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Effects of Climate Change on Bushfire Threats to Biodiversity, Ecosystem Processes and People in the Sydney Region

Sydney Bushfires, February 1, 2002

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Table of Contents

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EXECUTIVE SUMMARY

Fire management in the Sydney region is challenging due to the regularity of major fires and the juxtaposition of key values (e.g. property, biodiversity, water supply) in the region. Risks posed by adverse fire regimes to these values are largely unquantified. The prospect of changes to fire regimes via climate change and urban expansion heightens the challenge. In this project the FIRESCAPE landscape fire regime simulator was used to explore:

1) Potential effects of climate change on fire regimes
2) Effects of alternative management scenarios (via use of prescribed fire) on fire regimes
3) Effects of climate change on the outcome of these management alternatives
4) The quantitative nature of bushfire risk to people, property, biodiversity and catchment values.

FIRESCAPE captures the key processes that determine the incidence and spread of fires (weather, terrain and fuels). This simulator therefore provides a means for examining the influences of changes to fire weather (climate change) and fuel (management via prescribed burning) on fire regimes, and quantification of risk to key values.

Methods

The simulation model was parameterised and calibrated for three case study areas (Blue Mountains – BM; Woronora Plateau – WP, and Hornsby/Ku-ring-gai - HK) within the Sydney region, using relevant terrain, vegetation, fuel and a time series of daily weather for 1977 – 2003. Effects of climate change were simulated using forecasted 2050 climate and fire danger scenarios (‘Low’ and ‘High’) by Hennessy et al. 2005. These were based on adjustments to weather data for 1977 – 2003 for Sydney. Effects of climate change were assessed relative to simulated results under contemporary climate (i.e. 1977 – 2003).

Effects of variations in spatial strategy of prescribed burning (i.e. patterns of treatment across landscapes) and effort (i.e. annual area treated) were simulated. Four different spatial strategies were explored: Edge (treatment of urban edge blocks only); Unconstrained (random selection of blocks irrespective of proximity to the urban interface); Constrained (selection of ‘edge’ and non-edge blocks on an evenly weighted basis), and; Linear, arrays of blocks arranged in intersecting north-south and east-west linear segments. Effort was varied by selecting different proportions of blocks for annual treatment within each strategy (i.e. 0, 4, 8, 12, 16, 20 % of available blocks). This resulted in treatment of differing numbers of blocks among strategies. Blocks were selected annually on the basis of an estimation of potential fire intensity (i.e. effects of terrain and fuel status). Simulations were repeated for 50 cycles of the basic 26 year time series (contemporary or 2050 climate).

Indicators of consequential risk to urban, biodiversity and catchment values were examined. Mean area and intensity of edge blocks burnt by unplanned fires indicated the component of urban risk (about 50%) contributed by landscape condition. Area weighted distribution of adverse inter-fire intervals (IFI) is an indicator of extinction probability for key plant functional types. Annual probability of high intensity fires (> 10,000 kW/m) and very large fires (> 20,000 ha) may indicate adverse effects on arboreal mammals and increased propensity for soil movement.

Results

The mean area of simulated unplanned fires was linearly reduced by increasing prescribed burning effort (approximate 1 ha reduction for every 3 ha. treated). Spatial strategy of prescribed burning did not strongly affect mean area of unplanned fires. Similar effects of prescribed burning were found on respective probabilities of large (> 1,000 ha) and uncontrollable (i.e. > 4000 kW m^{-1} intensity) fires.

Climate change caused area burned by unplanned fires to increase by 7 – 13 % (Low scenario) and 26 -35 % (High scenario). This shift was more than twice that of the predicted increase in FFDI. Similar reductions in average interval between unplanned fires resulted (7 – 11% Low and 11 – 24% High). Climate change intensified some local patterns of high unplanned fire recurrence within BM and WP. Such trends may have significant consequences for urban protection in these areas.

Climate change (High scenario) increased mean unplanned fire area by > 20% at low levels of prescribed burning effort and by about 15% at high levels of effort (Unconstrained strategy only). Similar effects of climate change were found on annual probability of uncontrollable fire (> 4,000 kW/m). The effect on annual probability of large fires (> 1,000 ha) was lower (an increase > 5 % High scenario). A five fold increase in overall prescribed burning effort would be needed to counteract these effects in most of the case studies. Much smaller increases in effort were predicted to be needed to counteract effects of the Low climate change scenario.

The area of urban interface burnt by unplanned fires declined linearly with increasing levels of prescribed burning. This decline was greater under the Edge treatment compared with the other strategies for BM and WP. Contemporary levels of prescribed fire (circa. 1 % of landscape treated) yielded a small (circa. 5%) reduction in relative risk (i.e. compared with zero area treated). Intensity of fires at the urban edge was less strongly affected by prescribed fire strategy and effort. In both BM and WP, the Edge treatment decreased the intensity at the urban edge at a greater rate as a function of effort than the other strategies. Climate change increased the area of urban edge affected by unplanned fires under the Unconstrained strategy. Again, a five fold increase in prescribed burning would be required to counteract these effects. Effects of climate change on urban edge intensity were small, with a slight increase evident in all case studies and at most levels of prescribed burning effort.

The adverse IFI distribution under contemporary conditions (weather and prescribed burning) corresponded with low plant species extinction risk within the predominant Dry Sclerophyll vegetation. Climate change did not substantially alter this. High levels of Unconstrained prescribed burning (circa. 15 - 20 % of the landscape treated per annum) increased the IFI distribution to levels corresponding with high extinction risk. Climate change did not strongly alter this trend either. The IFI distribution within Wet Sclerophyll forest under contemporary conditions corresponded with high plant species extinction risk. Increasing levels of prescribed fire tended to mitigate this effect. Climate change did not strongly affect these trends.

Increasing levels of prescribed fire reduced crown fire probability (i.e. > 10,000 kW m^{-1} intensity). Climate change increased crown fire probability by about 20-25% at low levels of treatment in BM and WP but effects were lower in HK. At least a five fold increase in prescribed burning would be required to counteract the effect of the high climate change scenario. The percentage of rainforest affected by unplanned fire declined with increasing levels of prescribed fire. Climate change markedly increased the area of rainforest burnt irrespective of prescribed burning effort.

Discussion and Conclusions

Simulated fire regimes were sensitive to prescribed burning strategy, effort and climate change. Characteristics of unplanned fires were predominantly sensitive to level of prescribed fire effort, rather than spatial strategy. The chief exception was the effect of the Edge strategy on area and intensity of edge blocks burnt. Climate change, particularly the High scenario, generally affected fire regimes more than variations in spatial strategy of prescribed burning. Fire regimes were generally less sensitive to variations between case studies. Choices concerning prescribed burning effort in particular will be critical, governed by the following responses to increasing prescribed burning effort.

1. Zero prescribed fire maximised indicators of risk to urban and catchment values and species and communities deemed to be sensitive to fire intensity (SFI). It did not, however, result in high levels of risk to IFI sensitive plant biodiversity.
2. Current levels of prescribed burning (i.e. circa. 1 % of landscape treated per annum, approximating the Constrained strategy) resulted in a 5 % reduction to risk indicators for urban, catchment and SFI biodiversity values (relative to the level under zero prescribed fire). Current levels of burning did not result in high levels of risk to IFI sensitive biodiversity when assessed at a whole of case study scales. Local effects may warrant scrutiny.
3. Treatment of 3 % of the landscape per annum resulted in a 10 – 15 % reduction to risk indicators for urban, catchment and SFI biodiversity values, compared with zero prescribed fire.
4. Treatment of about 5-10 % of the landscape per annum resulted in a 25 – 45 % reduction to risk indicators for urban, catchment and SFI biodiversity values, compared with zero prescribed fire, without raising levels of risk to IFI sensitive biodiversity to a critical level.
5. Unconstrained treatment of 10 – 20 % of the landscape per annum resulted in a 50 – 70 % reduction to risk indicators to urban, catchment and SFI biodiversity values, compared with zero prescribed fire, but concurrently increased risk to IFI sensitive biodiversity to a critical level (i.e. high extinction probability).

6. Risk to key values cannot be eliminated by any level of plausible or hypothetical treatment.
7. Operational and economic constraints will have a major bearing on choices. The cost of prescribed burning will have a major effect on future choices and will benefit from economic impact analyses.

Climate change effects on risk are predicted to be appreciable, as follows.

1. Indicators of risk to urban, catchment, SFI and IFI sensitive biodiversity values could increase by 5 - 20 % under the High scenario and < 10 % under the Low scenario.
2. The change in risk to IFI sensitive biodiversity resulting from changes to the fire regime is unlikely to have a major effect on extinction probability at a whole of case study scale. More localised, potential adverse effects on IFI sensitive biodiversity were simulated in BM and WP. These may be significant and require further research.
3. Effects on extinction probability of SFI biodiversity are unknown and require further research.
4. The level of change in risk due to climate change is slightly reduced at very high levels of Unconstrained prescribed burning (> 10 % of landscape treated per annum).

The level of increase in prescribed burning required to counteract the adverse effects of the High climate change scenario is about five fold. A smaller increase in prescribed burning (up to twice current levels) may be needed to mitigate effects of the Low scenario. Cost will dictate the degree to which future increases in risk are ameliorated by prescribed burning or other management approaches. Other factors that affect the built environment will govern overall levels of risk to people and property (i.e. the results summarise the contribution to urban risk posed by the condition of the landscape only).

Only one potential vector of climate-driven change to fire activity (i.e. 2050 fire weather) was dealt with in this study. Changes to ignition rates (lightning) and fuel that may result from climate change were not addressed in detail. Preliminary sensitivity analyses (BM case study only) indicated that a 10 % decline in fuel accumulation under current climate may reduce mean annual area burned by about 20%. Such an effect would cancel out the increase predicted under the High climate change scenario, leaving overall area burned relatively unchanged. This scenario could represent the combined outcome of more severe fire weather and increased dryness. Effects of elevated CO_2 on plant growth could counteract effects of future dryness on fuels, but such effects in local ecosystems are uncertain. Thus there is potential for fire activity to either increase or decrease in the Sydney region as a consequence of climate change. Given that human ignition sources may increase as a function of human population pressure, a net increase in area burned is possible, with concomitant shifts in risk. The complexity of fire management will also increase in the future due to the need to consider additional values such as emissions, carbon sequestration and, human health (i.e. smoke effects).

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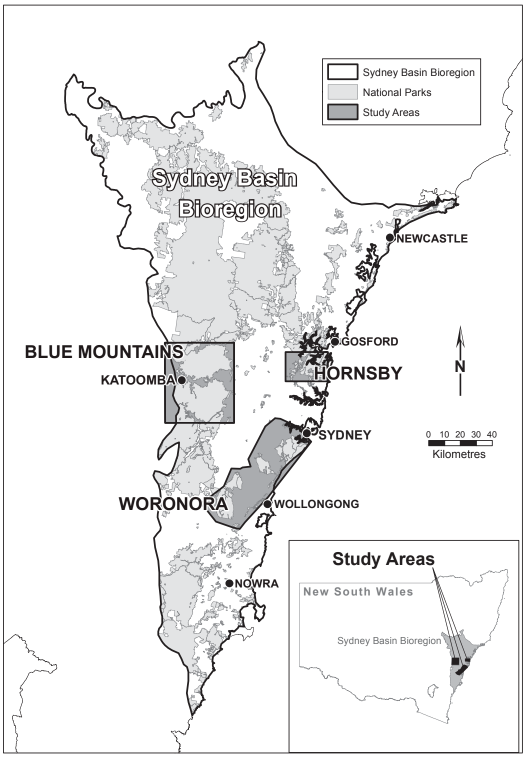
REPORT INTRODUCTION

The Sydney Basin IBRA Bioregion (Thackway and Creswell 1995) covers an area of about 3.6 million hectares ranging from the Hunter Valley in the north, the western margins of the Blue Mountains/Great Divide in the west to the Clyde and Shoalhaven River catchments in the south (Fig. 1). The terrain is diverse and rugged across the bulk of the region, reflecting the predominant sedimentary geology and elevation of >1000m on the highest parts of the Great Divide. The region also contains one of the largest tracts of relatively intact native vegetation in eastern NSW along with the largest city in Australia (i.e. Sydney) including satellite development on the Central Coast, the Illawarra and Blue Mountains. These landscapes are fire prone, in common with similar temperate regions of the world, due to the inherent attributes of weather, terrain, vegetation and ignition sources (natural and human).

The biodiversity of the region is significant due to the range of communities and species, endemics and rare taxa. Beadle (1981) noted that the region contains the second-most significant area of sclerophyll plant diversity on the continent. This diversity is situated in a complex array of eucalypt (e.g. *Eucalyptus, Corymbia, Angophora, Syncarpia*) open-forests, woodlands, shrublands and heathlands that swathe the sandstone dominated landscapes. Embedded within this sclerophyll matrix are patches of warm temperate and sub-tropical rainforest, often associated with gullies (topographic fire refugia) and/or volcanic intrusions (Adam 1992). The rare gymnosperm, *Wollemia nobilis* is found in such patches.

Figure 1
The Sydney Bioregion, with the location of case study landscapes indicated by shaded areas.



The significance of biodiversity in the region (among other geomorphic, scenic and land use values) has been recognised in the declaration of the Greater Blue Mountains World Heritage Area (GBMWHA). The GBMWHA (1.03 million ha) covers a large proportion of the native vegetation in the region, including about 2000 vascular plant species and about 100 eucalypt species. The bulk of Sydney's water supply catchments are situated within these landscapes, to the west and south of Sydney. The largest catchment, Warragamba, lies within the GBMWHA in the southern Blue Mountains, encompassing the Cox, Kowmung, Wollondilly and Warragamba rivers.

The natural vegetation of the region forms an arc around the bulk of the metropolitan area of Sydney and the outlying ribbons of coastal development. Significant areas of urban development abut or are intermingled with bushland. Fragments of bushland of varying size are also embedded within the urban matrix. Exposure of urban development to major bushfires is exacerbated by the prevailing north and westerly winds that are typically associated with periods of severe fire danger (Gill and Moore 1996, Bradstock et al. 1998, Hennessy et al. 2005). The sclerophyll plant communities exhibit relatively rapid rates of fine fuel accumulation (Conroy 1996). This characteristic, combined with their shrubby structure and the relatively frequent recurrence of severe fire weather results in conditions conducive to uncontrollable fires. Major episodes of fire resulting in large areas burned and loss of property occur somewhere in the region once or twice a decade (Cunningham 1984, Conroy 1996, Gill and Moore 1996, Bradstock and Gill 2001).

Most of the natural vegetation within the region, including much of the GBMWHA, lies within National Parks, Nature Reserves and Conservation Areas that are managed by the NSW National Parks and Wildlife Service (NPWS) within the NSW Department of Environment and Climate Change (Fig. 1). Other significant areas of fire-prone bushland are managed by the Sydney Catchment Authority (SCA) and Local Government. In effect, the bulk of fire management issues fall within the domain of public land managers and the NSW Rural Fire Service which has formal responsibility for co-ordination of planning and management of operations.

The configuration of biodiversity and human assets provides a challenging arena for fire management. Fires therefore pose significant but largely unmeasured risks to people and their property. The ensuing fire regimes have a complex ecological interplay with biodiversity. Appropriate fire regimes have a positive role in replenishing the structure and composition of vegetation and habitats through effects on plant recruitment, competition and reproduction. Inappropriate fire regimes may have negative effects on species populations through disruption of resilience capacity, and directly affect the quantity and quality of water for Australia's largest urban area. The challenge of resolving risks to these values posed by bushfires and climate change in the Sydney basin is nationally significant. The trade-offs needed to manage risk to these values are largely heuristic and speculative (e.g. Gill et al. 2003). The prospect of climate change adds further uncertainty and heightens the challenge.

In this study we examined approaches to understanding risks to key values – particularly biodiversity and urban assets - and the way these may vary under climate change.
We used three case study areas within the region that pose important contrasts in terms of the balance of urban versus bushland extent, overall size, proximity to coast, elevation, diversity of terrain and significance of catchment protection issues. They therefore offer deeper insight into the influences of biophysical variation on fire regimes within the region and thus provide a representative basis for assessing the robustness of differing management strategies and sensitivity to climate change scenarios.

• Aims and objectives of the study

This study used the opportunity provided by landscape-scale simulation modelling to explore:

1. Potential effects of climate change on fire regimes
2. Effects of alternative management scenarios (via use of prescribed fire) on fire regimes
3. The way these alternative management outcomes may be altered by climate change
4. The quantitative nature of bushfire risk to people, property, biodiversity and catchment values
5. The degree to which risk will be modified by typical biophysical and land use variation within the region.

Simulation modelling provides a means to explore the dynamics of fire regimes that emerge from complex interactions of climate, weather, fuel and landscape patterns. Such explorations over large spatial/temporal scales are rarely possible through field experiments or analyses of historical data. In addition, the combination of alternative management approaches, climates and multiple values produces a plethora of scenarios. Simulation is probably the only feasible means to rigorously explore and compare these scenarios.

Modelling of this kind is best regarded as an exploration of the sensitivity of the system – particularly the sensitivity of fire regimes to key drivers (weather, fuel, management) in this case. The results should not be treated as explicit forecasts but rather as an indication of the relative change in risk that may ensue from changes to drivers and fire regimes. The distinction between forecasting as opposed to exploration of system sensitivity is subtle but important and the conclusions provided below need to be interpreted accordingly.

The study used recent work on prediction of changes to fire weather (i.e. Hennessy et al. 2005), specifically drought indices and the McArthur Forest Fire Danger Index (FFDI), for this purpose. Such work provides valuable insight into one aspect of global change (i.e. fire weather) and its potential role in altering bushfire risks. Other aspects of global change may also effect fire regimes in the near future, such as: changes to niches of key species that provide the bulk of bushfire fuel; effects of climate on plant productivity (also affecting fuel); effects of elevated CO_2 on plant growth and decomposition of dead plant material (also affecting fuel), and; changes to ignition rates – lightning and anthropogenic. The specific effects of such factors and their interactions are largely unknown within the study area, though some speculative trends are discussed below. As a result, the potential effects of these factors were not included in the modelling.

This study therefore constitutes a partial exploration of climate change effects on bushfire risk. In particular it offers insights into the degree to which a change in fire weather translates into a change in area burned, fire frequency (or length of inter-fire interval IFI), fire intensity and fire season. Such effects remain largely unexplored in the scientific literature. The potential effects of other global change factors on fire regimes are considered in discussion below.

It is emphasised that the approach used here, contrasted the predicted outcome of forecast changes to weather in 2050 on fire regimes, with outcomes (observed and simulated) under contemporary weather. We did not attempt to simulate progressive changes in fire regimes that may be caused by transient change in climate over the next 50 years.

METHODS

• Study landscapes

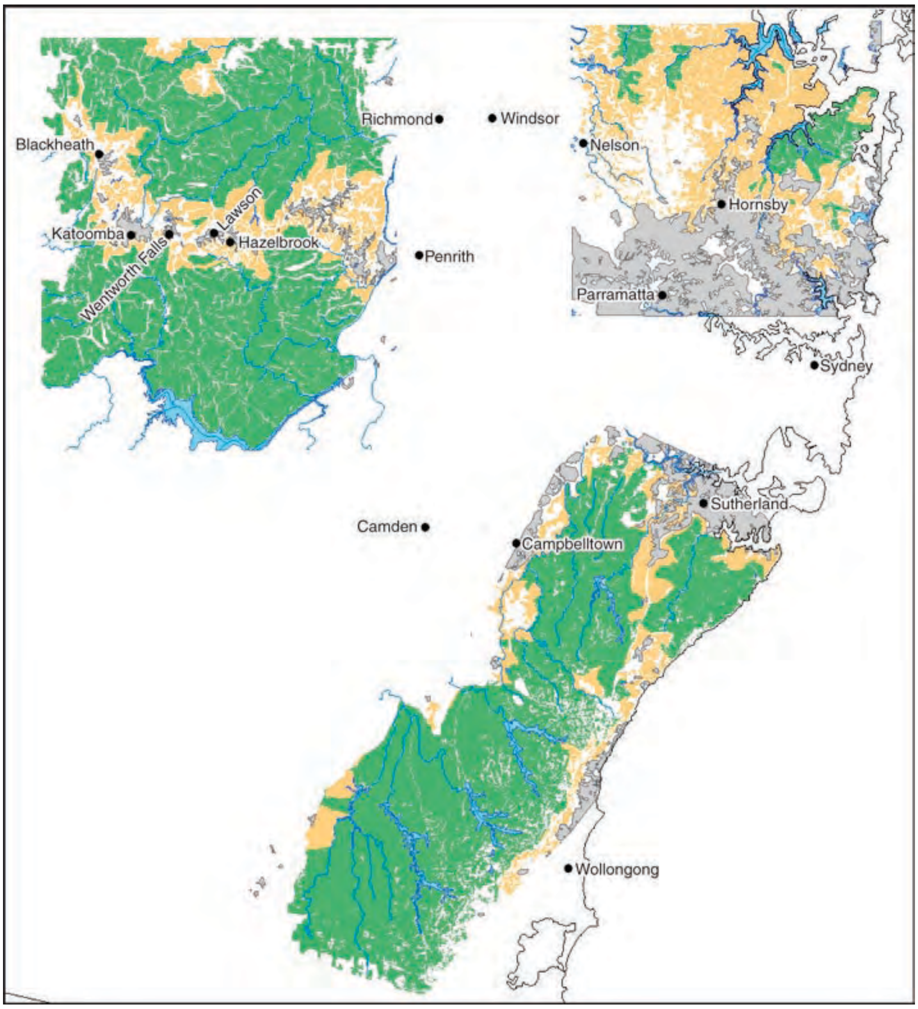
This study focuses on three areas (totalling 4100 square kilometres) within the greater Sydney basin which form discrete case study areas in the north, south and west of the region (Figs 1 & 2). These case studies encompassed respectively:

1) **HK** - **The Hornsby Local Government area** (covering much of the Hornsby Plateau), including the **Ku-ring-gai Chase and Marramarra National Parks**. The case study area (68,231ha) extends from Broken Bay in the north, south to Sydney's northern suburbs.

2) **BM - Blue Mountains Local Government area**, including much of the Blue Mountains National Park, along with significant portions of the Kanangra Boyd and Wollemi National Parks and the Warragamba catchment, including land managed by the Sydney Catchment Authority (case study area 187,718 ha).

Figures 2a & 2b The distribution of blocks used to define alternative strategies of prescribed burning in the case studies (see Fig. 1).

