MEASURING THE HEALTH IMPACT OF BUSHFIRE SMOKE

COMPANION VOLUME FOR THE FIRE DECISION SUPPORT TOOL FINAL PROJECT

C.P. (Mick) Meyer, Martin Cope, Sunhee Lee, Stuart Young and Melita Keywood
CSIRO Marine and Atmospheric Research

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The smoke plume from the Coonabarabran bushfire in New South Wales (January 2013). Photo supplied.

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Executive Summary

Fires are a part of the Australian landscape and a part of the Australian experience. While they recycle nutrients to ecosystems, they damage ecosystems, property, infrastructure and human life. They also emit large quantities of trace gases and particles to the atmosphere, and this smoke can negatively impact human health of either the population close to the fire, or populations several hundred kilometres away from the source as smoke undergoes long range transport.

These fires can be severe and extreme or can be small prescribed fuel reduction burns designed to reduce the severity of extreme wildfires. In the later case, while the distance this smoke may be transported is low (tens of kilometres), many prescribed burns are conducted in or near the rural-urban interface where residential human population is significant. Communities exposed to smoke from these events are increasingly concerned about the negative effect this smoke may have on the health of community members.

The aim of this study was to develop a Risk Tool Kit that can be used to quantify the health impact risk of smoke from wildfires and prescribed burning and to provide tools that fire managers can use to plan prescribed burning activities that minimise health impacts from the prescribed burning smoke.

The assessment of the risk of smoke impact on human health requires

1) Tools for developing a spatial emissions inventory suitable for input into chemical transport models
2) A chemical transport model
3) Tools for verifying the performance of the two, particularly for predicting surface air quality.

At the start of this project, emissions from biomass burning in temperate forest ecosystems were calculated from a very simple algorithm that proved to be inadequate for many fire classes, particularly higher intensity fires. High intensity fires are difficult to access and consequently emissions data for these was sparse. A comprehensive model appropriate for Australian vegetation to estimate emissions of both climatically active gases and particles and toxic compounds was required for many applications including assessment of climate impacts, emissions reporting, trading and management, population health impacts, agricultural impacts, and the development and implementation of management policy. The challenge was to describe emission properties using input parameters that could be quantified spatially across Australia. These input parameters were most commonly remotely sensed products (fire extent), interpolated from national observation networks (weather and climate) or generated from reliable spatial biogeochemical models (intensity, fuel loads, fuel moisture).

At the start of this project a number of air quality/fire models were being developed by the research community, however it was unclear how suitable these were for application to southern Australia. Issues that affected their suitability included accuracy of model formulation as well as access to and availability of data sets in the format required to drive the models. For example, it is rarely possible to port a model developed for other locations (such as North America) to another jurisdiction where differences in reporting formats exist for vegetation, fire area, local weather data, etc. Despite it being well known that smoke injection height has a significant impact on long range transport of smoke, chemical transport models often simulated plume rise poorly.

At the start of this project there were a number of observational data sets available to the Risk Tool Kit Project to assess the performance of the available models from around south-eastern Australia; these included air quality data from Aspendale Victoria (operated by CSIRO Marine and Atmospheric Research-

CMAR; Keywood et al., 2011), Ovens and Manjimup (collected as part of the Clean Air Research Program by CMAR; Meyer et al., 2008 and Reisen et al., 2011), Huon Valley (collected as part of the Huon Valley Air Quality Study by CMAR; Meyer et al., 2011) and air quality data collected by EPA Victoria and NSW OEH from the Victorian and NSW air quality monitoring networks. Also available were remotely sensed data products including aerosol data from CALIPSO (Cloud-Aerosol Lidar Infrared Pathfinder Satellite Observations) and aerosol optical depth (AOD) data from MODIS (Moderate Resolution Imaging Spectro-radiometer) on the NASA Aqua and Terra satellites.

This project involved a review of the available models and the costs of implementing them for southern Australia in order to determine the most suitable model for the Risk Tool Kit considering available time and computing resources (Meyer et al., 2012). Recommendations of this review resulted in the development of a framework for the Risk Tool Kit consisting of tools for developing a spatial emissions inventory suitable for input into chemical transport models, a chemical transport model, tools for verifying the performance of the emissions inventory and chemical transport model, tools for determining population exposure, and knowledge of risk associated with smoke exposure.

The performance of the chemical transport model was assessed by comparison with air quality observational data collected in Melbourne during the 2006/2007 Alpine Fires. The model predicted air quality parameters across Melbourne well and was able to accurately predict the timing of air pollution events (such as elevated ozone) when fire emissions and the correct plume rise parameters were included in the chemical transport model. Comparison with MODIS imagery of smoke showed qualitatively that the chemical transport model was able to effectively model the plume transport (Meyer et al., 2012).

The Risk Tool Kit was then applied to three case studies to determine the risk of health impacts from smoke exposure:

1) the Victorian Alpine fires of 2006/2007 – a large fire event of long duration;
2) the Kilmore East fire on Black Saturday (7 February 2009) – an extreme event, large in area and intensity, but of relatively short duration; and
3) a series of high intensity prescribed burns in the Huon Valley, Tasmania in Autumn 2010 – a series of prescribed burns that consumed a very large fuel load and created an extensive smoke plume that created a major controversy due the perception that it would have a significant impact on the local population.

The greatest health impact was from the 2006/2007 Alpine fires, which burned 1.1 million hectares over a period of 60 days. Smoke from the fire significantly impacted all of Victoria, including the Melbourne airshed, and produced a risk equivalent to an increase in mortality of 84 largely due to the long duration of the event which led to widespread smoke dispersion that reflected seasonal climatology. In contrast, the impacts from the Kilmore East fire and the Huon Valley prescribed burning events were minor, primarily because smoke did not impact regions of high population density and the events were of short duration and therefore determined by the weather of the day. All events emitted large amounts carbon to the atmosphere; however the size of the emission was not a reliable indicator of risk to health (Meyer et al., 2013).

In addition, a proof-of-concept tool has been developed to provide agencies with the information they need to decide where and when prescribed burning can occur that will minimise impact of smoke on populated centres. The concept was applied for the Ovens Valley, Victoria, the main population centre of which is 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 the 2003 and 2006 wildfire seasons, dense smoke persisted in the valley for several weeks. This valley is characterised by protracted fumigation events with smoke originating from a combination of different sources, either fresh or aged.

The approach, described in detail by Meyer et al. (2013) applied an inverse modelling technique to assess the relative contribution of emissions in each cell of a gridded domain on defined receptor cells. The spatial pattern of impact risk for the test month, April 2009, was not predictable from topography or distance from the source. This modelling approach offers promise of developing detailed climatologies of smoke dispersion relevant to fire districts. Development of the method will continue beyond the life of the current project.

Finally, a “Users Guide” describing how to configure and run The Air Pollution Model (TAPM) model for several practical applications has been produced in this project.

The study has demonstrated that smoke impact on regional populations can, on occasions, be the greatest risk from a fire event, far outweighing the direct risks at the fire front. Given the imperative to increase the extent of prescribed burning to 5% of public managed lands in Victoria, smoke impact on people is only likely to increase. New modelling approaches and applications of existing modelling systems have been developed to assist fire managers and planners to limit the risks of these impacts and will continue to be a priority of the Risk Tool Kit team members beyond the life of the current CRC.

Smoke dispersion models are computationally intensive, and are not currently in a form that can be incorporated directly in the Risk Tool Kit for tactical applications. However a project currently in progress, developed in part from the Risk Tool Kit experience shows promise of delivering a modelling framework suitable for operational planning within the next three years. This involves adapting the US modelling framework, BlueSky, to southern Australia.

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Glossary/Abbreviations

AOD	Aerosol Optical Depth
AVHRR	Advanced Very High Resolution Radiometer
CALIPSO	Cloud-Aerosol Lidar Infrared Pathfinder Satellite Observations
CCAM	Conformal Cubic Atmospheric Model
CMAR	CSIRO Marine and Atmospheric Research
CO	carbon monoxide
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CTM	Chemical Transport Model
DEPI	Department of Primary Industries
EPA Victoria	Environment Protection Authority Victoria
FireDST	Fire Impact & Risk Evaluation Decision Support Tool
µg	microgram 10^{-6} g
MODIS	Moderate Resolution Imaging Spectrometer
NO2	nitrogen dioxide
NOx	oxides of nitrogen
NSW OEH	New South Wales Office of Environment and Heritage
O3	ozone
PM2.5	mass of particles less than 2.5 µm in diameter
PM10	mass of particles less than 10 µm in diameter

1 Introduction

This document reports Milestone 4.1.3 of the Risk Took Kit Project of the Bushfire CRC. The Risk Tool Kit Project is a sub project of the Fire Impact & Risk Evaluation Decision Support Tool (FireDST) and is focused on the regional and local impacts from bushfire smoke. The objective of the Risk Took Kit Project was to develop a risk analysis framework for smoke impacts on health in urban and rural areas. The impetus for establishing this Project arose from the extreme fire event of Black Saturday 2009 that focused community and government attention on the risks associated with fire and extreme weather.

Community concern about the impact of smoke on health arises because of two factors. The first is the direct impact of smoke from the event itself on the health of the population close to the event source as well as up to several hundred kilometres away from the source as the smoke undergoes long range transport. The second arises from the management response to reduce risk of extreme fire events that has the potential to contribute to additional adverse health outcomes. Usually this management involves small prescribed fuel reduction burns designed to reduce the severity of extreme wildfires. However these fuel reduction burns produce smoke that impacts communities immediately surrounding the prescribed fires. While the distance this smoke is transported is generally low (up to tens of kilometres), many prescribed burns are conducted in or near the rural-urban interface where human population is significant.

The atmosphere is the medium transporting the impacts of fires from the fire ground to the wider region. It is becoming increasingly apparent that the health impacts can be significant. The challenge for a toolkit is to integrate smoke dispersion predictions, which typically require substantial computing resources into a short-term modelling framework. The aim of this project therefore was to identify the issues of smoke dispersion that are relevant to operational fire management and fire risk assessment.

Research questions being addressed here are:

- How effective and accurate are current models of smoke emission and dispersion in describing the vertical and horizontal distribution and concentration of smoke?
- What are the impacts on air quality from smoke emitted from extreme fires?
- How do these compare with the impacts from prescribed burning?
- What are the risks to population health from exposure to smoke from both sources?
- What are the options and requirements for developing them into operational tools for planning and incident control?

2 Methodology

2.1 Framework

To understand the impacts of smoke emitted from fires (extreme and prescribed burning fires), a smoke impact modelling framework has been developed.

The framework consists of:

1) Tools for developing a spatial emissions inventory suitable for input into chemical transport models (this provides information on the location and rate of smoke emissions);
2) A chemical transport model (this provides information on the dispersion of smoke and possible chemical transformations in the smoke plume);
3) Tools for verifying the performance of the two, particularly for predicting surface air quality (these provide information on how well the emission inventory/chemical transport model system is able to simulate smoke emission and dispersion);
4) Tools for determining population exposure (this provides information on the location of people and their proximity to smoke sources); and
5) Knowledge of risk associated with smoke exposure (this provides information on how many people will experience a particular health outcome at specific particle loadings).

2.1.1 EMISSION MODEL

Emission sub-models provide information on fuels, consumption and emission factors. They require a large amount of local information, although the general form of these sub-models is well established (Russell-Smith et al., 2009). Information required to drive the model includes fuel loads, burning completeness, fire area, patchiness, and emission factors. Also required is information on the rate of fire spread in order to convert from the time frequency of fire area data (usually daily) to hourly emission rates required for smoke dispersion modelling. While much of this information exists (e.g. fuel load models and databases for Victoria), it is currently not collated into a single database. Additionally, emission factors are generally based on overseas data, and thus an important component of the current project has been the refinement of these factors based on measurements carried out in Australia. This project has involved the collation of this information and the development of tools to convert the formats to those required for input to the chemical transport model (Meyer, 2012).

2.1.2 CHEMICAL TRANSPORT MODEL (CTM)

Fire modelling is a complicated multi-scale process, from the flame reaction zone on millimetre scale to the synoptic weather scale of hundreds of kilometres (Mandel et al., 2011). Weather has a major influence on wildfire behaviour; in particular, wind plays a dominant role in the fire spread. A forecast model for Australia may require an Australia wide grid in order to incorporate all influences on local meteorology.

This also poses a question of model scale. The model grid needs to be large enough to include large scale mixing features, yet small enough to depict valleys and local topographical features. Thus a model with a grid-nesting capability is required. After reviewing all available models and the costs of implementing them for southern Australia (Meyer et al., 2012) the modelling approach adopted in the Risk Tool Kit was:

(1) The Cubic Conformal Atmospheric Model (CCAM) with the CTM for regional modelling; and

(2) The Air Pollution Model (TAPM) for local operational planning.

2.1.3 OBSERVATIONAL DATA

Observational data available to assess the performance of the models included air quality data from Aspendale Victoria (Bayside Air Quality Station operated by CMAR; Keywood et al., 2011), Ovens and Manjimup (collected as part of the Clean Air Research Program by CMAR; Meyer et al., 2008 and Reisen et al., 2011), Huon Valley (collected as part of the Huon Valley Air Quality Study by CMAR; Meyer et al., 2011) and air quality data collected by EPA Victoria and NSW OEH from the Victorian and NSW air quality monitoring networks. Also available were remotely sensed data products including aerosol data from CALIPSO (Cloud-Aerosol Lidar Infrared Pathfinder Satellite Observations), aerosol optical depth (AOD) data from MODIS (Moderate Resolution Imaging Spectro-radiometer) on the NASA Aqua and Terra satellites.

