

distribution of time varying ground-level pollutant concentrations, either using the default Eulerian grid-based approach (used here) or a Lagrangian particle approach targeted at important point sources. The air pollution component was run coupled with the meteorological component.

The performance of TAPM has been verified in a number of previous meteorological and dispersion studies, e.g. Luhar and Hurley (2003), Hurley (2006), and Luhar et al. (2006).

8.7.2 Model setup

We select a horizontal model domain of 1000 km × 1000 km centred at the location (133°17′E, 15°0′S), which corresponds to 315.421 km E and 8340.952 km N in the Australian Map Grid (AMG84) coordinate system, with a grid resolution of 10 km × 10 km (i.e. 101 × 101 grid points). TAPM is generally not suitable for larger domains because of the neglect in the model of the curvature of the earth. The largest domain that can be specified is 1500 km × 1500 km. It was not computationally feasible to select a finer resolution than 10 km, given the large size of the selected model domain. For typical air pollution applications where the domain of interest is much smaller, e.g. a city or an area around an industrial facility, TAPM is applied with multiple grid nesting, which enables a better spatial resolution as well as a more realistic imposition of meteorological and dispersion boundary conditions. However, this was not possible in our application due to the large size of the area of interest. The vertical model levels are staggered, with the lowest ten of the 20 vertical levels used being 10, 50, 100, 150, 200, 300, 400, 500, 750 and 1000 m above the ground level and the highest model level being 8 km. The model default databases of soil properties, topography, and the monthly sea-surface temperature were used (see http://www.cmar.csiro.au/e-print/open/hurley_2005e.pdf). The values of the deep soil volumetric moisture content for the months January–December were 0.2, 0.2, 0.15, 0.1, 0.05, 0.05, 0.05, 0.05, 0.05, 0.05, 0.1 and 0.15, respectively, with a value of 0.05 reflecting dry conditions. The input synoptic meteorological fields were obtained from the Australian Bureau of Meteorology's GASP (Global AnalySis and Prediction) analyses given at 6-hourly intervals with a horizontal grid resolution of 1º × 1º.

The input hourly $PM_{2.5}$ emissions rates (g m^{-2} s^{-1}) determined at a resolution of 1 km × 1 km were averaged over the model resolution. It was assumed that biomass burning is the only source of $PM_{2.5}$ in the Top End.

TAPM (v3.5) was run with the gravitational settling and deposition (with no chemistry) option, but any chemical transformation (e.g. secondary organic aerosol generation) were not considered. This is in contrast to the simulations by Luhar et al. (2008) who treated $PM_{2.5}$ as a tracer.

It difficult to explicitly account for the effects of plume rise associated with the strong buoyancy of hot gases and particles from vegetation fires in a large-scale numerical model when the emission sources are given in gridded form. TAPM can explicitly account for the buoyant rise of a point source plume (e.g. an isolated forest fire) but not of grid-based emissions. In its default mode, the model mixes the gridded surface emissions over the first vertical level (i.e. 10 m). To approximately account for the buoyant plume rise, we mixed the surface emissions up to the third model level (i.e., 100 m), however the improvement to the fit beteen observed and measured surface concntrations of PM_{10} and $PM_{2.5}$ was minor. Because the dominant fuel in the Top End is fine fuel (i.e. grass, leaf litter and twigs less than 6 mm in diameter), the associated buoyancy flux (and hence the plume rise) is generally much lower than in the case of a forest fire (Freitas et al., 2006).

8.7.3 Observed aerosol concentrations

Measurements of aerosols made at the Casuarina Campus site of the Charles Darwin University for the period 1 May – 31 October 2007 were used for comparison with the model simulations. At the site, which is about 12 km north-east of the city of Darwin and is termed as Darwin in the following discussion, both $PM_{2.5}$ and PM_{10} were measured using a Partisol Dichotomous air sampler, and PM_{10} was also measured using a Tapered Element Oscillating Microbalance (TEOM).

Figure 8-9a shows the time series of the 24-hour averaged (or daily) concentrations of $PM_{2.5}$ and PM_{10} measured using Partisol, and those of PM_{10} measured using TEOM at Darwin. There are some gaps in the data, and Partisol data subsequent

to 12 September 2007 were not available. According to the Partisol data, the advisory Air NEPM of 25 μg m^{-3} for the daily $PM_{2.5}$ is exceeded thrice, while the Air NEPM of 50 μg m^{-3} for the daily for PM_{10} is exceeded once.

In the scatter plot in Figure 8-9b, there is a very good correspondence between the daily PM_{10} concentrations measured using Partisol and those measured using TEOM (r^2 = 0.91 and slope of the best fit line = 0.93). The scatter plot in Figure 8-9c suggests the following best-fit relationship between the daily $PM_{2.5}$ concentrations measured using Partisol and the corresponding PM_{10} concentrations:

$$PM_{10} = 1.12\,PM_{2.5} + 5.9\,. \qquad (1)$$

The above relationship is very similar to $PM_{10} = 1.13\,PM_{2.5} + 6.9$ (r^2 = 0.88) reported by Luhar et al. (2008) between the observed $PM_{2.5}$ and PM_{10} concentrations at Darwin for the dry seasons of the years 2004 and 2005. The tight correlation between PM_{10} and $PM_{2.5}$ with a regression slope close to unity suggests that the aerosol is primarily due to biomass burning. The regression slope of 1.12 indicates that around 12% of PM_{10} is not included in the $PM_{2.5}$ fraction, or equivalently, that on average 88% of PM_{10} due to biomass burning consists of $PM_{2.5}$. The intercept value of 5.9 μg m^{-3} can be considered as a background PM_{10} component in the size range 2.5 – 10 μm that may be due to non-biomass burning sources such as maritime haze and mineral dust. Based on the above relationship, one can assume that the PM_{10} emission factor for the savanna and grassland is 1.12 × 4.54 = 5.08 g per kg of dry fuel burned.

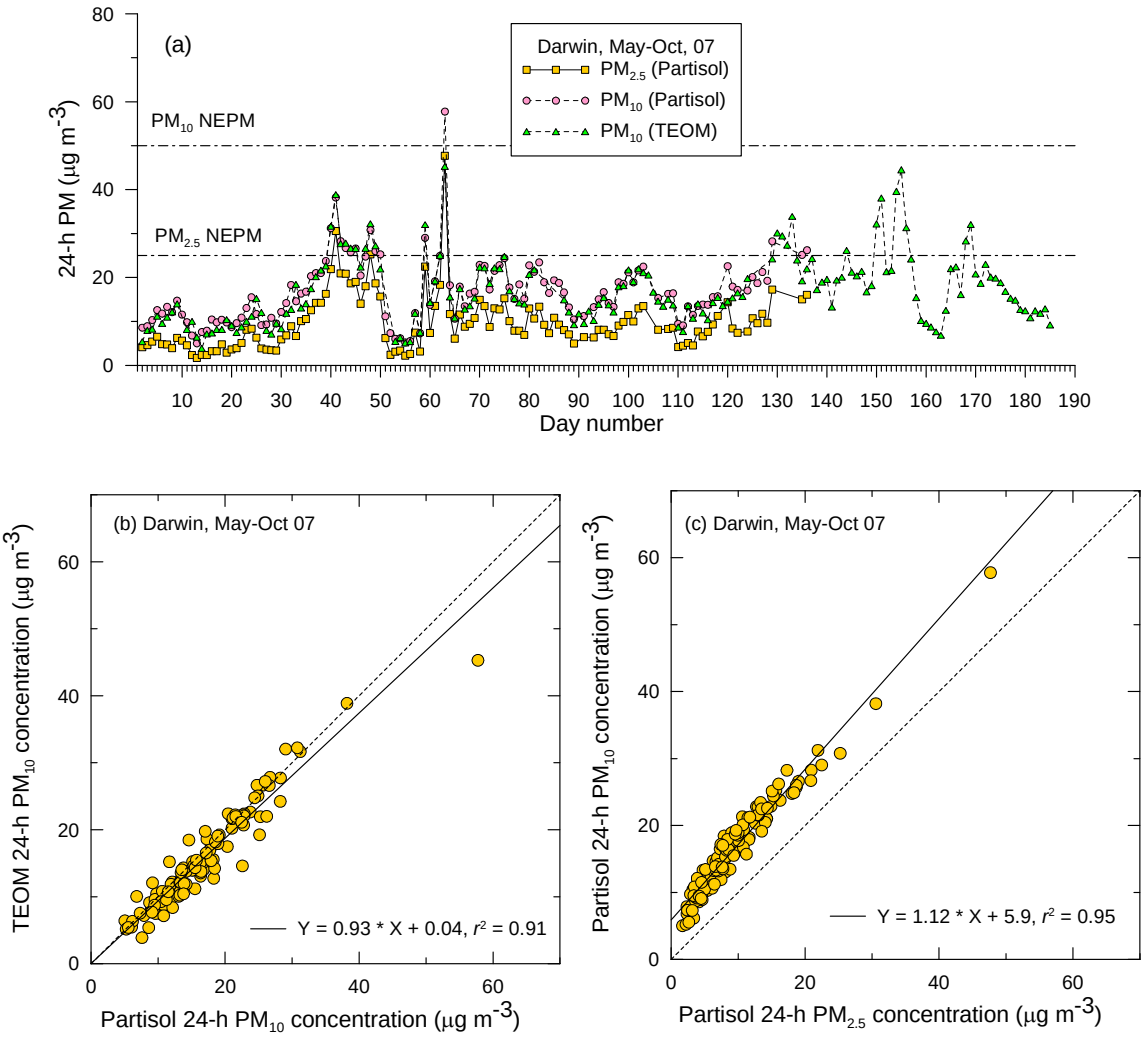


Figure 8-9: (a) Time series of the 24-hour averaged (or daily) concentrations of $PM_{2.5}$ and PM_{10} measured using Partisol, and those of PM_{10} measured using TEOM at Darwin; (b) scatter plot of the 24-hour averaged PM_{10} concentrations from Partisol vs. those from TEOM (sample size N = 111); (c) scatter plot of the 24-hour averaged $PM_{2.5}$ concentrations vs. the corresponding PM_{10} concentrations from Partisol (N = 123).

8.7.4 Comparison with the modelled $PM_{2.5}$ concentrations

In Figure 8-10a, the modelled time series of the daily $PM_{2.5}$ concentration is in good agreement with the observed one, with the model performing reasonably well at predicting the observed peaks.

The model tends to underpredict the lower concentrations, which may be partly because non-biomass burning sources such as background maritime haze, mineral dust and other pollution sources (e.g. motor vehicles), although much less dominant, also

contribute to $PM_{2.5}$ concentrations. In addition, possible generation of secondary organic aerosols, neglected in the modelling, may be a significant contributor. Less likely reasons include the inability of the model to account for the possible recirculation of particular matter once it leaves the model domain and for contributions from sources outside the model domain; a larger-scale model will need to be considered to address these issues.

The model overpredicts some of the higher concentration values, and as a result gives a higher number of exceedences of the Air NEPM than the observed: 8 as opposed to 3.

Departure of the modelled wind direction from the observed one, which may be due to model approximations and parameterisations as well as the quality of input data including synoptic analyses, can also result in a modelled concentration being substantially different from the observed one at the same time and location, especially when only a few scattered sources are present. The resolution (10 km × 10 km) used in the model is rather coarse, and a finer resolution may have improved the predictions. A proper accounting of plume rise in the model may have also led to better predictions.

The scatter plot in Figure 8-10b between the observed and modelled concentrations suggests a good performance by the model. The correlation coefficient is $r = 0.69$ and the slope of the best fit line is 0.92. This model performance is better than that reported in Luhar et al. (2008) for the year 2004.