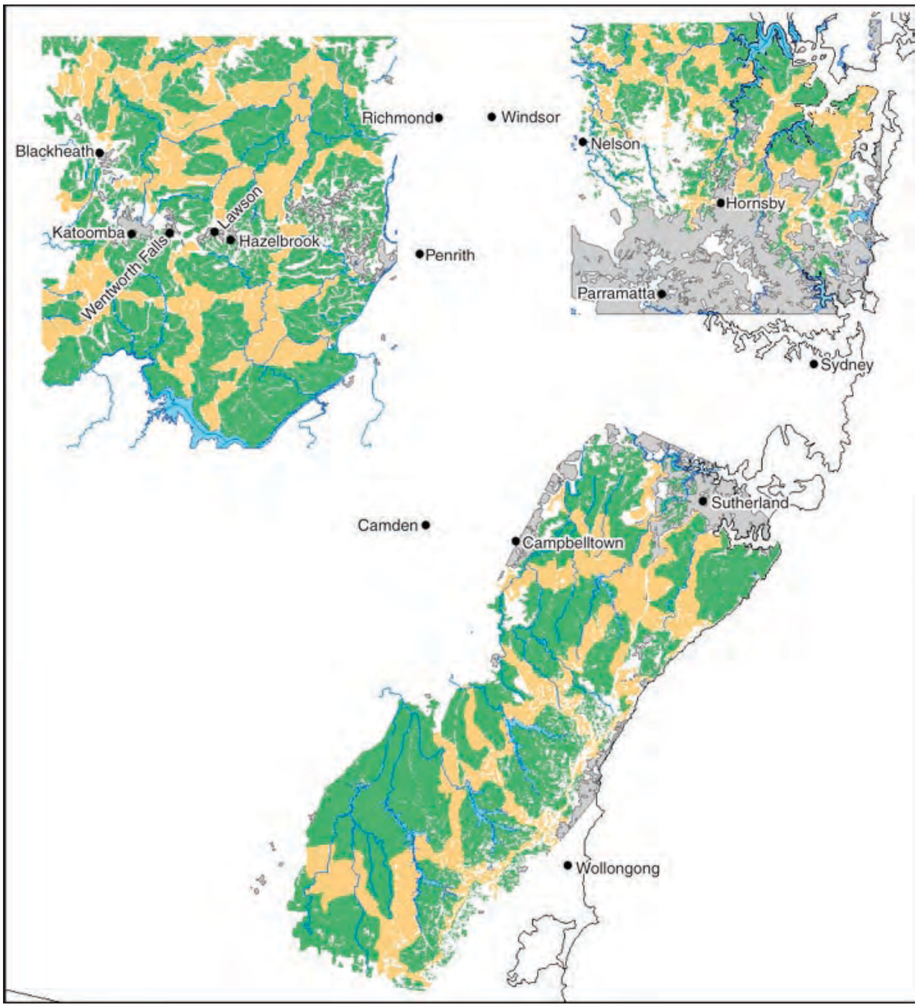
Figure 2a Edge blocks are located adjacent to urban areas (treated under Edge and Constrained strategies), while non-edge blocks (treated under Unconstrained, Constrained and Linear strategies) are situated more remotely.



3) **WP - The Woronora Plateau**, containing Heathcote, Dharawhal and Royal National Parks and the Metropolitan Water Catchments (Woronora, Cataract, Cordeaux, Nepean and Avon reservoirs) and managed by the NPWS and SCA. These lands abut the urban area of greater Wollongong to the east and part of the urban corridor of southern Sydney (e.g. Engadine, Heathcote). Case study area was 154,287 ha.

The potential for fire spread in all three areas was naturally constrained to varying degrees by a range of landscape features and development patterns (Fig. 2). The exposure of urban interface (e.g. length and juxtaposition to natural fire spread direction) varied among the case studies (see below and Fig. 2). For example, the Hornsby (HK) area contained significant areas of development to the west and south of bushland which are 'upwind' of the usual path of major fires. Development in the Blue Mountains (BM) case study, is surrounded by bushland in most directions. Much of the development in the Woronora (WP) case study (i.e. the Illawarra) lies in the potential path of major fires, but is situated beneath a major escarpment.

Figure 2b The layout of the Linear strategy in each case study. Groups of blocks were selected to form continuous line segments in map. Effort was varied through selection of differing numbers of segments for annual treatment (see text).



• Model application

We use a modified version of the landscape fire model FIRESCAPE (Cary and Banks 1999) and a new prescribed burning model implemented in the LAMOS landscape modelling shell (Lavorel 2000) to perform all simulations for this study.

FIRESCAPE is a process-based dynamic simulator that generates spatial patterns of fire regimes in topographically complex landscapes. It explicitly deals with effects of weather, fuel and terrain at a variety of spatial and temporal scales (Keane et al. 2003). It operates over a gridded landscape (e.g. 1 hectare cells) containing a vegetation type, elevation and slope. FIRESCAPE uses high temporal and spatial resolution data on weather and fuel to explicitly spread fires from point ignitions. Along with modules to represent, weather, fuel and ignitions, FIRESCAPE utilizes topographic information (slope, aspect and altitude) to govern the initiation and spread of fires

FIRESCAPE uses daily weather variables, (maximum and minimum temperature, precipitation, vapour pressure, wind speed and direction) and interpolates these to hourly values during a fire (Cary and Banks 1999). Lapse rates are applied to the appropriate weather variables at each elevation. A soil dryness Index (e.g. Mount SDI) is calculated each day, using relevant equations (see below). SDI is used to calculate a daily drought factor (DF) by incorporating the time and amount of the last rainfall event (Noble et al. 1980). Fuel loads are modelled using an approach first proposed by Olson (1963), which empirically accounts for both litter decomposition and input rates. This approach has been found to be suited to Australian systems (e.g. Raison et al. 1983). A spatial resolution of 1 hectare was used to model fire spread and underlying fuel and terrain influences in this study.

The principal outputs from FIRESCAPE are maps of annual fire patterns, timing and intensity which can be analysed to obtain the spatial and temporal variation in the major components of the fire regime. FIRESCAPE simulations can be performed for long time periods, allowing insights into the development of fire regime patterns (fire frequency, intensity and season – Gill 1975) to be developed across large landscapes. For example, Cary (2002) used FIRESCAPE to examine responses of fire regimes to climate change in the ACT region. King et al. (2006, 2008) modelled long term responses of fire regimes and vegetation under varied strategies of prescribed burning in western Tasmania. FIRESCAPE therefore provides an experimental basis for exploring the effects of variation in drivers such as climate and management. FIRESCAPE is therefore the most appropriate simulation platform currently available for investigating long term responses of fire regimes to climate change and other factors, in eucalypt dominated vegetation.

• Ignition

Both natural (lightning) and anthropogenic ignition sources were modelled. Historical information on ignitions (de Ligt 2005ab) was examined to determine spatial and temporal patterns. Source of ignition was found to be highly variable and uncertain. No reliable information was available for temporal bias (i.e. seasonal or inter-annual scales) in unplanned anthropogenic ignitions over the three study sites. A stochastic function was therefore used to determine the number of unplanned anthropogenic ignition events each day of the year. By contrast, locations for unplanned anthropogenic ignitions were based on distance from roads after the study by de Ligt (2005a), who found declining incidence with increasing distance from roads (Fig. 3a). Initial annual, unplanned anthropogenic rates were therefore set at a level that approximated contemporary trends and then further adjusted during model calibration (see below).

Figure 3a Anthropogenic ignition probabilities as a function of distance from road

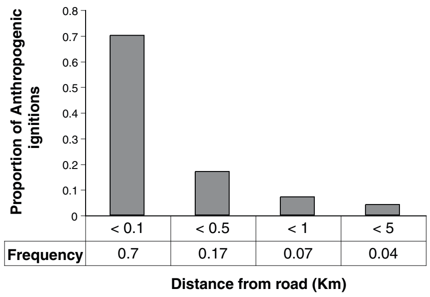
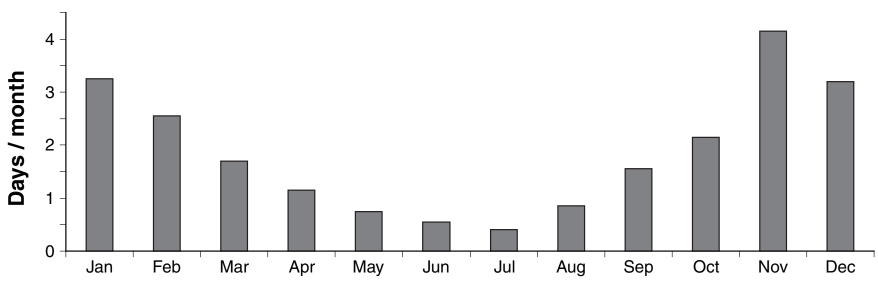


Figure 3b Mean prescribed fire area for 1977 to 2003 in the case study landscapes and the modelled outputs for the same period.

No data were found to indicate any spatial bias for natural, lightning ignitions in the region. As a consequence the location of lightning strikes was modelled randomly. Thunder days (Kuleshov et al. 2002) in the Sydney region have a strong autumn/summer/spring bias (Fig. 4) and these data were initially used to calibrate intra-annual patterns of lightning ignitions in the simulations. Rates of lightning ignitions (i.e. as a proportion of thunder days) were determined during model calibration, in conjunction with anthropogenic ignitions, to arrive at a net modelled, annual ignition rate that approximated contemporary patterns. Modelled ignition rates from both natural and anthropogenic sources were then held constant under predicted 2050 climate scenarios.

Figure 4 Monthly distribution of thunder-days in Sydney (Kuleshov et al. 2002)



• Fire spread

After ignition, the model spreads fire deterministically from a cell to its immediate neighbours based on an elliptical spread equation (Van Wagner 1969). The rate of spread of the head fire (ROS) is calculated for each individual burning cell based on the current fire conditions and vegetation type. Rates of spread in other directions (e.g. flanking and backing fires) are also calculated. Equations based on McArthur's Forest Fire Danger Meter are used for forests (Noble et al. 1980). For shrub/heathlands, equations from Catchpole et al. (1998) are applied. In grasslands, ROS is calculated using the model of Cheney et al. (1998). Fire line intensity (kWm^{-1}) is calculated from the ROS and the fuel load (Byram 1959). Fire line intensity is used to determine the conditions under which a fire will extinguish (Cary 2002, McCarthy and Cary 2002).

The version of FIRESCAPE used in this project does not simulate fire propagation via fire brands (i.e. 'spotting'). This mode of propagation may enhance the ability of fires to bridge fuel discontinuities. This has important ramifications for interpretation of results: effects of prescribed burning on area burned via creation of fuel discontinuity will be maximised. The McArthur equation was modified to increase the ROS under high wind conditions, increasing from 1 to 3 times for wind speeds from 30 to 80 kph in accordance with empirical evidence (e.g. McCaw et al. 2008). Account was taken of wind speed reduction for shrub and grass fires under canopies based on Tran and Pyrke (1999).

• Weather

The study period was constrained by both the availability of reliable fire observations (de Ligt 2005b) and by the period covered by the climate change study of Hennessy et al. (2005). Both contemporary and 'adjusted' weather (i.e. to represent 2050 climate change) used in the modelling were based on a 26 year time series from 1st of July 1977 to 30th June 2003 spanning at least 5 cycles of the southern oscillation index (Australian Bureau of Meteorology 2008). Basic daily data for this period were used to produce inputs for FIRESCAPE. Two future climate scenarios for 2050 were produced by Hennessy et al. (2005) from the CCAM (Mark2) model, one for low (referred to here as the Low scenario) and a second for high rates of global warming (High scenario) based upon IPCC scenarios (SRES 2001). These involved adjustments of relevant variables (e.g. temperature, humidity, wind speed) for the 1977 – 2003 daily time series (methods described in Hennessy et al. 2005).

Observed weather data from Sydney was used as the basis for simulations for both the Hornsby and Woronora Plateau under both contemporary and predicted 2050 climate. Both areas are at similar altitude (although the southern extremity of the Woronora study area reaches 700 metres) and they have a similar degree of exposure to coastal influences. The Blue Mountains study on the other hand, had a stronger influence of westerly winds during the fire season, less of the moderating influence of the sea on temperature and humidity provided to maritime locations and is at a higher altitude (e.g. Katoomba 1030 m). For this reason, observed weather data from Katoomba was used for the Blue Mountains case study. Climate projections for the Blue Mountains were not produced by Hennessy et al. (2005) and therefore a relevant set of 'adjusted' data had to be created by applying the same degree of predicted change used for Sydney (i.e. the Hornsby and Woronora case studies), to the Katoomba observations (authors' unpublished data).

As mentioned above, FIRESCAPE requires six daily weather variables as input. The CSIRO climate scenarios provide maximum temperature, precipitation, 3 p.m. humidity and wind speed. To calculate minimum temperature, we used the minimum temperature from the weather station data and assumed no change in diurnal temperature range (DTR) when projecting to the future. We also assumed no change in wind direction for the climate projections. During the course of a simulated fire, temperature, wind and vapour pressure were interpolated on an hourly basis by the simulator. Wind speed and direction and vapour pressure were modified by the long-term mean differences between 3 p.m. readings and readings at other times over the 24 hour period.

Relevant lapse rates for temperature and precipitation in each case study were determined from a number of other weather stations, beside Sydney (Table 1). Relative humidity was recalculated for each hourly value of vapour pressure and temperature. Lapse rates were applied to temperature and precipitation. For maximum temperature the lapse rate is 6.5^{o}C per 1000 metres. For minimum temperature the lapse rate is 2^{o}C per 1000 metres. These values are empirically based upon differences in nearby weather stations at various altitudes. The change in precipitation with altitude was calculated linearly as:

$$p_a = (b + m \times a)p_r$$

where p is the precipitation, a the elevation and b and m were empirically derived station specific parameters (Table 1).

Study area	Station (BoM)	b	m
Blue Mountains	Katoomba (63039)	0.56	0.00043
Woronora Plateau	Sydney (66062)	1	0.0003
Hornsby / Ku-Ring-Gai	Sydney (66062)	1	0

Table 1 The Woronora precipitation lapse rate (see text) is based on mean annual difference in rainfall between Bankstown, Campbelltown, Bowral and Moss Vale. Katoomba is calculated on rainfall in Lithgow, Lidsdale and Mt Boyce. No interpolation was used in Hornsby as there are no suitable stations to provide data over the elevation range and this range is less than 300 metres.

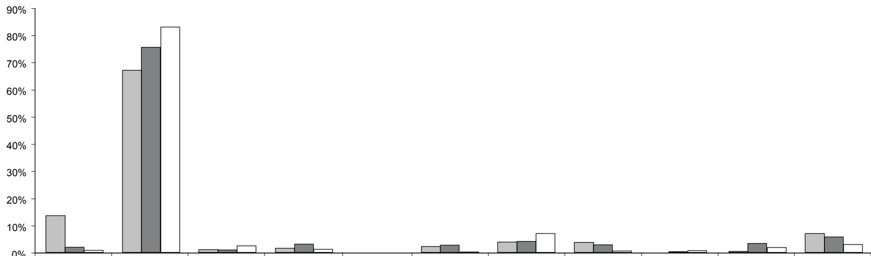
Soil dryness was calculated by summing daily precipitation to a 200mm limit after subtracting a constant amount for interception (1.27mm) and run-off (7%) and removing an amount based on season and temperature for evapo-transpiration from tables for Sydney using the method of Mount (1972). These rates were held the same for all three study sites.

• Fuel

Vegetation maps for the region were obtained from the NSW Department of Environment and Climate Change (DECC) using the classification of Keith (2004). Eleven vegetation formations were represented within the case studies, of which Dry Sclerophyll Forest with a shrubby understorey (DSF-S) represents approximately 75% of the total of all study areas (Fig. 5a, Appendix 1). The vegetation formations were used as a template for estimation of fuel type and corresponding accumulation patterns. Fuel accumulation curves are given for differing vegetation formations in Fig. 5b. These were calculated using the Olson (1969) model, based on data from Conroy (1996), Van Loon (1977) and Cary and Golding (2002).

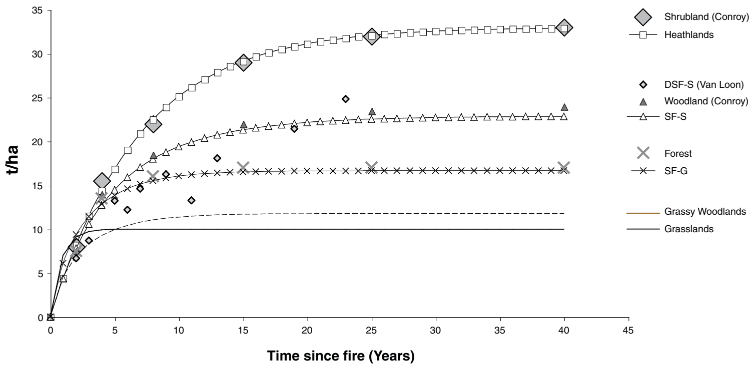
The spatial distribution of community types and their constituent fuel curves were held constant during this study. We did not attempt to model changes in both the vegetation distributions and fuel dynamics arising from changes to growth and decomposition under changing climate.

Figure 5a Proportion of vegetation formations in the three case study areas



	DSF (grass sub)	DSF(shrubby sub)	Forested Wetlands	Freshwater Wetlands	Grasslands	Grassy Woodlands	Heathlands	Rainforests	Saline Wetlands	WSF (grassy sub)	WSF (shrubby sub)
Blue mtns	13.57%	66.99%	0.99%	1.54%	0.00%	2.14%	3.74%	3.66%	0.00%	0.39%	6.98%
Woronora	1.94%	75.43%	0.89%	3.03%	0.00%	2.60%	4.02%	2.83%	0.29%	3.26%	5.71%
Hornsby	0.80%	82.92%	2.38%	1.17%	0.00%	0.09%	6.98%	0.49%	0.58%	1.71%	2.88%

Figure 5b Modelled fuel curves and observations from Conroy (1996) and Van Loon (1977)



• Model calibration

The model was calibrated to achieve correspondence between contemporary fire area data (1977 to 2003 – see de Ligt 2005b, NSW DECC unpublished records) and modelled fire area, under contemporary weather and management. As discussed above, there was considerable uncertainty in the field data on the causes and seasonality of ignition events and fuel accumulation patterns within the study area. Model calibration was therefore carried out by iteratively adjusting these variables within the bounds of this uncertainty. The aim was to produce a plausible set of model parameters that produced a close match between modelled and observed data on fire regimes for the study area.

The model was initially run with data on weather (1977-2003), vegetation, mean fuel accumulation and the recorded history of prescribed burning for the same period. Contemporary prescribed burning was simulated using the recorded average area treated per annum for 1977 - 2003 (Fig. 3b). These data represent a long term average level of treatment of 0.8 % (BM), 1.8 % (WP) and 0.9 % (HK) of the landscape per annum. These are probable overestimates of the area burned historically because prescribed fires tend to be patchy within target areas (see below). In subsequent prescribed fire simulations (see below), spatial heterogeneity within treated areas was more explicitly represented.

A model run of 1300 years (50 repetitions of the basic weather time series) was used to generate initial output (i.e. the average number of fires, area burnt and fire size distributions). These data were compared to the history of unplanned fires for the same period. Only fires greater than 1 hectare were considered in this study.

As noted, the ratio of anthropogenic to natural ignitions was highly uncertain (i.e. there were many unattributed ignitions – de Ligt 2005ab) as was inter-annual variability in anthropogenic ignition frequencies and spatial distribution of natural ignitions. However, the proportion of total ignitions from either source was found to affect modelled fire size. Therefore, systematic adjustments were made to the following parameters for each study area to more closely fit the model outputs with the recorded history of fires in the region:

1. Probability of a ground strike on a thunder day
2. Daily probability of attempted anthropogenic ignitions
3. Changes to fuel loads as described below.

Within the limits of available data it appears that natural ignitions in the region are more likely to occur during periods of high fire danger and thus lead to larger fires. This was not the case with anthropogenic ignitions. Records (e.g. DECC unpublished) indicated that these occurred with regular frequency during the year regardless of fire danger and at a more restricted set of locations (near roads – de Ligt 2005a) thus producing smaller fires on average.

• Effects of climate change on fire regimes

The climate data used to generate 2050 scenarios of change in FFDI by Hennessy et al. (2005) were used to simulate effects of change in fire weather on area burned, inter-fire interval and fire intensity. The calibrated model was initially run for the 26 year time series (50 cycles – see above) of contemporary weather (1977 – 2003) plus the two corresponding 'adjusted' scenarios representing low and high levels of 2050 climate change (see above). Effects on number of fires, area burned, fire recurrence interval and spatial patterns of modelled fires under climate change were explored.

This approach provides a sensitivity analysis of fire regimes under the range of weather conditions experienced at the present and those predicted for 2050. We did not attempt transient simulations to explore effects of a potential continuous shift in climate from the present to 2050.

The method of Hennessy et al. 2005 does not alter the structure of the daily time series (i.e. the order of days characterized by particular levels of fire danger). The analysis therefore only represents the effects of adjusting the key weather variables on an individual daily basis to reflect climate change. It is therefore relatively conservative as it does not consider the possibility of higher order temporal change (i.e. changes to sequence of days) that could result from climate change. In particular, this approach does not consider changes to the length of periods of days without rain. This can have a significant effect on short-term changes in fuel moisture and resultant fire danger index (Noble et al. 1980).

• Effects of fire management on fire regimes

Different prescribed burning approaches were explored. These were composed of differing levels of "effort" (i.e. area treated per annum) and contrasting spatial patterns of treatment ("strategies"). They were chosen to represent contemporary approaches and commonly debated alternatives. All approaches involved the treatment of blocks defined on the basis of attributes (i.e. size, topographic and boundary features such as roads, trails, streamlines) that reflected current practices.

	Edge blocks	Mean area (ha)	Non-edge blocks	Mean area (ha)	Linear	Mean area (ha)
Blue Mntns	303	173	431	250	12	3577
Hornsby	573	62	60	160	12	1515
Woronora	101	170	342	320	11	3543

Table 2 Number and mean area of treatment blocks used for differing prescribed burning strategies in the three case studies

Blocks in each case study were classified as either abutting the urban edge or not (i.e. remote from the urban edge (Table 2, Fig. 2). Edge blocks contained areas defined and used as Asset Protection and Strategic Fire Advantage Zones under current Bushfire Risk Management Plans for the case study areas. These blocks were smaller on average than non-edge blocks and blocks in the Hornsby case study were smaller than those in the other case studies, reflecting the nature of the landscape and development patterns (Table 2, Fig. 2). The Hornsby case study was also composed of a much higher proportion of edge blocks (55 %) compared with the other case studies (12-13 %), as a result of these influences.

Four contrasting prescribed burning strategies were compared (Fig. 2):

i) **Edge** - treatment of blocks abutting the urban interface only

ii) **Unconstrained** - treatment of blocks at random

iii) **Constrained** - treatment of both "edge" and "non edge" in a 1:1 ratio

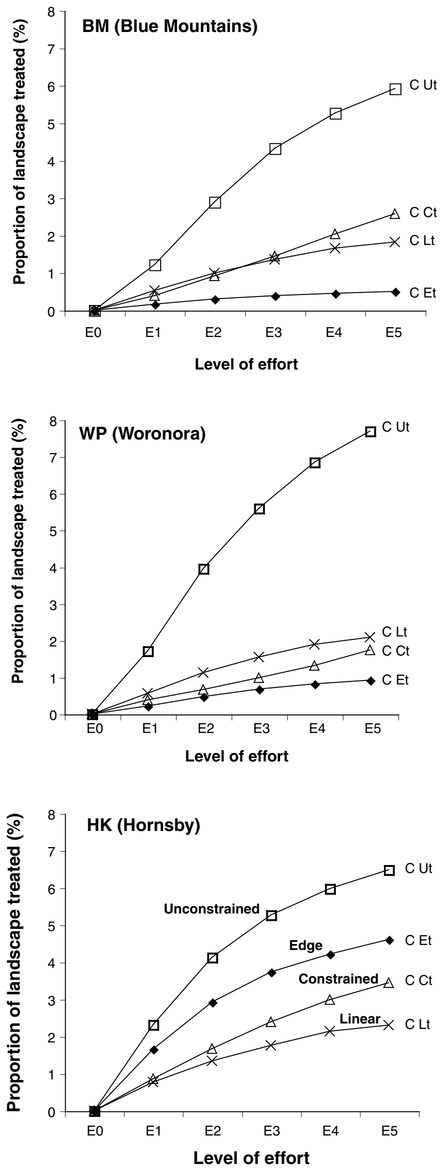
iv) **Linear** – strategic treatment of blocks in the landscape in linear arrays running approximately north–south and east-west (Fig. 2b).