2.1.4 POPULATION EXPOSURE

Population exposure to ground level PM2.5 (mass of particles less than 2.5 µm in diameter) can be estimated by combining the average hourly PM2.5 concentrations across the model grid for the duration of the event with the population distribution. The population distribution in Victoria and southern NSW is shown in Figure 1. The data were sourced from the 2011 census at SA1 resolution which comprises districts of approximately 300 people. This identifies the population centres at a finer resolution than the model grid, however for rural areas; it tends to smooth the population density across many model grid cells. For the most accurate assessment of impacts, the resolution of both the population distribution and the smoke dispersion should be matched to minimise the smoothing coincident peaks in population and PM concentration

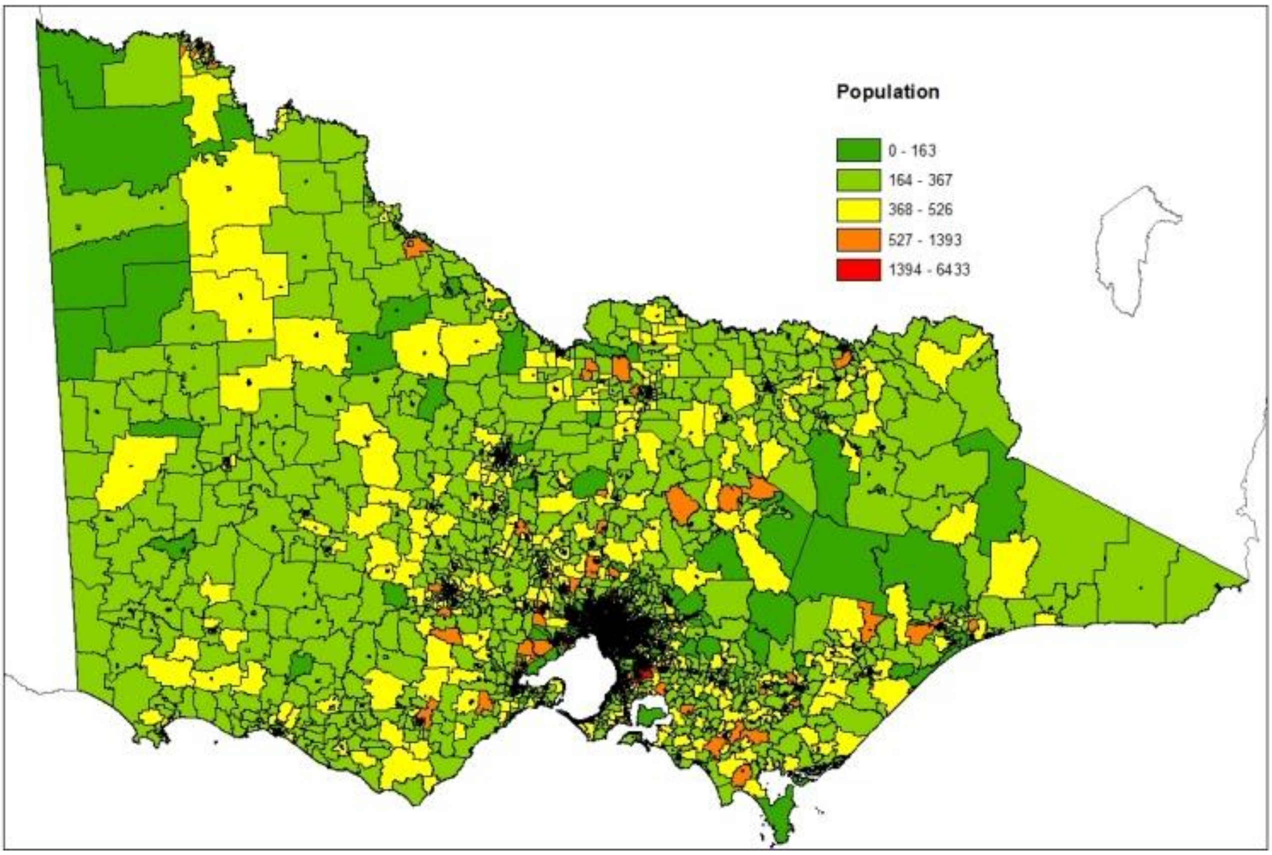

Figure 1 Spatial distribution of population density in Victoria in 2011.

2.1.5 HEALTH RISK

The impact of negative health outcomes resulting from exposure to PM2.5 has been determined in numerous epidemiological studies involving cohorts of large groups over long time periods. In their comprehensive review of PM10 (mass of particles less than 10 µm in diameter) impacts on health, Pope and Dockery (2006) conclude that an impact of at least 1% increase in mortality per 10 µg m^{-3} increase in PM10 concentration is consistently reported. Many individual studies, particularly those investigating both mortality and morbidity in susceptible population groups, report much higher sensitivities for PM2.5 impact. Table 1 lists some of the PM2.5 health risk rates for studies conducted in the US, Europe and Melbourne.

Table 1 Relative health risks for exposure to 24-hour average PM2.5.

Location	Health Risk	Reference
Harvard Six Cities study (8096 white participants from various cities of the USA followed from the mid-1970s to 1998)	Increase of 10 µg m^{-3}: 16 % increase in mortality, 28% increase in cardiovascular disease, 8% increase in respiratory	Laden et al. 2006
Women's Health Initiative cohort study, including 65,893 post-menopausal women	Increase of 10 µg m^{-3}: 76 % increase in cardiovascular mortality	Miller et al. 2007
ACS-CPS-II study linked air pollution data with the individual data of approximately 500,000 adults from the USA, followed from 1982 to 1998	Increase of 10 µg m^{-3}: 6 % increase in all mortality, 12% increase in cardiovascular disease	Pope et al. 2002; Pope et al. 2004
Los Angeles October 2003 wildfires	Increase of 10 µg m^{-3}: 3 % increase in respiratory hospital admissions, 5% increase in asthma hospital admissions, 4% increase in chronic obstructive pulmonary disease admissions	Delfino et al. 2009
Madrid 2003-2005	Increase of 25 µg m^{-3}: 7% increase in hospital admissions, 8% increase in cardiovascular admissions, 7% increase for respiratory admissions.	Linares et al. 2010
Melbourne, Alpine fires 2006/2007	increase of PM2.5 of 6 µg m^{-3}: 4.5% increase in out-of-hospital cardiac arrest	Dennekamp et al. 2011

Health impact (I) is the combination of population exposure to the pollutant and health risk, i.e.

$$I = C \ x\ P\ x\ D\ x\ R \qquad \text{Equation 1}$$

where:

I = health impact (deaths due to exposure to particulate matter per unit area);

C = increase above ambient of mean 24-hour surface concentration of particulate matter;

P = population density (people per km^2);

D = Standardised annual death rate deaths per 100,000 people per year). Standardised death rates use the age distribution of total persons in the Australian population at 30 June 2001 to normalise the crude death rate recorded for the statistical area. It is the baseline used to assess average risk; and

R = risk (increase in death rate).

Integrating *Equation 1* across the smoke affected region yields the total health impact for an event.

2.2 Performance of the model

Modelling smoke emission and dispersion from fires is an inherently difficult task due to the challenges of accurately representing plume rise and the variable spatial scale of the emissions. The methods used to introduce fire emissions estimates into the TAPM-CTM framework are described in the next section.

2.2.1 MODELLING FIRE EMISSIONS IN THE TAPM-CTM FRAMEWORK

The plume rise behaviour of a vegetation fire is strongly dependent on the fire heat flux, which in turn, is dependent on the fuel loading, the fuel moisture content, the slope of the terrain, and atmospheric drivers such as the air temperature, moisture and wind speed. In TAPM-CTM, fires can be treated as a matrix of point sources, providing a capability to inject the emissions into upper levels of the model thus representing plume rise due to the high temperatures of the fire plume. This is a critical consideration given that ground level concentrations may vary exponentially with the release height of the source.

For this assessment of the effect of plume rise, the transport and chemical transformation of emissions from the Victorian region during the 2006/2007 Victorian Alpine fires (both anthropogenic and fire-related) were modelled using TAPM–CTM. Details of the model set up are given in Meyer et al. (2012). The results presented here were for a simulation of December 2006. Emissions from the Alpine fires were modelled as individual point sources, with hourly emission rates for the modelling period determined by scaling daily area-based emissions resolved to 1 km by the diurnal time-course of fire rate of spread.

The plume rise parameter in TAPM-CTM requires estimates of the fire buoyancy flux, which are represented in these model runs as a range of prescribed heights (1000, 2000, 3000 m), based on observations from the CALIPSO satellite. The CALIPSO overpass on 20 December 2006 (Figure 2) which captured the fire plume from the a day of extensive fire activity in the Alpine fire complex shows that the smoke was located between 1000 m and 4000 m above ground level.

To determine the effective average plume rise height, four model scenarios were run. These were: the base case (or scenario 0) in which fires were absent but other sources of reactive pollutants were included; scenario 1 where the plume height was 1000 m; scenario 2 where plume height was 2000 m and scenario 3 where the plume height as 3000 m. The results of the model runs are presented as Taylor diagrams (Taylor, 2001), which graphically summarize how closely a pattern of model results matches the observations. The strength of the model/observation fit is quantified in terms of the correlation (the dotted radial lines labelled 0.1 to 0.99 in Figure 3 and Figure 4) and the ratio of modelled to observed variance (the concentric lines labelled 0 to 2).

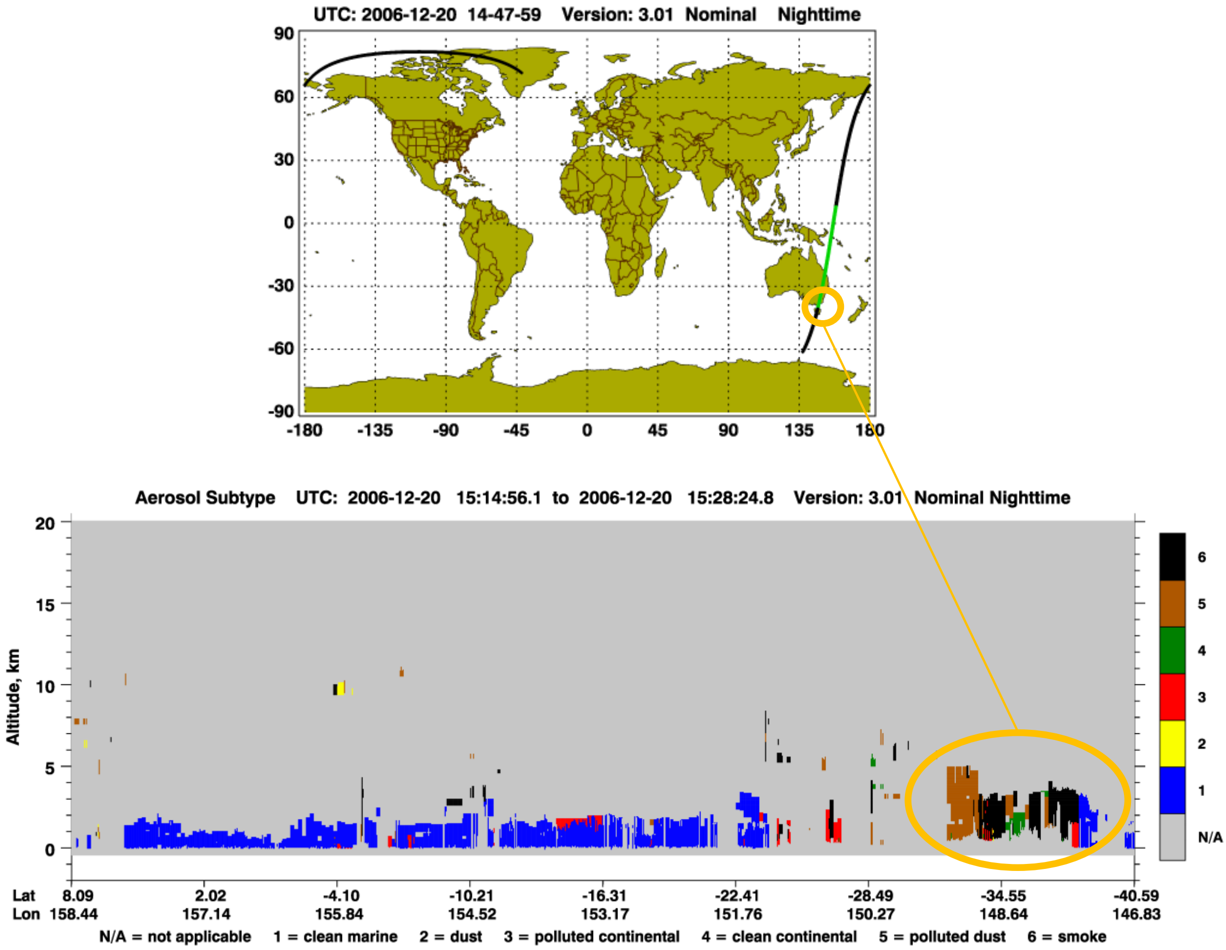

Figure 2 Aerosol sub-type curtain plot (bottom) for the CALIPSO satellite trajectory (top) for 20 December 2006. The circled area (bottom) shows the diagnosed aerosol sub-type for the trajectory segment which passes through south-eastern Australia. The smoke sub-type corresponds to the Alpine fire.