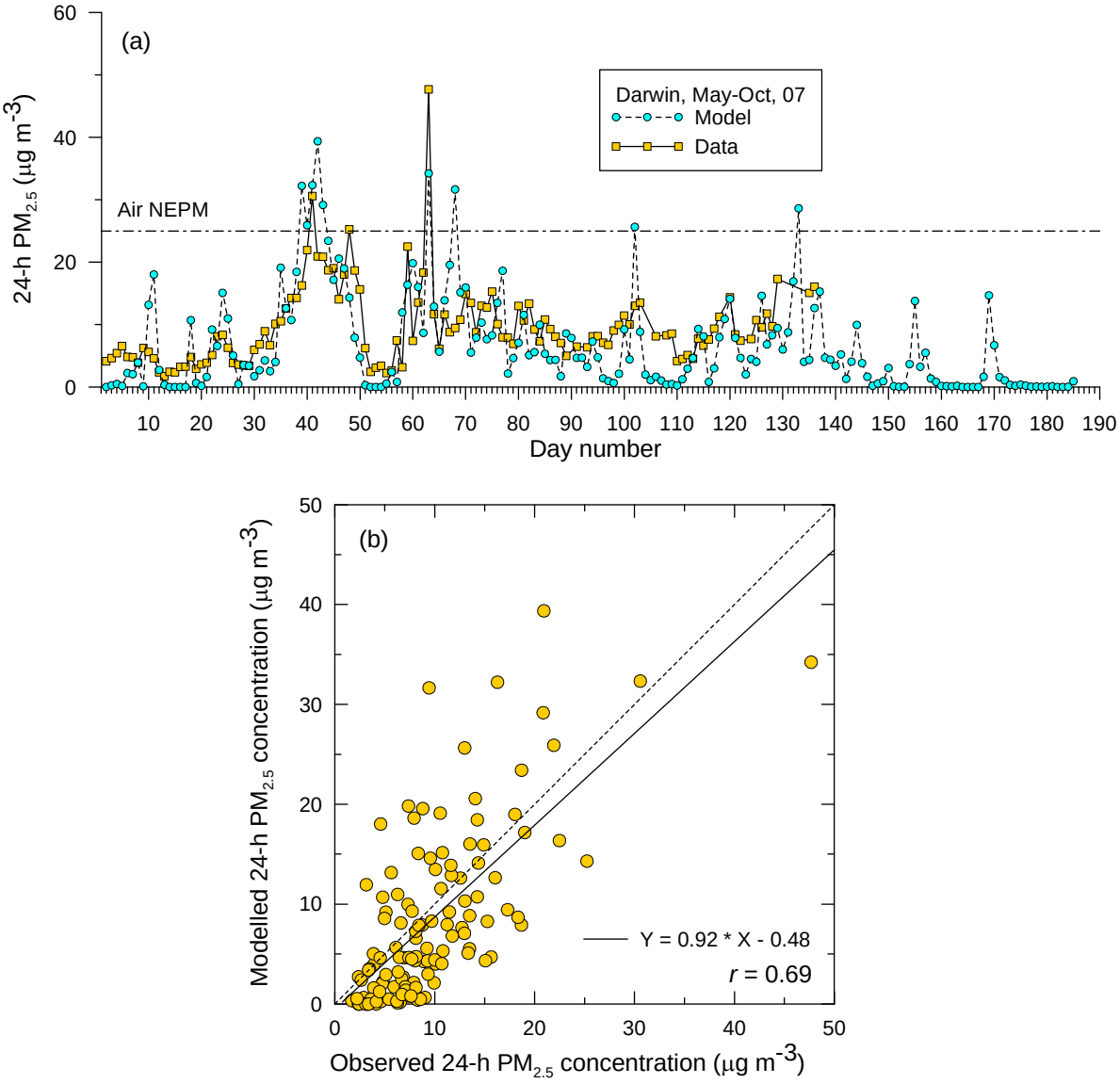


Figure 8-10: (a) Time series of the observed and modelled 24-hour averaged $PM_{2.5}$ concentrations at Darwin for the period 1 May – 31 October 2007, and (b) scatter plot of the observed and modelled 24-hour averaged $PM_{2.5}$ concentrations.

8.7.5 Comparison with the modelled PM_{10} concentrations

Figure 8-11a presents the observed time series of daily PM_{10} together with the modelled variation determined using a PM_{10} emission factor of 5.08 g per kg of dry fuel burned and a background PM_{10} level of 5.9 $\mu g\ m^{-3}$ (Equation (1)). The model performance is good, especially during May–August. A scatter plot of the observed (Partisol) and modelled PM_{10} concentrations is shown in Figure 8-11b.

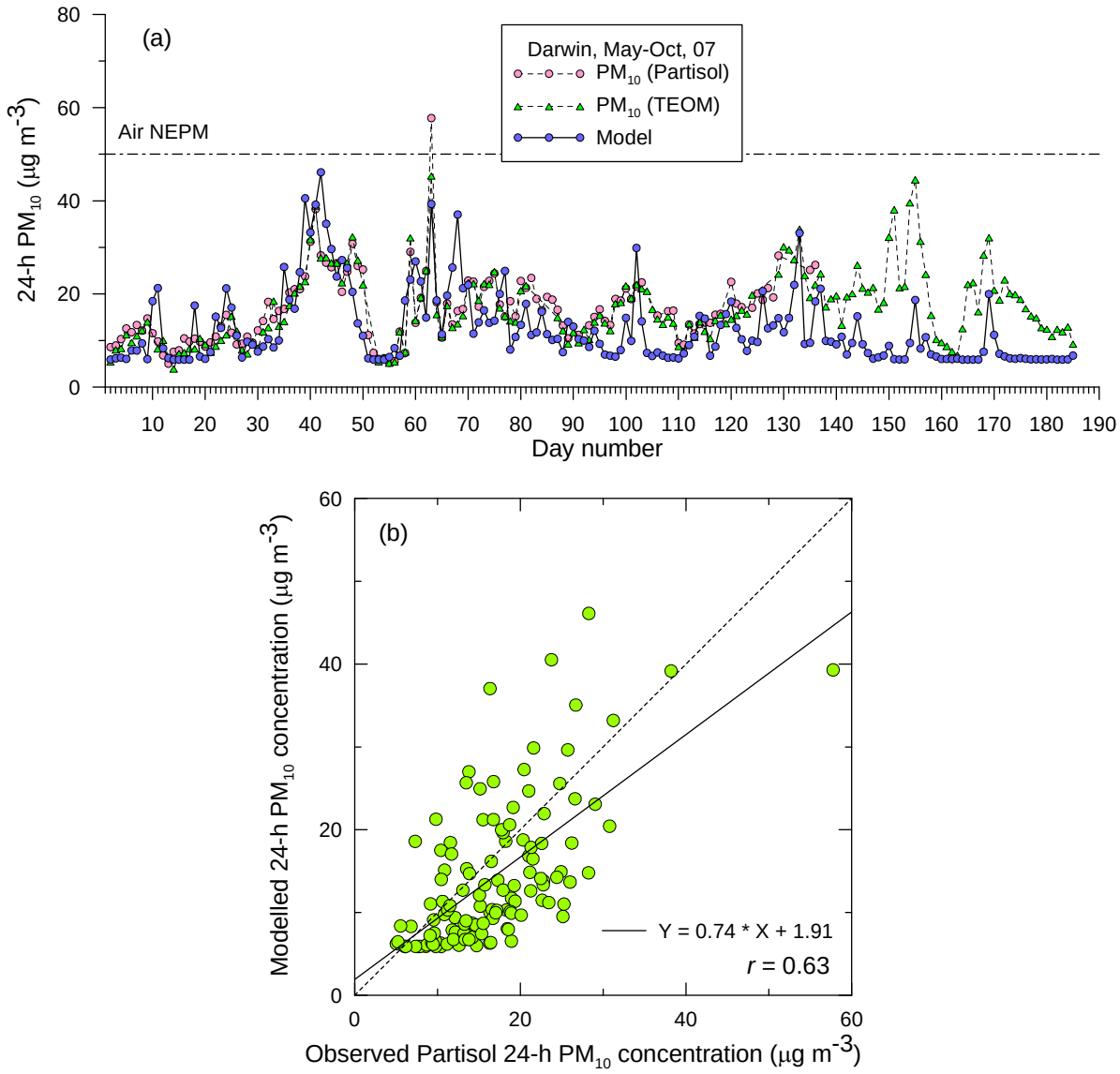


Figure 8-11: (a) Time series of the observed and modelled 24-hour averaged PM_{10} concentrations at Darwin for the period 1 May – 31 October 2007, and (b) scatter plot of the observed (Partisol) and modelled 24-hour averaged PM_{10} concentrations.

8.7.6 Modelled $PM_{2.5}$ exceedences

Given the good performance by the model at simulating the aerosol fields near Darwin, one can go further and examine the modelled aerosol distribution over the whole Top End. In particular, in Figure 8-12 to Figure 8-16 we present the distribution

of the modelled number of exceedences of the Advisory NEPM for the daily $PM_{2.5}$ for each of the two-monthly periods during 1 May – 31 October.

In Figure 8-12 for the year 2003, the exceedences during May–June mainly occur over the north-west. This is also true for the subsequent two months (Figure 8-12b), but with a smaller number of exceedences. For the period September–October (Figure 8-12c), the exceedences are more extensively distributed, but they are not as intensive as for May–June.

For the year 2004 (Figure 8-13), the exceedences are mostly concentrated over the north-west of the Top End for May–August, but they become much more widespread and intense in the subsequent two years.

For the year 2005 (Figure 8-14), the period May–June has the largest number of exceedences, which are concentrated over the north-west.

In Figure 8-15 for the year 2006, there are relatively fewer exceedences during May–June. However, they become more numerous during the next two months, mostly concentrated over the north-west. For September–October, the exceedences are more widely distributed.

In the year 2007 (Figure 8-16), the north-west encounters the most number of exceedences during May–August, while the period September–October has the most extensive distribution of exceedences.

In the above figures, the distribution of exceedences is consistent with the spatial coverage and intensity of the emissions presented in Figure 8-4 to Figure 8-8. It is also apparent that in the north-west, which includes Darwin, most exceedences occur during early to mid part of the burning season. One reason for the high number of exceedences in the north-west is its high fuel load.

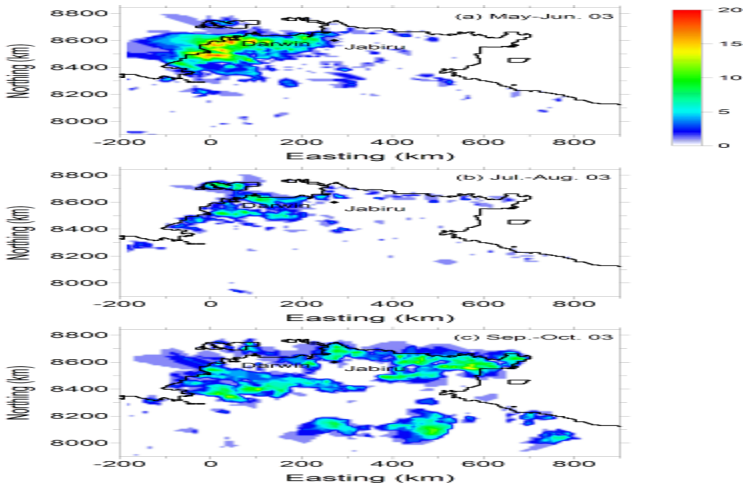


Figure 8-12: Modelled distribution of the number of exceedences of the Advisory NEPM (of 25 μg m^{-3}) for the daily mean $PM_{2.5}$ concentration for three two-monthly periods during 1 May – 31 October 2003.

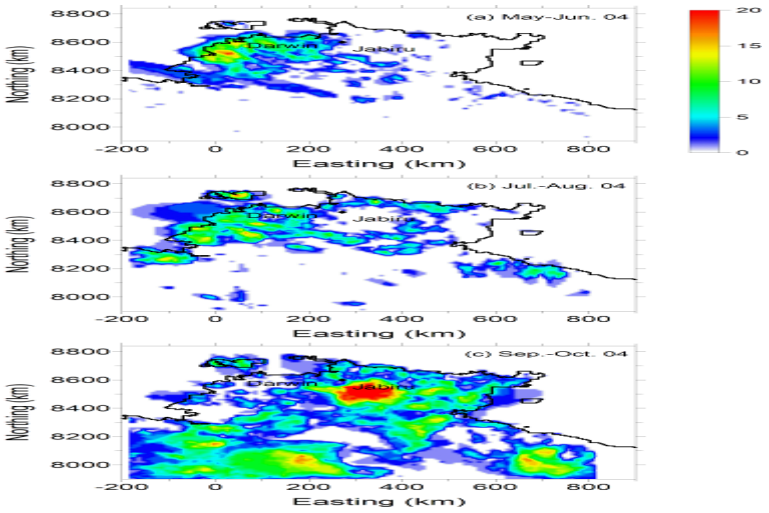


Figure 8-13: Modelled distribution of the number of exceedences of the Advisory NEPM (of 25 μg m^{-3}) for the daily mean $PM_{2.5}$ concentration for three two-monthly periods during 1 May – 31 October 2004.