The Edge strategy was targeted solely at reducing fire intensity adjacent to the urban interface, on the presumption that such a reduction will reduce ember attack and flame contact on buildings and provide safer conditions for fire suppression (Bradstock et al. 1998; Bradstock and Gill 2001). It can be regarded largely as a defensive strategy that does not seek to reduce incidence and potential spread of unplanned fires in the wider landscape. The latter three strategies, by contrast, seek to affect chance of incidence and spread and can therefore be regarded as part of an offensive strategy. By definition, the Constrained strategy mixed both roles.

Variations in annual level of treatment effort in each case were achieved by varying the number of blocks selected. Effort was varied across 5 standard levels of available blocks (4, 8, 12, 16 and 20%) for each strategy. These percentages yielded differing numbers of blocks treated for a given effort level among the strategies (Fig. 6) resulting in differing overall proportions of the total landscape treated (Fig. 6). The highest level of treatment (20%) equated to 'saturation' of effort, due to the minimum 5 year interval between prescribed fires that was imposed in the model.

Figure 6 Proportion of landscape treated for 4 prescribed burning strategies at 5 levels of effort

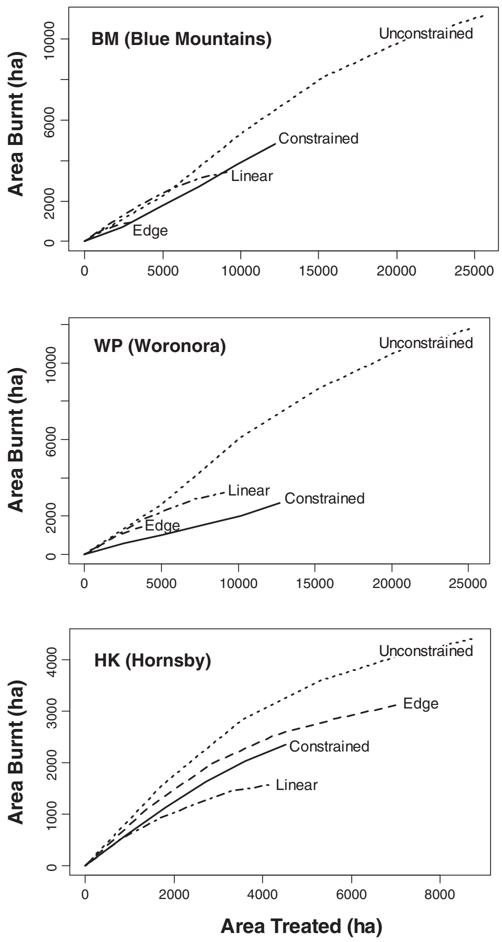
The number of blocks selected for treatment, at each level of effort, were 4, 8, 12, 16 and 20% of the total available.



Selection of blocks for treatment was based on an estimation of potential unplanned fire intensity derived from slope, aspect, vegetation class and fuel age, using the methodology developed by (Bradstock et al. 1998). Blocks with highest potential intensity were given priority for treatment, within the constraints of spatial strategy. Fire intensity potential was re-estimated each year within simulations and the priority list for treatment was re-ordered accordingly.

When a block was selected for treatment, each cell within was burnt with a probability of ignition success based upon a linear function of time-since-last-fire after Penman et al. (2007), reflecting typical conditions of relatively low FFDI (see below). This resulted in the burning of relatively consistent proportions of selected blocks (Fig. 7). Variations in effort had little effect on these proportions, though there were more distinct variations between strategies in some landscapes (Fig. 7), with overall, mean ratios of treated:burnt area of 2.41 (BM), 2.87 (WP) and 1.77 (HK). Only cells with litter fuel loads > 5 t ha^{-1} were burnt. Once treated, a block was unavailable for treatment for the next 5 years (minimum rotation period) and a post-treatment, residual fuel load of 5 t ha^{-1} was specified in accordance with results of local studies (Morrison et al. 1996). Prescribed fire intensity was estimated at an FFDI of 2, using the available fuel load. This value was chosen to represent typical conditions for prescribed burning. This study did not attempt to constrain prescribed burning to periods of suitable weather: it was assumed that sufficient days of appropriate weather were available (see Bradstock et al. 1998), irrespective of level of treatment. Rainforest was excluded from prescribed burning.

Figure 7 Relationship between area treated and area burned under different prescribed burning strategies



Simulations were carried out for each case study landscape, using ignition rates, level of replication and the climate scenarios described above. Outputs included number of fires (planned and unplanned), area burned (planned and unplanned), fire intensity (Noble et al. 1980; Gill et al. 1987), annual probability of large fires (e.g. > 1000 ha), annual probability of uncontrollable fires (i.e. intensity > 4000 kWm^{-1}) and length of inter-fire interval (IFI). Effects of prescribed burning strategy and effort on these outputs were explored in ways that were relevant to management values, described below.

• Risk assessment for people, property, biodiversity and ecological values

Various fire regime diagnostics were used to assess risk to biodiversity and people. Risks to people and property will be a direct outcome of the length of urban interface burnt by unplanned fires and the mean intensity of such fires at the interface (i.e. within 100 metres of the edge – Bradstock et al. 1998; Bradstock and Gill 2001; Gill 2005). Bradstock and Gill (2001) estimated that about half the total risk to people and property at or near the urban edge was attributable to factors which govern the spread of fires in bushland landscapes beyond the urban interface. The other half of the problem is accounted for by factors which govern the condition of the built environment. This project dealt with quantification of the component of risk that arises from landscapes and their condition. It did not deal with factors within the built environment. The results presented here need to be treated accordingly (i.e. they constitute an estimation of one key component of overall risk to people and property). Therefore, the area of edge blocks burnt by unplanned fires and the mean intensity of such fires within these blocks was estimated under the differing prescribed burning and climate scenarios. These measures provide an indicator of risk ensuing from the arrival of fires at the urban interface as a function of the condition of the landscape.

Biodiversity effects were evaluated using a TPC (Thresholds of Potential Concern) approach based on life history attributes of plant species and their responses to fire regimes. Bradstock and Kenny (2003) provided a spatio/temporal TPC approach for assessing the status of plant species in relation to length of inter-fire interval (IFI). This approach was developed in the Sydney region where IFI length is known to effect the persistence of particular plant functional types – particularly obligate seeders. Short IFIs can cause population decline or extinction, through disruption of reproduction, whereas long IFIs may result in decline via senescence and an absence of subsequent recruitment in unburnt conditions (Bond and van Wilgen 1996).

The approach developed by Bradstock and Kenny (2003) allows judgement of fire regimes across landscapes at a functional type level. At the level of individual species, it is best used to assess effects of IFI variation on widely distributed obligate seeder plant species, or species with potential habitat covering large areas. It is likely to be unsuitable for assessment of likely effects IFI variation on species with highly restricted distributions/habitat. More specific population viability models (e.g. Keith et al. 2008) may be better suited for such species. Use of such models is beyond the scope of this report but opportunities exist for linkages with the fire regime simulation results. While insights from this approach allow some direct assessment of effects of fire regimes on plant species they may also provide some indirect insight into effects on animal habitat (Kenny et al. 2004).

The approach deals with the additive effects of IFIs that are either too short or too long for species to persist at landscape scales. When such intervals prevail over the bulk of a landscape (i.e. > 50%) extinction at a landscape-scale may be likely. The approach therefore has a direct linkage to extinction probability of key functional types that are well represented in the most common vegetation formation within the region. Critical IFIs for the complex of shrubby, dry sclerophyll woodland/open forest are defined as < 7 years and > 30 years (e.g. Bradstock and Kenny 2003, Kenny et al. 2004; Table 3). These represent significant risk of exposure of obligate seeder plant species in the juvenile phase in the former case, and significant risk of senescence in the latter. Critical IFI 'thresholds' for other vegetation formations are given in Table 3. These are based on a state-wide analysis of plant life history data (Kenny et al. 2004).

Regime	Vegetation formation	IFI Threshold of Potential Concern
DSF-S	Dry Sclerophyll forest with shrubby understory	• Inter-fire intervals less than or equal to 7 years • No fire for more than 30 years
Heath	Heathland	• as above
DSF-G	Dry Sclerophyll forest with grassy understory Grassy woodlands	• as above
Wetland	Forested wetlands Freshwater wetlands Saline wetlands	• as above
WSF	Wet Sclerophyll forest with shrubby understory Wet Sclerophyll forest with grassy understory	• Inter-fire intervals less than 14 years • No fire for more tthan 40 years
Rainforest	Rainforest	• Any fire

Table 3 Inter-fire interval “Thresholds of Potential Concern” (see text) for the 10 modelled vegetation formations.

The IFI distribution for each combination of prescribed burning strategy, effort level and climate change was estimated from an evenly spaced sample of 4 % of the cells in each case study landscape. The IFI was estimated from data for the last 100 years of the each 1300 year simulation in the target cells.

Fire intensity in the wider landscape was also recorded and analysed to measure impacts on human protection and biodiversity conservation. An intensity of 4,000 kW m^{-1} demarcates the upper limit of effective fire suppression. The proportion of fires above this level of intensity in the wider landscape will provide a general indicator of likely suppression difficulty.

Fires > 10,000 kW m^{-1} in dry sclerophyll forests of this region are usually high intensity crown fires (Cheney 1981, Hammill and Bradstock 2006). Such fires may be lethal to arboreal vertebrates such as Possums, Gliders and Koalas, which are common in the region (Lunney et al. 2007, Gill and Bradstock 1995). Lethal intensity may cause temporary extinction only, not permanent extinction, because the potential can exist for the species to migrate back into the area from pockets of non-lethal intensities or from outside of the burned area (Gill and Bradstock 1995). The fire regimes suited to the permanent presence of such species in a location are different to those in which the species is temporarily present or excluded. Insights into the relationship between occurrence and spatial extent of fires of lethal intensity and probability of extinction at landscape scales are therefore lacking (cf. Mackey et al. 2002). We therefore used this measure of high fire intensity (annual probability in an average cell or ‘point’ in the landscape) as a coarse indicator of vulnerability of “intensity-sensitive” taxa – particularly arboreal mammals.

Rainforest (Table 3) can be regarded as a special case of a plant community that is sensitive to fire intensity, as it may contain plant species that are strongly disadvantaged by a single fire (Kenny et al. 2004). While many rainforest species resprout (e.g. Clarke et al. 2005) resilience to subsequent fire may be reduced.

High intensity fires may temporarily alter soil-surface water repellency (i.e. in the first year after fire). A temporary increase in the potential for movement of material from upper slopes (Shakesby et al. 2007) may result. Adverse consequences for water quality in reservoirs may then occur through increases in nutrient concentration in streams, though evidence of significant effects of fire on run-off in local catchments is lacking (Tomkins et al. 2008). Other factors, such as high intensity rainfall events, are required to trigger upper slope soil movement on slopes during the vulnerable post-fire phase. We therefore used high intensity fire probability as a coarse indicator of vulnerability of soils to post-fire movement.

• Analyses

We present and discuss the mean trends of simulated response data. The means were calculated using the 50 simulation cycles as quasi-replicates. These are not true replicates because an identical, deterministic time series of weather was used in each cycle, though there was considerable stochasticity in ignition timing and subsequent effects on fuel. We do not present estimates of variation around the mean as these were found to be small (authors' unpublished data; also see King et al. 2006).

RESULTS

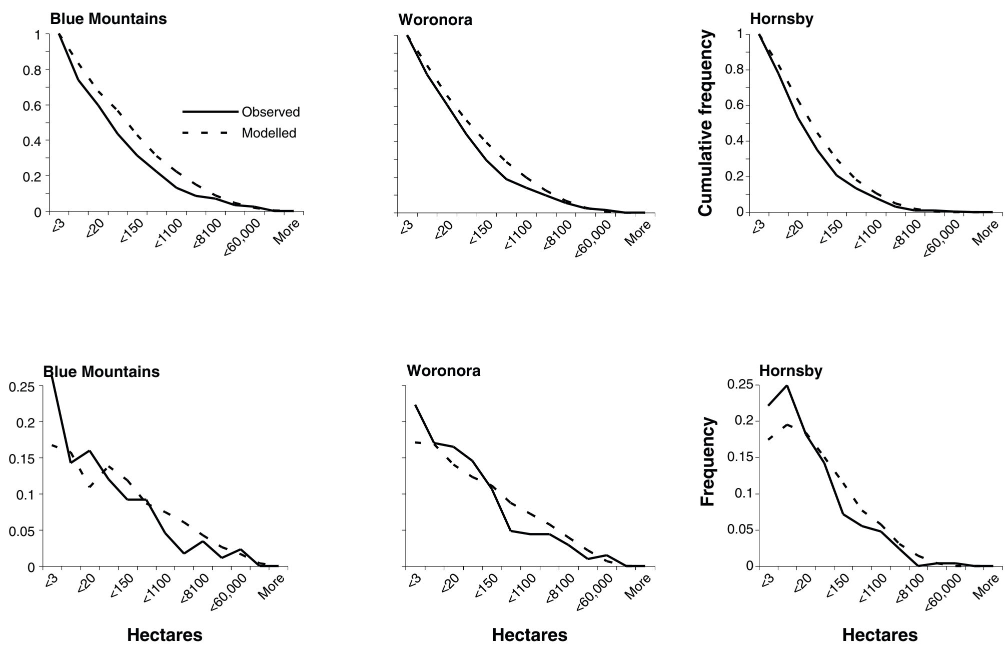
• Model calibration

Adjustments were made to the fuel curves (Fig. 3) for the predominant Dry Sclerophyll Forest with shrubby understorey (van Loon 1977, Conroy 1996) to achieve correspondence between results of model simulations and observed fire regimes (1977 - 2003). Heathland fuel accumulation was not adjusted. Adjustments, compared to lines of best fit, for Dry Sclerophyll in all three areas, were: WP + 18%; HK - 6%, and; BM - 8%. This resulted in maximum fine fuel loads of: 27 t/ha for WP and 21 t/ha HK and BM.

Following calibration, the simulated distribution of fire sizes under contemporary climate was compared to the observed period (1977 – 2003; Fig. 8). Middle sized fires tended to be over predicted (i.e. occurred with greater frequency than observed (Fig. 8) compared to very small (< 10 ha) and large fires (> 1000 ha). Simulated length of IFI was negatively related to the size of chosen landscapes among the case studies (i.e. BM = 19.7; WP = 20.3; HP = 25.7 years) possibly reflecting factors such as degree of fragmentation due to urban development.

Figure 8 Observed and predicted distributions (raw and cumulative) of fire size (Log) for the period 1977 to 2003 for the three case studies.

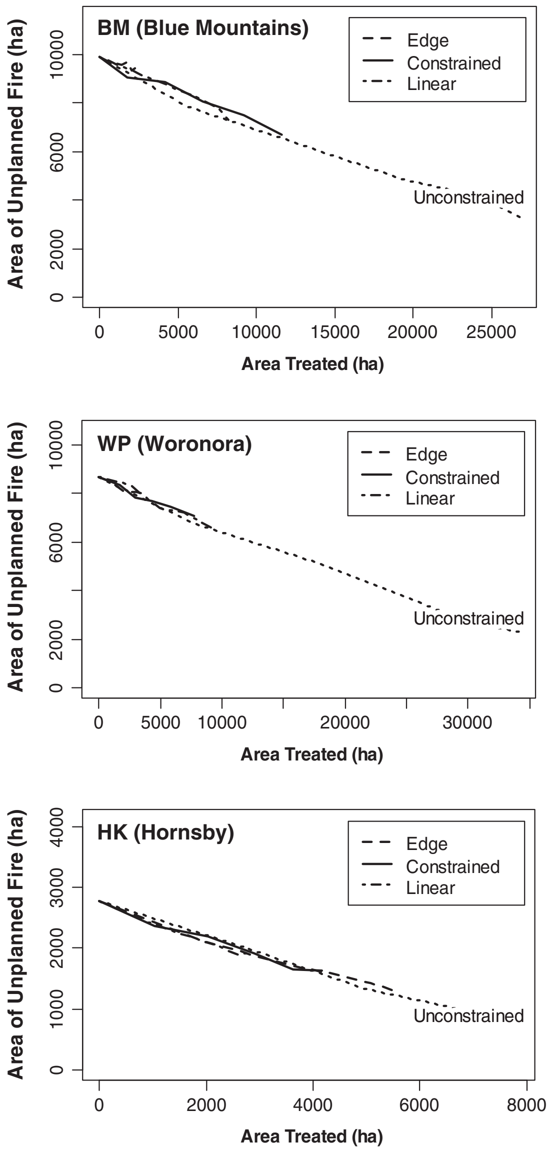
Predicted fire sizes were simulated using weather and prescribed burning history over the same period.



• Effects of fire management

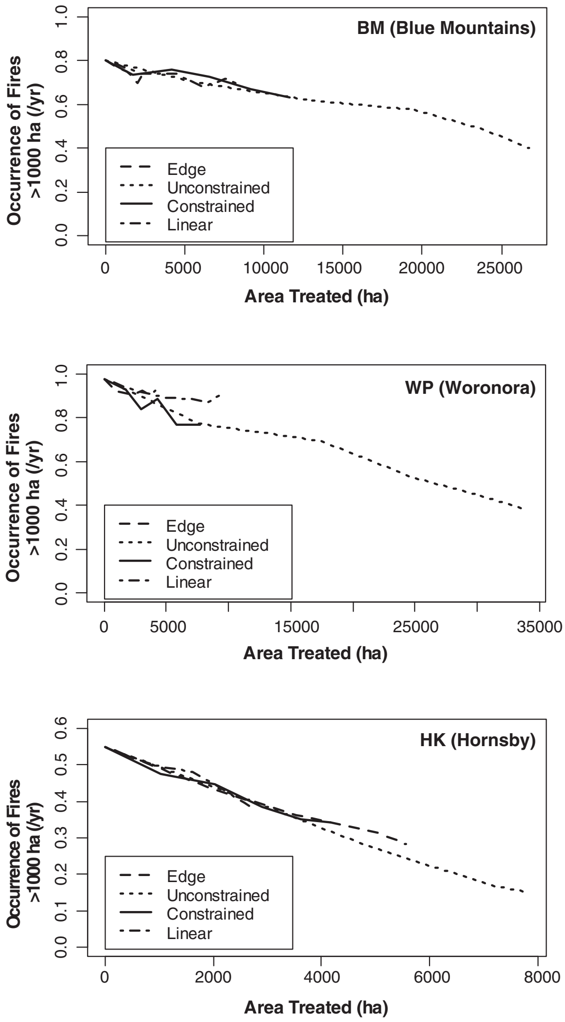
The mean area of simulated unplanned fires was reduced by increasing prescribed burning effort, in all case studies (Fig. 9). Spatial strategy of prescribed burning did not, however, strongly affect mean area of unplanned fires across the restricted range of comparable treatment levels. The relationship between mean unplanned fire area and prescribed burning effort was linear and negatively correlated. As a result, high levels of the Unconstrained treatment (beyond the range shared among treatments) yielded greater reductions in mean unplanned fire area compared to the other strategies. Thus for BM, the Linear and Constrained strategies reduced unplanned fire size by about 30% under maximum effort (circa. 8% of landscape treated), whereas the equivalent reduction (i.e. maximum effort > 15% of the landscape treated) under the Unconstrained strategy was about 60%. Similar effects occurred in the other case studies, though the disparity between the treatment effects was lower in HK.

Figure 9 **Effect of prescribed burning effort and strategy on mean size of unplanned fires**



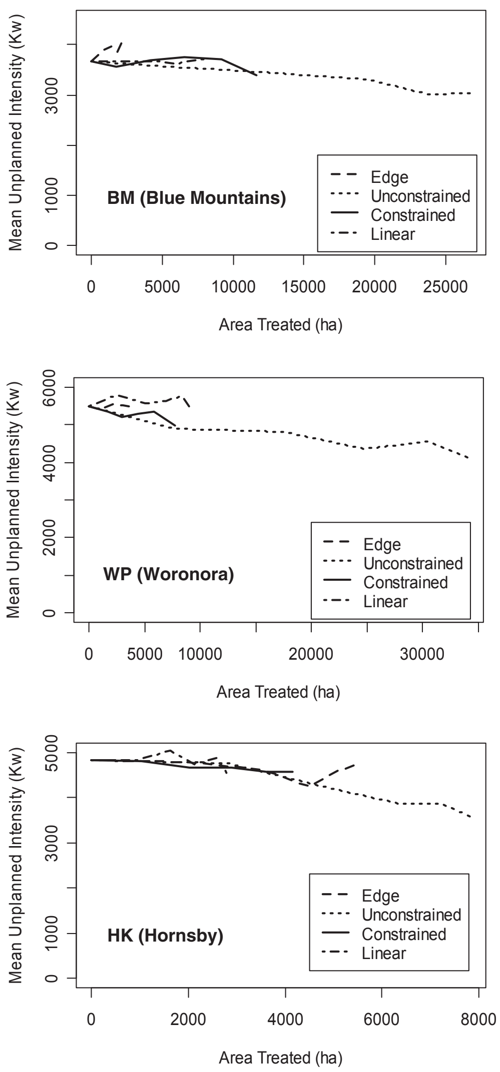
The simulated probability of large fires (> 1,000 ha) was reduced with increased prescribed burning effort in all case studies (Fig. 10). Prescribed burning strategy did not strongly affect large fire probability, except in WP where this probability tended to be higher under the Linear treatment compared with the other strategies. Under contemporary conditions, probability of large fires was higher in BM and WP (about 0.8 – 1.0 large fires per annum) than HK (about 0.5 large fires per annum). The simulations for BM corresponded with historical trends: e.g. Bradstock et al. (in review) estimated 51 days of large fire ignitions in the Blue Mountains for 1960 – 2003. The relationship between simulated large fire probability and level of prescribed burning effort was linear. High levels of the Unconstrained treatment therefore yielded greater reductions in large fire probability than the other strategies: e.g. Unconstrained treatment of 20% of the landscape per annum halved the probability of large fires. Moderate levels of treatment (circa. 5 % of landscape treated per annum) reduced large fire probability by 10 – 20 %.

Figure 10 Effect of prescribed burning effort and strategy on mean, annual probability of a large unplanned fire (> 1,000 ha)



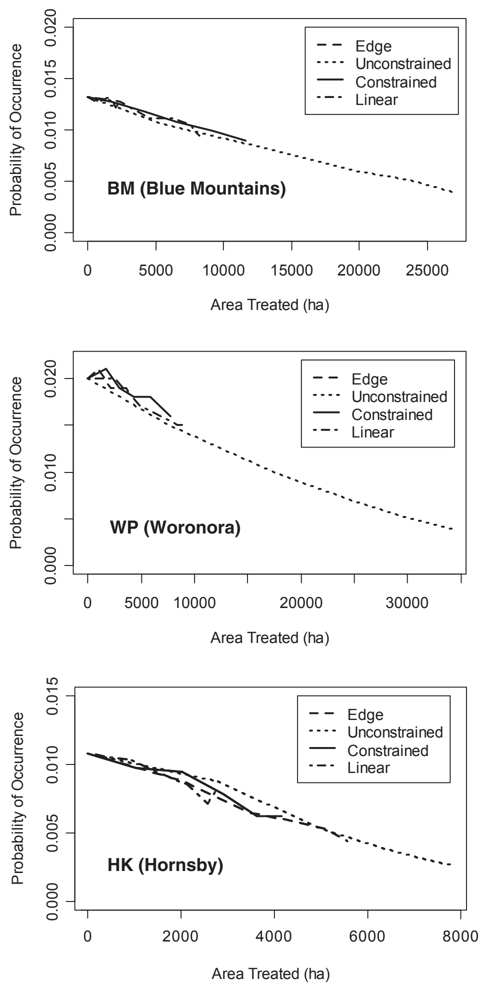
The mean intensity of unplanned fires was weakly affected by prescribed fire strategy and effort. In WP the mean intensity was generally unaffected by prescribed burning strategy and effort, and remained mostly in the 3500 – 4000 kW/m range (Fig. 11). For HK, there was little effect of strategy and effort level (circa. 4,500 kW/m average), with the exception of the Unconstrained strategy. Lower mean intensities resulted under the Unconstrained strategy and mean intensity declined with increasing effort to about 3500 - 4000 kW/m. For BM, mean intensity (circa. 4000 – 5500 kW/m range) declined with increasing effort for the Linear and Unconstrained strategies, with a tendency for the Constrained and Edge strategies to yield higher mean intensities than the other strategies. High levels of Unconstrained treatment yielded a greater reduction in mean intensity than the other strategies.