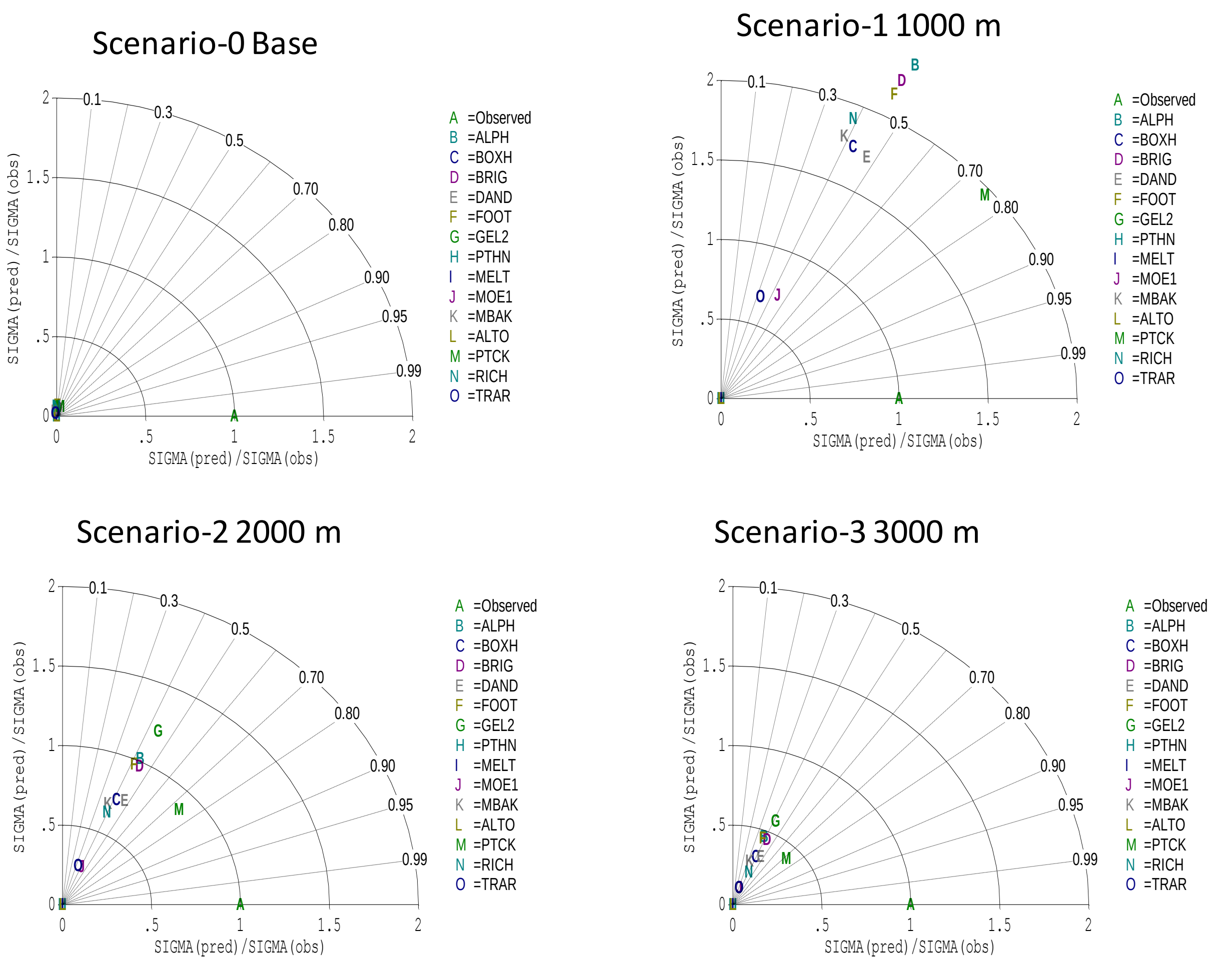

Figure 3 Taylor diagrams of average PM2.5 simulated for the four scenarios, using the 9 km grid, December 2006. ALPH is Alphington, BOXH is Box Hill, BRIG is Brighton, DAND is Dandenong, FOOT is Footscray, GEL2 is Geelong South, PTHN is Point Henry, MELT is Melton, MOE1 is Moe, MBAK is Mooroolbark, ALTO is Altona, PTCK is Point Cook, RICH is Richmond and TRAR is Traralgon.

Figure 3 shows the Taylor diagrams for PM2.5 simulated for the four scenarios. In the base case the data are crowded about the origin as is expected since this is the case where no fire emissions were included and all other sources of aerosol were small. In the other three plume rise scenarios, the correlation between observations and predictions is around 0.4, however the average ratio of predictions to observations is closest to 1 for the 2000 m plume rise scenario. For the 1000 m scenario, predictions are generally greater than observed (pred/obs >1.5), while for the 3000 m scenario the predictions are generally lower than observed (pred/obs < 0.5). The 2000 m plume rise scenario also does the best job of predicting ozone (Figure 4). The technique was also applied to a range of other pollutant species; for NO_2 all three plume rise scenarios produce results similar to the base case suggesting that NO_2 is more strongly influenced by non-fire sources, however for CO (for which fire is the largest source), the 1000 m plume rise scenario gives the best results.

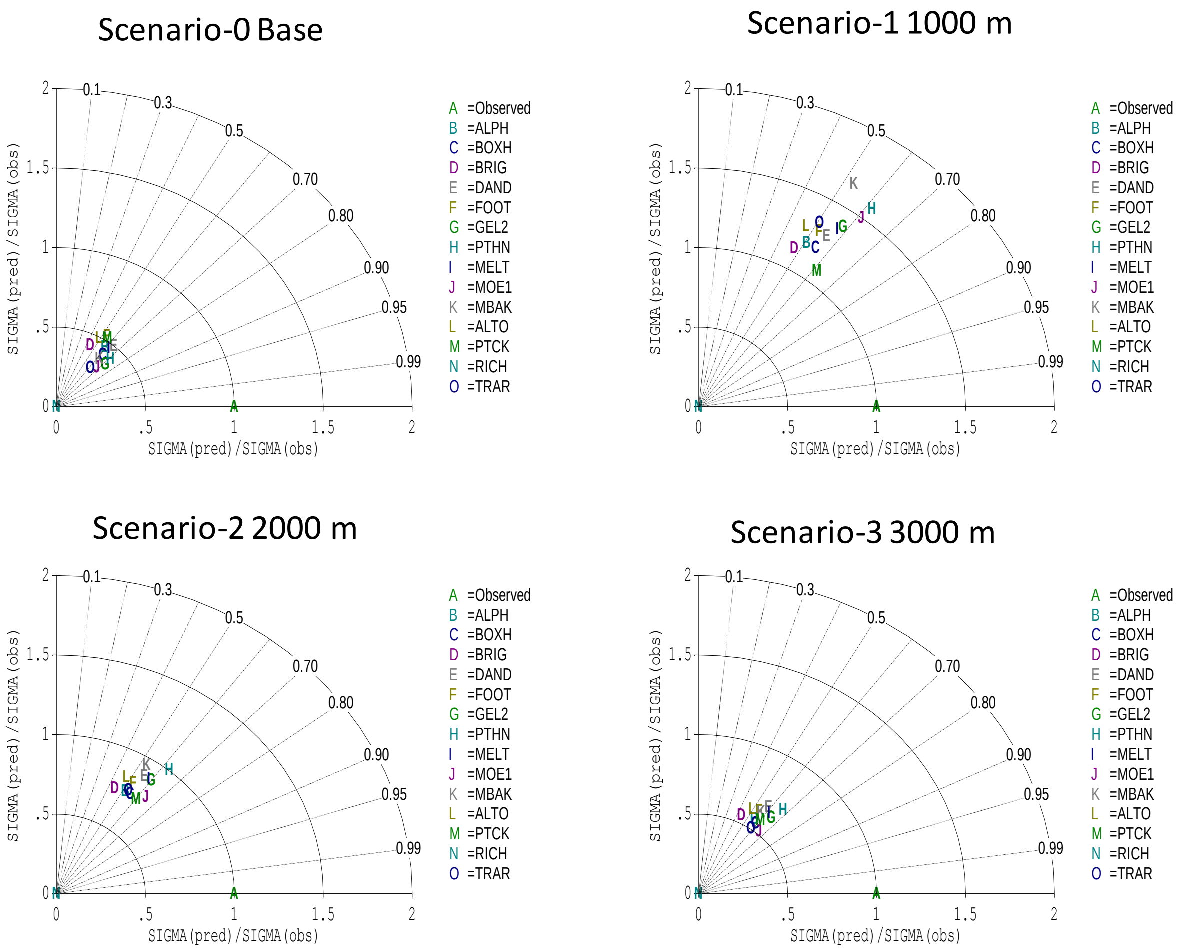

Figure 4 Taylor diagrams of average ozone simulated for the four scenarios, using the 9 km grid, for 2-31 December 2006.

2.2.2 MODEL ACCURACY

Verification of model accuracy by comparing model predictions against surface observations is essential before modelled smoke dispersion can be used to assess population impact risks. The data summarised in the Taylor diagrams presents a good overall summary, and shows that model predictions across the Melbourne airshed are good. However it is also important to compare the time series of modelled and observed concentrations at a selection of observing stations to confirm the model dynamics hour-by-hour rather than just as monthly averages. The Melbourne airshed was chosen for this comparison because of the number of sites in the airshed for which air quality data are available (these sites belong to the network of EPA Victoria air quality stations). Figure 5 compares the observed and modelled concentration time series for 1-hour ozone. Time series plots are given for the Moe monitoring station in the Latrobe Valley, Brighton in Melbourne and Geelong South to the south-west of Melbourne. The modelled concentrations are plotted for the four scenarios – the base and three plume rise scenarios. The model is unable to reproduce the high observed ozone concentrations unless the Alpine fire is included in the simulation. When the fire is included, the model is able to predict the timing of the high ozone events for the majority of events, and (as shown above), the magnitudes of the observed and modelled peaks agree best for scenario 2 (plume rise of 2000 m).

Only two stations in the Melbourne network, Footscray and Alphington, included instrumentation that measured PM2.5 concentration directly. However all stations included instruments that measured particle light scattering which is highly correlated with PM2.5 concentration. This correlation can thus be applied reliably to estimate PM2.5 from the light scattering data for stations that did not directly measure PM2.5 concentration. Figure 6 suggests there were three significant fire plume impacts at Geelong South, four to five impacts at Brighton and five impacts at Moe. The model was able to predict the majority of these impacts within a window of ± 1 day. With the exception of Moe where the PM2.5 impacts of the fires were under predicted, the modelled PM2.5 concentrations for the three plume rise scenarios generally span the observed PM2.5 concentrations.

