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Meteorological controls on atmospheric particulate pollution during hazard reduction burns

Giovanni Di Virgilio[1], Melissa Anne Hart[1,2], Ningbo Jiang[3]

[1]Climate Change Research Centre, University of New South Wales, Sydney, 2052, Australia
[2]Australian Research Council Centre of Excellence for Climate System Science, University of New South Wales, Sydney, 2052, Australia
[3]New South Wales Office of Environment and Heritage, Sydney, 2000, Australia

Correspondence to: Giovanni Di Virgilio (giovanni@unsw.edu.au)

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Abstract. Internationally, severe wildfires are an escalating problem likely to worsen given projected changes to climate. Hazard reduction burns (HRB) are used to suppress wildfire occurrences, but they generate considerable emissions of atmospheric fine particulate matter, which depending upon prevailing atmospheric conditions, can degrade air quality. Our objectives are to improve understanding of the relationships between meteorological conditions and air quality during HRBs in Sydney, Australia. We identify the primary meteorological covariates linked to high $PM_{2.5}$ pollution (particulates < 2.5 μm diameter) and quantify differences in their behaviours between HRB days when $PM_{2.5}$ remained low, versus HRB days when $PM_{2.5}$ was high. Generalised additive mixed models were applied to continuous meteorological and $PM_{2.5}$ observations for 2011-2016 at four sites across Sydney. The results show that planetary boundary layer height (PBLH) and total cloud cover were the most consistent predictors of elevated $PM_{2.5}$ during HRBs. During HRB days with low pollution, the PBLH between 00:00 and 07:00 h (local time) was 100-200 m higher than days with high pollution. The PBLH was similar during 10:00-17:00 h for both low and high pollution days, but higher after 18:00 h for HRB days with low pollution. Cloud cover, temperature and wind speed reflected the above pattern, e.g. mean temperatures and wind speeds were 2 °C cooler and 0.5 m s^{-1} lower during mornings and evenings of HRB days when air quality was poor. These cooler, more stable morning and evening conditions coincide with nocturnal westerly cold air drainage flows in Sydney, which is associated with reduced mixing height and vertical dispersion, leading to the build-up of $PM_{2.5}$. These findings indicate that air pollution impacts may be reduced by altering the timing of HRBs by conducting them later in the morning (by a matter of hours). Our findings support location-specific forecasts of the air quality impacts of HRBs in Sydney and similar regions elsewhere.

1 Introduction

Many regions experience regular wildfires with the potential to damage property, human health, and natural resources (Attiwill and Adams, 2013). Internationally, the frequency and duration of wildfires are predicted to increase by the end of the century (e.g. Westerling et al., 2006; Flannigan et al., 2013). Wildfire frequency and duration have increased in western North America since the 1980s (Westerling, 2016). Their frequencies have also increased in south-eastern Australia over the last decade (Dutta et al., 2016), with a predicted 5-25 % increase in fire risk by 2050 relative to 1974-2003 (Hennessy et al., 2005), a risk compounded by climate change (Luo et al., 2013). In an effort to mitigate the escalating wildfire risk, fire agencies in Australia, as is the case internationally, conduct planned hazard reduction burns (HRBs; also known as prescribed or controlled burns). HRBs reduce the vegetative fuel load in a controlled manner and aim to lower the severity or occurrence of wildfires (Fernandes and Botelho, 2003).

Both wildfires and HRBs generate significant amounts of atmospheric emissions such as particulate matter (PM), which can impact urban air quality (Keywood et al., 2013; Naeher et al., 2007; Weise et al., 2015), and consequently public health (Morgan et al., 2010; Johnston et al., 2011). Of particular concern are fine particulates with a diameter of 2.5 µm or less, '$PM_{2.5}$'. Increased $PM_{2.5}$ concentrations are related to health effects including lung cancer (Raaschou-Nielsen et al., 2013) and cardiopulmonary mortality (Cohen et al., 2005). These impacts can be more severe for vulnerable groups, like the young (Jalaludin et al., 2008), elderly (Jalaludin et al., 2006) and individuals with respiratory conditions (Haikerwal et al., 2016).

Sydney, located in the south-eastern Australian state of New South Wales (NSW), is the focus of this study because HRBs make a significant contribution to PM pollution in this city and the surrounding metropolitan region (Office of Environment and Heritage, 2016). Sydney is Australia's largest city with 4.9M inhabitants (ABS, 2016). Approximately 130,911 ha in NSW was treated by HRBs during 2014-15 (RFS, 2015) and this figure is projected to increase annually (NSW Government, 2016). Smoke events between 1996 and 2007 in Sydney attributed to wildfires or HRBs were associated with an increase in emergency department attendances for respiratory conditions (Johnston et al., 2014). Hence, a potential consequence of HRBs is that Sydney's population experiences poor air quality and its associated health impacts (Broome et al., 2016). Furthermore, the eastern Australian fire season is projected to start earlier by 2030 under future climate change (Office of Environment and Heritage, 2014). This could restrict the period within which HRBs can occur, potentially exposing populations to particulates over more concentrated time-frames.

Sydney is located in a subtropical, coastal basin bordered by the Pacific Ocean to the east and the Blue Mountains 50 km to the north-west (elevation 1189 m, Australian Height Datum). Its air quality is influenced by mesoscale circulations, such as terrain-related westerly drainage flows in the evening, and early morning, easterly sea breezes in the afternoon (Hyde et al., 1980). These processes interact with synoptic-scale high-pressure systems (Hart et al., 2006). A recent study by Jiang et al. (2016b) further examined how synoptic circulations influence mesoscale meteorology and subsequently air quality in Sydney. The results showed that smoke generated by wildfires and HRBs makes a significant contribution to elevated PM levels in Sydney, in particular, under a combined effect of typical synoptic and mesoscale conditions conducive to high air pollution. However, analysis of the local (i.e. city-scale) meteorological processes that influence air quality during HRBs is still sparse. Previous research focusing on a single site in Sydney found that $PM_{2.5}$ concentrations were higher during stable atmospheric conditions and on-shore (easterly) winds (Price et al., 2012). Elsewhere, $PM_{2.5}$ concentration was mainly influenced by the receptor-to-burn distance and wind hits during HRBs (Pearce et al., 2012). We therefore have three aims: 1. summarise the temporal variation in $PM_{2.5}$ concentrations in Sydney and how this relates to HRB occurrences; 2. characterise $PM_{2.5}$ pollution sensitivities to meteorological and HRB variables to identify the primary covariates connected to high pollution; 3. identify the differences in covariate behaviours between HRB days when $PM_{2.5}$ pollution is low, versus burn days when pollution is high. Achieving these aims will help efforts to forecast the air pollution impacts of HRBs in Sydney, and more broadly, in Australia or elsewhere in the world.

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2 Data

2.2 Meteorological, air quality and temporal variables

Continuous time series of hourly meteorology and $PM_{2.5}$ (µg m^{-3}) observations between January 2005 and August 2016 inclusive were obtained from four air quality monitoring stations (Chullora, Earlwood, Liverpool and Richmond) in the NSW Office of Environment and Heritage (OEH) network in Sydney (Fig. 1). Monitoring stations are located at varying elevations and in semi-rural, residential and commercial areas (Table 1). These four locations were chosen because they have the longest, uninterrupted record of $PM_{2.5}$ measurements in Sydney. Prior to 2012 $PM_{2.5}$ was measured using tapered element oscillating microbalance (TEOM) systems. Since 2012 beta attenuation monitors (BAM) have been used to measure $PM_{2.5}$. Although there appear to be effects from instrument change, such effects are generally small if compared to the daily-to-day or hourly fluctuations in $PM_{2.5}$ levels.

To compare how $PM_{2.5}$ concentrations varied over daily and monthly timescales, we also obtained hourly measurements of PM_{10} (µg m^{-3}), nitrogen dioxide (NO_2) (parts per hundred million - pphm) and oxides of nitrogen (NO_X) (pphm) from these stations. Meteorological variables included in our analyses were: surface wind speed (m s^{-1}), wind direction (°), surface air temperature (°C) and relative humidity (%). Hourly global solar radiation (W m^{-2}) data were available at the Chullora station only, but were subsequently omitted as a predictive variable (see: 3.3.1 Model selection).

Hourly total cloud cover (okta) and mean sea level pressure (MSLP; hPa) were obtained from the Australian Bureau of Meteorology (BoM) Sydney Airport weather station (WMO station number 94767). These are included as covariates in models for the four monitoring sites. Twenty-four hour rainfall totals (mm) were approximated for each OEH station from the BoM weather station that is nearest (Fig. 1).