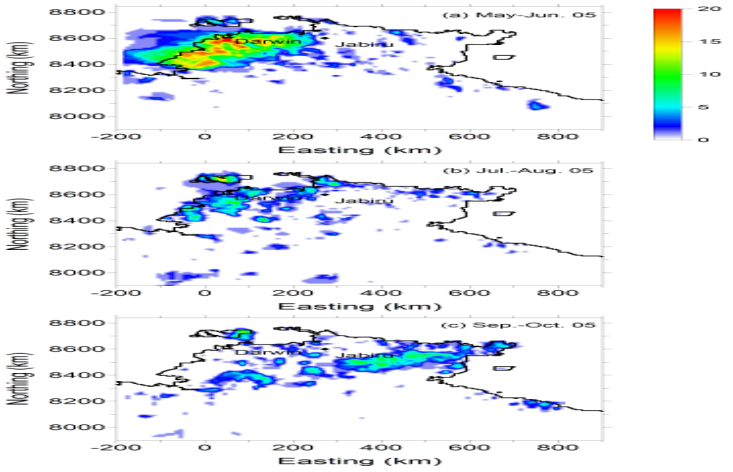


Figure 8-14: Modelled distribution of the number of exceedences of the Advisory NEPM (of 25 μg m^{-3}) for the daily mean $PM_{2.5}$ concentration for three two-monthly periods during 1 May – 31 October 2005.

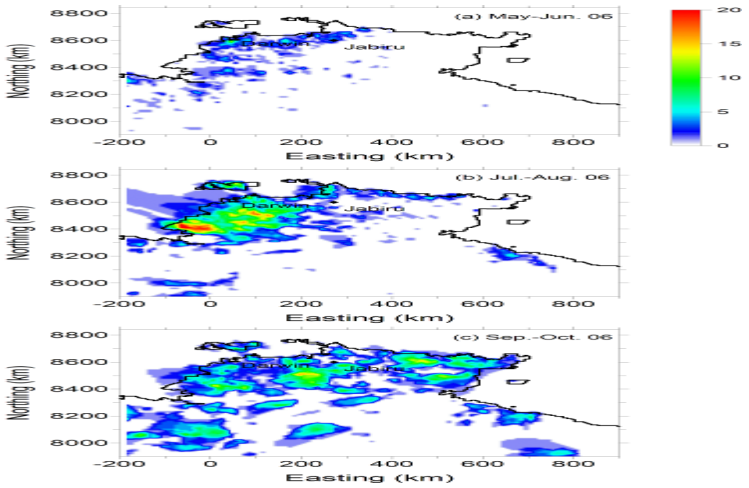


Figure 8-15: Modelled distribution of the number of exceedences of the Advisory NEPM (of 25 μg m^{-3}) for the daily mean $PM_{2.5}$ concentration for three two-month periods during 1 May – 31 October 2006.

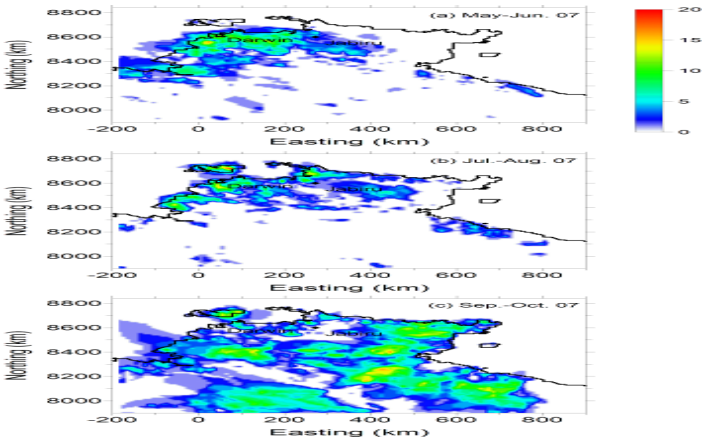


Figure 8-16: Modelled distribution of the number of exceedences of the Advisory NEPM (of 25 μg m^{-3}) for the daily mean $PM_{2.5}$ concentration for three two-month periods during 1 May – 31 October 2007.

The modelled numbers of exceedences of the Advisory NEPM for the daily $PM_{2.5}$ for Darwin and Jabiru are given in Table 8-2. The highest number of exceedences at Darwin for the period considered (i.e. May–October) was 11 in the year 2007, whereas the lowest was 3 in the year 2006. At Jabiru, highest number of exceedences was 11 in the year 2004, and the lowest was 2 in the year 2005. At Darwin, the highest exceedences are determined to be in the early part (May–June) of the period modelled, except for the year 2006, whereas there is no such trend at Jabiru.

Table 8-2. Modelled number of exceedences of the Advisory NEPM (of 25 μg m^{-3}) for the daily mean $PM_{2.5}$ concentration.

Year	Months	Darwin	Jabiru
2003	May–Jun.	9	1
	Jul.–Aug.	1	0
	Sep.–Oct.	0	2
2004	May–Jun.	5	3
	Jul.–Aug.	3	0
	Sep.–Oct.	0	8
2005	May–Jun.	6	1
	Jul.–Aug.	1	1
	Sep.–Oct.	0	0

2006	May–Jun.	0	2
	Jul.–Aug.	3	1
	Sep.–Oct.	0	0
2007	May–Jun.	8	3
	Jul.–Aug.	3	2
	Sep.–Oct.	0	1

8.8 Modelling secondary aerosols and ozone concentrations

The model simulations described in the previous section assume that $PM_{2.5}$ is a stable tracer with no formation mechanisms following emission and losses occurring only through deposition. This is a simplification of reality, because, clearly, reactive chemistry does proceed during transport leading to formation or destruction of gaseous species and formation of secondary aerosol. Ignoring these can lead to underestimating the $PM_{2.5}$ emissions, underestimating the impact of smoke on the background rural air quality, and unjustifiably concluding that goodness of fit between observed and modelled $PM_{2.5}$ concentrations implies accurate emissions estimates. In this section, we apply a three-dimensional weather and air pollution modelling system TAPM-CTM that includes chemical reactions to predict aerosol and ozone concentrations across the Top End.

The system comprises the following components.

- The same emission inventory is used as with TAPM run except it includes natural aerosol such as sea salt and biogenic emissions.
- A numerical weather prediction system, CSIRO's TAPM (Hurley 2005), which was used for the prediction of meteorological fields including wind velocity, temperature, water vapour mixing ratio, radiation and turbulence.
- A chemical transport model (CTM; Cope et al., 2004), which was coupled with the Carbon Bond 2005 (CB5; Yarwood et al. 2005) mechanism for modelling photochemical transformation.

- A secondary organic aerosol (SOA) module based on components of the methodology documented in Pun and Seigneur (2007) and references therein. The partitioning coefficients recommended by Pun and Seigneur were adjusted to provide a best fit to the SOA formation rates observed in the CSIRO environmental smog chamber
- The Model for an Aerosol Reacting System (MARS; Saxena et al. 1986) for modelling secondary inorganic $PM_{2.5}$ production was employed.

Combining the CTM with TAPM allows us to predict the impact of smoke on the background air quality of the region. Figure 8-17 shows the observed 24-h $PM_{2.5}$ concentrations in Darwin from May to October 2007 against modelled $PM_{2.5}$ concentrations using TAPM with and without CTM. Modelled $PM_{2.5}$ concentrations are higher for TAPM with CTM compared to TAPM without CTM indicating that at least some of the $PM_{2.5}$ mass observed in Darwin may be secondary aerosol formed in the smoke plume during transport, and is relatively more significant at higher atmospheric $PM_{2.5}$ concentration. This is shown more clearly in Figure 8-18 where secondary aerosol formation is significant only during major smoke events such as days 10, 40, 61 and 65 of the daily time series. On days of lower $PM_{2.5}$ concentrations the models tend to under predict in comparison to the observed concentrations.

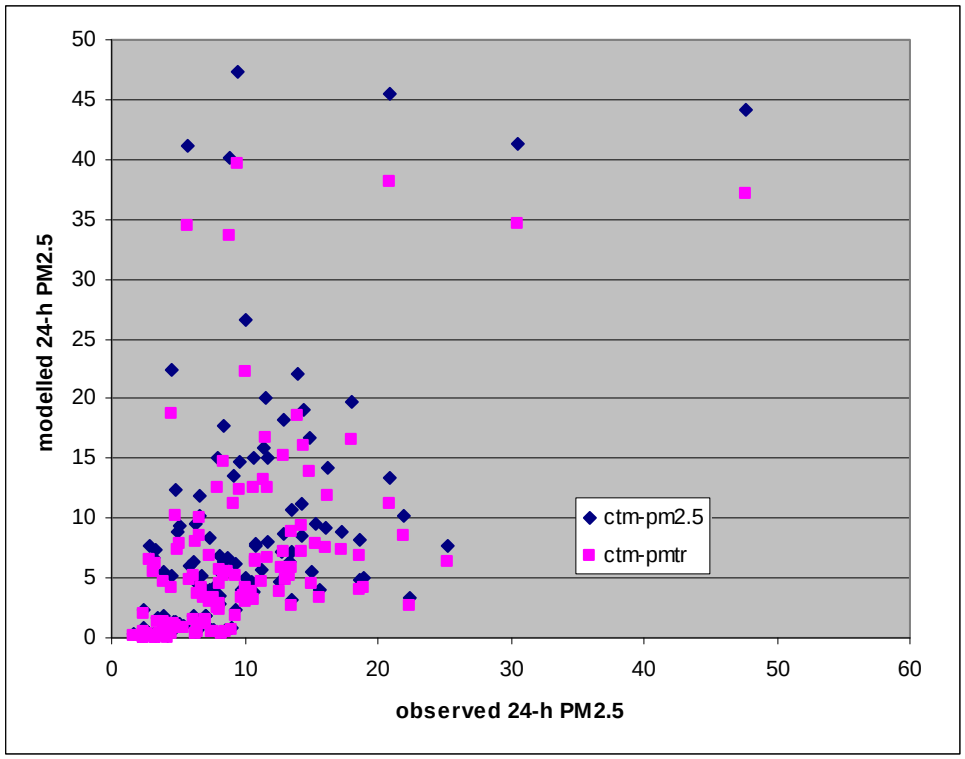


Figure 8-17: Comparison plot of CTM $PM_{2.5}$ (ctm-pm2.5) of all sources with chemical reactions and CTM $PM_{2.5}$ as tracer (ctm-pmtr) without any chemical reactions

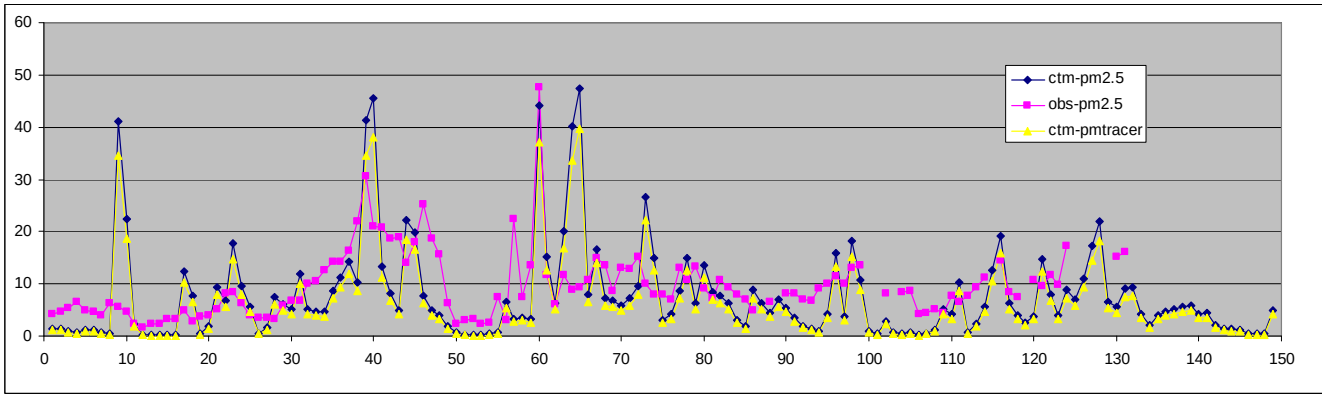


Figure 8-18: Time series of the observed and modelled 24-hour averaged concentrations of $PM_{2.5}$ using TAPM with and without CTM for the period 1 May – 31 October 2007.