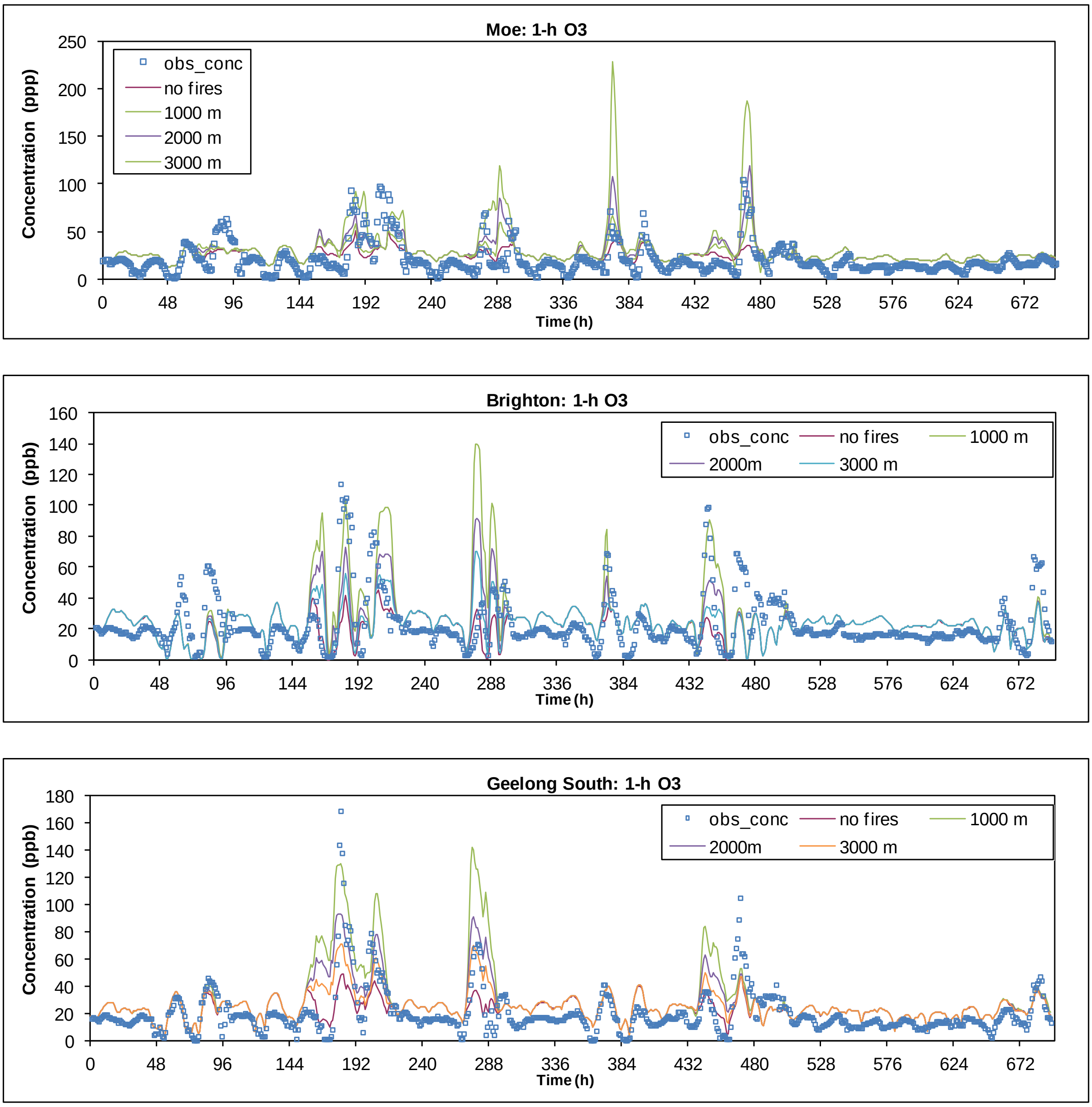

Figure 5 Observed and modelled 1-hour concentration time series for ozone for Moe, Brighton and Geelong South. The concentration time series are for December 2006.The modelled concentrations are shown for the four scenarios described in the text: Base – no fires; 1000 m- Alpine fire plume injected at 1000 m above ground level; 2000 m- injected at 2000 m; 3000 m- injected at 3000 m.

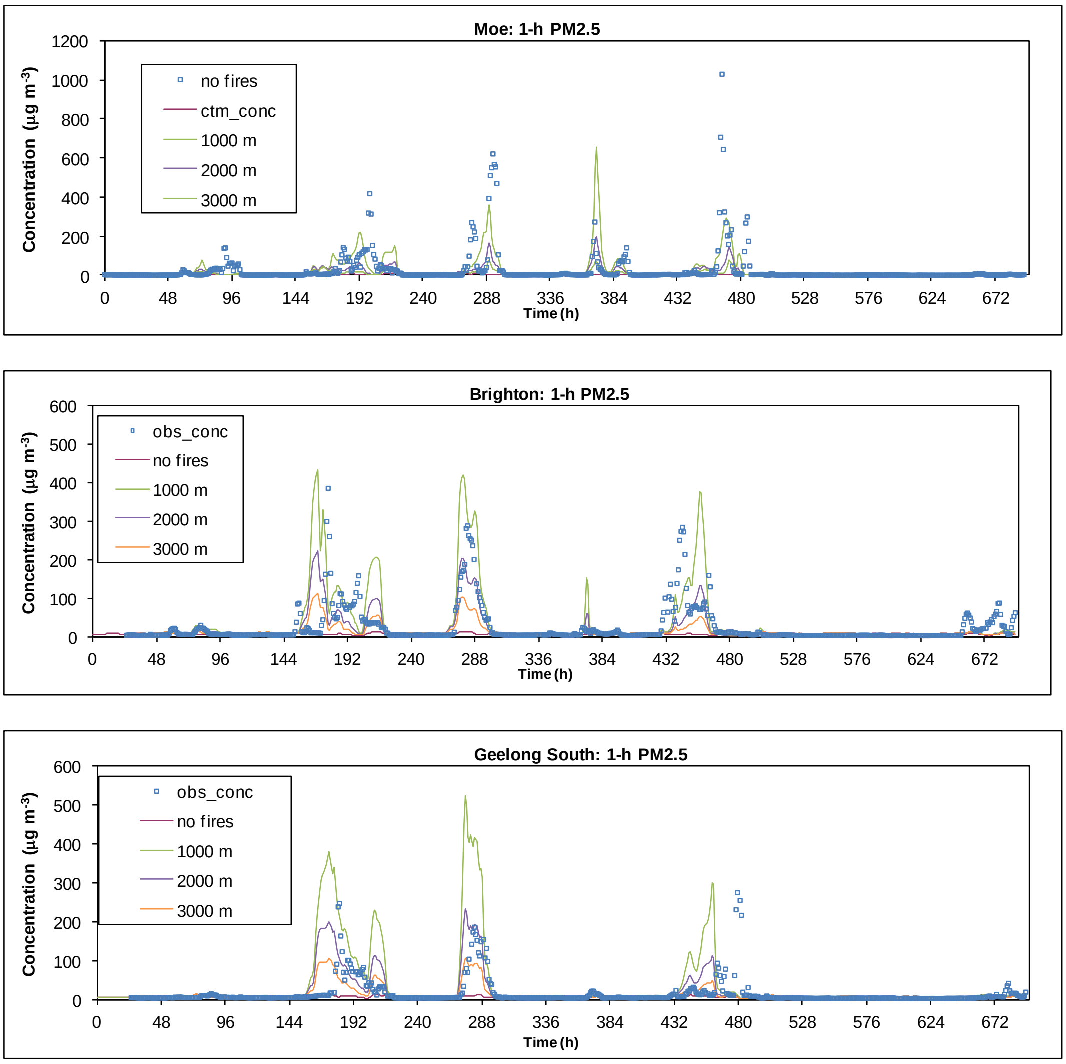

Figure 6 Observed and modelled 1-hour concentration time series for PM2.5 for Moe, Brighton and Geelong South. The concentration time series are for December 2006. Note that the observed PM2.5 has been generated from 1-hour nephelometer light scattering observations using a linear regression between PM2.5 and nephelometer data derived from observations at Footscray and Alphington monitoring stations in Melbourne for December 2006. The legend is the same as for Figure 5.

Figure 7 shows a comparison of the observed and modelled 1-hour ozone and PM2.5 concentration distributions throughout December 2006 at the Brighton monitoring station. Again the modelling system accurately estimated the observed distribution of extreme events (for either ozone or PM2.5) only when emissions from the Alpine fires were included. In the case of ozone the 3000 m scenario tended to underestimate the observed range, the 1000 m scenario overestimated the range and the closest agreement occurred with the 2000 m plume rise scenario. In the case of PM2.5, the model was more challenged to reproduce the inter-quartile range (25th to 75th percentiles) than for ozone, and the best agreement occurred for the 1000 m plume rise scenario.

MODIS imagery was used to evaluate the effectiveness of the modelling system to predict the dispersion of the black carbon component of PM2.5 produced by the Alpine fires. Figure 8 shows a comparison of the modelled spatial distribution of elemental carbon and visible light images from MODIS. The figures cover the period 3-10 December for days where the fire plume was easily distinguished from the cloud cover. This period also included two days when smoke was detected in the Melbourne airshed at the CSIRO atmospheric monitoring station and across the EPA observing network. The figure demonstrates qualitatively that the model effectively modelled the plume transport.

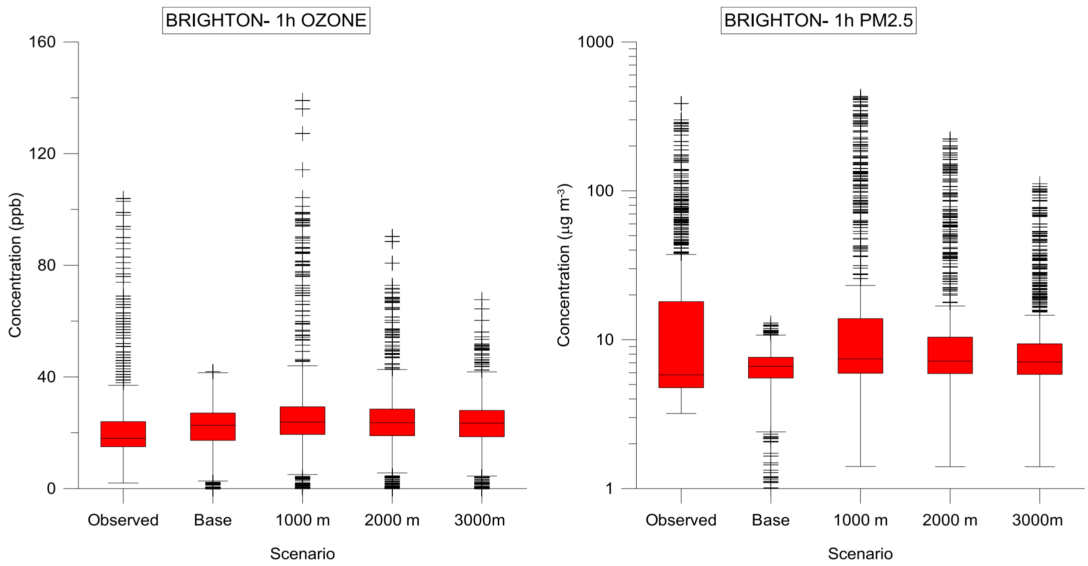

Figure 7 Box and whisker plots of the observed and modelled distributions of (left) 1-hour ozone, (right) 1-hour PM2.5 (derived from nephelometer data) for Brighton monitoring station for December 2006. Base - no fires; 1000 m - Alpine fire plume injected at 1000 m above ground level; 2000 m - injected at 2000 m; 3000 m- injected at 3000 m.

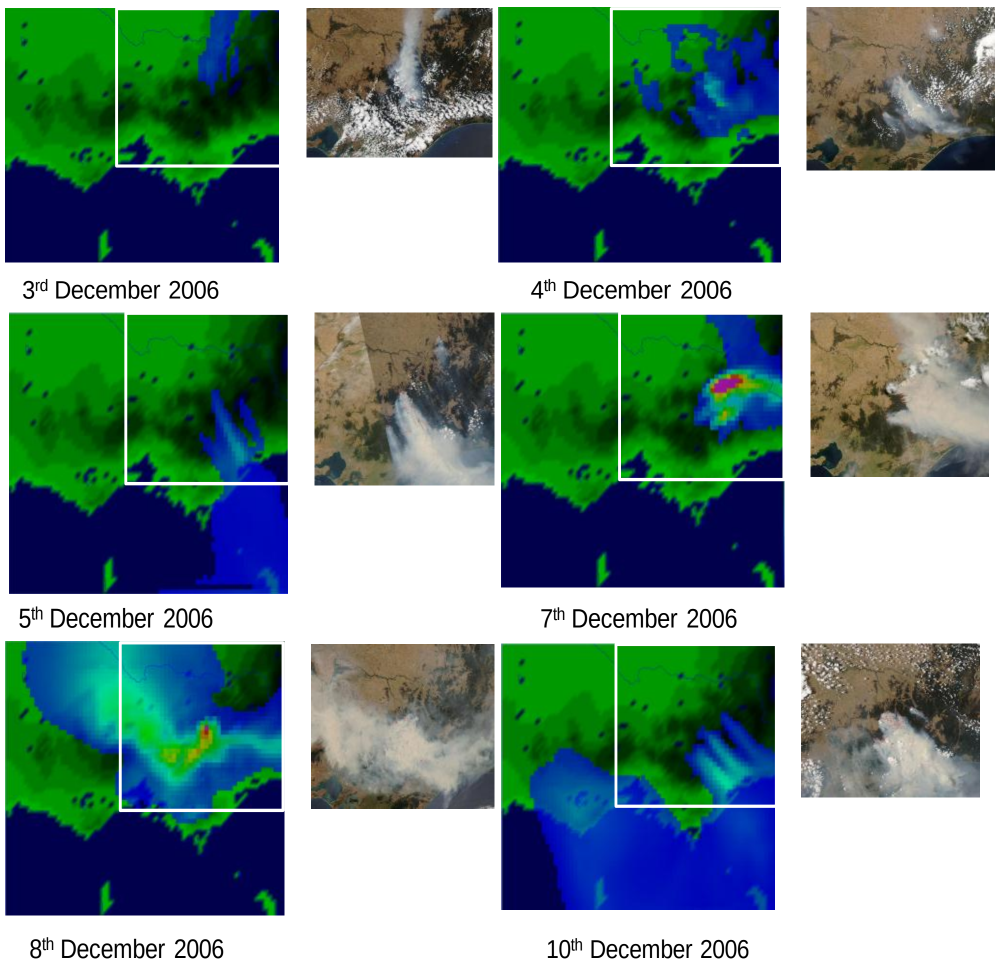

Figure 8 Comparison (inset) of the modelled spatial distribution of modelled elemental carbon (PM2.5) with MODIS Aqua satellite visible images. Comparisons are shown for 3, 4, 5, 7, 8 and 10 December 2006 (selected for having a relatively clear sky view of the Alpine fire smoke plume).

