Local and regional smoke impacts from prescribed fires

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Abstract. Smoke from wildfires poses a significant threat to affected communities. Prescribed burning is conducted to reduce the extent and potential damage of wildfires, but produces its own smoke threat. Planners of

10   prescribed fires model the likely dispersion of smoke to help manage the impacts on local communities. Significant uncertainty remains about the actual smoke impact from prescribed fires, especially near the fire, and the accuracy of smoke dispersal models.

To address this uncertainty, a detailed study of smoke dispersal was conducted for one small (52 ha) and one large (700 ha) prescribed fire near Appin in New South Wales, Australia through the use of stationary and

15   handheld pollution monitors, visual observations and rain radar data, and by comparing observations to predictions from an atmospheric dispersion model. The 52 ha fire produced a smoke plume about 800 m high and 9 km long. Particle concentrations ($PM_{2.5}$) reached very high peak values (>400 $\mu g/m^3$) and high 24 hour average values (>100 $\mu g/m^3$) at several locations next to or within ~500 m downwind from the fire, but low levels elsewhere. The 700 ha fire produced a much larger plume, peaking at ~2000 m altitude and affecting

20   downwind areas up to 14 km away. Both peak and 24 hour average $PM_{2.5}$ values near the fire were lower than for the 52 ha fire, but this may be because the monitoring locations were further away from the fire. Some lofted smoke spread north against the ground-level wind direction. Smoke from this fire collapsed to the ground during the night at different times in different locations. Although it is hard to attribute particle concentrations definitively to smoke, it seems that the collapsed plume affected a huge area including the towns of

25   Wollongong, Bargo, Oakdale, Camden and Campbelltown (~120,000 ha). $PM_{2.5}$ concentrations up to 169 $\mu g/m^3$ were recorded on the morning following the fire. The atmospheric dispersion model accurately predicted the general behaviour of both plumes in the early phases of the fires, but was poor at predicting fine-scale variation

in particulate concentrations (e.g. places 500 m from the fire). The correlation between predicted and observed varied between 0 and 0.87 depending on location. The model also completely failed to predict the night time collapse of the plume from the 700 ha fire.

This study provides a preliminary insight into the potential for large impacts from prescribed fire smoke to NSW
5   communities and the need for increased accuracy in smoke dispersion modelling. More research is needed to better understand when and why such impacts might occur and provide better predictions of pollution risk.

1. Introduction

Smoke from wildfire has caused pollution events in large Australian cities on many occasions. For example, the
10   alpine fires in northeast Victoria in 2009 caused serious smoke pollution across Melbourne on several days over a period of a month (Tham et al., 2009). Pollution from wildfire is recognised as a health issue in Australia (Johnston et al., 2011;Hanigan et al., 2008) and globally (Jayachandran, 2009;Sapkota et al., 2005). As air pollution standard become stricter across Australia and steps are taken to reduce emissions from industrial and transport sources, so the relative contribution of wildfire smoke to total particulate matter becomes greater.

15   Prescribed burning is intended to reduce the risks to the community from unplanned fires. Primarily, this is focussed on reducing loss of life and damage to property. However, in terms of smoke pollution, there is a trade-off between the reduction of smoke from reduced wildfire activity resulting from prescribed burning treatment and the pollution from the prescribed fires themselves. The distribution of smoke impact from prescribed fires is likely to differ from that of wildfire. Wildfires are usually large with strong convective forces injecting smoke
20   high into the atmosphere, which can lessen the impact to the local community, but can affect communities up to hundreds of kilometres away (Sapkota et al., 2005;Jayachandran, 2009). In comparison, the smoke from prescribed fires may linger near the fire. Also, wildfires are often located in remote areas such as National Parks while prescribed fires are often located within or near the communities that fire management authorities intended to protect. This means that prescribed fires could have high local impacts.

25   While the potential hazard from prescribed fire smoke has been recognised (Haikerwal et al., 2015), most of our understanding of prescribed fire smoke exposure stems from research into wildfires (Reisen and Brown, 2009). There have been studies of firefighter exposure at prescribed fires (Miranda et al., 2010;Adetona et al., 2013;Reisen et al., 2011) but the effect of prescribed burning on smoke exposure to the local community has

rarely been addressed in research (Reisen and Brown, 2009). To our knowledge there have only been two studies attempting to quantify the spatial distribution of prescribed burn smoke pollution close to the source (Meyer et al., 2013;Pearce et al., 2012), and only one of those was empirical (Pearce et al., 2012). Pearce et al (2012) studied 55 prescribed fires at Savannah River (Georgia, USA) using a grid of pollution monitors and

5   found that $PM_{2.5}$ concentrations reduced to background levels within 2 km downwind of the fire (all other factors being equal).

Given the risks of smoke pollution affecting residents, many fire management agencies consider likely smoke impacts in their prescribed fires planning, using atmospheric dispersion models to predict the spread of particulate matter (PM) in the air. However, the use of dispersion models for predicting fire smoke is

10   problematic because they are (mostly, if not all) designed to predict large-scale movements of pollutants (Pearce et al., 2012) and the models are not well validated against observational data (Price et al., 2012).

Recent research on the effectiveness of prescribed burning has shown that over the long term it increases the total area burned in Australian eucalypt forests because approximately 3 ha of prescribed burning are required for every ha reduction in wildfire area (Price and Bradstock, 2011;Boer et al., 2009;Bradstock et al., 2012). A

15   similar increase in total area burnt due to prescribed burning is likely to occur in many regions globally (Price et al., 2015). However, this does not necessarily mean that the smoke impact is 3 times greater because less fuel is consumed in a prescribed fire. Nevertheless, the increased area burnt and the possibility for high local impact from prescribed fire smoke highlights the need to understand more about the dispersal of prescribed fire smoke.