Given its role in the turbulent transport of air pollutants (Seidel et al., 2010; Pal et al., 2014; Sun et al., 2015; Miao et al., 2015), we included planetary boundary layer height (PBLH) as an explanatory variable. PBLH has previously been derived from observational meteorological data by Du et al. (2013) and Lai (2015), using a method which they found was an effective estimate of the PBLH and its relationship with PM concentrations. Although direct PBLH measurements would be ideal, these are unavailable for the study domain at appropriate spatial and temporal resolutions. Hence, we derived PBLH estimates at the location of each monitoring station from a subset of the meteorological data following the method used by the above authors (Eq. (1) and Eq. (2)).

$$PBLH = \frac{121}{6}(6-s)(t-td) + \frac{0.169s(ws+0.257)}{12f\ \ln(\frac{h}{l})} \quad (1)$$

$$f = 2\Omega\ \sin\theta \quad (2)$$

where s is a stability class that estimates lateral and vertical dispersion; t is surface air temperature and td is surface dew point temperature (approximated for the location of each station using the method proposed by Lawrence (2005)); ws is wind speed; h is wind speed altitude in m for a given monitoring station; l is the station's estimated surface roughness index, f is the Coriolis parameter in s^{-1}; Ω is the earth's rotational speed (rad s^{-1}) and θ is the station latitude. The stability typing scheme was based on the Pasquill-Gifford (P-G) stability categories (Turner, 1964), via a turbulence-based method using the standard deviation of the azimuth angle of the wind vector and scalar wind speed.

We calculated the 24-hour mean for hourly meteorological and $PM_{2.5}$ measurements, where wind direction was vector-averaged (i.e. averaging the u and v wind components). Log-transformations were applied to $PM_{2.5}$ and rainfall. Applying transformations to the remaining explanatory variables did not greatly reduce heterogeneity.

Temporal variables trialled for inclusion in analyses included day of the year, weekday, week, month (all representing different seasonal terms) and year (because air quality varies from year to year). A Julian date variable was incorporated to represent the longer-term trend in $PM_{2.5}$ concentrations.

2.3 Burns

Historical records of HRBs conducted between January 2005 and August 2016 in NSW were obtained from the NSW Rural Fire Service (RFS), the firefighting agency responsible for the general administration of HRBs. There were a total of 9200 fire polygons in this data set prior to data conditioning (see: 3 Methods). HRBs are conducted predominantly in Autumn and Spring, and often at weekends, typically, with burns lit in the early morning. Most

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historical HRBs have occurred to the west and north-west of Sydney (Fig. 2). Additional predictive variables derived from the HRB data (all daily values) were: total number of burns, total burn surface area (ha), median burn elevation (m), median fire duration (days) and median fire distance from the geographic centre of the monitoring stations (km).

It is important to note that other potential sources of $PM_{2.5}$ emissions in Sydney include motor vehicles, soil erosion and occasional dust storms. Use of domestic wood-fired heaters can also make a substantial contribution to $PM_{2.5}$ concentrations during Winter months (when HRBs are generally not conducted). However, between 2011 and 2016, average $PM_{2.5}$ air quality index (AQI) values were higher on days when either HRBs or wildfires occurred relative to days when there were no fires (Fig. 3).

3 Methods

3.1 Statistical approach: generalised additive mixed models

Generalized additive models (GAMs) (Hastie and Tibshirani, 1990) offer an appropriate approach with respect to air quality research because relationships between covariates are often non-linear, an issue which can be addressed within the GAM framework. In addition to the seasonal pattern of hazard reduction burning, $PM_{2.5}$ concentrations in Sydney also show daily, monthly, seasonal and annual variation. Adding terms to a GAM to account for these temporal variations fails to deal with residual autocorrelation completely, as is evident in the autocorrelation function (ACF) of the residuals (Fig. S1, Supplementary Material). Given the residual autocorrelation and non-independence of the data, we used a generalised additive mixed modelling (GAMM) approach to take account of the seasonal variation and trends in the data. GAMMs can combine fixed and random effects and enable temporal autocorrelation to be modelled explicitly (Wood, 2006). We assumed a Gaussian distribution and used an identity link function. Cubic regression splines were used for all predictors except wind direction and day of year which used cyclic cubic regression splines, because there should be no discontinuity between values at their end points. Experimenting with alternative smooth classes did not drastically affect model results or diagnostics. Smoothing parameters were chosen via restricted maximum likelihood (REML). We implemented GAMMs with a temporal residual auto-correlation structure of order 1 (AR-1). More complex structures (e.g. auto-regressive moving average models; ARMA) of varying order or moving average parameters produced marginally higher Akaike information criteria (AICs) (e.g. mean = 259.6) than models with AR-1 auto-correlation (mean AIC = 259.02). Omitting a correlation structure entirely produced the largest AICs (mean AIC = 279.5). In all cases, the AR models for the residuals were nested within month (nesting within week and year was also trialled, but produced higher AICs). Auto-correlation plots obtained by applying the GAMMs using the AR-1 structure showed that short-term residual autocorrelation in the residuals had been removed relative to using GAMs (Fig. S1-2 in Supplementary Material).

3.2 $PM_{2.5}$ trend estimates, monthly and daily means

We first used the GAMM framework to estimate the annual trend in the weekly mean concentrations of $PM_{2.5}$ for 2005–2015, split by season, with Julian day as the only predictor. Monthly and daily mean $PM_{2.5}$, PM_{10}, NO_2 and NO_X concentrations for all years were also compared to assess how concentrations of each pollutant varied with these timescales. The latter analyses were performed using *R* software for statistical computing (R Development Core Team, 2015) and the *openair* package (Carslaw and Ropkins, 2012). The annual trend and subsequent statistical analyses described below were performed using *R* software and packages *mgcv* (Wood, 2011) and *nlme* (Pinheiro et al., 2017).

3.3 Identifying the meteorological and burn variables related to elevated $PM_{2.5}$

To assess how $PM_{2.5}$ concentrations vary in relation to the meteorological, burn and temporal variables, the GAMMs were applied to each monitoring site separately and focused on the period January 2011-August 2016. There were comparatively fewer HRBs conducted prior to 2011, hence the choice of this timeframe. For each station, we split the data into two subsets: 1) for all days when HRBs were conducted and the $PM_{2.5}$ concentration was less than the median $PM_{2.5}$ concentration for the location in question, 'low pollution days'; 2) for all HRB days when the $PM_{2.5}$ concentration was greater than the median value for the location in question, 'high pollution days' (the minimum/maximum number of observations in each low/high subset was in the range 179-189). The time series were conditioned in this manner to better characterise the differences in covariate behaviours between burn days when pollution remains low versus burn days and elevated $PM_{2.5}$. Since our focus is specifically on $PM_{2.5}$ concentrations during HRBs, days when wildfires had occurred were excluded.

3.3.1 Model selection

Using the GAMM framework described above, we started with a model where the fixed component included all predictive variables. We used variance inflation factors (VIF) to test variables for collinearity (Zuur et al., 2010). We sequentially dropped covariates with the highest VIF and recalculated the VIFs, repeating this process until all VIFs were smaller than a threshold of 3.5. Following this process, explanatory variables were dropped from the initial model if they were not statistically significant in any case. As a result, global solar radiation, relative humidity, burn elevation, burn duration, weekday, week and year were excluded.

An intermediate model included HRB distance as a covariate. This revealed that beyond a maximum distance of approximately 300 km from monitoring sites, the influence of HRBs on air quality appears negligible (Fig. S3, Supplementary Material). Subsequent models excluded burn distance and burns > 300 km from the geographic mean centre of the monitoring stations. Hence, the fixed component of our optimal model used the following predictors: PBLH, MSLP, temperature, total cloud cover, rainfall, wind speed, wind direction, number of burns per day, total area burnt per day, day of year and Julian day.

3.4 Diurnal variation in relation to elevated $PM_{2.5}$

Meteorological covariates relevant to high $PM_{2.5}$ concentrations were identified via the GAMMs based on criteria of statistical significance at more than one location, or where the influence of covariates on $PM_{2.5}$ showed a marked distinction between pollution conditions. We then used the hourly meteorological data for these select covariates to compare their mean diurnal variation on burn days with low versus high pollution. The 95 % confidence intervals of these diurnal means were calculated using bootstrap re-sampling.

4 Results

4.1 Temporal variation in $PM_{2.5}$ concentrations

There is an increasing inter-annual trend in weekly mean $PM_{2.5}$ concentrations in all seasons during 2011 to 2015, especially in summer and winter (Fig. 4). Mean $PM_{2.5}$ concentrations range from 6 - 10 µg m^{-3}. Mean monthly $PM_{2.5}$ averaged over all years shows increasing concentrations from early autumn (March), peaking in May, then decreasing towards the end of winter, before increasing again from early spring (Fig. 5a). Notably, mean daily $PM_{2.5}$ concentrations (averaged over all years) are higher at weekends relative to other pollutants (PM_{10}, NO_2 and NO_x; Fig. 5b).