Figure 11 Effect of prescribed burning effort and strategy on mean unplanned fire intensity



The point-scale, mean annual probability of uncontrollable fires (i.e. > 4000 kW m^{-1} intensity) was relatively low (circa. 1 in 100 years in BM and HK; circa. 1 in 50 years in WP). This reflected the combination of overall rate unplanned fire recurrence (see below and Fig. 14) and mean intensity (Fig. 11). Probability of fire of uncontrollable intensity was mostly insensitive to treatment strategy but declined with increasing treatment level (Fig. 12). Very high levels of Unconstrained treatment more than halved the probability of uncontrollable fires in all case studies (Fig. 12). Decline in uncontrollable fire probability at high levels of Unconstrained treatment was greatest in WP and HK.

Figure 12 Effect of prescribed burning effort and strategy on the annual probability of unplanned fires exceeding a controllable intensity (i.e. > 4,000 kW/m) at a point in the landscape



• Effects of climate change on fire regimes and prescribed fire

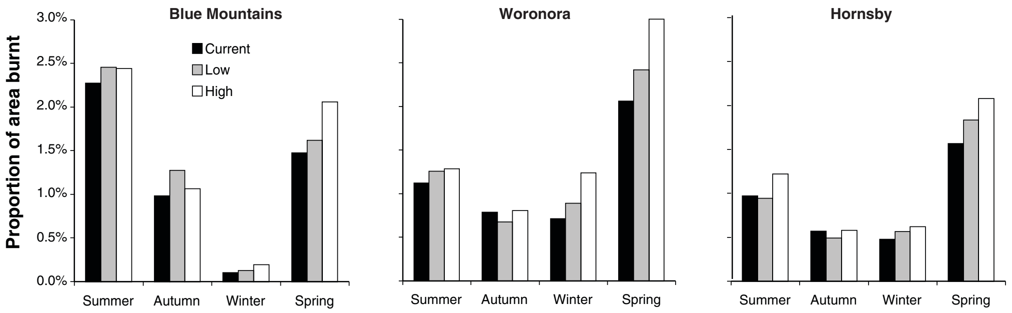
The relative change in the annual sum of maximum FFDI for the sample period of 1977 - 2003 was similar for both Sydney and Katoomba (Table 4). The monthly trends in FFDI showed that November had a relatively low, mid spring FFDI and changed little with the two climate projections. This was possibly due to the relatively small number of days without rain in this month. Although the total rainfall was reduced for both climate projections, rainfall events were not reduced below 2 mm. With this formulation, the number of days without rain remained unchanged. This had the effect of moderating the drought factor and in turn the FFDI calculation. Given that October and December are peak periods of high fire risk and that November has the highest number of thunder days of any month, it is likely that estimated of effects of climate change on fire activity in spring were conservative.

		∑FFDI	Area burnt	Decline in inter-fire interval
Blue Mountains	Low High	4% 11%	13% 19%	9% 11%
Woronora	Low High	5% 12%	12% 35%	11% 24%
Hornsby	Low High	4% 12%	7% 26%	7% 20%

Table 4 Relative change from current (modelled) conditions for annual ∑FFDI, average area burnt and average inter-fire interval of unplanned fires.

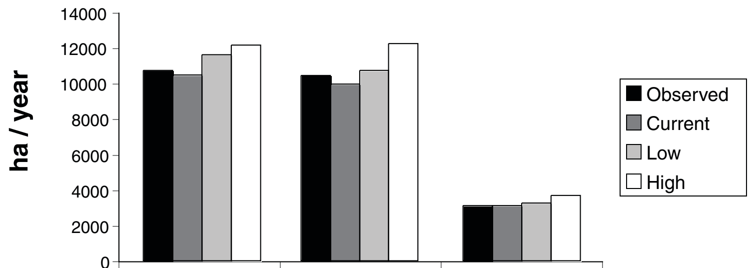
Simulations for all case study areas showed a strong bias toward spring-summer as the seasons of highest area burned (Fig. 13). This matched trends in recent fire history (authors' unpublished data). All case studies showed a strong increase in area burned in relation to climate change in spring, with the High climate change scenario causing a larger increase than the Low scenario. Responses to climate change in summer were more limited. In WP and BM, both High and Low scenarios produced a similar increase, whereas in HK only the High scenario increased area burned. BM showed a tendency toward higher area burned in autumn, particularly under the Low scenario.

Figure 13 Predicted seasonal proportion of landscape burnt by unplanned fires under contemporary and future climate for the case studies



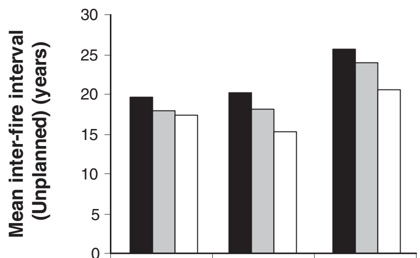
The overall level of change in simulated area burned under climate change (7 – 13 % Low and 26 -35 % High) exceeded the change in FFDI, by a factor of about two (Table 4, Fig. 14). Commensurate reductions in average IFI (7 – 11% Low and 11 – 24% High) were simulated under climate change (Table 4, Fig. 14).

Figure 14a Predicted mean annual area burned (prescribed plus unplanned) under contemporary and future climate in comparison to observed area burned (1977 – 2003) for the case studies



	Blue Mtns	Woronora	Hornsby
Observed	10719	10429	3119
Current	10450	9963	3102
Low	11593	10711	3259
High	12136	12238	3684

Figure 14b Predicted mean interval between unplanned fires under current and future climate assuming contemporary levels of prescribed fire



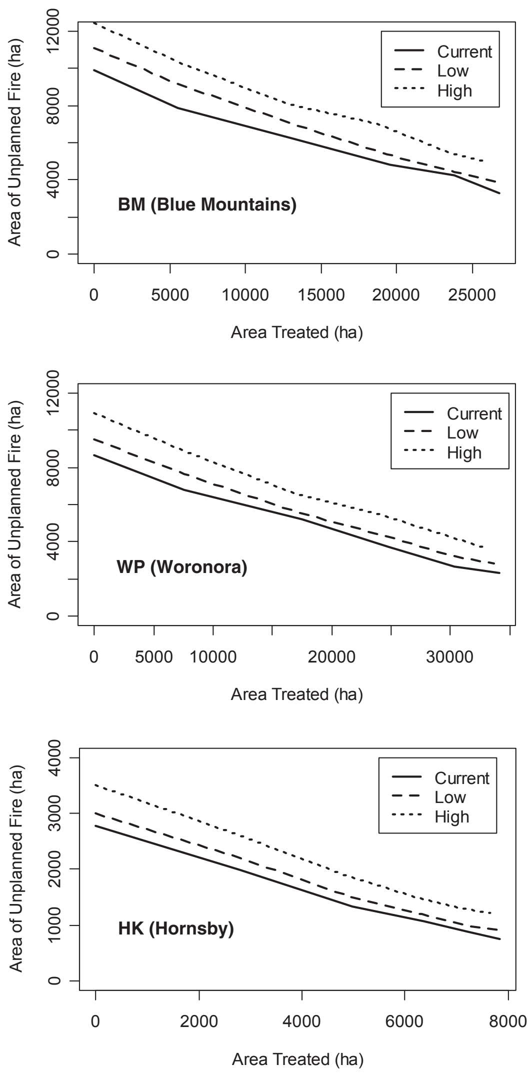
	Blue Mtns	Woronora	Hornsby
Current	19.7	20.3	25.7
Low	18.0	18.1	24.0
High	17.4	15.3	20.6

There was bias in the modelled, contemporary spatial pattern of IFI in all case studies, but this was most pronounced in BM and WP (Appendix 2). For example, shorter IFI was evident in east of BM (Lower Blue Mountains area) and in the north-east of WP (e.g. the Royal National Park). Homogeneous areas with relatively short IFI (5-10 years) were simulated in these localities. These biases reflected probable influences of modelled ignition sources and direction of spread of fires: i.e. the influence of westerly winds. The pattern of this effect in the Royal National Park was partially influenced by the prominence of the heathland. These spatial patterns accord with known trends in fire history (de Ligt 2005b, authors' unpublished data). Climate change reinforced these patterns, meaning that the overall simulated trend for shorter IFI was relatively evenly spread throughout the case study landscapes (Appendix 2). Nonetheless, the level of area exposed to relatively short IFI markedly increased under climate change. The implications for human protection and IFI sensitive biodiversity are discussed below.

By contrast, the intensity of unplanned fires was less strongly affected by climate change (Appendix 3). Strong variability in the modelled spatial patterns of intensity was evident in all case studies under contemporary climate. This reflected the diverse terrain of the region, with a bias toward higher intensities on upper, westerly aspects and ridge-tops, reflecting known patterns of fire severity captured in remote-sensing studies (Chafer et al. 2004, Hammill and Bradstock 2006, 2008). Climate change did little to alter these, particularly in BM and HK (Appendix 2). In WP there was an increase in mean intensity under climate change in some of the areas exposed to short IFI (e.g. the north eastern portion – see above).

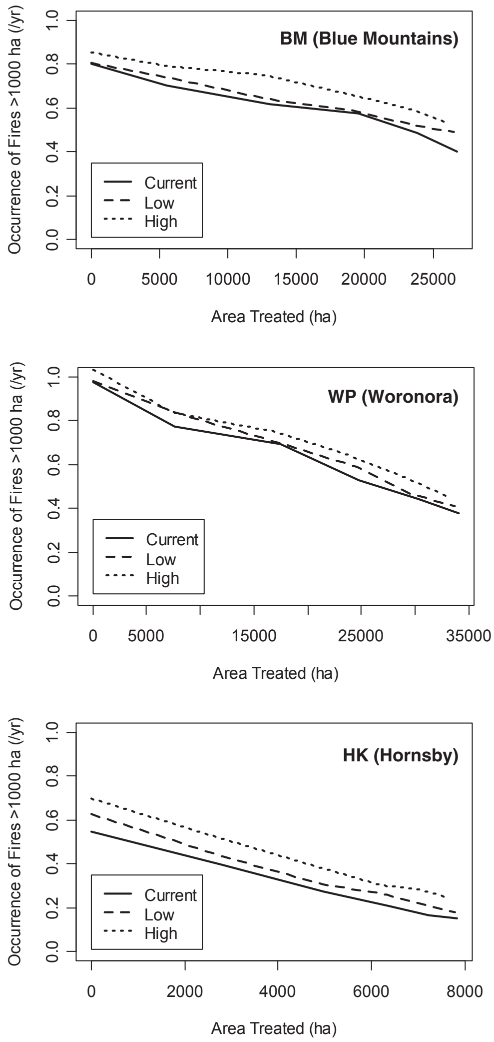
Climate change altered the relationship between level of prescribed burning and mean unplanned fire area. In all cases, for the Unconstrained strategy (Fig. 15), climate change (High scenario) increased mean unplanned fire area by > 20% at low levels of treatment effort and by about 15% at high levels of treatment. An increase in overall prescribed burning effort of about 5% of the total landscape would be required to counteract the effect of the High climate change scenario in all case studies, particularly at lower treatment levels.

Figure 15 Effect of climate change on mean unplanned fire area in relation to Unconstrained prescribed burning effort



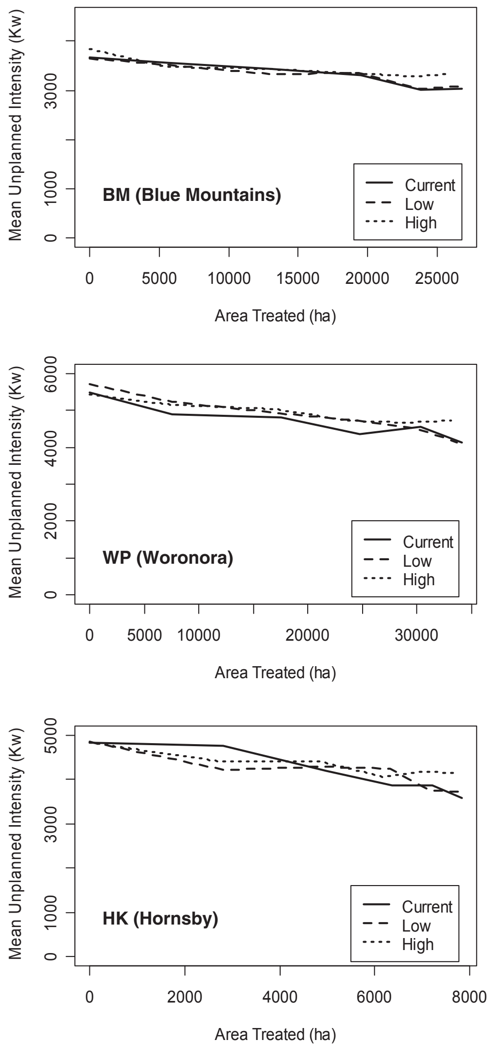
The probability of large fires (> 1,000 ha), was increased under the High climate change scenario (Fig. 16) by about 5% in BM and WP and > 10% in HK. Effects of Low climate change were less marked (i.e. half the level of increase in probability predicted for the High climate scenario).The increase in Unconstrained prescribed burning effort (additional % of landscape treated per annum) needed to counteract climate change (High scenario) varied from about 2% in WP to > 5 % in HK. Much smaller increases in effort were predicted to be required under the Low climate change scenario.

Figure 16 Effect of climate change on mean, annual probability of a large unplanned fire (> 1,000 ha) in relation to Unconstrained prescribed burning effort



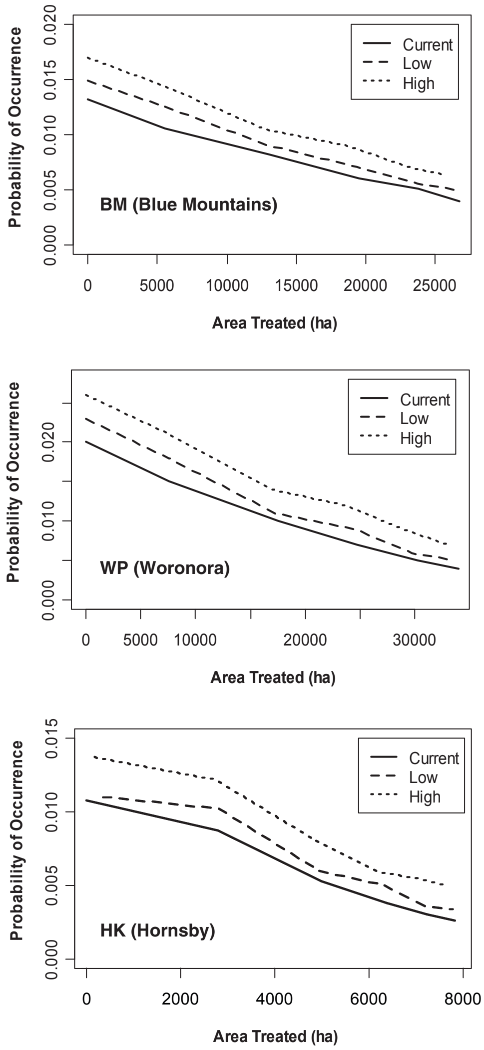
Climate change did not strongly affect the outcome of prescribed burning (Unconstrained strategy) on mean fire intensity (Fig. 17). In BM there was a tendency for mean intensity to increase at high levels of effort under the High scenario. In WP there was a small but more consistent effect of climate change on intensity across levels of effort. In HK there were varied effects of climate change, with a slight reduction in intensity at low levels of effort and the converse at higher levels.

Figure 17 Effect of climate change on mean, unplanned fire intensity in relation to Unconstrained prescribed burning effort



The probability of uncontrollable fires (> 4,000 kW/m) was increased by climate change (20 – 25 %) in BM and WP, but not appreciably in HK (Fig. 18). The effect of climate change diminished at high levels of Unconstrained prescribed fire. A substantial increase in prescribed burning effort (i.e. 5% of total landscape treated) was required to offset effects of climate change in BM and WP.

Figure 18 Effect of climate change on the annual probability of unplanned fires exceeding a controllable range of intensity (i.e. > 4,000 kW/m) at a point in the landscape, in relation to Unconstrained prescribed burning effort



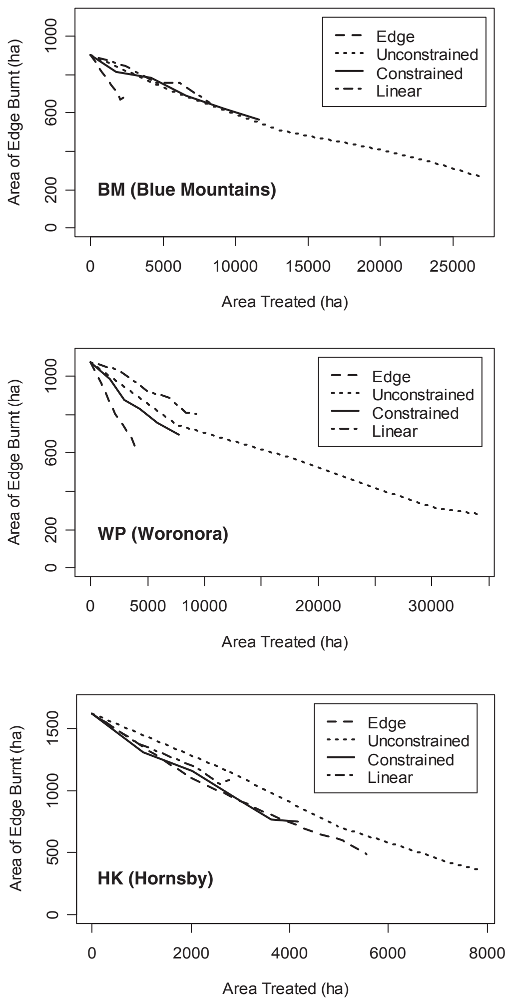
• Risk assessment for people, property, biodiversity and ecological values

- Protection of the urban interface

The area of urban interface (Table 2) affected by unplanned fires was sensitive to prescribed burning strategy and level of treatment in all case studies (Fig. 19). Affected area declined with increasing level of treatment, though this decline was greater under the Edge treatment compared with the other strategies for BM and WP (Fig. 19). Contemporary levels of prescribed fire (circa. 1 % of landscape treated, Fig. 4b – approximating the Constrained strategy) yielded a small (circa. 5%) reduction in relative risk (i.e. compared with zero area treated). The spatial pattern of unplanned fires under contemporary conditions (climate and management) indicated important differences between case studies (Appendix 2). Relatively high levels of unplanned fire recurrence were simulated adjacent to urban areas in the Lower Blue Mountains – Nepean River area of BM. In WP high levels of unplanned fire are simulated adjacent to Bundeena. These results conform to historical patterns of destructive fire (Cunningham 1984, Bradstock and Gill 2001).

For BM, the relationship between treatment effort and affected area differed little among the ‘non-edge’ strategies. For WP, the Constrained strategy more effectively reduced affected area than the Linear and Unconstrained strategies (Fig. 19). For HK, Edge, Constrained and Linear strategies differed little in their effects on affected area, but more effectively reduced affected area with increasing effort than the Unconstrained strategy (Fig. 19).

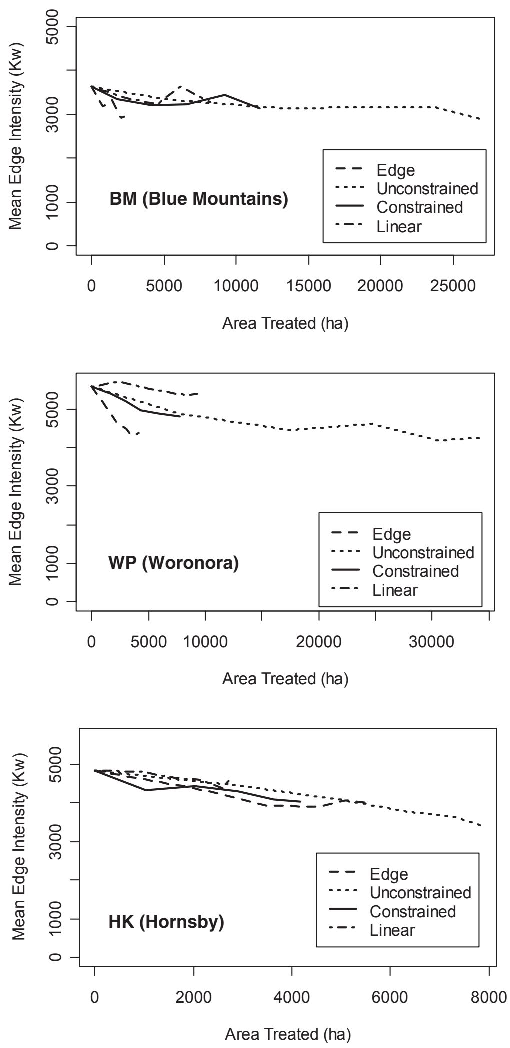
Figure 19 Effect of prescribed burning effort and strategy on mean area of the urban interface burnt by unplanned fires



In the larger landscapes (BM and WP), a greater reduction in affected area occurred under moderate to high levels of treatment by non-Edge strategies, compared with the Edge strategy (Fig. 19). Thus treatment of > 12,000 ha per annum (> 7 % of the landscape) by non-Edge strategies achieved a reduction in affected area of > 30% and a treatment level of > 25,000 ha per annum (> 14% of the landscape, Unconstrained treatment only) achieved a reduction of > 65%. This effect was largely absent in HK: higher levels of the Unconstrained strategy caused little decrease in affected area compared with the Edge and Unconstrained strategies (Fig. 19).

Intensity of fires at the urban edge was less strongly affected by prescribed fire strategy and effort (Fig. 20), particularly in the HK case study. In both BM and WP, the Edge treatment decreased the intensity at the urban edge at a greater rate as a function of effort than the other strategies. In WP the linear treatment decreased intensity at a lower rate than the other strategies. The Unconstrained strategy did not decrease urban edge intensity more than the Edge treatment at maximum level of effort in BM and WP, but in HK there was a slight decrease in this regard.

Figure 20 Effect of prescribed burning effort and strategy on mean intensity of unplanned fires within blocks adjacent to the urban interface



Climate change increased the area of urban development that may be exposed to simulated high levels of unplanned fire activity (Appendix 2). This effect was most pronounced in BM, where such an increase may affect urban/rural development in the lower Blue Mountains and western Sydney, fringing the Nepean River. Simulated effects in this regard in the other case studies were less marked (Appendix 2).