This study measured the actual dispersal of prescribed fire smoke, primarily to determine the air quality impact

20   on residents and the accuracy of the dispersion model prediction. Two prescribed fires were studied, one large (700 ha) and one small (52 ha). A mix of permanent, temporary fixed and handheld monitors were used to measure the temporal and spatial variation of the particle pollution at distance from within the fire perimeter to 30 km away. This was supplemented with measurement of the smoke plume via visual observation and, 3-D rain radar data. The observations were compared to predictions from the TAPM atmospheric dispersion model

25   (Hurley et al., 2005). Anecdotal evidence suggests that the local impact could be high and the accuracy of the model be low for a variety of reasons. These include: a) The injection height is likely to be lower than assumed in the model, meaning most of the smoke will remain in the local area; b) light winds favoured for prescribed burning also make the smoke linger; 3) temperature inversion during the night are common in the prescribed burning season, and this might increase pollutants at ground level long after the fire has ceased.

2. Methods

Two prescribed fires were observed in the field around the Cataract Scout Camp near Appin NSW (Figures 1 and 2). The vegetation around the camp is 'Sydney coastal dry sclerophyll forest' (Keith, 2004) comprising a mix of Eucalypt trees and shrubby understorey. The first fire was 52 ha, lit on 22nd August 2015, targeted patches of forest within the scout camp. The second was 700 ha, lit on 9th October 2015 and burnt the area surrounding the camp in an arc from north through west to south. Pre-burn fuel assessments estimated the surface fuel load to be 15 t/ha and total fuel load to be 23 t/ha across the burn areas (S. Chadwick, pers. comm.).

A range of measurements were collected at the fire ground and surrounding areas:

1. Small monitoring stations were placed at two locations at the Scout Camp by the NSW Office of Environment and Heritage (OEH). The stations were within 50 m of burnt forest in the 52 ha fire but 220 m and 800 m away from the 700 ha fire (Figure 2). Each station was equipped with a variety of monitoring instruments, including a TEOM monitor (reference monitor hereafter) for measuring concentrations of $PM_{2.5}$ (particulate matter less than 2.5 microns in diameter). Reported $PM_{2.5}$ values were 6-minute averages.

2. A handheld particle monitor (Dustrak II) was used to sample $PM_{2.5}$ at a variety of locations around the outside of the fire, with distances ranging from 20 m to 18 km from the flames (Figure 2). Six locations all within 1 km of the fires were sampled repeatedly approximately twice per hour and more distant locations were sampled once or twice during the day. Due to the different areas burnt, most locations were closer to the 52 ha fire than the 700 ha fire (mean distance 332 m cf 693 m). Sampling was conducted until 6 pm of the burning day, with a single repeat measurement at ~8 am on the morning after the fire. Readings from the DustraK II monitor were 1-minute averages. For locations with several $PM_{2.5}$ observations, peak and 24 hour average (beginning at 10 am) values were estimated. In the case of the handheld monitor, only locations with three or more $PM_{2.5}$ observations were used, and the periods between observations was assumed to have a value of 4 µg/m^3 (the mean pre-fire value). This is a simple, but conservative estimate of possible daily exposure.

3. The handheld monitor was calibrated against a reference monitor (TEOM) by taking 13 measurements next to the reference monitor, spread across the two fires and at different times of the day, and using linear regression to estimate the relationship between the two monitors.

4. For the period of the 700 ha fire, air quality data were also obtained from OEH for 11 long-term monitoring stations located in the Illawarra and Sydney metropolitan regions(between 15 and 60 km from the fires). Most of the 11 stations measured PM_{10} (particular matter less than 10 microns in diameter) but not $PM_{2.5}$ concentrations at hourly intervals. ANOVA was used to test whether PM_{10} and $PM_{2.5}$ levels were elevated at each monitor during the fire (up to 6 pm) or in the night (6 pm – 10 am the following morning) compared to the full record (from 00.01 hours 9/10/2015 to 23.59 hours 10/10/2015).

5. Visual observations of the fires were made from a suitable vantage point (920 m from the 52 ha fire and 4.5 km away from the 700 ha fire), consisting of the vertical angle of the top of the plumes (measured using a clinometer) and compass bearings for the top, left and right side of the plumes.

6. Radar data from the Bureau of Meteorology rain radar at Appin (approximately 5 km from the fires) was examined using the 3-D Rapic visualisation program (Purdham, 2007). The data consists of reflectance values for radial pixels of 14 'slices' at different vertical tilt angles from the radar. The software can display a 3-D profile (e.g. cross section or planar view) of the plume at 6 minute intervals during the day. Rain radar has been used in several studies to measure the vertical and horizontal spread of wildfire smoke (Jones and Christopher, 2010;Fromm et al., 2012;Cruz et al., 2012). However, radar detects objects at least 100 μm in diameter, so is measuring larger particles in the smoke than the $PM_{2.5}$ fractions recorded by the other instruments. The data were used to produce a series of polygons outlining the extent of the plumes at several times of the day.

7. An automatic weather station was located inside the fire for the 700 ha fire. It was not deployed for the 52 ha fire so measurements from a permanent meteorological station at Lucas Heights (24 km northeast of the fire) were used (Bureau of Meteorology, unpublished data). In addition, upper atmospheric data were sourced for the Sydney Airport weather station (Bureau of Meteorology, unpublished data), 46 km from the fires, from which the C-Haines index of atmospheric instability (Mills and McCaw, 2010) was calculated for 7 am local time. This index, and its precursor (the Haines index) have been shown to be correlated with extreme fire behaviour, and in particular with the development of huge pyro-cumulous smoke plumes (Mills and McCaw, 2010).