4.2 Meteorological and burn variables related to $PM_{2.5}$

Adjusted R^2 values for high pollution models were between 0.40 and 0.56, and between 0.35 and 0.50 for the low pollution models (Table 2). PBLH and total cloud cover were the most consistent predictors of elevated $PM_{2.5}$ during HRBs (Table 2). On high pollution days, PBLH had a statistically significant, negative influence on predicted $PM_{2.5}$ concentrations at all locations (Fig. 6). This influence was generally more linear on high pollution days, relative to low pollution days. Notably, fitted curves for $PM_{2.5}$ – PBLH were steeper at lower altitudes (< 800 m) in the high pollution condition. Cloud cover had a negative influence on predicted $PM_{2.5}$ concentrations that was significant in all but one case (Table 2), though fitted curves do not appear to differ noticeably between pollution conditions (Fig. 7). Although temperature and wind speed showed a more variable pattern of statistical significance (Table 2), they exhibited marked differences in behaviour between low and high pollution days. During high pollution at Richmond and Chullora, temperature had a negative, curvilinear influence on fitted $PM_{2.5}$ values (Fig. 8). This negative influence reverses at temperatures > 20 °C. In contrast, the $PM_{2.5}$ – temperature relationship was weak and linear during low pollution days. Wind speed had a significant influence on $PM_{2.5}$ only at Earlwood (Table 2). During low pollution days, this association is negative, whereas on high pollution days there is a positive influence on $PM_{2.5}$ at low wind speeds which reverses at speeds above 2 m s^{-1} (Fig. 9). During HRBs and high pollution, wind direction curves show peaks at approximately 150 degrees at Liverpool (south-easterly flows), and also increase between ca. 230 and 300 degrees at Liverpool and Earlwood (south-westerly to north-westerly flows) (Fig. 10). Earlwood frequently experiences north-westerly flows during Spring, Autumn and Winter, whilst south-westerly flows are common during the same seasons at Liverpool (Fig. S4, Supplementary Material).

The remaining meteorological predictors either did not show marked differences between pollution conditions or were statistically significant in only one instance. Rainfall generally had a negative influence on $PM_{2.5}$ during HRBs (Fig. S5, Supplementary Material). MSLP had a positive association with higher $PM_{2.5}$ concentrations during low and high pollution (Fig. S6, Supplementary Material), though this association was only significant during high pollution at Richmond (Table 2).

HRB frequency had a significant and positive influence on $PM_{2.5}$ only for the high pollution condition (Table 2 and Fig. 11), whereas the influence of burn area was negligible in all cases. Curve gradients for day of year start increasing at day ninety (autumn) during high pollution days, however, this predictor was significant in only one instance (Fig. S7, Supplementary Material). The influence of Julian day on $PM_{2.5}$ showed significant non-linear, increasing trends in all instances.

4.3 Differences in covariate behaviours on HRB days with low versus high $PM_{2.5}$

Having identified the most informative and consistent meteorological predictors using the GAMMs, we compared their mean diurnal variation during the occurrence of HRBs and low versus high $PM_{2.5}$ pollution:

4.3.1 PBLH

Taking Liverpool as an example, between 00:00 and 07:00 h during low pollution days when HRBs have occurred, the PBLH is on average 100-200 m higher than during high pollution days (Fig. 12; see Fig. S8-10 in the Supplementary Material for the other monitoring stations). From late morning (ca. 10:00 h) until early evening (c. 19:00 h), the PBLH altitudes of both $PM_{2.5}$ conditions are very similar, but after 19:00 h the PBLH is again higher during low pollution.

4.3.2 Total cloud cover

During HRBs, mean diurnal variation of cloud cover is between 2 and 7 % greater during the mornings and evenings of low pollution, compared to high pollution days (Fig. 12). In contrast, there is minimal difference in cloud cover during the early afternoon of both conditions.

4.3.3 Temperature

The temperature is 1 to 6 °C warmer between 00:00-08:00 h and 20:00-23:00 h during HRBs and low $PM_{2.5}$, in comparison to burns coinciding with high pollution (Fig. 12). However, there is a clear reversal in this trend from mid-morning to late afternoon during burns and high $PM_{2.5}$ when mean temperature is several degrees warmer than during HRBs and low pollution.

4.3.4 Wind speed

Mean diurnal wind speed is approximately 0.5 m s^{-1} higher in the mornings and after 18:00 h during burns and low air pollution in comparison to speeds during high $PM_{2.5}$ (Fig. 12). In contrast, there is a minimal difference in wind speeds between 12:00 and 18:00 h.

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Air quality in Sydney is generally good. On the occasions when it is poor, atmospheric particulates are the principal cause, and HRBs are potentially one source of high particulate emissions. Sydney's population is projected to increase (~63 %) to over 8 million by 2061 (ABS, 2013), with much of the expansion occurring at the urban-bushland transition. Even if air quality remains stable, these demographic changes will increase exposure to particulate pollution. However, we observed increasing annual trends in $PM_{2.5}$ concentrations. In addition, projected decreases in future rainfall (Dai, 2013) and increases in fire danger weather are likely to increase fire activity and lengthen the fire season (Bradstock et al., 2014), thus amplifying fire-related particulate emissions. Changes in measurement instrumentation have a potential to introduce systematic biases in these annual $PM_{2.5}$ trends. However, the instrumentation changes that occurred in 2012 are likely to have a minimal impact on the trends identified in this analysis, as is consistent with the increasing $PM_{2.5}$ trends shown in two EPA analyses (NSW Government, 2016; 2017). Moreover, the trends start increasing from 2011 during spring and winter, which precedes the instrumentation change.

Relative to other pollutants such as NO_x and NO_2, $PM_{2.5}$ concentrations are higher at weekends. $PM_{2.5}$ concentrations also start increasing in autumn with peaks in winter and spring. These patterns may reflect the timing of HRB occurrences, which occur mainly in autumn, spring and at weekends, though there is also increased domestic wood-fired heating during winter. Consequently, conducting multiple, concurrent HRBs during these periods might exacerbate $PM_{2.5}$ concentrations that are already high relative to baseline.

$PM_{2.5}$ concentrations tend to be dominated by organic matter (57%) during peak HRB periods in autumn. There is also contribution, in order of apportion, from elemental carbon, inorganic aerosol, and sea salt. This compares to summer months when sea salt plays a larger role, with organic matter making up just 34% (Cope et al. 2014). Other days where national $PM_{2.5}$ concentration standards have been exceeded have been attributed to wildfires and dust storms. $PM_{2.5}$ concentrations also tend to be higher across the Sydney basin during winter due to smoke from wood fire heaters used for residential heating, however, exceedances of standards due to these emissions are rare (EPA, 2015).

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5.1 Primary covariates affecting $PM_{2.5}$ and how they differ during low and high pollution

PBLH was the most consistent meteorological predictor of $PM_{2.5}$. It had a significant, negative influence on $PM_{2.5}$ at all locations during HRBs and 'high pollution days'. There was a marked difference in mean diurnal mixed layer heights between low and high pollution conditions in the early morning (00:00-07:00 h) and from 20:00 to 23:00 h, with the PBLH being approximately 100-200 m lower at these times during HRBs and high $PM_{2.5}$. During these two time periods whilst the PBLH is low, mean cloud cover, temperature and wind speeds are also lower relative to their magnitudes at corresponding times during low pollution. Essentially, these early hours of cold, stable conditions with minimal turbulence (i.e. conditions that are conducive to temperature inversions) prevent the dilution of $PM_{2.5}$. These subdued conditions often coincide with the night time/early morning westerly cold drainage flows and low mixing heights (inhibiting vertical dispersion), leading to the build-up of $PM_{2.5}$ during mornings (Lu and Turco, 1995; Hart et al., 2006; Jiang et al., 2016b). These pollution-conducive conditions are similar to those identified in Jiang et al. (2016a) as being related to a ridge of high pressure extending across eastern Australia, resulting in light north-westerly winds. These synoptically driven flows, although light, tend to enhance nocturnal drainage flows, inhibit afternoon sea breeze formation, and allow the transportation of pollutants across the Sydney basin to the coast. There is also a large difference in mean diurnal temperatures between low and high pollution conditions from late morning to early evening, with temperatures 3-4 °C warmer during high pollution. During warmer daytime conditions, $PM_{2.5}$ can be potentially higher without fire events, for instance, because these conditions tend to be coincident with increased precursor emissions and generation of secondary organic aerosols in the air. Furthermore, the fact that early morning and late evening temperatures tend to be lower during high pollution conditions may indicate the presence of temperature inversions which hinder atmospheric convection, leading to the collection of particulates that cannot be lifted from the surface. Cold morning temperatures can also result in stronger drainage flows into the Sydney basin. Consequently, if HRBs are being conducted during early mornings in the hills and mountains to the west of Sydney, this could result in the dispersion of particles from such sources, possibly into populated areas.

These findings indicate how the timing of HRBs can be altered to reduce their air pollution impacts in Sydney. Conducting HRBs when the PBLH is forecast to be higher ought to help reduce their air quality impacts in Sydney. More specifically, conducting HRBs later in the morning (for example by a matter of hours) is one way of potentially reducing HRB air quality impacts, because the PBLH generally starts increasing rapidly in height from 07:00 until 12:00 h. Fires conducted early in the morning when the PBLH is at its lowest, and temperatures are cool will promote effects such as fire smoke residing near ground-level. One constraint concerning later burn times is that wind speed typically increases as the day progresses. However, the maximum mean diurnal wind speed was approximately 3 m s^{-1} and occurred at 15:00 h. This is considerably lower than the RFS' upper-limit of 5.56 m s^{-1} for conducting safe HRBs (Plucinski and Cruz, 2015). An additional caution for conducting burns later in the afternoon is that onshore coastal breezes can develop during afternoons. The optimal timing of burns will also be dependent on other factors such as burn intensity, lighting method, fuel/soil moisture and geographic location.