Climate change increased the area of urban edge that may be affected by unplanned fires under the Unconstrained strategy (Fig. 21). The high climate change scenario had a marked effect in this regard. In all case studies, high climate change effectively negated the effect of an increment in prescribed burning effort: i.e. an increase of about 4-5% of total landscape treated would be required to mitigate effects of climate change. Similar effects (not illustrated) were found for other strategies. Effects of climate change on urban edge intensity were small, with a slight increase evident in all case studies and at most levels of Unconstrained prescribed burning effort (Fig. 22).

Figure 21
Effect of climate change on mean area of the urban interface burnt by unplanned fires in relation to Unconstrained prescribed burning effort

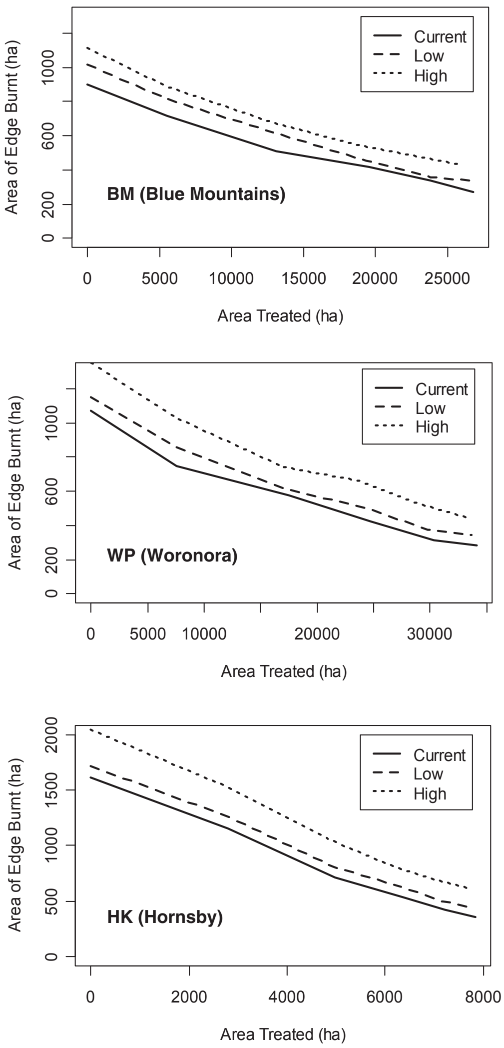
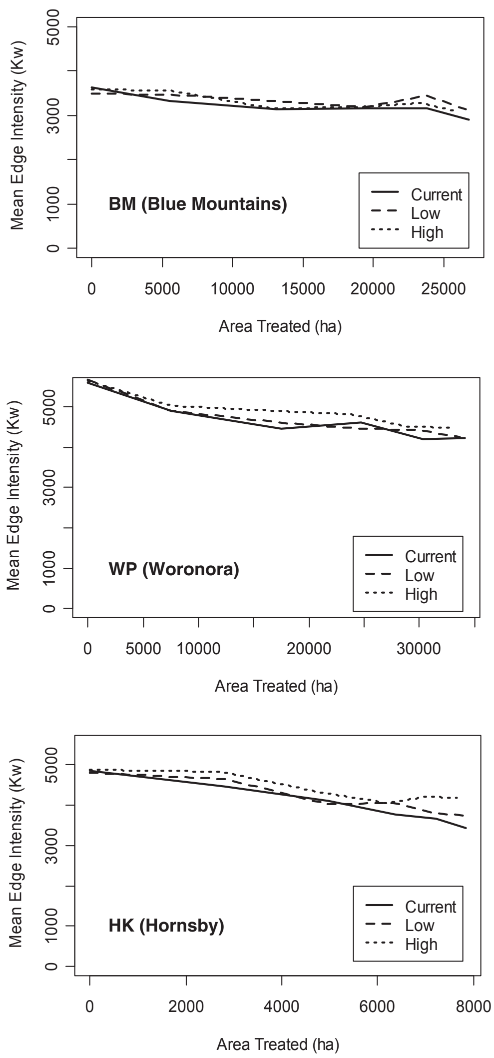


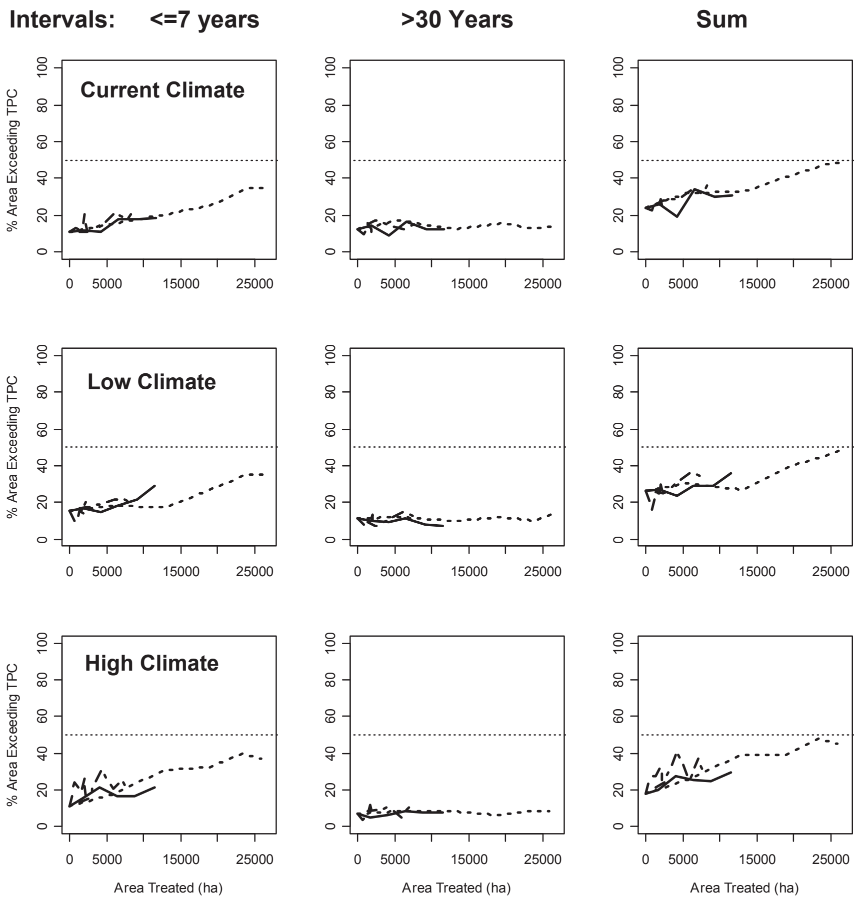
Figure 22
Effect of climate change on mean intensity of unplanned fires within blocks adjacent to the urban interface, in relation to Unconstrained prescribed burning effort



Conservation of biodiversity sensitive to IFI

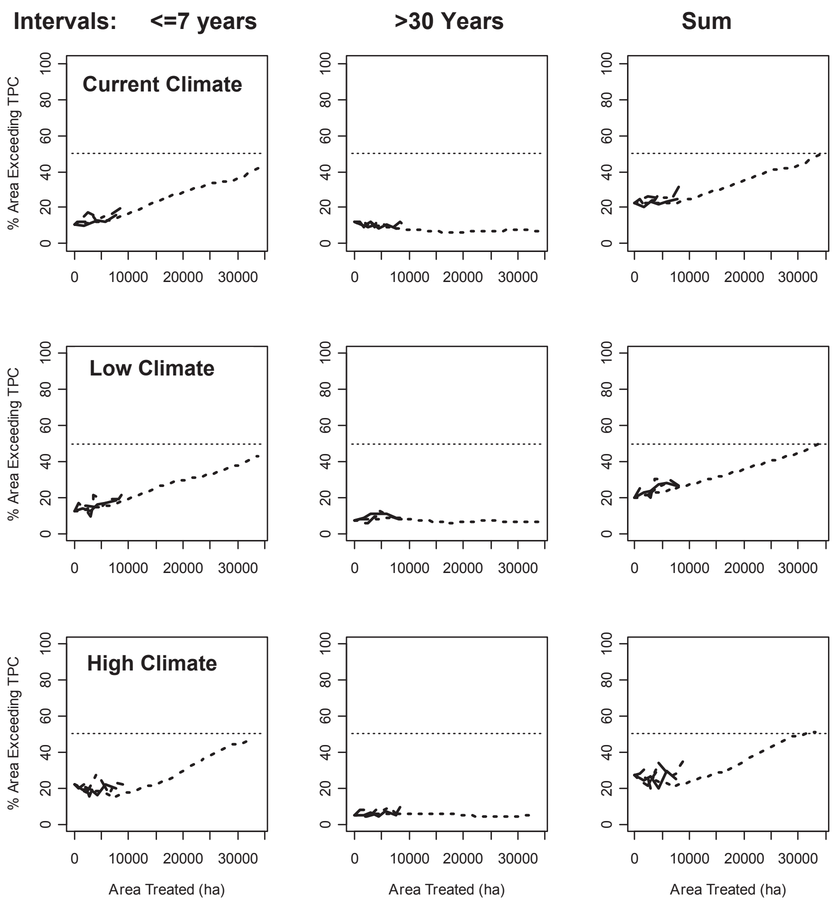
The modelled distribution of adverse IFI under contemporary conditions in BM (i.e. prescribed burning average of circa. 1000 ha per annum or < 1% of the landscape – Fig. 4b, approximating the Constrained spatial strategy) was relatively benign for the predominant Dry Sclerophyll vegetation: i.e. the sum of adverse short and long intervals was around 20% (Fig. 23). Thus current fire regimes are predicted to be within a landscape-level range that is compatible with the persistence of plant functional types sensitive to IFI. As a result, large-scale intervention to reduce extinction risk in this regard does not appear to be necessary. Climate change caused little change to this scenario. While there were some shifts (Fig. 23) in the adverse IFI distribution, these were not large enough to change this conclusion. General reductions in IFI distribution occurred under climate change (authors' unpublished data).

Figure 23 Proportions of BM (Blue Mountains) Dry Sclerophyll Formations affected by adverse inter-fire intervals (short < 7; long > 30; sum - short + long), in relation to prescribed fire strategy, effort and climate change



Localised effects may be significant in BM due to the spatial pattern of probability of occurrence of adverse fire regimes (Appendix 4). Relatively high annual probability of occurrence of fires producing an adverse, short IFI were concentrated in the Dry Sclerophyll vegetation in the eastern half of BM (Appendix 1, authors' unpublished data). These effects are swamped by results for other areas (i.e. low annual probability of adverse IFI) when the case study area is analysed as a whole (Fig. 23). These localised fire regimes were the outcome of the simulated pattern of unplanned fire activity illustrated in Appendix 2. The local effects of these particular fire regime patterns may be important as they may have adverse consequences for species in situ, with wider significance if rare or threatened taxa are affected (Gill et al. 2003). Exploration of these effects via targeted population modelling (see above) is warranted. Climate change (Appendix 4) expanded these potential effects over a significant proportion of Dry Sclerophyll vegetation in the eastern portion of the case study.

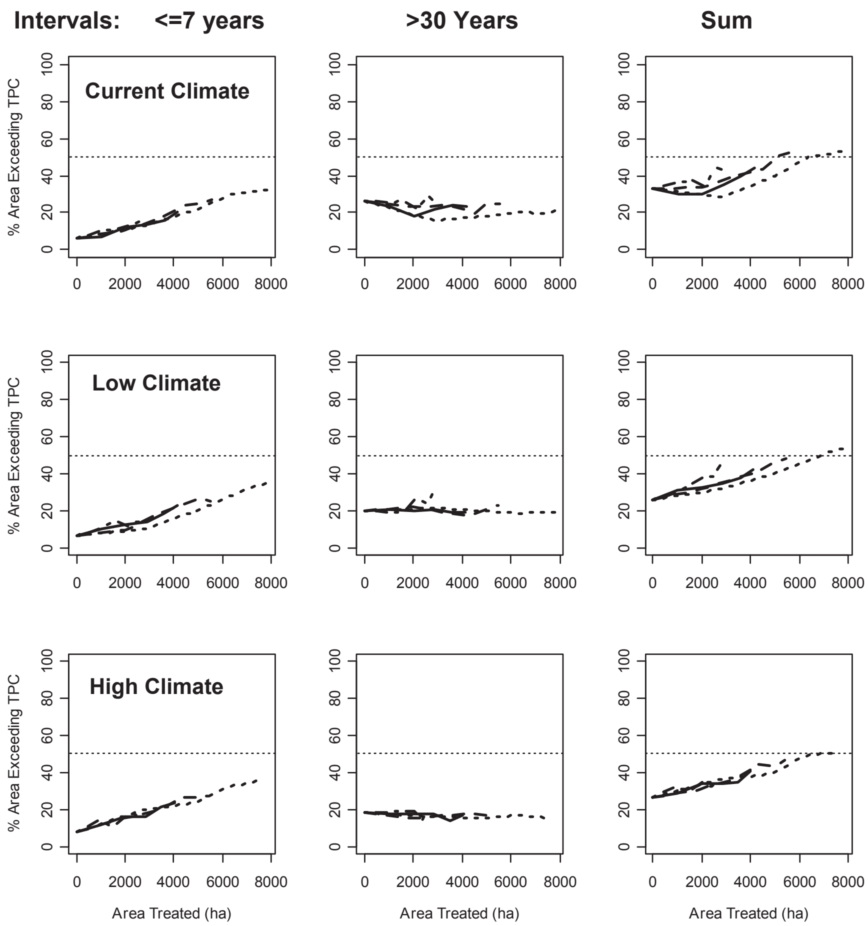
Figure 24 **Proportions of WP (Woronora Plateau) Dry Sclerophyll Formations affected by adverse inter-fire intervals (short < 7; long > 30; sum - short + long), in relation to prescribed fire strategy, effort and climate change**



Similar conclusions apply to the other case studies (Figs 24 & 25), though the total level of adverse IFI in HK was higher at low treatment levels (circa. 30%) than in BM and WP. In HK climate change tended to reduce the average percentage of adverse IFI through a reduction in long adverse intervals (> 30 years), indicating a possible link between landscape fragmentation and IFI. Localised effects of contemporary climate and management were not predicted to be significant within Dry Sclerophyll vegetation in WP and HK (Appendix 4). Instances of simulated high probability of occurrence of fire producing an adverse IFI for Dry Sclerophyll were confined to very small patches (Appendices 1 and 4). Climate change did little to alter this trend (Appendix 4).

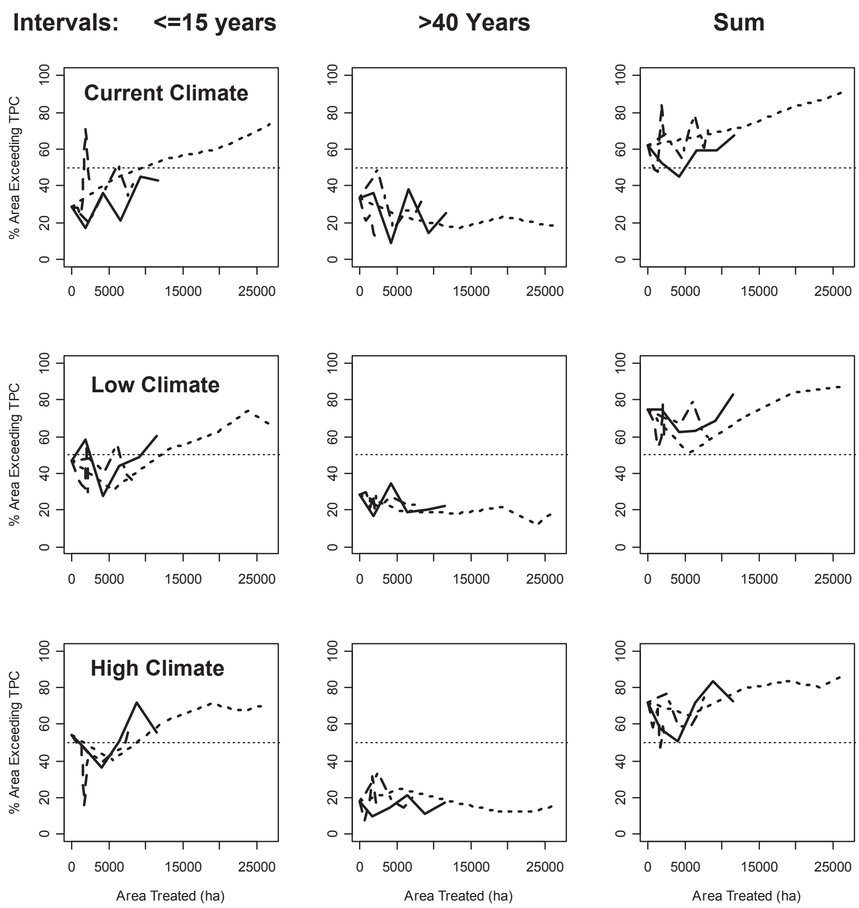
Prescribed burning was found to substantially alter the landscape-level adverse IFI distribution in dry sclerophyll (Figs 23, 24, 25). The adverse, short IFI component increased with treatment level for all strategies under contemporary climate in all case studies (Figs 23, 24, 25). It was only under high levels of Unconstrained treatment that the proportion of adverse, short IFI reached levels where risk of extinction may be relatively high (Figs 23, 24, 25). This trend was most pronounced in WP and least in HK. Long, adverse IFI were relatively unaffected by either strategy or treatment level, and affected about 10% of the area of these plant communities in BM. A similar level was affected in WP but there was a decline in the percentage at high levels of Unconstrained treatment. The long, adverse IFI level was higher in HK (circa. 20%) and also declined at high treatment levels. Thus risk of extinction from lack of fire under contemporary weather and ignition rates is predicted to be low and would be relatively unaffected by choice of prescribed fire strategy in all cases.

Figure 25 Proportions of HK (Hornsby/Ku-ring-gai) Dry Sclerophyll Formations affected by adverse inter-fire intervals (short < 7; long > 30; sum - short + long), in relation to prescribed fire strategy, effort and climate change



Overall (for all case studies – Figs 23, 24, 25) the results indicate that the only prescribed fire strategy likely to result in a high risk of extinction of interval-sensitive plant species (i.e. sum of adverse inter-fire intervals > 50% of landscape occupied by Dry Sclerophyll plant communities) under a contemporary climate was a high level of Unconstrained treatment (circa. 20 % of the landscape treated per annum). This prediction was underpinned by the trend in adverse, short IFI in particular. Climate change did not strongly affect this trend (Figs 23, 24, 25).

Figure 26 **Proportions of BM (Blue Mountains) Wet Sclerophyll Formations affected by adverse inter-fire intervals (short < 15; long > 40; sum - short + long), in relation to prescribed fire strategy, effort and climate change**



Trends in wet sclerophyll forest were more varied among the case studies (Figs 26, 27, 28). Under contemporary levels of prescribed burning (see above) and contemporary weather, levels of total adverse IFIs exceeded 50% of the area containing this vegetation in all case studies. This corresponds to high risk of plant extinction. This effect was due mainly due to a high percentage of both short and long adverse IFI in BM and WP (Figs 26, 27). These effects are evident in the maps of annual probability of occurrence of fires producing an adverse IFI in these case studies (Appendices 1 and 4). By contrast in HK, adverse, long IFI outweighed the occurrence of short, adverse IFIs (Fig. 28, Appendices 1 and 4). Increasing levels of prescribed fire tended to increase the overall level of adverse IFIs in all case studies (Figs 26, 27, 28).This was driven by an increase in short, adverse IFIs, which outweighed a decrease in long, adverse IFIs. Climate change did not strongly alter this trend (Figs 26, 27, 28), though there was some trend toward greater annual probability of occurrence of adverse IFI in Wet Sclerophyll in BM and WP (Appendices 1 and 4).

Figure 27 Proportions of WP (Woronora Plateau) Wet Sclerophyll Formations affected by adverse inter-fire intervals (short < 15; long > 40; sum - short + long), in relation to prescribed fire strategy, effort and climate change

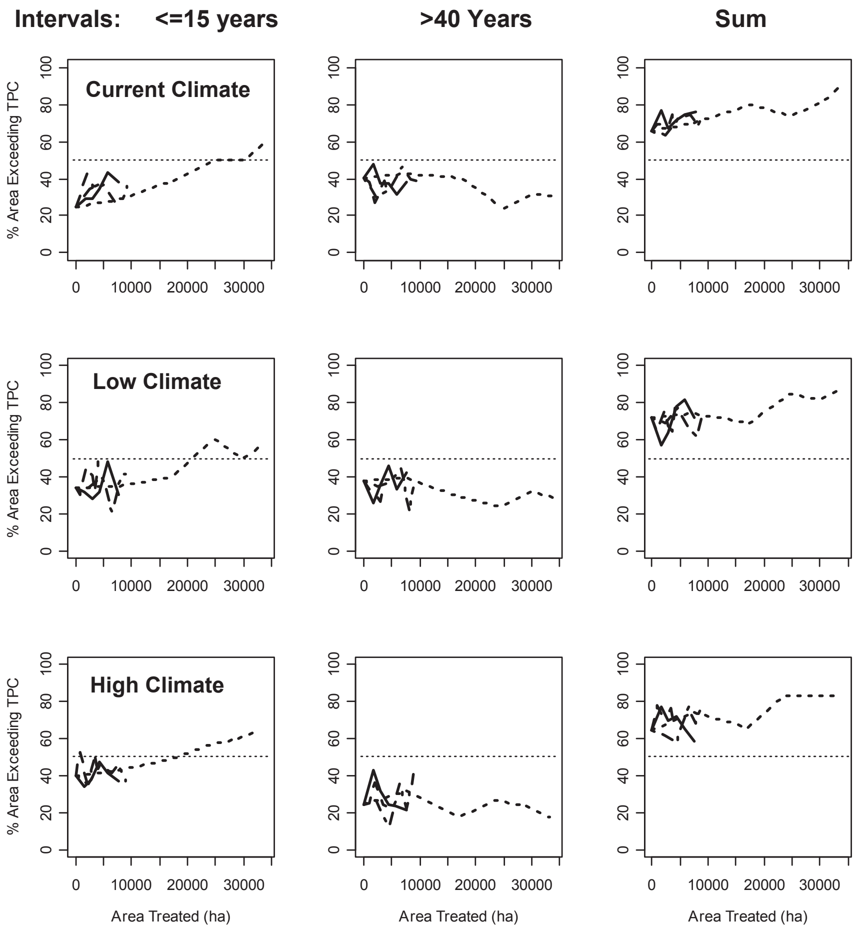
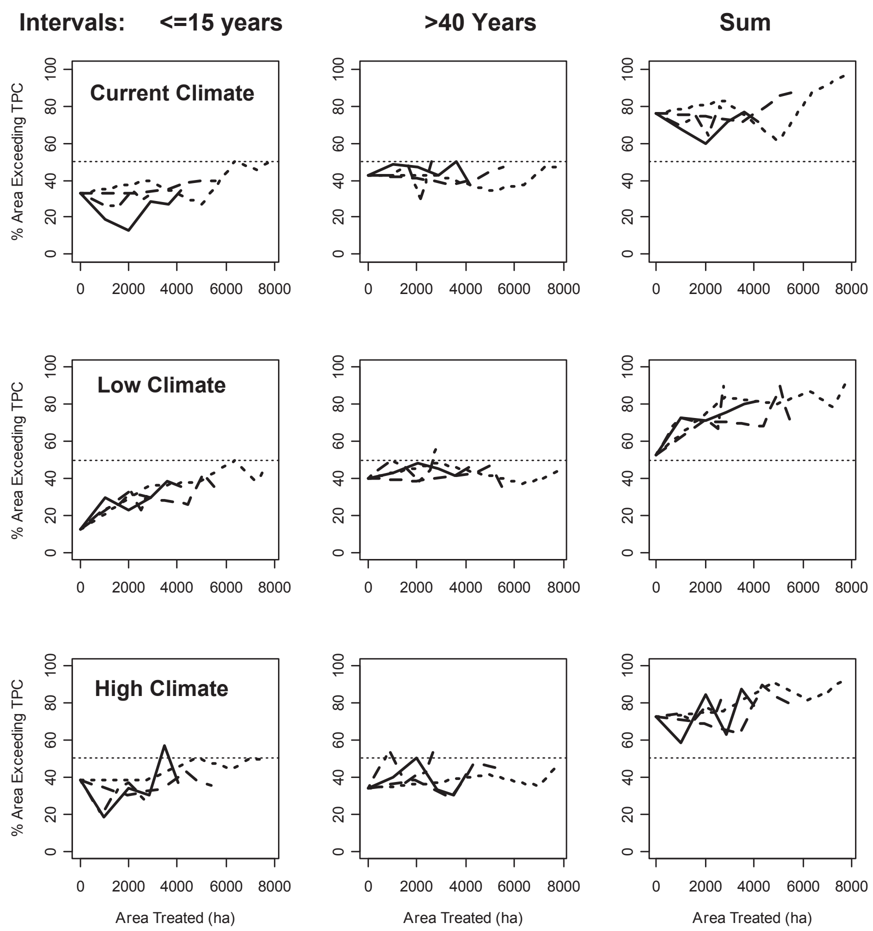


Figure 28 **Proportions of HK (Hornsby/Ku-ring-gai) Wet Sclerophyll Formations affected by adverse inter-fire intervals (short < 15; long > 40; sum - short + long), in relation to prescribed fire strategy, effort and climate change**



Conservation of biodiversity and catchment values potentially sensitive to high fire intensity

Annual, point scale, crown fire probability (i.e. > 10,000 kW m^{-1} intensity) was relatively low (Fig.29). Values were higher in WP than in BM and HK, but relatively unaffected by treatment strategy. There was a negative effect of level of treatment on annual crown fire probability (Fig. 29). Climate change increased probability of occurrence by about 20-25% at low levels of treatment in BM and WP and to a lesser degree at high levels of treatment (Unconstrained strategy only: Fig. 24). Climate change effects were less evident in HK. The high climate change scenario effectively negated the effects of moderate levels of treatment (circa. 3000 ha per annum) under contemporary climate (Fig. 30).