The accuracy of the CTM implantation in Darwin is better assessed from the predicted ozone enhancements during smoke incursions which are more easily identified in the air quality time series. There is direct correlation between $PM_{2.5}$ and ozone concentrations (Figure 8-19). This is also seen in the modelled concentrations however in the latter, the ozone levels are significantly over-predicted. This indicates that the current model definition needs further refinement and work is progressing in this area.

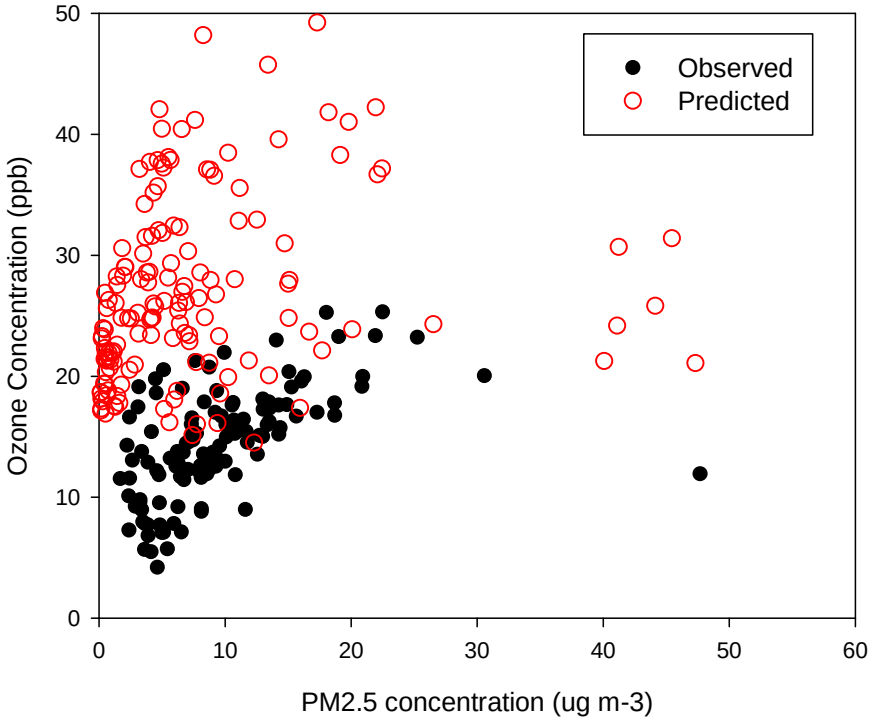


Figure 8-19 Relation between (a) observed 24-hour mean $PM_{2.5}$ and ozone concentrations in Darwin between May and October 2007 (closed circles) and (b) modelled concentration using TAPM + CTM (open circles)

8.9 Conclusions

In the present study, aerosol emissions from biomass burning in the Top End for five years 2003–2007 were calculated at a high spatiotemporal resolution (1 km, 1 1h) using satellite derived fire scars and hot spots, together with a fuel load distribution. The highest $PM_{2.5}$ emissions were for the year 2004 (0.68 Tg) and the lowest for the year 2005 (0.28 Tg). The high resolution of the derived emission distribution enabled modelling of aerosol concentrations in the atmosphere. The modelled $PM_{2.5}$ concentration agreed well with the measured levels near Darwin. The results support the robustness of the emission calculation methodology of Meyer et al. (2008) and Luhar et al. (2008), and reinforce their conclusion that measured hotspots with no associated fire scars must be included in the methodology. The modelling suggests that in the north-west of the Top End, which includes Darwin, most exceedences of the Advisory $PM_{2.5}$ NEPM generally occur during early to mid part of the burning season. One reason for

the high number of exceedences in the north-west is its high fuel load. The highest number of exceedences at Darwin for the period considered (i.e. May–October) was 11 in the year 2007, whereas the lowest was 3 in the year 2006. At Jabiru, highest number of exceedences was 11 in the year 2004, and the lowest was 2 in the year 2005. At Darwin, the highest exceedences are determined to be in the early part (May–June) of the period modelled, except for the year 2006, whereas there is no such trend at Jabiru.

9 GENERAL DISCUSSION

The monitoring program has clearly shown that air quality in rural areas, while generally good, is impacted by smoke from a range of biomass combustion sources. Prescribed fire is a significant source in southern Australia, savanna woodland are a major source in Northern Australia and there is emerging evidence that crop stubble burning might be the source of serious air quality issues in some agricultural regions. At Manjimup the four exceedences of the NEPM were caused by plume strikes, mostly from Spring burns but one occasion from an Autumn fire. Average $PM_{2.5}$ concentrations were substantially higher during the prescribed burning season that at other periods and were higher than average concentrations in Melbourne. The site at Ovens in NE Victoria was similarly affected during the autumn prescribed burning season; with air quality obviously smoke impacted almost 50% of the time. Darwin air quality was affected by savanna fires throughout the dry season, particularly mid season during June and July Beyond Darwin, the modelled spatial distribution of NEPM exceedences suggests some localities, which fortunately are sparsely populated, regularly experience poor air quality throughout the fire season. However, overall, air quality in Manjimup, Ovens and Darwin locations met the $PM_{2.5}$ NEPM.

Wagga Wagga has a more serious issue of fine particle pollution than the other three sites, and the correlation between the seasonal and diurnal timing of the exceedences and stubble burning activity within the regions argues strongly that stubble burning is the cause. Therefore it is clear that biomass burning is impacting the air quality in regional communities with respect to $PM_{2.5}$ and may well be the dominant source of particulate pollution in an otherwise clean environment.

The concentrations of other classes of criteria pollutants such as ozone and VOCs are also determined by fire activity, however ambient concentrations' rarely approach air quality limits and never exceed them.

In summary, then, the general perception that smoke from prescribed burning activities has an impact on the air quality of these rural population centres appears to be valid. This leads to the question of whether there are management options available to reduce the air quality impacts of prescribed burning without eliminating the practice. There are several classes of options:

1. Strategies to reduce the emissions at the source;
2. Strategies to avoid plume strike on population centres; and
3. Strategies to reduce the impact of plume strike on air quality.

The first strategy derives from the determinants of emissions defined in Equations 3-1 and 3-2. Emission factors for most trace species vary with combustion efficiency (MCE = [CO2]/([CO]+ [CO2], Hao et al. 1996; Ward et al. 1996).MCE varies with the rate of oxygen supply, and hence fire intensity. Factor that determine intensity such as fuel class , fuel moisture, fuel load and fuel density affect MCE and hence emission factors. Grasses tend to have high MCE (~0.95), whereas forest fuels have lower MCE (~ 0.9: Ward et al. 1996; Andreae and Merlet 2001). $PM_{2.5}$ emissions increase with lower MCE, i.e. with lower temperature, more smouldering fires. Therefore prescribed burning conducted when fuel moisture content is high, as occurs in Spring are likely to emit $PM_{2.5}$ at higher rates than Autumn burns when the fuel is drier. Higher fuel loads, in contrast, may support more intense fires which might favour lower $PM_{2.5}$ emission rates. Fuel loads are determined, in part by the interval between fires.

Strategies to avoid plume strike on population centres, or to minimise the risk of plumes from several fires combining require accurate forecasts of plume trajectories; these exist and in some cases are available as operational products. Alternatively regional transport models could be used to identify which forest area within a region pose the highest risks for smoke impact on population centres towns in order to focus planning resources effectively. This approach is currently being developed for application in the Riverina to identify potential pollution source regions for Wagga Wagga..

Strategies to reduce the impact of plume strike on air quality, include scheduling burns for periods of lower photochemical activity. Ozone production in plumes from spring fires observed in Manjimup was greater than plumes from Autumn burns. Similarly exceedences on the ozone NEPM in Ovens occurred only during the wildfires in December 2006. The issue is important not only for addressing air quality regulatory requirements, but also for population health issues. Secondary reaction products found in aged smoke plumes may have toxic properties to the primary emission products. For example, secondary aerosol, which is composed predominantly of ultrafine particles, presents different heath risks compared to the fine particles produced during

combustion. Secondary aerosol formation rate is determined by photochemistry and therefore is seasonally dependent (Keywood and Cope, 2008).

The application of these strategies is dependent on the development of air quality transport models appropriate to the region. Such model are being developed and used to test management scenarios (e.g. Tian et al. 2008). TAPM predictions of $PM_{2.5}$ in the NT presented in this study indicates that the development of similar tools for Australian vegetation is feasible. However there are still significant information gaps that need to be addressed including:

- Limited availability of air quality data in rural and remote regions in Australia,
- Limited information on the emission properties of $PM_{2.5}$ and related species from combustion of Australian fuels, particularly the effect of fuel size and fuel moisture and vegetation types; and
- Limited information on the nature of reactive chemistry in smoke plumes and the secondary reaction products.

A useful next step is to apply the current model to other regions in Australia, particularly Vic, WA and NSW, and to extend the model to wildfire emissions. At present the primary concerns of fire managers who plan and execute prescribed burning programs centre on safe fire management, damage to live vegetation and impacts on biodiversity. These already pose a major challenge for balancing incompatible objectives. The strategies suggested above for also addressing air quality issues while presenting additional challenges, fit within the current framework. An immediate requirement is to develop, in conjunction with fire management authorities, sets of practical management scenarios that can be tested for feasibility and impact on regional air quality.

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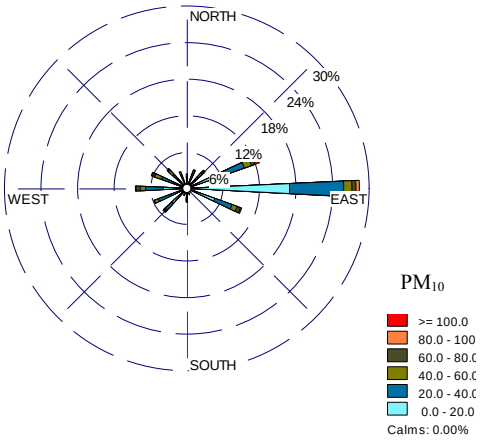
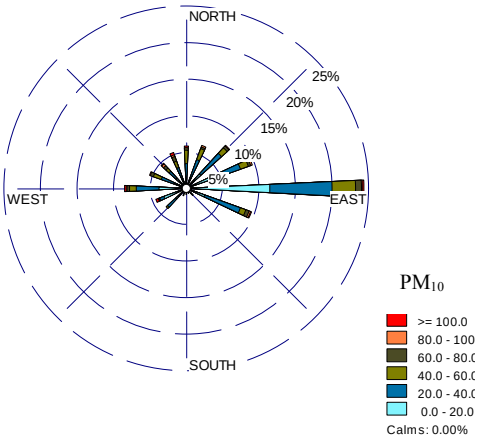
APPENDIX A –

Appendix 1: Pollution roses for Albury and Wagga Wagga – summer, spring and winter 2003-2007