Although there were similarities in the influence of covariates between locations, these associations often varied spatially. For example, mean diurnal PBLH and temperature were lower at Richmond in the early morning and at night in comparison to the other locations (Fig. S10, Supplementary Material). Richmond is further inland than the other monitoring sites and is thus closer to the mountain range to the west of Sydney. The insights gained into the spatial variation in the behaviour of covariates can support efforts to create location-specific particulate pollution forecasts.

The north-westerly signal apparent for several locations during HRBs and high pollution may reflect the fact that, overall, the majority of burns are conducted to the west, north and north-west of Sydney (Fig. 2). From a management perspective, comparatively greater attention might be devoted to adapting burn operations in these regions. However, the daily vector-averaging applied to the wind data will smooth out the signal associated with diurnal changes in wind directions (and speeds), e.g. between drainage flow and sea breezes. Thus, to some degree, the signal of wind influence may be suppressed, which is potentially one explanation why wind does not emerge as a more important factor from the GAMM models.

Using a different analysis approach, Price et al. (2012) found that the optimum radius of influence of landscape fires on $PM_{2.5}$ was 100 km for Sydney. We found that whilst close-proximity fires influenced air quality, fires up to approximately 300 km from monitoring stations also potentially influenced $PM_{2.5}$. Longer-range exposures on regional scales, particularly from multiple HRBs in an air-shed can impact communities at considerable distance under certain atmospheric transport conditions (e.g. Liu et al., 2009).

Multiple concurrent burns are more likely to adversely affect air quality in Sydney, as indicated by the statistically significant, positive influence of the number of concurrent HRBs on $PM_{2.5}$ during high pollution days. In general, greater numbers of concurrent burns within a given air shed are likely to result in greater quantities of particulate emissions. The area of these burns would also determine the amount of particulate emissions generated. However, HRB total area per day was not an effective predictor. This finding ought to be interpreted cautiously because of uncertainties about how accurately the area actually burnt was recorded within the polygons representing HRBs. In particular, to date it can be difficult to obtain timely and accurate estimates of the actual area burnt daily.

6. Conclusions

Fine particulate concentrations are increasing in Sydney, and given projected increases in fire danger weather, intensification in fire activity is expected to further amplify fire-related $PM_{2.5}$ emissions. We identified the key meteorological factors linked to elevated $PM_{2.5}$ during HRBs. In particular, diurnal variation of the PBLH, cloud cover, temperature and wind speed have a pervasive influence on $PM_{2.5}$ concentrations, with these factors being more variable and higher in magnitude during the mornings and evenings of HRB days when $PM_{2.5}$ remains low. These findings indicate how the timing of HRBs can be altered to minimise pollution impacts. They can also support locality-specific forecasts of the air quality impacts of burns in Sydney and potentially other locations globally. In addition to mitigating wildfire risk, globally HRBs are used for forest management, farming, prairie restoration and greenhouse gas abatement. Future research should incorporate more sophisticated fire characteristics such as plume height and fuel moisture into analyses, and also consider the influence of climatic phenomena on particulate pollution. Synoptic features can also be incorporated into a future GAMM analysis, as well as modelling the diurnal evolution of $PM_{2.5}$ pollution due to HRB occurrences.

Author contribution

G. Di Virgilio, M. A. Hart and N. Jiang conceived the research questions and aims. G. Di Virgilio designed and performed the analyses with contributions from all co-authors. G. Di Virgilio prepared the manuscript with contributions from all co-authors.

Competing interests

The authors declare that they have no conflict of interest.

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Tables

Table 1. The area type, elevation, location, inter-annual (2005-2016) mean and standard deviation (SD) $PM_{2.5}$ concentration (μg m^{-3}) of each monitoring site.

Site	Area Type	Elevation (m)	Lat	Lon	$PM_{2.5}$ Mean	$PM_{2.5}$ SD
Chullora	Mixed residential/commercial	10	-33.89	151.05	7.56	4.13
Earlwood	Residential	7	-33.92	151.13	7.26	4.34
Liverpool	Mixed residential/commercial	22	-33.93	150.91	8.27	4.85
Richmond	Residential/semi-rural	21	-33.62	150.75	6.85	6.29

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Table 2. Adjusted R^2, F and p-values for the smoothers of the optimal generalised additive mixed models (GAMM) applied to each monitoring site on days when hazard reduction burns occurred and with the data split into low and high air pollution conditions. Asterisks denote statistical significance: *** = $p < 0.001$; ** = $p < 0.01$; * = $p < 0.05$.

Pollution Condition	Chullora		Earlwood		Liverpool		Richmond	
	Low	High	Low	High	Low	High	Low	High
	R^2	R^2	R^2	R^2	R^2	R^2	R^2	R^2
	0.44	0.43	0.43	0.56	0.50	0.50	0.35	0.40
Variable	F	F	F	F	F	F	F	F
PBLH	6.0 ***	9.5 **	2.1	5.8 **	2.9 *	14.1 ***	0.9	7.7 ***
MSLP	1.1	0.0	0.3	1.7	0.3	0.8	0.5	4.8 *
Temperature	1.1	1.5	0.1	3.1	1.5	8.8 ***	0.0	4.1 *
Cloud cover	10.5 **	5.9 ***	4.4 *	5.2 **	9.3 ***	17.7 ***	1.4	5.9 *
Rainfall	0.2	0.7	5.1 *	3.5 *	5.7 **	0.7	2.4	4.0 *
Wind direction	2.2 *	1.2	3.3 **	6.4 ***	0.6	4.9 ***	3.0 **	0.6
Wind speed	0.8	2.0	3.3 *	3.7 **	2.1	0.0	0.7	1.5
HRBs daily frequency	0.4	6.1 **	0.4	4.1 *	0.7	3.7 *	0.2	14.0 ***
HRBs area burnt daily	8.2 ***	1.3	4.0 **	0.1	1.8	2.9	3.1	0.2
Day of Year	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.9 *
Julian Day	11.7 ***	4.9 **	9.5 ***	6.2 ***	16.7 ***	6.1 *	15.7 ***	0.7

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Figures

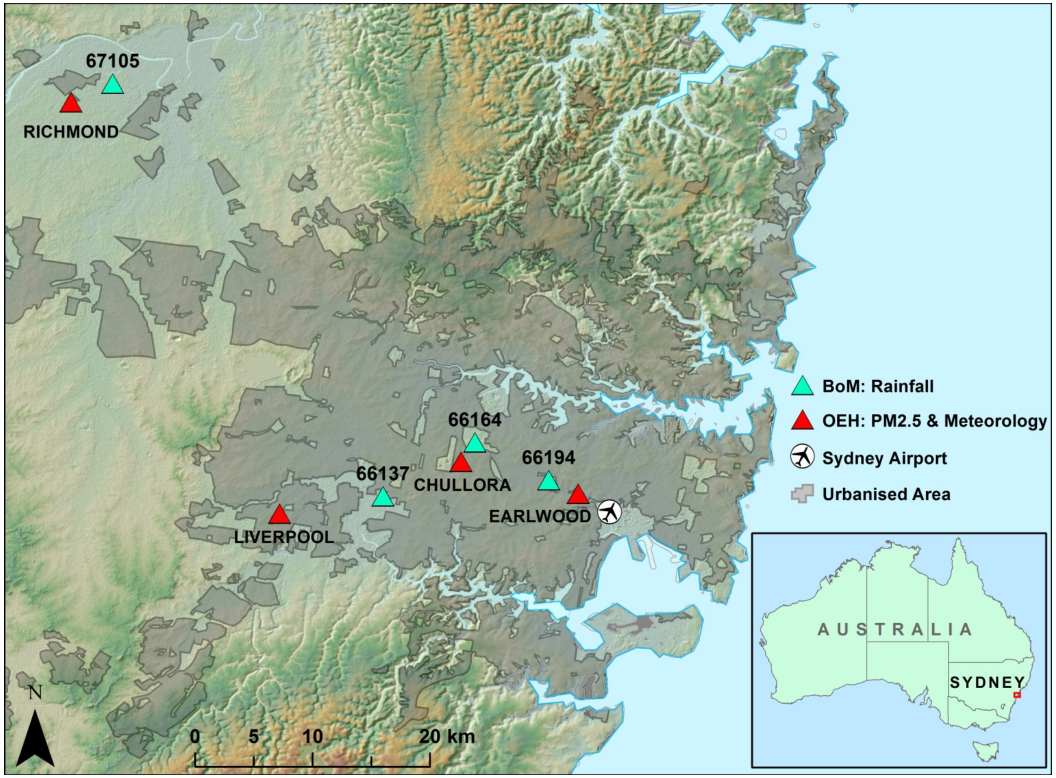

Figure 1. Locations of meteorological/$PM_{2.5}$ monitoring stations in the New South Wales Office of Environment and Heritage network in Sydney, Sydney Airport meteorological station, and Bureau of Meteorology (BoM) stations (with station numbers) from which rainfall data were obtained.