Figure 29
Effect of prescribed burning effort and strategy on the annual probability of unplanned crown fire (i.e. > 10,000 kW/m) at a point in the landscape

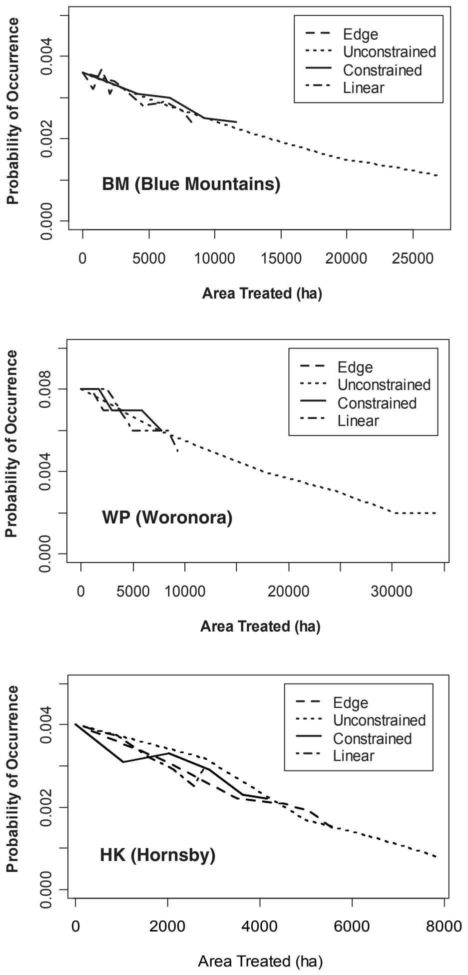
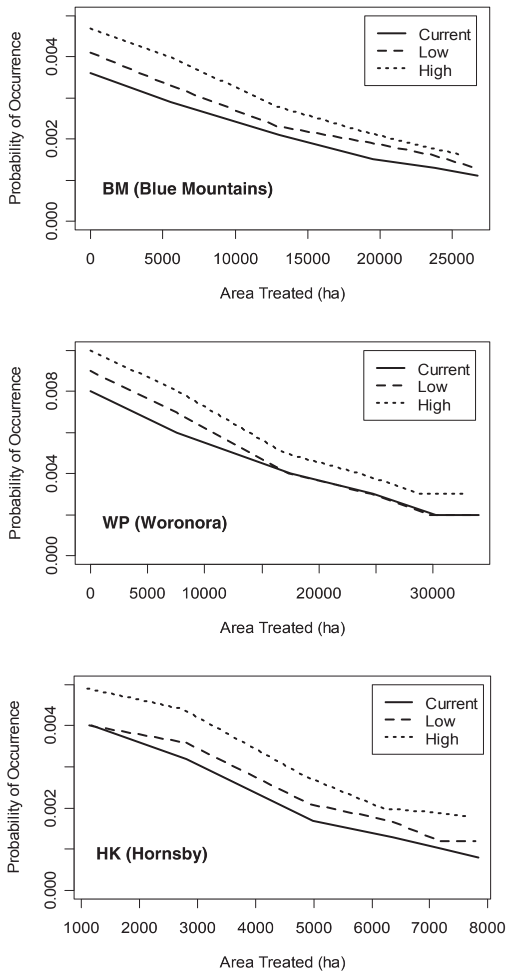


Figure 30
Effect of climate change on the annual probability of unplanned, crown fire (i.e. > 10,000 kW/m) at a point in the landscape, in relation to Unconstrained prescribed burning effort



Rainforest showed a different pattern of response (Fig. 31), in part due to the presumed deleterious effect of a single fire (Table 3, Appendices 1 and 4). Overall, the percentage of rainforest affected by unplanned fire declined with increasing levels of prescribed fire. Low to intermediate levels of prescribed fire (all strategies) resulted in adverse incidence of fire (i.e. > 50 % of area affected) in BM and HK. The corresponding level in WP was lower. High levels of Unconstrained treatment caused a significant drop in area of rainforest burnt (Fig. 31).Climate change markedly increased the area of rainforest burnt across all levels of prescribed burning effort in all case studies (Fig. 32, Appendices 1 and 4).

Figure 31 Effect of prescribed burning effort and strategy on the proportion of rainforest burnt once by unplanned fire

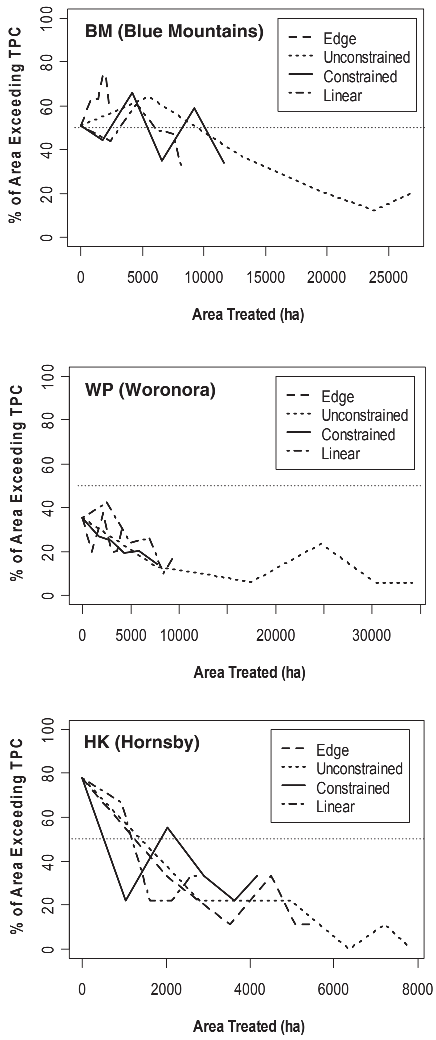
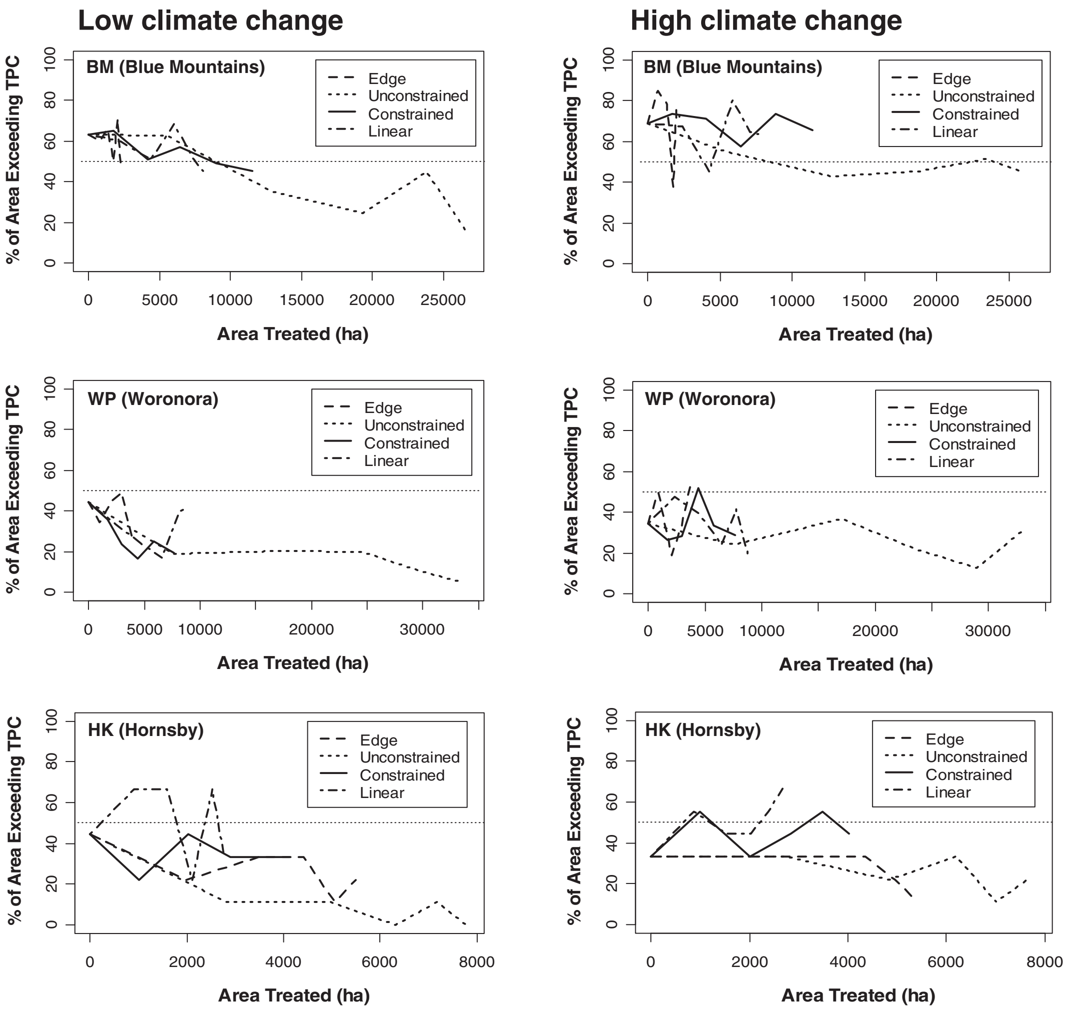


Figure 32 Effect of climate change on the proportion of rainforest burnt once by unplanned fire, in relation to Unconstrained prescribed burning effort



DISCUSSION

Simulated fire regimes were sensitive to prescribed burning strategy, effort and climate change. These effects were variable among case studies though there were many common patterns. In general though, characteristics of unplanned fires were predominantly sensitive to level of prescribed fire effort, rather than differences in spatial strategy. The chief exception was the effect of the Edge strategy on measures relevant to urban risk (i.e. area and intensity of edge blocks burnt – Figs 19, 20). Climate change, particularly the High scenario, generally affected fire regimes more than variations in spatial strategy of prescribed burning.

Fire regimes were generally less sensitive to variations between case studies (e.g. Figs 9 – 12), despite the fragmented nature of HK and the greater diversity of terrain in BM. Adverse IFI distributions for differing vegetation formations indicated effects of fragmentation in HK, with a tendency for greater proportions of adverse long intervals in this case study. This caused some subtle variations in response to management and climate change in HK compared to the other case studies. Nonetheless, the general trend in indicators of risk, in response to management and climate change, was robust across a range of biophysical variation that is representative of the region.

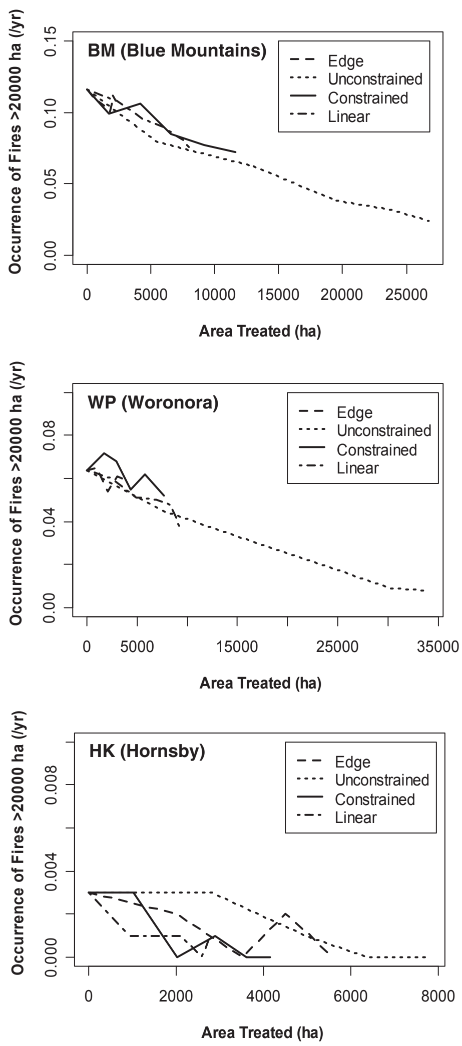
• Consequential Risk Assessment

These results can be used to provide perspectives on a wide range of important values (i.e. biodiversity, catchment integrity and human protection). They provide a quantitative basis for assessment of sensitivity of risk to variations in prescribed fire. Choices concerning prescribed burning effort in particular will be critical because of the overall sensitivity of fire regimes to annual level of treatment. While spatial strategy had much weaker overall effects on fire regimes, some values such as urban protection are predicted to be strongly sensitive to spatial strategy (Figs 19, 20) as found by Bradstock and Gill (2001) using a simpler modelling approach. These choices can be summarised as follows:

- The opportunities for successful fire suppression (a function of fire intensity) within the landscape are predicted to increase with increasing prescribed burning effort in all case studies (Fig. 12);
- The annual probability of large fires (> 1000 ha) was simulated to decrease with increasing prescribed burning effort (Fig. 10);
- Risks (based on indicators) to urban values, catchment values and biodiversity, deemed to be sensitive to fire intensity (SFI), decreased with increasing prescribed burning effort, in all case studies (Figs 19, 20, 29).
- Risks to plant biodiversity, sensitive to IFI, increased with increasing prescribed burning effort, in all case studies.
- Urban edge risk was reduced to a greater degree by the Edge strategy, compared with other strategies, for a given level of annual treatment (Figs 19, 20). High levels of Unconstrained treatment (i.e. > 10% of landscape treated per annum) are required to match the level of risk reduction achieved under much lower levels of Edge treatment (Figs 19, 20).

These results are based on a fundamental trade-off between fire intensity and length of inter-fire interval that emerge from the nexus of fire behaviour models, fuel dynamics, weather and terrain and various assumptions that are captured in the model. Thus, with increasing prescribed fire effort, measures of fire intensity, unplanned fire and IFI decreased. This is because the modelled, net area burned per annum (sum of planned and unplanned fires) increased with increasing prescribed burning effort (inferred from Fig. 9, authors' unpublished data). This trend was amplified by responses of rare but very large fires (> 20,000 ha; simulated occurrence about one in 10 - 20 years in BM and WP under contemporary climate – Fig. 33). While differing levels of prescribed fire caused some reduction in their probability of occurrence (Fig. 33), a simulated reduction in the absolute size of the largest fires was lacking (authors' unpublished data). These effects have been observed in other applications of FIRESCAPE in differing ecosystems (e.g. King et al. 2006, 2008), and as an outcome of other simulation models (e.g. Bradstock et al. 2006).

Figure 33 Effect of prescribed burning effort and strategy on the annual probability of very large unplanned fires (i.e. > 20,000 ha)



The overall result can be summarised on the basis of yield (see Bradstock in press): for each hectare of unplanned fire that is “reduced” (i.e. yielded), more than 3 hectares have to be treated (inferred from Fig. 9). Even allowing, for ‘patchy’ treatment of blocks (see above), the ratio of decline in unplanned area to area burnt by prescribed fire was < 1. Thus prescribed burning increases the overall level of burning in the landscape, with consequent effects on fire regimes outlined above.

Differences in prescribed burning strategy create a spatial weighting of the intensity versus IFI component of the above trade-off. The Edge, Constrained and Linear strategies resulted in shorter IFI and lower average intensity in the target areas, whereas the Unconstrained strategy diluted these effects by spreading treatments more widely. Thus an emphasis on use of prescribed fire at the urban edge (Edge strategy) is predicted to lower risk to property to a greater degree per hectare treated than the Unconstrained strategy (Figs 19, 20). The IFI distribution across the landscape will be relatively unaffected by this approach, but adverse short IFIs may be highly concentrated in target areas such as Asset Protection and Strategic Fire Advantage Zones. This may be acceptable provided that species that are sensitive to such a fire regime are mainly located elsewhere. Such an approach is less effective at mitigating chance of high intensity fires (Fig. 12) and consequent risks to other components of biodiversity and catchment values. Alternative choices will weight the balance of risk in a different way.

Pragmatic considerations are also important: e.g. there is evidence that prescribed burning adjacent to the urban interface is considerably more expensive than in remote areas (Bradstock et al. 1998, Bradstock and Gill 2001). Thus for a given amount of expenditure it may be feasible to treat more area via non-Edge strategies. The ratio of cost per hectare of treatment among alternative strategies will be crucial: e.g. in BM and WP, about three times the amount of Unconstrained treatment is required to match the effect of Edge treatment in terms of reduction of area of urban edge burnt by unplanned fires (Fig. 19). The disparity between strategies was more marked in terms of unplanned fire intensity at the urban edge (Fig. 20). The simulations indicate that Unconstrained treatments of > 15 % of the landscape per annum may be required to match the effect of the highest level of Edge treatment in BM and WP in this regard.

This example illustrates the way risk is weighted by choice of strategy and other factors. Is there an optimum approach (prescribed burning strategy and level of treatment) that acceptably mitigates risk across all these values? If so, how robust will such a strategy be, given the predicted responses to climate change? The modelling results provide a basis for answering these questions, which can be summarised by the following results and interpretation.

1. Zero prescribed fire did not result in high levels of risk to IFI sensitive plant biodiversity but maximised indicators of risk to species and communities deemed to be sensitive to fire intensity (SFI). It is not known, however, if these fire regimes correspond to high extinction probability of SFI biodiversity in the case study landscapes. Formal relationships of this kind are unavailable at present.
2. Zero prescribed fire maximised indicators of risk to urban, SFI biodiversity and catchment values.
3. Current levels of burning (i.e. circa 1 % of landscape treated per annum – fig. 4b) resulted in a relatively small degree of risk reduction to urban, catchment and SFI biodiversity indicators (circa. 5 % change from level under zero prescribed fire).

4. Current levels of burning did not result in high levels of risk to IFI sensitive biodiversity when assessed at a whole of case study scales. IFI in specific landscapes, however, could pose high levels of risk to IFI sensitive species in Dry Sclerophyll (within BM) and Wet Sclerophyll (WP). Such effects may need to be scrutinised through targeted modelling, but are not predicted to be ameliorated by higher levels of prescribed burning.
5. Increments such as doubling or tripling the current level of prescribed burning (e.g. treatment of about 3 % of the landscape per annum) resulted in a further reduction in risk (circa.10 – 15 % below the level under zero prescribed fire) to indicators of urban, catchment and SFI biodiversity values.
6. Treatment of up to 10 % of the landscape per annum yielded a 25 – 45 % reduction in risk, compared to zero prescribed fire, to indicators of urban, catchment and SFI biodiversity values without raising levels of risk to IFI sensitive biodiversity to a critical level.
7. Unconstrained treatment of 10 – 20 % of the landscape per annum yielded a 50 – 70 % reduction in risk, compared to zero prescribed fire, to indicators of urban, catchment and SFI biodiversity values, but concurrently increased risk to IFI sensitive biodiversity to a critical level (i.e. high extinction probability).
8. Risk to key values cannot be eliminated by any level of plausible or hypothetical treatment. An appreciable probability of large and high intensity fire will remain even at relatively high levels of treatment (i.e. residual risk – Gill 2005).
9. Operational and economic constraints will have a major bearing on choices. A doubling of prescribed burning level would plausibly require a major increase (i.e. doubling) in funding and resources, in return for a small reduction in urban interface risk. Further increases in prescribed burning beyond this would involve commensurate financial increases.

The results provide a basis for a fuller evaluation of return on investment, by quantifying components of consequential risk to key values. This has important consequences for debate about the utility and effects of prescribed burning. Such an analysis is beyond the scope of this report but is probably critical for deciding future directions in fire management. Economically feasible increases in effort (e.g. a doubling of expenditure and consequent area treated), irrespective of spatial strategy, are predicted to yield small reductions in risk to urban or catchment values. Such increases are, however, also unlikely to significantly increase or decrease risk to biodiversity. Both conclusions, based on modelled results, run contrary to perceptions and arguments that are frequently aired in debate about fire management. Paradoxically, the levels of prescribed burning needed to fulfil these perceptions are likely to require levels of funding (e.g. > ten times current expenditure/ resources) that are improbable.

The resultant conclusion is that small changes in risk can be maximised by refinement of strategy – crudely represented here by the major alternatives. The results indicated that the Edge and Constrained strategies (emphasis on treatment of urban interface blocks) reduced exposure of the urban interface to a greater degree (per ha treated) than the other strategies in two of the case studies (Fig. 19). Given that contemporary management represents a variant of the constrained strategy, further development of this approach would appear to be warranted. Consideration of alternatives such as expansion of areas of urban interface that are permanently modified (i.e. creation of 'defensible space' – Gill 2005) may be warranted, if these prove more cost effective in the light of economic analyses. This would provide scope for allocation of resources for more strategic landscape burning (e.g. upper slopes and ridges which are readily accessible for suppression) which may yield a higher return on expenditure (i.e. amount of risk mitigated per dollar spent).

• Climate change

Climate change effects on risk are predicted to be appreciable and the level of change over contemporary weather and management can be summarised as follows.

1. Indicators of risks to urban, catchment, SFI and IFI sensitive biodiversity values could substantially increase, principally through an increase in mean area of unplanned fires, large fire probability, average fire intensity and average area of urban edge burnt by unplanned fires. The level of increase in risk, based on these measures, is generally 5 - 20 % under the High scenario and < 10 % under the Low scenario.
2. The change in risk to IFI sensitive biodiversity resulting from changes to the fire regime is unlikely to have a major effect on extinction probability at a whole of case study scale. More localised, potential adverse effects on IFI sensitive biodiversity were simulated in BM and WP. These may be significant and require further research (see above).
3. Effects on extinction probability of SFI biodiversity are unknown and require further research (see above).
4. The level of change in risk due to climate change is slightly reduced at very high levels of Unconstrained prescribed burning (> 10 % of landscape treated per annum).