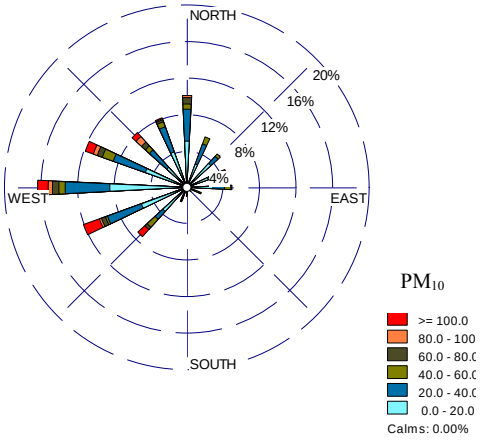
January hrs 0 to 5



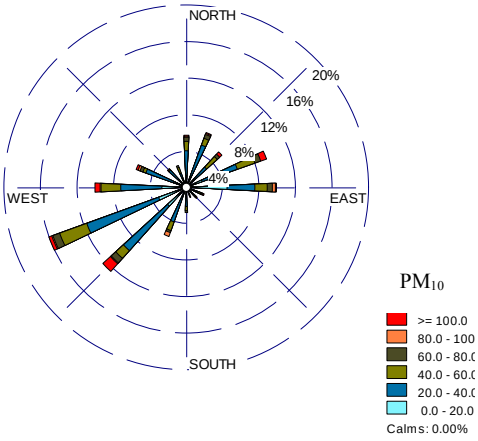
January hrs 6 to 11



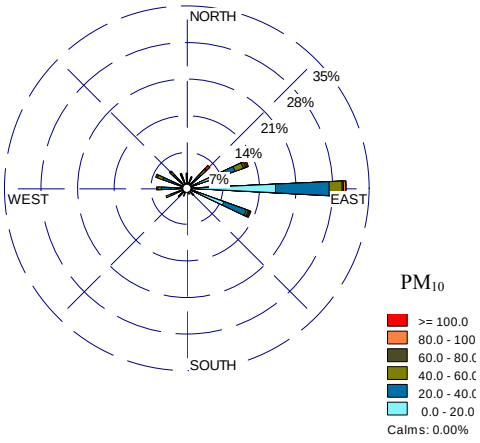
January hrs 12 to 17



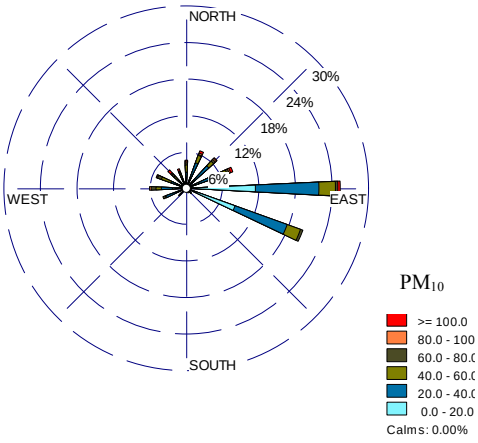
January hrs 18 to 23



February hrs 0 to 5

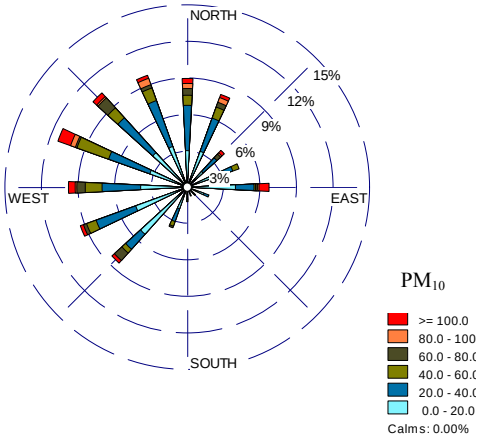


February hrs 6 to 11

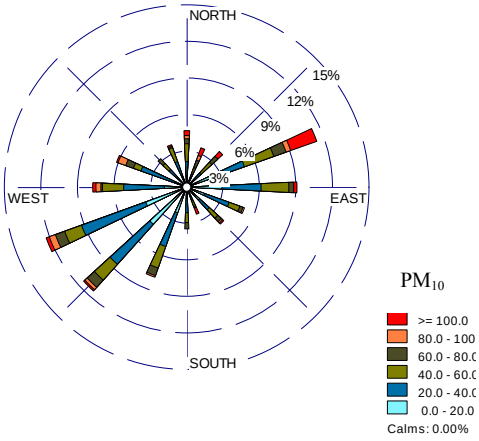


February hrs 12 to 17

February hrs 18 to 23



June hrs 0 to 5



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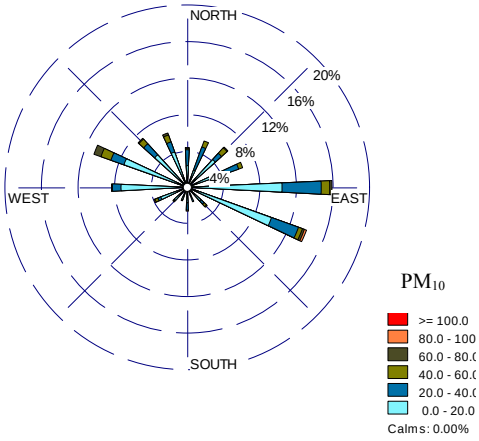
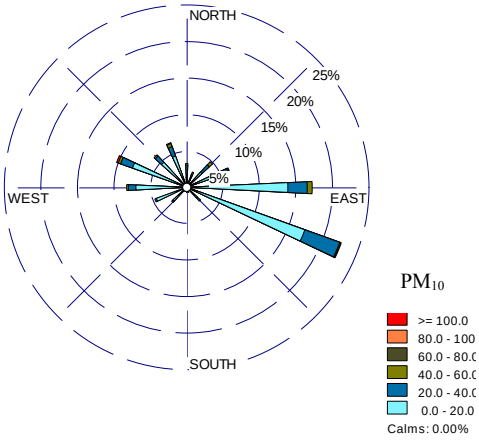


June hrs 12 to 17



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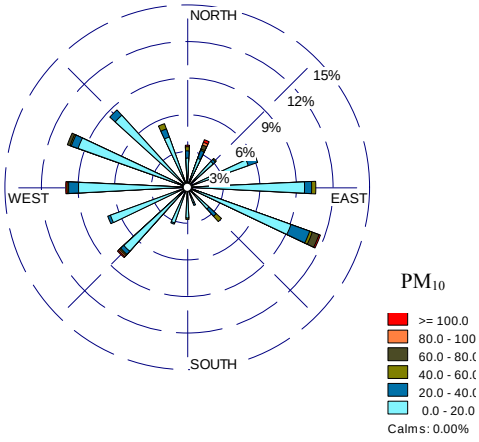

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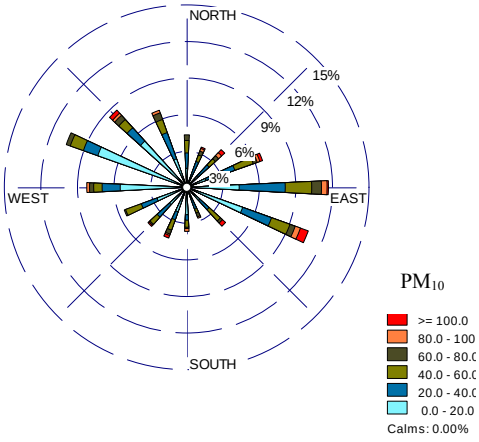

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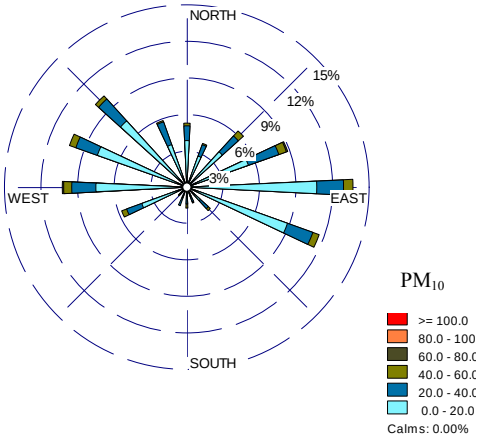

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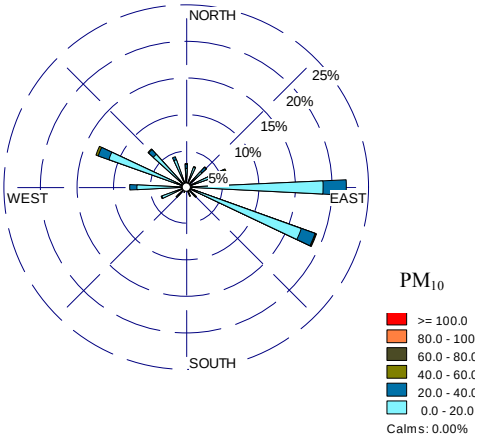
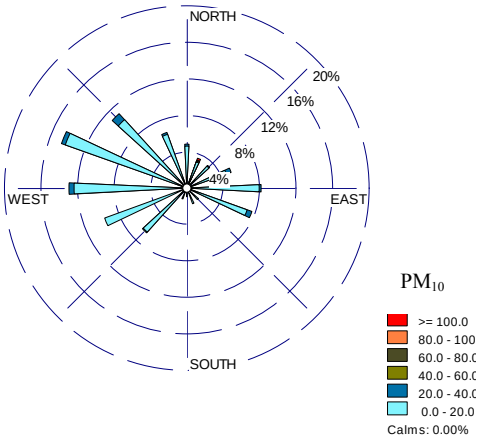
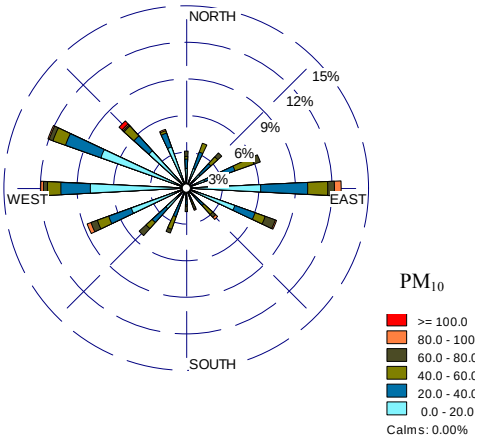


July hrs 18 to 23



August hrs 0 to 5



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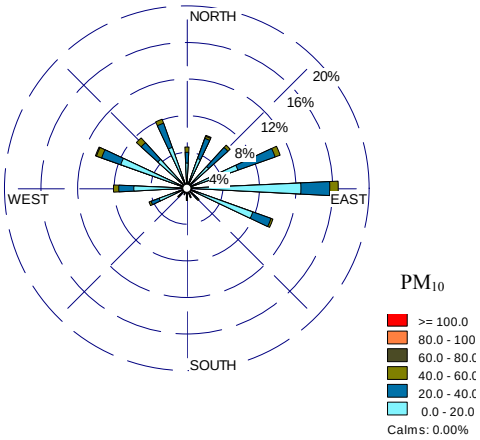

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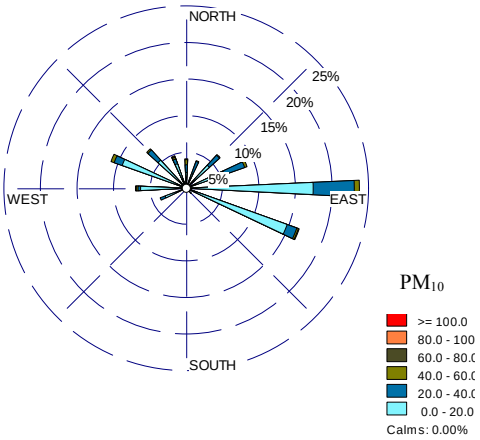

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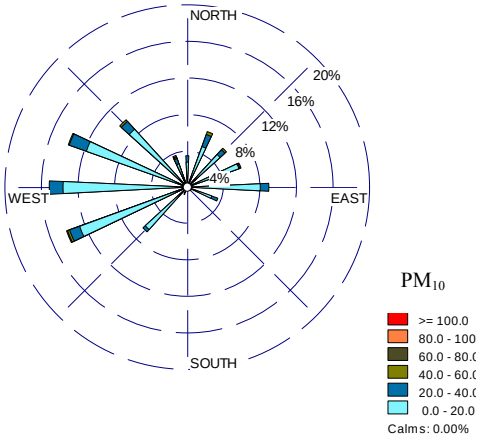

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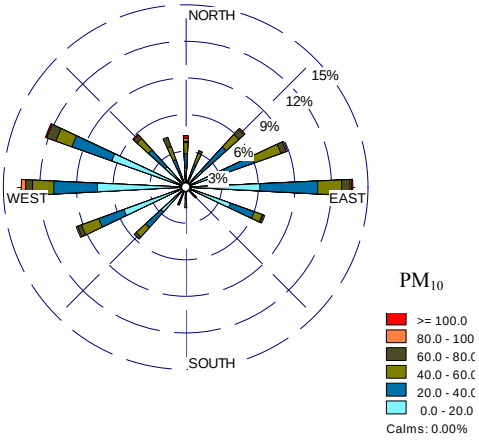

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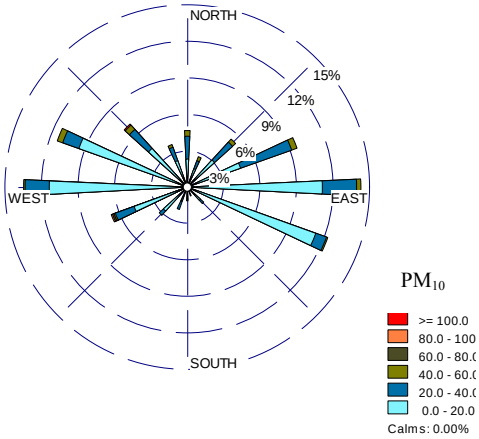