8. Smoke plume predictions were generated by the Rural Fire Service smoke forecasting system 24 hours before the proposed burn. The smoke forecasting system couples the TAPM (a commonly-used air quality dispersion model in Australia, (Hurley et al., 2005) with CCAM (a weather forecasting model, (Engelbrecht et al., 2009;McGregor, 2015). The system uses forecast weather up to 5 days before the fire

with an assumed injection of smoke at a height of 1000 m and produces predicted grids of PM$_{2.5}$

concentrations with 1 km resolution, for 18 vertical strata (10 – 8000 m) at hourly intervals until 24 hours

after the fire. The RFS system applies a standard equation from the National Pollutant Inventory (1999) to

convert the area and fuel load of the fire into an emission rate and duration entered into TAPM. The

5   correlation coefficient was calculated between predicted hourly values at 10 m elevation and actual readings

from four locations near the fires.

9.   Some supplementary data were also obtained from carbon monoxide monitors in near the entrance of shafts

at two nearby coal mines operated by a mining company.
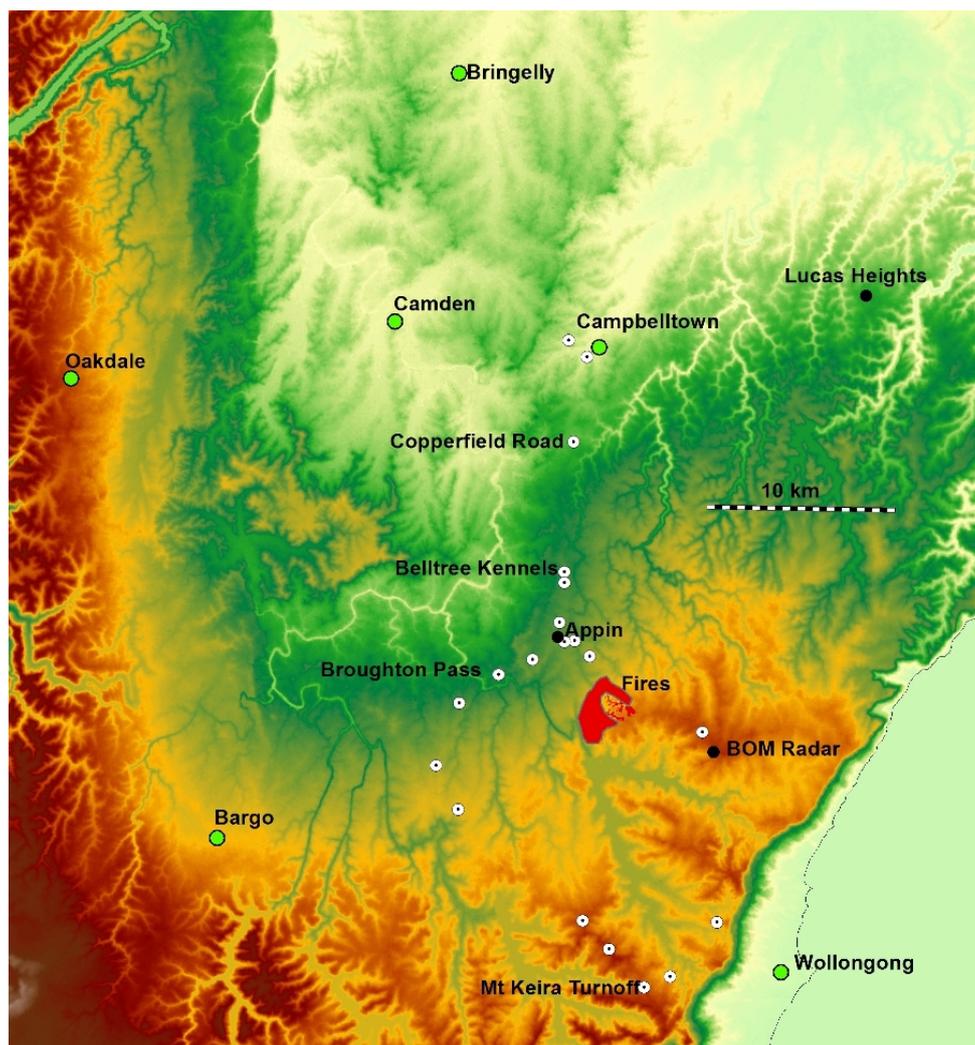
Figure 1: Location of the fires (red polygons) and relevant landmarks. Green dots are permanent air
quality monitors, white dots are readings from handheld air quality monitor. Black dots are other landmarks.
The background is elevation (pale green = 0 m, brown = 700 m).
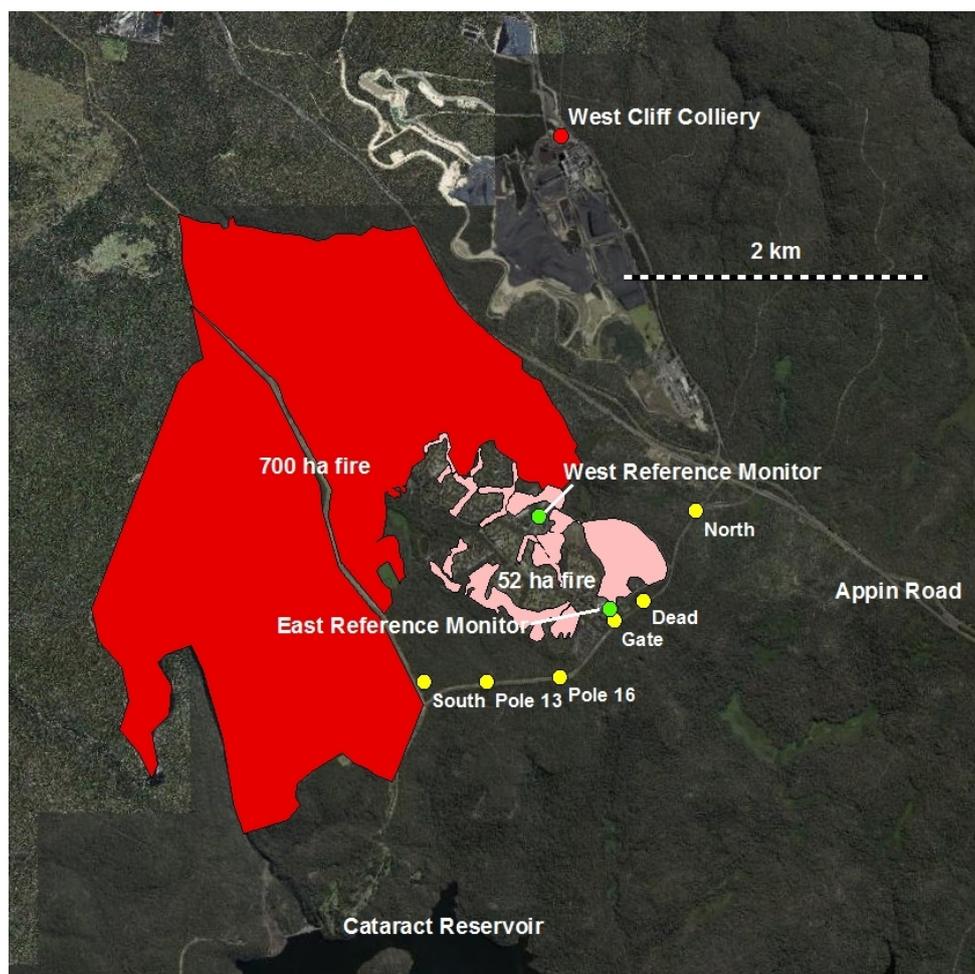
Figure 2: Detail of the fires and nearby air quality measurement locations. Green dots are reference air quality monitors, yellow dots are repeat measurements using the handheld air quality monitor. Red dots are other permanent monitors. The background is air-photo mosaic from Land and Property Information NSW.