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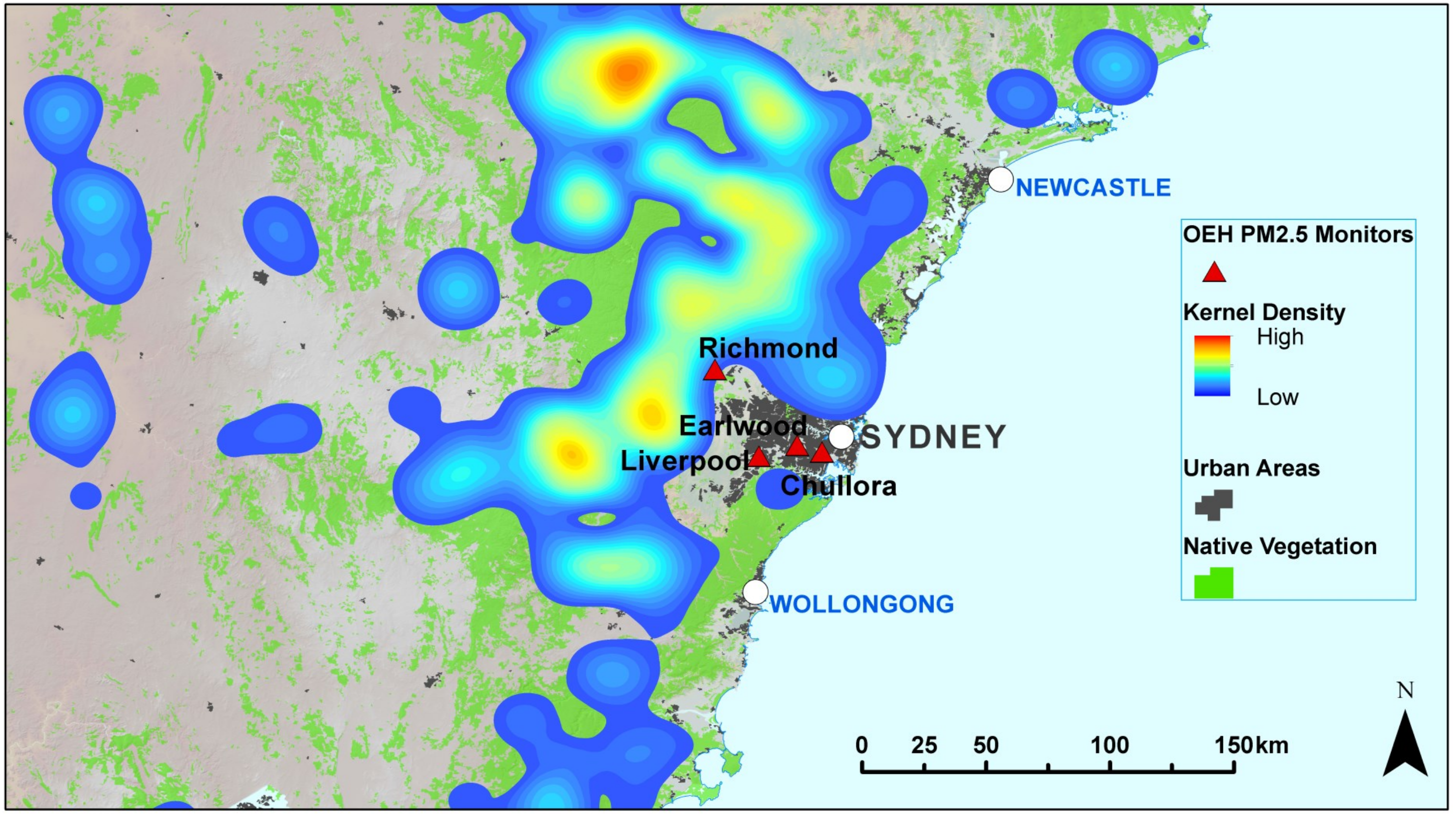

Figure 2. Kernel density function (magnitude-per-unit area) for hazard reduction burns (HRB) conducted in the vicinity of Greater Sydney (2004-2016). The warmer the colour of the kernel density surface, the more/larger HRBs that have occurred in that area. The kernel density calculation is weighted according to fire surface area.

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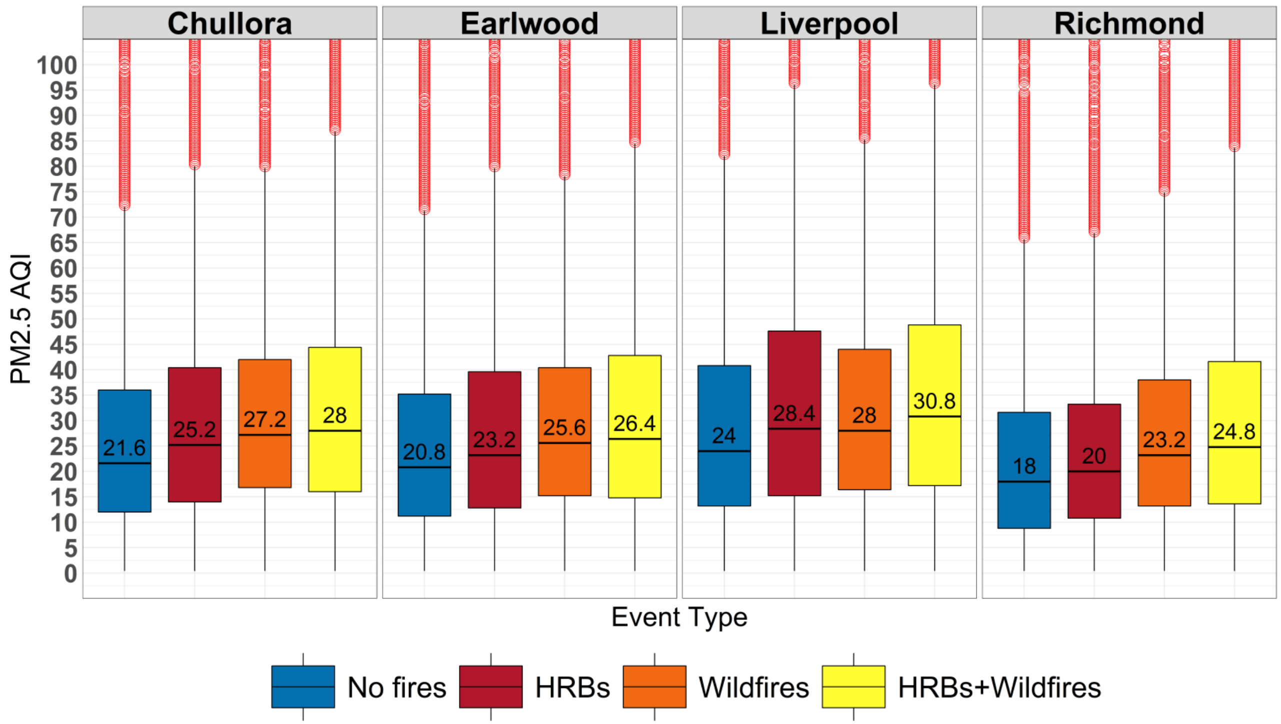

Figure 3. Boxplots showing the variation in $PM_{2.5}$ air quality index values (AQI) at four measurement sites in Sydney between 2011 and 2016 during days when there were no fires (neither hazard reduction burns (HRBs) or wildfires), days when only HRBs occurred without coincident wildfires, days when wildfires occurred without coincident HRBs, and days with concurrent HRBs and wildfires. Horizontal black lines on boxplots are median $PM_{2.5}$ AQIs and their corresponding values are shown above these lines. Red circles are outliers.