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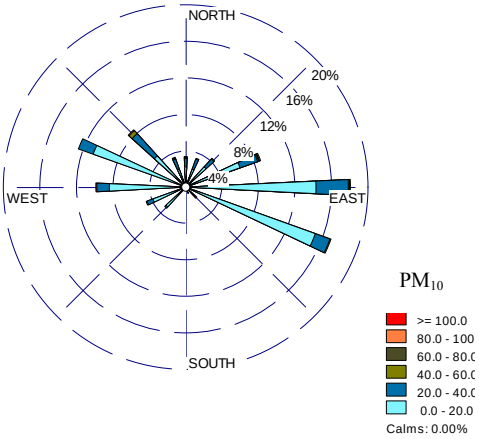

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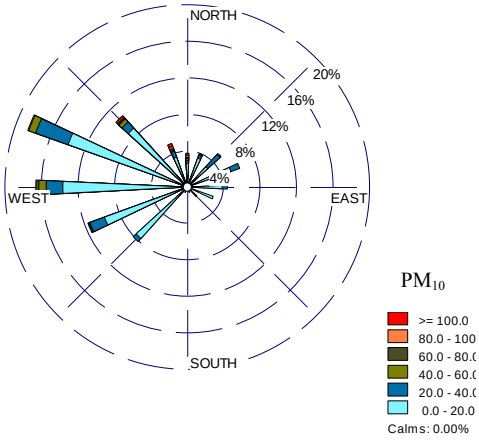

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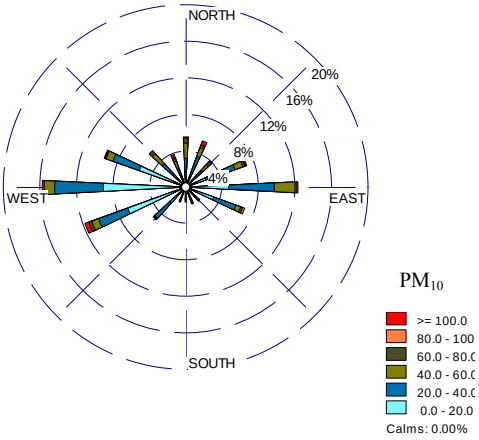

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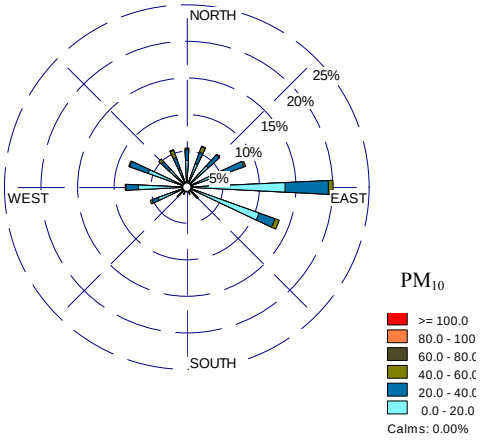

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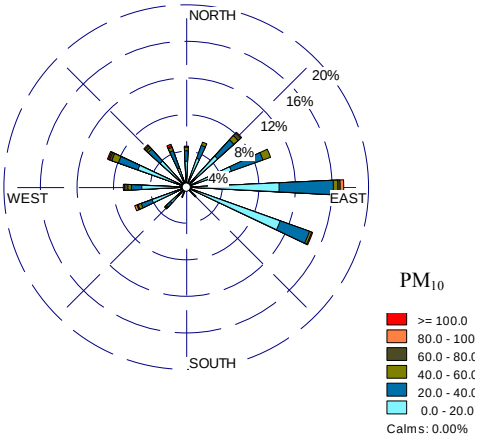

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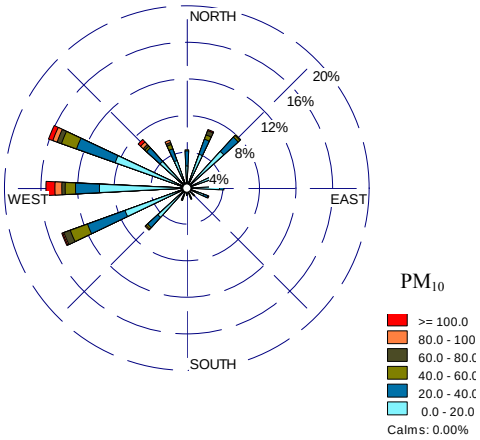

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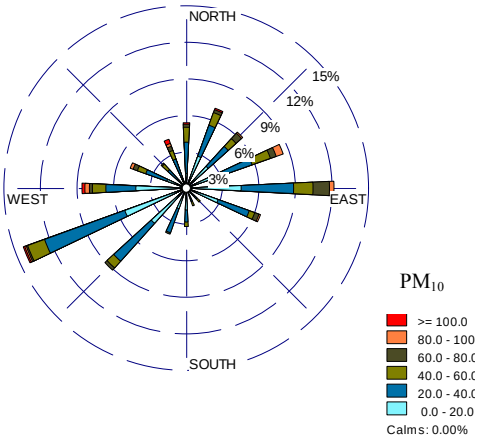

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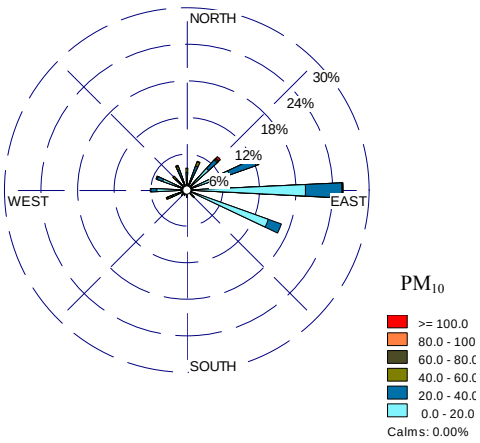

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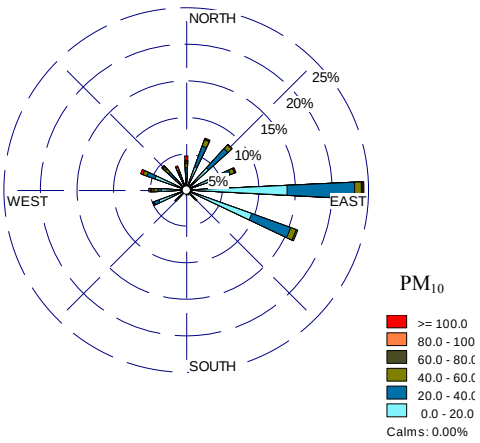

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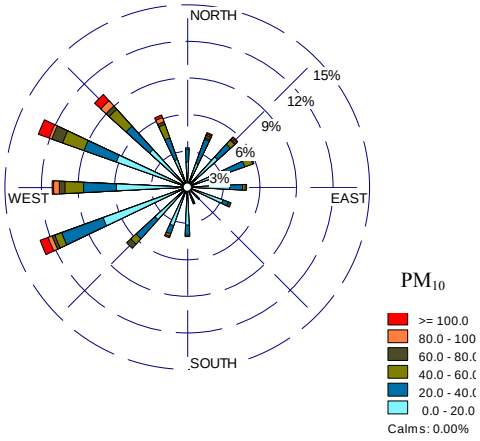

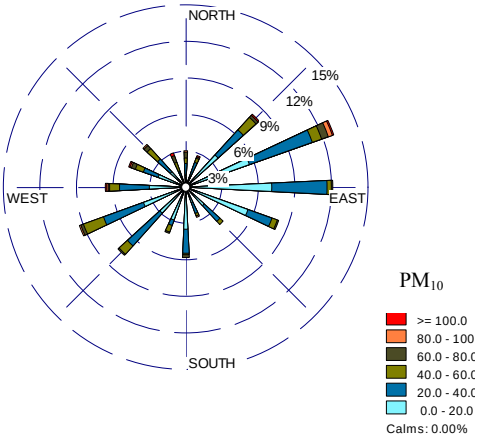

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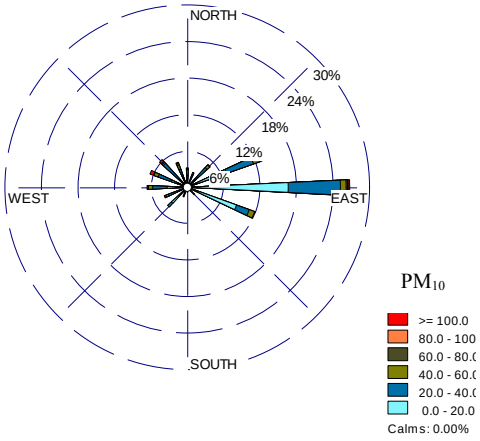

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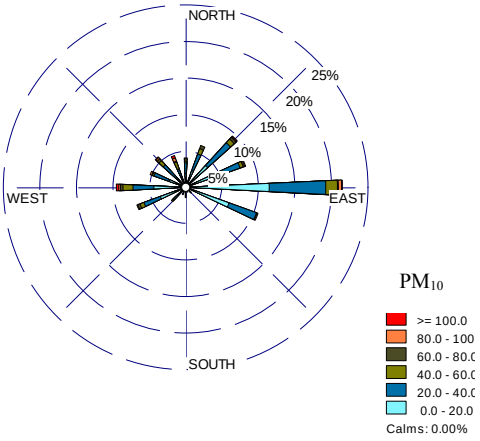

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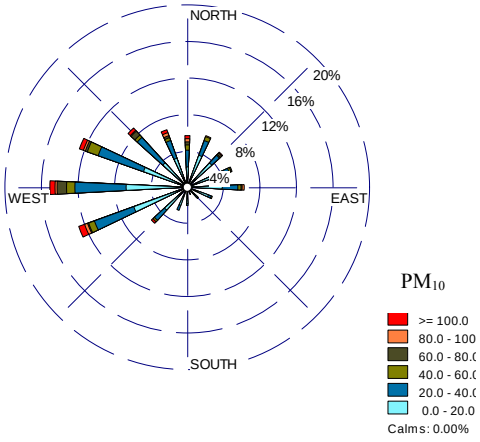

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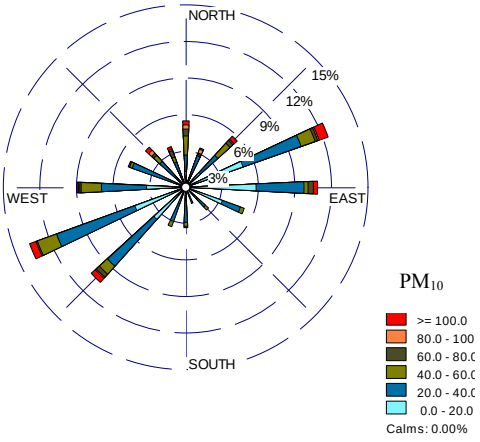

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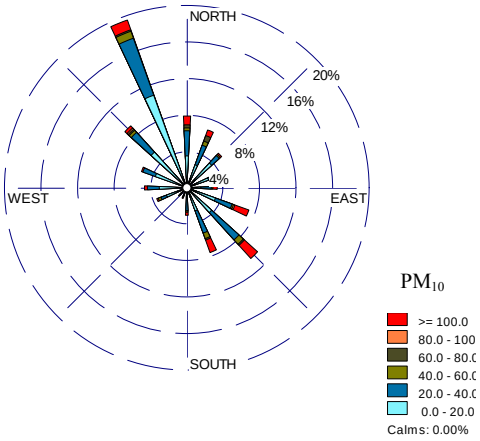

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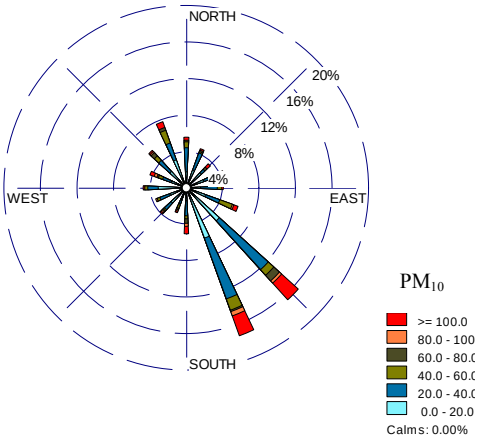