3 Case Studies

To assess the significance of smoke impact on health risk, three recent fire events that cover a wide range of potential impacts were investigated. They were:

(1) The Victorian Alpine fires of 2006/2007;

(2) The Kilmore East fire on Black Saturday (7th February 2009); and

(3) A series of high intensity prescribed burns in the Huon Valley, Tasmania in autumn 2010.

The first was a large fire event of long duration, the second was an extreme event, large in area and intensity, but of relatively short duration, and the third was a series of prescribed burns that consumed a very large fuel load and generated an extensive smoke plume that created a major controversy due the perception that it would have a significant impact on the local population.

3.1 Big wildfire - Victorian Alpine Fire Complex (2006/2007)

During the summer of 2006/2007 (December 2006 – February 2007), Victoria was affected by the longest recorded fires in the State's history. During this time the State was ravaged by 690 separate wildfires, including the major Great Divide Fire, which devastated one million hectares over 69 days. On several occasions, thick smoke haze was transported over Melbourne's central business district and PM10 concentrations at several EPA Victoria air quality monitoring sites peaked at over 200 µg m^{-3} (four times the National Environment Protection Measure Ambient Air Quality Standard for 24-hour PM10.

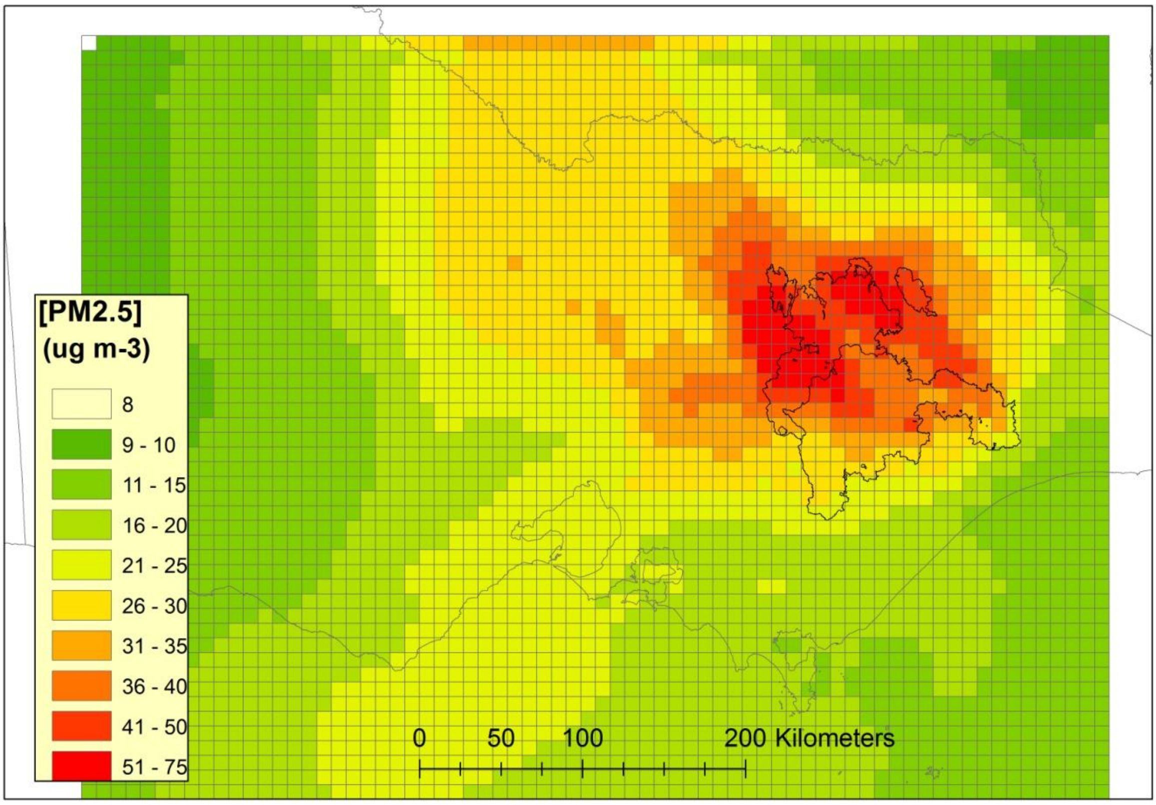

Figure 9 Average surface mean 24-hour PM2.5 concentration during the 2006/2007 Alpine fires.

Using methodologies reported in Meyer et al. (2012), the emissions of trace species from the smoke were estimated and incorporated into TAPM-CTM to predict daily PM2.5 concentrations across Victoria and Southern NSW. The model grid was 80 x 80 cells each 9 km square. The model was run from 2 December

2006 to 31 January 2007. The mean daily PM2.5 concentrations were calculated for each cell and then averaged across all days of the model run to produce the distribution of average 24-hour surface PM2.5 (Figure 9). This was corrected for the background PM2.5 to estimate the increase in mean 24-hour PM2.5. The background PM2.5 concentration (i.e. the average PM2.5 due to all sources other than the Alpine fire) was estimated as the 5th percentile of the frequency distribution of 24-hour PM2.5 in each cell. The 5th percentile was selected to ensure that we accounted for the urban contribution to the non-fire background concentration since values below this percentile are dominated by non-urban background concentrations.

As a first approximation we can take the increase in mortality to be a useful indicator of health risk associated with exposure to smoke. As discussed above, the impact on health of PM2.5 (Table 1) varies widely across groups and studies, however the average impact reported by Pope and Dockery (2006) of an increase in death rate of 1% per 10 µg m^{-3} increase in mean 24-hour for PM10 across the total population should give a conservative estimate of risk for PM2.5 that will support comparisons across the three case studies. The study can be extended in the future to account for variation in risk with age group and health status. Applying Equation 1 to each grid cell produces the distribution of health risk across the domain (Figure 10). Most of the risk is located in the Melbourne airshed, particularly the inner eastern and south-eastern suburbs; however the large regional towns, particularly Geelong, Ballarat, Bendigo, Shepparton and Albury/Wodonga were affected. Integrating across the model domain indicates an additional 84 deaths may have occurred due to exposure of the Victorian population to smoke from this fire, which is an increase in the average annual death rate of approximately 0.3%. This is a substantial health impact.

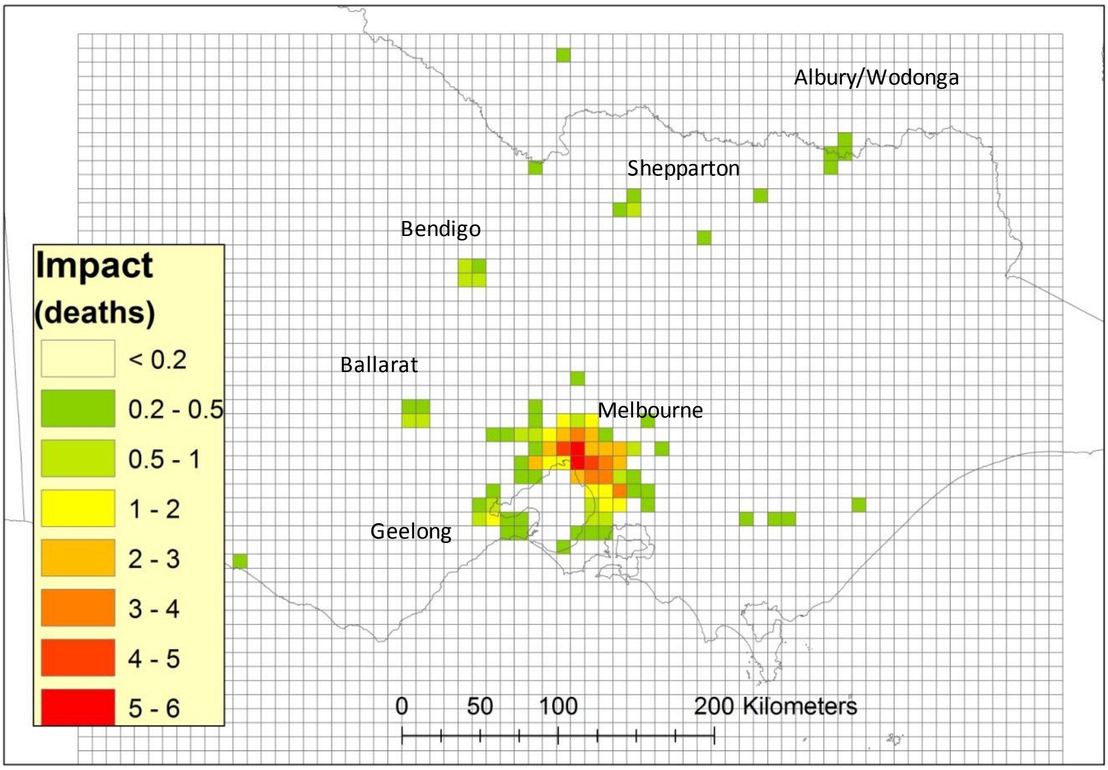

Figure 10 Modelled mortality increase in Victoria due to modelled PM2.5 associated with smoke from the Victorian Alpine Fire complex. PM2.5 increase above the background PM2.5 concentration for the 60 days of impact was multiplied by the population distribution from the 2011 census and the standardised death rates for 2006-2011. For this analysis we assumed a 1% increase in mortality per 10 µg m^{-3} increase in 24-hour mean PM2.5, a conservative estimate.

3.2 High intensity, short duration - Kilmore East Fire

The Kilmore East fire was the most significant contributor to fatalities during the Black Saturday fires. It burned 50,000 ha in less than 12 hours on 7 February 2012 and accounted for 70% of the fatalities. Cruz et al. (2012) provide an analysis of the development of the fire that was driven by a combination of extremely dry fuel and near-gale to gale force wind. The rate of fire spread was very fast (between 68 and 153 m min^{-1}). In addition, strong winds aloft and the development of a strong convection plume led to the transport of embers that resulted in the lighting of spot fires up to 33 km ahead of the main fire front. The change in wind direction between 17:30 and 18:30 turned the 55 km long eastern flank of the fire into a headfire and a pyrocumulonimbus cloud formed that injected smoke into the lower stratosphere. The extremely fast development and progression of the Kilmore fire (Cruz et al., 2012) posed significant difficulties for the modelling of the dispersion of smoke associated with this fire. In particular uncertainty exists around the points of ignition, the fire progression pattern and the effects of the grid size employed in the modelling framework.

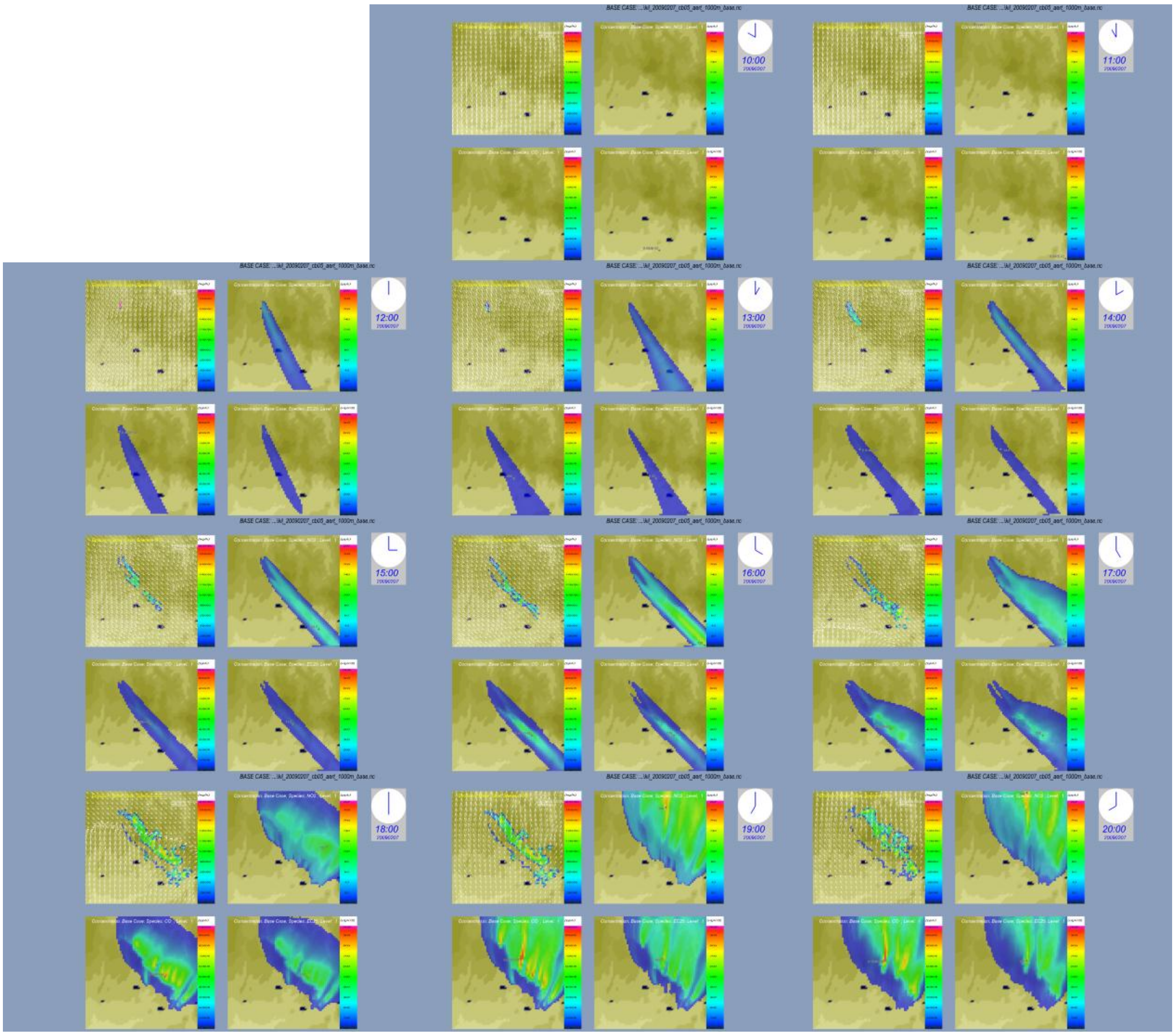

Figure 11 Time sequence of TAPM-CTM model output of the Kilmore fire at 3-hourly intervals. Top left plot of each set of four is the emission field (representing the fire), right top plot of each set is NO2 concentration (ppb), bottom left plot is CO concentration (ppm) and the bottom right plot in each set is the elemental carbon concentration (µg m^{-3}).