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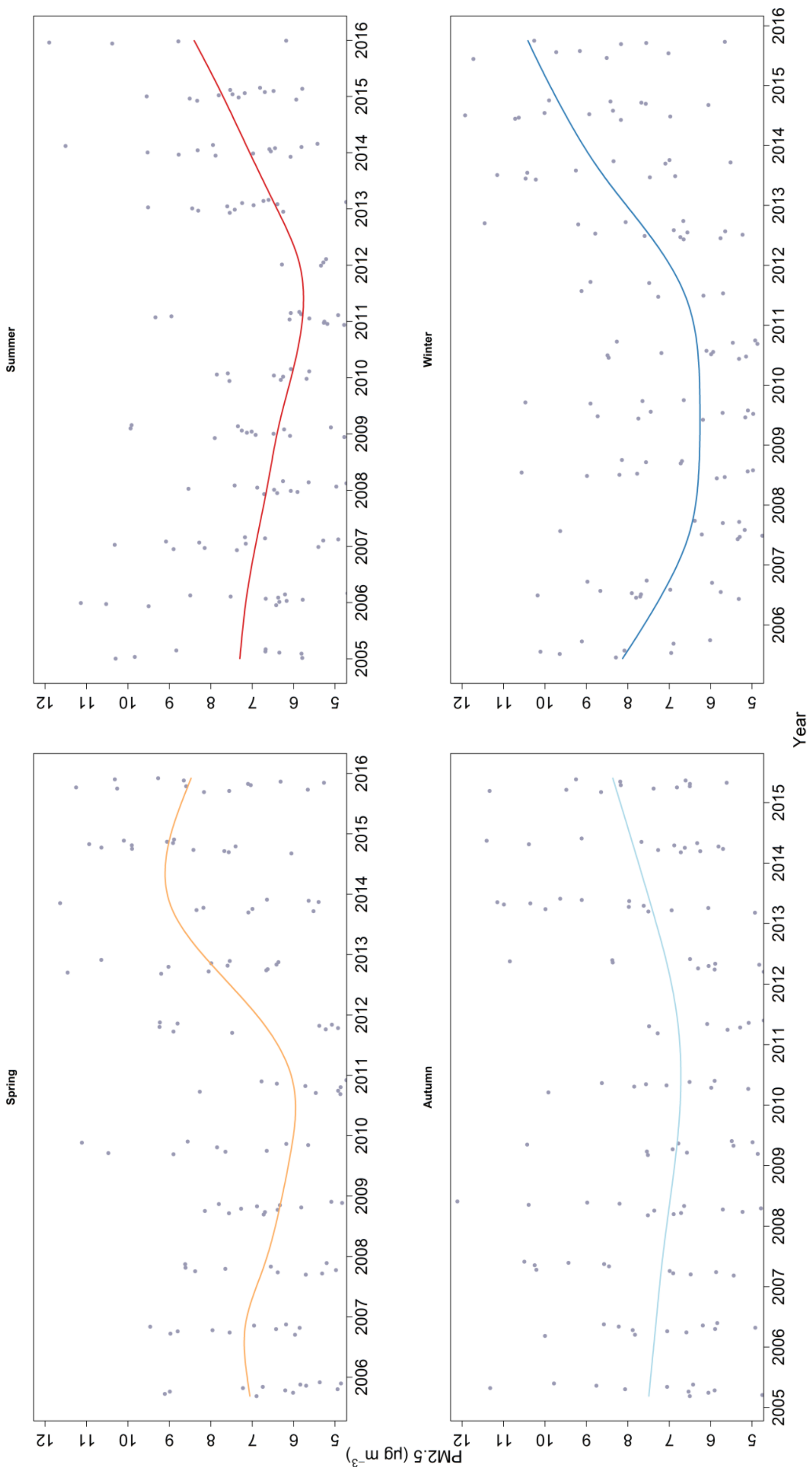

Figure 4. Annual trends in the weekly mean concentrations of $PM_{2.5}$ in Sydney, split by season for 2005 - 2015.

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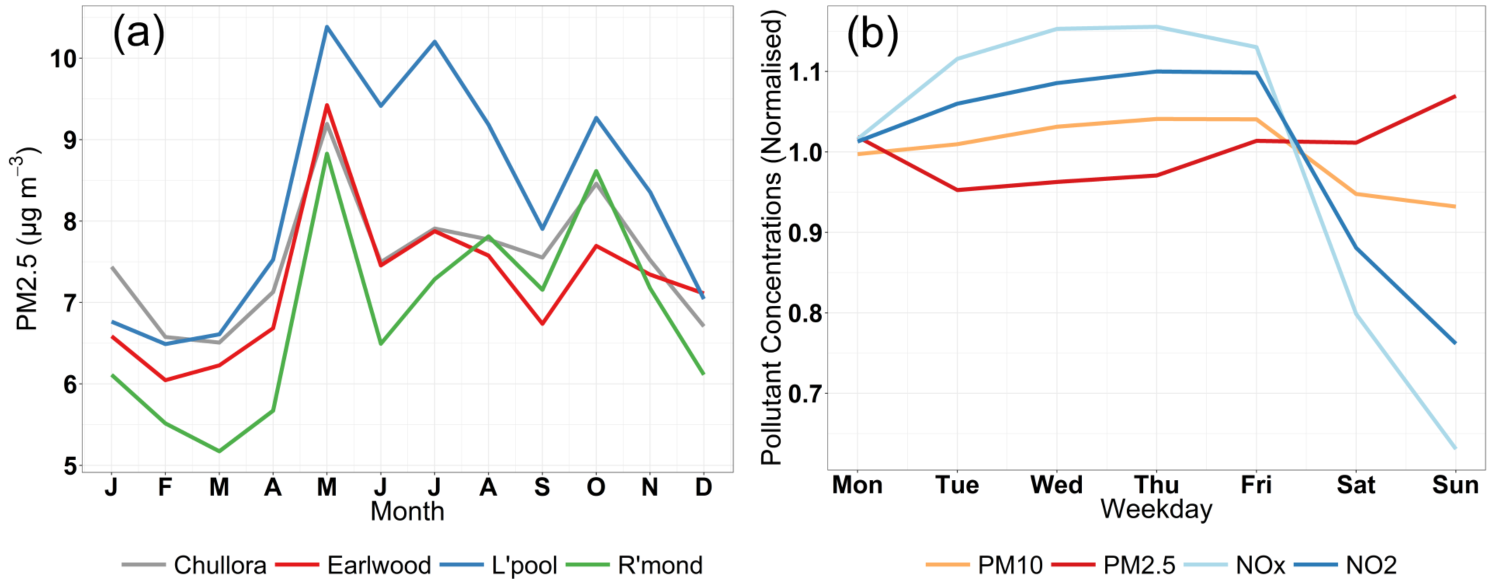

Figure 5. Mean monthly $PM_{2.5}$ concentrations for the period 2011 to August 2016 at four air quality monitoring sites in Greater Sydney (a). Mean daily normalised concentrations of $PM_{2.5}$ compared to the variations of PM_{10}, NO_2 and NO_x (b).

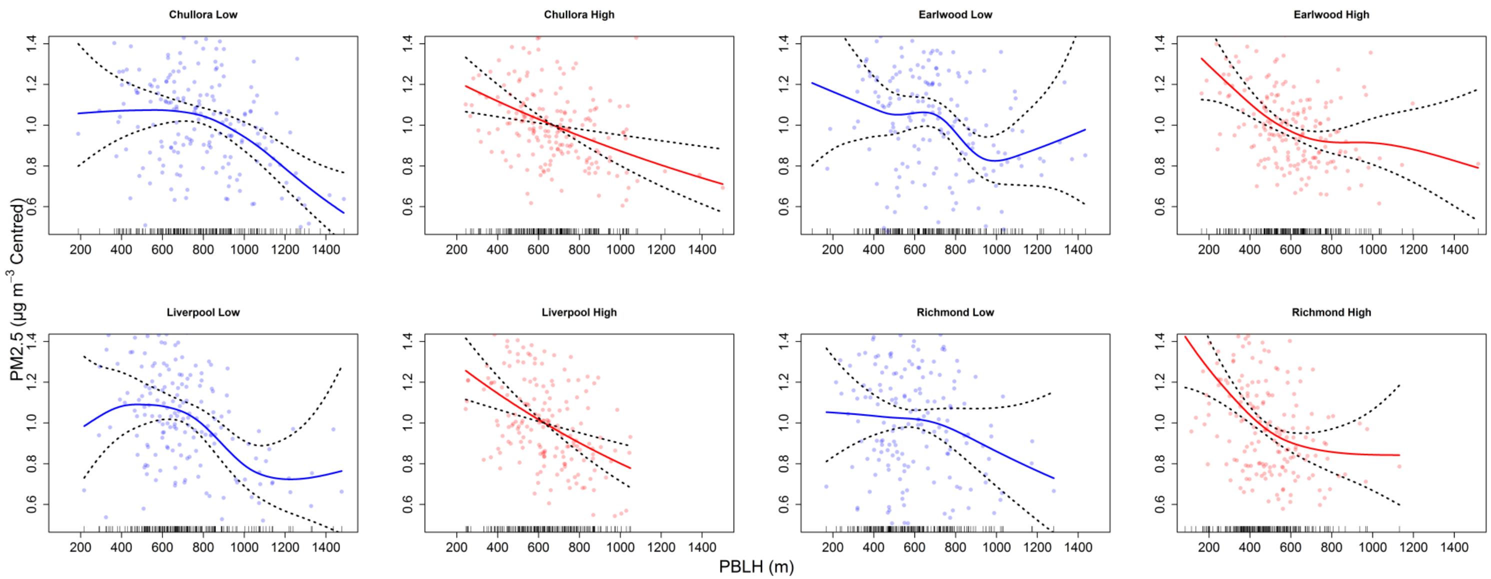

Figure 6. The contribution by the planetary boundary layer height (PBLH) component of the generalised additive mixed model (GAMM) linear predictor to fitted $PM_{2.5}$ values (µg m^{-3}, centred). The solid lines are the fitted curves. Dotted lines are 95% confidence bands. Dots are partial residuals.