These consequences can be further assessed by estimating the degree of change in management effort (prescribed fire) needed to hold the contemporary level of risk constant. The modelled results suggest that the required level of change in management effort is large: e.g. an increase of > 5 % of the landscape treated (High climate change scenario). An increase in management effort of this kind would require a large increment in resources and funding (see above), possibly beyond realistic limits. A more feasible level of increase in prescribed fire (e.g. up to twice current levels) will only partially offset effects of the High climate change scenario but may substantially mitigate the level of increased risk under the Low scenario.

The general conclusion is that risks to urban, catchment and biodiversity values are likely to increase under the scenarios of 2050 climate change within the range of feasible prescribed burning options. The likely level of increase in risk, while appreciable, should not be regarded as catastrophic in relation to urban and catchment values. Consequences for SFI biodiversity remain unknown but such changes in risk are predicted to be insufficient to pose a general threat to IFI sensitive biodiversity, though exceptions may occur at a local scale in BM and wider linkages between effects on plant diversity, habitat attributes and animal responses need exploration. The quantification of change in risk provides an opportunity for analysis of economic consequences. This may provide scope for estimation of the cost benefits of other risk mitigation measures for urban protection, such as building and garden design, town planning and vegetation modification, aimed at enhancing 'defensible space'. The modelling indicates that the problem of elevated risk under climate change is unlikely to be solved by prescribed burning alone. A wider range of solutions, incorporating prescribed fire, will therefore be required.

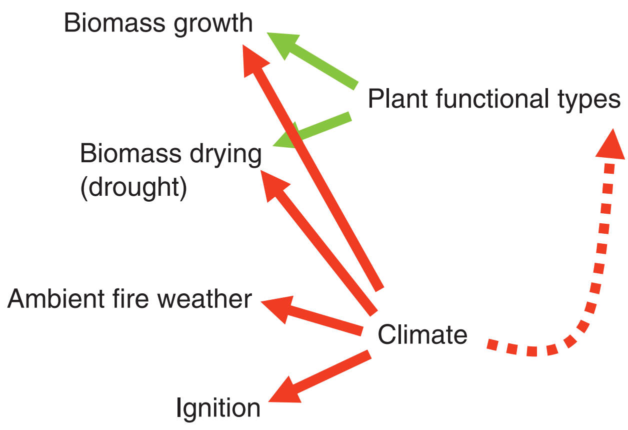
• Model limitations and the scope of potential climate change effects

The modelling of potential climate change effects on fire regimes and risk used in this study are limited and conservative. As noted, the scenarios of 2050 fire weather provided by Hennessy et al. (2005) involve adjustments of daily maximum values of weather variables based on a contemporary time series of days, but do not attempt to change the sequence of days and associated length of periods without rain. The method of estimation of fuel moisture, and thus availability to burn, used here (Mount SDI – see above), is empirically based and may not be appropriate under all possible climate projections where components of weather may change in different ways. For example, the scenarios indicate changes in temperature and precipitation that lead to dryer fuels, while trends in humidity and wind speed indicate possible increases in fuel moisture.

Ultimately, this study dealt only with one potential vector of change to fire activity (i.e. 2050 fire weather) and its effect on fire regimes. The results presented here are therefore similar in scope to previous attempts to simulate changes in fire regimes in Australia (Cary 2002), though correlative approaches based on statistical models have been employed in other localities (e.g. Pausas 2004 – Spain; Flannigan et al. 2005 – Canada). Such approaches are lacking in Australia to date.

Other major vectors of change include changes to ignition rates and changes to fuel that may result from climate change (Fig. 34). Changes to the season and frequency of lightning under climate change could alter ignition rates, but there are few data to indicate possible trends. Expansion of urban development may also increase anthropogenic ignition rates (Keeley and Fotheringham 2001). Fuel availability may be directly affected by changes to plant productivity under altered climate and atmospheric CO_2. It may also be affected by changes in plant community composition. Such effects remain largely unexplored in detail but some generalisations are possible. The consequences of a predicted trend toward a drier climate (CSIRO 2007) in many parts of southern Australia can be inferred from comparisons of fuel accumulation curves from forested areas under different contemporary rainfall regimes (e.g. Raison et al. 1983; Cary and Golding 2002). Both rate of accumulation and quasi-equilibrium fuel load tend to be positively correlated with rainfall in relevant temperate eucalypt forest types. Thus a trend toward a drier climate in the Sydney region could result in lower fuel levels. Increased atmospheric carbon dioxide concentrations could have the opposite effect by enhancing growth of woody plants and reducing the palatability of leaves (e.g. Gleadow et al. 1998, Wang 2007).

Figure 34 A summary of pathways of climate change effects on fire regimes.

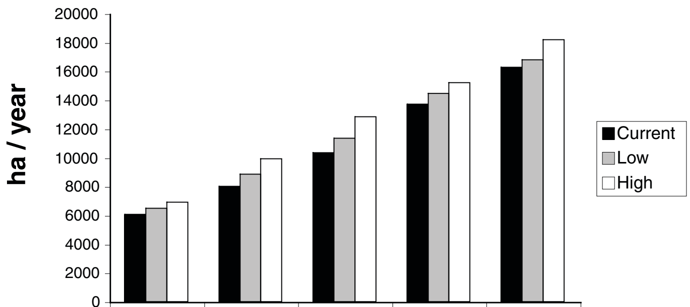


Red (darker) arrows indicate direct effects of climate on the fundamental drivers of fire. Indirect effects on vegetation composition/structure and resilience are indicated by the dashed line.

Direct effects of plant functional types on primary drivers of fire are indicated by green arrows.

The possible implications of these changes can be inferred from sensitivity analyses conducted for BM (Fig. 35) using the model. These indicate that changes to fuel accumulation have appreciable effects on mean annual area burned. For example, a 10 % decline in fuel accumulation under current climate was simulated to reduce mean annual area burned by about 20%. A 20% decline in fuel decreased annual area burned by about 40%. Commensurate changes occurred under higher levels of fuel and future climate scenarios. This analysis can be used to indicate possible interactive effects of changes to fire weather and fuels under climate change. For example, a 10% reduction in fuel under the High climate change scenario yielded a mean annual area burned that was similar to that simulated for contemporary climate and fuel (Fig. 35). In this case the increase in area burned due to 2050 fire weather was offset by the effect of a reduction in fuel. This scenario could represent the combined outcome of more severe fire weather and increased dryness. A more severe effect of a drying climate on fuel (i.e. 20% reduction in fuel) would result in an overall drop in mean annual area burned (Low or High 2050 scenario).

Figure 35 Effect of changes in fuel accumulation (+ 10 and 20%) and fire weather (Low and High 2050 scenarios) on mean annual area burned in BM.



	-20%	-10%	Obs	10%	20%
Current	6052	8003	10341	13701	16280
Low	6473	8865	11353	14447	16793
High	6898	9930	12839	15187	18192

These scenarios do not account for effects of elevated CO_2 on plant growth. These potential effects add further complexity. If increased plant growth and decreased litter composition result then these effects could offset any decline in fuel caused by a drying climate. Such effects would then tend to reinforce the effect of more severe fire weather, causing an increase mean annual area burned. Thus a 10% increase in fuel combined with the High climate change scenario yielded a 50% increase in area burned. Such a scenario would cause a major increase in risk to most key management values, relative to those discussed above. The effect of elevated carbon dioxide could therefore be pivotal in determining fire regime responses to climate change, but at this stage remains poorly understood within local, forested environments (Hughes 2003, Steffen and Canadell 2005). We conclude that there is potential for fire activity to either increase or decrease in the Sydney region as a consequence of climate change. The relative changes in ignitions, fuels and weather are sufficiently uncertain to preclude a definitive resolution of their net effects at this stage. Given that human ignition sources may increase as a function of human population pressure, the balance among these drivers may favour a net increase in area burned, with concomitant shifts in risk.

• Further research

This report summarises an extensive exploration of management and climate change effects on consequential risks arising from bushfires. A simulation modelling exercise of this scope is unprecedented in Australia and elsewhere, notwithstanding earlier attempts at simulating consequences of climate change (Cary 2002). It is the first time FIRESCAPE has been applied to landscapes fragmented by large areas of urban development. Nonetheless improvements to the modelling approach would be worthwhile and beneficial. Among the possibilities, further development of the model to simulate initial attack suppression would be of value. This would allow more comprehensive exploration of the interaction between fuel modification and suppression – a capability that would significantly enhance the potential for understanding how investment in prevention and suppression may be adjusted to produce an optimal outcome (i.e. greatest reduction in risk for a given level of overall expenditure). Improvements to fundamental knowledge of fire behaviour, fuels and spatial variation in weather conditions across landscapes would be of great value to future modelling.

Further exploration of ignition scenarios would also be of immediate benefit. Given the capability to represent lightning and anthropogenic ignitions in the modelling approach, an obvious priority is the need to explore future scenarios of change that may result from climate effects on lightning and demographic/economic effects of people. Further research on the spatial pattern of lightning ignitions would be of benefit. Clear cut linkages between population density and fire incidence around other major urban centres such as southern California (Keeley and Fotheringham 2001) provide impetus for this. Previous studies using models from around the world (Cary et al. in press) have suggested that after weather, ignition rates are the next most important contributor to area burnt by unplanned fire.

Indicators of three key values – urban, biodiversity and catchment – were explored above. Some of these indicators (e.g. probability of high intensity fires) require further research to quantify their linkages to consequential risk, as outlined above. Fire management is also likely to become more complex because of the necessity to encompass other values. Epidemiological evidence suggests that bushfire smoke may have negative effects on human health via air quality (e.g. Johnson et al. 2002, 2006). Carbon sequestration and emissions of greenhouse gases from bushfires (Cook et al. 2005, Mackey et al. 2008) are emerging issues that will heighten the challenge for bushfire managers. The modelling platform and simulation results presented here provide an immediate basis for exploration of these additional key values. Finally, as emphasised above, linkage of the simulation approach to economic models may be of great utility, as a means of exploring the cost-benefits of alternative approaches to resource allocation and investment in fire management. The key is the quantification of consequential risk that emerges from a process-based understanding of the response of fire regimes to their fundamental drivers.

CONCLUSIONS

Modelled fire regimes in the Sydney region were strongly sensitive to level of prescribed burning. Climate change also affected fire regimes, while differences in the spatial strategy of prescribed burning had smaller, but important effects. These findings have major consequences for bushfire risk management in the Sydney region.

The level of prescribed burning had an overarching effect on bushfire risk to urban assets. The level of risk to urban and other assets that is mitigated by current treatment levels is small (circa. 5% of the total level sustained under zero prescribed burning). For urban values this constitutes a reduction in only one component of overall consequential risk (i.e. the contribution made by condition of landscape versus that made by condition of the built environment – each component accounts for half of the overall risk – see above). For values in situ (e.g. biodiversity sensitive fire intensity and catchment stability) there are also other factors that need to be considered (some unknown) that have to be taken into account in an overall estimation of risk.

An increase in prescribed burning, within a range that is probably feasible within constraints of resources, budgets and opportunities (e.g. a 50 – 100 % increase in effort) is predicted to further mitigate the landscape contribution to overall risk to urban and other assets by about another 5%. Such an increase is unlikely to significantly change the level of risk to biodiversity that is sensitive to the length of inter-fire intervals. Localised effects, particularly on threatened species with restricted distributions, may however require closer scrutiny. While much larger reductions in risk to urban and other assets could be achieved, these would involve a level of investment in prescribed burning an order of magnitude beyond current levels. An elevation of risk to important elements of biodiversity is predicted to ensue. No level of hypothetical treatment is predicted to totally eliminate risk (i.e. there is residual risk - Gill 2005).

Climate change is predicted to elevate risk to human and other assets through a fire- weather-driven increase in area burned. The net effect of the High 2050 climate change scenario was to elevate risk beyond a level that could be mitigated via a 50 – 100 % increase in prescribed burning effort (i.e. a level that is likely to financially feasible). The level of change in fire regimes under this scenario of climate change was not predicted to significantly elevate risk to fire-interval sensitive biodiversity, with the exception of some areas within BM and WP case studies. The Low 2050 climate change scenario was predicted to change risk to a lesser degree, within a range that may be offset by a feasible increase in prescribed burning effort.

These scenarios do not account for other vectors of change to the fire regime that may result from climate change. Increased dryness may lower fuel availability to a degree that reduces area burned and consequent risk. Such effects could offset the positive effects of changed fire weather on area burned summarised above. Possible effects of carbon dioxide fertilisation under higher, future atmospheric concentrations of this greenhouse gas could counteract this effect. Considerable uncertainty surrounds this latter, important process. The net result is that predictions about changed fire regimes must be treated as being highly uncertain until a more complete resolution of the relative effects of the various drivers has been achieved. While much evidence points to an increase in fire activity in the region, other factors may either exacerbate or counteract this trend.

Further experimentation and modelling will contribute to an improvement in our understanding of bushfire risks and their sensitivity to both management and global change. There is now a pressing need to use the modelling approaches developed here in conjunction with economic analyses to better understand the cost-benefits and trade-offs involved in decision-making. This could yield immediate benefits in terms of optimum investment for mitigation of bushfire risks. The impetus to consider risks to other values such as human health (via bushfire smoke), plus effects of fires on carbon sequestration and greenhouse gas emissions, amplifies this need.

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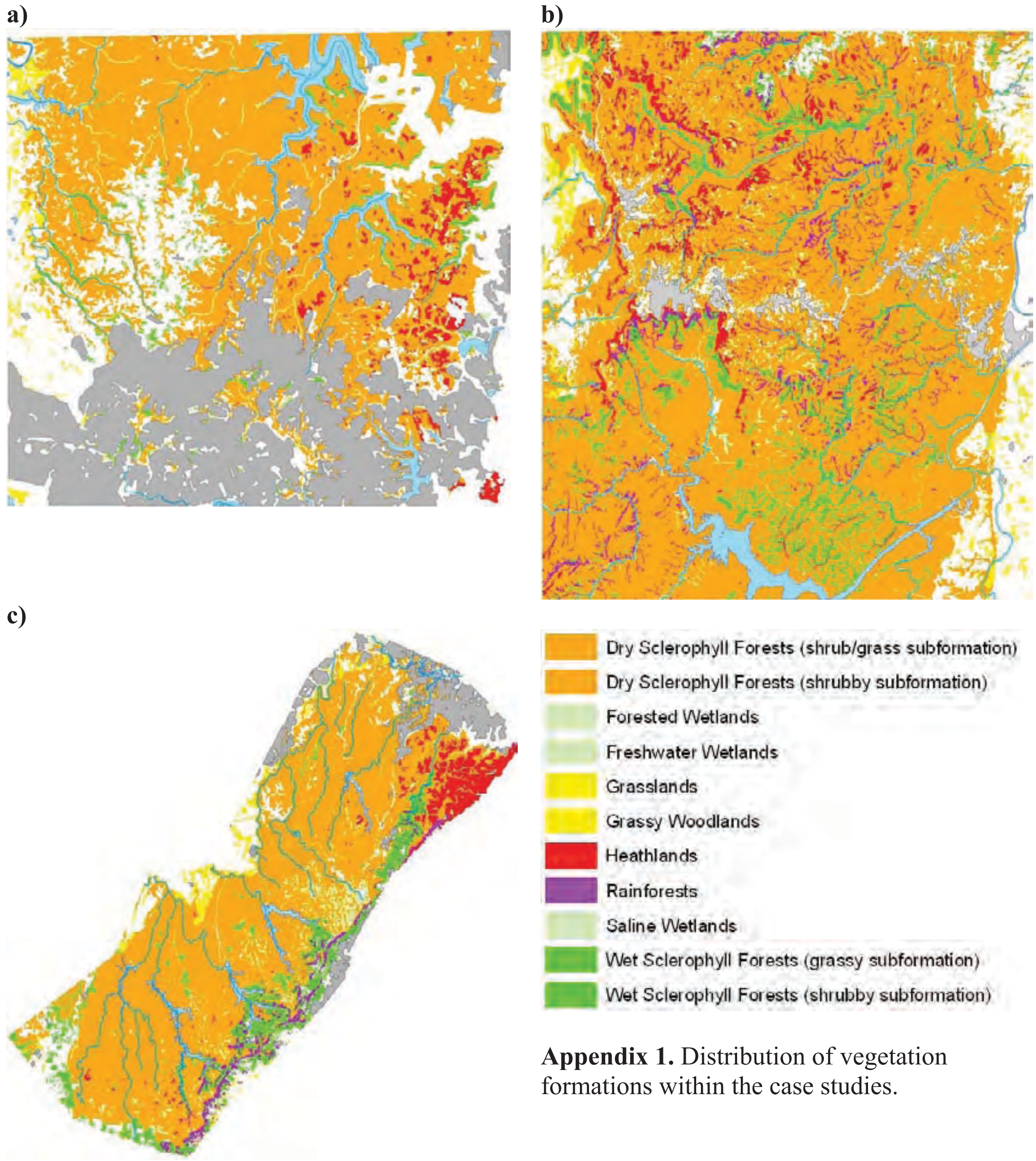
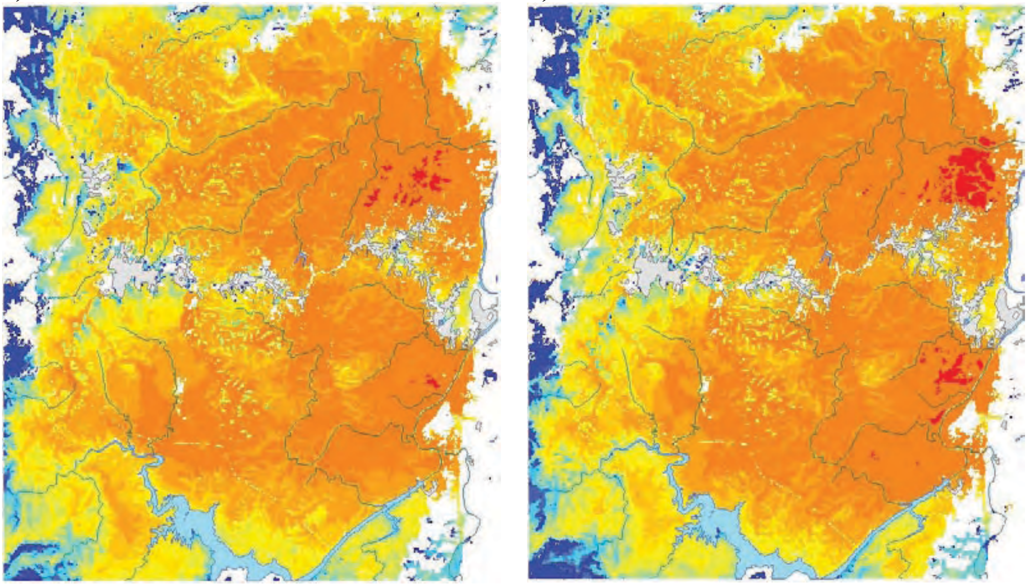
APPENDICES



Appendix 1. Distribution of vegetation formations within the case studies.

a) b)



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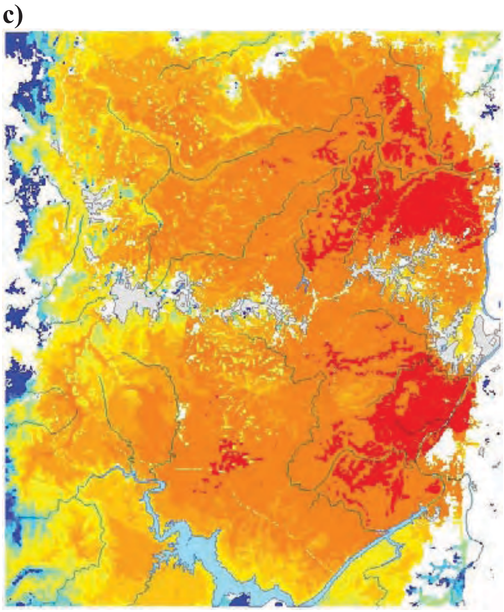
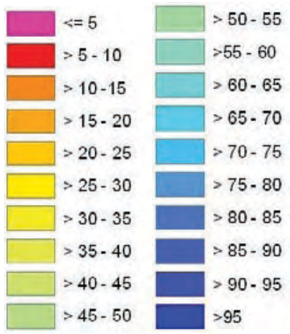
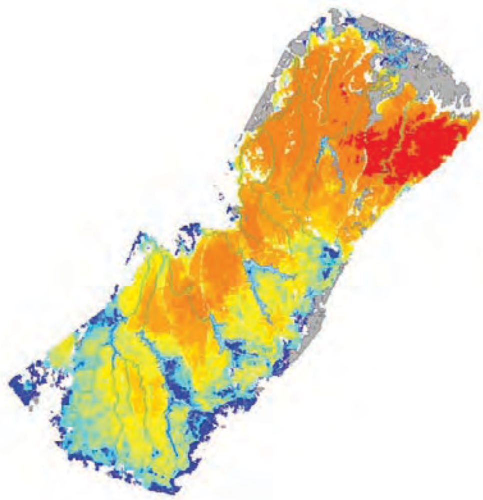


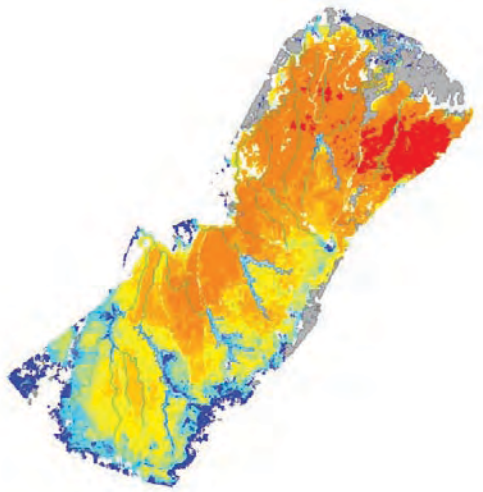


Appendix 2a. Spatial distribution of inter-fire intervals of unplanned fires for current a) and two climate projections (Low b) and High c)) in the Blue Mountains (BM) case study. Data calculated over 1300 years of simulation. Built-up areas are shown in grey, rivers and water bodies in light blue.

a)



b)



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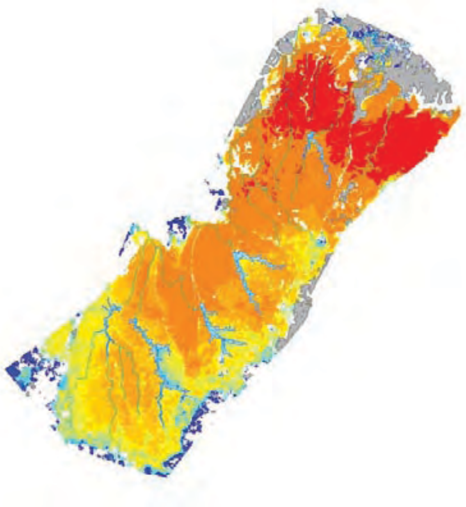
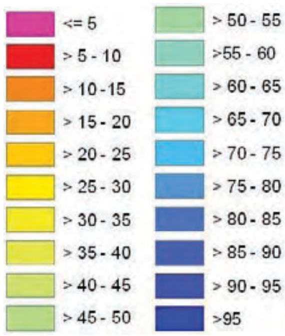
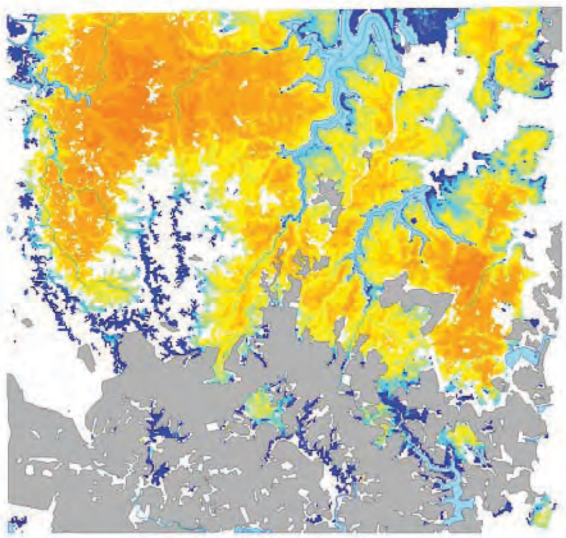


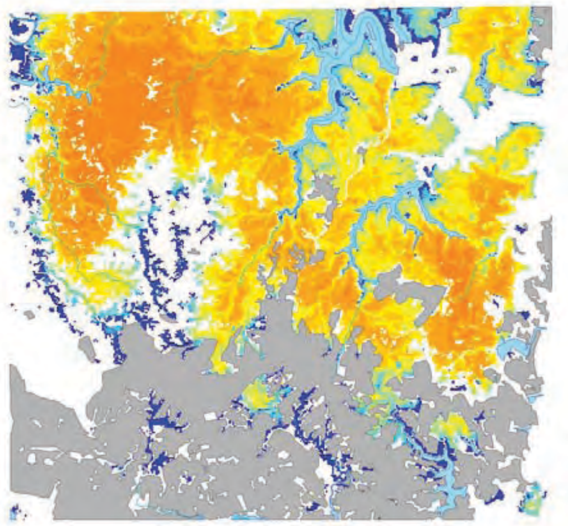


Appendix 2b. Spatial distribution of inter-fire intervals of unplanned fires for current a) and two climate projections (Low b) and High c)) in the Woronora Plateau (WP) case study. Data calculated over 1300 years of simulation. Built-up areas are shown in grey, rivers and water bodies in light blue.