The influence of each of these factors on estimates of the number of people impacted by smoke dispersion was assessed using TAPM-CTM. Figure 11 shows progression of the smoke plume from ignition at 11:45 until containment at approximately 20:00 Australian eastern summer time. Strong northerly to north-westerly winds caused the fire to progress along a south-easterly track and the fire front grew extensively over the next 8 hours. The smoke plume followed this track until 17:00 when the change in wind direction to a south-easterly moved the plume to north-west. This was associated with a dramatic increase in the area covered by the plume and the concentration of species such as CO, NO_2 and elemental carbon in the plume.

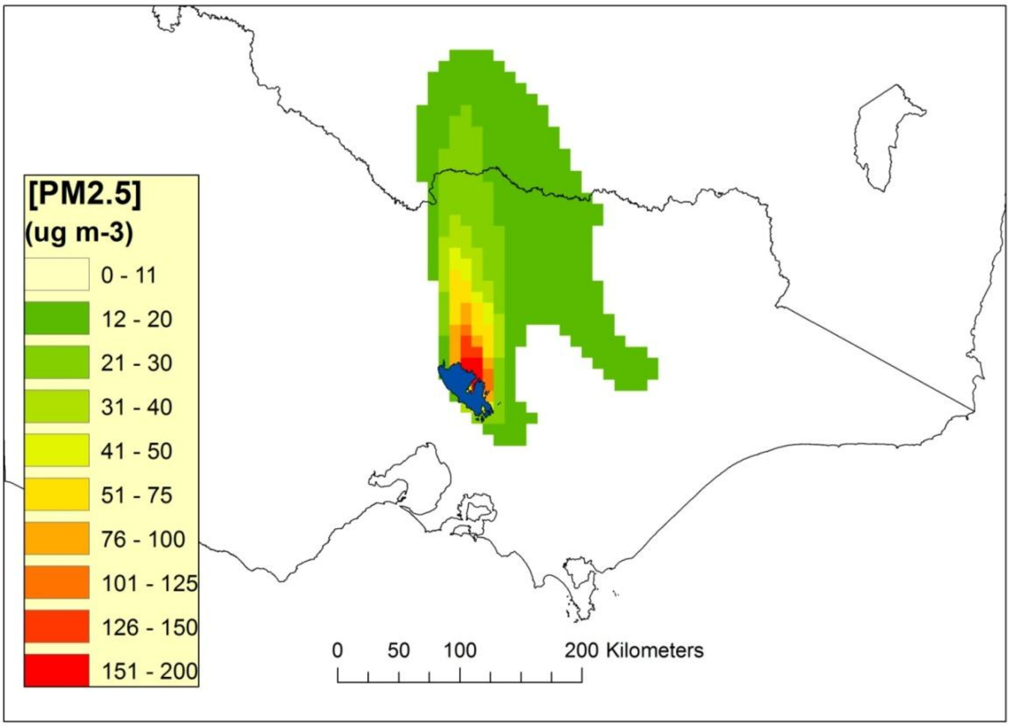

Figure 12 Mean 24-hour surface concentration for PM2.5 resulting from the Kilmore East fire on the 8 and 9 February 2009. The final fire affected is shown in blue.

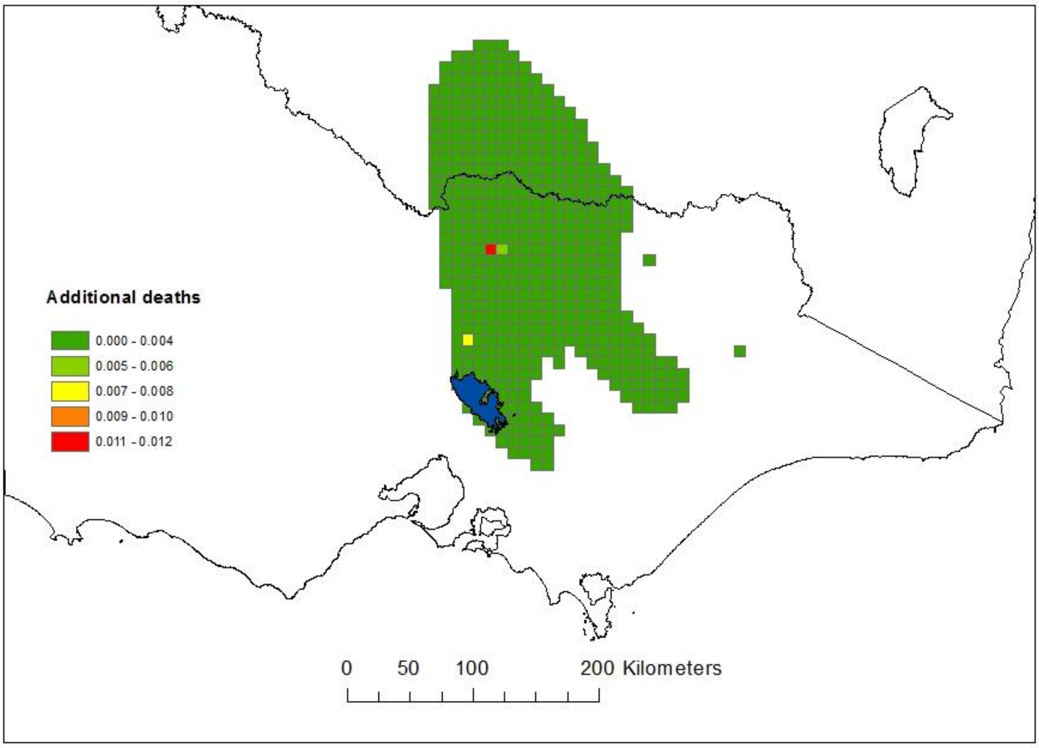

Figure 13 Modelled Increase in deaths due to potential exposure to modelled PM2.5 emitted from the Kilmore East fire.

The distribution of surface PM2.5 (Figure 12) was determined mostly by dispersion following the wind change in the late afternoon when the plume rise diminished and the depth of the mixed layer declined thus mixing the plume down to the surface; the smoke plume prior to the wind change passed high above the regions to the south-east of the fire zone and dispersed in the troposphere and lower stratosphere over the Tasman Sea. Consequently, it was the regions to the north and east of the fire that were most affected. The population density in the impacted region is quite low; the only large towns and cities impacted were Seymour and Shepparton (Figure 13). Integrating across the domain gives a total increase in deaths of 0.1; a very minor risk, and negligible compared to the 173 deaths that occurred on the fire ground (Teague et al., 2010).

3.3 High intensity regeneration burns - Huon Valley

In rural areas, air quality is generally good, however there are occasions when this is not the case. Rural pollution events mostly result from domestic, agricultural or forestry activities, and often involve smoke from biomass combustion. In the Huon Valley, Tasmania, smoke from prescribed burning has been the subject of public debate. The commonly accepted view is that regeneration burning following forestry logging operations is the major source of particulate matter (PM) pollution; however, in the absence of reliable ambient air quality monitoring it is impossible to confirm the veracity of this perception.

The major anthropogenic sources of PM in the Huon Valley are prescribed burning, domestic wood-fuelled heaters, windblown dust from roads, motor vehicles, and domestic waste incineration. To determine the contribution of these sources to the ambient surface concentration of PM in the Valley, two air quality monitoring stations were installed; one at a rural site, the Department of Primary Industry Research Station near Grove; and one in an urban area, Geeveston. The rural site was expected to be affected mostly by prescribed burning while the urban site was expected to be influenced by all anthropogenic PM sources. Ambient surface PM concentration was monitored continuously between March 2009 and November 2010; both sites were impacted by smoke from the two biomass combustion sources, including prescribed burning and domestic woodheaters. However these sources are active in different seasons; the prescribed burning season is March and April, and the domestic woodheaters season extends from May to September (Meyer et al., 2011). The summer is a period of negligible biomass combustion. The study concluded that most of the PM pollution in the towns was due to domestic woodheaters emissions; however prescribed burning caused significant pollution events in both years of monitoring. The most severe event was the period from 10-21 April; it lead to widespread public criticism and concern about health impacts and was probably the most publicised smoke impact event in recent years in Tasmania. However, the study did not investigate in detail the patterns of smoke dispersion in the region and therefore was not able to comment on the regional PM exposure and health impacts. Hence the study provides an excellent third case study for Risk Tool Kit Project.

The modelling challenges in this system involved the time course of the emissions and plume rise. Neither the fuel load, nor the rate of fuel consumption have been measured for high intensity burns in this region, however we estimated fuel loads of 100 t ha^{-1} as described in Meyer et al. (2011). Based on the fuel consumption model the Fire Emission Product Simulator (Anderson et al., 2004) we assumed that the fuel will burn at an approximately exponentially declining rate over a period of 24 hours (Figure 14). We also set the ignition time of each burn at 12:00 h. The effect of variations in these parameters was explored by Meyer (2013), and the impact on the dispersion pattern was found to be relatively minor. Based on the boundary layer mixing height during the event, plume injection height was set at 100 m, 300 m and 1000 m. An injection height of 100 m represents conditions occurring in the late evening, while an injection height of 1000 m models a plume that has sufficient energy to penetrate to the free troposphere. The population distribution of the Huon Valley is shown in Figure 15.

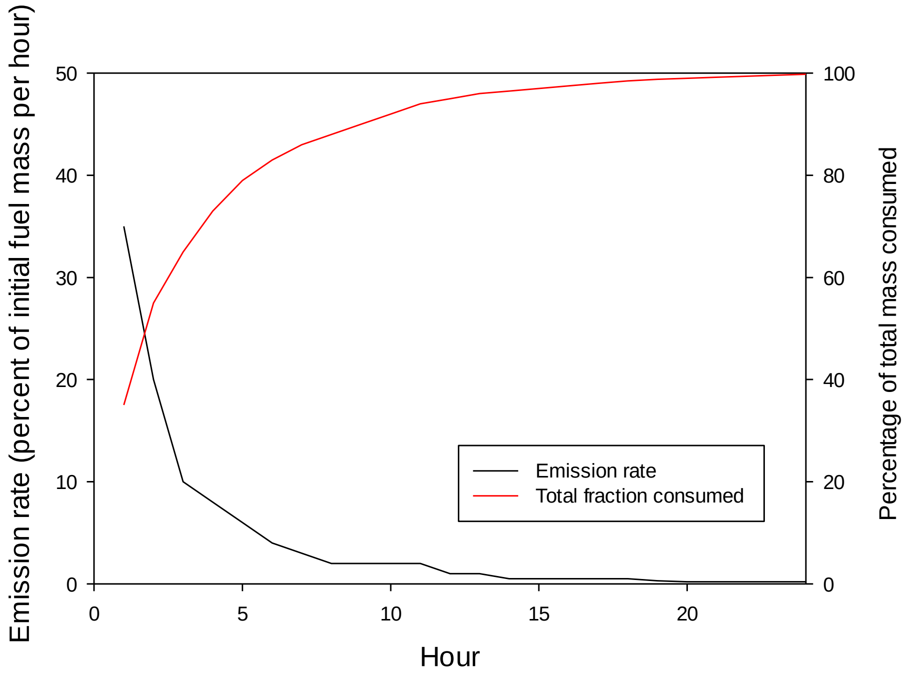

Figure 14 Prescribed rate of fuel consumption and emission rate in high intensity regeneration burn.