Table 1: Locations for particulate monitor readings, along with the maximum PM$_{2.5}$ reading, time of that reading and 24 hr average value Locations other than the reference monitors were estimated assuming value of 4μg/m^3 for non-observed periods.

Location	ID (see map)	Easting	Northing	52 ha fire				700 ha fire			
				Reading Count	Max PM$_{2.5}$	Time of Max	24 hr mean	Reading Count	Max PM$_{2.5}$	Time of Max	24 hr mean
Eastern Reference Monitor	1	299739	6208372	240	372.3	11.15	100.7	240	871	11.05	122.4
Western Reference Monitor	2	299293	6208941	240	1105	15.10	116.7	240	200	11.4	33.6
North	3	300321	6208976	12	13.2	14.00	4.59	13	35.1	15.25	5.1
Dead	4	299976	6208379	16	1006	13.40	209.7	11	66.1	11.15	8.3
Gate	5	299784	6208251	13	531.5	14.07	78.1	15	164.8	11.17	19.0
Pole 16	6	299428	6207880	11	862.6	14.51	51.5	14	195.9	11.55	22.3
Pole 13	7	298951	6207849	11	195.8	14.54	22.1	11	216.2	14.09	23.6
South	8	298534	6207849	11	13.4	14.18	4.94	15	752.1	10.57	40.9
Cataract Road	10	298258	6208293	0				4	275.4	14.05	18.9

3. Results

3.1 Calibration of the Dustrak monitor

The 13 calibration points taken when the portable (handheld) monitor was co-located with the reference monitor spanned a range from 4 to 221 µg/m^3 recorded on the reference monitor and 18 to 1163 µg/m^3 on the portable monitor. The calibration provided a correction factor of 0.223 for the portable monitor (i.e. it overestimated PM$_{2.5}$ levels by nearly a factor of 5), with a correlation coefficient of 0.834.

3.2 Plume characteristics

The 52 ha fire was lit at 9.45 am on August 22nd 2015, and was complete by approximately 5 pm. The wind direction (at Lucas Heights) was 300° (north-westerly) at ignition and gradually swung to 340° (northerly) until 3 pm when it remained stable until midnight. Wind speed was generally moderate (5-15 km/hr) (Figure 3a). The maximum ambient air temperature was 26.7^{o}C and the minimum relative humidity was 33% (at 3.20 pm) on the day. The C-Haines index (Mills and McCaw, 2010) of atmospheric instability at 7 am at Sydney Airport was 6.33 (which is the 80th percentile value for the index at that station; BOM unpublished data), indicating a relatively unstable atmosphere. Visual observations estimated that the plume developed about 30 minutes after ignition and gradually grew to its maximum height of approximately 460 m at 12.30 pm, thereafter gradually reducing to be less than 200 m by 4.30 pm (Figure 4a). The width peaked at 3900 m at 12 noon. The rain radar data detected a plume extending southeast to a maximum of 9 km (peaking at 11.24 am, Figure 5a). The maximum plume height recorded from the rain radar was 900 m at 1.34 pm.
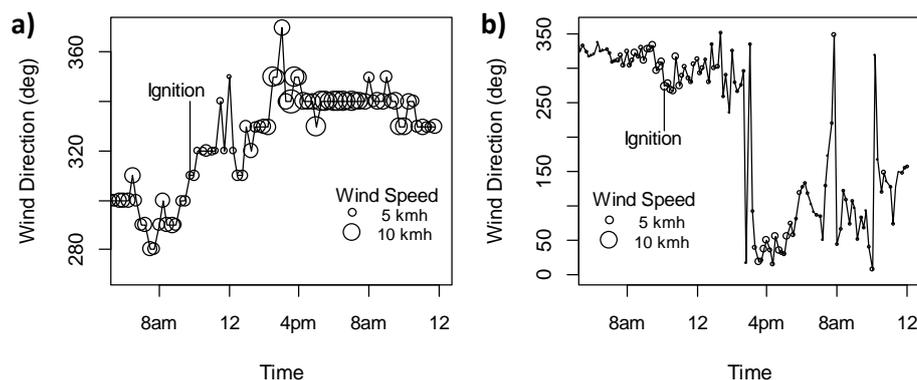
Figure 3: Wind direction and speed recorded during the fires: a) from Lucas Heights for the 52 ha fire and b) from the mobile weather station for the 700 ha fire.