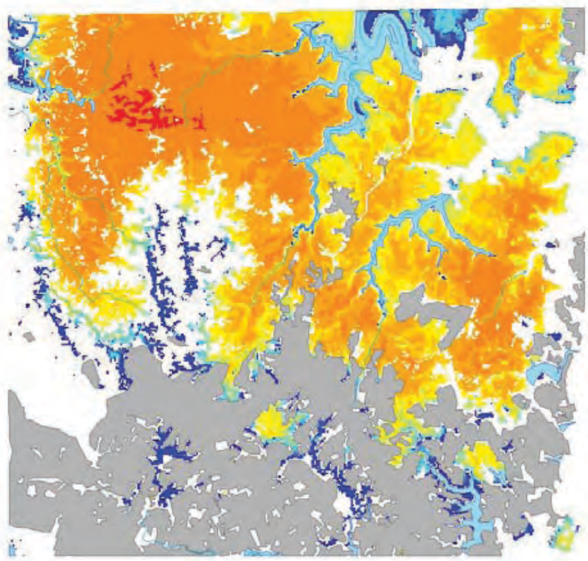
a)

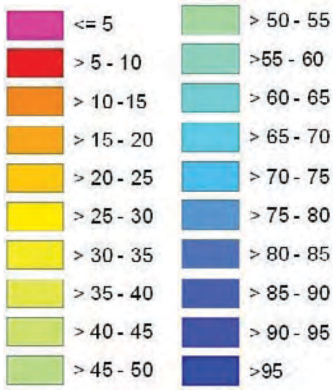


b)

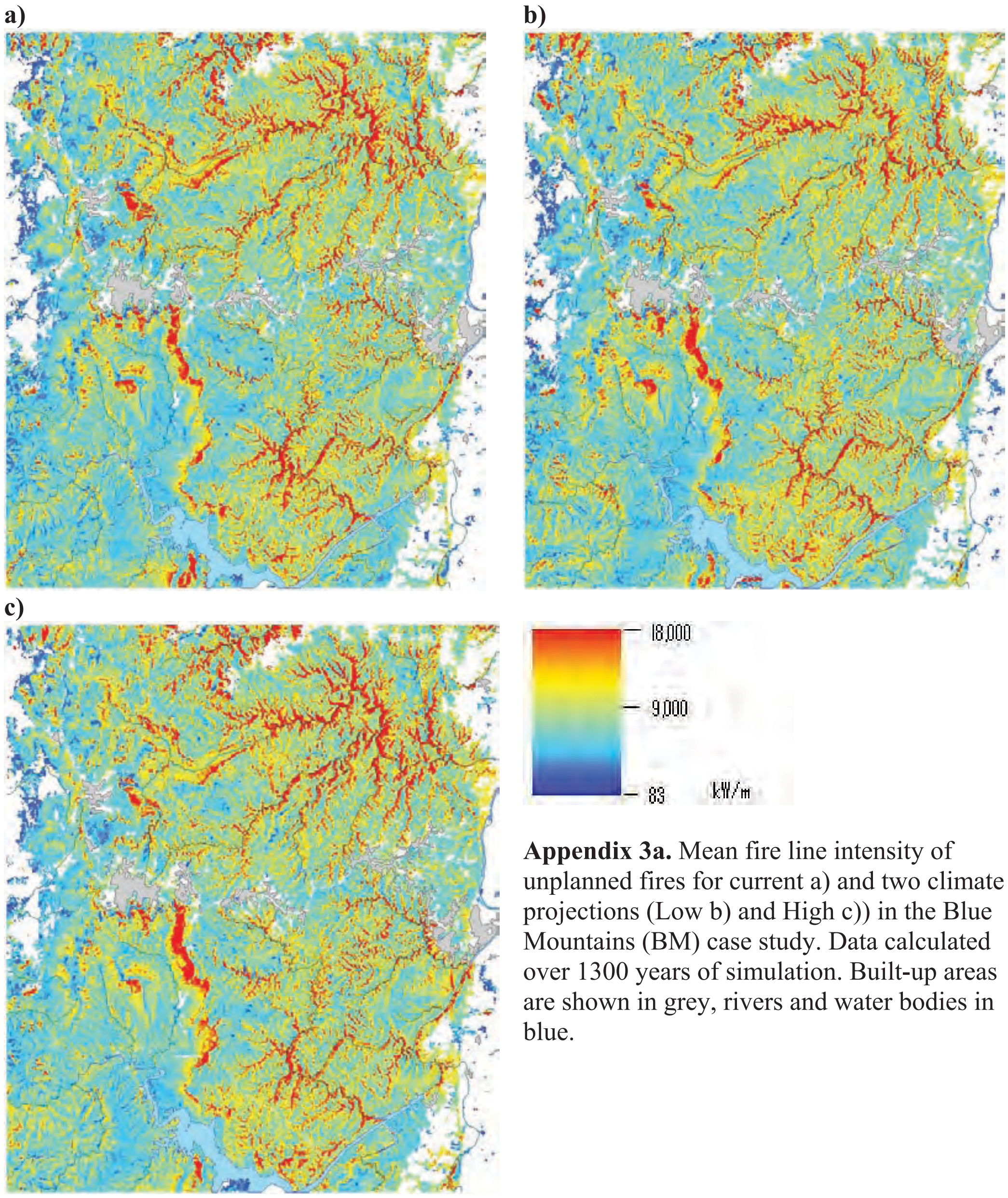


c)



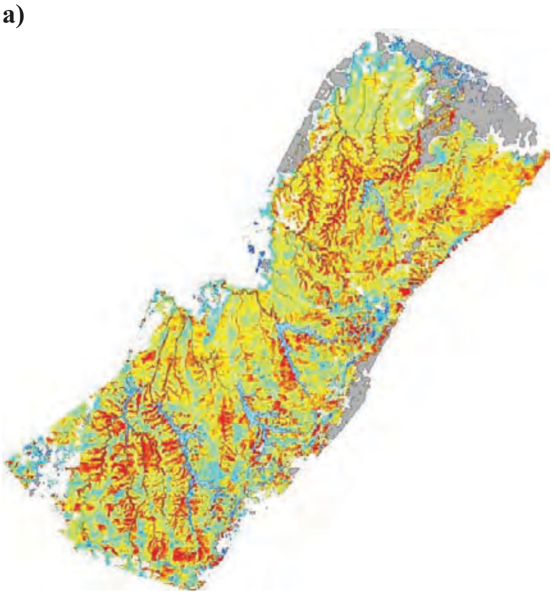


Appendix 2c. Spatial distribution of inter-fire intervals of unplanned fires for current a) and two climate projections (Low b) and High c)) in the Hornsby (HK) case study. Data calculated over 1300 years of simulation. Built-up areas are shown in grey, rivers and water bodies in light blue.

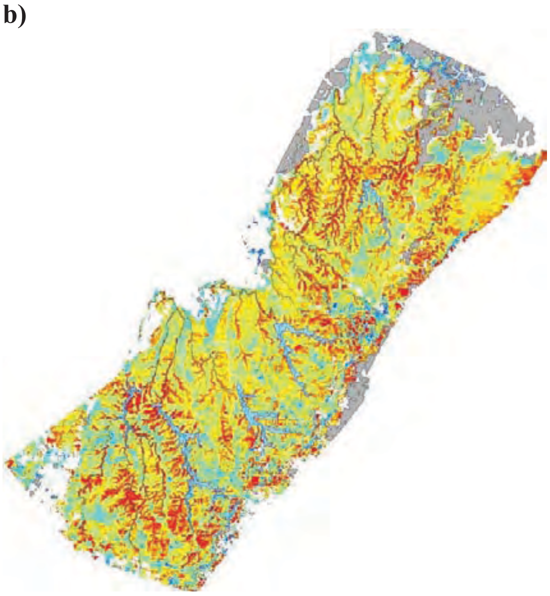


Appendix 3a. Mean fire line intensity of unplanned fires for current a) and two climate projections (Low b) and High c)) in the Blue Mountains (BM) case study. Data calculated over 1300 years of simulation. Built-up areas are shown in grey, rivers and water bodies in blue.

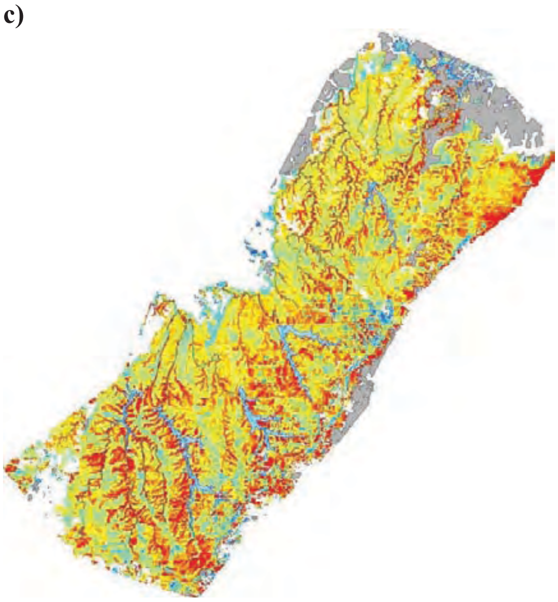
a)

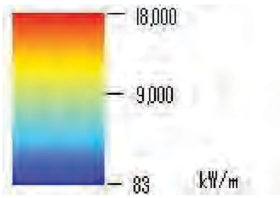


b)

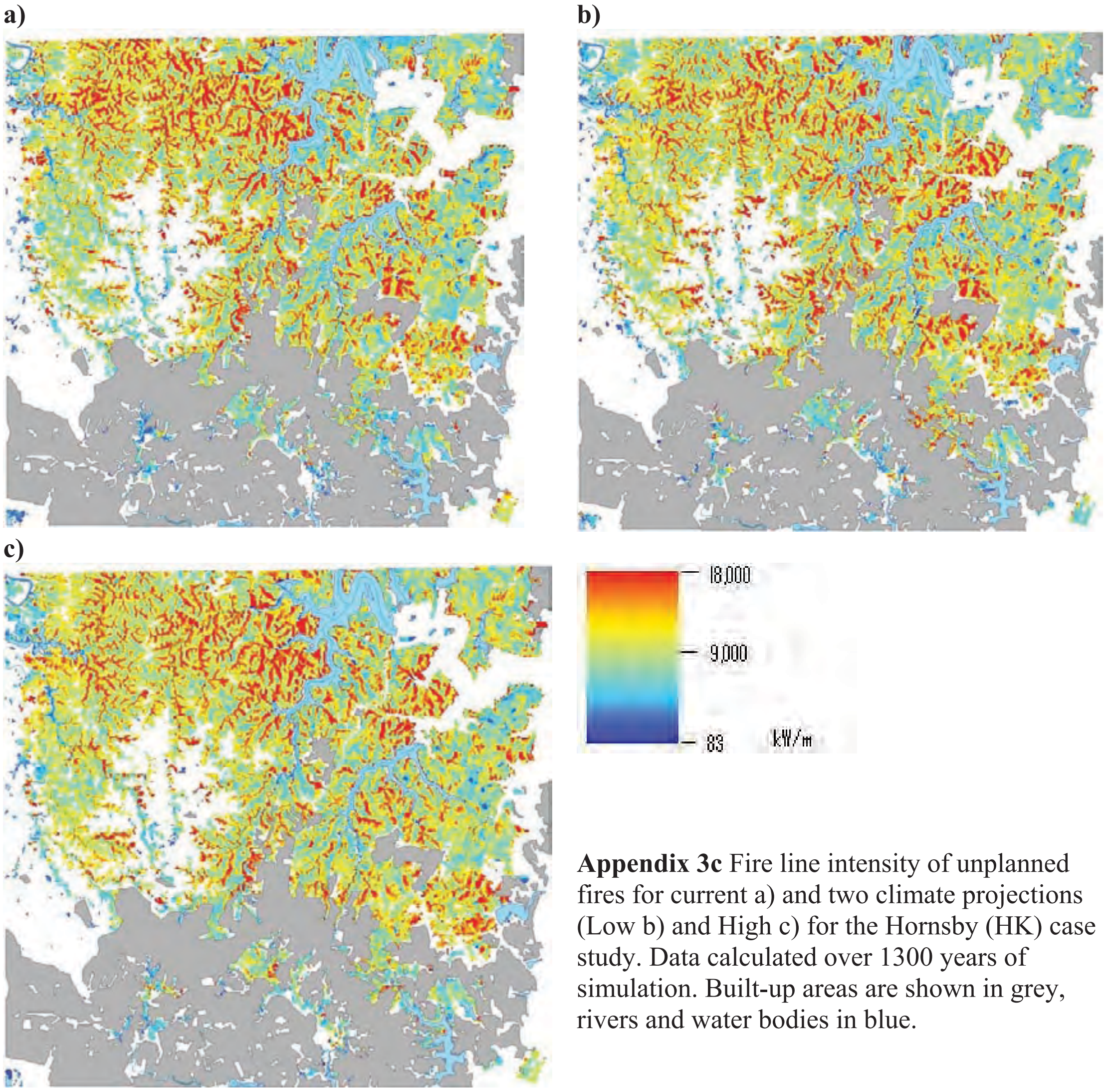


c)

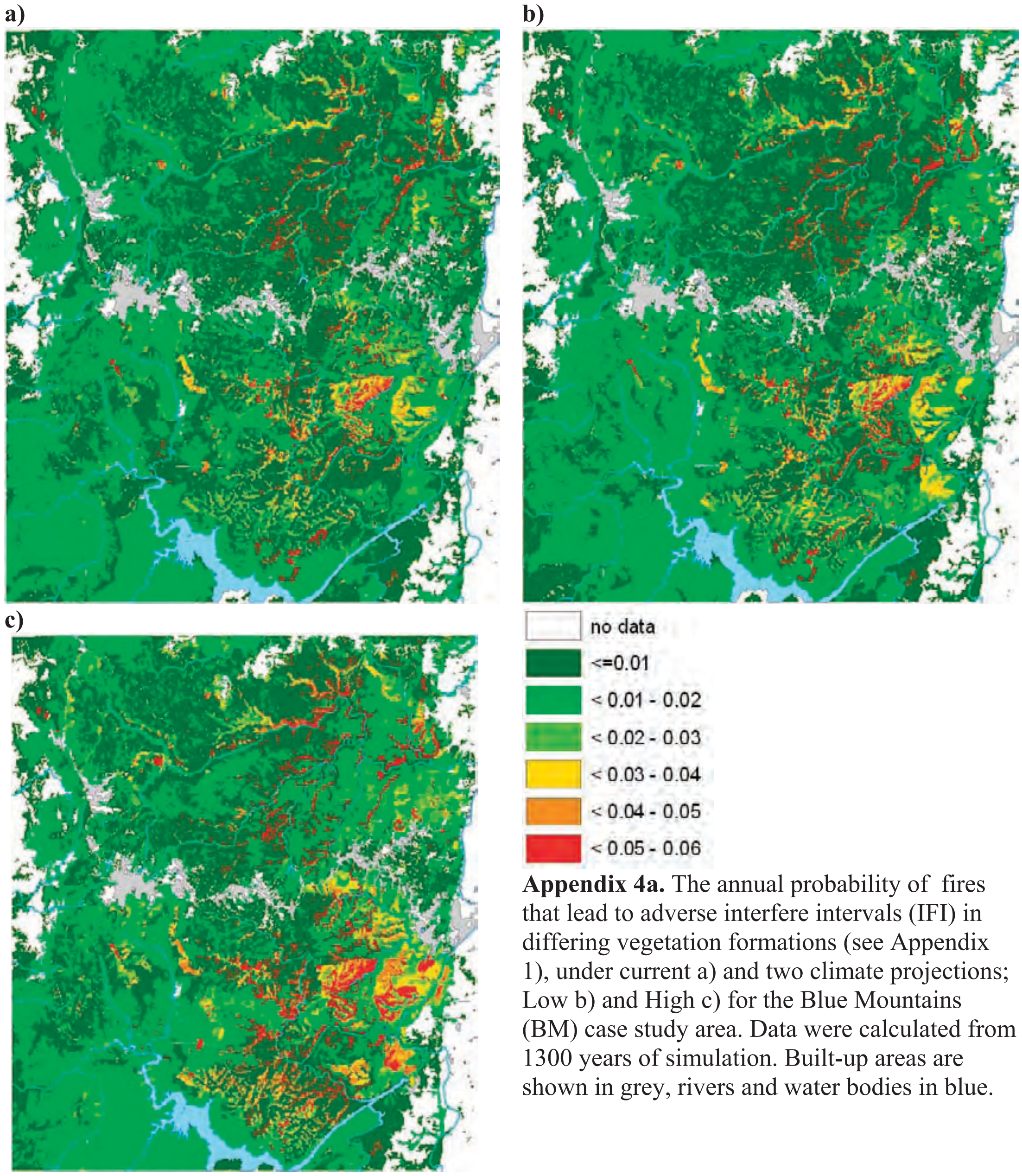




Appendix 3b. Mean fire line intensity of unplanned fires for current a) and two climate projections (Low b) and High c)) in the Woronora Plateau (WP). Data calculated over 1300 years of simulation. Built-up areas are shown in grey, rivers and water bodies in blue.



Appendix 3c Fire line intensity of unplanned fires for current a) and two climate projections (Low b) and High c) for the Hornsby (HK) case study. Data calculated over 1300 years of simulation. Built-up areas are shown in grey, rivers and water bodies in blue.

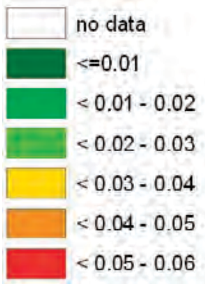


Appendix 4a. The annual probability of fires that lead to adverse interfere intervals (IFI) in differing vegetation formations (see Appendix 1), under current a) and two climate projections; Low b) and High c) for the Blue Mountains (BM) case study area. Data were calculated from 1300 years of simulation. Built-up areas are shown in grey, rivers and water bodies in blue.

a)

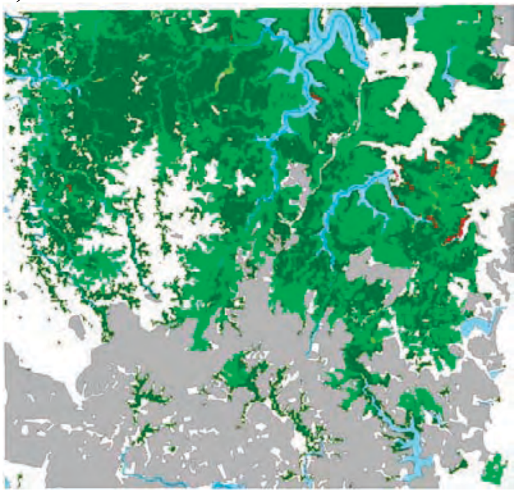
b)

c)

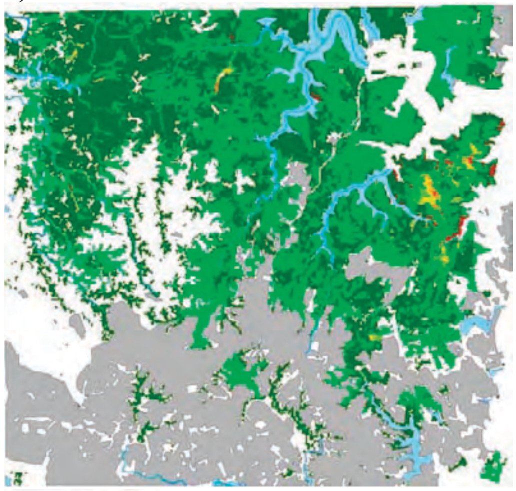


Appendix 4b. The annual probability of fires that lead to adverse interfere intervals (IFI) in differing vegetation formations (see Appendix 1), under current a) and two climate projections; Low b) and High c) for the Woronora Plateau (WP) case study area. Data were calculated from 1300 years of simulation. Built-up areas are shown in grey, rivers and water bodies in blue.

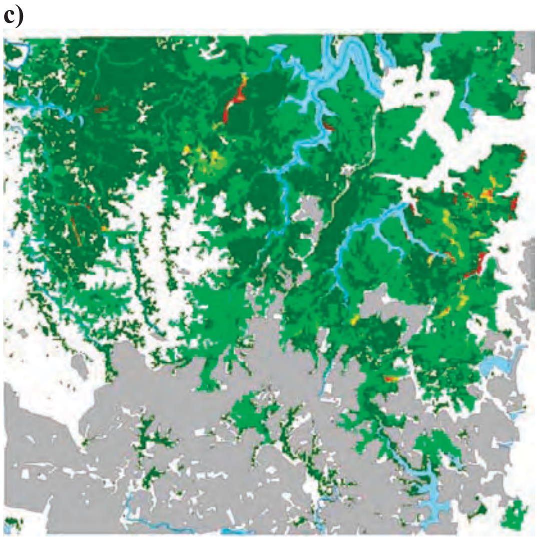
a)

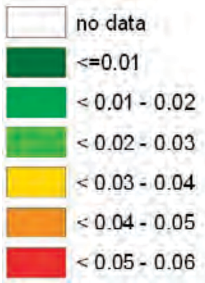


b)



c)





Appendix 4c. The annual probability of fires that lead to adverse interfere intervals (IFI) in differing vegetation formations (see Appendix 1), under current a) and two climate projections; Low b) and High c) for the Hornsby (HK) case study area. Data were calculated from 1300 years of simulation. Built-up areas are shown in grey, rivers and water bodies in blue.

End of Report