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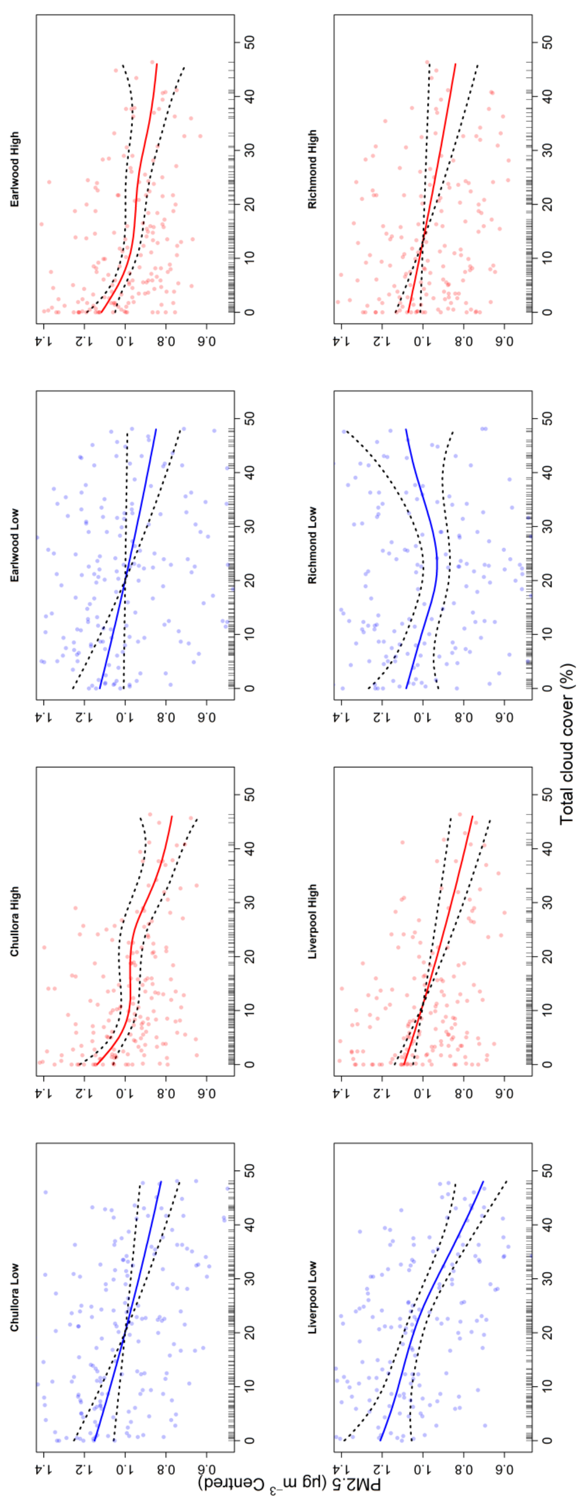

Figure 7. The contribution by the cloud cover component of the GAMM linear predictor to fitted $PM_{2.5}$ values (μg m^{-3}, centred).

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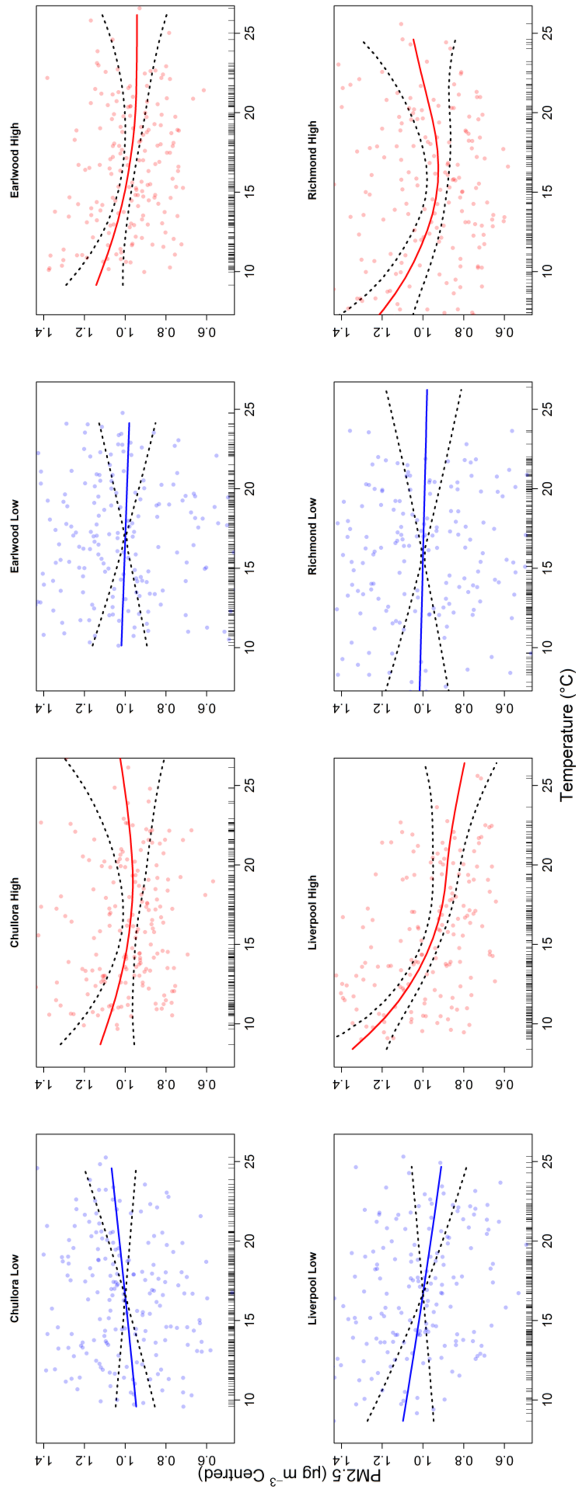

Figure 8. The contribution by the temperature component of the GAMM linear predictor to fitted $PM_{2.5}$ values (µg m^{-3}, centred).

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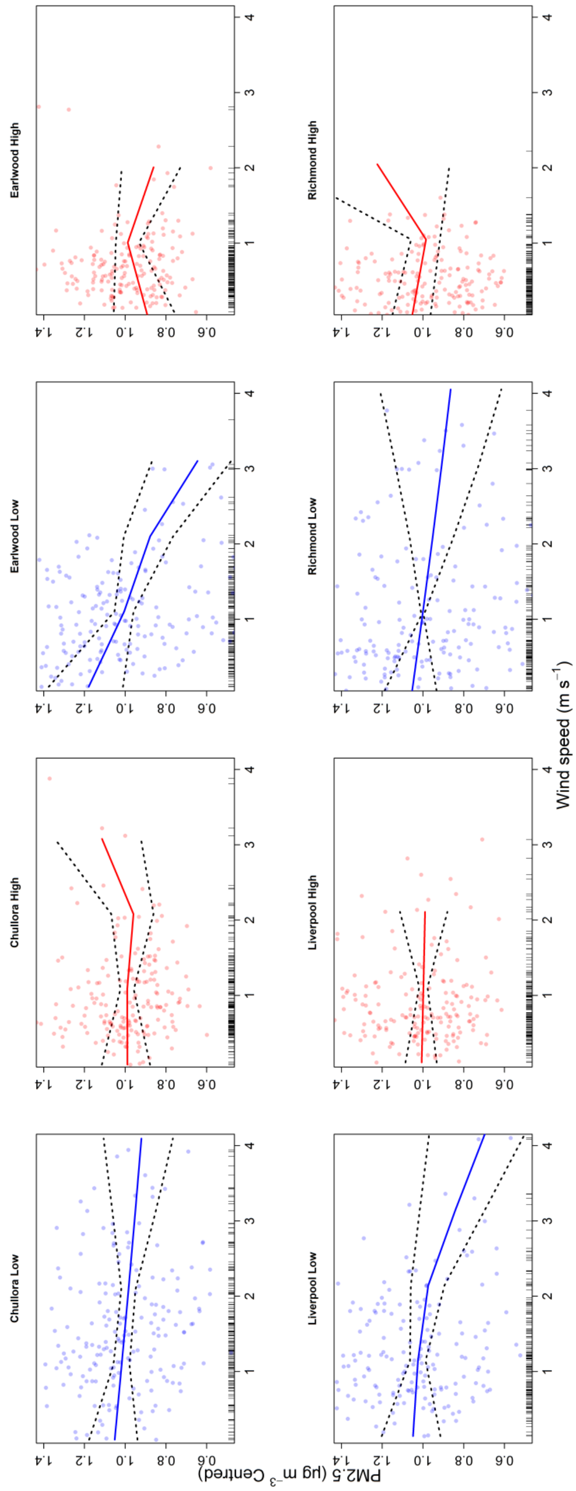

Figure 9. The contribution by the wind speed component of the GAMM linear predictor to fitted $PM_{2.5}$ values ($\mu g\ m^{-3}$, centred).