The 700 ha fire produced a plume that was higher, far more extensive and lasted longer than the 52 ha fire. It was lit at 10.15 am on October 9th and smoke was still rising at 6.45 pm (near sunset). Aerial incendiaries were dropped within the burnt-out perimeter at about 4 pm to ensure the completion of the burn within one day. The wind at the fire ground was light and variable, being westerly at the ignition time and shifting to north-easterly toward 3 pm, and never exceeding 5 km/hr for the data collection period (Figure 3b). Maximum ambient air temperature was 28.1°C and minimum relative humidity was 36% on the day (similar conditions to the 52 ha fire). The 7 am C-Haines index was -0.95 (the 7th percentile value), indicating a very stable atmosphere. The plume was visible almost immediately and built gradually to a height of ~1000 m at 1 pm (Figure 4b). The plume grew rapidly at 2.45 pm to 1400 m altitude and then again at 3.45 to 2200 m (probably coinciding with aerial incendiaries). The plume remained in the same general location, to the south of the fire, but gradually expanded all day to reach a maximum visual width of 8.5 km. Notably, from about 2.30 pm, a section of the plume above ~1000m altitude began to spread north. The general size and location of the plume in the early hours was confirmed by the rain radar, which detected a relatively stable, circular plume above and slightly to the south of the fire with a diameter of ~6 km (Figure 5b). The height of the plume observed on radar was 1000 m for most of the day, with a significant boost in the late afternoon, reaching a maximum height of 2000 m at 4.48 pm.
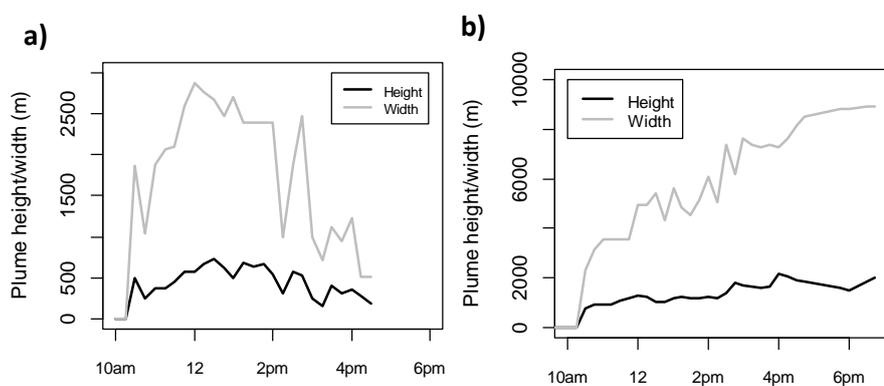
Figure 4: Visual observations of the development of plume height and width for a) the 52 ha fire and b) the 700 ha fire.
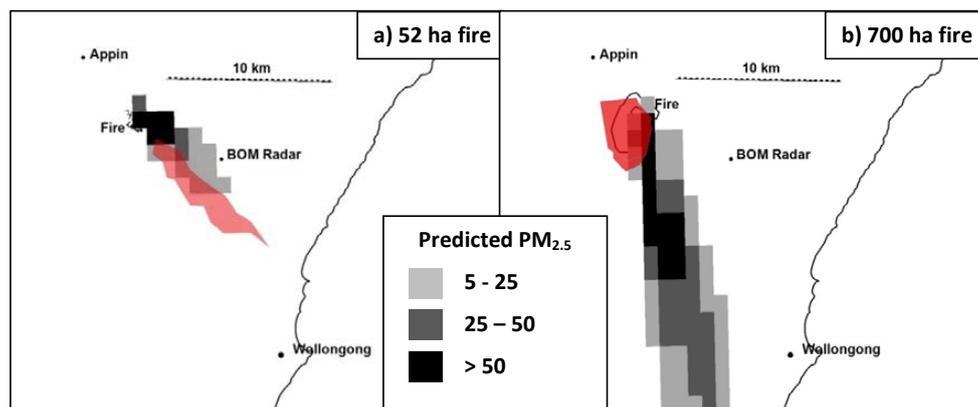
Figure 5: Smoke plumes for the each fire as observed from the 3-d radar data at 12 noon and predicted surface PM$_{2.5}$ concentrations from the dispersion model.

3.3 Pollution measurements

The two reference air quality monitors near the fires showed wide fluctuations in pollutant levels which were partially in synch with each other (Figure 6a and b, Table 1). In both fires, PM$_{2.5}$ concentrations reached > 900 µg/m^3 at the western monitor while the fire was alight, but less than 400 µg/m^3 at the eastern monitor. Night time peaks were observed at both monitors in both fires, at about midnight for the 52 ha fire and 4 am for the 700 ha fire (Figure 6a and b). In both cases, the concentrations were ~200 µg/m^3. Light rain fell on the fire ground at about 6 am on 23rd August and no pollution was detected from the 52 ha fire after this. Average 24 hour concentrations at the western and eastern reference monitors were 117 and 101 µg/m^3 respectively for the 52 ha fire and 122 µg/m^3 and 34 µg/m^3 for the 700 ha fire (Table 1).