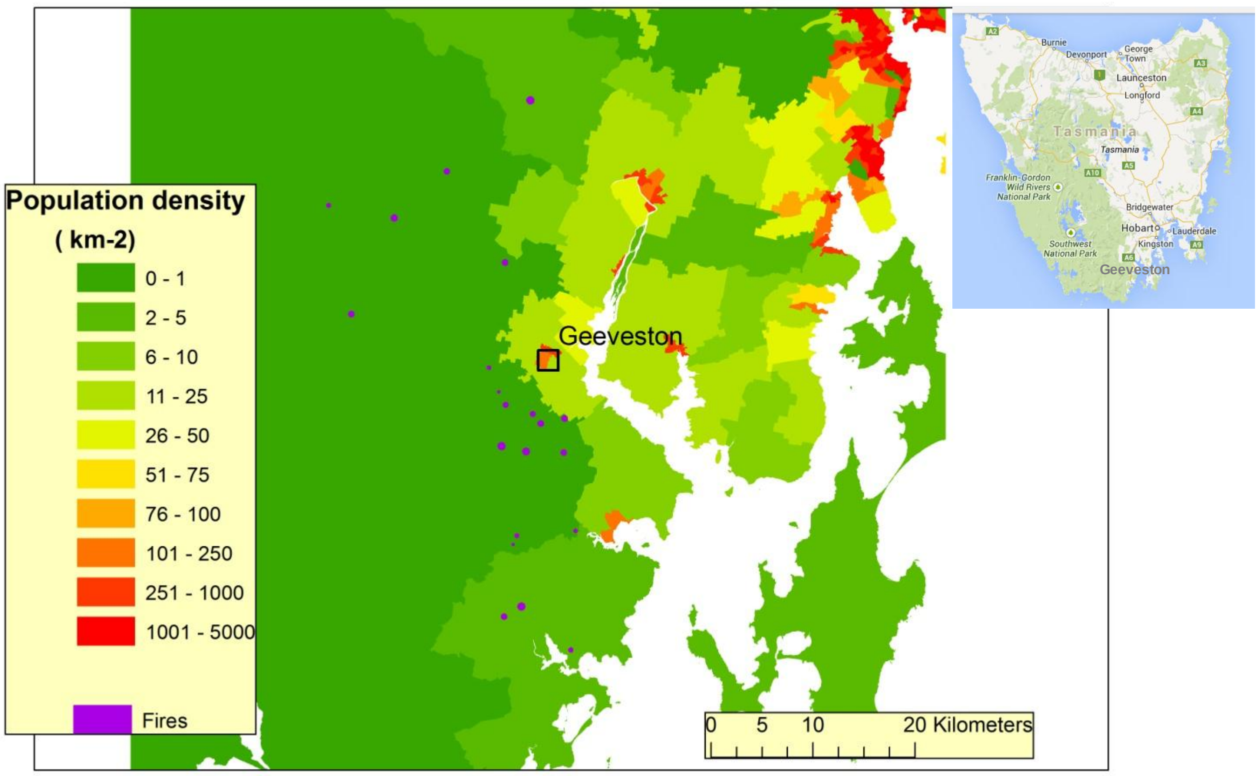

Figure 15 Distribution of population density in the Huon valley. Based on Australian Bureau of Statistics 2011 Census.

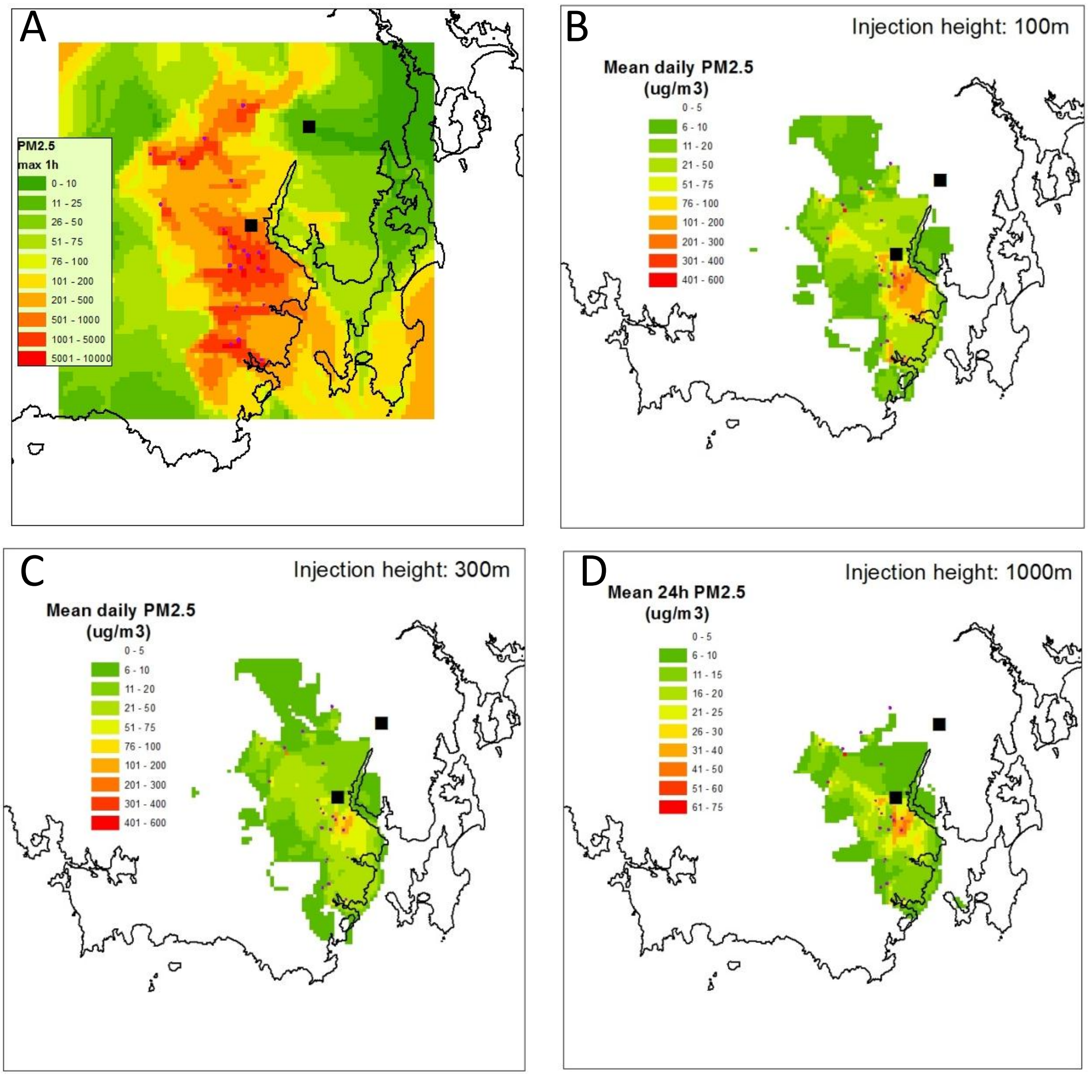

Figure 16 Modelled surface PM2.5 concentration during the 2010 smoke event in the Huon Valley, Tasmania. A: maximum 1-hour PM2.5 concentration, Plume injection height 300m. B: mean 24-hour PM2.5 concentrations during the event, plume height 100 m. B: mean 24-hour PM2.5 concentrations during the event, plume height 300 m. D: mean 24-hour PM2.5 concentrations during the event, plume height 1000 m. The locations of the monitoring stations are indicated by black squares. Grove is the northern most square and Geeveston is the southern square. Locations of the fires are shown by the purple points.

Most of the prescribed burns were south and west of Geeveston, the closest less than 7 km away. The extent of the smoke plume impact is shown by the distribution of maximum 1-hourour average PM2.5 concentrations during the event (Figure 16a), and was largely confined to the lower regions of estuary and the upper reaches of the Huon Valley west of Huonville. The monitoring site at Grove was not impacted. This confirms that the smoke was not widely dispersed, contrary to public perception, but mostly affected uninhabited regions of the Valley. The degree of impact is shown in Figure 16 b, c and d as the mean 24-hour surface PM2.5 concentration. There is little difference in the dispersion patterns for the three different plume heights. Quantitatively, the 100 m injection height, which predicts a mean PM2.5 concentration increase above background at Geeveston of 32 µg m^{-3}, agrees best with the observations at Geeveston of 27 µg m^{-3}. The significant smoke impacts mostly occurred at night suggesting that it was smouldering emissions from the heavy fuel continuing through the late afternoon and evening that contributed most of the observed PM.

3.4 Summary of case studies

The three case studies are summarised and compared in Table 2. The dispersion of smoke from persistent fires, such as the 2006/2007 Alpine fires, is determined by the seasonal climatology, with the likelihood that even when population centres are upstream of the prevailing winds, there will be occasions when dispersion carries plumes to the city airshed. In contrast, severe wildfires of short duration (e.g. Kilmore East fire) develop strong convection columns that disperse smoke high into the troposphere, minimising the risk of ground strike; although local regions downwind of the event will invariably be impacted to some degree. Prescribed burns, because they are managed, can be timed to avoid plume strike on sensitive regions. It is also clear from Table 2 that the magnitude of the smoke emission does not imply the magnitude of the smoke impact. This is particularly relevant to the third case study, which caused significant public comment at the time largely due to the visibility of the plume. Contrary to perception, the modelling of this event showed that smoke did not accumulate in the Huon valley, and, as was confirmed by Meyer et al., (2011) and Reisen et al., (2013) posed only a relatively minor risk to the population.

Table 2 Summary of the relative risk of health impact due to PM2.5 emitted during the fire event. Baseline deaths are the number of deaths that occur due to other factors other than fires. The baseline is the annual deathrate standardised by the age distribution of the Australian population in 2001

					Impacted zone			
Fire	Start date	Days	Area (km^2)	Emission (k t C)	Area (km^2)	Population (1000s)	Baseline (deaths y^{-1})	Impact (deaths)
Alpine	8/12/2006	60	11,400	28,700	212,600	5,370	34,611	84
Kilmore	8/02/2009	1	925	1,340	50,200	270	1,745	0.10
Huon	16/04/2010	6	5.5	25	1,275	4.36	30	0.015

3.5 Ensemble analysis of smoke dispersion from the Kilmore East fire

A key feature of FireDST project (of which the Risk Tool Kit project is a sub-project) is the capacity to analyse the effects of uncertainties in the model inputs. For smoke dispersion, the uncertainty derives from uncertainties in the fire location, rate of spread, plume rise, and uncertainties in the combustion parameters (burning efficiencies, fire patchiness, and emission factors for each fuel class). Although all are important, to demonstrate the feasibility of the probabilistic approach, we considered uncertainty in one area only, the fire ignition point. Fire ignition point was selected as it contributes the greatest uncertainty to the time course and fire progression in the PHOENIX fire spread model, which in turn has large implications for the impact of the fires.

The analysis was performed on for the Kilmore East fire case study, for a high resolution domain comprised of an 80 x 80 grid of 1 km^2 cells centred on Kilmore. The Kilmore fire progression was simulated using PHOENIX five times with different ignition locations; the base-case scenario where the ignition point was reported ignition site; plus four alternatives in which the ignition point was offset by 500 m to the four cardinal points of the compass (North, East, South and West) from the actual ignition site. The hourly concentration maps of each of the key smoke tracers (PM2.5, CO, O_3 and NO_X) for each fire were then overlaid to produce maps of the hourly maximum tracer concentration in each grid cell for the five scenarios (Figure 17) and maps of the percentage overlap of the fire simulations for each species (Figure 18). The former show exposure risk and the latter show the consistency between the scenarios. A full description of the process is contained in French (2014). Production of both the ensemble maps will be helpful in quantifying and displaying the uncertainty in the smoke maps due to variability in the underlying fire.

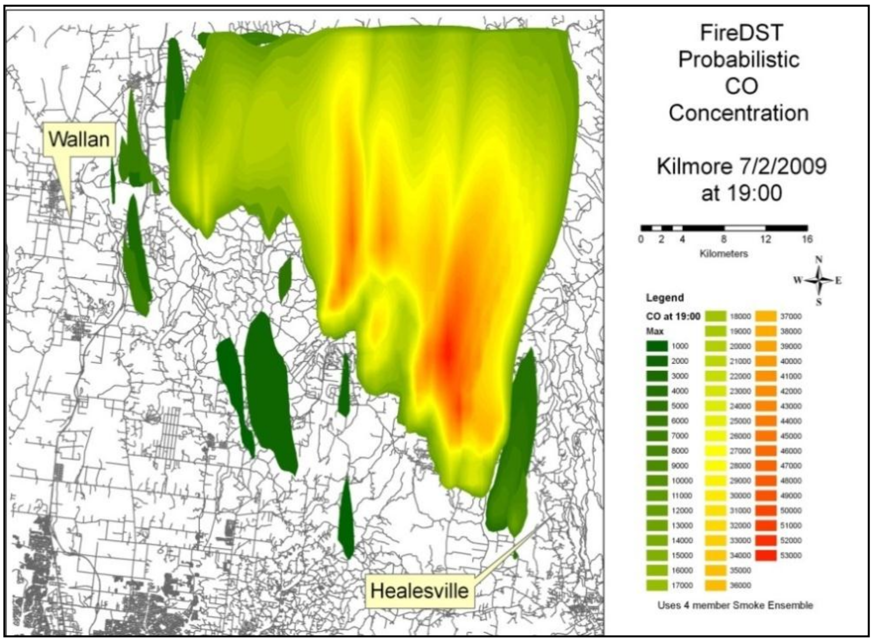

Figure 17 Probabilistic map of the concentration of CO at 19:00 for the four simulations of the Kilmore fire. This concentration map must be used in conjunction with the probability map.