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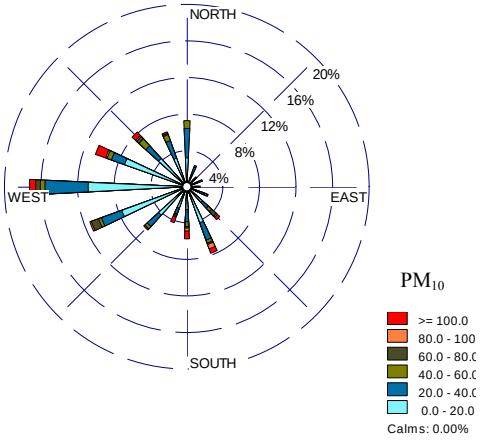

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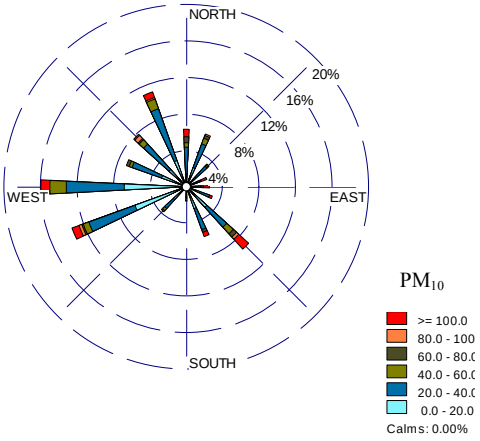

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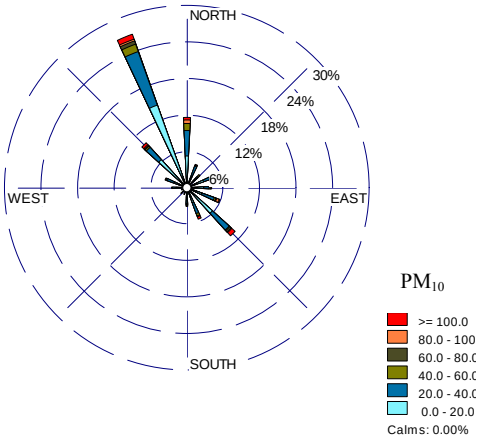

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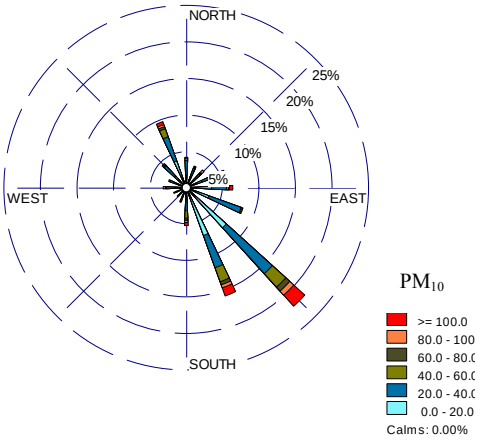
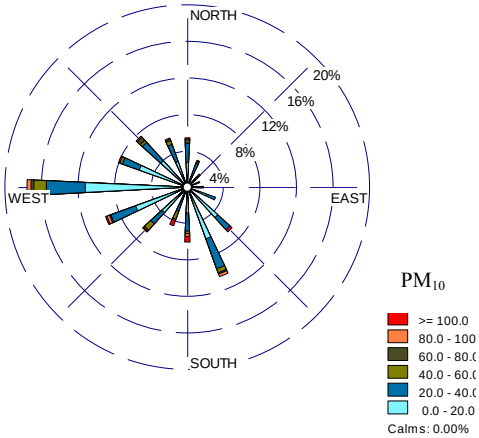


February hrs 12 to 17

February hrs 18 to 23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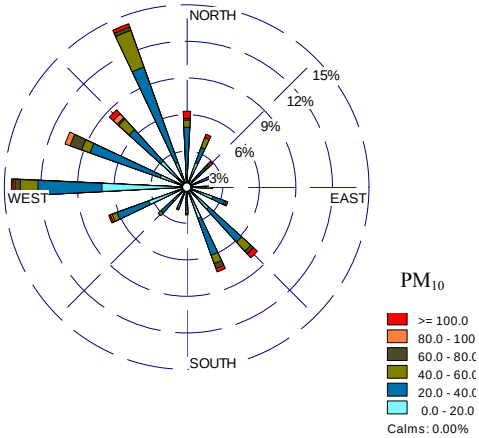
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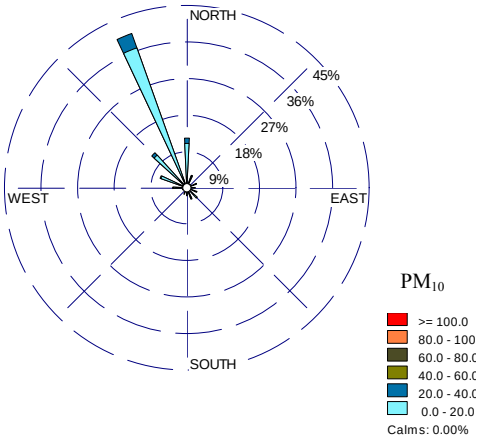
June hrs 0 to 5



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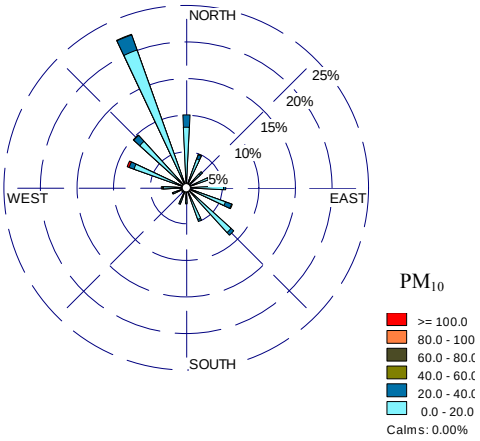

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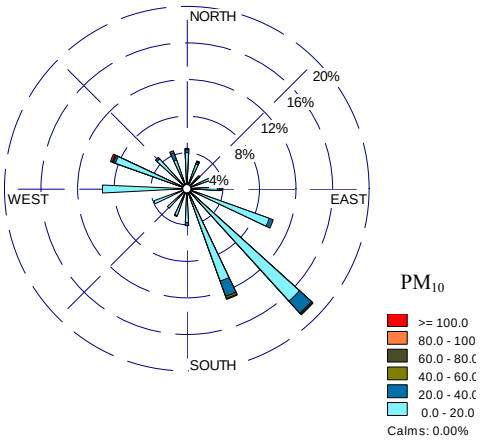

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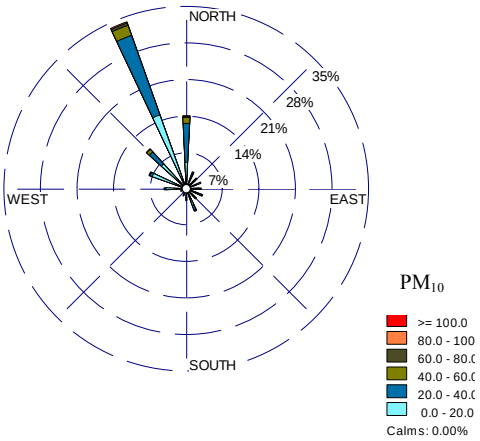

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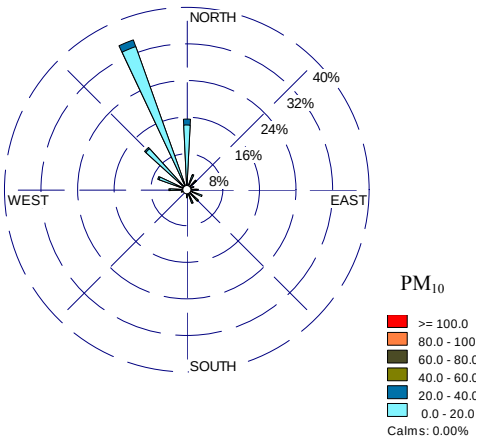

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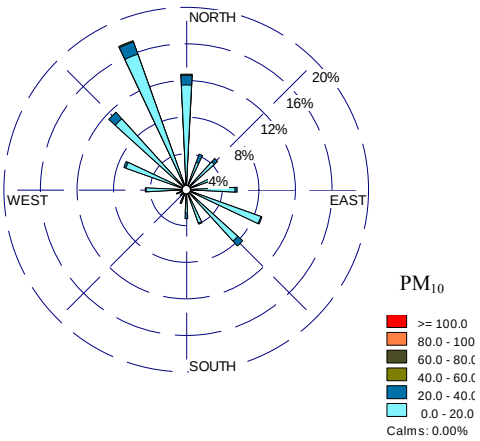
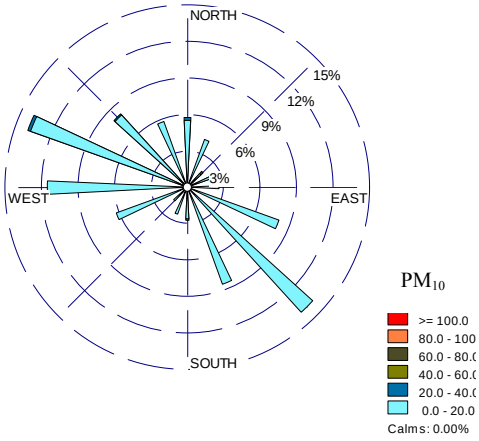


July hrs 12 to 17

July hrs 18 to 23



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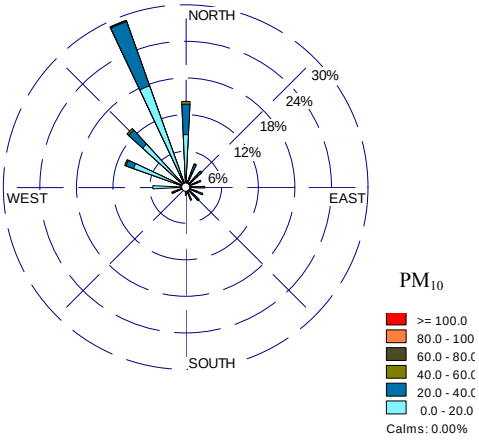

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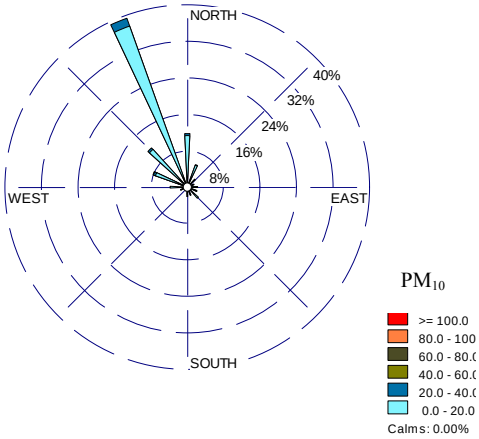

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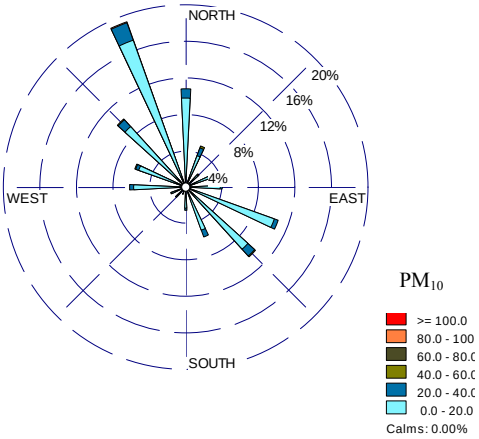

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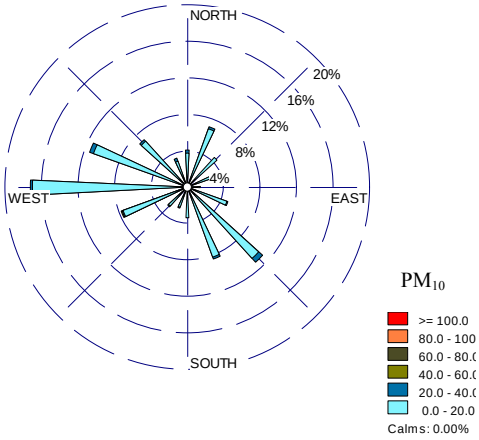