The handheld monitor showed high PM$_{2.5}$ concentrations at downwind locations close to both fires (with no smoke pollution upwind), but the patterns at each location differed between the fires because the 700 ha fire was further from the monitoring locations and the wind tended to be slightly more northerly (Figure 6 c and d). Several locations close to the 52 ha fire recorded peak concentrations > 500 µg/m^3 and 24 average concentrations > 50 µg/m^3 (Table 1). Only one location recorded concentrations > 500 µg/m^3 in the 700 ha fire, and none had average 24 values >50 (Table 1).

A survey further afield revealed that a shroud of smoke from the 700 ha fire had settled in the catchment of the Cataract Dam to the south of the fire, with accompanying high PM$_{2.5}$ concentrations, including a 1 minute reading of 45 µg/m^{3}at Mt Keira (14 km south of the fire) at 2.38 pm. There was also a smoke haze in

Wollongong which may have been due to the fire (though no readings were taken there). The rain radar did not detect this southward spread of ground-level smoke. A survey along roads in the evening revealed an obvious shroud of wildfire smoke and high concentrations approximately 4 km west of the fire (108 $\mu g/m^3$ at 6.51 pm at 550 Wilton Road, 89 $\mu g/m^3$ at 6.55 pm at Broughton Pass (see Figure 1). A revisit to the fire area on the morning of 10th October found low $PM_{2.5}$ concentrations in the immediate vicinity of the fire, but high concentrations at more distant locations northwest of the fire, including 169 $\mu g/m^3$ at Appin Colliery (8.40 am) and 37 $\mu g/m^3$ in Appin town centre at 8.42 am. An obvious shroud of smoke was present at the Belltree Kennels 5.5 km north of the fire, recording 157 $\mu g/m^3$ (9.02 am) while at Copperfield road on the southern end of Campbelltown (12 km north of fire) the reading was 52 $\mu g/m^3$ (9.06 am) (Figure 1). Concentrations in central Campbelltown were lower (21.0 $\mu g/m^3$ and 7.4 $\mu g/m^3$ at Hyde Park and the Freeway intersection respectively).

Particulate data from the regional air quality network confirmed that smoke from the 700 ha fire had settled at ground layer in several distant areas at different times (Table 2). It is difficult to isolate a smoke effect in the PM signal from these permanent monitors because several of them have high urban background concentrations that fluctuate during the day. Nevertheless, a combination of the presence of sudden peaks and statistical comparison of periods showed that monitors to the south of the fire (Wollongong and Bargo) had significantly elevated PM levels during the fire (which may be a normal, smoke free pattern), the monitor to the west had a sudden spike at 8 pm (Oakdale, peak PM_{10} level = 102.9 $\mu g/m^3$), and monitors to the north had greatly elevated levels later during the night (Camden, Campbelltown and Bringelly). No other fires were reported in the region to explain these peaks (S. Chadwick, pers. comm.). Monitors even further north in Sydney also showed significantly elevated values at night, but these were lower than those in Cambelltown and Camden. The 24 hour average PM concentrations were much lower than the peaks (last column in Table 2), with none of the $PM_{2.5}$ monitors sustained levels above 25 $\mu g/m^3$ and none of the PM_{10} monitors above 50 $\mu g/m^3$. Supplementary carbon monoxide data from two mines showed a strong peak at Appin (2 km northwest of the fire) from 11.30 pm lasting until 9 am (maximum level 15.0 ppm, 24 hr mean 3.9 ppm) and at Westcliff (500 m north) between 2.30 am and 4.30 am (maximum level 11.5 ppm, 24 hr mean 1.7) (Figure 6e). It seems that smoke affected the ground level in downwind directions probably as far as Wollongong (15 km) during the fire, and the fire also generated a large plume which collapsed overnight affecting places as far as 30 km from the fire, beginning before 7 pm to the west of the fire and after midnight to the north.
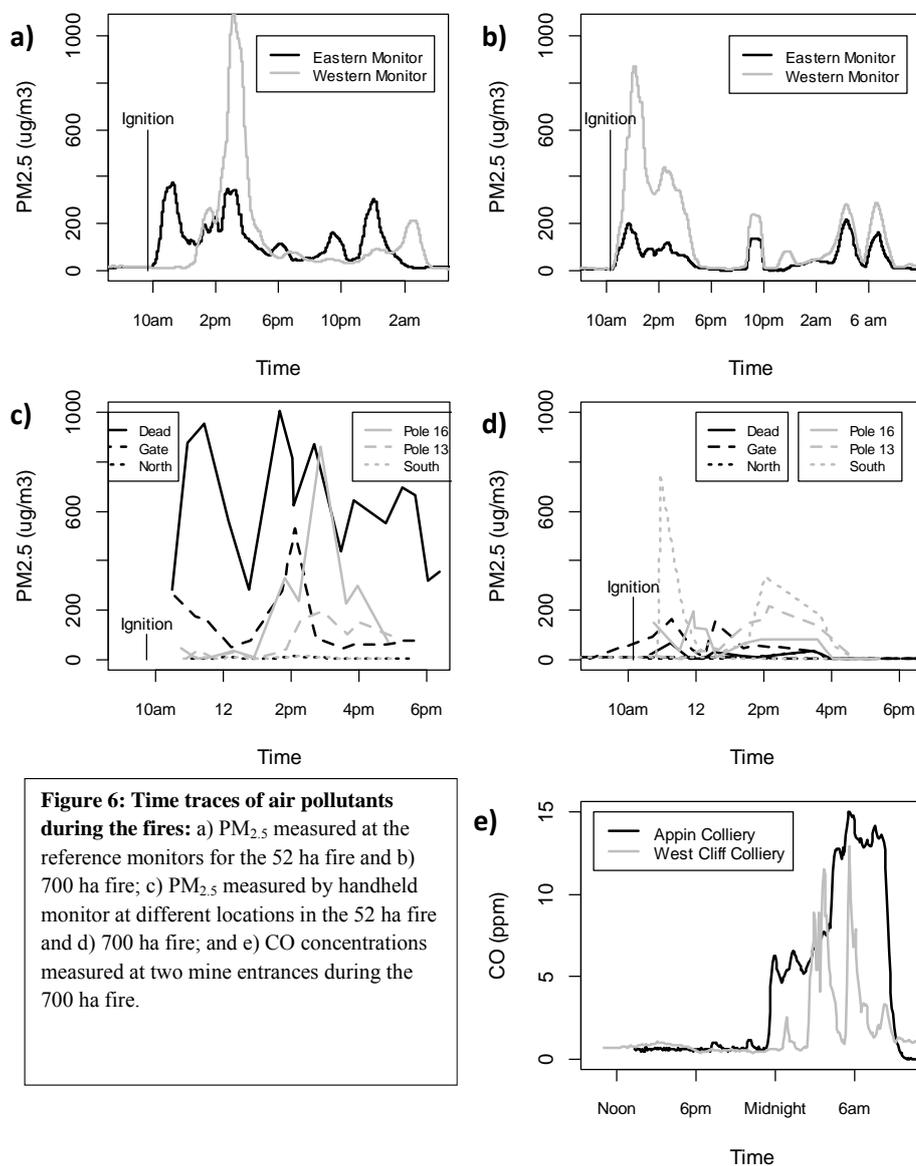
Figure 6: Time traces of air pollutants during the fires: a) PM$_{2.5}$ measured at the reference monitors for the 52 ha fire and b) 700 ha fire; c) PM$_{2.5}$ measured by handheld monitor at different locations in the 52 ha fire and d) 700 ha fire; and e) CO concentrations measured at two mine entrances during the 700 ha fire.