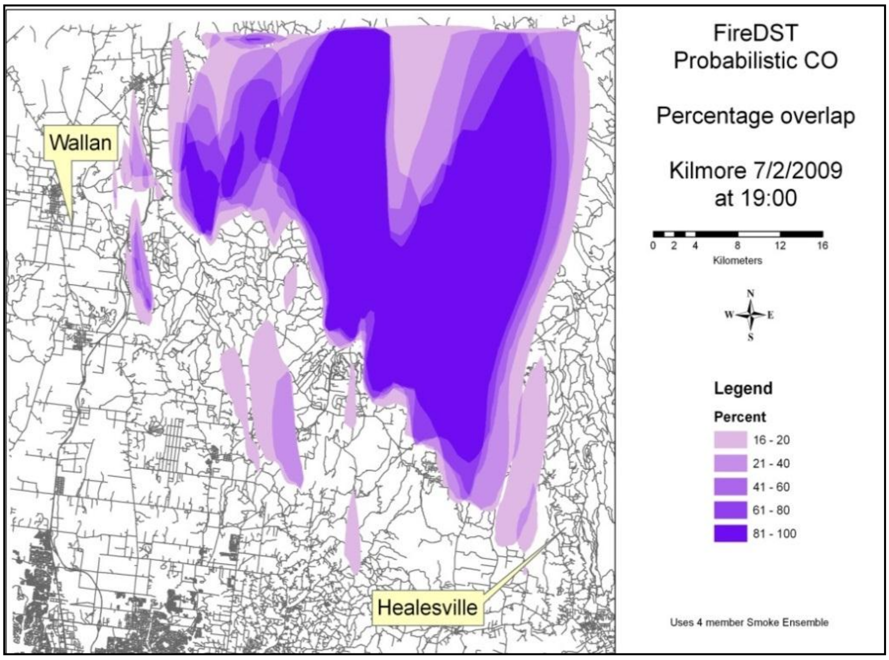

Figure 18 Probabilistic map of the percentage overlap (probability) of CO at 19:00 for the four simulations of the Kilmore fire. This probability map must be used in conjunction with the probabilistic concentration.

4 Tools for agencies

4.1 Minimising smoke impact from prescribed burns

This project involved the development of a tool based on a numerical modelling system that can be used to undertake an assessment of the risk to communities resulting from smoke generated by fuel reduction burning. The modelling system was used in an inverse mode to investigate the relationship between sensitive receptor and upwind source regions. With the Ovens Valley in north-eastern Victoria as an example, we used the chemical transport model to simulate the dispersion of PM2.5 emitted daily between 11:00 and 16:00 during April 2009 from each 3 km x 3 km grid cell in a 50 x 50 cell domain centred on Harrietville. From these data we then assessed the relative impact of each source cell on any receptor cell within the domain. Taking the towns of Myrtleford, Harrietville and Mt Beauty (located in the nearby Kiewa Valley) as test cases, we found that the greatest likelihood of smoke impact was from fires close to the receptor cell, however more distant sources were also significant, with the strongest located on the valley slopes. Vegetated source areas in the bottom of the valleys and on ridges had the least impact. Harrietville and Mt Beauty, which lie in different valleys, nevertheless had similar source risk distributions, in contrast to Myrtleford, which lies downstream of Harrietville on the Ovens River, and has a totally different source risk profile. Significantly, there was no indication of a prevailing flow for any of the three receptor cells i.e. the spatial pattern of impact risk for the test month, April 2009, (Figure 19) was not predictable from topography or distance from the source and therefore the modelling approach offers promise of developing detailed climatologies of smoke dispersion relevant for individual fire districts.

Development of the method will continue beyond the life of the current project. This system can be used to provide a self consistent framework for testing smoke transport screening approaches for use by fire managers for planning prescribed burning schedules.

4.2 Users guide for using TAPM for modelling smoke dispersion from defined fire events

Some of the modelling tools applied in the Risk Tool Kit project are sufficiently well developed to provide potential for application in regional offices. The CSIRO dispersion model TAPM, which was developed in part for use by environmental consultants, is sufficiently robust, simple to apply, and fast to run on a personal computer, to be of use for planning and analysis by staff in regional offices. A Users Guide describing how to configure and run the model for several practical applications has been prepared by Meyer (2013).

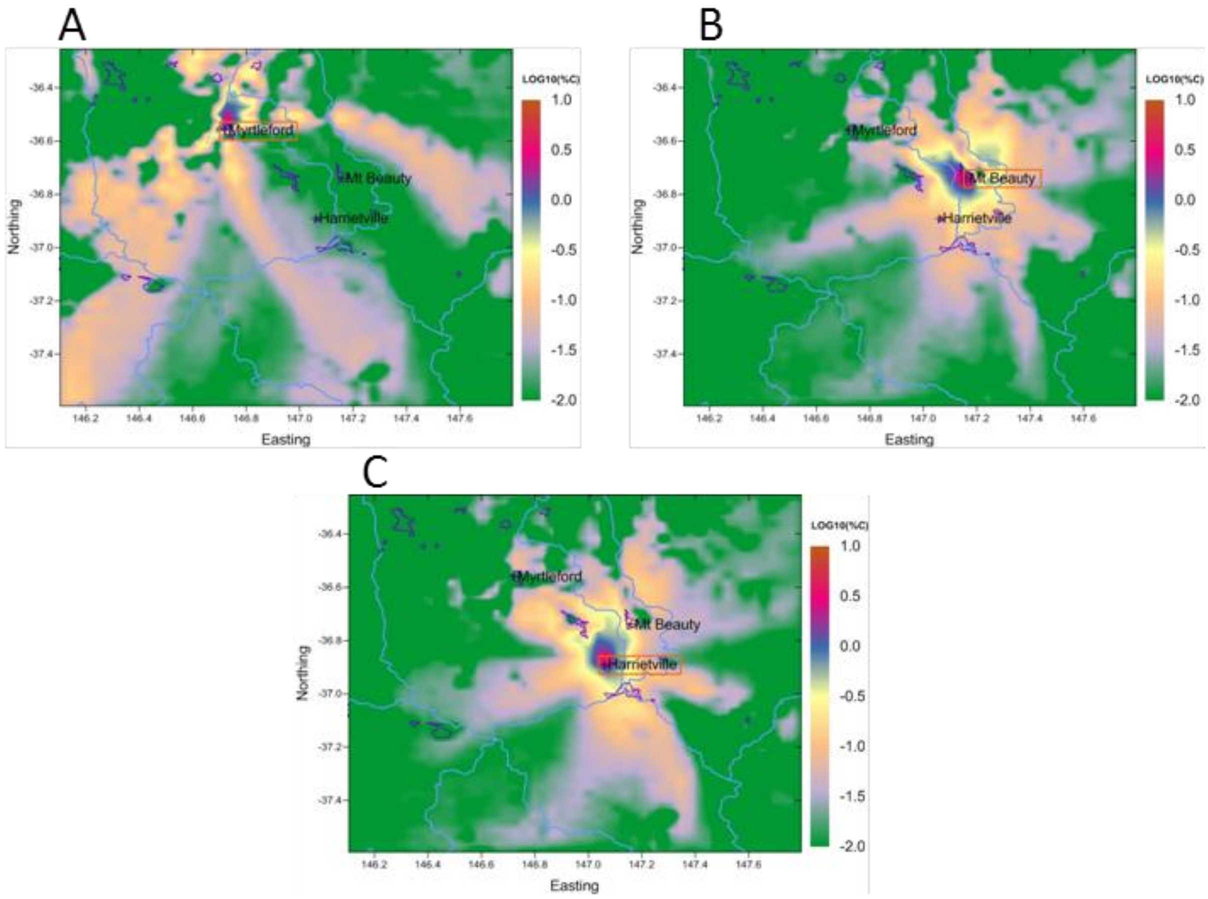

Figure 19 Relative contribution (plotted on a log scale) of smoke emissions released from every 3 x 3 km^2 grid cell in the modelling domain for A: Myrtleford, B: Mt. Beauty, and C: Harrietville

5 Summary and Future Directions

The study has demonstrated that smoke impact on regional populations can, on occasions, be the greatest risk from a fire event, far outweighing the direct risks at the fire front. Given the imperative to increase the extent of prescribed burning to 5% of public managed lands in Victoria, smoke impact on people is only likely to increase. New modelling approaches and applications of extant modelling systems have been developed to assist fire managers and planners to limit the risks of these impacts and will continue to be a priority of the FireDST team members beyond the life of the current CRC.

Smoke dispersion models are computationally intensive, and are not currently in a form that can be incorporated directly into the Risk Tool Kit for tactical applications. However the Department of Primary Industries (DEPI) Smoke Transportation Modelling and Smoke Emission Modelling Project currently in progress, developed in part from the Risk Tool Kit experience, shows promise of delivering a modelling framework suitable for operational planning within the next three years. This involves adapting the US modelling framework BlueSky to southern Australia.

The current emissions model, though effective, is weak in a range of areas. These include

1. Quantifying the hourly time-course of combustion within the fire scar boundaries. Where the time-course of fire spread is resolved to an hour or less, the challenge is to accurately describe the emissions of each of the fuel components. While consumption of the fine fuels can be adequately modelled, the duration and extent of combustion of other fuel components (coarse woody debris (CWD), elevated fuels and the canopy) are poorly known; and
2. The emission factors of PM2.5, greenhouse gases and reactive organic compounds that affect plume chemistry are relatively poorly characterised for Australia forests and woodlands. The first can be addressed through linking the fire spread predictions of area and intensity to the smoke emission model. With improved knowledge of combustion dynamics in CWD and living fuels and the combustion processes that determine emission factors, it may be possible to estimate variations in emission factors with properties of fire intensity (e.g. flame length and heat release rate).

In addition, there is still significant uncertainty about the health effects of bushfire smoke, so that there is a need to quantify the correlation between population exposure to PM2.5 and morbidity. There is still mainly because of the challenges associated with exposure assessment (e.g. exposure is unpredictable, pollutants are diverse and complex and effect seen in small subgroup of people). Studies examining the health impact of severe bushfire smoke pollution are limited and further research is needed to understand the potentially unique health effects of smoke exposure from bushfires. This will be addressed in the DEPI Smoke Health Impact Study, currently in the second year of a three-year program.

The aim of the DEPI Smoke Health Impact Study, is to assess the cardiovascular and respiratory health effects from exposure to particulate matter air pollutants emitted from bushfire smoke. The study has three components:

1. A review and, to the extent possible, a meta-analysis of published studies of impacts of bushfire smoke on health. This component is complete and has been prepared for publication. Due to the diversity of methodologies applies a detailed meta-analysis was not possible.
2. A retrospective analysis of surface PM2.5 concentration in regional Victoria during the 2006/2007 Alpine fire event and
    a. Hospital admission episodes codes for cardiovascular & respiratory disease;
    b. Emergency department visits for cardiovascular & respiratory disease; and
    c. Ambulance attendance for out of hospital cardiac arrest (non-traumatic).

    The smoke concentrations were modelled in case study 1. With the caveat that the work is still in progress there appears to be an increase in the percentage of emergency department visits for asthma cases correlated with exposure to bushfire smoke (Dennekamp et al., 2014).

3. An analysis of Health impacts from smoke exposure during the 2013/2014 prescribed burning season.

6 CSIRO MAR publications for this project

Peer-reviewed publications

Keywood, M., C. P. Meyer, Iinuma Y. and K. Emerson (2014). What does smoke from wildfires do to air quality and atmospheric chemistry in an Australian city? Atmospheric Environment submitted April 2014.

Meyer, C. P., M. Cope and S. Lee (2013). Assessing the population exposure risk of regional population to smoke from fires. 20th International Congress on Modelling and Simulation, Adelaide, 1st to 6th December http://www.mssanz.org.au/modsim2013/A3/meyer2.pdf.

Meyer, C. P., M. Cope and S. Lee (2013). Smoke impacts from prescribed burning in Victoria; developing a risk climatology, 20th International Congress on Modelling and Simulation, Adelaide, 1st to 6th December http://www.mssanz.org.au/modsim2013/A3/meyer2.pdf.

BFCRC/Internal-reviewed publications

Meyer, C. P. (2012). Review technology transfer options and review project data requirements from agency fire data bases and delivery timelines. Risk Tool Kit Project, Bushfire CRC Progress Report 2.1.2. CSIRO 10p.

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Meyer, C.P., Cope, M., Lee, S. (2013) Assessing population exposure risk to smoke from bushfires. Presentation at AFAC 2013, Melbourne 2-5 September 2013.

Meyer, C.P. (2013) Smoke Dispersion modelling. Presentation at the Research Advisory Forum 8, Perth, May 2013.

7 Acknowledgements

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Ralph Smith (Lead end user)	Manager, Environmental Protection Branch Dept. Fire & Emergency Services (DFES), West Australia
Liam Fogarty	Department of Environment & Primary Industry, Victoria
Dr Simon Heemstra	*New South Wales Rural Fire Service (NSW RFS)*
Mark Chladil	*Tasmanian* Fire Service (TFS)
Fergus Adrian	Queensland Rural Fire Service (QRFS)
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Lyndsey Wright	Research Manager, Bushfire CRC

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FOR FURTHER INFORMATION

Marine and Atmospheric Research
Mick Meyer
t +61 3 9239 4686
e mick.meyer@csiro.au

CONTACT US

t 1300 363 400
+61 3 9545 2176
e enquiries@csiro.au
w www.csiro.au