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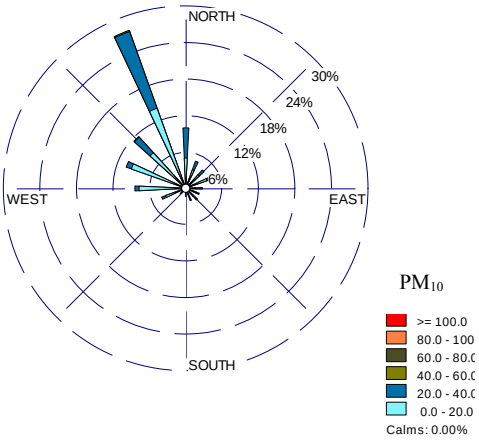

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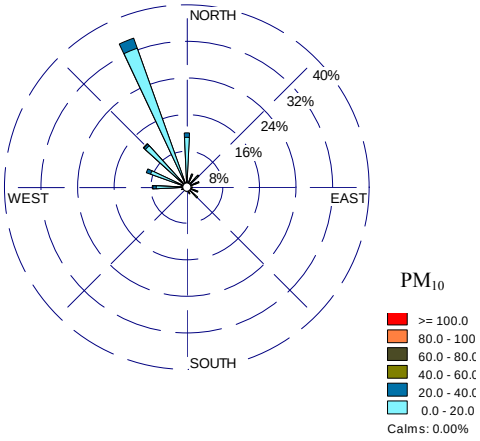

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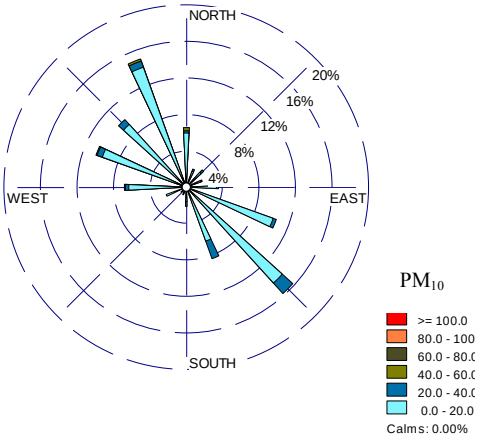
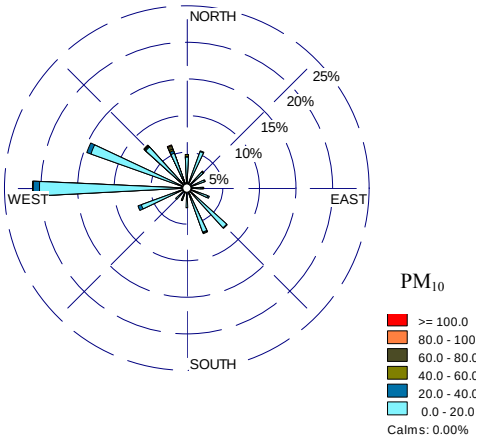


September hrs 18 to 23



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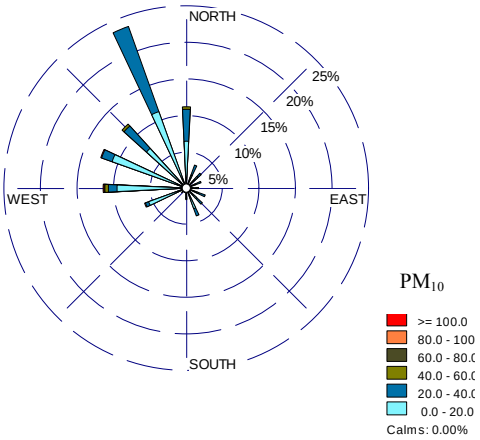

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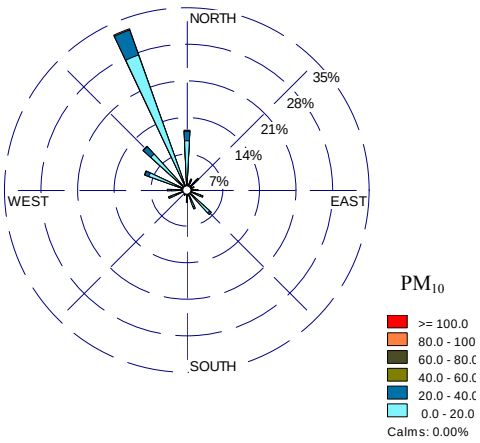

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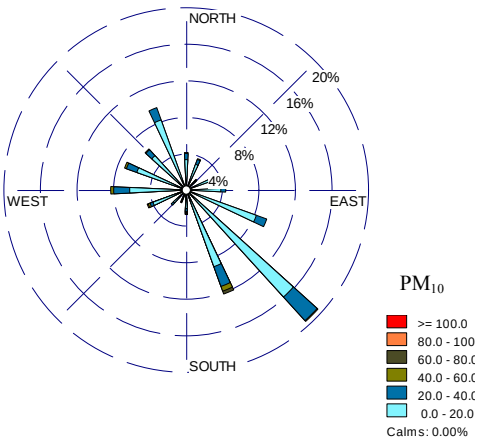

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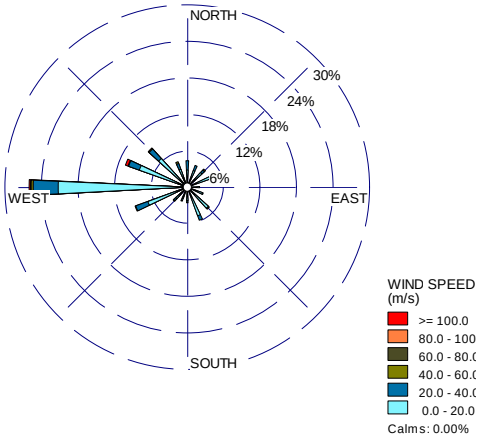

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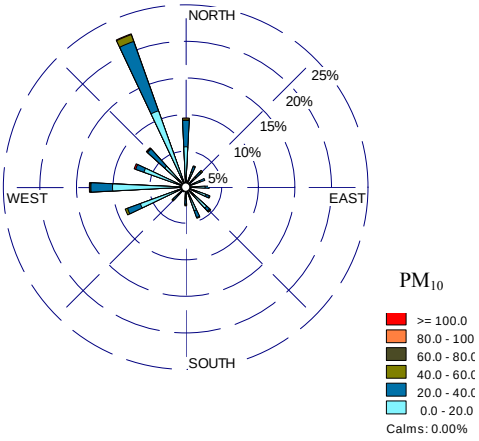

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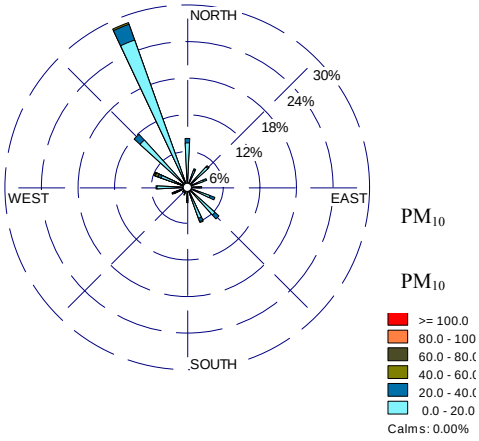

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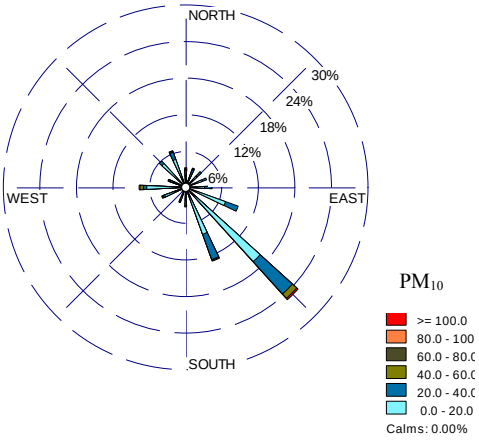

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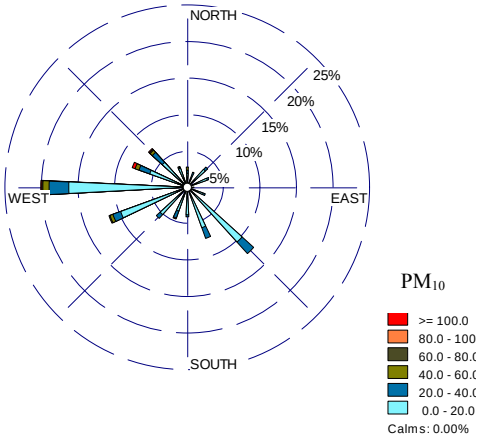

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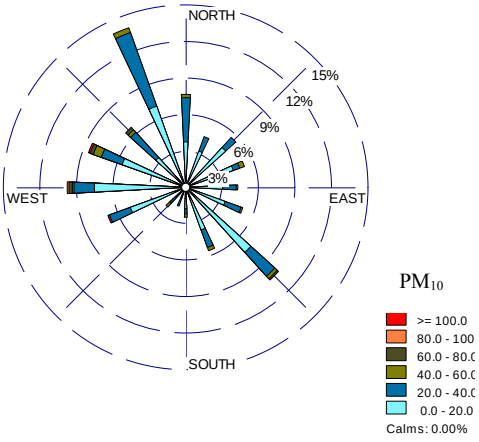

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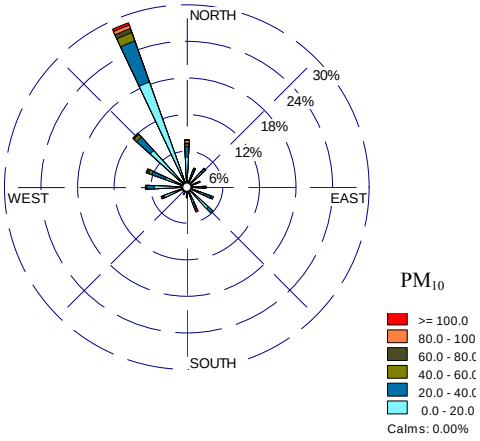

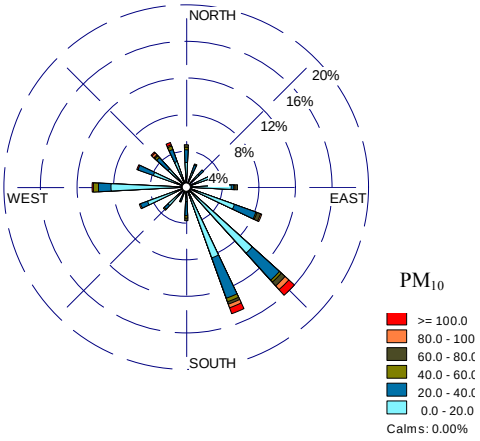

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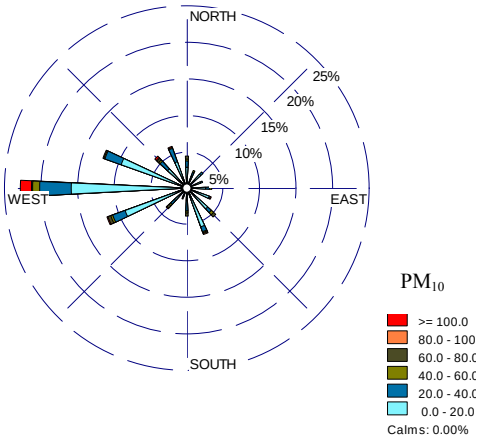

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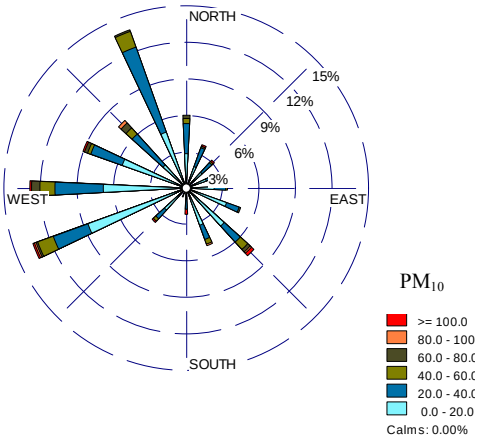

APPENDIX B

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Contact Us

Phone: 1300 363 400
+61 3 9545 2176

Email: enquiries@csiro.au

Web: www.csiro.au

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