Inspection of the aerological diagrams for 6 am and 4 pm at Sydney Airport suggests that smoke was capped by

an inversion layer at about 1300 m all day with a complex wind shear developing in the afternoon around the

boundary layer below the inversion (Simon Louis, pers. comm.). This atmospheric condition may have been

responsible both for the northerly spread of the lofted air and the subsequent collapse to the surface.

Table 2: Summary of particulate observations from the permanent air quality network. The data are the mean values in four periods: before the 700 ha fire (Midnight 9/10/2015 – 10 am), during the fire (10 am – 6 pm), during the night (6 pm – 10 am 10/10/2015) and the following day (10 am – 6 pm). 'Raised During' and 'Raised at Night' report ANOVA tests comparing two particular periods to all the others. P values are reported (* = p < 0.05, *** = p < 0.001).

Monitor (measure)	Before fire	During Fire	Post-fire night	Following day	Raised During	Raised at Night	24 hr mean
South of Fire							
Bargo_PM_{10}	16.1	24.6	17.0	17.1	*		19.5
Wollongong_PM_{10}	21.7	33.3	25.3	24.6	*		28.0
Wollongong_$PM_{2.5}$	6.9	7.8	10.7	8.1			9.8
West of Fire							
Oakdale_PM_{10}	17.5	18.7	25.8	21.5			23.4
North of Fire							
Camden_PM_{10}	22.2	29.1	37.0	22.3		***	34.5
Camden_$PM_{2.5}$	11.8	8.2	24.1	11.3		***	18.8
Campbelltown_PM_{10}	23.8	38.5	41.1	22.0		***	40.6
Bringelly_PM_{10}	27.1	32.8	36.0	25.2		*	34.9
Liverpool_PM_{10}	23.5	32.7	35.8	24.6		***	34.8
Liverpool_$PM_{2.5}$	12.0	4.5	22.2	11.0		***	16.3
Sydney City							
Chullora_PM_{10}	22.7	25.7	28.8	24.0		*	27.8
Chullora_$PM_{2.5}$	11.1	6.0	12.5	11.6			10.3
Earlwood_PM_{10}	22.9	24.4	28.6	25.3		*	27.2
Earlwood_$PM_{2.5}$	9.0	6.7	11.2	11.4			9.8
Randwick_PM_{10}	21.3	27.8	25.8	20.3	*	*	25.4
Rozelle_PM_{10}	23.3	24.8	26.1	23.6			25.7

3.4 Comparison with the smoke plume predictions

As shown in Figure 5, the smoke plume forecasting system predicted the surface smoke (as PM2.5) to spread

downwind much as it actually occurred for the first several hours of each fire. However, the model predicted

some northward spread of surface smoke that did not occur. For the 52 ha fire, the model predicted that $PM_{2.5}$

concentrations would continue to rise until 3 pm (peaking at 111 µg/m³) and remain above 25 µg/m³ in areas up

to 7 km SE of the fire until 8 pm before reducing and shifting to the northeast of the fire by midnight (predictions for noon are shown in Figure 5) . The correlation between predicted and actual values was 0.559 at the east monitor, 0 at the west monitor, 0.396 at the "North" location and 0 at "Pole 16". The ratio (predicted/actual) of values at the four locations ranged from 0.15 to 9.8 (large under-estimate to large overestimate).