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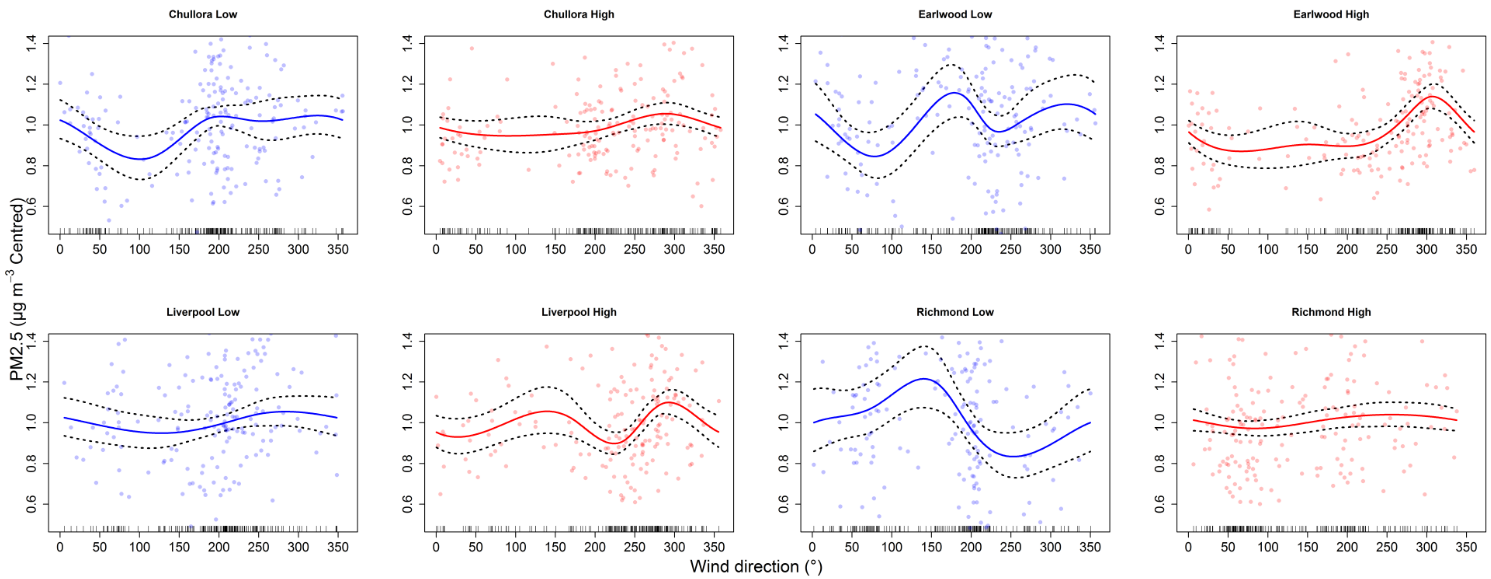

Figure 10. The contribution by the wind direction component of the GAMM linear predictor to fitted $PM_{2.5}$ values (µg m^{-3}, centred).

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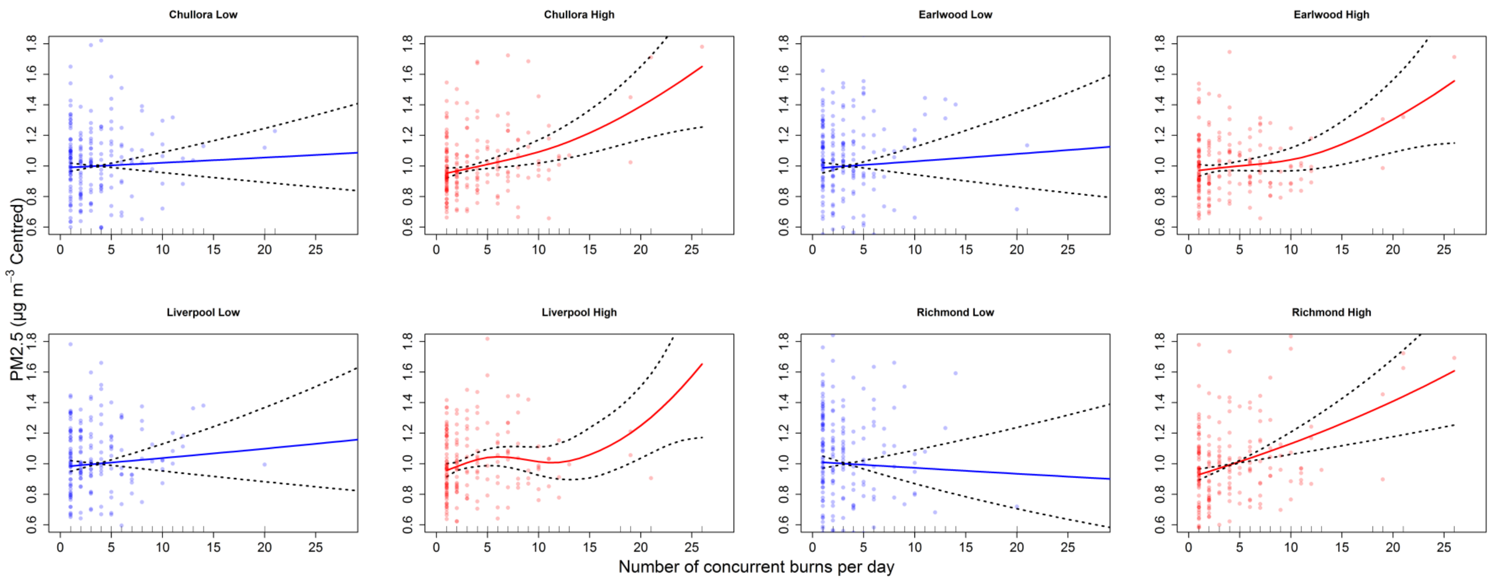

Figure 11. The contribution by the hazard reduction burn (HRB) daily frequency (number of concurrent burns per day) component of the GAMM linear predictor to fitted $PM_{2.5}$ values (µg m^{-3}, centred).

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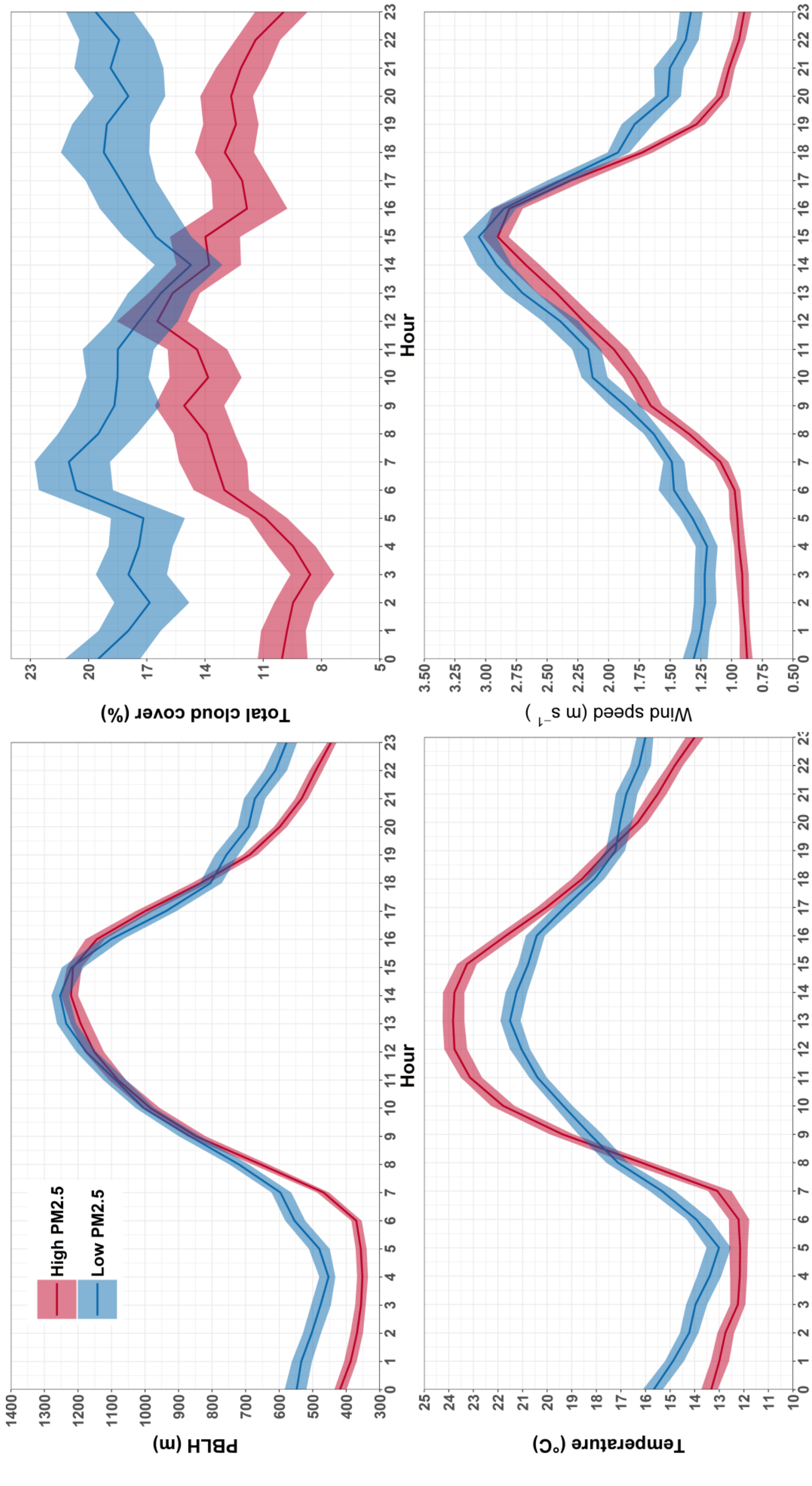

Figure 12. Mean diurnal variation of hourly PBLH, total cloud cover, temperature and wind speed for low versus high $PM_{2.5}$ pollution during HRBs at Liverpool, Sydney (see Figures S8-S10 for other stations). Shading represents the 95 % confidence intervals of the means.