For the 700 ha fire, the model predicted ground level $PM_{2.5}$ concentrations to be high in a narrow plume to the south of the fire, remaining above 25 $\mu g/m^3$ at distances up to 12 km away until 5 pm, much as actually happened (though both the visual observation and the rain radar indicated that the majority of the plume remained near the fire). By 7 pm, predicted concentrations had reduced to below 25 $\mu g/m^3$ in all places and had become wider (less directional). By 11 pm, predicted concentrations were everywhere below 1 $\mu g/m^3$, whereas in reality the plume was collapsing by this time with high concentrations in many areas. The correlation between observed and predicted values was high at both reference monitors (r = 0.867 and 0.646, n=13) and "Pole 16" (r = 0.679). However, there was no correlation at the "North" site (r = 0, n = 6). The ratio (predicted/actual) of values at the four locations ranged from 0.13 to 2.9.

4. Discussion

The data from these two fires demonstrates that the smoke impact from prescribed fires can be severe. Peak $PM_{2.5}$ levels > 1000 $\mu g/m^3$ and 24-hour average levels of 209 $\mu g/m^3$ were recorded at one location 100 m away from the 52 ha fire and other locations up to 500 m downwind from that fire also recorded very high levels. The $PM_{2.5}$ levels during the fire are similar to those found Pearce et al (2012) at similar distances to prescribed burns in savanna vegetation, including their peak value of 180 $\mu g/m^3$ ~ 9 km from one of the fires. Four locations near the 52 ha fire and three near the 700 ha fire recorded 24-hour average $PM_{2.5}$ above the national standard (25 $\mu g/m^3$ for average daily $PM_{2.5}$ exposure). The 700 ha fire generated generally lower peaks and sustained PM levels, but spread low level pollution over a huge area. This large-scale impact is the most novel finding from this study. Relatively low 24-hour average levels at the permanent network of monitors suggested that this pollution was not a serious health hazard, but individual 1-minute $PM_{2.5}$ readings from the handheld monitor above 150 $\mu g/m^3$ at some locations on the morning after the fire indicate that it might have been a localised hazard in certain areas.

The larger fire lofted more smoke to a higher elevation, and spread pollution over a much larger area than the smaller fire. However, it is difficult to compare the local impact of the fires (~500 m from the fire) because the distance and bearing of the monitors to the fire were different between the fires and the wind was blowing from a slightly different direction. The data show higher local $PM_{2.5}$ levels in the 52 ha fire, but this may simply because the monitors were closer or more directly downwind than in the 700 ha fire. The atmosphere was much more stable in the larger fire (as reflected in the C-Haines index values) and also the winds were lighter. This presumably explains why the larger fire plume was trapped under an inversion layer during the night and possibly why shear caused some of the smoke to spread northward against the surface wind direction during the afternoon. The fact that the 700 m fire lofted smoke to approximately twice the height of the 52 ha was presumably because it generated more convective energy as opposed to any atmospheric effect (we would have expected the more stable air to limit the lofting height). The higher plume in the 700 ha fire will counter the fact that a greater total amount of smoke was produced to ameliorate the local smoke concentrations but only to a limited extent. Pearce et al. (2012) found a positive relationship between fire size and PM concentrations (range of fire sizes 10 – 1100 ha).

The atmospheric dispersion model predicted the distribution and concentration of pollution accurately during the daylight hours of both fires, but failed in three important respects. First, it did not capture the fine-scale variation in PM concentrations that is important when predicting local impacts. This is partly because of the coarse resolution of the model (1 km pixels), but may also reflect poor representation of the ambient wind or even the interaction between the wind and the fire itself. Second, the model failed to predict the northward spread of lofted smoke in the 700 ha fire in afternoon, though it did predict smoke spread at higher elevation to be different to the surface spread. Third, it failed to predict high PM concentrations during the night as a result of a temperature inversion.

This study is a preliminary attempt to apply empirical techniques to quantify the air quality impact of prescribed fires and the accuracy of a smoke prediction system. Many more fires would need to be studied before a systematic picture of the impacts of prescribed fires can be developed. Research should focus on 1) understanding the spatial scale of the smoke hazard from prescribed fires; 2) understanding when and why they can have bad outcomes; and 3) further testing of the atmospheric dispersion models. This kind of research is difficult because it requires equipment and personnel to be on 'standby' for a planned fire. The fires used in this project were postponed 12 times (including one abandoned altogether). The reference monitors need to be

deployed two days before the fire to enable calibration and final notification that burns will proceed often do not allow this amount of time. One short-cut method that could be used would be a historical analysis of $PM_{2.5}$ data from the permanent air quality network, looking at days when known prescribed fires were lit nearby one of the monitors, assuming such events did occur.

We acknowledge that the uncertainty in the $PM_{2.5}$ concentrations provided by the Dustrak monitor. In this study, it was found that the monitor overestimated the reference values by almost a factor of 5. Dustraks are commonly used for field $PM_{2.5}$ measurements (Innis, 2013) and overestimation by a factor of 1.7 and 2.2 times the reference are reported (McNamara et al., 2011;Trent, 2006). Hence, the overestimation in this study still appears high.

5. Conclusion

This study found particulate pollution levels to exceed national air quality standards in the vicinity of two prescribed fires, suggesting that local residents would be strongly impacted. Low level impact occurred over a very large area (120,000 ha) from the larger (700 ha) fire, and in some low lying places this may have caused an exceedance. The large area affected was possibly caused by a combination of two effects: wind shear causing smoke to spread against the surface wind direction and a temperature inversion at night that caused smoke to collapse to the ground. The TAPM smoke dispersal model predicted the general behaviour of the plumes in the early hours, but not the counter-wind spread or the collapse of the plume.

It is likely that these findings are general: high local impact and a risk of regional low level impact, increasing with the size of the prescribed fire. However, many more fires would need to be studied to be confident of these conclusions.

Agencies that conduct prescribed fires should aim to improve their prediction of smoke behaviour to minimise the air quality impacts on local communities.

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